MODELING ECOSYSTEM SERVICES
IN A MEDITERRANEAN AREA RIVER BASIN

(MODELIZACIÓN DE SERVICIOS ECOSISTÉMICOS EN UNA CUENCA HIDROLÓGICA
DEL ÁREA MEDITERRÁNEA)

Tesis doctoral

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ABSTRACT

Modeling is the process to idealize real-world situations through simplifications in order to obtain a model. However, model estimations lead to uncertainties that have to be evaluated formally. The role of the sensitivity analysis (SA) is to assign model output uncertainty based on the inputs and can increase confidence in model, however, it is often omitted in modelling, usually as a result of the growing effort it involves. In addition, the balance between accuracy and simplicity is not easy to assess. For this reason, when a model is developed, it is necessary to test it in order to understand its behavior and to include, if necessary, more complexity to get a better response.

Ecosystem services are the conditions and processes through which natural ecosystems, and their constituent species, sustain and fulfill human life. The relevance of ecosystem services and the need to better manage them and their associated benefits have stimulated the emergence of models and tools to measure them. InVEST, Integrated Valuation of Ecosystem Services and Tradoffs, is one of these ecosystem services-specific tools developed by the Natural Capital Project (Stanford University, USA). As a result of the growing interest in measuring ecosystem services, the use of InVEST is anticipated to grow exponentially in the coming years. However, apart from model
development, making a model involves other crucial stages such as its evaluation and application in order to validate estimations.

The work developed in this Thesis tries to help in this relevant and imperative phase of the modeling process, and does so in two different ways. The first one is to conduct a sensitivity analysis of the model, which consists in choosing and applying a methodology in an area and analyzing the results obtained. The second is related to the in-stream processes that are not modeled in the current model, and consists in creating and applying a methodology for testing the streams role in the InVEST nutrient retention model in a case study, analyzing the results obtained.

The results of this Thesis will contribute to the understanding of the uncertainties involved in the modeling process. It will also illustrate the need to check the behavior of every model developed before putting them in production and illustrate the importance of understanding their behavior in terms of correctly interpreting the results obtained in light of uncertainty. The work in this Thesis will contribute to improve the InVEST platform, which is an important tool in the field of ecosystem services. Such work will benefit future users, whether they are researchers (in their future research), or technicians (in their future work in ecosystem conservation or management decisions).
RESUMEN

La modelización es un proceso por el que se obtienen modelos de los procesos del ‘mundo real’ mediante la utilización de simplificaciones. Sin embargo, las estimaciones obtenidas con el modelo llevan implícitas incertidumbre que se debe evaluar. Mediante un análisis de sensibilidad se puede mejorar la confianza en los resultados, sin embargo, este paso a veces no se realiza debido básicamente al trabajo que lleva consigo este tipo de análisis. Además, al crear un modelo, hay que mantener un equilibrio entre la obtención de resultados lo más exactos posible mediante un modelo lo más sencillo posible. Por ello, una vez creado un modelo, es imprescindible comprobar si es necesario o no incluir más procesos que en un principio no se habían incluido.

Los servicios ecosistémicos son los procesos mediante los cuales los ecosistemas mantienen y satisfacen el bienestar humano. La importancia que los servicios ecosistémicos y sus beneficios asociados tienen, junto con la necesidad de realizar una buena gestión de los mismos, han estimulado la aparición de modelos y herramientas para cuantificarlos. InVEST (Integrated Valuation of Ecosystem Services and Tradoffs) es una de estas herramientas específicas para calcular servicios eco-sistémicos, desarrollada por Natural Capital Project (Universidad de Stanford, EEUU). Como resultado del creciente
interés en calcular los servicios eco-sistémicos, se prevé un incremento en la aplicación del InVEST.

La investigación desarrollada en esta Tesis pretende ayudar en esas otras importantes fases necesarias después de la creación de un modelo, abarcando los dos siguientes trabajos. El primero es la aplicación de un análisis de sensibilidad al modelo en una cuenca concreta mediante la metodología más adecuada. El segundo es relativo a los procesos dentro de la corriente fluvial que actualmente no se incluyen en el modelo mediante la creación y aplicación de una metodología que estudiara el papel que juegan estos procesos en el modelo InVEST de retención de nutrientes en el área de estudio.

Los resultados de esta Tesis contribuirán a comprender la incertidumbre involucrada en el proceso de modelado. También pondrá de manifiesto la necesidad de comprobar el comportamiento de un modelo antes de utilizarlo y en el momento de interpretar los resultados obtenidos. El trabajo en esta Tesis contribuirá a mejorar la plataforma InVEST, que es una herramienta importante en el ámbito de los servicios de los ecosistemas. Dicho trabajo beneficiará a los futuros usuarios de la herramienta, ya sean investigadores (en investigaciones futuras), o técnicos (en futuros trabajos de toma de decisiones o gestión ecosistemas).
Chapter 1

1. INTRODUCCION

1.1. BACKGROUND

Ecosystem services are the conditions and processes through which natural ecosystems, and their constituent species, sustain and fulfill human life. Natural ecosystems maintain biodiversity and produce services that benefit people in numerous ways. Although largely neglected in the past, their relevance has grown substantially in recent years (Daily, 1997; MA, 2005).

