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La evaluación de los servicios de los ecosistemas como herramienta para planificar la restauración ecológica de cuencas hidrográficas

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LA EVALUACIÓN DE LOS SERVICIOS DE LOS ECOSISTEMAS COMO HERRAMIENTA PARA PLANIFICAR LA RESTAURACIÓN ECOLÓGICA DE CUENCAS HIDROGRÁFICAS

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Ph.D dissertation

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Por

Mattia Trabucchi

Memoria presentada para optar al grado de Doctor por la Universidad de Zaragoza

Zaragoza, Noviembre 2012

Este trabajo ha contado con la dirección del Dr. Francisco A. Comín Sebastián, Profesor de Investigación del Consejo Superior de Investigaciones Científicas (CSIC) en el Instituto Pirenaico de Ecología.

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CERTIFICA:

Que Mattia Trabucchi ha realizado bajo mi dirección el trabajo,

para optar al grado de Doctor por la Universidad de Zaragoza, que aquí presenta con el título:

"La evaluación de los servicios de los ecosistemas como herramienta para planificar la restauración ecológica de cuencas hidrográficas"

Que el trabajo se ajusta a los objetivos establecidos en el Proyecto de Tesis Doctoral aprobado el 5 de julio de 2011, por el Departamento de Geografía y Ordenación del Territorio y ratificado por la Comisión de Doctorado el 14 de julio de 2012.

Y para que así conste, firmo la presente Certificación en Zaragoza el 17 de septiembre de 2012 para los efectos que sean oportunos.

Fdo.: Francisco A. Comín Sebastián

Índice

“Es necesario que enseñen a sus hijos, lo que nuestros hijos ya saben, que la tierra es nuestra madre. Todo lo que ocurra a la tierra, les ocurrirá también a los hijos de la tierra. Cuando los hombres escupen en el suelo, se están escupiendo así mismos. Esto es lo que sabemos: la tierra no pertenece al hombre, es el hombre el que pertenece a la tierra. Esto es lo que sabemos: todas las cosas están ligadas como la sangre que une a una familia. El sufrimiento de la tierra se convertirá en sufrimiento para los hijos de la tierra. El hombre no ha tejido la red que es la vida, solo es un hilo más de la trama. Lo que hace con la trama se lo está haciendo a sí mismo”

Extracto de la carta que envió en 1855 el jefe indio Seattle de la tribu Suwamish al presidente de los Estados Unidos, Franklin Pierce en respuesta a la oferta de compra de las tierras de los Suwamish en el noroeste de los Estados Unidos

A todos los becarios, investigadores, trabajadores del Instituto Pirenaico de Ecología que han hecho grande el estar en este centro estos años, por su altruismo y compañerismo, este aire que se respira aquí no se encuentra en otros sitios.

Resumen

La provisión de servicios por los ecosistemas podría empeorar considerablemente y rápidamente durante la primera mitad del presente siglo si no se restauran eficientemente ecosistemas degradados. Frente a la aproximación clásica de la restauración basada en sistemas de referencia a imitar, existe el reto de obtener metodologías para territorio amplio y complejo y no solo para un sitio con un tipo de ecosistema. Existen muchas opciones para conservar o fortalecer servicios específicos de los ecosistemas de forma que se reduzcan las elecciones negativas que nos veamos obligados a hacer o que se creen sinergias positivas con otros servicios de los ecosistemas. En esta tesis se ha desarrollado una metodología basada en la evaluación de servicios de los ecosistemas, como variables de estado, y del riesgo de erosión, como factor de disturbio, para establecer una jerarquización espacial de actuaciones de restauración a escala de cuenca hidrográfica. Para ello se ha realizado la evaluación de servicios de los ecosistemas, modelización de la erosión y se han utilizado sistemas de información geográfica (SIG) para la elaboración de cartografía jerárquica y análisis espacial. El área de estudio utilizada es la cuenca del Río Martín (Teruel, NE España, 1938 km^2) como unidad funcional que, por su susceptibilidad natural a la erosión y con su elevada heterogeneidad paisajística y diferentes usos del suelo (agrícola, minería, ganadera) se presta como un valioso territorio donde aplicar y testar la metodología propuesta. La cartografía elaborada para la estimación de las tasas de erosión ha sido extrapolada con el modelo RUSLE (Ecuación de pérdida de suelo revisada) utilizando un innovador índice de vegetación (GPVI). Este índice fue elaborado mediante una técnica de inteligencia artificial llamada programación genética, la cual fue calibrada con los datos de campo del factor C de RUSLE (muestreo de suelos, transectos de vegetación) del presente estudio. Los datos de campo utilizados para crear el mapa de erosión han sido complementados con imágenes satelitales Landsat 5-TM y mapas disponibles de las características del territorio (litología, uso del suelo, ortofotos aéreas). Las tasas de erosión observadas en la cuenca del Martín tienen una media de 13.8 t ha^{-1} $año^{-1}$ siendo notablemente mayores en la parte sur (20 t ha^{-1} $año^{-1}$) debido a su irregular orografía que en las zonas de llanura del norte (10 t ha^{-1} $año^{-1}$). Los servicios de los ecosistemas se evaluaron mediante indicadores obtenidos a partir de bases de datos nacionales y regionales complementados con datos de campo. Los datos son expresados para cada servicio en las unidades de medida correspondientes y se basan en el análisis de los mapas de diferentes datos físico-químicos y biológicos. Los datos de los servicios relacionados con el agua han sido proporcionados para la Confederación Hidrográfica del Ebro (CHE), los datos de acumulación de carbono en pies mayores han sido proporcionados por el Departamento de Recursos forestales del Centro de Investigación de tecnología y investigación agraria de Aragón (CITA). Los datos de acumulación de carbono en el suelo son disponibles en el Portal de Suelos Europeo (European Soil Portal). Las rutas de eco-turismo han sido descargadas de la pagina de

rutas wiki-loc y la pagina de senderos de Aragón. La retención de suelo fue modelizada combinando datos del factor C para estimar el porcentual de cobertura vegetal y las tasas de erosión del modelo RUSLE-SIG. Los servicios de los ecosistemas variaron también entre amplios y diferentes rangos. La acumulación de carbono varía entre 0 y 4648 t CO_2 eq en zonas menos densas de vegetación y 40442 y 118073 t CO_2 eq en las zonas forestales densas; la provisión de agua superficial en el norte varía entre 0 y 13 mm y 100 y 210 en el sur de la cuenca, principalmente en fondos de valles; el control de la escorrentía (recarga acuíferos) es más alto en zonas montañosas del sur de la cuenca con valores entre 8 y 81 mm año^{-1} con valores mínimos entre 8 y 34 mm año^{-1} en el norte y máximos de 81 mm año^{-1} en el sur; la retención del suelo se ha expresado en valores relativos que varían de 1 a 5 dependiendo de la relación entre porcentaje de cobertura vegetal y perdida del suelo (estimada por la RUSLE-SIG en 5 clases de muy baja a muy alta), con valor máximo de retención de suelo a coberturas mayores de 70% y erosión menor de 12 t ha^{-1} año^{-1}, y mínimo a zonas de cobertura inferior a 30% y erosión mayor de 17 t ha^{-1} año^{-1}. El servicio de eco-turismo se ha evaluado como presencia-ausencia, asignando valor 1 a las áreas de la cuenca que se observan desde los senderos usando la herramienta de visualización de cuenca en SIG (viewshed) y 0 en el resto de la cuenca que no se observa desde los senderos según el modelo digital del terreno utilizado. Tratándose de datos con unidades diferentes, entre ellos se utilizó una agrupación en el rango relativo de 1 a 5 de cada servicio por cortes naturales (Natural Breaks) en SIG, que genera clases cuyos límites se ubican donde hay diferencias relativamente grandes en los valores de los datos por cada servicio. Ecoturismo tenía un valor 0 o 1 según la ausencia o posibilidad de visualización del paisaje en el recorrer los caminos. El valor más elevado de un determinado servicio se considera un área de elevado valor definido como *hotspot*, que es un área de una importancia máxima para ese servicio. Análisis de solapamiento han sido realizados para entender las relaciones entre servicios. Finalmente a través de la creación de mapas jerárquicos los datos de erosión y servicios ecosistémicos han sido relacionados analizando la congruencia espacial y los patrones espaciales a diferentes escalas anidadas entre ellas, dándonos la posibilidad de analizar el comportamiento de los dos factores, y contrastar el factor de disturbio y las variables de estado a diferentes escalas espaciales. Se ha identificado la zona sur de la cuenca del área de estudio, como el área donde se presentan más servicios y se observan las tasas de erosión más altas debido a factores topográficos, entre otros. En ésta zona, y particularmente en las subcuencas con zonas mineras no restauradas (donde la erosión muestra tasas máximas y los servicios son muchas veces nulos y en subcuencas con altas tasas de erosión y alto número de servicios las acciones de restauración han de ser prioritarias si no se quieren perder servicios que benefician aguas abajo en la cuenca. Claramente según los objetivos del gestor las prioridades pueden modificarse y nuestra metodología fácilmente adaptarse. En la zona norte, llana y mayoritariamente usada para agricultura de cereal de secano, la erosión es relativamente baja y la provisión de servicios de regulación también. Es la zona de menor interés para realizar acciones de restauración dado que la mejora de los servicios no está asegurada y se podría entrar en conflicto con intereses de usos (*trade off*) de otros servicios (por ej., producción de alimentos) incluidos sociales. También se ha demostrado la utilidad de realizar evaluaciones a diferentes resoluciones espaciales para la mejor identificación de las zonas óptimas de restauración. Se propone un modelo conceptual general de toma de decisiones de restauración a escala de cuenca en función de la provisión de servicios de los ecosistemas y de los factores de alteración ecológica. Finalmente la metodología aquí propuesta, desarrollada con SIG con la creación de mapas jerárquicos, ha resultado fácilmente adaptable a la escala de paisaje. Esto hace que nuestro modelo dependiendo de la disponibilidad de datos, sea una herramienta útil y fácilmente aplicable para la restauración a escala de cuenca hidrográfica o de paisaje, donde los servicios ecosistémicos estén alterados por diferentes factores de disturbio.

1. Introduction

1.1. ***Ecosystem service trends in basin-scale restoration initiatives***

Human-induced changes and damage of the Earth's ecosystems make ecological restoration one of the key strategies of the present and beyond (Hobbs and Harris, 2001). Restoration is vital for stemming both the current loss of biodiversity and the associated decline of ecosystem services (Dobson et al. 1997; Millenium Ecosystem Assessment (MA) 2005). The purpose of restoration is to initiate, or accelerate, the recovery of an ecosystem with respect to its health, integrity and sustainability (SER 2004). Ecological restoration and associated efforts are rapidly increasing and are being implemented throughout the world (Clewell and Aronson 2007). This growth is supported by global and regional policy commitments, such as the Convention on Biological Diversity ([article 8(f)] 2007) and the Commission of the European Community (2008), among others. Restoration can be undertaken at different scales ranging from local and habitat-specific actions to the biome and regional levels. Although small-scale short-term projects can be valuable, these experiments do not resemble real-world ecosystem management. Many authors recognize the urgent need to greatly expand the scale of ecosystem restoration and conservation (Comín 2010; Moreno-Mateos et al. 2012; Naveh 1994; Palmer 2009; Hobbs and Norton 1996; Wohl et al. 2005). Large-scale ecosystem restoration is required to arrest and reverse the degradation of landscapes around the world, particularly focusing on biodiversity as a positive relationship has been observed between biodiversity and ecosystem services after restoration (Rey-Benayas et al. 2009). Also focus on river systems is encouraged as increasing evidence suggests that the biodiversity of freshwater ecosystems is among the most endangered in the world (Driver et al. 2005; Dudgeon et al. 2006; Jenkins 2003; WWF 2004).

The emerging policy focus on ecosystem services represents a significant shift in the objectives of restoration (Bullock et al. 2011). Economic valuation of ecosystem services has accentuated interest in using these services as a basis for restoration and conservation programs (Ehrenfeld 2000). European Environment Agency (EEA) initiated the EURECA project which is intended to contribute to a European Ecosystem Assessment is strong evidence of the institutional interest in integrating ecosystem services in future socio-economic decisions. Recent progress in the assessment and evaluation of ecosystem services is likely to increase the inclusion of ecosystem services in restoration planning and implementation (Fiedler et al. 2008; Martinez et al. 2008; Moberg and Ronnback 2003; Nelson et al. 2009; Reyers et al. 2009). While a single restoration project is unlikely to ameliorate the state of a large degraded basin, ecologists can help to identify combinations of projects that will best restore ecosystem services within watersheds. To obtain a full understanding of the services provided in a study area, research should ideally be conducted at multiple, nested scales, as environmental effects may be uncorrelated across scales (MA 2003), although the large-size, long-term ecological services and functions constrain or control the small-size, periodical ecosystem services and functions (Limburg et al. 2002). Such “strategic” restoration would prioritize the location, size and type of network of restoration projects needed for a watershed that can be compared with the stakeholder needs in order for it to provide optimal levels of ecosystem services (Zedler and Kercher 2005). Biophysical and, increasingly, socio-economic values are currently used to define priority areas for planning conservation and environmental management measures (Raymond et al., 2009) as well as for evaluating the benefits of restoration projects (Aronson et al. 2010; Palmer et al. 2005). However, the degree to which ecosystem services have been incorporated into basin-scale restoration actions to date is unclear. To address this knowledge gap, we conducted a survey of peer-reviewed international scientific literature to reveal global trends. Furthermore, we explored the emerging issues related to ecosystem service classification, mapping approaches, tools and software. We identified opportunities for the increased integration of ecosystem services in basin-scale restoration projects, suggesting a framework based on new hierarchical maps. This is based on congruence among threat maps (e.g., thresholds of impacts) and ecosystem service maps. The resultant new map will facilitate the targeting of threatened service supply at different scales. The inclusion of ecosystem services in restoration projects provides an opportunity for defining clear goals for generating public support and funding sources, which are necessary conditions to enhance the planning and implementation of restoration projects (Choi 2007; Ehrenfeld 2000; Hobbs 2007).

1.1.1. **Literature search and data extraction**

In order to understand how ES have been used in basin-scale restoration we search for peer-reviewed publications in using the ISI Web of Science from 1998-2010 (February) written in the English language, follow the methodology of Egoh et al. (2007). (http://www.newisiwebofknowledge.com). We limited our search to 1998 and beyond because this is when the terminology “ecosystem services” was introduced in the published literature by Daily (1997) and Costanza (1997). This publication, among others, created a clear increase in the number of studies citing ecosystem services (see Fig. 1 in Fisher et al. 2009). We searched for the term “restoration project” in an advanced search on ISI using the Boolean AND associated with a number of terms related to restoration (see Appendix 9.1.). For Data extraction we followed the data extraction methodology of Rey-Benayas et al. (2009) in part (see Appendix 9.1.), examining the titles and abstracts of each reference to determine how closely they aligned with our selection criterion of ecosystem services classification based on MA (2005) within basin areas, thereby determining their inclusion in this review. If the manuscript reported on measures of one or more ecosystem services and/or biodiversity in relation to restoration at the basin scale, the study was included. Tools and techniques associated with the included services were also discussed to understand the best way to include services in restoration projects in the future.

1.2. *Inclusion and trends of ecosystem services in restoration*

Our search identified a total of 414 studies related to the selected search terms. However, only 45 of these studies involved research addressing basin-scale restoration and also made reference to ecosystem services either explicitly or implicitly. Analysis of the 45 studies showed a clear increase in the integration of ecosystem services (or processes resulting in these services) in basin-scale restoration studies from 2006 onward (Fig. 1). Of the 45 studies, only 13 explicitly referred to ecosystem services as being an integral part of basin-scale restoration studies. Among these 13 studies, eight investigated only one ecosystem service; four studies measured two; and one study measured three ecosystem services. In the remaining 32 studies, the reference to ecosystem services was implicit in their reference to ecosystem providers expressed as processes and functions.

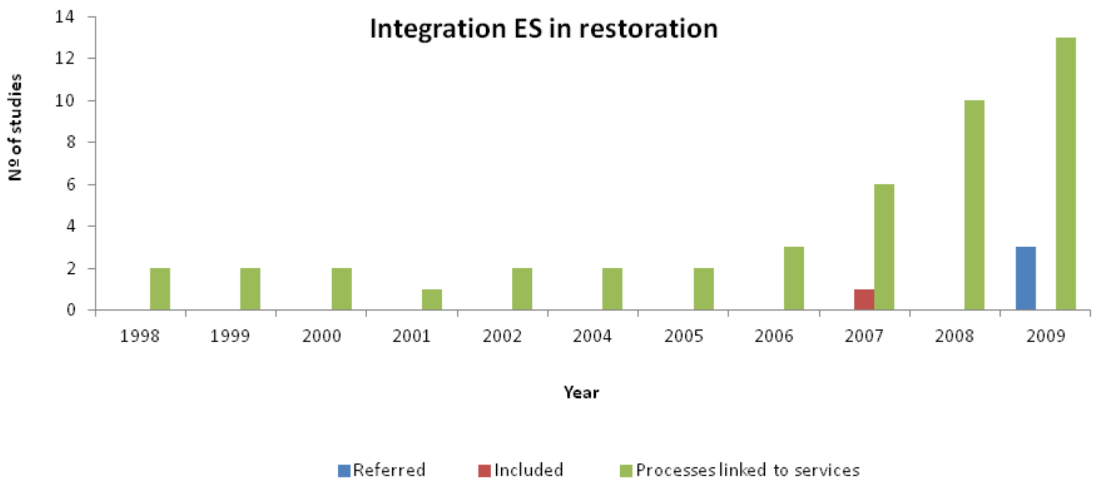


Fig. 1. Results of a review of the integration of ecosystem services in 45 restoration projects at basin scale over time. Graph shows number of projects that have either referred to or included services, reflecting the number of studies which have included services and those that have included processes potentially linked to services.

1.2.1. **Types of services that have been included**

Four categories of services were addressed in the 13 studies that made explicit reference to ecosystem services: supporting, regulating, cultural and provisioning services. The supporting service was the most common (appearing in eight studies), followed by regulatory (three studies), cultural (one studies) and provisioning services (one study). We note that these categories are not mutually exclusive; most of the restoration studies potentially included multiple services that were not stated, thus preventing the positive results of restoration from being represented in their totality, downplaying the effort undertaken. The supporting service of habitat/refugia/nursery functions, which is generally linked to target species that benefit from habitat restoration, was the most common. Flood/drought prevention, water regulation and erosion control also received attention in restoration studies, either through their explicit inclusion or through the inclusion of ecological processes linked to them. The provisioning services addressed in the studies were focused on water production in a river basin, while the cultural services were focused on landscape restoration and the local inhabitants' perceptions of the projects, which were evaluated by means of local surveys (see Table 8 in Appendix). Our review indicated that no study at the basin scale explicitly mapped ecosystem services targeting restoration; instead, they identified and, in some cases, mapped processes and Ecosystem Service Providers (ESPs), which are mostly habitats, species and populations that are in some way responsible for the provision of services.

1.2.2. **Classifying ecosystem services**

Despite the fact that ecosystem services now feature prominently in ecological studies and the many calls that have been made to introduce them into restoration plans (Dodds et al. 2008; Ormerod 2003; Peterson and Lipcius 2003), prior to 2006, few peer-reviewed studies on restoration at the basin scale actually did so. Our review found an increasing trend from this date onward towards the inclusion of this concept (Fig. 1). This growth may be due to an emerging societal consciousness that resources are becoming increasingly degraded and scarce (Costanza et al. 1997). The main reason for these declines is the rapid increase projected globally in the demand for food, fresh water, energy, and other resources over the next few decades, which implies greatly intensifying human impacts (Daily 2000). But the great catalyst was the MA work which made a thorough effort to assess the effects of policies on ecosystem services and human well-being in 2005 (MA 2005), and provided a base for further studies (Carpenter et al. 2009).

Notwithstanding the most difficult task in this review was the identification of ecosystem services, which was due to the lack of consistency and absence of the use of universally accepted classifications (e.g., Costanza et al. 1997; de Groot et al. 2002). Instead, the selected studies mostly referred to restoration of ESPs, ecological functions and processes to support biodiversity. This was a normal practice in past studies, where functions were identified and studied for years with no reference to services for humans, which they also provide (Fisher et al. 2009). Current debates about how to best define the distinction between ecosystem functions and services and how to classify the services to make them quantifiable in a consistent manner are ongoing (Fisher et al. 2009; de Groot et al. 2010). In a recent review, Rey-Benayas et al. (2009) also found that only a small minority of studies explicitly referred to the concept of ecosystem services, whereas a larger number referred to the concept of ecosystem function. In turn, Wallace (2007) found many relevant authors who examined the classification of ecosystem services combining means (processes) and ends (services) within the same category level, making the categories unusable for effective decision making. In our study case, for example, it was found that different services may be linked through processes, which may result in an unconscious double counting of services if services are not explicitly included in the study. The inconsistency in ecosystem service classification has been noted in many studies as Fu et al. (2011) highlighted in a recent review, causing uncertainty and a lack of reliability with respect to the estimation of the value of ecosystem services.

1.2.3. **Functions, processes and services?**

Ecosystem services are generated by ecosystem functions, which, in turn, are underpinned by biophysical structures and processes classified in the MA (2005). Moreover, biophysical processes are essential for the provision of ecosystem services, but processes are not synonymous with services (Tallis and Polasky 2009). Processes and functions become services if there are benefits for humans from them (Fisher et al. 2009); nevertheless, it is common to find many authors who treat them as synonyms (Wallace 2007). It is clear that a coherent and integrated approach for practical application of the concept of ecosystem functions and services in planning, management and decision making is still lacking (ICSU et al. 2008).

1.2.4. **Missed opportunities**

Every restoration project directly or indirectly aims to improve ecological processes, and based on the degree to which a degraded area is restored, it can potentially improve ecosystem services and create new ones, changing the conditions of degraded sites and improving the delivery of services. This is why some studies include multiple overlapping services, either intentionally or not. For example, in the present review, it was found that studies that attempt to restore habitat (see: Battin et al. 2007; Fullerton et al. 2006; Fullerton et al. 2009; Katz et al. 2007) for a target species (e.g., salmon) can be included among both supporting services (habitat provision) and provisioning services (food). Additionally, restoration of salmon habitat could enhance other services, such as regulating and cultural services (e.g., if the salmon are fished). However, the different services will often not be cited and are even less likely to be quantified. In studies addressing the dynamics of land use in a watershed, such as that of Rayburn and Schulte (2009), the addition of ecosystem service maps could complement, enrich and drive future land use scenarios as a basis for restoration planning.

Unfortunately, this lack of awareness regarding the use of ecosystem services is partially due to the poor understanding of the quantitative relationships between biodiversity, ecosystem components and processes and services. As de Groot et al. (2010) highlight, criteria and indicators are required to comprehensively describe the interaction between the ecological processes and components of an ecosystem and their services. Reaching this point, it is extremely important to create *standardized terms and definitions*, eliminating any doubts and inconsistencies and standardizing the classification and the methodology. Despite the tremendous resources required for this ambitious approach (Kremen and Ostfeld 2005), some progress has been made. If the opportunity to achieve concrete results is not to be lost, then it is time to standardize methodologies, definitions and key concepts to describe and quantify ecosystem services (de Groot et al. 2010; Wallace 2007).

1.2.5. **Learning from previous studies**

Given the amount of attention that the ecosystem services concept has received in the past few years, it seems surprising that the services are not yet widely used to drive and target restoration projects (e.g., at landscape and basin scale). A likely cause of this oversight is the use of a traditional *ad hoc* restoration approach instead of a more holistic view, which constitutes the basis of sustainability. We therefore need to move away from the *ad hoc* site- and situation-specific approach that has been prevalent in restoration activities (Hobbs and Norton, 1996). For example, in a river restoration project, a broad knowledge of the characteristics of the watershed and river is required to identify not only environmental impacts but also their origins (Comín et al. 2009). In the present review, Fullerton et al. (2006) can be a good example of ecological data required for future translation from process into services. They used land use maps, aerial photos and field observations to map riparian areas according to their in-stream functions (organic matter inputs, filtration of pollutants and sediment, bank stabilization, temperature control), linking them with services such as disturbance prevention and nutrient cycling. Fewer explicit guidelines are available at the landscape/basin scale beyond non-quantitative generalities about size and connectivity. The global-scale ecological decline (Global Footprint Network 2010) requires the development of general guiding principles for restoration projects to address the global challenges that humanity faces (Comín 2010). Development of these guidelines should be prioritized so that urgently required large-scale restoration can be planned and implemented effectively (Hobbs and Norton 1996).

1.2.6. **Mapping ecosystem services**

Unfortunately, the quantitative relationships between biodiversity, ecosystem components and processes and services are still poorly understood (de Groot et al. 2010). Current landscape maps normally include land cover and/or related uses. Quantifying ecosystem services in a spatially explicit manner and analyzing tradeoffs between them can lead to making more effective, efficient and defensible decisions related to natural resource.

Mapping ESPs is one of the most explicit methods for including ecosystem services in conservation activities (Egoh et al. 2007), though no consistent mapping protocol or official accepted framework exists that can be followed for this purpose. One of the main research questions to be resolved is how ecosystem services can be spatially mapped and visualized in a universal way (de Groot et al. 2010). In this review, ecosystem services were generally found to be both biotic (Grundel and Pavlovic 2008) or abiotic (Fullerton et al. 2006; Nienhuis et al. 2002) attributes, such as vegetation type (Vesk et al. 2008) or scenic rivers being mapped (Junker and Buchecker 2008). Mapping could also be applied in restoration planning, providing the opportunity to locate and quantify services for the purpose of making decisions and prioritizing future restoration activities. Unfortunately, the extent to which ecosystem services can be included in restoration studies remains largely untested, but there are some interesting new attempts focusing on some areas or some types of ecosystems of a territory (Orsi et al. 2011; Pert et al. 2010; Tong et al. 2007).

1.2.7. **Prioritization through mapped congruence**

Ecosystem services coupled with climate, demographic, economic and social models and data are becoming more common. The widespread use of geographic information systems (GIS), statistics and geostatistics currently provides a powerful and complementary suite of tools for spatial analysis in the agricultural, earth and environmental sciences (Burrough 2001). Studies at the basin and landscape scales have begun to include ecosystem service mapping and evaluation into management and restoration plans (see: Egoh et al. 2011; Nelson et al. 2009; O'Farrell et al. 2010; Wendland et al. 2010). These authors follow the common framework of comparing services with one or more datasets, such as datasets addressing biodiversity conservation, vegetation diversity, needs of the local population, or commodity production. Following these examples of data intersection, we suggest a framework based on evaluation of the congruence among degrading processes or threat areas (e.g., erosion, deforestation, point and non-point pollution areas) and ecosystem service maps (Raymond et al. 2009) for the generation of new hierarchical maps based on thresholds of impacts (e.g., estimation of erosion limits for soil formation) and services (e.g., the number per area or level of importance required for the wellbeing of the beneficiary).

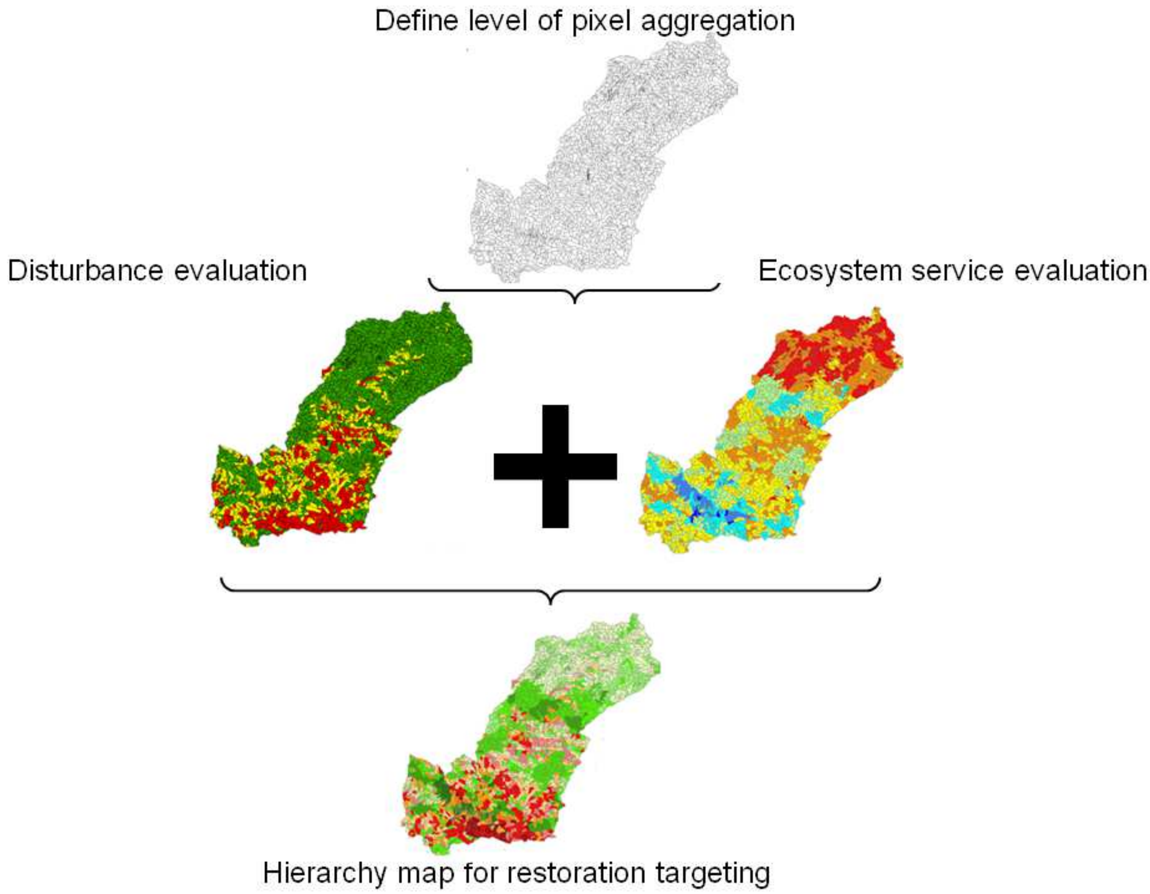


Fig. 2. Schematic map showing the building of a hierarchy map: first select a pixel aggregation or scale (e.g. subwatershed, river order) for data analysis; second, spatial analysis of the disturbance factor (e.g. mapping) and of the ecosystem services (e.g. mapping); third, reclassification and overlapping of the disturbance factor and the ecosystem services.

Congruence among ecosystem services or ecological processes and threats areas will be exported as a new map (Fig. 2) which will facilitate the targeting of threatened services supplied at different scales from the basin scale to the scale of the restoration site. This systematic approach is well recognized as the essential next step toward informing decision making for a systematic approach that combines the rigor of small-scale studies with the breadth of broad-scale assessments (Tallis et al., 2009). The development and application of these hierarchical maps is a step in this direction, providing the opportunity to obtain an overview of the ecological state of a basin to understand and locate key ecosystem service priority areas for the purpose of maintaining, improving or restoring strategically identified targets. In these cases, the resolution of the available data is key for the downscale approach to be effective. Changing the spatial scale from a basin to prioritized areas requires optimum dataset support, depending on the scale of the target (e.g., at finer scales, a smaller pixel size will be required) to achieve more accurate targeting. Depending on the cell size of our maps, we would be able to downscale gradually from the basin to the subwatershed until we arrive at more defined and specific threatened areas (e.g., slopes, opencast mines, riparian areas, forest patches)

In the next chapters we will provide a practical approach to the proposed framework for the creation of hierarchical maps based in erosion and ecosystem services maps in Martín Basin (NE Spain).

1.3. *Mapping erosion risk at the basin scale with opencast coal mines to target restoration actions*

Restoring eroded lands is a major objective to give back value to large parts of the world where erosion is a major environmental problem (Pimentel et al. 1995). However, defining areas for restoration in a vast territory requires establishing the magnitude of the problem and the benefits of the solutions at an adequate spatial scale (Boardman 2003).

Soil is often lost through erosion, a natural process that can be fostered by inappropriate land use and intense precipitation, among other factors (Garcia-Ruiz 2010). The European Union considers soil to be a nonrenewable resource, and soil degradation has strong impacts on soil and water resources (Montanarella 2000). The loss of topsoil and changes in its properties will cause the decline of the ecological processes that rely on it. Soil erosion increases the impact on streams through high sediment delivery, which has been identified as a leading cause of river degradation (USEPA 2000). Consequently, soil erosion causes the loss of the services provided by ecosystems (Van Wilgen et al. 1996) and knowing the spatial distribution of erosion rates is a primary step for planning restoration at the watershed scale.

In Mediterranean areas, developing efficient tools for decision making regarding land use management is a major objective (Simoncini 2009) because of the multiple environmental problems arising from the intensive use of the land since long ago (Tabara and Ihlan 2008), particularly problems related to erosion (Boardman et al. 2003, Bazzoffi 2009). Opencast mining is one such activity, which contributes mostly to erosion (Wu and Wang 2007). Opencast mines are sources of high sediment yield to rivers if restoration is not properly carried out (Balamurugan 1991; Taylor and Owens 2009). Subsequently, human intervention in failed reclamation areas, especially opencast mines with highly eroded slopes connected with the river network, is necessary to prevent water pollution and to slow irreversible erosion (Pimentel et al. 1995; Palmer et al. 2010).

Mapping ecological processes and restoring areas with high sediment delivery would help avoid irreversible degradation that removes nutrients and reduces fertility (DeFries and Eshleman 2004), thus limiting the sedimentation and eutrophication of nearby rivers, which would represent a potential hazard for the long-term sustainability of agriculture and ecosystem services at the basin scale (Krauze and Wagner 2007). For this reason, the number of projects on sediment-related river restoration at the river basin scale is increasing (Kondolf 1998; Ward and Fockner 2001; Pizzuto 2002; Pennisi 2004). Successful restoration projects on river basins require an understanding of sediment transport processes. This understanding is achieved by identifying the suspended sediment sources on the basis of sediment monitoring and modeling (Gao 2008).