At the same time, interest in measure ecosystem services has increased since their recognition as key factors in environmental management (Chambers et al., 2005; Goldman et al., 2008).

The importance of ecosystem services has stimulated the emergence of models and tools to measure them. This type of information is likely to be valuable in the decision making process. In addition, these tools would help in future
predictions such as how global change or changes in land uses will affect the production of ecosystem services. Policy makers require reliable predictions in order to design effective mitigation or adaptation measures.

Therefore, it is crucial to evaluate and minimize models uncertainty to avoid bias in decision making (Chavas, 2000; National Research Council, 2005). Rusell (1949) has illustrated this point with more optimism: ‘When one admits that nothing is certain one must, I think, also admit that some things are much more nearly certain than others.’ That is an encouraging statement especially for scientists whose basic vocation is to characterize and reduce uncertainty wherever possible.

InVEST (Integrated Valuation of Environmental Services and Tradeoffs), developed by the Natural Capital Project, is a platform that maps and quantifies ecosystem services and how they benefit society. The Natural Capital Project is a partnership combining research innovation from Stanford University and the University of Minnesota with the global reach of conservation science and policy at The Nature Conservancy and the World Wildlife Fund (www.naturalcapitalproject.org).

InVEST is a free and open-source software suite to inform and improve natural resource management and investment decisions. Despite the fact that the first version of this tool (version 1.0) is relatively new (Tallis et al., 2008), its development is an active effort will continue during the next years. During the period of the present Thesis research, several versions of InVEST have been used, each being the most up-to-date available at the time.
The modeling process—i.e. making a model, which is a simplified representation of the reality— is a huge and consuming time commitment, which should always be supported by previous work like occurs in other science areas.

Although model development is a huge and difficult task in itself, the completion of the first version is not the end of the modeling process. In addition to the development, making a model involves other crucial stages such as model evaluation and application.

For instance, it is mandatory to cross-validate every model with independent measurements in the situations for which it is designed to perform reliably.

Therefore, before applying the model, a sensitivity analysis (response of the output/s model to changes in inputs) has to be conducted and the results should be compared to the measured responses of these inputs in order to effectively follow the model evaluation and be confident in its estimations (Figure 1, Smith & Smith, 2007).
Due to the fact that a model is always a simplification of reality, the modeler decides which aspects of reality are important, which is one of the most important principle in modeling: the principle of parsimony or Ockham’s razor.

The principle of parsimony states that ‘of the two competing explanations, both of which are consistent with the observed facts, we regard it as right and obligatory to prefer the simpler’ (Barker, 1961).

In this way, the model developed should not be more complex than it is absolutely necessary to describe the required outputs. However, it is not easy to assess the balance between accuracy and simplicity.

For this reason, when a model is developed, it is necessary to test it in order to understand its behavior and to include, if necessary, more complexity to get a better response.
1.2. PROBLEM STATEMENT

As a result of the growing interest in measuring ecosystem services, the use of InVEST is anticipated to grow exponentially in the coming years. However, apart from model development, making a model involves other crucial stages such as the evaluation and application of the model in order to validate estimations.

The work developed in this Thesis tries to help in this relevant and imperative phase of the modeling process, and does so in two different ways.

The first one is the sensitivity analysis of the model, which consists in choosing and applying a methodology in an area and analyzes the results obtained.

The second is related to the in-stream processes that are not modeled in the current model, and consist in creating and applying a methodology for testing the nutrient retention in streams in a case of study and analyzing the results obtained.

The general statement of the problem under which this research has been conducted is therefore divided in two parts. The first part deals with performance of a sensitivity analysis of two of InVEST’s freshwater models (water provision and erosion control). The second part has to do, with the in-stream processes not currently modeled in a third of InVEST’s freshwater models: nutrient retention.
1.2.1. Sensitivity analysis

The sensitivity analysis is one of the essential stages during the development of a model and its components. The distinctive stages encompassed by the modeling process are: the conception of the conceptual model, the representational model, the computational model, code verification, and model validation.

However, model validation includes two imperative steps: calibration and sensitivity analysis. These analyses are significant in understanding how the model varies as a function of the input data and the model parameters (Saltelli et al., 2000).

The Natural Capital Project has developed the conceptual model, representational model, computational model and code verification for the entire InVEST platform. However, it is necessary to perform validation with real-world data. A previous research made by Terrado, et al. (2014) has calibrated three InVEST freshwater models in the Llobregat river basin (Spain, part of the Mediterranean Basin).

In the present Thesis, after the sensitivity analysis was performed, the calibration of the InVEST model in the Llobregat river basin was made based in this work (Terrado et al., 2014). Then, the sensitivity analysis of two of the InVEST models (water provision and erosion control) was made with two objectives.
The first objective is to get the results of the analysis in a Mediterranean river basin and be able to draw conclusions pertaining to this particular case, and possibly extrapolate them to other Mediterranean river basins.

The second, and more important objective is to develop a methodology which would be a basis for a possible subsequent implementation of sensitivity analysis in the entire InVEST suite. This would help the final user to make an automatic sensitivity analysis before the application of the model in his particular case of study.