1.4. *Mapping ecosystem services for management and targeting restoration efforts*

Human use and manipulation of ecosystems has increased rapidly over the last century. Currently, approximately 60% of worldwide ecosystem services are considered to be either degraded or used in an unsustainable manner (Millennium Ecosystem Assessment 2005). Agriculture and mining are vital human activities that generate essential products for human subsistence and well-being; however, both agriculture and mining have major impacts on the services provided by ecosystems (Power 2010). If we are to retain vital ecological functions, trends in ecosystem degradation need to be either halted or reversed through restoration actions (Global Footprint Network GFN, 2008; Comín 2010).

Mapping ecosystem services has become a popular tool for achieving different environmental objectives. Carreño et al. (2011) assessed the tradeoffs between the provisioning of ecosystem and economic services over the course of 50 years of land-use change in Argentina. Egoh et al. (2011) identified spatial priority areas for ecosystem services in grasslands in South Africa and evaluated whether biodiversity priority areas can be aligned with those for ecosystem services. Nelson et al. (2009) used a spatially explicit modeling tool to predict changes in ecosystem services, biodiversity conservation, and commodity production levels in a United States river basin. O' Farrell et al. (2010) engaged stakeholders and experts in identifying key services for determining the congruence between biodiversity priorities and ecosystem service *hotspots*. The inclusion of ecosystem services in environmental research will be a major challenge and will bring multiple positive advantages. For example, promoting the variety of ecosystem services that modern agricultural systems can provide would increase the value of agricultural areas in watersheds that require restoration (Swift et al. 2004). The need for aligning restoration objectives and ecosystem services has been recognized, and a growing number of studies are offering examples at appropriate local scales where this alignment has been attempted (Trabucchi et al. Submitted).

Planning the management and restoration of a region requires the identification and evaluation of the services provided by different types of land use and the prioritization of areas according to these findings. Two key issues have emerged from such planning. The first relates to the availability of data about ecosystem services (Troy and Wilson 2006). Detailed spatial information is needed to locate and quantify ecosystem services so that ecosystem services can be integrated into plans for management and restoration. The Millenium Ecosystem Assessment (MA 2005) attempted to address the lack of ecosystem service information required for decision making by assessing current knowledge, scientific literature and data. The findings of this study gave rise to the creation of ecosystem service databases at regional and national scales. The second issue pertains to recognizing the need for restoration initiatives that utilize ecosystem service information to reverse ecological degradation, recover habitats and restore biodiversity, ecological functions and services. Such restoration initiatives include erosion control, reforestation, removal of non-native species and weeds, re-vegetation of disturbed areas and the reintroduction of native species (SER 2004).

The 2006 Biodiversity Communication and its detailed Action Plan (Commission of the European Community 2006) acknowledged the need for restoration initiatives within the European Union. A recent review by Rey-Benayas et al. (2009) showed that ecological restoration supports biodiversity and ecosystem services by 44 and 25%, respectively, and that increases in both biodiversity and ecosystem services were positively correlated. Ecosystem service identification and evaluation is increasingly used to locate important natural resources and services for conservation, protection, restoration and management (Egoh et al. 2008; Nelson et al. 2009; O'Farrell et al. 2010; Reyers et al. 2009; Viglizzo et al. 2011). Furthermore, this information allows for the prioritization of investments (Johnson 1995). Areas for restoration can be selected in terms of their ability to reduce environmental risks while enhancing ecosystem service delivery. Clearly, we should be developing restoration programs that explicitly state priorities or goals (Forsyth et al. 2012) in the planning stages to guide investment decisions. Spatial congruence between areas targeted for restoration and areas that deliver ecosystem services needs to be examined and possibly aligned beforehand. Multi-scale analysis is especially important to Mediterranean ecosystems, which are characterized by high heterogeneity and provide society with a great diversity of ecosystem services at different scales (Martín-Lopez et al. 2012). River basins consist of a mosaic of ecosystems typically classified at subwatershed levels. Planning restoration at such scales requires the prioritization of subwatersheds according to their potential for delivering benefits from the restoration.

1.5. *Multi-scale approach for establishing restoration priorities in a degraded Mediterranean landscape through the evaluation of ecosystem services*

Soil erosion is a major threat to the continued provision of ecosystem services in large parts of the world (Brown 1981), particularly in arid and semi-arid areas (Gisladottir and Stocking 2005; García-Ruiz 2010). The future global change scenario corroborates the negative effects of increasing drought in Mediterranean regions on vegetation (Schroter et al. 2005), with runoff and sediment yields increasing in association with decreasing plant cover (from a certain threshold of cover) (Quinton et al. 1997). These suggested conditions are likely to result in greater amounts of soil being exposed to water and wind erosion (López et al. 1998). Additional factors that determine the predominance of erosion include the spatial scale, topographic thresholds, rainfall magnitude-frequency-duration characteristics, the initial soil moisture content and soil biological activity (Cammeraat 2002). Intensive agriculture and mining are land-use practices that are responsible for increasing erosion rates. These activities cause serious environmental problems across vast areas and result in enforced critical trade-offs for the associated societies (Zhang et al. 2007; Bernhardt and Palmer 2011; Carreño et al. 2011).

A key issue in semi-arid environments is determining how to prioritize areas for restoration to optimize erosion control. However, the challenge is increasingly how to combine this goal with the improved provision of vital ecosystem services, particularly water-related services and reduce the negative consequences for human development (Reynolds et al. 2007). Emerging policies are focused on ecosystem services and their inclusion in measures aimed at the restoration and the control of erosion. This represents a significant shift in the objectives of restoration (Bullock et al. 2011). Different organizations have set targets for ceasing biodiversity losses and the degradation of ecosystem services and restoring them 'so far as feasible' (EU Biodiversity Strategy 2020, MA 2003). To meet these policy objectives, there is growing interest in the development of tools and methods for identifying and evaluating ecosystem services and incorporating these measures into policies related to landscape planning, management and the allocation of environmental resources (Ruiz-Navarro et al. 2012; de Groot et al. 2010). This is particularly the case with regard to degraded areas and when attempting to understand trade-offs that arises related to land use and land cover planning (Rodríguez et al. 2006).

Mapping of ecosystem services has been identified as a useful aid in decision making during the allocation of efforts aimed at land use planning and management, particularly for the restoration of degraded areas (Reyers et al. 2009; Pert et al. 2010; Carreño et al. 2011). To obtain a complete understanding of the services provided in a study area, research should ideally be conducted at multiple, nested scales, as environmental effects may be uncorrelated across scales (MA 2003). The extent to which ecosystem services can be integrated into basin-scale restoration projects that are focused on reversing these trends remains largely untested, despite the recent and growing number studies focused on this broader topic (Fisher et al. 2009).

To understand how landscapes affect and are affected by biophysical and socioeconomic activities, we must be able to quantify spatial heterogeneity and its scale dependence (i.e., how patterns change with scale) (Wu 2004). Hierarchy theory is applied to the development and organization of landscape patterns and is best understood if tested across spatial and temporal scales (Bourgeron and Jensen 1993). Disturbance events that maintain landscape patterns and ecosystem sustainability are also spatial-temporal scale-dependent phenomena (Turner et al. 1993). Acknowledgment of this situation is critical for the development of management strategies aimed at ecosystem sustainability (McIntosh et al. 1994). Watershed risk analysis procedures can be used to consider the effects of rehabilitation treatments on watershed-level hazards, the consequences of inaction and the resources at stake (Milne and Lewis 2011). The combined analysis of areas that are important for the supply or provision of a suite of services employing erosion maps representing multiple scales should provide useful information for the establishment of priority areas for the restoration of watersheds (Orsi et al. 2011; Su et al. 2012; Trabucchi et al. 2012b). Historic restoration efforts have been primarily focused at a single scale (such as on stands or stream reaches) (Bailey et al. 1993; Milne 1994) and have relied on site-level information to direct restoration actions (Bohn and Kershner 2002). As a result, many restoration programs lack the ability to scale up their findings. This situation has prompted the call for the adoption of a multi-scale approach in planning ecological restoration policies (Ziemer 1997; Hobbs and Harris 2001; Comín 2010). Here, each restoration activity should be evaluated across a hierarchy of scales ranging from a broad region to an individual site, as the success of a local project depends on how well that project contributes to a comprehensive restoration strategy (Ziemer 1999; Palik et al. 2000). Landscape-level empirical studies are required for determining the kinds of scaling relationships that may exist and how variable or consistent they are (Wu 2004).

1.6. ***Objectives***

The general aim of this study is to check an approach for targeting and prioritizing sites for land management and restoration actions based on the assessment of ecosystem services in a Mediterranean semi-arid watershed with a marked spatial distribution of eroded areas.

The specific objectives are:

- Modelling the erosion in Martín River Basin.
- Evaluating a bundle of ecosystem services (water surface supply and flow regulation, soil retention and accumulation, carbon storage and ecotourism) and creating integrated maps of ecosystem services provision for the bundle of ecosystem services.
- Elucidating the spatial patterns of erosion and ecosystem services provision in Martín Basin.
- Create a spatial hierarchy of restoration actions against erosion for Martín Basin based on the evaluation of ecosystem services.

2. Study area

The Martín River watershed is a 2112 km^2 territory located in the south-central part of the Ebro River basin (NE Spain) (Fig. 3).

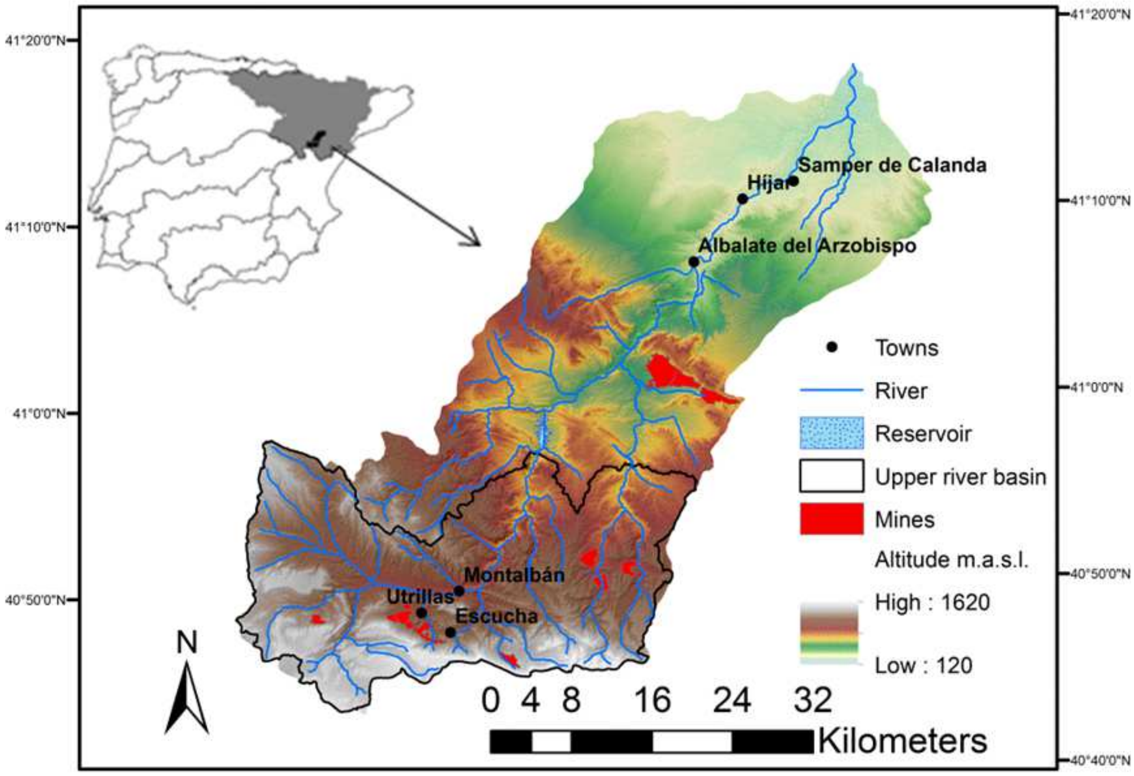


Fig. 3. Map of the Martín River Basin showing its hydrological network, and the upper (South) part and the lower (North) part of the basin, the last stream on the south east of the watershed is the Escuriza River.

The elevation ranges between 120 and 1620 m. a.s.l. The basin shows two major "regions" (Fig. 3): the highlands in the south (mean elevation 1100 m, 765 km^2) and the lowlands in the north (mean elevation 750 m, 1347 km^2), where most agricultural lands are established. The differences between these two zones are also marked by the presence of two dams, Escuriza (Fig. 3) and Cueva Foradada (maximum water storage capacities 6 and 22 Hm^3, respectively), which intercept sediments from the entire upstream area and disturb the natural river flow regime, creating a completely human-activity altered environment downstream. The climate is Mediterranean, with continental influence. The summers and winters are usually dry, and the annual average precipitation of the period 1970-2000 was 360 mm and is heterogeneously distributed both in space and time (Fig. 4). A few big storms are recorded every summer, more frequently in the upper (south) part of the watershed.

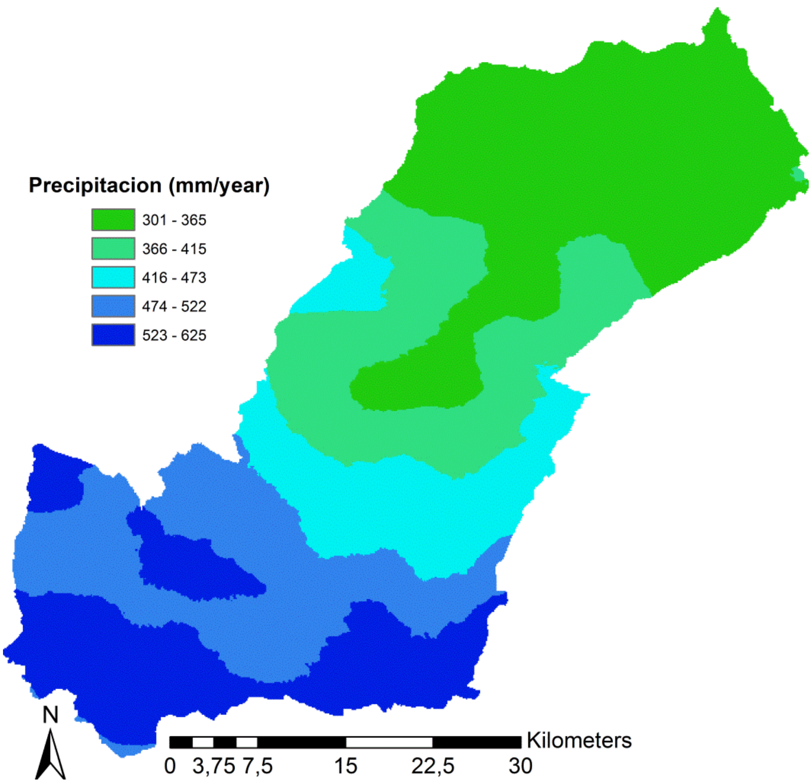


Fig. 4. Spatial rainfall pattern per year of the period 1970-2000 in Martín Basin.

The water deficit ranges between 530 mm and 758 mm, extending the dry period from May until October. The mean annual temperature range is 13-16 ºC, with minimum and maximum average temperatures of 5 and 25 ºC, respectively. Dryness, which has increased in recent years (Moreno-de las Heras et al. 2009), is the main limitation for natural plant development in the region and for the development of agriculture, which is the major socioeconomic activity in the lowland part of the basin (Fig. 5), covering 53% of this part of the basin. This land is mostly used for dry cereal farming (Foto 1 p. 39), except in the narrow belts along the river's sides near the villages, where an old canal network is still in use to irrigate vegetable and fruit tree fields. The meso-Mediterranean garrigue (*Quercus ilex*), accompanied by sabine (*Juniperus sabina*) in a few zones in the southern sector, is replaced northward by Kermes oak (*Quercus coccifera*), rosemary (*Rosmarinus officinalis*) formations, and steppe with small species (*Macrochloa tenacissima, Stipa tenacissima, Ligeum spartum, Tamarix africana, Juniperus phoenicea*). The only significant forests are located in the central part of the basin, and they consist mainly of Aleppo pine (*Pinus halepensis*). Riparian vegetation is extremely degraded because of the extensive cover of agricultural practices, and the intensive effects of some mining derived impacts and regulated river flows.

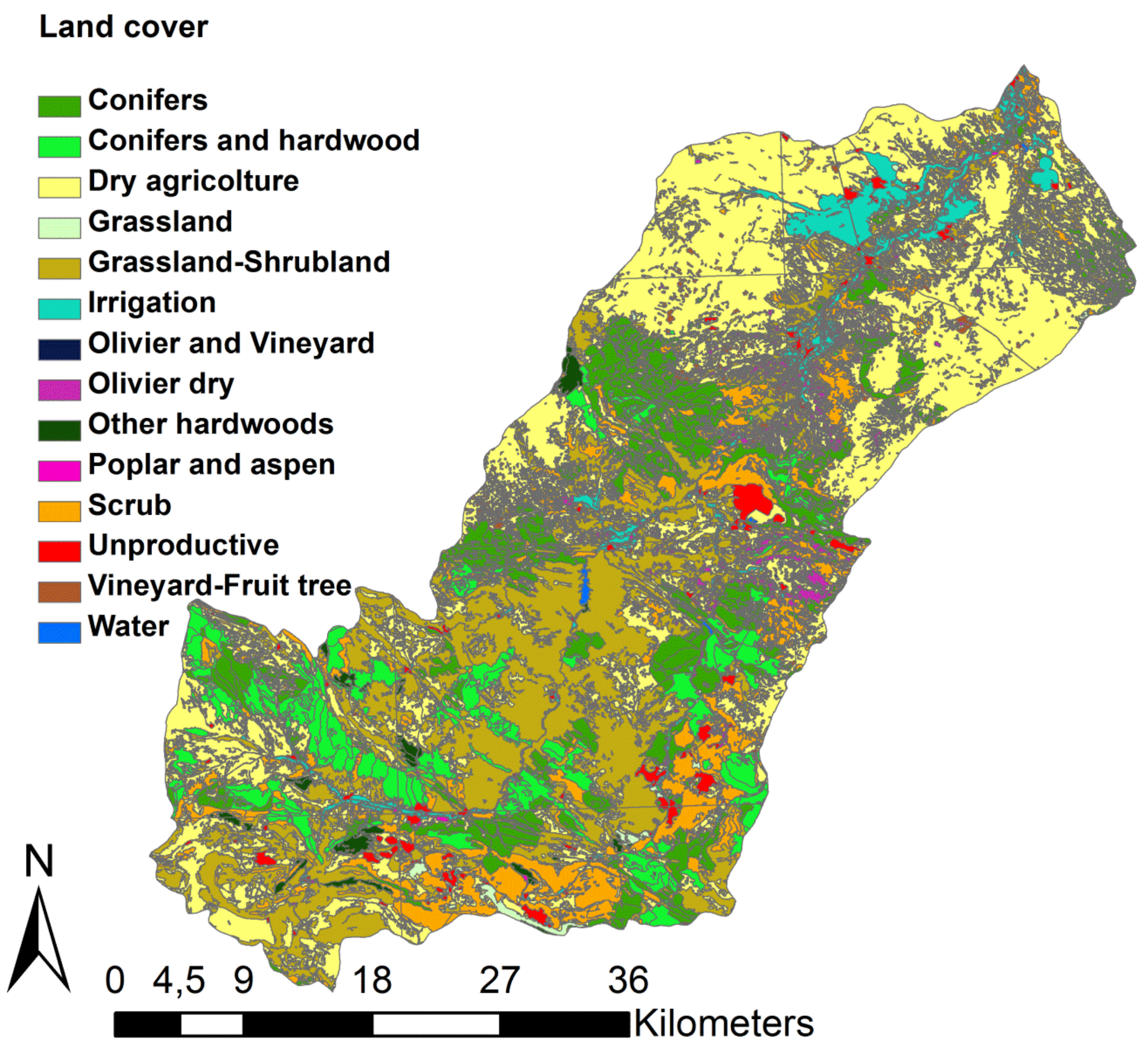


Fig. 5. Map of the Martín River watershed showing the different land cover units.

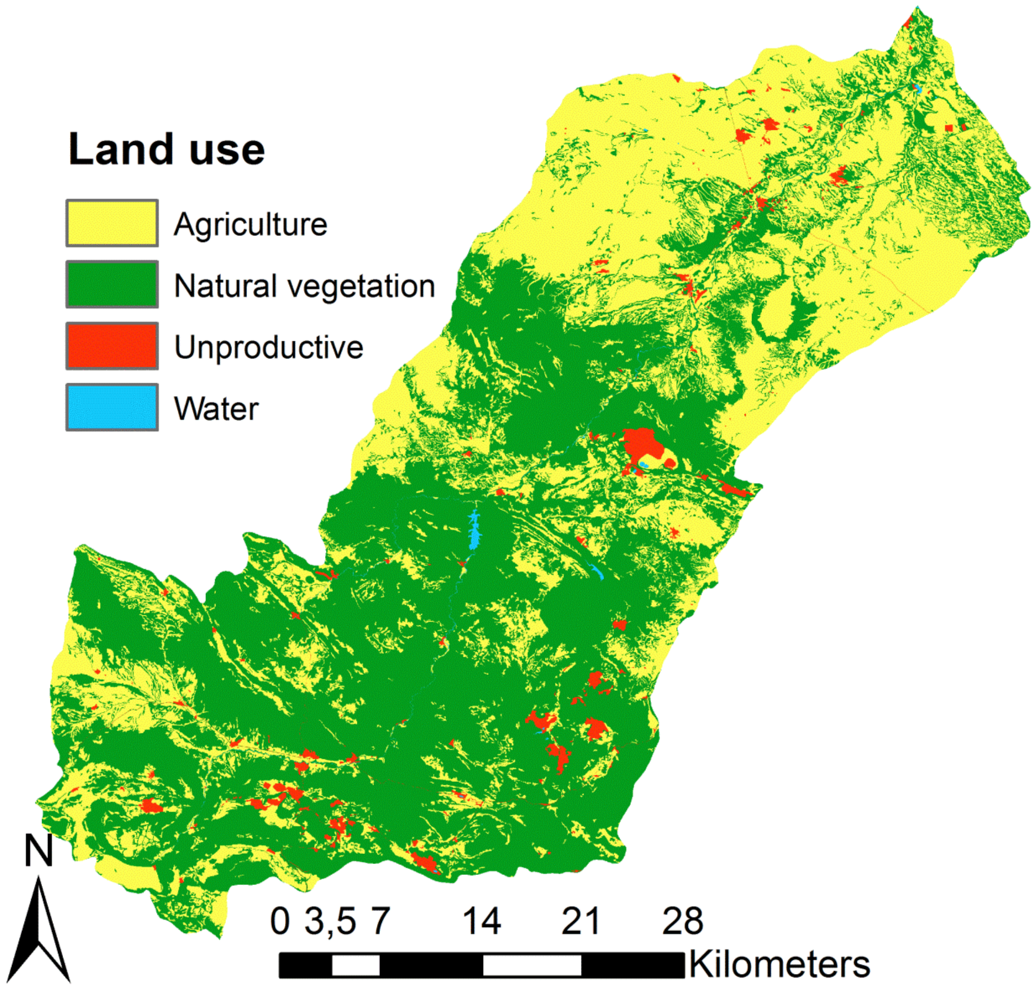


Fig. 6. Simplified map of land use in Martín Basin, red areas represent mines, quarries and towns.

Regosol is the most widespread soil type in the Martín River basin, covering 41% of the total area. This soil is composed of medium and fine-textured materials derived from a wide range of rocks, which are normally extensive in eroding lands (FAO-UNESCO 1988), particularly in arid and semi-arid areas and in mountain regions (Sánchez-Andrés et al. 2010). Rendsina-lithosol and cambisol, which are shallow soils with medium and fine-textured materials, cover 11.7% and 12.6% of the Martín Basin, respectively (Fig. 7). Calcic yermosol, defined as a surface horizon that usually consists of surface accumulations of rock fragments ("desert pavement") embedded in a loamy vesicular crust and covered by a thin aeolian sand or loess layer, extends over 8% of the study area. These qualities make these soils prone to erosion if combined with land cover-management misuse and steep slopes.

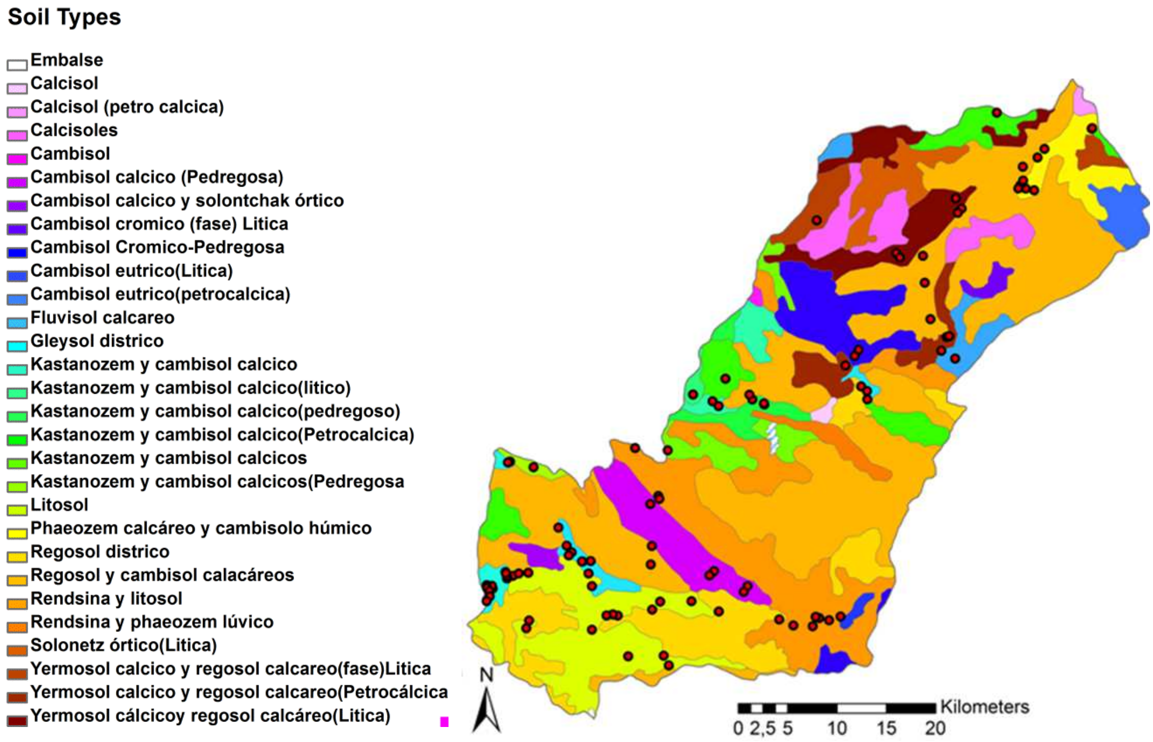


Fig. 7. Description of the study area by soil types showing with red-black dots soil sampling spots for soil analysis.

A large coalfield is located in the southern part of the basin (Fig. 3 and Fig. 6). Mining was the main socioeconomic activity for people living in this region from 1960 to 1990. After a period of great prosperity of opencast mining during the 1980's with 17 active opencast mines (27 Km^2), the activity has strongly declined and only three mines are currently operating (Comín et al. 2009). The mining zones (Fig. 8) contribute to the emission of sediment according to their restoration status.

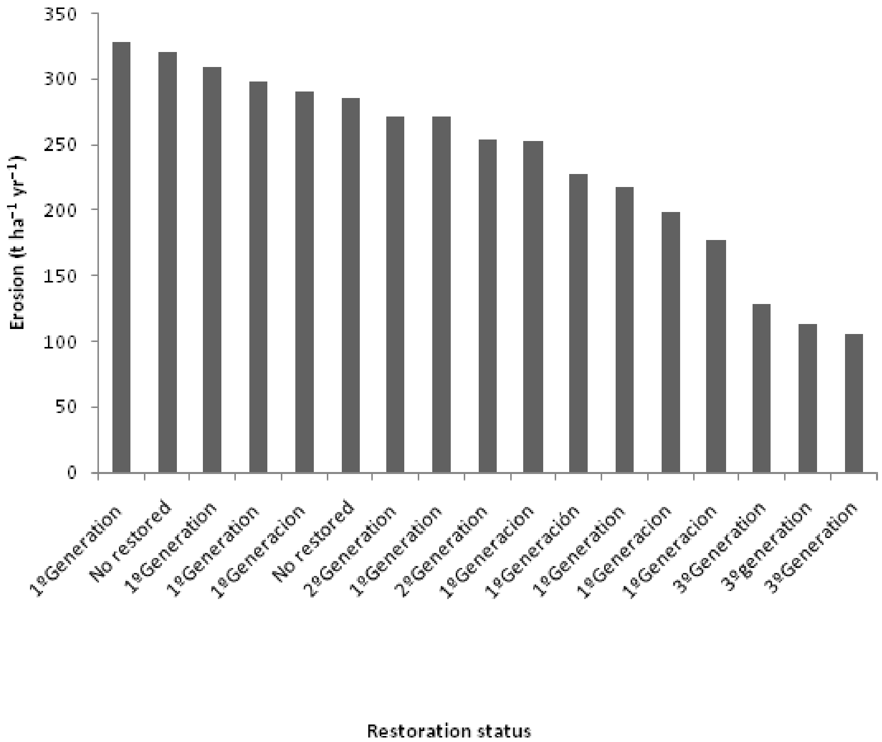


Fig. 8. Erosion rates by mine restoration status, the lower values correspond with the third generation restoration status (From Trabucchi et al.2012a).

These mining zones are classified as abandoned (Fig. 9 A, B), first generation (Fig. 9 C, D, p. 37), second generation Fig. 10), and third generation (Fig. 11). The first generation mines were restored by depositing materials following the platform-slope-ditch model during the 1980s. These mines have large areas of steep slopes (>22º), with ditches formed from rill erosion connected to the river network and an absence of pits (Fig. 9 C, D). The second generation group of mines was restored with the same model but with slopes of 15º and deep pit zones that accumulate runoff discharges (Fig. 10 C, D). Moreover, these mines have received extensive application of soil and plant material to restore the plant community (Fig. 10 A, C). The third generation mines were subject to a topographic restoration model that tries to simulate natural landforms and recreate a natural drainage to decrease peak flows (Nicolau 2003) (Fig. 11.). In addition to the three groups described above, a few mines abandoned after exploitation remain as non-restored mine zones in the region (Fig. 9 A, B,) due to the fact that laws forcing to mine companies to restore exploited lands were established in 1985.

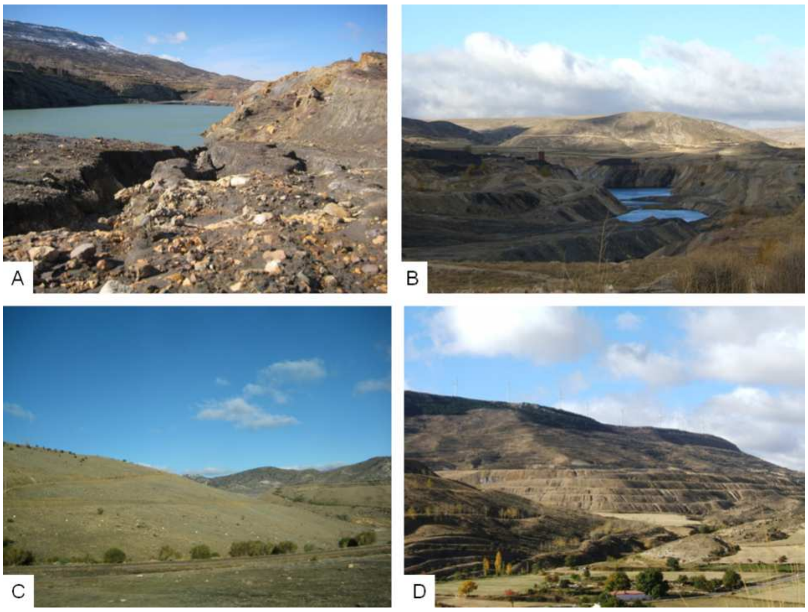


Fig. 9. Abandoned opencast coal mines (A and B, Palomar mine), where runoff reaches water bodies most of the time. Example of first generation mine restoration following the platform-bank model with a pyramidal topography of 30 degrees (C Murcielago mine in Utrillas and D Cueva la Hiedra in Montalbán).

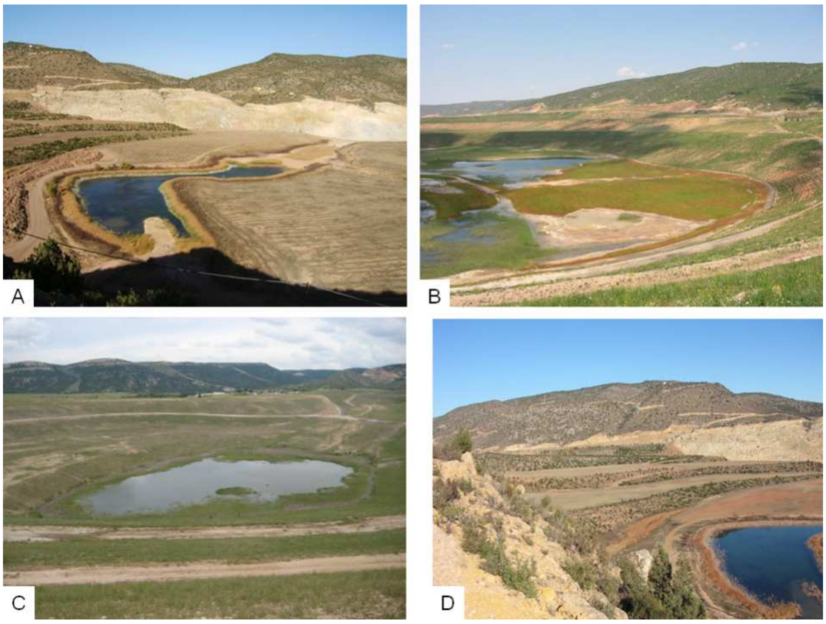


Fig. 10. Second generation of restoration (Barrabasa, A, and Corta Alloza, B, in Val Ariño) with a gentle slope (≤22º degrees) including creation of constructed wetland as part of the drainage system.

Fig. 11. Third generation of restoration (Utrillas) based on the understanding of the hydrological basin as a restoration unit. Reservoir works as a firebreaks line avoiding sediment emission to the natural watercourse and reducing peak flow.

PICTURES OF THE STUDY AREA

Foto 1. Tipos de zonas agrícolas frecuentes en la parte norte de la cuenca del Martín: Agricultura mecanizada con métodos tradicionales (en sentido horario desde la izquierda arriba) dos campos utilizados para secano que durante el invierno se quedan completamente expuestos a los agentes erosivos (Escatrón, Hijar). Campo dedicado a secano protegido residuos de cultivo cerca de Albalate del Arzobispo.

Foto 2. Diferentes paisajes dominantes en la parte central de la cuenca del Martín. En el sentido horario desde arriba a la izquierda: Olivares, campos de almendros y viñas; paisaje prevalentemente agrícola con una mayor componente natural de matorral; zona más rocosa y abrupta dominada por bajo matorral a la embocadura del embalse de Cueva Foradada conocida por su alto valor recreativo al interior del Parque Cultural del Río Martín; paisaje agrícola en Alloza.

Foto 3. Paisajes típicos de la parte sur de la cuenca (en sentido horario desde arriba e izquierda): Penyarroya, una atracción natural del Parque Cultural del Martín, se aprecian los bosques riparios formados por caducifolias; zona forestal en el municipio de Utrillas. "Humanización" del río en Obón, el bosque de ribera ha sido substituido por cultivos y caminos; zonas encañonadas del Río Martín, nótese los fenómenos de desprendimiento y acumulación de roca en las laderas de los montes

3. Methods

3.1. *Estimating erosion with RUSLE-GIS model*

Erosion rates have been estimated at the regional scale using the RUSLE model (Fu et al. 2005, Onori et al. 2006; Pizzuto 2002; Pennisi 2004). European environmental researchers (Panagos et al. 2011) have recently mapped a soil erodibility dataset at the European scale. The objective was to overcome problems of limited data availability for the application of the USLE (Universal Soil Loss Equation) model and to present a high quality resource for modelers who aim to estimate soil erosion at the local/regional, national or European scale. Following this direction, the location of eroded areas and the estimation of the average annual soil loss from rill and sheet erosion in the Martín Basin (Ebro Basin, Northeast Spain) were determined by using the RUSLE (Renard et al. 1997) and an updated version of USLE (Wischmeier and Smith 1978), coupled with GIS (Geographic Information System).

Many authors have used GIS/RUSLE models to estimate sheet wash erosion and non-point source material discharges in watersheds (Fu et al. 2005; Lim et al. 2005; Smith et al. 2007) and for environmental assessment (Boellstorff and Benito, 2005; Erdogan et al. 2007; Ozcan et al. 2008). An increasing number of studies on restoration ecology are using this model to identify potential restoration areas (Güneralp et al. 2003; Vellidis et al. 2003) and to design reclamation plans for degraded areas such as opencast mines (Toy et al. 1999; Martín-Moreno et al. 2008; Moreno-de las Heras et al. 2009).

Despite some uncertainties regarding RUSLE, such as the overestimation of soil loss on plots with low erosion rates and the underestimation of soil loss on plots with high erosion rates (Nearing 1998; Risse et al. 1993), we decided to use this model because it requires data that are relatively common and inexpensive to be processed with GIS. One of the highlights is the formulation of results that can be used for comparative or complementary future studies (Millward and Mersey 1999; Wang et al. 2003; Beguería 2006).

3.1.1. The RUSLE model

We used GIS commercial software (using a Spatial Analyst tool) to examine spatial variations in erosion using elevation data at a 20-m grid scale within the study area. Digital land cover data are available as shape files at the Aragon Territorial Information Centre (CINTA 2006). The Universal Soil Loss Equation (USLE) was used for this study because it is the most used empirical model that assesses long-term averages of sheet and rill erosion. This model is based on plot data collected in the USA (Wischmeier and Smith 1978). The USLE and its adapted version RUSLE (Renard et al. 1997) have been applied to various spatial scales and region sizes in different environments worldwide (Vrieling 2008).

The USLE and RUSLE are statistically based water erosion models related to six erosion factors (for a detailed description of the factors and data collection methods, see the appendix at points 9.3 and 9.4):

$$A = R * K * L * S * C * P$$

Where:

A is the average soil loss from sheet and rill erosion, reported here in tons per hectare per year (t ha^{-1} yr^{-1}) (Fig. 17, p. 65).

R is the rainfall-runoff factor and represents the erosion energy in MJ mm ha^{-1} h^{-1} yr^{-1} based on the methodology of Renard et al. (1997), and it represents the average annual summation (*EI*) values in a normal year's rainfall (Fig. 12 B).

K is the soil erodibility factor, which represents both the susceptibility of soil to erosion and the rate of runoff, as measured under the standard unit plot condition expressed in (t h MJ^{-1} mm^{-1}) (Renard et al. 1991) (Fig. 12 A).

Only *R* and *K* have units; those units, multiplied together, give erosion in units of mass per area and time. Each of the other terms scales the erosion relative to specified experimental conditions (>1 is faster than erosion under those experimental conditions, and <1 is slower). The remaining factors are non-dimensional scaling factors.

LS is the topographic factor describing the combined effect of slope length and steepness and is calculated with the approach of Moore and Wilson (1992) (Fig. 13 C),

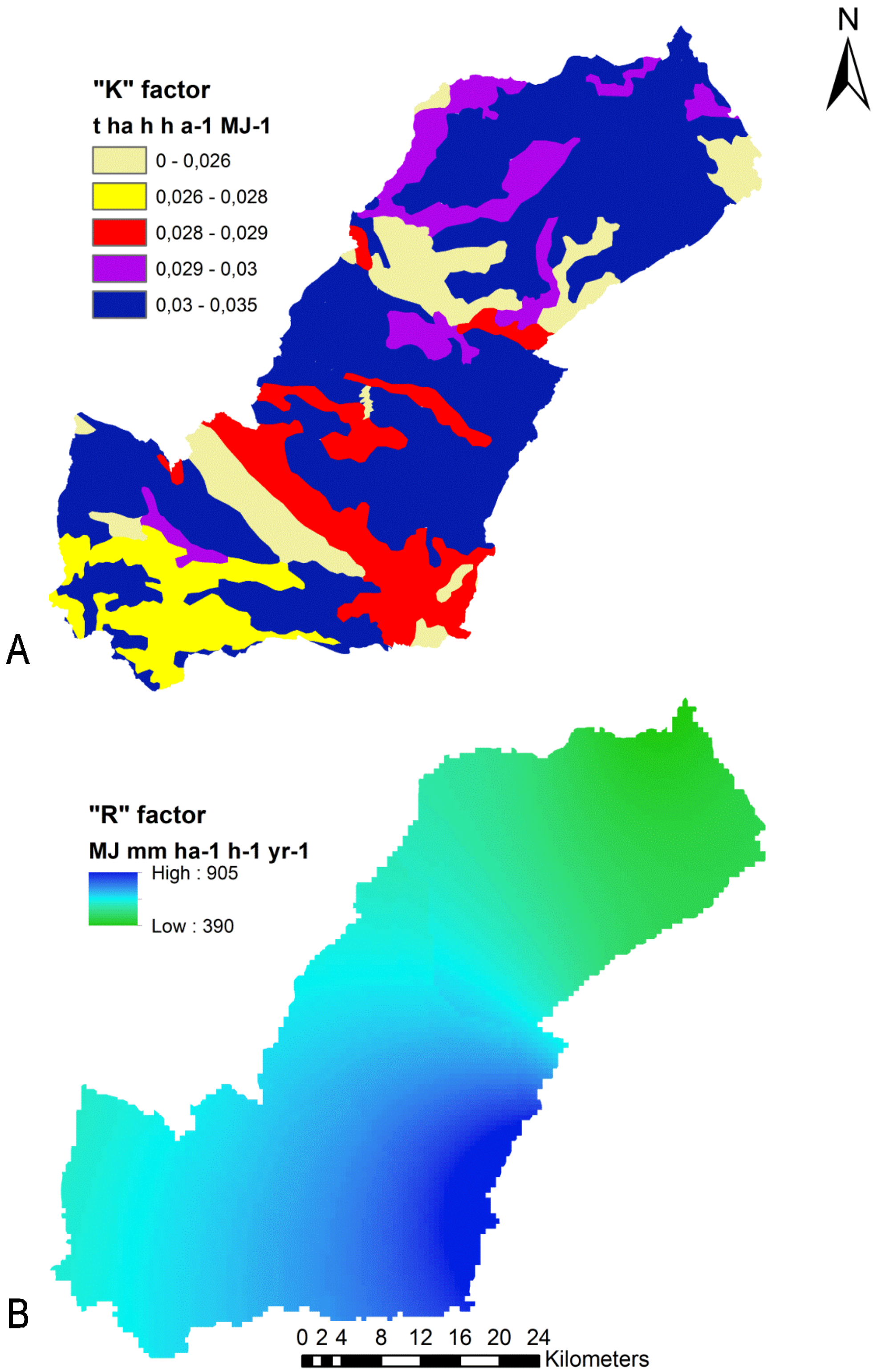


Fig. 12. Input data derived from the database of the Martín watershed: A) soil erodibility map (K-factor in RUSLE, Mg h MJ^{-1} mm^{-1}); B) rainfall erosivity map (R-factor in RUSLE, MJ mm ha^{-1} h^{-1} yr^{-1}).

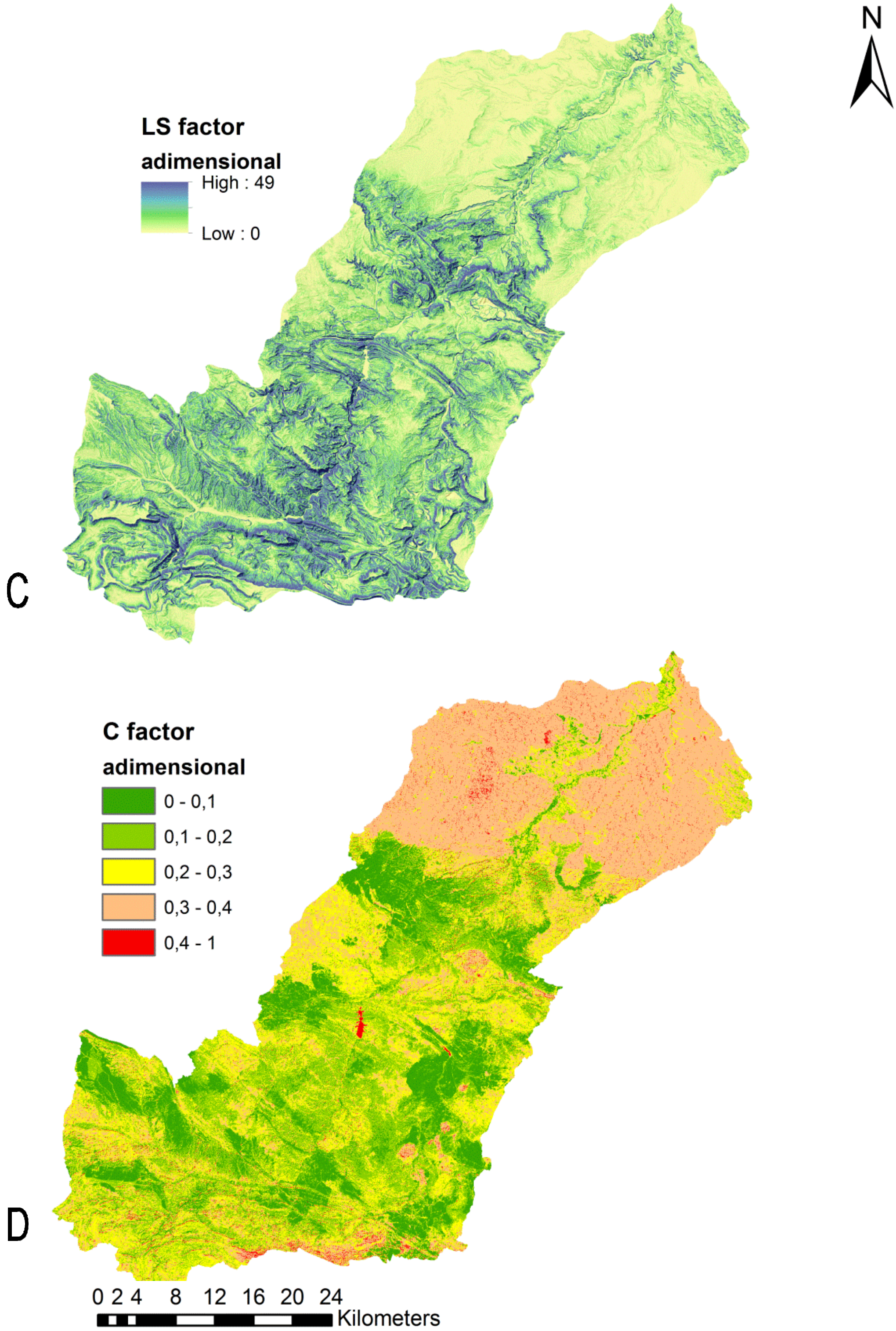


Fig. 13. Input data derived from the database of the Martín watershed: C) length slope factor (LS-factor en RUSLE) and D) crop management.

From the standpoint of soil conservation planning, the *C* factor is the most essential factor because land use changes that characterize, reduce or increase soil erosion are represented by this factor (Khanna et al. 2007); however, the *C* factor is also the most costly (in time, at least) to estimate locally and then to extrapolate from the local measurements to the entire system of interest. Vegetation cover acts as a buffer layer between the atmospheric elements and the soil, absorbing most of the energy of raindrops and surface water to decrease the volume of rain reaching the soil surface (Khanna et al. 2007). Soil constantly tilled or disturbed has a maximum potential for erosion (C=1). Soil not recently disturbed has a nominal value of 0.45. Live or dead vegetation and rocks reduce *C*, reaching a maximum of 1.0 in constantly tilled soil. In places where total ground cover by live or dead material remains, *C* is taken as 0. In this study, several field samples were collected to determine the *C* factor following the approach of González-Botello and Bullock (2012). The next step was to extrapolate the punctual *C* factor values to the entire study area using the Genetic Programming methodology described by Puente et al. (2011) to obtain Vegetation Indices (VI's) designed exclusively for our area. For a detailed description of the methodology used to calculate each factor, see the appendix 9.3.1., p. 151.

3.1.2. **Connectivity**

Connectivity means the physical linkage of sediment through the channel system, which is the transfer of sediment from one zone or location to another and the potential for a specific particle to move through the channel system (Hooke 2003). In an attempt to evaluate the sediment connectivity in the Martín River Basin, we created a buffer zone of 500 m wide at the sides of the main channel and its effluents. The area directly connected to the conveyor belt varies over different timeframes or under various flow conditions. We used this buffer size because it reflects a situation of moderate magnitude (Fryirs et al. 2007) over which sediments can readily reach the water without being intercepted by depositional areas. Then, we visually identified (color graduation) the higher eroded areas included in the buffer and marked them. In an effort to locate the areas and test the prediction of the model, we conducted a field and photographic survey in the degraded areas included within the buffer described by the model.

3.1.3. **Statistical analysis methodology**

To assess the relationship between erosion and the available covariates, a Generalized Linear Model (GLM, McCullagh and Nelder 1997) with a Gaussian response was selected. Among the various relevant factors that normally influence erosion, we chose cover, slope (*LS*), and rain (*R*) because they result in the best fit with erosion values. The response (erosion) and one of the covariates (*LS*) were log-transformed to reach normality. The regression models were fitted with the open-source R software (R Development Core Team 2010). For model selection an all-subset regression with *K*-fold cross-validation was performed (Miller 2002), with Bayesian Information Criterion (BIC) as selection criteria. The one-standard-deviation rule was applied for making the model selection more stable and for selecting the most parsimonious and adequate model (Hastie et al. 2009).

3.2. ***Ecosystem services surrogate in Martín Basin, description and analysis***

3.2.1. **Identifying and mapping services**

Identifying and selecting ecosystem services to be mapped should be based upon the ecological problems facing the study area (Wallace 2007). The Martín Basin, as with many other Spanish basins, has been deforested repeatedly, and erosion is a major environmental problem (García-Ruiz 2010) affecting the ecological functioning of the whole watershed. From de Groot et al. (2002), we selected a suite of regulating ecosystem services that are linked to major ecological functions: water flow regulation, surface water supply, carbon storage and soil retention and accumulation. We also investigated the potential for recreation/ecotourism services related to recreational-heritage activities that could be a major alternative or complementary socio-economic activity. We quantified and mapped these services to guide the prioritization of restoration actions and best management practices in the basin. The methods adopted and data used for quantifying and mapping are presented for every service.

3.2.2. **Surface water supply**

Surface water supply relates directly to the quantity of water available for human use. Surface water supply or water provision is predominantly regulated by meteorological factors but is also influenced by terrain features such as topography and vegetation cover, both of which determine the water balance of the ecosystem. Egoh et al. (2008) argued that many studies used volume of water produced and/or accumulated in an area as the ecosystem service surrogate of surface water supply and that runoff is positively correlated with water supply. Following this approach, a raster dataset of total runoff was obtained from the Spanish Integrated Water Information System (SIA http://servicios2.marm.es/sia/visualizacion/lda). Data were extracted from this national dataset and used as a surrogate surface water supply. The raster layer was expressed in mm/year per 1 km resolution cell size (Fig. 15 C). In this region, reservoirs are considered high water supply areas due to their capacity to provide water for human uses, though this is despite the fact that most of this water comes from other ecosystems and that reservoirs are artificially constructed systems.

3.2.3. **Water flow regulation**

Ecosystems can play a key role in regulating surface water flow, which is directly related to the water storage capacity of the ecosystem, the magnitude of the aquifer and characteristics of the vadose zone, the vegetation cover in terrestrial ecosystems and the water retention time in aquatic systems. Water flow regulation reduces the impacts of flooding and drought on downstream communities (Myers 1996). Important ground water recharge areas typically have low surface runoff volumes due to their increased infiltration capacity and high water storage. These characteristics, along with other factors such as plant cover, also limit erosion (Sophocleous 2002). Water recharge areas for the entire Ebro Basin have been mapped by the water authority Confederación Hidrografica del Ebro (CHE) and expressed in mm/year at 350 m resolution cell size (http://iber.chebro.es/geoportal/index.htm) using the Curve Number (USDA-SCS 1972). Data for the Martín Basin were extracted and used in this research (Fig. 14 B). Water flow regulation is an important service within the Martín Basin because of the negative impact of erosion and flooding on both natural and man-made systems. Vegetation cover plays a key role in the delivery of this service, reducing surface flows to nearby waterways. Therefore, reducing forest cover and density decreases moisture retention, which in turn reduces the growth of remaining trees and increases surface water yield from watersheds. These changes can be short-lived, however, and depend on climate, soil characteristics and the percentage and type of vegetation removal.

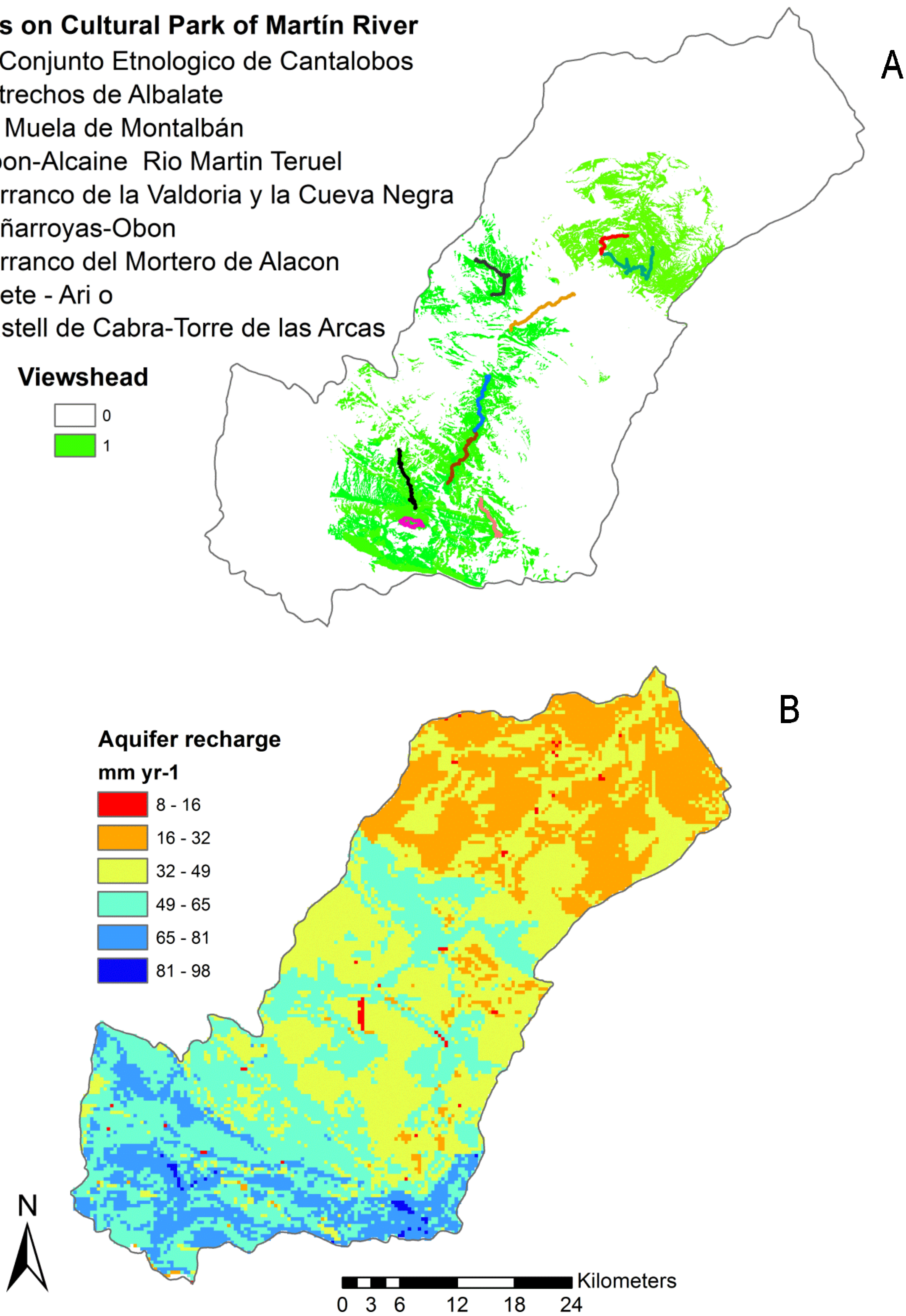


Fig. 14. Ecosystem services surrogates in Martín Basin: ecotourism paths and relative's viewshed (A), aquifer recharge (B).

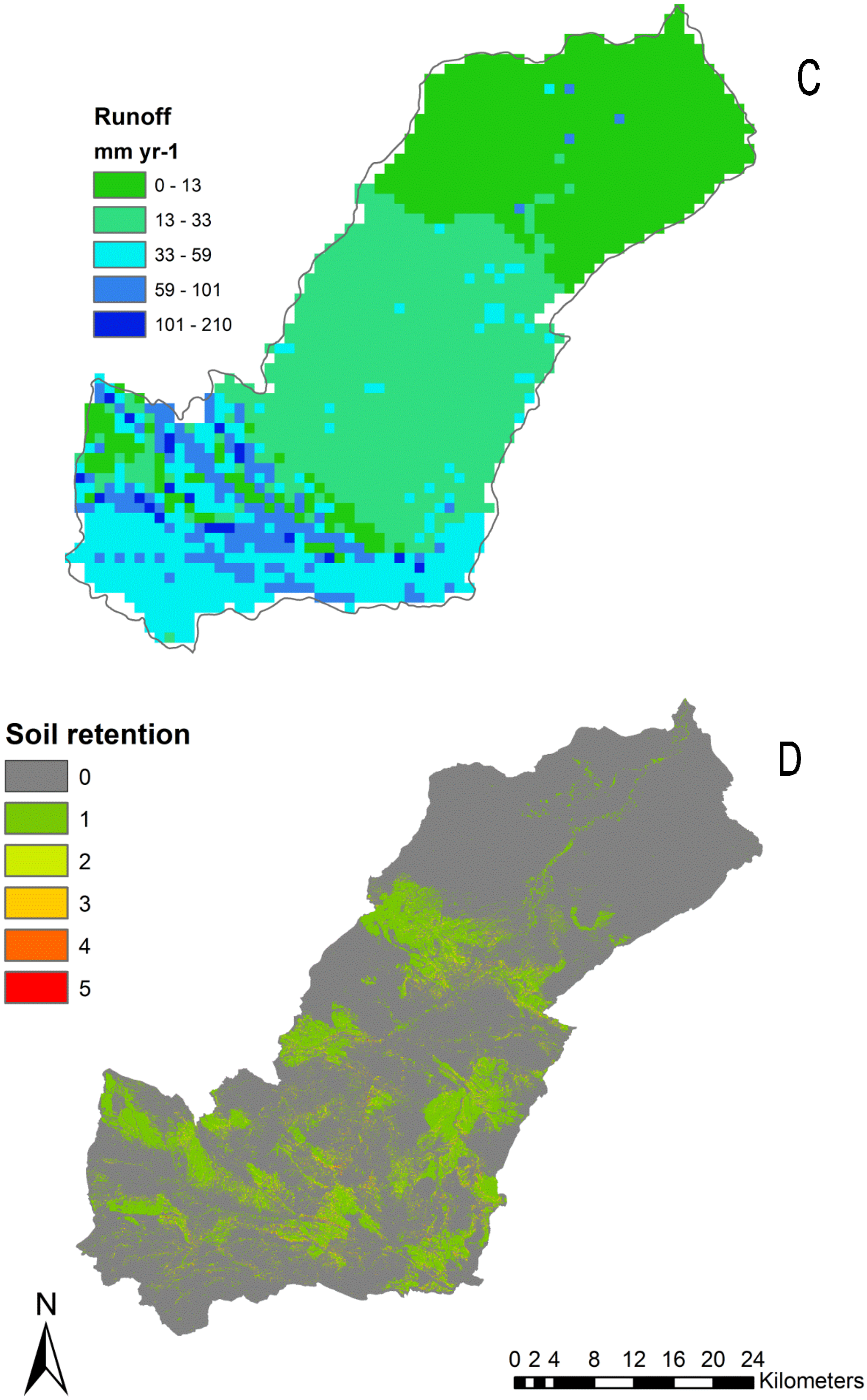


Fig. 15. Ecosystem services surrogates in Martín Basin: runoff (C), soil retention (D).

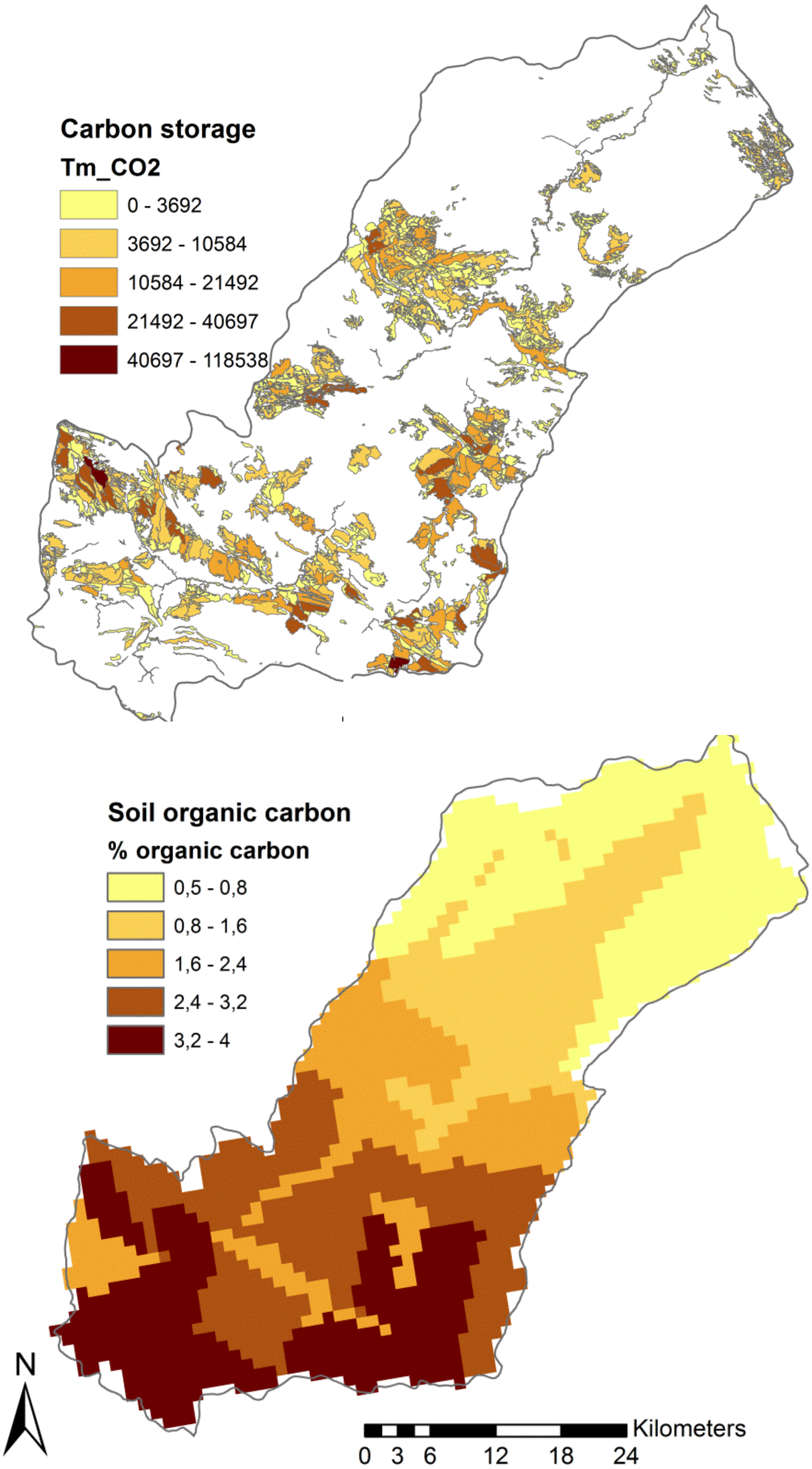


Fig. 16. Ecosystem services surrogates in Martín Basin: carbon storage (E), soil organic carbon (F).

3.2.4. **Carbon storage in woody vegetation**

The amount of carbon stored and its fixation rate was mapped across a large region, which included the Martín Basin, by the Agrifood Research and Technology Centre of Aragon (CITA unpublished, http://www.aragon.es/estaticos/GobiernoAragon/Departamentos/MedioAmbiente/Areas/03_Cambio_dimatico/06_Proyectos_actuaciones_Emisiones_GEI/estudio.pdf). This report focused on modeling different forest management alternatives for CO_2 sequestration, such as woody vegetation, and understanding the role of forests as CO_2 sinks. The method used estimates of biomass and CO_2 conversion using allometric equations (Montero et al., 2005) and data on tree diameters measured during the National Forest Inventory (IFN3 2005). Allometric equations related the diameter of a single tree species to the dry matter existing in different fractions or parts of the tree, i.e., the trunk, roots, leaves and branches of three different sizes. The information, which was linked to the sampling points of the National Forest Inventory, was extrapolated to surface units using the comprehensive 1:50.000 Spanish Forest Map (developed in coordination with the Third Spanish National Forest Inventory). GIS data layers for storage and sequestration rate, expressed in metric tons of CO_2 equivalent (t CO_2 eq), were available for the Martín River Basin in this cited report. The GIS layers were extracted as a polygon layer and converted to a raster layer to facilitate calculation (Fig. 16 E).

3.2.5. **Potential soil retention**

Soil erosion represents a hazard for the long-term sustainability of agriculture and the delivery of ecosystem services (Hajjar et al. 2008). Reduced soil retention results in increased sediment delivery to freshwater systems and degrades these systems (Gobin et al. 2004). Natural vegetation enhances soil retention and plays a vital role in ameliorating the impact of erosion on freshwater systems (Reyers et al. 2009). Quinton et al. (1997) found that a decrease in soil loss was particularly notable when the percentage of vegetation cover increased from 0 to 30% but there was little difference in the soil loss after vegetation cover values exceeded 70%. Trabucchi et al. (2012a) mapped erosion risk in the Martín Basin (expressed in t ha^{-1} yr^{-1}) using the RUSLE model (Renard et al. 1997). To extrapolate vegetation percentage cover, we used the cover factor of the RUSLE model, called the *C* factor (see Appendix 9.1), which is the cover-management term that represents the prior land use, crop canopy and surface cover (Renard et al. 1991) of our study area. Following the methods of Egoh et al. (2008), soil retention was mapped as a function of vegetation cover (%) and soil erosion estimations. Based on these data, vegetation cover densities were distributed in three classes: 0-30%, 30-70% and 70-100% (Fig. 15 D). Areas with vegetation cover greater than 30% and classified as having a very low to low erosion value were defined as having a potential to retain soil. A soil retention *hotspot* was defined as having a plant cover density greater than 70% with very low to low erosion values. Zones with cover densities of less than 30% and high to very high soil erosion values were extracted and identified as erosion-prone areas.