1.2.2. *In-stream processes*

When modeling, one of the most important principles is to prefer the simpler model that is still able to capture the reality processes (principle of parsimony or Ockham’s razor).

To this purpose, a good modeler strives to develop a model that is no more complex than what is absolutely necessary. On the other hand, over simplifications could bring in some uncertainty and a biased explanation of the processes involved.

Therefore, after a model is developed, it is mandatory to test its behavior to isolate possible problems, especially when new research provides a deeper insight into the underlying processes.

Currently, the three InVEST freshwater models only takes into account the contribution of the landscape (terrestrial) to the watershed, but not
the contribution of the in-stream (aquatic), simplifying this part of the watershed as a pipe.

However, recent work highlighted the role that in-stream processes play, especially for nutrient retention in the Llobregat river basin (Aguilera et al., 2012; 2013).

In an attempt to test if this new knowledge is worth being included in the InVEST nutrient retention model, a methodology for testing the role of the in-stream processes was designed.

This methodology was devised in order to test nutrient retention of the stream without the need to change the source code in InVEST.

This methodology re-uses the code already developed avoiding wasting time in making expensive code changes that turn out to be unnecessary and need to be reverted.

### 1.3. OBJETIVES AND RESEARCH QUESTIONS

The main aims of this research are to contribute to the improvement of a widely used tool in ecosystem services modeling, InVEST. First, I applied a sensitivity analysis method in two models (water provisioning and erosion control) in a case study and performed the necessary changes in the model’s code in order to be able to apply it. Second, I developed a methodology in order to include the in-stream processes in the nutrient retention model in the same case study and tested it.
The results of this work are important in and of themselves because of the conclusions obtained. However, this work also provides a guideline for a future implementation of sensitivity analysis and in-stream processes in InVEST suite.

These two additions in the model could improve the application of this important tool not only by supporting further research, but also in future environmental management projects and to better inform policies.

The main objectives of this study could be summed up to:

1. Designing and testing a sensitivity analysis method and use it in InVEST.
2. Apply this sensitivity analysis methodology in a Mediterranean basin and discuss the results as an example of utilization.
3. Develop a methodology in order to include in-stream processes in the InVEST nutrient retention model.
4. Test the methodology in the same basin and discuss the results.

1.4. RELEVANCE OF THIS THESIS

The relevance of this Thesis can be traced in two different ways:

Theoretical direction:

The work in this Thesis will contribute to the understanding of the uncertainties involved in the modeling process. It will also highlight the need to check the behavior of every model developed before its utilization and illustrate the importance to understand their behavior in terms of correctly interpreting the results obtained.
Applicability direction:

The models are powerful tools to address many issues. Improving them and applying the accumulated knowledge from new research will allow users to reduce uncertainty and increase the confidence in the results obtained.

The work in this Thesis will contribute to improve the InVEST platform, which is an important tool in the field of ecosystem services. Such work will benefit future users, whether they are researchers (in their future research), or technicians (in their future work in ecosystem conservation or management decisions).

1.5. THESIS OUTLINE

This Thesis includes work conducted for three years and half as part of the SCARCE project funded by the Spanish Ministry of Science and Innovation (Consolider-Ingenio 2010 CSD2009-00065).

The work on sensitivity analysis was published in two papers in Science of Total Environment (Sánchez-Canales et al. 2012; Sánchez-Canales et al. 2015). The in-stream processes is planned to be published in a future paper.

This Thesis is organized into 8 chapters.

Chapter 1 gives a brief background on the work in this Thesis, introduces the problems addressed, identifies the objectives, the relevance of this research, and a detailed plan.
Chapter 2 presents the state of the art in all the topics necessary to understand the work that will be presented, which include the theoretical concepts, but also the study area in where the work was carried out.

Chapter 3 presents the case study in which all the research was development (the Llobregat river basin, Spain).

Chapter 4 presents the methodology developed for the research.

Chapter 5 and 6 presents the sensitivity analysis of two of the freshwater models. All this research was reported in two manuscripts.

Chapter 7 investigates and tests the in-stream processes and their role in one of the freshwater model.

Chapter 8 summarizes the significant results obtained, and provides suggestions for future studies based on the conclusions drawn from this Thesis.
Chapter 2

2. STATE OF THE ART

2.1. MODELING THE REAL WORLD

Modeling arises in many major disciplines within science, engineering or social sciences as a useful way to make and test hypotheses, revise designs and theories in experimental sciences. Similarly, computational scientists use modeling to analyze complex, real-world problems in order to predict what might happen with some course of action (Shiflet and Shiflet, 2006).

Minsky (1965) defined a model as ‘anything to which experiment can be applied in order to answer questions about a system’. A system is a group of entities linked in some regular interdependence in which the modeler is interested in understanding the behavior of a particular system, the causes of possible changes in the system, and how sensitive it is to certain changes (Giordano et al., 2013).
The definition of model above does not imply that a model is a computer program; however, mathematical models are a subclass of models that can be implemented as computer programs. Using mathematics to understand a process in the real world is really convenient and oftentimes necessary before taking action or even predicting the future.