3.2.6. **Soil formation**

Accumulation of soil organic matter is an important process for soil formation and can be easily altered by habitat degradation and transformation (de Groot et al. 2002; Yuan et al. 2006). Organic carbon content (OCTOP) (%) in the topsoil layer (0-30 cm) was mapped by Jones et al. (2005) for the European Soil Database using a 1 km resolution grid cell (Fig. 16 F). Data were expressed as a percentage weight of organic carbon in the surface horizon by combining refined pedotransfer rules with spatial-thematic data layers of land cover and temperature. We used these data as a surrogate measure for the supporting ecosystem-service soil formation. Areas with a high organic content (>3.45%) were classified as *hotspots*.

3.2.7. **Potential recreation and ecotourism services**

Landscape as a visual experience holds considerable societal value. For rural tourism, the landscape is often the main attraction and can add significantly to the quality of life of the surrounding residents (Brabyn and Mark 2011). Agriculture and cattle breeding have historically been the most important social and economic activities in the Martín River Basin, with rural society taking shape around the agricultural and livestock cycles. Mining activities during the second half of the 20^{th} century not only changed the way of life in these rural communities, but it also changed the landscape in many parts of the watershed, particularly in the southern highlands. Since the end of the last century, many efforts have been made to promote tourism in the study area, which is rich in both natural and cultural resources. The basin is popular for its wide open spaces, scenery and the presence of the Martín River Cultural Park (http://www.parqueriomartin.com/en/), which is rich in both cultural heritage, including cave paintings, Iberian settlements and historical monuments, and natural sites, including caves, ravine waterfalls and mountain peaks. All of these cultural and natural sites are on hiking and mountain biking routes. The track locations were downloaded from Wikiloc (2011) and from the official web page of routes in Aragon (Senderos de Aragón 2011). The viewshed tool in ArcGIS (Environmental Systems Research Institute 2008) was used on the selected routes to calculate the potential viewing area (Fig. 14 A), which is important for providing an attractive visible environment for tourists (Reyers et al. 2009). The resultant maps were included as *hotspot* production areas following the methodology of O’Farrell et al. (2010).

While we acknowledge that many other cultural aspects and values exist within this region, these tourism routes and viewsheds capture the potential for attracting visitors and providing socio-economic benefits to the local populations, which are key factors for socio-economic development and could have a major regulating impact on the area.

3.2.8. Mapping spatial distribution of services and *hotspots* at basin and subwatershed scale

Maps of the selected ecosystem services were created following the methods of Egoh et al. (2008) and O'Farrell et al. (2010). In this study, data on surface water supply, flow regulation and soil formation had spatially continuous values that covered the whole basin, while data on the other services had spatially discrete values (e.g., the woody carbon storage layer was limited to forested areas and all other values were considered to be 0).

The original values of the ecosystem services in generally had a Poisson distribution, each map were reclassified into five classes that were determined using a Natural Breaks (O' Farrell et al. 2010) were generated classes are based on natural groupings inherent in the data. Class breaks are identified that best group similar values and that maximize the differences between classes. The features are divided into classes whose boundaries are set where there are relatively big differences in the data values (Environmental System Research Institute 2008). These five classes were renamed as very high, high, medium, low and very low. We assigned the value of 0 to the very low class of surface water supply, flow regulation and soil formation to avoid overlapping these services for the entire area because insignificant values mask potentially interesting results. The rest of the services of our suite have not been modified because they have a lower spatial distribution and include areas with no service flow at all (e.g., carbon storage is limited only in forested areas). Finally, service layers were overlapped one by one, and overlapping percentages were used to describe the spatial relationships between these services.

Hotspot maps were created for every single ecosystem service to identify, manage and conserve high service flow areas by extracting high and very high service values. In addition, multiple *hotspot* zones among services were identified and established by overlapping the *hotspot* layers of each of the different services following the methods of Egoh et al. (2008). Services were then generalized to the forth order catchments, which attempted to highlight the richness of services in every subwatershed by defining areas of land that are drained by a stretch of river of lower order than the main Martín River system. Sixty seven subwatersheds were distinguished in the Martín Basin. To identify service values for the subwatersheds (Fig. 22 B, p. 74), we utilized basin service maps using the GIS Spatial Analyst-Zonal Statistic tool (Environmental System Research Institute, 2008) and selected the majority statistical option (ArcGis resource center 2012), which determines the value that occurs most often out of all cells in the input in_value_raster that belongs to the same zone as the output cell. In our case, the majority statistical option attributes to every subwatershed the most frequent value of overlapping services for all of the cells in that subwatershed. When equal numbers of cells within a subwatershed received the highest and the second highest value, the lower value was assigned to the subwatershed. Despite this limitation, it is still considered to be the best statistical option for creating a general overview (Wu 2004). Following this overview for the whole Martín Basin (Fig. 22 A, p.74) and *hotspot* areas (Fig. 22 C, p. 74), the extraction of detailed overlapped-services maps (Fig. 22 C, p. 74) at the subwatershed scale was conducted. The same Zonal tool using the statistical majority option was applied at a subwatershed scale to select *hotspot* subwatersheds by the number of overlapped *hotspot* services (Fig. 22 B, D, p.74).

This process of downscaling facilitates the selection of areas in the region that are particularly vulnerable to environmental degradation and have a high supply of ecosystem services. We extracted from the erosion map generated by Trabucchi et al. (2012a), the mean erosion value for every subwatershed of the basin using zonal statistics with GIS. We then reclassified the erosion values and generated a new degradation map. Reclassification of this map was based on thresholds for soil formation in the study area defined as lightly (0-12 t ha^{-1} yr^{-1}) (Rojo 1990), medium (12-17 t ha^{-1} yr^{-1}) and highly (>17 t ha^{-1} yr^{-1}) (Moreno-de las Heras et al. 2011) degradation level (

Fig. 23 left, p. 76). This allows us to label subwatersheds according to the provisioning of ecosystem services and degradation status, establish a relative ranking of priorities for restoration actions to recover lost and degraded ecosystem service provisions. Table 1 includes the criteria to prioritize subwatersheds for restoration based on the combination of ecosystem service delivery and environmental risk of erosion on Martín Basin. Our priority is where already service flow and erosion are high because there is an elevated risk of losing these vital services if erosion is not counteracted with restoration/management actions and where restoration can work improving the delivery of ecosystem services.

Table 1. Combined ecosystem services delivery and environmental risk criteria for establishing priority areas for restoration in Martín Basin.

Environmental risk (erosion) → ------------------------ **Ecosystem service delivery** ↓	**Low**	**High**
High	Very low priority	High priority
Low	Tertiary priority	Secondary priority

3.2.9. Soil erosion priority areas

Scale-dependent disturbance dynamics have several important implications for land management (Turner et al. 1994). Martín Basin, as many areas in Spain is affected by erosion due to long history of deforestation, cattle grazing and mining (García-Ruiz 2010). Vegetation growth in the region is limited by semi arid condition (García-Fayos and Bochet 2009; Moreno-de las Heras 2011). Natural ecosystems play a vital role in ameliorating these impacts by retaining soils and preventing soil erosion. Erosion is counteracted mainly by structural aspects of ecosystems, especially vegetation cover and root systems (Gyssels et al. 2005) that can be stimulated with restoration actions, creating synergy among services (Bennett et al. 2009). As example, soil retention can stimulate soil accumulation service that will contribute in the maintenance of water quality in nearby water bodies (de Groot et al. 2002) among many others. Areas requiring these services are those vulnerable to erosion, as determined by the topography, rainfall, soil depth, and texture. Trabucchi et al. (2012a) mapped erosion risk using the RUSLE model in the study area at 20m cell size resolution which is recognized as the most appropriate scale for estimate soil loss in semiarid areas (Ruiz-Navarro et al. 2012). Reclassification of this map was based on thresholds for soil formation in the study area defined as lightly (0-12 t ha^{-1} yr^{-1}) (Rojo 1990), medium (12-17 t ha^{-1} yr^{-1}) and highly (>17 t ha^{-1} yr^{-1}) (Moreno-de las Heras et al. 2011) degradation level (

Fig. 23 right, p.76). Data were extended for every subwatershed as mean using zonal statistics tool. The belonging at one of the three categories above established automatically classified subwatershed of the basin as Low, Medium and High erosion grade.

3.3. ***Regional multi-scale spatial analysis***

3.3.1. **Delineation of subwatersheds among different spatial aggregation levels**

To perform a multi-scale analysis of erosion and ecosystem services, we distributed the basic information on these variables, available at a 20 m cell size, at three levels, or scales of aggregation, moving gradually towards a finer resolution. We used the ARCGIS watershed tool to perform this analysis. Following this approach, we created three drainage networks for the Martín Basin with different numbers of subwatersheds, which are described here.

We use three pixel spatial aggregations suitable for prioritization restoration actions, specifying the limit of pixels for flow accumulation, these being 20000 (level 1), 2000 (level 2) and 1000 (level 3).

The spatial arrangement of the Martín Basin at subwatershed level 1 contained 67 subwatersheds (Fig. 24 A left, p. 79), which presented a minimum area of 1.27 Km^2, a maximum of 120.9 Km^2 and an average of 28 Km^2. The second subwatershed, level 2, included 655 subwatersheds (Fig. 24B left, p.79), with a minimum area of 0.007 Km^2, a maximum of 12.1 Km^2 and an average of 2.87 Km^2. Finally, subwatershed level 3 consisted of 2534 subwatersheds (Fig. 24 C left, p. 79), with a minimum area of 0.006 Km^2, a maximum of 4.15 Km^2 and an average of 0.75 Km^2. These subwatersheds are the functional ecological units for the delivery of the majority of our selected suite of ecosystem services, determining erosion dynamics and planning of restoration actions. Classifying assessment units directly assists in resource management, including restoration.

Ecosystem service bundles and erosion maps were reclassified and summarized for every subwatershed level to create a new prioritization classification consisting of a combination of erosion rate thresholds and a number of services (Fig. 2, p. 22).

3.4. ***Comparison of management units***

To investigate service delivery and erosion at the finest scale, we selected two subwatersheds from the first level presenting contrasting topographic features and land use practices as a case study. Our selection was made to facilitate the assessment and utility of our multi-spatial level approach for prioritizing restoration measures. Subwatershed number 4, located in the northern lowland region and subwatershed number 63, located in the south mountainous area (Fig. 25 A, p.80), were selected for this analysis. They were further investigated at the second and third levels (Fig. 31, p.88) to determine the optimal management area for planning restoration policies and to develop an understanding of how patterns of congruence change with scale. Subwatershed number 4 (Foto 1, p.39) is a fairly homogeneous area that is mostly used for dryland and irrigation agriculture but also contains some remnant patches of shrubland. The erosion rate here was calculated to be 0.2 ± 64 t ha^{-1} $year^{-1}$. In contrast, subwatershed number 63 contains a mix of conifer and hardwood forest, shrubs, grassland-scrublands, abandoned and restored mines (Fig. 9, p.37) and dry agriculture areas. It has a calculated erosion rate of 0.5±165 t ha^{-1} $year^{-1}$.

4. Results

4.1. ***Erosion at the basin scale***

Based on the pixel resolution of the RUSLE model used (20 m cell), the mean erosion value for the Martín River Basin was 13.8 t $ha^{-1}yr^{-1}$, which is just over the maximum tolerable soil erosion that can occur and still permit crop productivity to be sustained economically (2.2 to 11.2 t ha^{-1} yr^{-1}) according to the RUSLE model of soils in the United States.

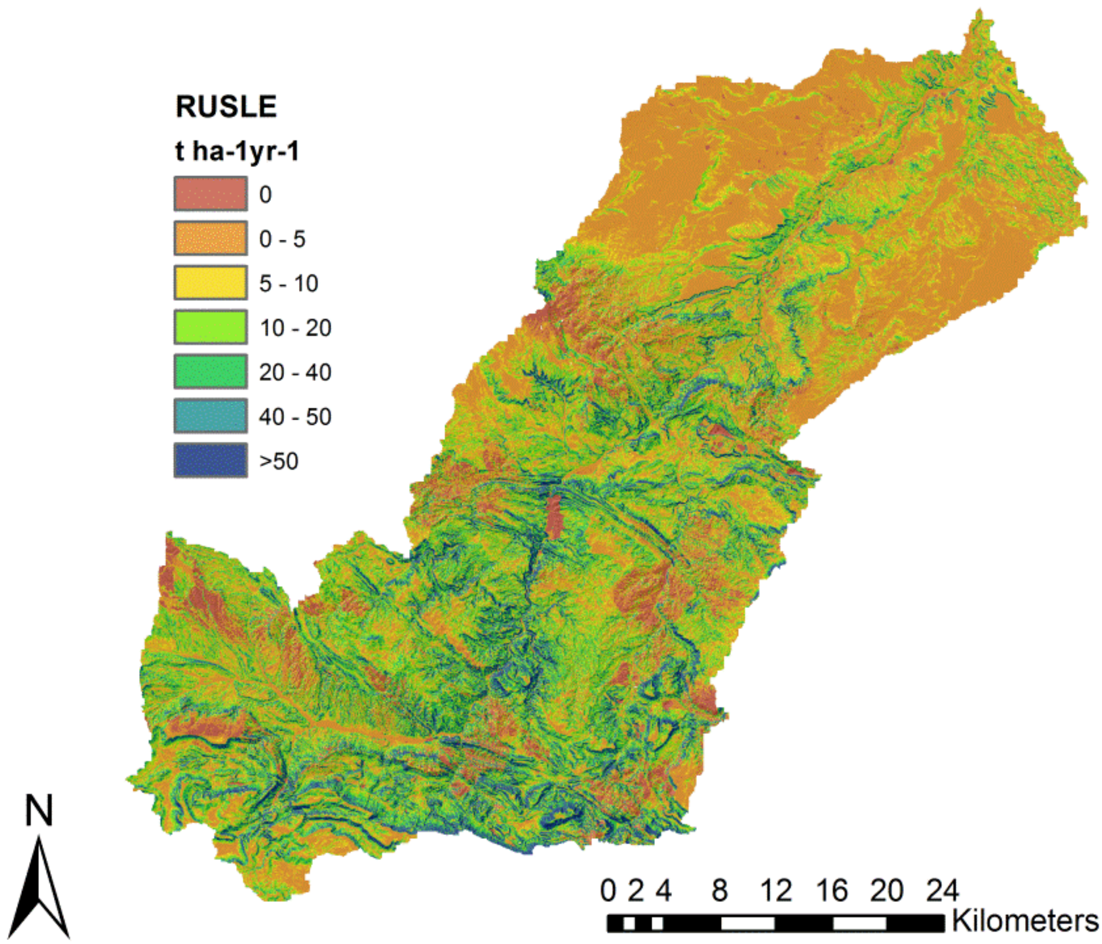


Fig. 17. Map of predicted soil erosion with the RUSLE model (A factor) in the Martín River Basin.

The spatial distribution of potential soil loss rates predicted by RUSLE and the watershed area related erosion rates are shown in Fig. 17. Two-thirds (69%) of the area of the Martín Basin have low and medium soil loss rates (less than 20 t ha^{-1} yr^{-1}), and one-third (31%) of the area, mostly located in the central and southern parts of the basin, has high (18% of the watershed area with 20-40 t ha^{-1} yr^{-1}) and very high (over 40 t ha^{-1} yr^{-1} in 13% of the area) erosion rates. A detailed description of the data estimated for each factor in the RUSLE equation is given in the appendix 9.4. p.151.

The soil loss is at a maximum in rendzina-lithosol, with and area-weighted average (w.a.) of 23.3 t ha^{-1} yr^{-1}, and in regosol, with a loss of 15 t ha^{-1} yr^{-1} (w.a). This soil distribution covers the greatest part of the steep slope areas in the Martín Basin (0≤LS≥49 (Fig. 7, p. 35).

Annual soil losses corresponding to the different land covers are shown in Table 2. Dry farming, which occupies 38.6% of the basin area, has a moderate value of potential soil loss of 10.1 t ha^{-1} yr^{-1}. Grassland-shrubland formations occupy 24.9% of the basin area, with a mean soil loss of 20.2 t ha^{-1} yr^{-1}. The mean estimates for conifers (12% of the basin) and the formations of conifer and hardwood (8%) are 12 t ha^{-1} yr^{-1} and 12.2 t ha^{-1} yr^{-1}, respectively. Scrub, irrigated agricultural, and unproductive land (mines, quarries, urban) cover 9.9%, 2.8%, and 1.5% of the basin area, respectively. Other cover (grassland, olive grove and vineyard, other hardwoods, poplar and aspen, vineyards, fruit trees) occupies 4.6% of the basin area.

Table 2. Statistic value of annual Soil loss (t ha^{-1} yr^{-1}) for the different Land Uses at the Martín River Basin

Land Use and Land cover	Area %	Min	Max	Mean	Standard deviation
Dry farming	38.6	0	403	10	15
Grassland-Shrubland	25	0	650	20	22
Grassland	1	0	290	25	29
Olivier dry	2	0	299	18	22
Vineyard-Fruit tree	1	0	191	12	16
Unproductive	1.5	0	354	23	30
Irrigation	3	0	260	7	12
Scrub	10	0	603	24	28
Poplar and aspen	0.5	0	232	15	21
Other hardwoods	1	0	241	13	20
Conifers	8	0	482	12	20
Conifers and hardwood	8	0	370	12	19

The final statistical model selected according to percentage of explained deviance (92%) and Akaike (1974) information Criteria (AIC) (Konishi and Kitagawa 2008) with a value of 473.8. Finally we selected the following model: log (Erosion) = log (*LS*) + *R* factor + Cover (Fig. 18).

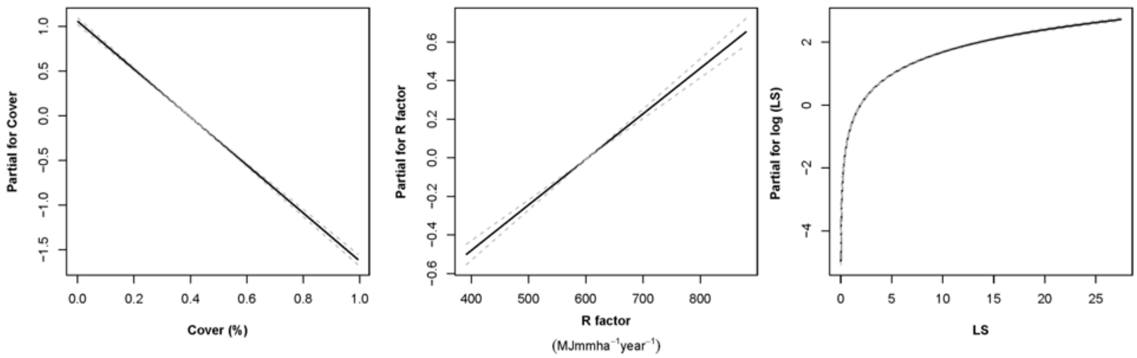


Fig. 18. Estimated effects of the covariates, with standard errors (SE). 95% confidence R factor is interval Cover %, LS factor is used R factor.

The log (*LS*) topographic factor explained 78% of the total explained deviance (Fig. 18), and contributed most of the variability of the values of predicted soil erosion. The percentages of plant cover explained only 21%.

Table 3. Estimated effects of the covariates, with standard errors (SE). Where used LS factor (log (LS) R (Rain) and C (Cover) factor.

	Estimate	SE	t value	p-value
(Intercept)	1.0141	0.0698	14.52	<2e-16
log(LS)	1.0252	0.0096	105.82	<2e-16
RAIN	0.0023	0.0001	18.87	<2e-16
COVER	-2.6843	0.0493	-54.36	<2e-16

For modelling purposes, the variable *C* factor was deleted because it was highly correlated with some covariates and its inclusion would cause co-linearity. In the full model, all possible two-term interactions were added.

Results obtained from all-subset regression with K = 10, (Fig. 19) shown as best models:

Model 1: log (Erosion) ~ log (LS) + Cover + Rain + Cover: log (LS)

Model 2: log (Erosion) ~ log (LS) + Cover + Rain

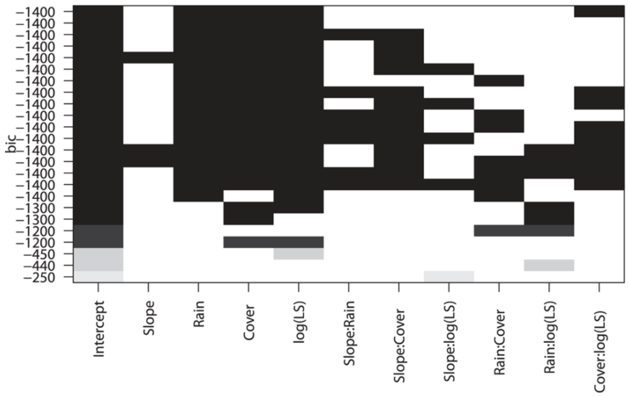


Fig. 19. Bayesian Information Criterium (BIC) of the different models obtained by all-subset regression (":" indicating interaction between the covariates).

According with the one-standard-deviation rule Model 2 was the most parsimonious (Fig. 19), with best Cross-validation score inside the interval $CV \pm s/\surd K$, being s the standard deviation of CV and K the validation samples (CV = 0.04, sd = 0.014, K = 10).

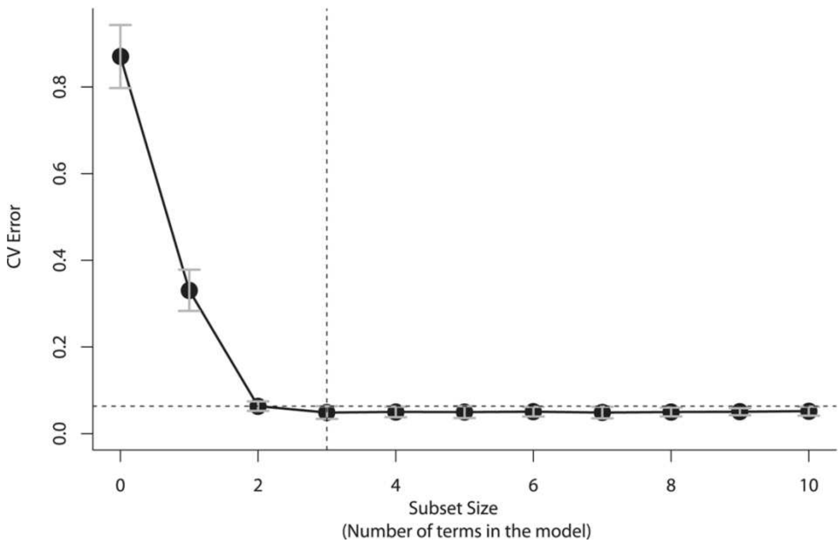


Fig. 20. Model selection with the one-standard deviation rule. (CV = cross-validation).

4.1.1. **Erosion in the coal mines**

In the Martín Basin, 8 mines are in good ecological status, as they are restored and preserved, and 9 are in bad ecological status, as they are either non-restored (3) or restored and degraded (6) (Comín et al. 2009). Five mines are closed basins; they have a surface design simulating natural geomorphology. The RUSLE estimates of soil loss in the mines ranged between 1.4 and 328 t ha^{-1} yr^{-1}. The lowest rates correspond with flat areas created for restoration that are used for dry farming purposes and with wetland areas created in the old exploitation pit, which receive all the drainage of the surrounding areas. Maximum values were registered in very steep ditches, on hill slopes and, overall, in abandoned, non-restored or deficiently restored mines, where it was not possible for plants to colonize because of steep zones and the use of overburden top soil material (Fig. 21). These areas are directly exposed to the eroding power of rainfall, generating high runoff.

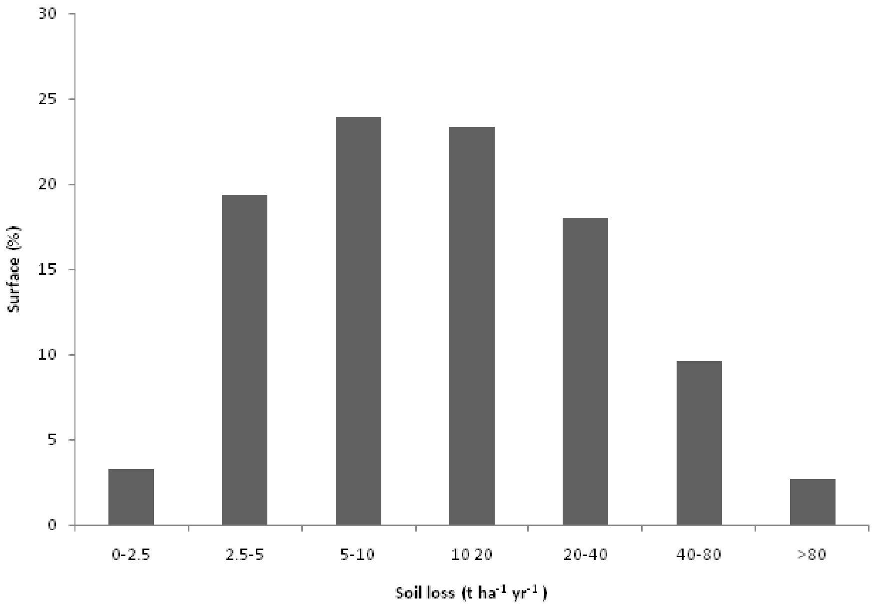


Fig. 21. Histogram of predicted soil erosion with the RUSLE model in the Martín River Basin.

Old, first generation mine restorations following sequences of platform-bank with a slope angle of 22º (Fig. 9, p. 37) have a range of 177-328 t ha^{-1} yr^{-1} for maximum values and mean values ranging between 17 and 54 t $ha^{-1}yr^{-1}$. Abandoned mines have a range of 116-320 t ha^{-1} yr^{-1} as maximum values and 17-44 t ha^{-1} yr^{-1} as mean values. The second generation corresponds to mines where restoration was performed following the same practices as in the first generation with lowered bank slopes (15º) (Fig. 10, p. 37). Intermediate erosion rates were estimated in these mine zones that still in exploitation-restoring process (17-25 t ha^{-1} yr^{-1}) recording maximum soil loss of 184 t $ha^{-1}yr^{-1}$ with medium value of 174 t ha^{-1} yr^{-1}. Micro-watersheds with gentle slopes and a drainage network were created for the mines restored under third generation concepts (Fig. 11, p. 38). In these areas, maximum soil loss estimates range between 106 and 98 t ha^{-1} yr^{-1}, while the mean values range from 16 to 23 t ha^{-1} yr^{-1}. It is clear that applying improved restoration techniques reduces soil loss in mine zones and that non-restored and deficiently restored mines are sites contributing the highest soil loss (Fig. 8, p. 36).

4.2. ***Ecosystem service provision and spatial distribution***

Water flow regulation, surface water supply and soil formation are all widespread services provided by, approximately, 79.5%, 67% and 61.5% of the study area, respectively (Table 4). Recreation and ecotourism is present in 36%, soil retention in 27% and carbon storage in 21.1%. See, Fig. 22 A, p. 74 for a general watershed view of the spatial distribution of the values of the services in Martín Basin.

Table 4. Percentages of the Martín Basin area where the ecosystem services listed are delivered. Between brackets is the percentage of the basin area where these services are delivered as hotspots (with high and very high values for the service).

Ecosystem service	Area (% of the total watershed area)
Water flow regulation	79.5 (42.4)
Surface water supply	67 (7.3)
Soil accumulation	61.5 (19.4)
Recreation/Ecotourism	36 (22)
Carbon storage	21.1 (2.4)
Soil retention	40.2 (19)

Water flow regulation has the largest *hotspot* area, which is defined as the percentage of an area where a given service is valued as high and very high, with 42.4% and carbon storage had the smallest with 2.4% (Table 4). Water flow regulation is governed by rainfall distribution but is strongly influenced by permeable, underlying geology, which is high in the mostly porous soils of the southern part of Martín Basin and facilitates groundwater recharge.

Surface water supply spread throughout the greater part of the basin. The highest values are located in the southern region and coincide with low values of soil formation.

Carbon storage and soil retention depend on the density of canopy cover and are mostly distributed according to an altitudinal pattern. Higher values correspond to a range of 600-

1100 m above sea level. At higher altitudes, both services decline to intermediate values. Certain riparian areas defy this altitudinal trend, having high values for both of these services and showing no relationship to altitude (Fig. 15 D, Fig. 16 E, p. 53-54).

Soil formation is predominantly found in the southern part of the study area, with very low or negligible values identified as one progresses towards the northern lowland areas of the basin.

Recreation and ecotourism services are found in some subwatersheds located in the southern-central and northern-central part of the basin along the river system. Many hiking and mountain biking routes start near the towns of Albalate del Arzobispo, Montalbán and Utrillas and extend outwards.

4.2.1. **Relationship between services**

The greatest overlap of services (3-5 services) was observed in mountainous areas of the south and central parts of the Martín Basin where dense plant cover, woodland and scrubland are located (Fig. 5, p. 33). A relatively small part (14%) of the Martín Basin is not delivering any of the selected suite of services. One and two services are provided in 25% and 25.8% of the basin area, respectively, and three services are provided in 21% of the area (Fig. 22 A).

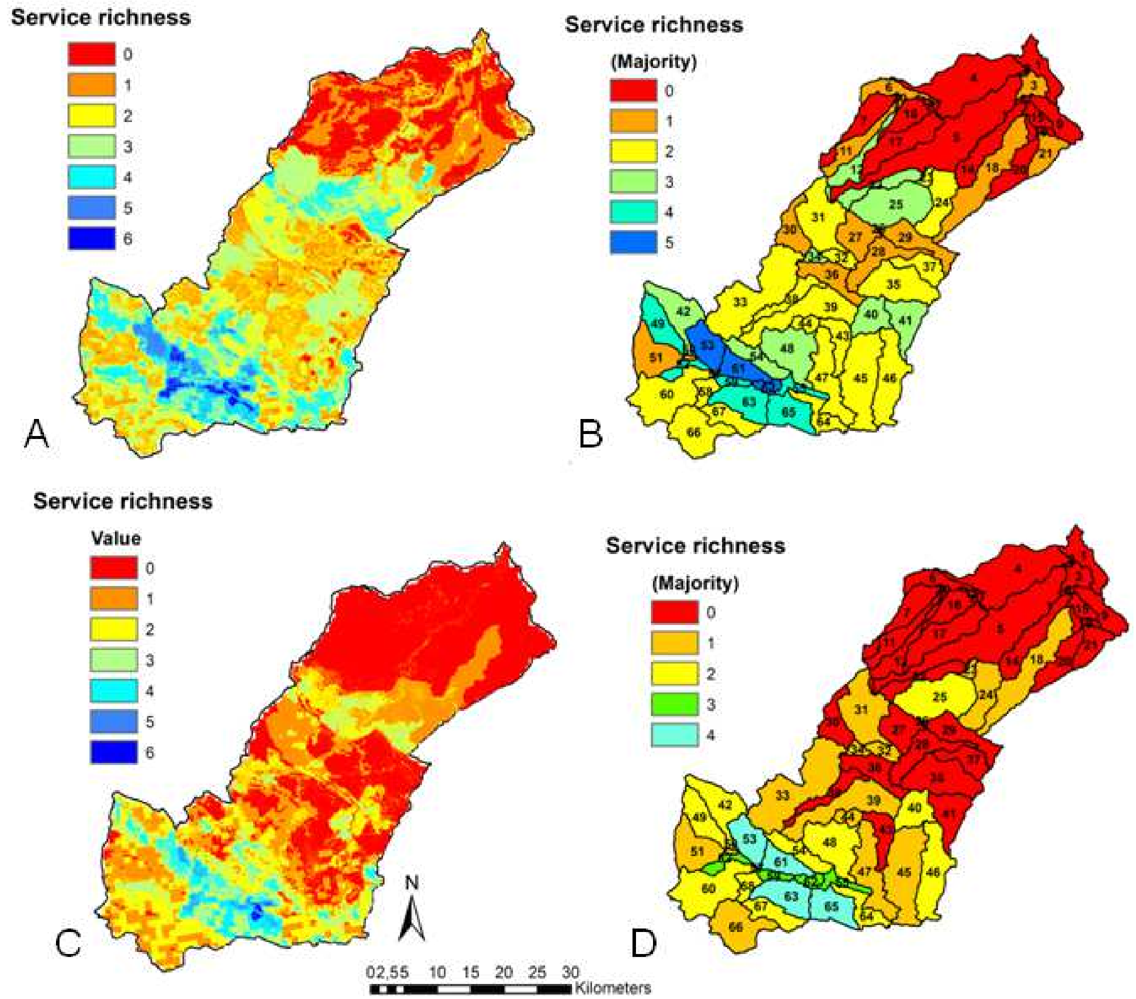


Fig. 22. Ecosystem services richness, number of services from the bundle of ES studied, in the Martín Basin (A). Services richness as hotspots (C). On the right respective service richness per subwatersheds (B) and as hotspot per subwatershed (D). The numbers in the subwatersheds are just correlative numbers to label them.

The spatial overlap among services is low in general. Most of the north and central parts of the watershed do not show any service overlap, while maximum number of services overlapping (5-6 services) is observed in small part of the mountain area in the south (Fig. 22 A). The maximum overlap between services was found between surface water supply and water flow regulation and accounted for 65% of the basin area (Table 5).