A real situation usually encompasses so many different pieces that it is impossible to express them all mathematically, thereby the first and crucial step in modeling is to decide which aspects are most important to keep. In this way, the modeler idealizes and simplifies the real-world, which is then distilled into mathematical expressions by applying approximations, theorems, and algorithms.

In modeling terminology, the three parts of every mathematical model are often referred to:

1) fixed parameters are values used to describe a process but remain constant regardless of changing conditions,

2) input variables are the values that are allowed to change in each model run in order to describe specific conditions, and

3) a relationship that joins both parts, for example an equation.

The three following approaches are appropriate ways for linking fixed parameters and inputs variables in a mathematical framework.

The first is to deduct directly a relationship from measured data, without any understanding of the processes involved. The second is to analyze the processes involved and then propose an equation that describes them. The last one is to do both: to propose the form of the equation informed by existing
knowledge about the processes, but setting the parameters in the equation with measured data. Each one has different strengths and weaknesses which makes them suitable for different purposes (Smith and Smith 2007).

The first approach usually yields descriptive and functional models. Deriving a model directly from measured data is used to describe the observed results in an experiment because this procedure is not limited by the need to understand the process.

However, if the bounds of the experiment change, the model built with this approach is unlikely to respond adequately. Thus, the application of this kind of model is usually limited within specific experimental conditions.

The second approach is a suitable way to develop mechanistic models that are predictive but usually non-functional. A model derived from scientific understanding of a system actually can explain for instance causes of observed changes.

However, due to the complexity of environmental and ecological processes, a complete understanding of the system is rarely possible. Fundamental rules in these sciences tend to be treated more like guidelines than strict rules; thus, the model structure is set up using these rules but it is necessary to conduct tests for each specific case, and to infer adjustments to improve the model accuracy.

This kind of models gives approximate solutions and might be too complex to use in many real-world applications because there are many input variables, and some of them are often lacking.
Using both approaches has advantages but also disadvantages. The model derived this way is predictive, mechanistic, and functional. It is predictive because it often fares well at predicting beyond the conditions of a specific experiment (better than a model derived purely from measured data).

In addition, this kind of model is mechanistic because it explains some of the underlying science of the process modeled (it maintains a certain level of scientific understanding and can be used to explain what is happening).

Also, a model obtained this way is functional because can also be applied in some real-world cases with adequate accuracy (it describes the empirical data with less accuracy than if the model was derived only from these data, but still with more accuracy than if the model was derived only from scientific understanding).

<table>
<thead>
<tr>
<th>MATHEMATICAL MODEL TYPE</th>
<th>APPROACH</th>
<th>ADVANTAGES</th>
<th>DISADVANTAGES</th>
</tr>
</thead>
<tbody>
<tr>
<td>Descriptive and functional</td>
<td>Derived directly from measured data</td>
<td>Understand the process unneeded</td>
<td>Application limited within specific experimental conditions</td>
</tr>
<tr>
<td>Mechanistic, descriptive and non-functional</td>
<td>Derived from scientific understanding</td>
<td>Can explain causes of observed changes</td>
<td>Need many inputs and difficult to use in real-world cases</td>
</tr>
<tr>
<td>Mechanistic, predictive and functional</td>
<td>Combined both approaches</td>
<td>Explains some science and can be applied in some real-world cases</td>
<td>Describes empirical data with less accuracy than 1st approach</td>
</tr>
</tbody>
</table>

*Figure 2. Different approaches and different types of mathematical models (based on Smith and Smith, 2007)*
2.2. MODELING ECOSYSTEM SERVICES

2.2.1. Ecosystem services and their importance

Ecosystem services are the conditions and processes through which natural ecosystems, and their constituent species, sustain and fulfill human life. Ecosystem services maintain biodiversity and the production of ecosystem goods (Daily, 1997).

The concept of ecosystem services has increased substantially in importance in the scientific literature since the 1990 (Costanza et al., 1997; Daily, 1997; de Groot, 1992). In the beginning of 2000’s, it was included in the Millennium Ecosystem Assessment as the benefits that people derive from nature (MA, 2005).

The classification of ecosystem services based on the groups previously defined by the MA established four main types (de Groot et al., 2010):

1) provisioning services,
2) regulating services,
3) cultural services, and
4) supporting services.

The need to identify and understand the services provided by ecosystems for avoiding their loss was highlighted in the MA. For example, many provisioning, regulating, cultural, and supporting services are related to water.
Some of the benefits that people most directly received from the services related to water include the provision of drinking water, irrigation water, hydropower, flood mitigation, and opportunities for recreation.

In this way, water yield is a relevant component of the processes that produce these services and their associated benefits; it contributes for example to water availability for consumptive use (e.g. drinking or irrigation) or in situ water supply (e.g. water for hydropower or fisheries) (Vigerstol and Aukema, 2011).

The services of natural ecosystems are very important to our societies because the services of ecological systems and the natural capital stocks that produce them are critical to the functioning of the Earth’s life-support system. They contribute to human welfare, both directly and indirectly, and therefore represent part of the total economic value of the planet (Costanza et al., 1997).

Management decisions can alter for example the distribution of water in the hydrologic cycle through land use change and adversely affect the associated benefits.

The relevance of the ecosystem services and the need of a better management of these services and their associated benefits have stimulated the emergence of models and tools to measure them.

Regarding the tools for modeling ecosystem services, there are two different classes: traditional hydrological tools (i.e. SWAT or VIC) and ecosystem services tools (i.e. InVEST or ARIES).
The traditional hydrological tools focus on ecosystem services drivers and require post processing for ecosystem services assessments. However, the ecosystem services tools represent a new kind of specific tools, focusing mainly on end services and visualization of these services across a landscape (Vigerstol and Aukema, 2011).

### 2.2.2. An ecosystem services tools: the InVEST model

InVEST is the acronym of Integrated Valuation of Ecosystem Services and Tradoffs. The choice of this tool for this research was made because it is an ecosystem services specific tool, and these tools are the most appropriate when the user is interested in a quick examination of specific hydrological services requiring minimal data.

Among the developed ecosystem services specific tools, InVEST uses biophysical relationships that, despite their limits and assumptions, are widely accepted by scientists as good estimates of physical reality.

In addition, the versions of this tool used in the present work have the advantage of being implemented using relatively transparent code that can be modified if needed (Vigerstol and Aukema, 2011).

InVEST, developed by the Natural Capital Project, is a platform that maps and quantifies ecosystem services and how they benefit society. It is a free and open-source software suite to inform and improve natural resource management and investment decisions (www.naturalcapitalproject.org).
InVEST modeling tools consist of a suite of models that estimate the levels and economic values of ecosystem services. InVEST includes models for quantifying, mapping, and valuing the benefits provided by terrestrial, freshwater, and marine systems.

In the suite for freshwater, there are three models that use land-use and land-cover (LU/LC) patterns for these estimations. The models run on a gridded map at an annual average time step, and the results can be reported in either biophysical or monetary terms, depending on the needs and the availability of information.

The biophysical models calculate the relative contribution of the different parts of the landscape to the provision of services. Then, InVEST allows to assign a monetary value to these benefits, however, this step is beyond the scope of this Thesis, and only the calculation in biophysical terms was made.

The conceptual framework of Haines-Young and Potschin (2010), under which ecosystem services are estimated, describes the pathway from ecosystem structures and processes to human wellbeing.

In the mentioned approach, the supply of all possible benefits is the service, whereas benefit is the use of a service by humans even if actual wellbeing has not changed. Each service could have different benefits, and each benefit could be estimated in terms of biophysical value.
Three services estimation are available in InVEST for freshwater: water provisioning, water purification, and erosion control (Figure 3).

Available water for hydropower production was the benefit assessed for the water provisioning service. Available water could be biophysically valued as the energy generated by this water available.

In InVEST, this part was named ‘Hydropower’ (in the tool interface, Figure 4) or ‘Reservoir Hydropower production’ (InVEST 2.2.2 User’s Guide, Tallis et al., 2011).

Avoided sedimentation in reservoirs and higher water quality are two benefits for the erosion control service (also a regulating service) provided by ecosystems through sediment and soil retention. In InVEST,
this part was named ‘Sediment’ (in the tool interface) or ‘Sediment retention model: Avoided dredging and water quality regulation’ (InVEST 2.4.4 User’s Guide, Tallis et al., 2013).

Higher water quality is the benefit assessed for the water purification service (a regulating service), defined in InVEST in terms of retention of total nitrogen (TN) and total phosphorus (TP), ultimately avoiding these nutrients from reaching streams. In InVEST, this part was named ‘Nutrient retention’ (in the tool interface) or ‘Water purification: Nutrient retention’ (InVEST 2.6.0 User’s Guide, Tallis et al., 2013).

Figure 4. The three Freshwater InVEST models (interface view)

The InVEST hydrological models can model different freshwater ecosystem services. These models are based on simplifications of hydrological relationships that are well-known, such as the Budyko curve
(which is part of the water provisioning model) or the Universal Soil Loss Equation (for the erosion control model).

Despite the fact that the first version of this tool (version 1.0) is relatively new (Tallis et al., 2008), its development is an active effort that will continue for the coming years. The most recent version of InVEST has been used throughout the period of this research.

All the versions of the InVEST models used in this Thesis (versions 2.2.2., 2.4.4., and 2.6.0.) run in ArcGIS. The landscape is broken into pixels, whose size is based on the scale of the input data, and performs calculations by pixel using the ArcGIS tools required (Figure 5).

The outputs of InVEST are GIS maps of intermediate modeling steps, final biophysical service levels and economic estimates (if the optional valuation step is run) for each pixel across the landscape. For example, in the InVEST erosion control model include the total sediment reaching a downstream point of interest (such as a dam), the capacity of each pixel to retain sediment, and an estimation of avoided dredge costs provided by retained sediment.