Table 5. *Proportional (%) overlap of ecosystem services in the basin and hotspots (hotspots in brackets).*

	Soil accumulation	**Carbon storage**	**Soil retention**	**Water flow regulation**	**Surface water**
Carbon storage	21.1 (1.26)				
Soil retention	10 (5.1)	18.7 (2)			
Water flow regulation	61.1 (16.3)	21 (2.23)	38 (13.2)		
Surface water	59.4 (4.4)	20.7 (0.35)	3.5 (1.95)	65 (6.75)	
Tourism	13.6 (4.3)	6.8 (0.18)	10 (4.5)	22,1 (11.5)	17.1 (5.6)

The percentage area of the basin with overlapped *hotspots* of these two services was 6.75% and was located in the southern region (Fig. 22 A). The soil retention and water surface-supply overlap areas accounted for 3.5% and had an overlapped *hotspot* area of just 1.95% of the basin, which was associated with forest ecosystems. Recreation and ecotourism services have a relatively high overlap with water flow regulation but a small overlap with other services, such as carbon storage and soil retention (Table 5).

The map of overlapped *hotspot* services generated using high and very high values for all of the services shows that a region comprising only 0.12% of the mapped areas incorporated all 6 services. The area is located in the southern part of the basin and corresponds with conifer forest (Fig. 22 C). Conversely, 41% of the basin, mostly in the northern part, is not delivering high or very high values for any service. Most of the areas classified as *hotspots* delivered one service (25.9%), two services (19%) and three services (9.2%). Only a small portion (0.71%) delivered five (Fig. 22 C).

4.2.2. **Subwatershed classification according to ecosystem service provision**

Applying the GIS Spatial Analyst tool and the majority statistic option within the zonal statistic module used to identify the greatest number of services found within each subwatershed, we did not find a subwatershed that provided all six services.

The distribution of the overlapping services by subwatersheds shows the same pattern as for number of services overlapping but let distinguish that subwatersheds 53, 61, 62, 63 and 65 are providing 4-5 services but only subwatersheds 53, 61, 63 and 65 are delivering 4 services as *hotspots* (Fig. 22 B, D). These subwatersheds occupy 3.1% of the total area of Martin Basin in its south part. They were also located in areas classified as having low and medium levels of degradation because of erosion (

Fig. 23).

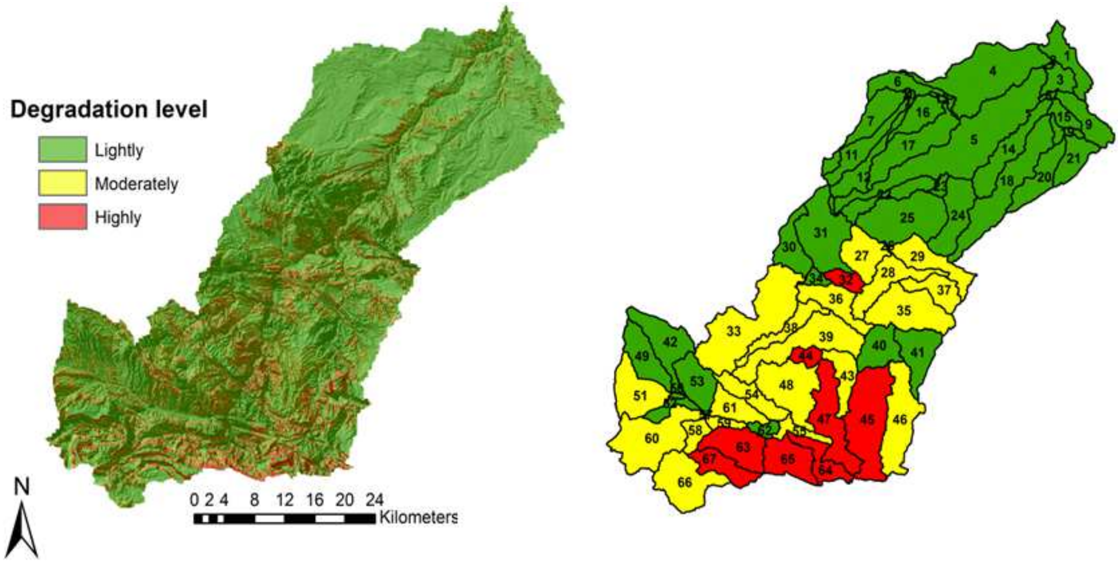


Fig. 23. Left: Spatial simplification of erosion classes in Martín Basin (Light: 0-12 t ha^{-1} yr^{-1}; Moderte: 12-17 t ha^{-1} yr^{-1}; high: >17 t ha^{-1} yr^{-1}. On the Right erosion represented per subwatersheds.

Subwatershed number 62 represents a focal point for surrounding subwatersheds that deliver at least 3 services (Fig. 22 C). In between the southern and the central part of the basin, nine other subwatersheds deliver at least three services (nº25, 22, 34, 40, 41, 42, 48, 54 and 12) and account for 7% of the total area. Nineteen subwatersheds deliver two services and accounting for 36.6 % of the basin area. Mostly of these subwatersheds corresponds with a low degraded status only subwatershed 48 and 54 were classified as having a moderately degradation level (

Fig. 23 right). In contrast, most of the subwatersheds in the northern part of the basin (13 subwatersheds) were delivering just one service, which was most commonly surface water regulation.

4.2.3. ***Hotspot*** **services at subwatershed scale**

Only four subwatersheds were classified as *hotspots* and included up to four services within their boundaries. They are located in the southern part of the basin (subwatershed 63, 65, 53 and 61) (Fig. 22 D). Subwatersheds 63 and 65 incorporate a vast mined area which has been restored (Fig. 3, p. 31), but is still classified as highly degraded, were as subwatershed 61 is mostly covered by conifer and hardwood and has a medium degradation level. All of these subwatersheds are found on steep slopes. In the same part of the basin, there are other subwatersheds (53, 55, 59, 52 and 62) that supply three services and mostly fall with the low and medium degraded level (

Fig. 23 right).

4.3. ***Multi-spatial-scale approach for establishing restoration priorities against erosion through the evaluation of ecosystem services at watershed scale***

Here we present the methodological approach for establishing a hierarchical spatial classification of restoration zones in Martin watershed based on the spatial analysis of erosion rates and ecosystem services assessments.

4.3.1. **Erosion patterns across subwatershed levels**

The landscape heterogeneity of the Martín Basin is a key determining factor explaining the erosion patterns in the region, with the northern area being predominantly flat and the southern area being mountainous, showing a considerable increase in slope, altitude and rainfall patterns. Contrasting the three spatial levels provides us with insights regarding how changes in spatial detail can facilitate the targeting of degraded areas. For example, in Fig. 24 A, we are able to clearly identify areas with high erosion values grouped in the south and a large portion of the northern area showing a low erosion value. By increasing the scale detail from the first level to the second level, we are able to differentiate three erosion thresholds in the northern region (Fig. 24 B, C).

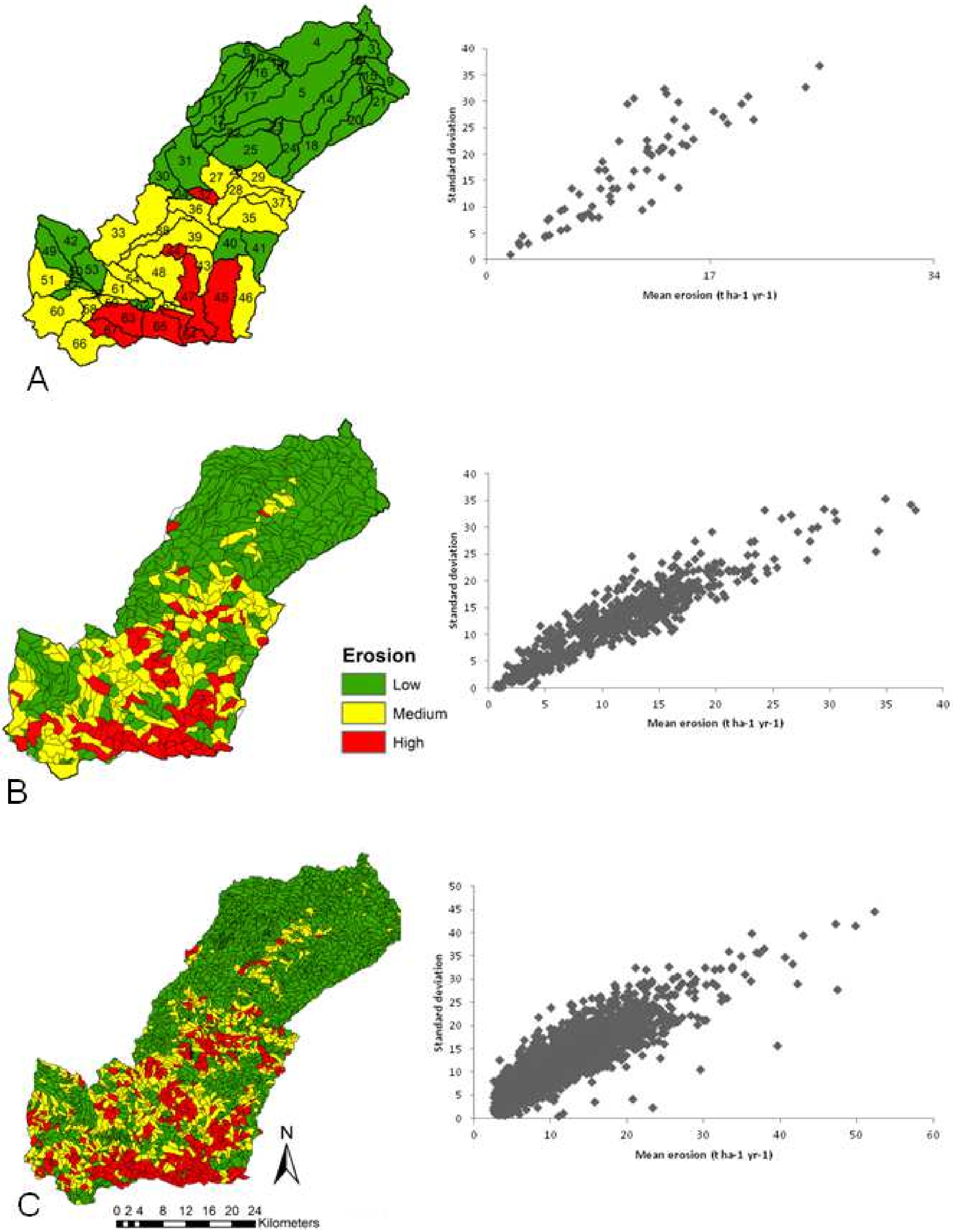


Fig. 24. Erosion map at first (A), second (B) and third (C) level. On the right of each map are plotted the relationship between mean erosion and standard deviation for each subwatershed.

Furthermore, some areas identified at level one as showing low erosion were re-identified as presenting both medium and high erosion regions when examined at the finer detail of level 3, facilitating more precise identification and location of areas for restoration. The results at different scales mostly highlight a fairly constant pattern across these scales (Fig. 24 right). The mean erosion rates (and the calculated standard deviations) exhibit similar values within single watersheds (Fig. 24 A, B, C, right) and a direct relationship was observed between the mean erosion rates and the calculated standard deviations. Subwatershed erosion rates that exceed the highest erosion threshold, indicating areas subjected to a high erosion risk, can be easily identified (Fig. 24 A, B, C). This pattern is repeated across different scales. However, the data dispersion increases as the detail of the analysis increases through the three levels of data aggregation. This is a fairly typical characteristic of ecological data (Levin 1992; Costanza and Maxwell 1994). At the third level, some subwatersheds with high standard deviations and mean erosion values in the low-to-medium erosion threshold range are identifiable (Fig. 24 C).

4.3.2. **Ecosystem service patterns across subwatershed levels**

There is a clear distinction in the ecosystem service supply across the study area (Fig. 22). The northern, lower, reaches of the watershed showed the lowest values, which increased toward the south of the basin. However, at the third level, the ecosystem service supply was highly differentiated (Fig. 26 C).

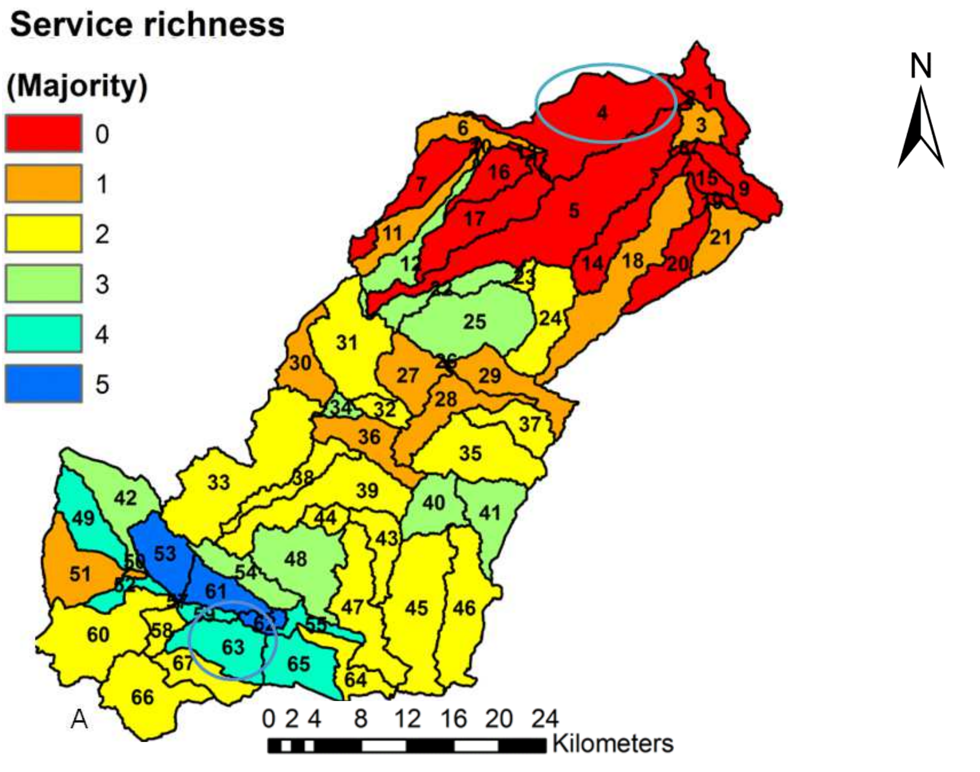


Fig. 25. Ecosystem services bundle map at first level. Highlighted by the blue circle show subwatershed number 4 (North) and 63 (South).

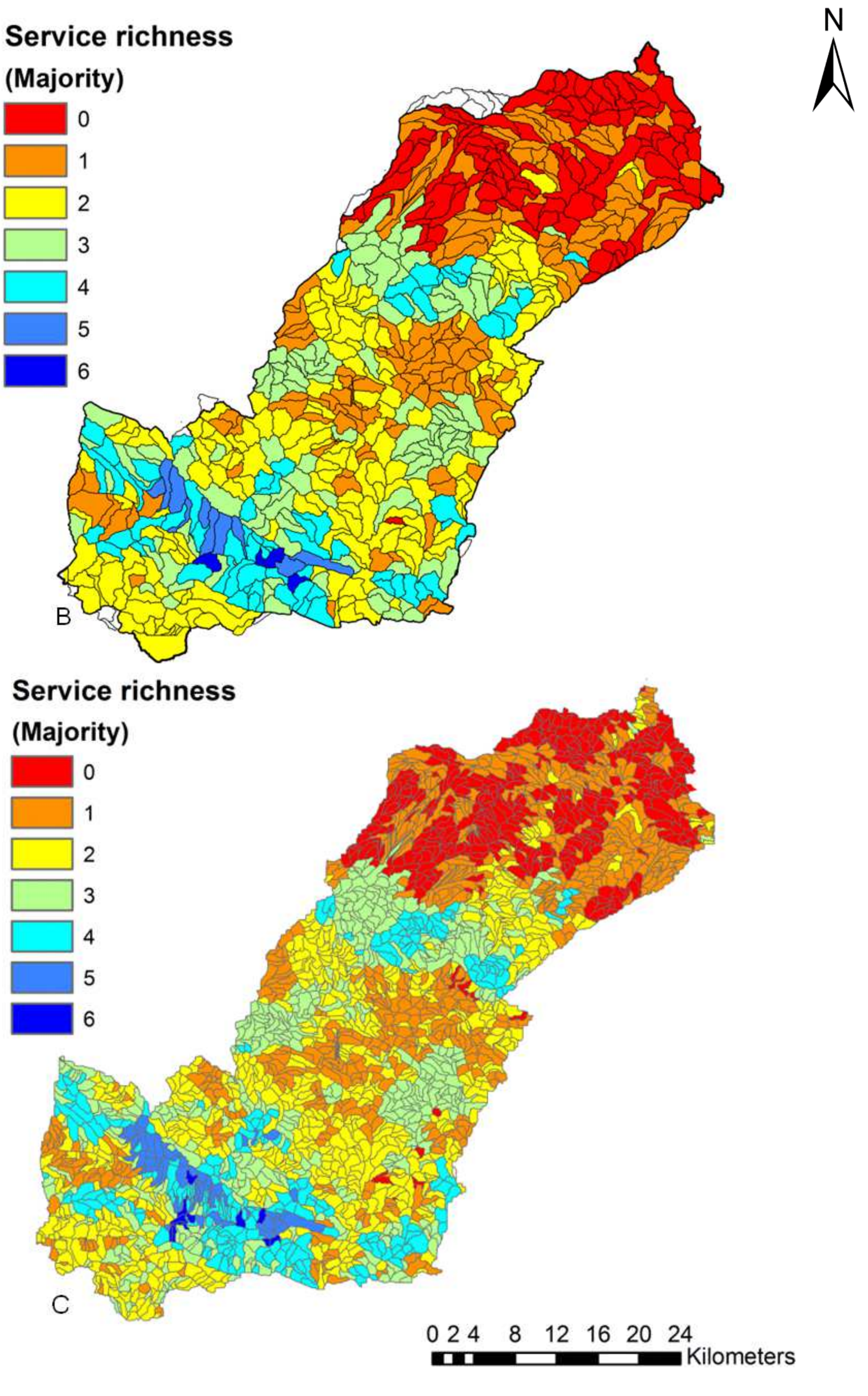


Fig. 26. Ecosystem services bundle map at second (B) and third (C) level.

Increasing the scale of analysis by decreasing the pixel aggregation up to the third level revealed previously masked ecosystem service values (Fig. 26). At the first level, the maximum number of services that overlap at the basin scale was five, but it increased to six as the resolution increased. Our method of calculation also influenced this trend. Here, we used the majority rule, which, when equal numbers of cells within a subwatershed received the highest and the second highest value, assigns the lower value to the subwatershed. In any case, at the lowest scale of pixel aggregation (higher detail), it is at the third level of analysis, the most detailed segregation of ecosystem services related to erosion is observed.

4.3.3. **Hierarchy maps and patterns across subwatershed levels**

In searching for a scale of analysis that offers adequate spatial differentiation of the relationship between the state factor and the degradation factor, we create hierarchy maps and plotted ecosystem service bundle overlaps against the average erosion rates per each subwatershed created in the three aggregation levels analyzed (Fig. 27, 28, 29).

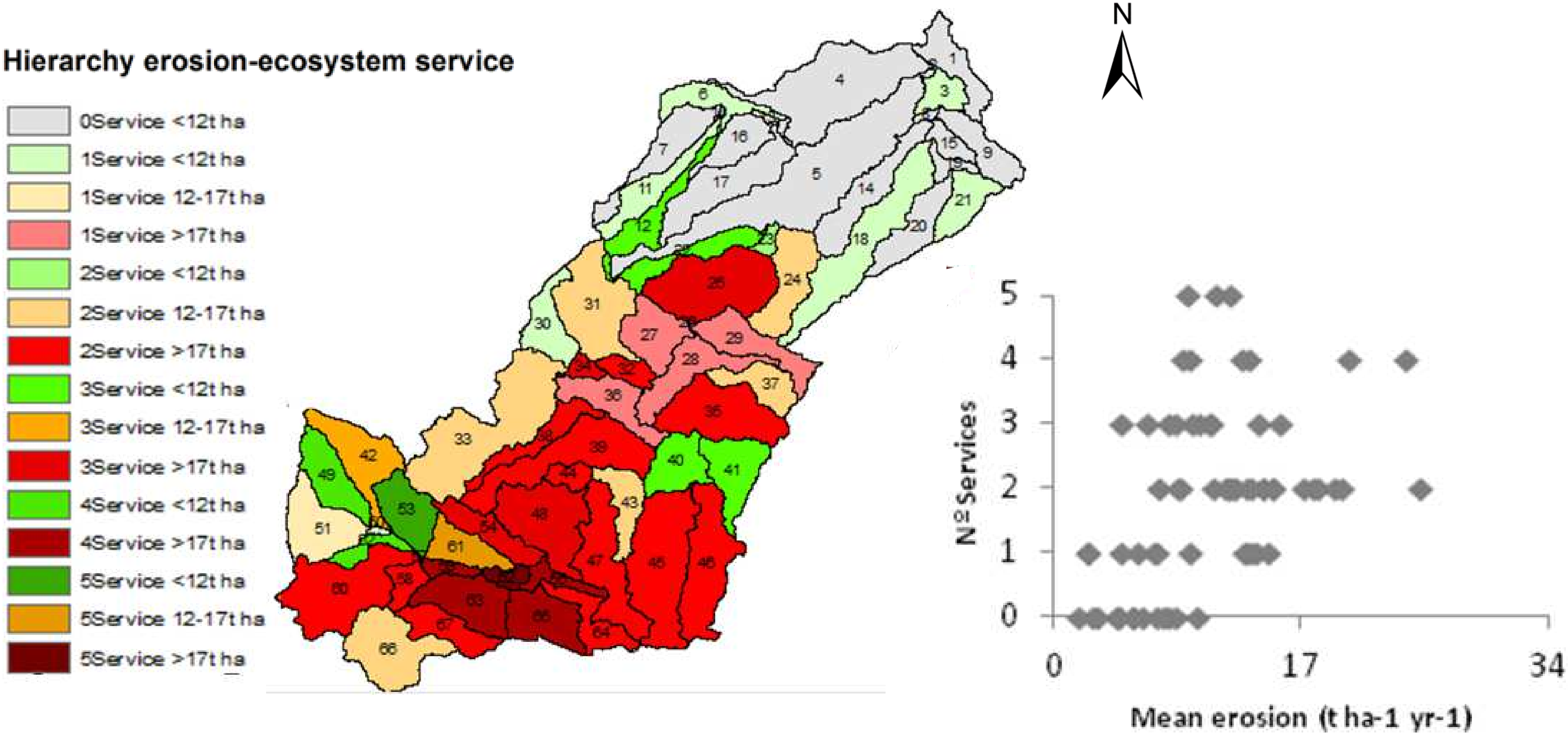


Fig. 27. Hierarchy map at first level (A). On the right is plotted the erosion mean values against numbers of eco.serv. per subwatershed (t ha is a abbreviation of t ha^{-1}yr^{-1}).

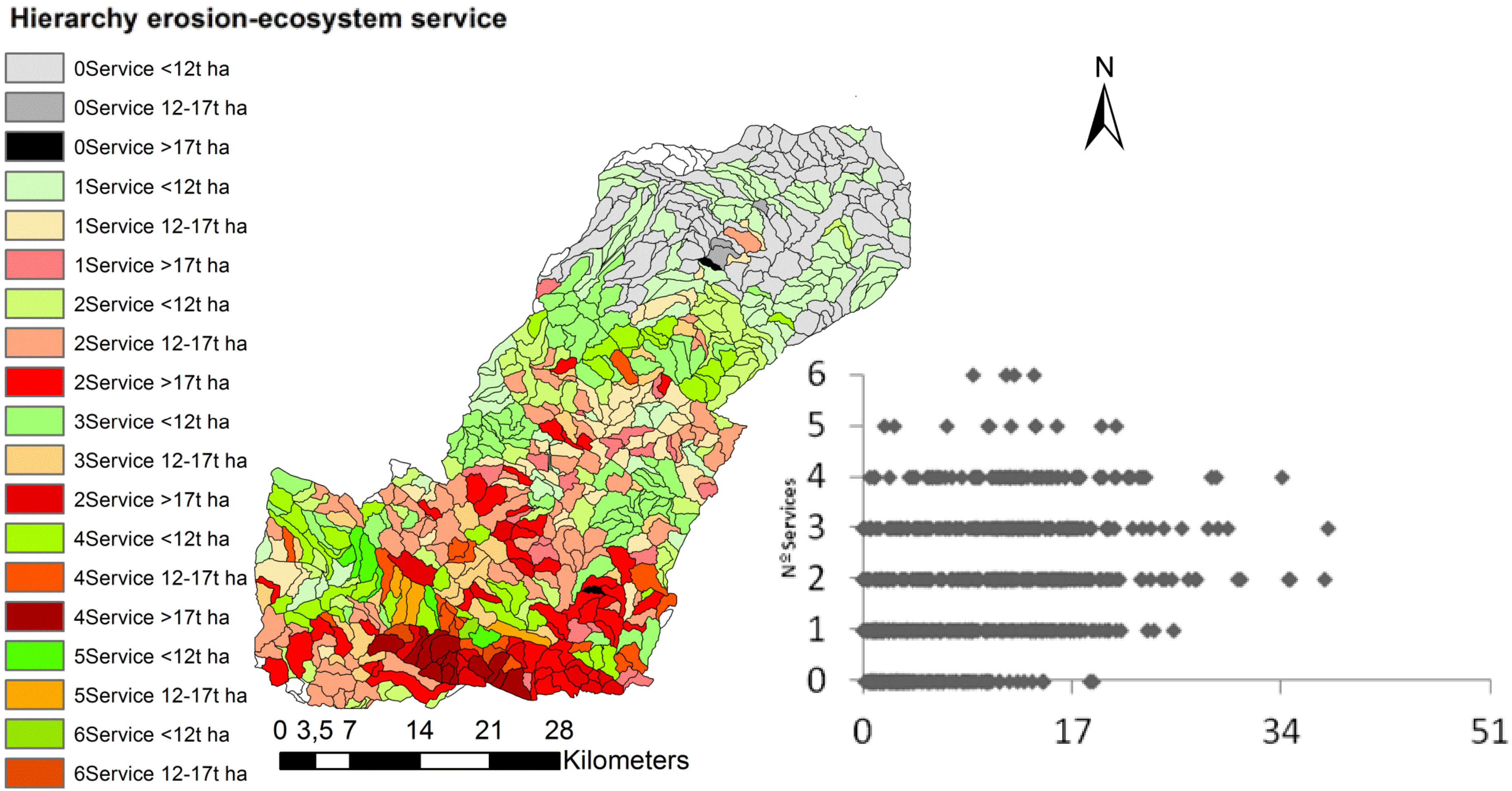


Fig. 28. Hierarchy map at second level (B). On the right is plotted the erosion mean values against numbers of eco.serv. per subwatershed (t ha is a abbreviation of t ha^{-1}yr^{-1}).

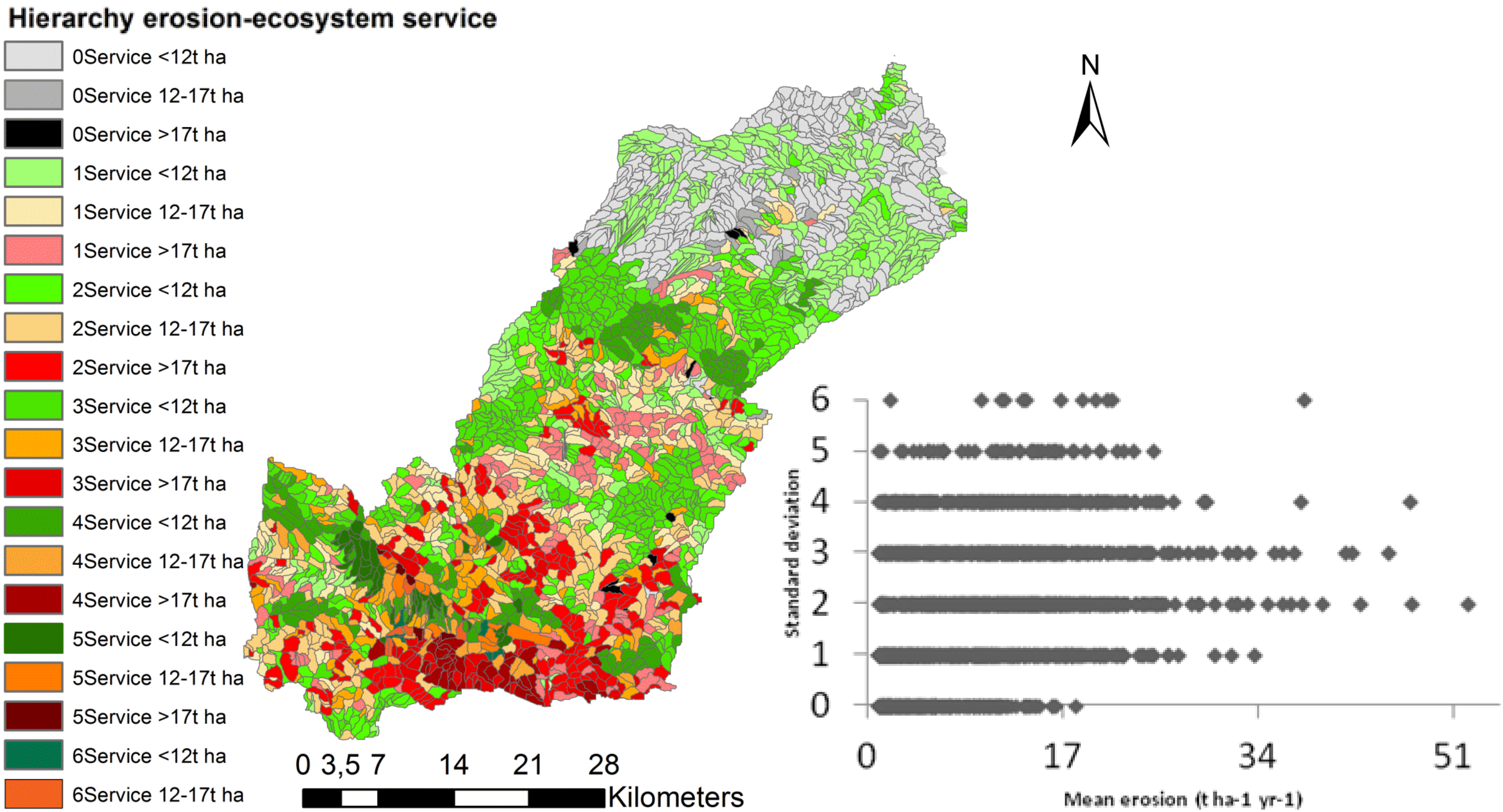


Fig. 29. Hierarchy map at third level (C). On the right is plotted the erosion mean values against numbers of eco.serv. per subwatershed (t ha is a abbreviation of t ha^{-1}yr^{-1}).

The first level of analysis did not highlight any subwatersheds with high erosion rates and either high or low ecosystem service values (Fig. 27). In contrast, at the third level of analysis, the combination of ecosystem services and erosion for these thresholds was clear, highlighting the problem of generalization at the first and second levels (Fig. 28).

Table 6. Combined ecosystem services delivery and environmental risk criteria (<12 (low), 12-17(Medium) , >17 (High) t ha^{-1} yr^{-1}) for establishing priority areas for restoration.

Environmental risk (erosion) → ------------------------ **Ecosystem service↓**	**Low**	**Medium**	**High**
High	Fifth priority	Tertiary priority	First priority
Low	Sixth priority	Forth priority	Secondary priority

Here, we have aligned three erosion thresholds for Martín Basin with high and low ecosystem service supplies, developing priority cases, or scenarios. Cases 3, 4, 5 and 6 present a lower risk of losing services through erosion, and strategies aimed at improving land-use practices should be targeted to these areas. Areas classified as high priority, cases 1 and 2 here, should be considered for restoration action so that ecosystem services vital for the entire basin will be reestablished and maintained. This decision support tool was derived from a data dispersion plot of erosion vs. ecosystem services (Fig. 27 right).

4.3.4. Hierarchical map of management units at the second and third subwatershed levels

The two case study subwatersheds, 4 and 63, provide contrasting examples that demonstrate the differences that are detectable across scales. At the second level, the same spatial heterogeneity is observed for ecosystem service delivery and the associated erosion (Fig. 30 A, B).