Then, depending on the service being modeled, the results may be combined in groups of pixels in subsequent modeling steps, giving the some results by sub-watershed or watershed).
Figure 5. The InVEST models used (versions 2.2.2., 2.4.4., and 2.6.0.) run in ArcGIS. The data may be input as table, raster or vector maps. All the calculations are made with ArcGIS tools. The results are tables and raster maps that could be shown by pixel, by watershed, and by sub-watershed depending of the result and the model.

a) InVEST water provision model or ‘Hydropower’ or ‘Reservoir Hydropower production’ (version 2.2.2)

This model informs about the total amount of water available in a basin. The amount of water provisioned from each cell in the landscape (water yield) is calculated as the annual amount of rainfall that does not evapotranspire, determined by the cell vegetation characteristics (Canadell et al., 1996). Water demands for
consumptive uses other than those evaluated are removed from the total yield before assessing the benefit.

This InVEST model (Tallis et al., 2011) has three steps referred to as: water yield, water scarcity, and valuation in terms of hydropower energy. Because this water available can be valued in terms of hydropower energy, the tool’s name in the ArcGIS interface appears as ‘Hydropower’.

The first step (water yield) uses data on average annual precipitation, annual reference evapotranspiration and a correction factor for vegetation type, soil depth, plant available water content, land use and land cover, root depth and elevation.

The second step adds saturated hydraulic conductivity and consumptive water use to determine water scarcity. The valuation model (third step) uses data on hydropower market value and production costs, the remaining lifetime of the reservoir, and a discount rate.

As it is explained before, in terms of sensitivity analysis the third step (valuation) is not taken into account in the present work. Consequently, only the first two steps (water yield and water scarcity) are explained. The biophysical models do not consider surface–ground water interactions or the temporal dimension of water supply.

In the InVEST model version used, the annual water yield (Yxj) for each pixel on the landscape (x=1,2,...,X) is calculated as shown below:
where, $AET_{xj}$ is the actual evapotranspiration (annual) on pixel $x$ for LULC$_j$ (LULC class code; e.g., 4 for forest, 3 for grass and shrubland, etc.), and $P_x$ is the annual precipitation on pixel $x$.

The approximation of the Budyko curve developed by Zhang et al. (2001) is used to calculate the evapotranspiration partition of the water balance ($\frac{AET_{xj}}{P_x}$) as follows:

$$Y_{xj} = \left(1 - \frac{1}{1 + w}\right)$$  \hspace{1cm} Eq.(1)

$$\frac{AET_{xj}}{P_x} = \frac{1 + \frac{1}{R_{xj}}}{1 + w}$$  \hspace{1cm} Eq.(2)

$R_{xj}$ is the Budyko dryness index on pixel $x$ for LULC$_j$ (Budyko, 1974) and values greater than 1 means pixels that are potentially arid (Arora, 2002; Budyko, 1974). This dimensionless index is defined by:

$$R_{xj} = \frac{k_{xj} ETo_x}{P_x}$$  \hspace{1cm} Eq.(3)

where $ETo_x$ is the reference evapotranspiration from pixel $x$ and $k_{xj}$ is the plant (vegetation) evapotranspiration coefficient associated with the LULC$_j$ on pixel $x$. $ETo_x$ represents an index of climatic demand and $k_{xj}$ is determined by the characteristics of vegetative characteristics in each $x$ (Allen et al., 1998).
\( w_x \) is the plant-available water coefficient on pixel \( x \). This dimensionless coefficient represents the relative difference in the way plants use soil water for transpiration (Zhang et al., 2001). It can be estimated as:

\[
 w_x = \frac{Z \cdot AWC_x}{P_x} 
\]

Eq.(4)

where \( Z \) coefficient is a seasonality factor that presents the seasonal rainfall distribution and rainfall depths (see Zhang et al., 1995; 2001; 2004; Milly, 1994) with values between 1 and 10.

The \( Z \) coefficient will approach in areas of winter rains 10, whereas in humid areas with rain events distributed throughout the year or regions with summer rains it will approach 1. \( AWC_x \) is the volumetric (mm) plant available water content on pixel \( x \).

Afterwards, the water scarcity value is calculated based on water yield and water consumptive use in the watershed of interest, as their difference.

**b) InVEST erosion control model or ‘Sediment’ or ‘Sediment retention model: Avoided dredging and water quality regulation’ (version 2.4.4)**

The InVEST erosion control model produces spatially explicit outputs at an annual average time scale (Tallis et al., 2013).
This model computes the total amount of sediment exported by estimating the average annual sediment generated by each parcel of land, employing a method based on the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) at the pixel scale.

The USLE integrates information on land use/land cover (LULC) patterns, soil properties, topography, rainfall and climate:

\[ USLE = R \times K \times LS \times C \times P \]

where \( R \) is the rainfall erosivity factor (a measure of the intensity and duration of rainfall events), \( K \) is the soil erodibility factor (which measures the susceptibility of soil particles to detachment and transport by rainfall and runoff), \( LS \) is the slope length-gradient factor (determined by the local topography, calculated according to Moore and Burch, 1986), \( C \) is the cover management factor (representing the specified crop and management relative to tilled continuous fallow) and \( P \) is the support practice factor (representing the effects of contouring or terracing).

Among these factors, \( R \), \( K \), and \( LS \) are physical factors, while \( C \) and \( P \) are mainly human-driven, depending on local agricultural and erosion-control practices. Each pixel’s contribution to the load of suspended sediment in the river is estimated by first calculating erosion with USLE, and then routing eroded soil down the flowpath and allowing retention by present vegetation.
The model quantifies the total amount of sediment retained by estimating how much soil is retained by vegetation on each pixel, both in terms of prevented erosion and in terms of captured sediment from upstream surface water running over a pixel.

In other words, the model first computes the difference between the sediment exported by a hypothetical bare soil and the actual land cover on that pixel, equivalent to RKLS x (1-CP). Next, the model adds the amount of sediment eroded on all upstream sources trapped by that pixel (Figure 6).

![Figure 6. Scheme of retained and exported sediment flows in the InVEST erosion control model.](image_url)
c) InVEST nutrient retention model or ‘Nutrient retention’ or ‘Water purification: Nutrient retention’ (version 2.6.0)

The InVEST nutrient retention model (Tallis et al., 2013) estimates the quantity of pollutants retained for water purification service from the landscape and the benefits of water purification are assessed for two pollutants (total nitrogen, TN, and total phosphorus, TP).

While these single measures ignore many other sources of water pollution, for the present work it is a representative way to estimate the water purification service.

This model has three steps referred to as: water yield, water scarcity, and valuation of those pollutants retained. The first step calculates annual average runoff from each parcel like in the first step of the InVEST water provision model.

The second step, estimates the amount of pollutant retained by each portion of the landscape (only the land is taking into account) based on the amount of pollutant exported from each land use by the export coefficients.

Those export coefficients, developed by Reckhow et al. (1980), are annual averages of pollutant fluxes derived from several field studies in the United States. Since these coefficients are experimental, a hydrological sensitivity score (HSS) is included in order to take into account the differences in condition between the experimental measures and the conditions where the model is being applied:
\[ ALV_x = HSS_x \cdot pol_x \quad \text{Eq.}(6) \]

where \( ALV_x \) is the Adjusted Loading Value at pixel \( x \), \( pol_x \) is the export coefficient at pixel \( x \), and \( HSS_x \) is the Hydrologic Sensitivity Score at pixel \( x \) which is calculated as:

\[ HSS_x = \frac{\lambda_x}{\lambda_w} \quad \text{Eq.}(7) \]

where \( \lambda_x \) is the runoff index at pixel \( x \), calculated using the following equation, and \( \lambda_w \) is the mean runoff index in the watershed of interest.

\[ \lambda_x = \log \left( \sum U Y_u \right) \quad \text{Eq.}(8) \]

where \( \sum Y_u \) is the sum of the water yield of pixels along the flow path above pixel \( x \) (it also includes the water yield of pixel \( x \)).

When loads per pixel are deduced, the quantity of downstream pixel retention can be calculated as surface runoff that moves the pollutant to the stream. The slope determines the routes water takes down flow paths, and the pollutant retained in each pixel is based on the land cover type’s ability to retain the modelled pollutant.

The model also estimates the amount of pollutant that reaches the stream (nutrient exported) by following the pollutant load of each pixel all the way downstream.
The model then aggregates the load that reaches the stream from each pixel to the watershed level and this load could be compared to a measured value and adjust export coefficients until the modelled load matches the measured one for each point of interest.

The model then decreases the estimated pollutant retained by the amount of ‘allowed’ pollution in the water body of interest to calculate the nutrient retention service. If a threshold is given, the service level is calculated in biophysical terms as follows:

\[
net_x = retained \\
Eq.(9)
\]

where \(retained_x\) is the amount of retention calculated as in the Equation 9, thresh is the total allowed annual load for the pollutant of interest (thresh\_p for phosphorous, thresh\_n for nitrogen) and contrib is the number of pixels on the landscape. Pixel values are then summed (nret\_sm) or averaged (nret\_mn) to the sub-watershed scale to give sub-watershed service outputs in biophysical terms.

2.3. SENSITIVITY ANALYSIS

Sensitivity Analysis (SA) is the study of the relationship between inputs and outputs of a computational model (Saltelli et al., 2000). Sensitivity analysis examines the behaviour of a model by identifying which model components have the most influence over its response by comparing output fluctuations
against variations introduced in the inputs. The variation in model could be introduced through changes in the variables or in the parameters.

Sensitivity analysis calculates the degree of correlation between model response and the value of a given input. The model is deemed sensitive to a particular input when small changes in this input generate significant variations in the model output. Sensitivity analysis is closely related to uncertainty analysis but does not necessarily identify the same inputs (Smith and Smith, 2007).

In the last years, sensitivity analysis has become essential in the development and evaluation of environmental models (Saltelli et al., 2000; Jakeman et al., 2006), being a powerful tool for helping in the development, calibration, optimization and application of computational models (King and Perera, 2013).

Using suitable techniques and frameworks, SA gives information such as model structure, correlation between input variables, model behavior with extreme values, and SA can be used as a decision making tool.

In SA, model inputs are modified within predefined ranges, representing all the plausible values of each model input often dictated by, but not limited to, the range of uncertainty of the input variable.

The model response is examined by comparing the magnitude of change between the response variables and that of each input variable.