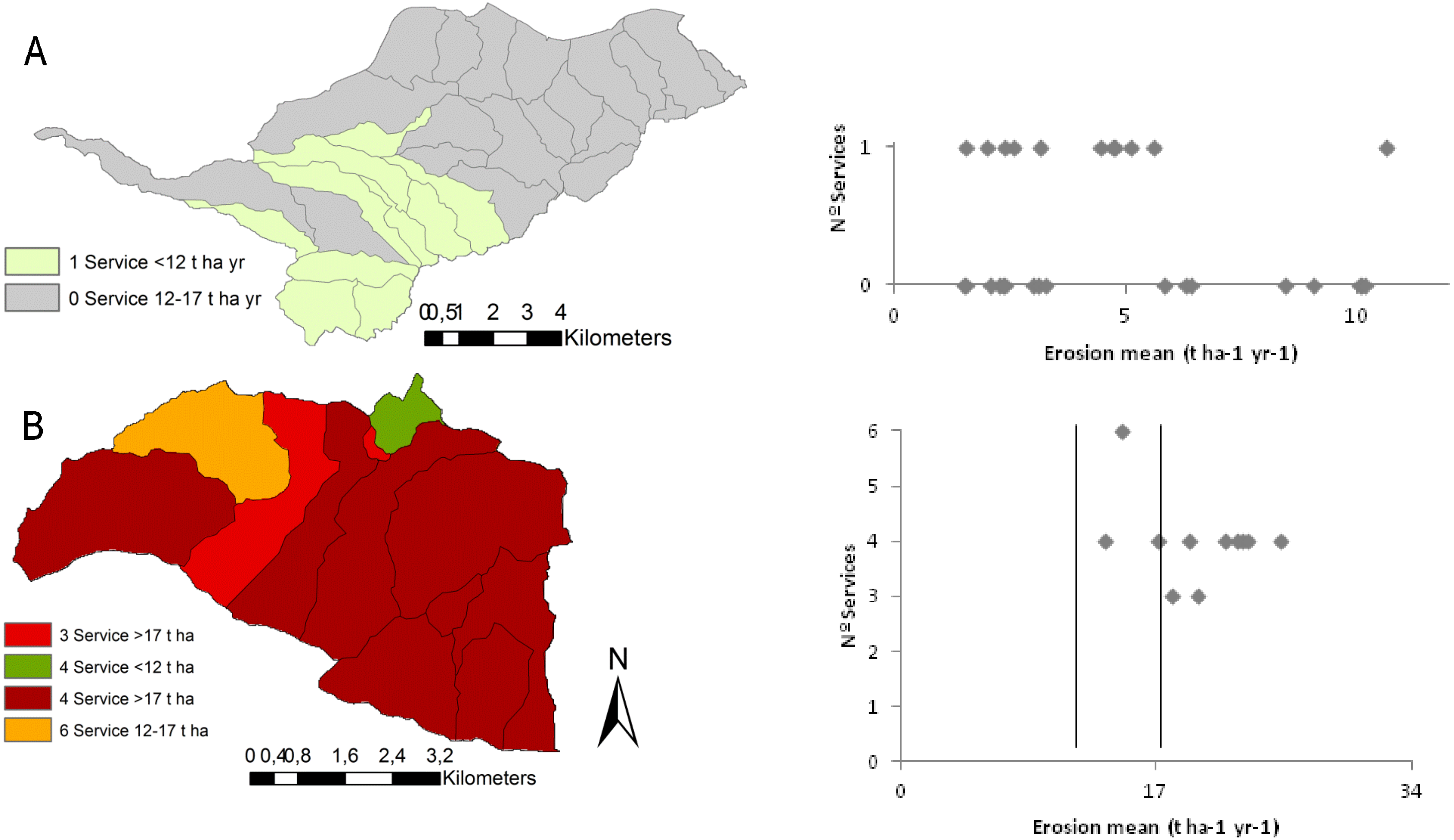


Fig. 30. Hierarchy map for subwatershed 4(A) and 63 (B) at second level. On the right of each map are plotted the erosion mean values against numbers of ecosystem service corresponding at each subwatershed. (t ha is a abbreviation of t ha^{-1}yr^{-1}).

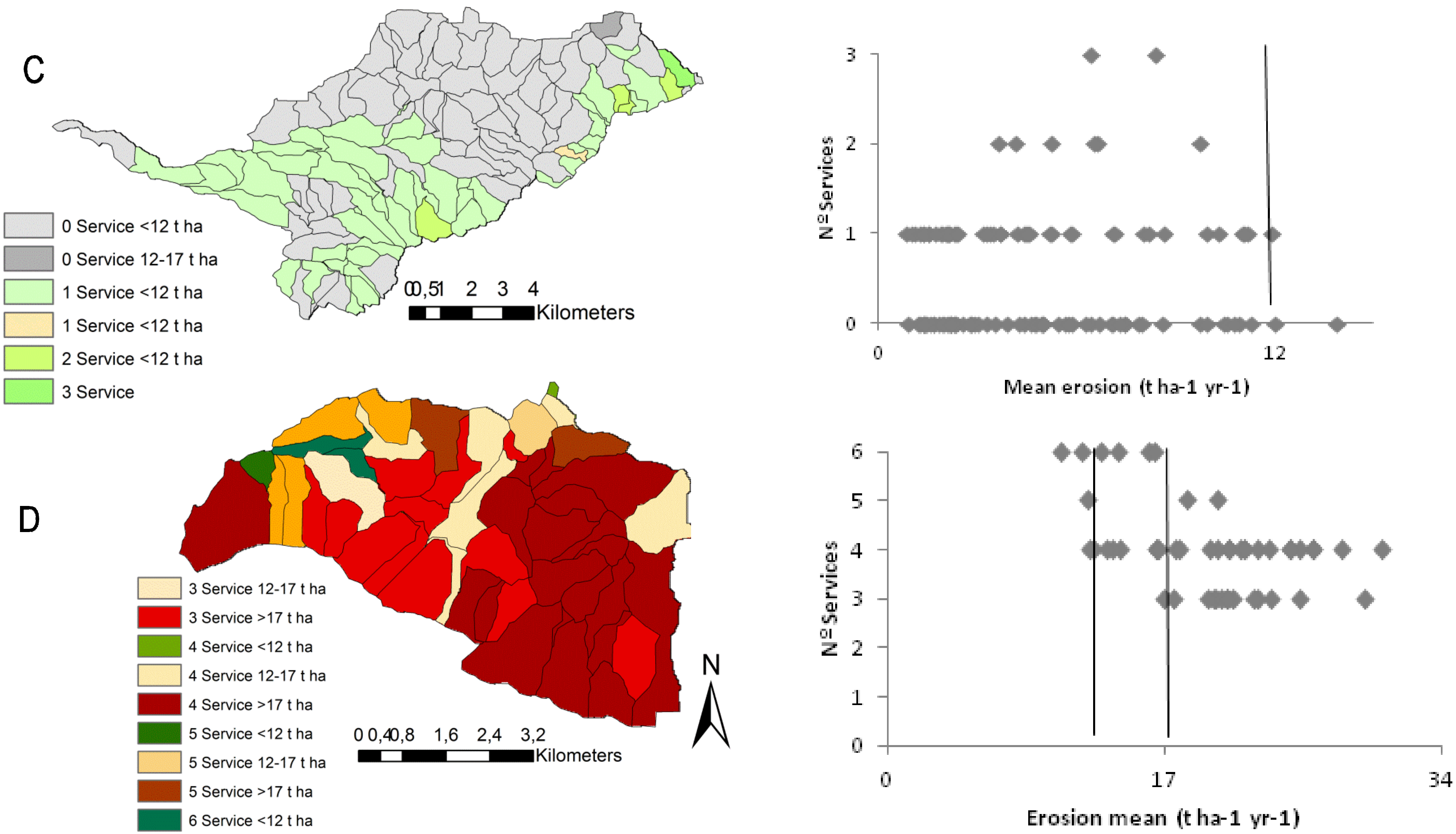


Fig. 31. Hierarchy map for subwatershed 4(C) and 63 (D) at third level. On the right of each map are plotted the erosion mean values against numbers of ecosystem service corresponding at each subwatershed. (t ha is a abbreviation of t ha^{-1}yr^{-1}).

Our two selected subwatersheds show marked differences in the number of services delivered, with 0-1 ecosystem services being observed for subwatershed 4 at second level associated with an erosion rate of <12 and 12-17 t $ha^{-1}yr^{-1}$ (Fig. 30 A) and 3-4-6 services being obtained in subwatershed 63 with an erosion rate > 17 t $ha^{-1}yr^{-1}$ (Fig. 30 B). In subwatershed 63 the priority restoration area is represented by 3 and 4 services and an erosion rate > 17 t ha^{-1} yr^{-1}, corresponding to the greater part of the subwatershed (Fig. 30 B). Moving from level two to level three, diversification increases (Fig. 31 C, D) and for subwatershed 4, the number of services now ranges from 0 to 3, but they are mostly associated with low erosion thresholds (Fig. 31 C). In subwatershed 63, at level three, the number of services per subwatershed ranges from 3 to 6 and most of the subwatersheds appear to present high erosion thresholds (Fig. 31 D).

5. Discussion

5.1. ***RUSLE for targeting restoration efforts***

This study demonstrates that the RUSLE model used with appropriate values for each factor is a powerful tool. Using the GP (Genetic Programming) methodology proposed by Puente et al. (2011) was proven as a reliable approach to generating specifically designed indices to estimate the *C* factor in contrast with traditional indices, such as those of the NDVI and SAVI family (Puente et al. 2011). We identified high-risk areas where soil conservation-restoration practices are needed. In the Martín River Basin, major efforts should be dedicated to retain soil in its southern high relief part and, especially, in the no-restored opencast coal mines to prevent the irreversible degradation of these zones. For this purpose, the results of this study are useful for identifying different zones of erosion risk at the watershed scale and at lower scales (e.g., subwatershed).

The average annual soil loss rate estimated using RUSLE and GIS for the Martín River Basin was 13.8 t $ha^{-1}yr^{-1}$. This estimation exceeds the estimated tolerable limits for soil formation of between 2 and 12 t ha^{-1} yr^{-1} in Mediterranean environments (Rojo 1990). These results compare well with other studies in similar areas (Renschler et al. 1999; Van Rompaey et al. 2003; Capolongo et al. 2008), confirming that the RUSLE-GIS generated estimates of soil loss in this study appear to be reasonable.

The spatial variation of erosion in the Martín Basin appears to be dominated by slope. The higher mean values of potential erosion were associated with zones located in the highlands with steep areas, including opencast coal mines that had the highest erosion rates even though large areas of many coal mine zones have been submitted to a restoration process. Although erosion varies greatly depending on the type of mine restoration, the steepest zones in the opencast mines match the highest erosion rates in the Martín River Basin because of the creation of large (sometimes 1 or more km^2) hillslope areas inside and surrounding the mines by means of excavation. The scale of the mined areas (0.14 – 7.2 km^2) in comparison with the pixel size of the DEM (400 m^2) supports our assumption. Rill and gully networks in these reclaimed systems can markedly limit water availability and modify the spatial distribution of soil moisture at the slope scale by reducing the opportunities for down-slope runoff re-infiltration and by concentrating the water flow along the channeling network (Biemelt et al. 2005; Moreno-de las Heras et al. 2010).

During the photographic field survey to evaluate the connectivity and eroded area prediction along the created buffer zone in the stream and river channels, we observed that some of the areas, appearing in the model analysis as high erosion areas, corresponded to bare rock and rock landslide phenomena (Foto 3); however, in the monitored areas, the model generally recognizes riverside degraded areas, as shown in Fig. 32. We also identified some mining areas that were degraded in the year of creation of the digital elevation model used here and that are now (2012) restored.

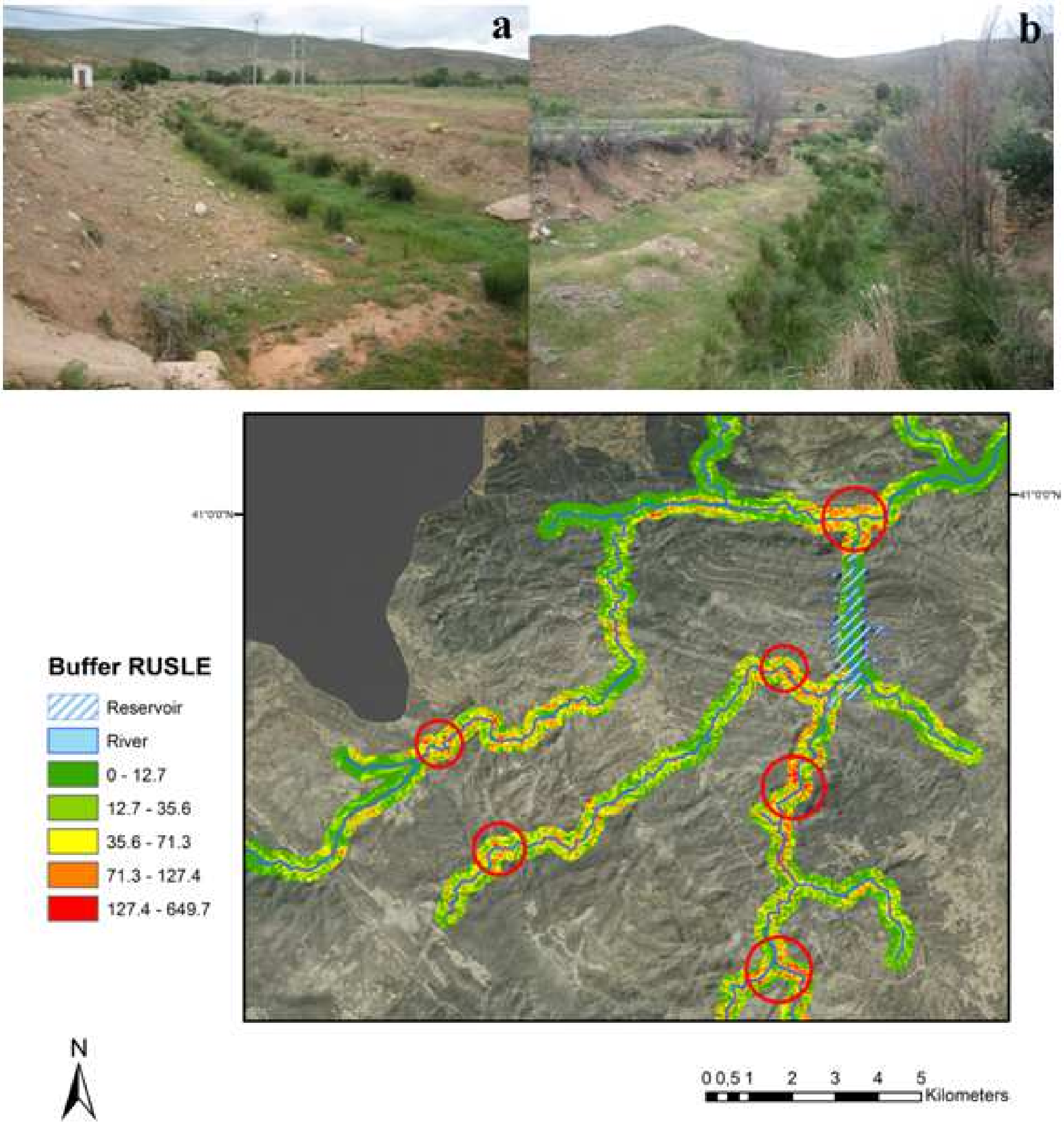


Fig. 32. Example of highly degraded riversides, founded using RUSLE-buffer map. In evidence on the bottom left (a), concrete ditch discharging straight in the river.

Road embankments have not been considered with a special focus in this paper, but during the photographic survey, we realized the magnitude of their impact on the river system

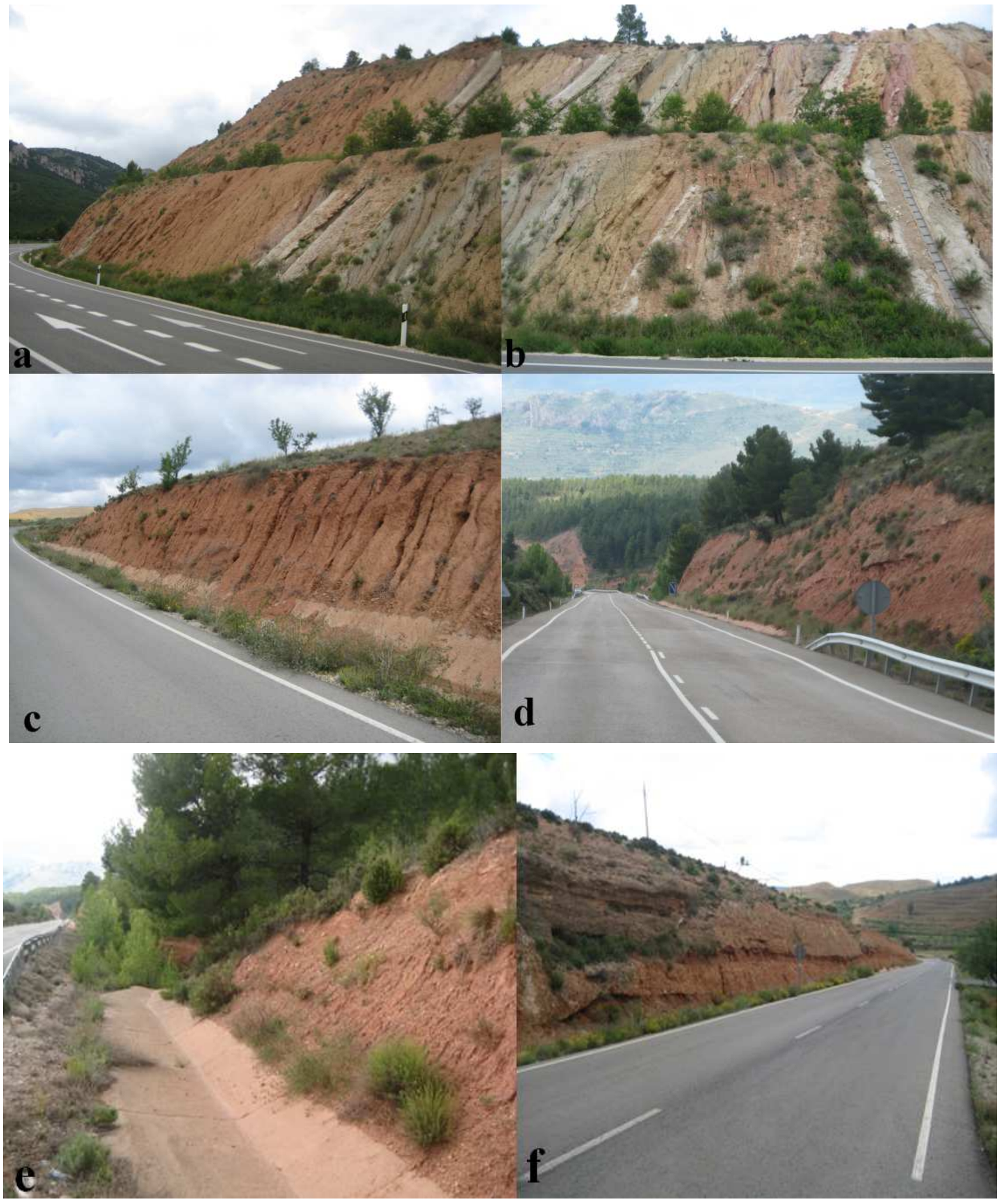


Fig. 33. Road embankments in different parts of the Martín Basin. In the central part (a, b) in the north (c) in the south (d, e, f).

These slopes are often directly connected by channels to the river network (Fig. 32 a on the bottom left of the picture and Fig. 33 e) without having any way to intercept the sediment through depositional areas. García-Fayos et al. (2009) argued that the stabilization of road slopes is doomed to failure if the ecological knowledge of the topographic thresholds that limit vegetation establishment is not taken into account at the time of road building. This argument is supported by the existence of road cuts with a slope gradient exceeding 45°, where intense erosion occurs, generating very high soil loss and impacts that other studies have already highlighted in Mediterranean sites near the study area (Bochet and García-Fayos 2004). We also highlight that some areas are highly degraded but are disconnected from the fluvial channel or are intercepted by depositional areas. These areas are not a direct threat to water bodies because they are not significant contributing areas. Management plans for a watershed should take into account the need to evaluate the importance of these areas with respect to different uses and the potential benefits of restoring these areas, assessing the effective value for the production of ecosystem services and the mechanical and monetary possibility of action (usually steep slopes) to restore it.

5.2. *Plant colonization and reclaimed slopes*

Moreno-de las Heras et al. (2011) suggest that natural plant colonization in Mediterranean-continental reclaimed environments requires vegetation cover of at least 30% and rill erosion rates below 17 t ha^{-1} yr^{-1}. In our case, 59% of the river basin has less than 30% plant cover, and 60% of the watershed has an erosion rate higher than 12 t ha^{-1} yr^{-1} and plant cover lower than 30%. This result is due to the very slow rate of plant recolonization and forest expansion, which occupies approximately 21% of the mountainous southern part of the basin.

Fifty-six percent of the mine areas are included in the acceptable soil loss range for plant colonization, but the erosion rate is higher than 17 t ha^{-1} yr^{-1} in 44% of the mine zones in the Martín Basin. In these latter zones, plant colonization is difficult, enabling the formation of rill networks depending on the degree of disturbance, slope length and available water, among other factors (Moreno de las Heras et al. 2010). The consequence is a high erosion rate that endangers the life span of these newly created habitats and the wetlands created in the pit of the restored mines, which were established by being filled with high loads of mined materials but are filled with eroded sediments. This siltation process also reduces other key ecological processes (e.g., sediment-water column exchanges, organic matter enrichment) and the biological structure of this type of ecosystem (Mitsch and Gosselink 2007; Gell et al. 2009).

In most of the (north) lowland and relatively flat part of the basin, which is dominated by agriculture, the estimated erosion rates are much lower (in general, <10 t ha^{-1} yr^{-1}). The high rates in this part of the basin are associated with river dynamics (bank erosion) and land use (Fig. 5, p. 33), prevalent cereal crops and scrublands. In the southern and central zones of the basin, which are covered by conifers and hardwoods, the estimated values of the *C* factor (the vegetation-related variable in the RUSLE equation; see appendix 9.4.4.) were, as expected, low because of the relatively high cover density. The *C* factor for vineyards and olive trees had typical intermediate values because of the vegetation-free zones between the rows of plants, which are common in this type of land use. However, for scrubland, the *C* values obtained reflect the low vegetation density of this land cover. Grassland was expected to show lower values than those obtained, but these values, again, depend on vegetation density, which is widely spaced. In any case, grasslands occupy only 0.5% of the whole area of Martín Basin.

Grassland-shrubland was found to be more susceptible to soil losses by water erosion than cropland, forests and plantations. A high erosion rate seems unlikely to occur in conifer plantations, but the relatively high rate observed in this land cover in Martin Basin is probably due to these artificial plantations being established with the highly regular spatial distribution of the trees in hillslope areas. These results are similar to those observed in other semi-arid areas labeled as poor soil environments with past human overexploitation (Erdogan et al. 2007). These results are also partially a consequence of the anthropogenic displacement of shrubs and forest from low slopes (Smith et al. 2007). Past agricultural practices in these zones have eliminated natural vegetation from the steep zones, leaving a difficult terrain for agriculture (García-Ruiz 2010). Other studies in Spain showed that reforestation followed by insufficient forest management may negatively affect both soil properties and the ecosystem's response to the erosive action of rainfall (Pardini et al. 2003).

Restoration planning to counteract erosion was approached with general reforestation actions extensively applied to large areas for most of the second half of the twentieth century. Now, more specific and autochthonous species are used for plant reforestation in Mediterranean areas (Pausas et al. 2004). Because slope plays a key role in erosion in the Martín Basin, restoration actions must focus on the mitigation of slope-based erosion impacts, which requires a more comprehensive restoration planning than just revegetating by planting trees.

The most efficient place from which to remove pollutants and nutrients from watershed discharges is the riparian zone (Welsch and Management 1991), before the water flows enter a stream channel. As most steep zones are located in the upper parts of the basin, the most important locations for protecting and restoring riparian buffers are along these headwater streams. Buffers disrupt lateral linkages within catchments, and they may include alluvial pockets of floodplains, fans or piedmont zones that occur at breaks in slopes along valley margins, disconnecting lateral connectivity in catchments (Fryirs et al. 2007). Solutions include low-cost erosion control techniques such as contour hedgerows across the slope in cropped fields or regenerated on the base of steeper inaccessible areas, where restoration actions are impossible or too expensive, to reduce runoff velocity and prevent pollution of the river network. Lasanta et al. (2001) and other studies showed that in Spain, the main process following the abandonment of hillslope cropping is the collapse of the terrace walls by landslides. Many areas identified in the highlands are affected by this problem (Fig. 34).

Fig. 34. Collapse of the terrace walls in Martín Basin.

Where possible, the recovery of decaying cropping terraces in the steep slopes will be a good soil conservation practice (Dunjó et al. 2003). Other techniques are stone terracing, where a stone embankment (Marienfeld 1994) around a hillside intercepts overland flow, enhances infiltration, and safely guides runoff off field. These are some of the major recommended engineering structures for controlling soil erosion.

Stimulating extensive livestock forage in depleted soil using leguminous forage crops (*Medicago sativa L.*) would improve the soil conditions in the valley floors (Prosperi et al. 2006). Because the shortage of nutrients in the Aragón region is the first limiting factor for plant colonization (Ries et al. 2000; Lasanta 2000), an enormous step forward will be the creation of a management plan for the use of organic waste as compost. This action would improve soil structure with organic matter and nutrients, taking advantage of this precious resource that is currently lost in landfills. This action will help plant colonization and consequently soil cover, which, when exceeding 60%, can significantly reduce soil erosion in semi-arid environments (Sauer and Ries 2008). These combined benefits will result in increased and sustained crop yields as well as enhancements to multiple ecosystem services.

5.3. *Ecosystem service mapping, value of the approach*

The presented approach is valuable because areas are selected according to the ecosystem services they deliver for a river basin, which is a useful tool for prioritizing restoration at a watershed scale when the information is evaluated alongside the area's environmental risk of degradation. This is a substantial change of focus for restoration and management planning. Previously, restoration actions were planned mostly at an ecosystem scale used reference ecosystems to define restoration actions (SER 2004) and didn't take into account that regions are made up of mosaics of ecosystems. Additionally, land and natural resource management are usually based on maintaining basic features and, using the combined evaluation of multiple services, provides a tool to plan an objective-based strategy to maximize multiple service provisions according to the mosaic of ecosystems forming an area (Aronson et al. 2006). The evaluation and categorization of different ecosystem services is based on two factors: the consideration of multiple ecosystem services and the approximation of the value of ecosystem service to obtain zones where high and very high services overlap, which increases the value of these selected zones. The inclusion of multiple ecosystem services, particularly those that are strongly related to key ecological processes and ecosystem functions, provides a more complete understanding and a stronger basis for making comparisons between zones (Swift et al., 2004; Carpenter et al. 2009). Targeting restoration prioritization at a basin scale is significant because basins are mosaics of ecosystems, and most restoration plans focus on single ecosystem types (Palmer 2009). It is true that some correlation exists between ecosystem services, but this is also the case for ecological processes regulating ecosystems (de Groot et al. 2010).

A critical issue in mapping ecosystem services is data quality and availability. Mapping involves GIS overlay analysis and geoprocessing to combine input layers from diverse sources to derive the final ecosystem service map. Difficulties encountered with deriving ecosystem service maps relate to the scale, age and accuracy of the input layers (Troy and Wilson 2006). An appropriate level of precision is vital if end maps are going to direct restoration and management. In our case, soil formation and water supply maps had a large cell-size unit (1 km) that should be re-sampled to direct further detailed analysis. Comparability of data is essential to meet the goal of establishing priority areas and objectives for restoration and land-use management.

Our analysis focused on several ecosystem services based on ecological processes and/or characteristics that are the most significant for sustaining the ecological functions of the whole basin. Most of the services are regulation services that enhance ecosystem resilience, and many have a synergy with provisioning and cultural services (Bennett et al. 2009). It's clear that our results need to be supported by data for additional ecosystem services to support and define precise future decisions about management and restoration actions in the study area. However, the methodological approach presented here is the basis for more comprehensive studies that will include a stakeholder perspective to understand which services are important for a sustainable development in the basin (Forsyth et al. 2012).

5.4. ***Trade-offs between services***

High values of some ecosystem services, especially provisioning services, are sometimes inversely related to other services, which challenges the sustainable use of the whole basin (Bennett et al. 2009; Viglizzo et al. 2011). Our results show that most part of Martín Basin is important for the delivery of at least one service within our selected suite of services. Only a few small areas produce very high numbers of services. The high degree of clustering between services points to a synergistic relationship between most of the services selected, and this has also been highlighted by other studies (Naidoo and Ricketts 2006; Nelson et al. 2009). As expected, the areas important for carbon storage, soil accumulation and retention and water flow regulation were clustered with different overlapping percentages. It is well known that trees stabilize soil with their roots, contribute to organic carbon accumulation due to the formation of leaf litter and facilitate water infiltration and storage, which facilitates plant-growth and the storage of carbon dioxide (Durán Zuazo and Rodríguez Pleguezuelo 2008; Winjum and Schroeder 1997). In any case, trade-offs among services are possible. Bellot et al. (1999) highlight that a landscape created by human management can increase plant biomass and the use of water by wild vegetation, agriculture and the human population while also reducing runoff that affects reservoir storage, deep drainage and the aquifer recharge.

However, it is not always valid to say that an area rich in services has a good ecological status. If restoration focuses on just one service, tradeoffs among services can create declines in some ecosystem services (MA 2005; Tallis et al. 2008) and could lead to negative impacts on biodiversity or provisions for other services. Use of suitable indicators for quantifying ecosystem services at a regional scale is challenging because major ecosystem services vary across different ecosystems. Too many indicators may confuse the public and decision makers, while too little will invalidate the results (Su et al. 2012). It is important to select or develop indicators that reflect the potential of the system to sustain the yield of each service (McMichael et al. 2005). When planning and managing restoration, considering a number of ecosystem services with the intent of improving the balance of the selected services is an objective-based strategy that offers a long term benefit to the whole socio-ecological system rather than just to a few structural and functional ecosystem characteristics (Kremen and Ostfeld 2005; Palmer et al. 2009). An example of this is the case presented by Barbier et al. (2008) that demonstrates the negative, long-term socio-ecological impacts after the conversion of mangroves to shrimp farming. Another example is the case of using alien species monocultures for cellulose production (*Eucalyptus*), which causes a reduced water yield from catchments among other service trade-offs (Samraj et al. 1988).

5.5. *Guidelines for watershed management and restoration*

Using this approach, we were able to identify subwatersheds located in the northern part of the lowlands of Martín Basin that only supplied one service of our suite. There were 24 subwatersheds marked in this area, representing 39.5% of the basin area. Most of these (13) did not provide any ecosystem service, and eleven of these subwatersheds provided only one to two services (Fig. 22 B, p. 74).

Conversely, subwatersheds that delivered an increased number of ecosystem services, often with high value, were located in the southern part of the basin in the highlands, which is also the area where major impacts from mining activities originate.

These results suggest that alternative decisions should be made regarding the spatial allocation of restoration actions at the basin scale. Is it better to restore services in the northern part of the basin, which currently provides mostly just one service, and manage this part of the basin to enhance multiple services simultaneously? Or is it better to restore key impacted and degraded areas in the southern part, which are already providing high values of multiple services, because of their importance in assuring the continuous delivery of services?

Placing the major restoration emphasis on the southern region would improve ecological functions as erosion is a major detractor in the provision of ecosystem services, negatively affecting soil retention, water supply, and the biodiversity based services. Adopting this strategy would increase the delivery of ecosystem services throughout the entire basin because the lowlands depend on ecological processes taking place in the highlands. For example, some surface water supplied in the highlands may become available in the lowlands due to run-off or human-managed systems acting as reservoirs and canals. The six services that we have focused on have high values in the highland area of the basin, and their proper maintenance will stimulate synergy among services ameliorating the flow of services throughout the basin. In addition, the northern lowlands are dominated by agricultural production. Prioritizing restoration in the lowland region of the basin would compromise the benefits obtained from extensive agricultural farming and would likely affect the positive social atmosphere required for producing an efficient restoration project.

The marked spatial heterogeneity of this basin largely governs the distribution of ecosystem services. Our findings clearly note the need for an integrated approach for land-use management and restoration prioritization. This is particularly relevant in watersheds with large agricultural areas (Zhang et al. 2007) and/or where intensive extractive activities, such as mining, are of key economic importance for the population of the region. Integrative strategies should focus on enhancing ecosystem service delivery through restoration of *hotspots* or subwatersheds that offer high numbers of ecosystem services while simultaneously promoting sustainable land-use practices in areas where ecosystem services are limited. Table 6. Combined ecosystem services delivery and environmental risk criteria (<12 (low), 12-17(Medium) , >17 (High) t ha-1 yr-1) for establishing priority areas for restoration providing a framework for decision-making with regards to the prioritization of areas within a watershed based on the approach presented here: the combination of improving ecosystem service delivery and reducing environmental risks of degradation.

In the Martín Basin, restoration efforts in the southern region could focus on the protection, stabilization and enhancement of existing synergies between services in areas where service values are relatively low. Restoration action should focus on increasing soil retention by reestablishing forest ecosystems, thereby stimulating ground water recharge, soil accumulation, carbon sequestration and climate regulation, which will positively influence ecosystem services in other parts of the basin. Bennett et al. (2009) showed that when investments are made in securing regulating services, provisioning and cultural services also increase, resulting in an increased resilience of the local ecosystems. These restoration actions should be followed up with the development of forest management plans to increase carbon forestry and protect important headwater areas. In these areas, vegetation management will be essential for improving the cover to prevent irreversible degradation.

In the northern lowland area of the Martín Basin, a best management practice approach would ensure long-term provisioning of agriculturally derived benefits. The adoption of good agricultural practices, including conservation tillage and adaptation to threats of climate change, should be encouraged. Additional management practices could include the use of manure and biomass residues (e.g., straw mulching), which will help to improve soil organic carbon levels (Jones et al. 2005), thereby reducing soil and water losses (Su et al. 2007). The implementation of multi-crop rotation strategies would also increase the level of soil organic carbon (West and Post 2002) and improve soil structure, making soils more resilient (Lal, 1997). The establishment of leguminous forage crops on low productive areas would improve livestock production (Delgado 2000). This would require the use of native and adapted species to avoid potential negative impacts on the ecosystems.

Special attention must be given to the mining areas because they are the major source of sediment in the basin (Trabucchi et al. 2012a). These mines have been restored using a variety of restoration techniques and strategies at different times (Moreno de- las Heras et al. 2008). The opportunity to create new services in restored areas exists and has been demonstrated on several restored mines in the Martín Basin that have been planted with crops and fruit trees. However, in order for these areas to be sustainable, best agricultural practices need to be adopted due to the high susceptibility of their soils to erosion and the very low soil organic carbon content. Furthermore, wetlands created in the old mine pits can provide multiple functions at a smaller scale, including recreation and education, and contribute multiple services at a larger/watershed scale, which could be accomplished in this semi-arid area through re-establishing a network of sites for biodiversity development (Moreno-Mateos et al. 2009).

Mapping multiple ecosystem services provides a useful framework for management and restoration planning at the watershed scale. Detailed spatial prioritization of restoration actions will require analysis of ecosystem services and tradeoffs at a finer spatial resolution. Watersheds or basins have fractal characteristics, so fractal methods of analysis can be effective in predicting ecosystem service patterns at multiple scales (Halley et al. 2004).

5.6. *Restoration implications from multi-scale analyses*

Landscapes are complex systems that require multi-scale analyses if they are to be appropriately managed and if the outcomes of interventions are to be anticipated (Hay et al. 2001). Basin-scale analyses (such as that performed in our case study area, the Martín Basin) appears to represent an appropriate extent scale for evaluating our methodology as the basin is considered the optimal functional ecological unit of management or, at least, that where more intensive interactions occur between human use of the resources and ecological processes (Golley 1994), both of which determine ecosystem services. Exploring a variety of spatial scales is a necessary exercise for understanding resource distribution (Lewis et al. 1996; White and Walker 1997). In our case, different spatial scales (levels of analysis) were used to investigate the spatial locations of possible restoration actions and the dynamics of ecosystem services associated with erosion. The type of multiscale spatial analysis performed in the Martín Basin to assess ecosystem services, which has frequently been suggested (Kremen and Ostfeld 2005; Hein et al. 2006; de Groot et al. 2010), proved useful for identifying sites to be targeted for restoration (to ameliorate erosion) that simultaneously increase the provision of a selected bundle of ecosystem services.

An initial assessment of the Martín Basin at low spatial detail is able to provide a general understanding of this territory and to identify the general status of broad areas in the basin, which is useful for a preliminary general statement of types of restoration action according to differences in the environmental risk and ecosystem service provision in these areas (Trabucchi et al. 2012a). Chu et al. (2003) described this need for obtaining a broad-scale understanding related to system dynamics so that it will be possible to explain cause-effect relationships in detail. The introduction of additional hierarchies or levels facilitates the integration of more detailed information. Our third level of analysis was found to be key in determining watershed processes and the mechanism of ecosystem degradation (Nakamura et al. 2005). As expected, reducing pixel aggregation increased spatial differentiation and detail and facilitated the location of areas for the prioritization of restoration and management actions. The second and especially, the third level of analysis followed a bottom-up approach. This approach increased the accuracy of the identification of site-scale areas to be targeted for action and provides a defensible basis for hypothesis testing in field experiments. We explored the third level (highest resolution) in detail, as this scale is expected to be the most economically suitable for directing restoration actions. In our case study, this level of analysis corresponded closely to the scale of opencast mine areas, which present a mean average area of 1.5 Km^2.

The fine-scale analysis highlighted subwatersheds or geographical areas in the basin where restoration actions to control erosion should be prioritized hierarchically to maintain or increase the provision of ecosystem services. This would not have been possible if we had only undertaken a single broad level (first level) of analysis.

5.7. *Developed approach for including priority restoration areas*

As a first step in restoration planning, a regional analysis aims at constructing an overview of ecosystem conditions to identify altered areas in need of management action (Nakamura et al. 2005). To manage a river basin efficiently, objectives must be established and restoration priorities identified (Kondolf and Micheli 1995). This understanding is essential to achieve the optimal and efficient allocation of limited resources (Palik et al. 2000; Suding 2011), especially at a broad scale, where costs can grow exponentially. In the Martín Basin, areas presenting few services and low erosion rates were found to be predominant in the flat northern areas, which have historically delivered provisioning services related to food production. In this homogeneous landscape with an oligotrophic environment (low precipitation, low soil organic matter content), restoration actions would be disproportionately expensive compared with the benefits that would be derived from such actions.

We have adapted a simple risk decision support matrix previously used in watershed risk analysis (Milne and Lewis 2011) to facilitate the selection of priority areas for restoration (Table 6 p. 86).

A hierarchical mapping approach could be used for a variety of purposes, particularly in exercises related to site location (Palik et al. 2000; Palik et al. 2003). Area selection can be further refined by coupling the generation of hierarchy maps for prioritizing subwatersheds with desired biological or physical ecological indicators (e.g., water quality, land use, erosion) (Niemi and McDonald 2004), combinations of which can be chosen to infer cause and effect relationships (such as explanatory environmental variables and responses manifested as changes in ecosystem services) (Nakamura et al. 2005). Furthermore, alternative state models, emphasizing internally reinforced states and recovery thresholds, can help in guiding restoration efforts (Suding et al. 2004). These thresholds could include types of pollution (e.g., nutrients, suspended soil, gas emissions) and general environmental disturbance thresholds (e.g., fires, floods) (Groffman et al. 2006).

Ecological problems often require the extrapolation of fine-scale measurements for the analysis of broad-scale phenomena (Turner et al. 1994). The generation of hierarchical maps that allow the evaluation of restoration activity across a hierarchy of scales, ranging from a broad region to an individual site (Ziemer 1999), appears to be a logical and efficient way of locating key potential restoration areas. It is well recognized that restoration and landscape ecology exhibit an unexplored mutualistic relationship (Bell et al. 1997; Li et al. 2003). Our proposed framework integrates multi-scale studies, representing a key interest in landscape ecology (Turner et al. 1994; Hay et al. 2001; Brandt 2003; Burnett and Blaschke 2003; Wu 2004), with the type of hierarchical prioritization used in restoration ecology (Lee and Grant 1995; Palik et al. 2000; Cipollini et al. 2005; Nakamura et al. 2005; Comín et al. 2009) and the growing field of ecosystem service research (Fisher et al. 2009; Reyers et al. 2009; de Groot et al. 2010; Su et al. 2012). Such a multidisciplinary approach has been recommended to make restoration plans more attractive (Benayas et al. 2009; Bullock et al. 2011; Trabucchi, et al. 2012b) and to enhance research and the application of the three disciplines. Here, the focus of ecological restoration shifts from the site-scale studies adopted in the past aimed at the reestablishment of historical abiotic conditions to promote the natural return of the vegetation (Dobson et al. 1997; Bell 1998; Prach et al. 2001) or the reestablishment and improvement of animal habitat (Huxel and Hastings 1999; Bond and Lake 2003) to broad analyses of environmental conditions at regional scales. This vision is supported by modern restoration practices, which acknowledge the importance of ecosystem patterns and processes occurring at landscape scales (Nakamura et al. 2005). During the nested analysis, various spatial and field assessment data can be added as layers to complement and enrich the analyses and improve the precision of prioritization according to the proposed objectives making our methodology extremely adaptable at each single case of research purpose.

5.8. *Investigation of possible trade-offs in restoration prioritization*

Ecosystem service trade-offs are defined as situations in which one service is increased or improved at the expense of another (Bennett et al. 2009) and can arise from the differing interests of social agents (Martín-López et al. 2012). Analyzing the spatial patterns of ecosystem service bundles allows us to understand how services are distributed across a landscape, how the distributions of different services compare and where trade-offs and synergies among ecosystem services might occur (Raudsepp-Hearne et al. 2010). The presented approach highlights where potential ecosystem service improvement can be achieved through restoration and consequently, which trade-offs can be established between the services evaluated here (carbon storage, soil formation and retention, water flow regulation, surface water provisioning, eco-tourism), which contribute positively to natural resource enhancement and those that contribute negatively to natural resource conservation, which are typically provisioning services based on human extractive activities as intensive agriculture and mining. Conventional agricultural practices degrade the soil structure and soil microbial communities due to mechanical activities such as plowing, but management practices can also protect the soil and reduce erosion and runoff (Lupwayi et al. 1998; Holland 2004). The Martín Basin, especially its northern region, displays clear evidence of trade-offs between regulatory and provisioning services, which is an issue that has been noted in many other regions of the world (Rodríguez et al. 2006; Power 2010). Management decisions often focus on the immediate provisioning of a commodity or service at the expense of this service or another ecosystem service at a distant location or in the future (Power 2010). However, win-win scenarios are possible when appropriate land-use practices, such as conservation tillage, crop diversification and legume intensification, are applied (López et al. 1998; Prosperi et al. 2006; Trabucchi et al. 2012a). The potential success of integrating these approaches depends on the maintenance of ecological integrity and cohesion (Gómez-Sal and González-García 2007). Therefore, it may be possible to manage agro-ecosystems to support a diversity of ecosystem services while still maintaining or even enhancing certain provisioning services (Power 2010; Nainggolan et al. 2011). Understanding the benefits and costs of different types of management practices is necessary to allow the establishment and maintenance of sustainable agro-ecosystems (Dale and Polasky 2007).

Due to the predominant natural land cover in the southern part of the Martín Basin, the trade-offs among ecosystem services in this region are of another type and are more difficult to identify because they also exhibit many synergies and dependent ecological processes (section 5.4. p.84). For example, most of the ecosystem services produced in perennial vegetation areas, such as under forest cover, are related to water (e.g., purification, regulation) and these, in turn, are linked to soil (e.g., accumulation, retention) (Klijn et al. 1996; Milne and Lewis 2011; Powlson et al. 2011). While there are clear synergies, there are also potential trade-offs. For example, increasing carbon storage through the planting of fast-growing trees for CO_2 accumulation (a carbon storage service linked to climate regulation) or cellulose production (a provisioning service) may reduce the surface water supply and could also result in the salinization and/or acidification of soils, with consequent decreases in ecosystem services associated with grasslands and reduced resilience of such systems (Bot and Benites 2005; Cespedes-Payret et al. 2009).

Identifying trade-offs is an important step that allows policy makers to understand the long-term effects of preferring one ecosystem service over another and the consequences of focusing only on the present provision of a service, rather than the future (Rodríguez et al. 2006).

5.9. *Possible methodological limitations and future research needs*

5.9.1. Data management

Spatial analysis typically involves GIS overlay analysis and geoprocessing to combine diverse sources of input layers in deriving a desired map. This analysis is often complicated by differences in parent scales, years of creation, accuracy levels, modeled data and minimum mapping units for each input layer (Troy and Wilson 2006). There is no single “correct” or “optimal” scale for characterizing spatial heterogeneity, but comparisons between landscapes using pattern indices must be based on the same spatial resolution and extent. Indeed, a comprehensive empirical database containing pattern metric “scalograms” and other forms of multiple-scale information on diverse landscapes is crucial for achieving a general understanding of landscape patterns and developing spatial scaling rules (Wu 2004). The relationship between ecosystem service delivery and the regulation of environmental factors, such as erosion, may also change according to the spatial scale of analysis (Jackway and Deriche 1996). An analyst's job will often include assembling many layers with different resolutions to obtain a final map that is suitable for management purposes. Ecosystem services, such as the ecological functions and processes from which they are derived, may change in relation to the spatial pattern of observation (Hein et al. 2006; Hurteau et al. 2009), posing a major challenge for mapping these services. It is difficult to define the most appropriate scale of a study, as the resolution at which the phenomena of interest operate and are operated upon may not be immediately apparent (Rutchey and Godin 2009). Thus, in most cases, the best practice may be to adopt the highest resolution affordable (Haines-Young and Chopping 1996) but there must be a threshold for increasing the resolution (decreasing the grain size) of the analysis which once surpassed provides not so useful information as it is not related to functional aspects of the ecosystem (basin in our approach) or could result on excess of resources used in the analysis versus value of the information obtained. Furthermore, high-quality databases and new sampling approaches that support research at broader spatial and temporal scales are critical for enhancing ecological understanding and supporting further development of restoration ecology as a scientific discipline (Michener 1997).

5.9.1. **Statistical analysis**

Selecting appropriate statistical procedures and asking the right questions is vital for meeting targets (Marcot 1998). This study employed one of several available methods for aggregating spatial data to analyze ecosystem service bundles. We used the majority rule method because of our interest in identifying the major number of services present at each spatial level (Trabucchi, submitted). Although this is probably the most commonly used rule in ecological and remote sensing applications (Wu 2004), it would be interesting to compare how different aggregation methods affect the characteristics of ecosystem service bundles. The use of rules, such as maximum, minimum and average rules and others available in GIS zonal statistical tools can have a marked effect on the obtained results (Smith et al. 2007).

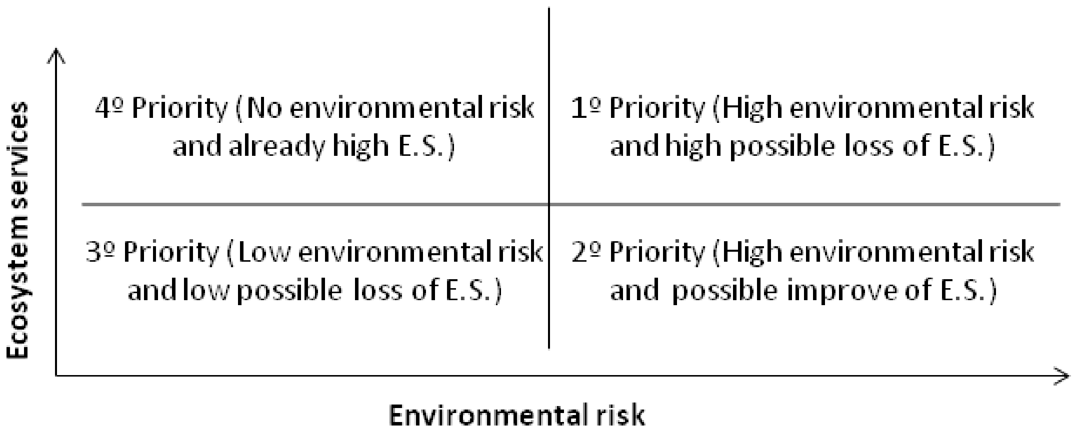
5.9.2. **Validation of the framework**

In the Martín Basin, some subwatersheds at the third level, classified as being of high priority for restoration (presenting erosion of > 17 t ha^{-1} yr^{-1} and >3 ecosystem services), coincide with closed mines. This finding confirmed both the appropriateness of the size of the subwatersheds generated at this level as well as the erosion and ecosystem service categorization applied for prioritizing restoration. However, future studies are needed to investigate the application of hierarchical maps at a mine scale, where these data are available, to further validate the approach presented here.

5.10. ***Flexibility of the framework***

Identifying goals for restoration and prioritizing restoration efforts are subjective processes to some extent (Palik et al. 2000) and the framework showed here can easily be modified to achieve different restoration targets.

Table 7 Schematic approach based on ecosystem services (E.S.) and environmental risk for prioritize restoration action.



A more generalized scheme to prioritize restoration actions at watershed scale is proposed with references to the relative value of the disturbance factor and the ecosystem services (Table 7). In general, restoration priority is given to zones with relatively high risk of degradation (high values of the disturbance factor) as decreasing this risk should contribute to maintain the existing high values of ecosystem services and/or to increase low values. And lower priority is given to zones with low environmental disturbance as maintenance or improvement of ecosystem services is not at risk in these zones. Here zones with high environmental risk and high number of services (or value) are prioritized for restoration due to the risk of degradation and loss of highly valued services. Second priority is given to zones with high environmental risk and low ecosystem services as it is expected that ecosystem service provision will increase after ameliorating the disturbance factor. Third priority for restoration is assigned to zones in a watershed with low environmental risk and low ecosystem services, as it is expected that ecosystem services provision will increase after decreasing the disturbance. And final position in the range of restoration priority is assigned to zones with low environmental risk and high ecosystem services as no risk exists for ecosystem services provision.

This generalization can be re-ordered in similar way to those plankton mandalas by Margalef (1983; 1997) to express the potential relationships among parts of an ecosystem under different environmental conditions which work both in the same direction to favor restoration (synergy: erosion × ecosystem services) or which both set against (erosion/ecosystem services) some ecosystem characteristics (Fig. 35.). This type of mandala can be interpreted for re-thinking the restoration priority criteria for different zones of a watershed, but also as a framework for a potential synergistic effect among restored (with the above criteria) zones in a watershed. Restoration to decrease environmental risks of zones with high ecosystem services will ensure the provision of ecosystem services with effects in other zones of the watershed with lower ecosystem services and high environmental risk, which in turn may favor increasing ecosystem services provision in zones with low environmental risk. Further development of the approach presented will be necessary to show if this dynamic aspect of influences between zones of a watershed restored with the priority criteria presented here takes place. Practical cases of restoration at watershed scale performed with this approach will provide advances in this way.

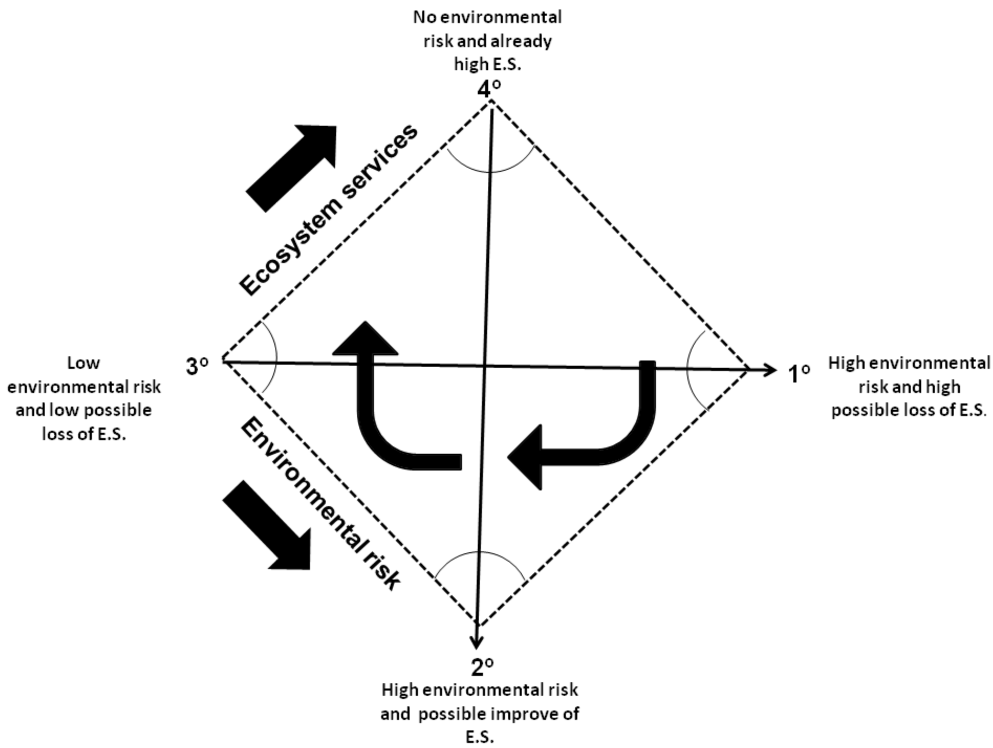


Fig. 35 Mandala for criteria priorization for different zones of a watershed

6. Conclusions

- The publication of the Milenium Ecosytem Assessment promoted using ecosystem services into restoration studies after 2006. However different approaches have been followed until now reflecting the diversity of types of restoration plans and that there is not a general framework to include ecosystem services in restoration plans (with *ad hoc* methods and general definitions) which could facilitate defining and evaluating ecosystem services. Mapping in a complementary way the evaluation of ecosystem services and factors of environmental disturbance is a solution for the hierarchic prioritization of restoration actions in a territory.

- The RUSLE model (Revised Universal Soil Loss Equation) has been used to estimate the erosion risk in the watershed of Martn River (Aragón, NE Spain) using an innovative vegetation index to evaluate the key C factor (cover factor), obtaining better results than other vegetation index available. One of the RUSLE model limitations is that it does not include deposition and remobilization processes which take place down slope resulting in laminar erosion. Results from RUSLE-GIS applied to Martín Basin resulted in average erosion rates similar to those obtained for other zones with similar environmental characteristics.

- The average soil erosion in Martín Bain is 13.8 t ha^{-1} $year^{-1}$. The south part of the basin, highlands, is that with high erosion because of marked orography (24 t ha^{-1} $year^{-1}$) while in the north part, lowlands where mostly dry agricultural use is established, relatively low erosion was estimated (10 t ha^{-1} $year^{-1}$). The erosion map generated with this model let analyze the major zones in the watershed as sediment sources. Pasture areas are those with the highest erosion rates (25 t ha^{-1} $year^{-1}$), although this type of habitat only covers 1% of the total watershed area; schrublands, which also include some bare soil zones mixed with schrublands and mine zones, also have high erosion rates (23, 24 t ha^{-1} $year^{-1}$, respectively for schrublands with some bare soil zones and firsts generation restored mine zone).

- The main advantage of this method for estimating erosion rates at landscape scale is that it is easy for implementation after some information which is relatively easy to obtain nowadays. This methodology has been shown the utility of remote sensing techniques for basic and applied studies, both at basin and regional scales (10-100,000 km^2). Based on our experimental studies, we think that RUSLE-GIS, can improve the estimation of soil erosion rates and, consequently, can be a useful tool for land use management, conservation and restoration at basin scale.

- A bundle of ecosystem services soil retention and accumulation, water supply and regulation and carbon storage, (selected based on their relevance for the ecological functioning of the watershed) and ecotourism as a cultural service, were evaluated using surrogates, (organic carbon in the topsoil, carbon dioxide stored in forest vegetation, runoff, aquifers recharge), and mapped in Martín Basin. Water runoff regulation, surface water supply and soil formation are those present in large areas of the watershed (80%, 67% y 62%, respectively), while ecotourism, soil retention and carbon storage are provided in smaller areas of the watershed (36%, 27% and 21%, respectively). *Hotspot* areas (zones with provision of an ecosystem service at high value) are located in the south part of the basin (highlands). Water regulation has the largest *hotspot* area covering 42.4% of the watershed and climate regulation the smallest one (2.4%).

- The spatial distribution of the ecosystem services related to water follows the spatial rainfall pattern. Surface water supply is present in the entire watershed and coincides with low values of soil accumulation in the low part of the basin. Climate regulation and erosion control depend on the plant cover; both are distributed increasing with altitude. Soil accumulation dominates in the south part decreasing towards the north, lowlands. Ecotourism is located mostly in the south and central parts of the watershed, and close to towns where trekking routes start.

- The highest service overlapping, 3-5 ecosystem services, is observed in mountain zones of the south and central parts of the basin, in accordance with relatively high plant cover, mostly forest and shrubland. One or two services are provided in 25% of the watershed and 3 services in 21%. Four and five services represent the 10% and 2.6% respectively of the basin. Six services together are delivered only in a small area, 0.67%, near Montalbán. The area providing not any of these ecosystem services is 14% located in the north part of the basin. Ecosystem service overlapping is high in general. Those services related with water show the highest overlap, 65% of the watershed and 6.75% of the watershed shows overlap of some *hotspot* services.

- As it has been observed by other author's mountainous areas provides greatly ecosystem services. Mountain areas of Martín Basin, with high relief and relatively high rainfall provide a higher number of services with higher value than lowlands highly influenced by human activities and used for agricultural production in semiarid landscapes. The same characteristics favoring the provision of ecosystem services may contribute to their decrease if land and natural resources management and use are not adequate or if important environmental risks, as erosion, exist in these zones. This is the case for a few subwatersheds of Martín Basin with sparse plant cover and others with open coal mines where erosion rates higher than 17 t ha^{-1} $year^{-1}$ and only 1-2 services are delivered.

- In this work an approach for establishing a spatial hierarchic classification of zones for restoration has been proposed bases on the analysis of the spatial distribution of erosion, as the factor of environmental risk, and the evaluation of ecosystem services, as the state variables, using 20 m × 20 m basic data. This methodological approach was followed after analysis at three spatial scales (level, of analysis defined as different number of pixels containing data aggregated as the basic data of spatial analysis). This let identify the most adequate spatial level analysis for selecting priority areas for restoration in Martín Basin. In order to establish criteria for prioritize sites for restoration, three erosion thresholds related to ecological thresholds for the establishment of vegetation in the study area were used combined with the provision of ecosystem services. Mapping the results, it is observed how the spatial diversification and precision of areas proposed with different priority category for restoration increase as the spatial aggregation of analysis decreases (definition increases). Mine zones, with areas of about 1.5 km^2 and erosion rates over 17 t ha^{-1} $year^{-1}$and high provision of ecosystem services, are distinguished between those small subwatershed selected as high priority areas for restoration at the fine grain size analysis. The first pixel aggregation level of analysis in Martín Basin is useful to distinguish large areas of the basin and potential general strategies for their restoration or management.

- This approach, combining the evaluation of the factor regulating the environmental disturbance factor, erosion, and the evaluation of ecosystem services, as state variables, and its graphical representation with GIS, constitutes a logic and practical approach for establishing a hierarchy of sites for restoration. Basic data availability with good resolution and the analysis of interest by stake holders may be further requirements to be incorporated for further development of the approach.

- A conceptual framework is derived from this work with easy application for the same purpose to other territories with environmental disturbance for ecosystem service provision (but for provisioning services). At the watershed scale, it is recommended to establish a hierarchy of area for restoration as follows: first priority to those areas with high environmental risk and high provision of ecosystem services (in order to decrease the environmental risk of losing high ecosystem services provision); second priority to those areas with high environmental risk and low provision of ecosystem services (where some ecosystem service gain can be obtained after restoration, decreasing the environmental disturbance); third priority for restoration in those areas with low environmental risk and ecosystem services provision (expecting to gain ecosystem services after performing some improvement of the environmental conditions); and not acting in areas with low environmental risk and high ecosystem services provision (as there is no risk of losing the provision of ecosystem services).

- As a general conclusion, this work has shown that the assessment of ecosystem services is a useful tool to plan the ecological restoration and land management of a territory made of a mosaic of ecosystems.

7. Conclusiones

- La inclusión de los servicios de los ecosistemas en los estudios de restauración a escala de cuenca se ha incrementado desde el año 2006 bajo el impulso del Millenium Ecosystem Assessment, pero los enfoques adoptados para este fin hasta ahora son diversos. Esto se debe a la herencia de la utilización de enfoques *ad hoc* en los planes de restauración del pasado (ej. restauración de hábitat de una única especie) y a la inexistencia en la actualidad de un marco general a seguir para la inclusión de los servicios de los ecosistemas en los planes de restauración (basado en una metodología y definiciones generales) que podrían hacer la localización y evaluación y localización de los servicios más directa y sencilla. La complementación cartográfica de los servicios con factores que amenazan la continua provisión de los mismos parece una solución para priorizar jerárquicamente las necesarias acciones de restauración.

- Se ha usado el modelo RUSLE (Ecuación de pérdida de suelo Revisada), un modelo de tipo empírico para evaluar el riesgo de erosión en la cuenca del Río Martín utilizando un innovador índice de vegetación para la evaluación de un factor clave (factor *C* cobertura) del modelo elegido que ha demostrado obtener mejor resultados que los disponibles en la actualidad. La limitación de este modelo es que no incluye fenómenos de deposición y retransporte que ocurren en un perfil ladera-abajo ya que resulta en un valor de erosión laminar. La implementación de RUSLE SIG (Sistema de Información Geográfica) ha permitido la estimación de la tasa media de erosión, obteniendo valores muy similares a los deducidos en otras áreas con características similares.

- El aporte medio anual de sedimentos en la cuenca del Río Martín fue de 13.8 t ha^{-1} $año^{-1}$. La parte sur de la cuenca resulta ser la más afectada por erosión influenciada por la orografía acentuada de la zona (24 t ha^{-1} $año^{-1}$). La parte baja, norte, de la cuenca (principalmente campos agrícolas), es donde se registran las menores tasas de erosión (10 t ha^{-1} $año^{-1}$).

- El mapa de erosión generado por el modelo permitió analizar las principales áreas fuentes de sedimento. El pastizal resulta ser el área con más altas tasas medias de erosión (25 t ha^{-1} $año^{-1}$) aunque solo representa el 1% del área de la cuenca, los matorrales; las zonas de suelo desnudo como las minas y algunas zonas de matorral-pastizal, fueron las responsables de las más altas tasas de erosión generadas en la cuenca (23, 24 t ha^{-1} $año^{-1}$ respectivamente), todas ellas en su parte sur.

- La principal ventaja de esta metodología es la sencillez de su implementación a partir de fuentes de información relativamente fáciles de adquirir hoy en día.

- La metodología desarrollada en este estudio ha demostrado la utilidad de las técnicas de teledetección para realizar estudios básicos y aplicados, tanto a escala de cuenca como a escala regional (10-10,000 km^2). Sobre la base de nuestros resultados experimentales, creemos que RUSLE-SIG, no obstante sus limitaciones y el uso del índice de vegetación como el GPVI, podrían mejorar la predicción de las tasas de erosión del suelo y la consecuente planificación, gestión, conservación y restauración del suelo a escala de cuenca.

- Mediante la evaluación y el cartografiado de un conjunto de servicios ecosistémicos en la cuenca del Río Martín se ha observado que la regulación hídrica, la producción de agua dulce superficial (escorrentía) y la fertilidad del suelo son los servicios más extendidos con respectivamente el 80%, 67% y 62% del área de la cuenca, mientras que los servicios de ecoturismo (36%), control de la erosión (27%) y regulación climática (captura y almacenamiento de carbono en pies mayores) (21.1%) se proveen en extensiones menores de la cuenca. Las áreas *hotspot* están concentradas en las partes sur, más alta, de la cuenca. La regulación hídrica tiene el área más vasta de *hotspot* cubriendo el 42.4% de la cuenca y la regulación climática la más reducida con un 2.4%, ambas en el sur de la cuenca.

- Los servicios relacionados con el agua siguen el patrón espacial de la lluvia. La producción de agua dulce superficial está presente en toda la cuenca y coincide con bajos valores de acumulación de suelo en el sur de la cuenca. La regulación climática y el control de la erosión dependen de la densidad de la vegetación y están distribuidos mayoritariamente siguiendo el patrón espacial de aumento de altitud. La acumulación de suelo predomina en la parte sur reduciéndose progresivamente hacia la parte norte, baja, de la cuenca. El servicio de ecoturismo se distribuye en distintas áreas del sur y centro-norte de la cuenca y, en muchos casos, se centran cerca de los pueblos de Albalate del Arzobispo, Montalbán, y Utrillas, ya que las rutas senderistas empiezan en estos núcleos urbanos o sus cercanías.

- El más alto valor de solapamiento, de 3 a 5 servicios se ha observado en áreas montañosas del sur y del centro de la cuenca correspondiendo con una alta densidad de cobertura vegetal forestal y de matorral. Uno y dos servicios son provisionados en el 25% y 25.8% de la cuenca respectivamente y tres servicios en el 21%. Cuatro y cinco servicios representan el 10% y 2,6%, respectivamente, de la cuenca. Seis servicios juntos se encuentran solamente en un área pequeña, 0,67%, en el entorno natural alrededor del pueblo de Montalbán, mientras un 14% de la cuenca no proporciona ninguno de los servicios seleccionados en ese estudio. El solapamiento espacial entre servicios es grande en general. Los servicios relacionados con el agua son los que tienen el más alto porcentaje de solapamiento con el 65% de la cuenca y 6.75% de los correspondientes *hotspot* en la parte sur de la cuenca, lo cual indica sinergia en la presencia de estos servicios.

- Como se ha observado por otros autores, las partes montañosas con su relieve heterogéneo y mayores precipitaciones son capaces de generar un mayor número de servicios y de mayor valor que las zonas de llanura altamente antropizadas y dedicadas a producción agrícola con clima semiárido cual es el caso de la cuenca del Martín. El mismo relieve aportador de heterogeneidad bio-geofísica y riqueza paisajística puede acentuar la disminución de estos servicios en estas zonas si su gestión y uso no son los apropiados poniendo en peligro la continua producción de servicios capaces de influir en el buen estado ecológico de toda la cuenca si existen factores alteradores importantes, como la erosión. Este es el caso de algunas subcuencas o zonas de la parte sur de la cuenca del Martín con escasa cobertura vegetal y otras en donde se ubican zonas mineras en las que se han estimado tasas de erosión mayores de 17 t ha^{-1} $año^{-1}$ y 1-2 servicios de los ecosistemas.

- Se ha elaborado un marco para la definición y jerarquización de zonas de restauración a escala de cuenca hidrográfica basado en mapas jerárquicos de la erosión, como factor de alteración, y de los servicios de los ecosistemas, como variables de estado, para identificar zonas prioritarias de restauración basadas en umbrales de erosión y números de servicios producidos utilizando datos básicos en mapas con pixel de 20 m × 20 m. Esta metodología incluye tres niveles espaciales de análisis, en este caso de la cuenca del Martín, definidos por tres niveles diferentes de agregación espacial de los pixeles que forman los mapas de erosión y de provisión de servicios, que son agregados para evaluar su congruencia espacial. La elaboración de mapas y patrones multi-escala (con diferente agregación de pixeles) ha permitido identificar la resolución ideal de análisis espacial para seleccionar áreas prioritarias de restauración de la erosión para mejorar la provisión de servicios ecosistémicos en la cuenca del Martín. Para la clasificación de zonas se establecieron tres umbrales de erosión, que coinciden con límites ecológicos para el establecimiento de la vegetación en el área de estudio, contrapuestos con el número de servicios. Graficando los resultados notamos como reduciendo la agrupación espacial de los pixeles, creando subcuencas más pequeñas, el grado de precisión y la diversificación en la definición de zonas de la cuenca con diferentes valores combinados de erosión y de servicios ecosistémicos aumenta. Entre las subcuencas generadas al tercer nivel de agregación de pixeles, destacan zonas mineras que tienen un área media de 1,5 Km^2, al sur de la cuenca, donde se observan tasas de erosión mayores de 17 t ha^{-1} $año^{-1}$ y alto número de servicios. Una escala de análisis a escala de cuenca fluvial sirve para tener una visión amplia de condiciones diferenciadas entre partes amplias de la cuenca.

- Esta aproximación, combinando en una escala el factor de alteración, la erosión, y el factor de estado, la provisión de servicios de los ecosistemas, constituye un enfoque lógico y práctico para la selección y establecimiento de una jerarquía de áreas de restauración; y su representación gráfica mediante SIG, una herramienta útil para la selección y establecimiento de una jerarquía espacial de áreas de restauración en la cuenca hidrográfica. La disponibilidad de los datos con una resolución óptima y el análisis de necesidades de los interesados se requieren para que este enfoque pueda mostrar todo su potencial.

- De este estudio se deriva un marco conceptual adaptable y fácilmente aplicable a la definición de zonas de actuación de mejora del funcionamiento ecológico natural en otras cuencas o territorios que tienen perturbaciones ambientales que amenazan la provisión de servicios del ecosistema (exceptuando los de producción). A la escala de cuenca hidrográfica, se recomienda ordenar las actuaciones de restauración priorizando las zonas identificadas a la escala espacial adecuada como de alto factor de disturbio y de provisión de servicios (por existir riesgo de alteraciones y de pérdida de servicios); seguidas de zonas con alto factor de disturbio y baja provisión de servicios (por haber una potencial ganancia de servicios derivada de la restauración que obraría disminuyendo el factor de alteración), y con menor interés de realizar actuaciones de restauración ambiental las zonas con bajo riesgo de alteración ambiental y donde puede existir una mejora en la provisión de servicios; dejando sin actuaciones las zonas con bajo impacto y alta provisión de servicios, ya que no existe riesgo para la provisión de estos servicios.

- Como conclusión general se ha comprobado con este trabajo que la evaluación de los servicios de los ecosistemas es una base útil para la planificación de la restauración ecológica y la gestión de un territorio formado por un mosaico de ecosistemas.

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9. Appendix

9.1. ***Literature search and data extraction***

We followed the methodology of Egoh et al. (2007) using the Web of Science (http://www.newisiwebofknowledge.com) to search for peer-reviewed publications from 1998-2010 (February), written in the English language. We limited our search to 1998 and beyond because this is when it was consistently introduced the terminology "ecosystem services" in published literature by Daily (1997). These publications among others create a clear increase of studies which cite ecosystem services (see Fig. 1in, Fisher et al., 2009). We were using the term "restoration project" AND in the advanced search on ISI using the Booleans AND associated with the following search terms: "ecological restoration", "restoration planning", "ecological rehabilitation", "ecological reclamation", "ecological management", "water quality", "priority area", "area identification", "stream restoration", "planning restoration", "restoration plan", "landscape restoration", "river basin", "watershed", "catchments" and "restoration goals". This search identifies a total of 414 studies. We then conduct a search on this sample with EndNote, selecting for Any Field the phrases "ecosystem services", "restoration", and, as they are sometimes used interchangeably, "watershed", "basin" and "catchment". Due to the small result obtained (three) we just use the words "ecosystem services" and "restoration" but the result was small (eleven) so we decided to read the abstracts of all 414 to search for all papers that include ecosystem services that are not quoted literally according to the classical definitions used by MA (2005).

9.1.1. **Data extraction**

We partially followed the data extraction of the methodology of Rey-Benayas et al. (2009) examining the titles and abstracts of the 414 references to determine how closely they aligned with our selection criteria of ecosystem services within basin areas thereby determining their inclusion in this review. If the manuscript reported on measures of one or more ecosystem services and/or biodiversity in relation to restoration at the basin scale the study was included. During this research we eliminated 310 because we do not consider the theory modelling, animal restoration, review, conservation projects, marine projects or the studies just not consider the river basin scale and excluded other 28 papers from the sample because not pertinent (energy, dental medicine, radiology etc.). Finally we selected 45 by their implicit link reference to ecosystem services related with a basin restoration projects. After an accurate reading we selected and included for revision 13 studies that make clearly reference to ecosystem services in a river basin restoration context and classify them following the criteria at the MA (2005). In the following Table 8 they are listed.

9.2. ***Articles reviewed***

Table 8. Ecosystem services founded in 13 basin-scale restoration plans.

Ecosystem service	**Method of identification**	**Source**
Supporting		
Biodiversity support	Aerial Photo Land cover Stream sinuosity	(Rayburn and Schulte 2009)
Biodiversity support, Habitat (Salmon)	Land use, Human population growth	(Fullerton et al. 2009)
Biodiversity support, Habitat (oak Savanna)	Species diversity, avian community richness	(Grundel and Pavlovic 2008)
Biodiversity support, Habitat (birds)	Vegetation provisioning habitat survey	(Vesk et al. 2008)
Biodiversity support, Habitat (Salmon)	Old data restoration Project	(Katz et al. 2007)
Biodiversity support, Habitat (Salmon)	LULC, Aerial Photo, field observation	(Fullerton et al. 2006)
Biodiversity support, Habitat-Cultural	Abiotic and biotic variables	(Nienhuis et al. 2002)
Biodiversity support, Habitat (native plant communities)	Unclear	(Cuevas and van Leersum 2001)

Cultural		
Cultural landscape	Questionnaire local habitants	(Schaich 2009)
Regulatory		
Flood-Drought prevention	Predict impact on climate change published	(Palmer et al. 2009)
Temperature regulation - Habitat (Salmon)	Climate models LULC, Population dynamics, Hydrology model	(Battin et al. 2007)
Water purification	Clean water act	(Novotny 1999)
Provisioning		
Water production	Water monitoring	(Cobourn 1999)

References of the rest papers selected (32)

(Shirazi et al., 1998; Urbanska, Erdt et al., 1998; Bowler 2000; Curnutt et al., 2000; Palmeri and Trepel 2002; Campbell and Mazzotti 2004; Martinez-Abrain, Sarzo et al., 2004; Groninger 2005; Schulte, Pidgeon et al., 2005; Noss et al., 2006; Twedt et al., 2006; Wightman and Germaine 2006; Alexandridis et al., 2007; Mcintire et al., 2007; Johnson et al., 2007; Rumps et al., 2007; Spanhoff and Arle 2007; Wang et al., 2007; Brudvig and Mabry 2008; Mollot and Bilby, 2008; Montgomery and Eames, 2008; Pavao-Zuckerman, 2008; Robbins and Lewis 2008; Sogge, Sferra et al., 2008; Baron, Gunderson et al., 2009; Bradley and Wilcove, 2009; Castillo and Figueroa, 2009; Cha et al., 2009; Fullerton et al., 2009; Howie et al., 2009; Lane and Texler, 2009; Likens et al., 2009; Papanastasis, 2009)

Alexandridis, T. K., V. Takavakoglou, et al. (2007). "Remote sensing and GIS techniques for selecting a sustainable scenario for Lake Koronia, Greece." Environmental Management 39(2): 278-290.

Baron, J. S., L. Gunderson, et al. (2009). "Options for National Parks and Reserves for Adapting to Climate Change." Environmental Management 44(6): 1033-1042.

Bowler, P. A. (2000). "Ecological restoration of coastal sage scrub and its potential role in habitat conservation plans." Environmental Management 26: S85-S96.

Bradley, B. A. and D. S. Wilcove (2009). "When Invasive Plants Disappear: Transformative Restoration Possibilities in the Western United States Resulting from Climate Change." Restoration Ecology 17(5): 715-721.

Brudvig, L. A. and C. M. Mabry (2008). "Trait-based filtering of the regional species pool to guide understory plant reintroductions in Midwestern oak savannas, USA." Restoration Ecology 16(2): 290-304.

Campbell, M. R. and F. J. Mazzotti (2004). "Characterization of natural and artificial alligator holes." Southeastern Naturalist 3(4): 583-594.

Castillo, J. M. and E. Figueroa (2009). "Restoring Salt Marshes Using Small Cordgrass, Spartina maritima." Restoration Ecology 17(3): 324-326.

Curnutt, J. L., J. Comiskey, et al. (2000). "Landscape-based spatially explicit species index models for Everglades restoration." Ecological Applications 10(6): 1849-1860.

Cha, S. M., Y. S. Ham, et al. (2009). "Evaluation of pollutants removal efficiency to achieve successful urban river restoration." Water Science and Technology 59(11): 2101-2109.

Groninger, J. W. (2005). "Increasing the impact of bottomland hardwood afforestation." Journal of Forestry 103(4): 184-188.

Howie, S. A., P. H. Whitfield, et al. (2009). "Can Analysis of Historic Lagg Forms Be of Use in the Restoration of Highly Altered Raised Bogs? Examples from Burns Bog, British Columbia." Canadian Water Resources Journal 34(4): 427-440.

Johnson, T. E., J. N. McNair, et al. (2007). "Stream ecosystem responses to spatially variable land cover: an empirically based model for developing riparian restoration strategies." Freshwater Biology 52(4): 680-695.

Martinez-Abrain, A., B. Sarzo, et al. (2004). "Unforeseen effects of ecosystem restoration on yellow-legged gulls in a small western Mediterranean island." Environmental Conservation 31(3): 219-224.

Mollot, L. A. and R. E. Bilby (2008). "The use of geographic information systems, remote sensing, and suitability modeling to identify conifer restoration sites with high biological potential for anadromous fish at the Cedar River Municipal Watershed in western Washington, USA." Restoration Ecology 16(2): 336-347.

Montgomery, J.A. and Eames, J.M., 2008. Prairie Wolf Slough Wetlands Demonstration Project: A Case Study Illustrating the Need for Incorporating Soil and Water Quality Assessment in Wetland Restoration Planning, Design and Monitoring. Restoration Ecology, 16 (4), págs.618-628.

Noss, R. F., P. Beier, et al. (2006). "Recommendations for integrating restoration ecology and conservation biology in ponderosa pine forests of the southwestern United States." Restoration Ecology 14(1): 4-10.

Palmeri, L. and M. Trepel (2002). "A GIS-based score system for siting and sizing of created or restored wetlands: Two case studies." Water Resources Management 16(4): 307-328.

Papanastasis, V. P. (2009). "Restoration of Degraded Grazing Lands through Grazing Management: Can It Work?" Restoration Ecology 17(4): 441-445.

Pavao-Zuckerman, M. A. (2008). "The Nature of Urban Soils and Their Role in Ecological Restoration in Cities." Restoration Ecology 16(4): 642-649.

Rumps, J. M., S. L. Katz, et al. (2007). "Stream restoration in the Pacific Northwest: Analysis of interviews with project managers." Restoration Ecology 15(3): 506-515.

Schulte, L. A., A. M. Pidgeon, et al. (2005). "One hundred fifty years of change in forest bird breeding habitat: Estimates of species distributions." Conservation Biology 19(6): 1944-1956.

Shirazi, M. A., P. K. Haggerty, et al. (1998). "The role of thermal regime in tundra plant community restoration." Restoration Ecology 6(1): 111-117.

Sogge, M. K., S. J. Sferra, et al. (2008). "Tamarix as habitat for birds: Implications for riparian restoration in the southwestern United States." Restoration Ecology 16(1): 146-154.

Spanhoff, B. and J. Arle (2007). "Setting attainable goals of stream habitat restoration from a macroinvertebrate view." Restoration Ecology 15(2): 317-320.

Urbanska, K. M., S. Erdt, et al. (1998). "Seed rain in natural grassland and adjacent ski run in the Swiss Alps: A preliminary report." Restoration Ecology 6(2): 159-165.

Mcintire, E.J.B., Schultz, C.B. & Crone, E.E., 2007. Designing a network for butterfly habitat restoration: where individuals, populations and landscapes interact. Journal of Applied Ecology, 44(4), págs.725-736.

Robbins, J.L. & Lewis, L.Y., 2008. Demolish it and They Will Come: Estimating the Economic Impacts of Restoring a Recreational Fishery1. JAWRA Journal of the American Water Resources Association, 44(6), págs.1488–1499.

Twedt, D.J., Uihlein, W.B. & Blaine Elliott, A., 2006. A spatially explicit decision support model for restoration of forest bird habitat. Conservation Biology, 20(1), págs.100–110.

Wang, Z.Y. et al., 2007. Principles of river training and management. Int. J. Sediment Res., 22(4), págs.247-262.

Wightman, C. S. and S. S. Germaine (2006). "Forest stand characteristics altered by restoration affect Western Bluebird habitat quality." Restoration Ecology 14(4): 653-661.

9.3. *RUSLE factors description*

$$A = R * K * L * S * C * P$$

A is the average soil loss from sheet and rill erosion, reported here in tons per hectare per year (t ha^{-1} yr^{-1}).

R is the rainfall-runoff factor, representing the erosion energy in MJ mm ha^{-1} h^{-1} yr^{-1} according to the methodology of Renard et al. (1997), and it is the average annual summation (EI) values in a normal year's rainfall. The erosion index is a measure of the erosion force of a specific rainfall event. When other factors are constant, storm losses from rainfall are directly proportional to the product of the total kinetic energy of the storm (E) times its maximum 30-minute intensity (I).

The *K* factor is the soil erodibility factor, which represents both the susceptibility of soil to erosion and the rate of runoff, as measured under the standard unit plot condition expressed in (t h MJ^{-1} mm^{-1}) (Renard et al. 1991). In RUSLE, factor *K* considers the whole soil, and factor Kf considers only the fine-earth fraction, i.e., material with a <2.00 mm equivalent diameter. For most soils, Kf = *K*

Only *R* and *K* have units; those units, multiplied together, give the erosion in units of mass per area and time. Each of the other terms scales the erosion relative to specified experimental conditions (>1 is faster than under those experimental conditions; <1 is slower). The remaining factors are non-dimensional scaling factors.

The *LS* factors are topographic factors describing the combined effect of slope length and steepness, and they are calculated with the approach of Moore and Wilson (1992) as a function of the net contributing area ($\chi\eta$) and the slope angle (θ, radians).

$$LS = \left(\frac{\chi\eta}{22.13}\right)^{0.4} . \left(\frac{\sin\theta}{0.0896}\right)^{1.3} \qquad \text{Formula 1}$$

L It is the ratio of soil loss from the field slope length to that from a 22.1-meter length on the same soil type and gradient. *S* is the slope steepness, representing the effect of slope steepness on erosion, and the ratio of soil loss from the field gradient to that from a 9% slope under otherwise identical conditions.

9.3.1. **Data collection (Measurement for estimating the factors)**

The *R-factor* map for the area was implemented by Angulo-Martínez and Beguería (2009) following the methodology proposed by Renard et al. (1997) using the SAIH system (automatic hydrological information network) of the Hydrographic Confederation of the Ebro River. Each meteorological station provides precipitation data at a time resolution of 15 min. The system began in January 1997 and is the only dense network in the region providing data at a sub-daily resolution.

More than 110 selected rainfall series were used from those authors to calculate *R*-factor values for the periods May 2005–May 2006, May 2006–May 2007 and May 2008. No high time resolution data were available for the 1955–2008 period, so they used an approximation based on daily rainfall data (Angulo-Martínez and Beguería 2009).

Point estimates were interpolated by means of smoothing splines with the geostatistical analysis package of the GIS software to create *R-factor* maps.

9.3.1.1 ***K* Measurements**

The study assessed the soil erosivity factor *K* by selected areas, following the land covers and soil types that were sampled. Because of the lack of detailed soil maps for the study area, it was necessary to analyze the soil samples. A total of 97 sites generally encompassing the spatial variability of soil type-land cover combinations were sampled in triplicate (Fig. 7, p.35) (Foto 4).






Foto 4. Muestreo en diferentes tipos de suelo en la cuenca del Río Martín con su correspondiente cubierta vegetal.

The *K*-factor values were determined from soil texture data (Romkens and Wang 1987) according to the following equation:

$$K_{text} = 0.0034 + 0.0405 \exp\left[-0.5\left(\frac{\log D_g + 1.659}{0.71}\right)^2\right]$$ (Formula 2)

where Ktext is a soil erodibility factor (Mg h MJ^{-1} mm^{-1}) and Dg is the geometric mean weight diameter of the primary soil particles (fraction < 2 mm). Dg was determined using a Coulter laser diffraction particle size analyzer (Coulter LS 230) for the 2–2000 µm fraction, following removal of organic matter. The K-factor values were then corrected to reflect the effect of stones in the soil surface on soil erodibility (Box 1981), according to the following equation:

$$K = K_{text} \exp^{(-0.0278St)}$$ (Formula 3)

where *St* is the weight of stones in the topsoil, expressed as a percentage of the total weight of the topsoil.

We interpolate field data with the soil map, excluding bare rocks and predominantly rocky areas. We generate, for each type of soil, an averaged corresponding value of *K.*

9.3.1.2 ***LS* Measurements**

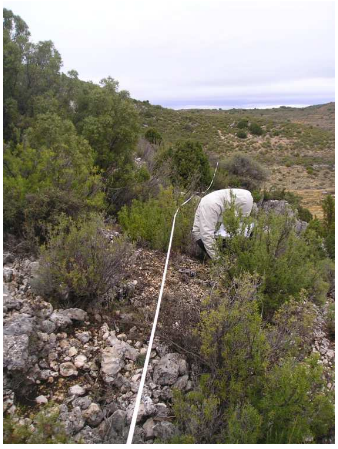
We evaluated *LS* with the flow accumulation tool (ArcMap) using a DEM (Digital Elevation Model) from the Aragón Territorial Information Centre (CINTA) (2006) and a watershed delineation tool to consider the topographical and hydrological effects on soil loss (Fig. 13, C p. 48).

This approach is easy to run within a GIS application and has been satisfactorily used in other Mediterranean areas, such as northwest Spain (Martínez-Casasnovas and Sánchez 2000) and southern Italy (Di Stefano et al. 2000).

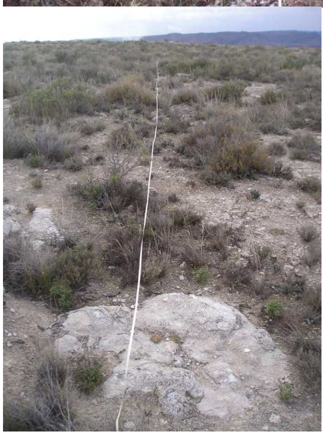
9.3.1.3 **C Measurement**

The field measurement procedures were adapted from the RUSLE manual, and we followed the methodology of González-Botello and Bullock (2012). The measurements were taken at 20 different random points along the 30 m transects placed at the perpendicular direction to the predominant slope, and human trampling in the area was reduced as much as possible .

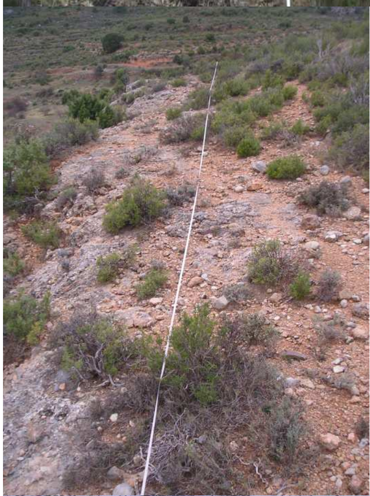
A plumb was dropped at each point. Then, the surface cover percentage was visually estimated within a 10-cm-diameter micro-plot around the plumb using five linear categories (0 = 0-1%, 1 = 1-25%, 2 = 25-50%, 3 = 50-75%, and 4 = 75-100%) (Foto 5).











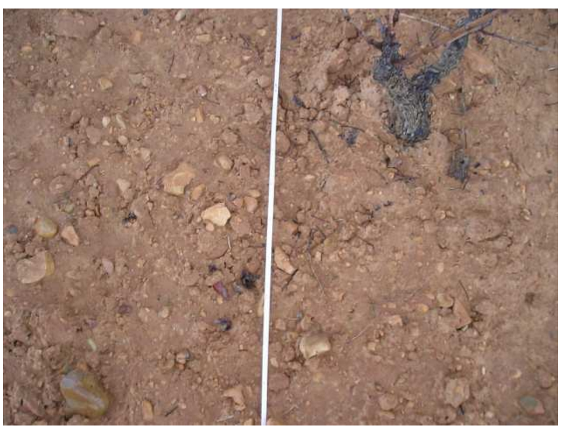



Foto 5. Transectos en la cuenca del Rio Martín para evaluar el factor C del modelo RUSLE. Vegetation transect in different part of the Martín Basin for different vegetation cover

In the absence of long-term experiments, where soil loss is measured from field plots to obtain estimates for any variety of site conditions, it is possible to estimate the *C* factor using a standard calibration of sub-factors. Wischmeier and Smith (1978) identified three major sub-factors that determine the effectiveness of vegetation in limiting soil erosion on rangelands. The first sub-factor includes the canopy cover sub-factor (above-ground plant biomass and the height that raindrops fall from the plant to the soil surface). The second sub-factor includes the soil surface cover (composed of non-eroding material such as rocks and organic litter, plant basal area). The third sub-factor is the residual and tillage sub-factor (root biomass effects and other organic matter in the soil avoiding compaction and facilitating surface stabilization). Prior to the fieldwork, to identify representative sampling sites, a detailed examination of satellite imagery and topographic maps of the river basin was conducted. The *C* factor (Fig. 13 D, p.48) was estimated for each sampling site by using the following equation derived from data in Wischmeier and Smith (1978; Table 10):

$$C = 0.45(e^{[-0.012 \cdot b]}) \cdot (1 - p \cdot e^{[-0.328 \cdot h]}) \cdot e^{(-0.039 \cdot g \cdot [0.24/r]^{0.08})} \quad \text{(Formula 4)}$$

Where h is the canopy height; p the percentage of canopy cover; r is the surface roughness; b was defined by primary productivity according to the methodology described by Weltz et al. (1987); and g is the surface cover. This equation is similar to equations described by Weltz et al. (1987) and Renard et al. (1997).

Once the field measurements were obtained, the next step was to extrapolate the punctual *C* factor values to the entire area of study using a Landsat 5 image corresponding to the sampling period, courtesy of the National Geographic Institute (Ministry of Development) through the National Plan for Remote Sensing (http://blogpnt.wordpress.com/). There are three main approaches to the problem of extracting C from satellite imagery as tools to generalize local field plot samples to a broad area: the classified thematic map method, the Vegetation Index method, and the more complex Linear Spectral Mixture Analysis (LSMA). We use the Genetic Programming methodology described by Puente et al. (2011) to obtain Vegetation Indices (VIs) designed exclusively for our area of study. Genetic programming (GP), as stated by Koza (1992) and Poli et al. (2008), is an evolutionary computation (EC) technique inspired from the principles of biological evolution that is used to create computer programs that learn a user-defined function. The GP approach is able to evolve a population of computer programs. That is, generation by generation, GP stochastically transforms populations of programs into new, hopefully better, populations of programs.

In the GP process, programs are usually expressed as syntax trees rather than as lines of code. The variables and constants in the program (in this case, the reflectance values of NIR and Red bands) are leaves of the tree, which are called Terminals, while the arithmetic operations (+, - and ÷) are internal nodes, called Functions. The sets of allowed Functions and Terminals together form the primitive set of a GP system. In our GP, the Terminal set will be represented by information on the spectral bands, such as Red, NIR, and Green. The function set will be represented by all arithmetic operations (+, -, *, and ÷) because these kind of functions are widely used in common VIs. Both sets of terminals and functions form our primitive set that the GP system will use to create composite operators.

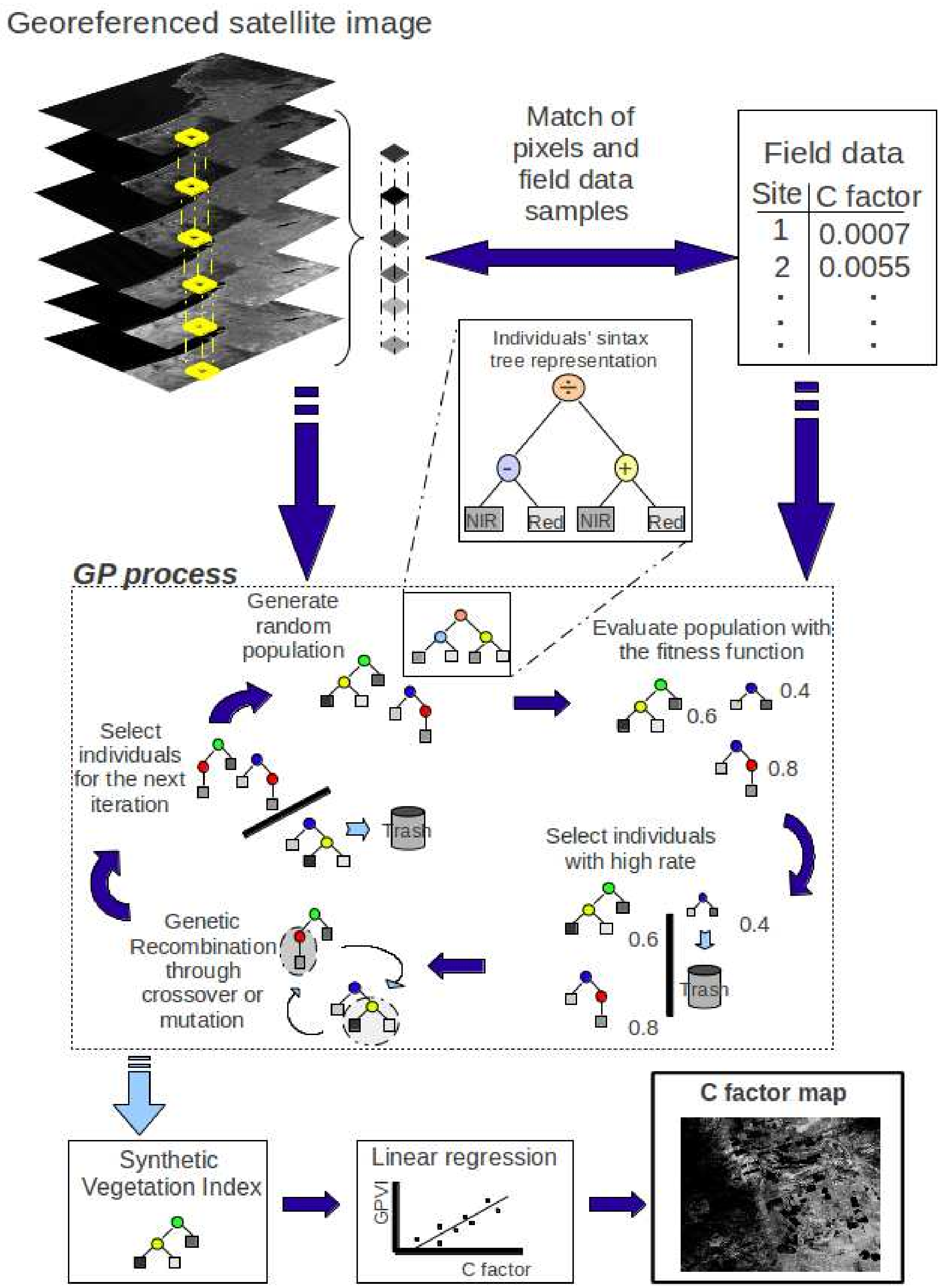


Fig. 36. General flowchart of the methodology to estimate C from vegetation indices synthesized by GP (from Puente et al. 2011).

Fig. 36 shows the flowchart of the procedure developed to generate novel VIs to estimate C. A run of the GP algorithm consists of the following steps. First, satellite imagery is georeferenced to prepare the input data through the identification of all pixels in the spectral bands that match each sample in the field data. Then, the primitive set is defined as follows:

F = {+, − , ∗ , / }

T = {Red, Gr, Bl, NIR, SWIR1, SWIR2, a, b, aG, aR, aNIR, aSWIR1, NDVI, EVI, TSAVI, GV I, SASI },

where Red, Gr, Bl, NIR, SWIR1, and SWIR2 represent the image bands of the Landsat5-TM satellite. Moreover, aG, aR, aNIR, and aSWIR1 characterize the angle between the three consecutive bands considering the previous satellite channels (see [10] for details about how to obtain such angles). In these expressions, the a and b terminals represent the soil line parameters. Finally, to complete the terminals, we consider the most common vegetation indices that are represented by NDV I, EVI, TSAVI, GVI, SASI. The initial population of solutions is now generated. Then, each individual is evaluated by the fitness function. The fitness function is based on the correlation coefficient ρ(x,y) that indicates the strength and direction of the linear relationship between the factor *C* and the evolved vegetation indices. The correlation is 1 in the case of an increasing linear relationship and -1 in the case of a decreasing linear relationship. In this work, we choose to apply the absolute value operator of ρ(x,y) because the closer the coefficient is to either -1 or 1, the stronger the correlation between the variables. Hence, the fitness function is defined as follows:

$$Q = \max(|\rho_{x,y}|) \text{ , such as } \rho_{x,y} = \frac{cov(x,y)}{\sigma_x \sigma_y} = \frac{E((x-\mu_x)(y-\mu_y))}{\sigma_x \sigma_y} \text{ , (Formula 5)}$$

where E is the expected value and cov means covariance. x represents the RUSLE's C factor, y is the evolved vegetation index and ρx,y is defined within the range {ρx,y : −1 ≤ ρx,y ≤ 1}.

The next step is to select candidate solutions to rank all individuals and discard the solutions with low fitness. Then, the genetic recombination between selected trees is performed through crossover, and mutation is then applied. Finally, the next population is created using the stochastic universal sampling method. These steps are iterated until the maximum number of generations is reached. From the population of the last generation, only the best solution is kept to perform regression analysis. The result of the linear regression is a C factor map for the entire region.

Additionally, we calculated cover percentage to obtain information about how cover influences erosion. We developed a *C* factor map with the assumption that Cover Density (CD)

= 0.00 if the *C* factor is 0.45 and 1.0 if the *C* factor is 0. By simple algebra, we can say that CD = 1.0 - (1/0.45)*C = 1.0 -2.22*C. This calculation is an empirical estimate of CD based on the end-member extreme cases, and it includes cover effects from both ground cover and canopy cover (including live and dead material).

9.4. *Factor results*

9.4.1. *R* values

In the Martín Basin, the rainfall erosivity factor (*R*) had a mean value of 603 MJ mm $ha^{-1} h^{-1} yr^{-1}$, with a minimum value of 390 MJ mm $ha^{-1} h^{-1} yr^{-1}$ in the north, a predominantly flat area, and a maximum value of 905 MJ mm $ha^{-1} h^{-1} yr^{-1}$ in the southern highlands, where steep zones are located. The large variation in the rainfall erosive factor is a consequence of the highly variable interannual and seasonal precipitation behavior, with long dry periods alternating with some wetter periods (October-November), although April-May and summer often benefit from the triggering high-intensity storms (Peña et al. 2002).

9.4.2. K values

The soil erosivity factor (*K*) ranged from 0.022 t ha h ha^{-1} MJ^{-1} mm^{-1} to 0.041 t ha h h a^{-1} MJ^{-1} mm^{-1}. Because the value of K belongs to the most common soil, which is more widespread in the basin, we decided to use a constant value for a *K* factor of 0.03, that is, an average value.

9.4.3. *LS* values

The *LS* factor map, generated from the DEM, had a mean value of 3.7, and the *LS* ranged from 0 to 49. The *LS* ranges 0-1, 1-2, 2-4, 4-8, 8-16, >16 covered 32%, 17%, 19%, 18%, 11% and 2%, respectively, of the study area. Thus, the values of *LS* in the Martín River Basin are distributed mainly in the 3 lower ranks (68%), with 0-1 being the most abundant at 32%. Only approximately 2% of the basin shows steep, long slopes that favor very high erosion.

9.4.4. *C* values

The *C* factor map (Fig. 13 D, p. 48) had a mean value of 0.28 in the Martín Basin. The highest mean values were associated with unproductive uses, such as open coal mines and dry agriculture (Table 9).

Table 9 C value for the different land covers at the Martín Basin

Land Use-Land Cover	MEAN
Dry agriculture	0.37
Grassland-Shrubland	0.26
Grassland	0.28
Olive tree	0.29
Vineyard-Fruit tree	0.34
Unproductive	0.35
Water	0.36
Irrigation	0.27
Scrub	0.25
Poplar and aspen	0.17
Other hardwoods	0.13
Conifers	0.13
Olive tree and Vineyard	0.32
Conifers and hardwood	0.11

The lowest values were related to conifers and hardwood and poplar and aspen. In the riparian areas with dense scrubland, the mean value was 0.25 (Table 9).

Agradecimientos

Parece que he llegado al final del túnel y es el momento de mirar atrás, haré lo mejor para recordar y agradecer tantas, pero tantas personas dispersas por el mundo que han sido fundamentales para el desarrollo de estas “cuatros ojas” que tanto sudor costaron.

Estoy con la cabeza como un bombo que no da para más…mañana deposito no os lo toméis a mal si me he olvidado de alguien meteré un errata corrige y añadiré unos achuchones!!

En primer lugar agradecer mi director Francisco Comín por haberme engañado trayéndome a Zaragoza desde Spaghettiland para embarcarme en esa odisea que fue mi tesis, gracias sobre todo por el grande apoyo laboral y humano especialmente en el último tirón, siempre te seré agradecido. No te lo he dicho nunca pero Viaroli me envió para demostrarte que los italianos no somos vagos jejejeje.

Ahora el International Team!

En orden de aparición el incansable Stephen V. Smith que se vio aterrizar en casa un becario perdido y bastante incapaz de entender los contenidos y el americano a la primera en el CICESE de Ensenada y no obstante todo me ha seguido enseñado a distancia con ENOOOOORME paciencia, como un padre a un hijo en la redacción de nuestro artículo. A rueda va Cesar Puente su becario que está triunfando en la vida por San Luis Potosí, gracias por haberme dedicado vuestro tiempo y ayuda, ¡¡QUE VIVA MEXICO CABRONES!!

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