Normally, sensitivity analysis techniques enable to explore entire interval of definition for each input factor and do not require any assumption about the model's characteristics (such as linearity or additivity). The sensitivity analysis could be local or global:
• **Local Sensitivity Analysis (LSA):** studies the effect in model response by varying an input, keeping the others constant. The result is a partial derivative for each input – or an approximation of these. The partial derivative explains the rate of change in model response in relation to the rate of change from the input.

• **Global Sensitivity Analysis (GSA):** gives the effect on the output by varying all the inputs over their ranges at the same time. However, a parameter sampling procedure is required, such as Monte Carlo methods, and the result is more complex to interpret than that of LSA. In Saltelli et al. (2000) the statistics used to summarize the information obtained are described, and the GSA are classified into:
  1) global screening methods such as Morris method (Morris, 1991; Campolongo et al., 2007);
  2) variance decomposition methods such as Extended Fourier Amplitude Sensitivity Testing (Extended-FAST) (Saltelli et al., 1999);
  3) regression/correlation-based methods such as the standardized regression coefficients (SRCs) method (Saltelli et al. 2008).

Due to the high complexity of environmental models, the use of the GSA applications has been limited because of their high computational cost (Campolongo et al., 2007; Yang 2011). Despite of this limitation in applying GSA, during the recent years modelers have understood the potentialities of each GSA method applied to complex models (e. g. Massmann and Holzmann 2012; Herman et al. 2013; Zhan et al. 2013).
The Morris method (Morris, 1991) is one particular GSA method that could be considered the state of the art and recommended for its computational efficiency. The typical application for this tool is where many parameters and available resources do not allow to specify probability density functions for a full Monte Carlo analysis. Moreover, if the Morris method indicates that the parameters are independent, it is unnecessary to use methods that take into account the interaction between multiple parameters such as the Monte Carlo analysis. In fact, from a practical point of view, the Morris method was chosen because it needs a relatively low number of model runs in order to produce valid results. In this regard, it is worth noting that each experiment performed in this work (350 model runs) amounted to about 100 CPU hours. Other SA methods, which require more than 5000 models runs demand computation times that are very difficult to satisfy in practice.

2.3.1. The Morris Method

The Morris method (Morris, 1991) is a specialized randomized One-At-a-Time (OAT) sensitivity analysis (SA) methodology. This methodology was chosen because it is very efficient and reliable to identify and rank sensitive inputs in a model (Campolongo et al., 2007; Morris, 1991).

As all SA methods, the purpose of the Morris screening method is to determine the relative influence of each input ($x_i$), defined in the region $\Omega$ of $R^n$, on the model response ($y$) with a minimum number of model runs.
The assumption behind the design of an OAT SA is that the model is most sensitive to some inputs that cause the largest variations in the output, when all the changes are made by the same relative amount.

The main idea of the Morris method is to use the output’s partial derivative with respect to each input \( \frac{\partial y}{\partial x_i} \), employing an OAT-type calculation scheme, where all input variable at each step are fixed, except the one which effect is being calculated.

The standardized effect of a positive or negative \( \Delta \) change (or step) of an input can be calculated with its Elementary Effect (EE – Morris 1991). If \( x_i \) is considered to be a random variable whose sample space is \( \Omega \), each elemental effect, \( EE_i \), will be a random variable whose probability distribution function is denoted by \( F_i \).

The two indices proposed by Morris (1991), namely the mean (\( \mu \)) and standard deviation (\( \sigma \)) of the set of EEs for each input variable, are calculated using:

\[
\sigma_i = \sqrt{\frac{1}{r} \sum_{n=1}^{r} (EE_n - \mu_i)^2}
\]

Eq.(10)

\[
\mu_i = \frac{\sum_{n=1}^{r} EE_n}{r}
\]

Eq.(11)

Campolongo et al. (2007) introduced a different index, the mean of absolute values (\( \mu^* \)) instead of the conventional mean (\( \mu \)). It is calculated as:
\[ \mu_i^* = \frac{\sum_{n=1}^{r}|EE_n|}{r} \]

*Eq.(12)*

The use of the \( \mu^* \) instead of the \( \mu \) is necessary to avoid cancellation of terms within the sum using a simple average as a measure of sensitivity. The index \( \mu_i^* \) gives a better importance measure, without any non-monotonic input to model behavior that could occur in \( \mu_i \), because \( \mu_i^* \) avoids cancellation effects that may happen for \( \mu_i \).

When \( \mu_i^* \) is high with respect to other inputs, the model response is highly sensitive to this input variable. Conversely, an input with a low value of \( \mu_i^* \) has small sensitivity associated to its input variable and causes a relatively low change in output. In addition, high \( \sigma_i \) values indicate possible interactions with other variables and/or that these inputs have a non-linear effect on the output.

The practical problem is that, due to the impossibility (or great complexity) to obtain \( \frac{\partial y}{\partial x_i} \) analytically, the determination of each \( EE_i \) must be approximated numerically:

\[ EE_i(x) = \frac{[y(x_1, x_2, \ldots, x_{i-1}, x_i + \Delta, x_{i+1}, \ldots, x_k) - y(x)]}{\Delta} \]

*Eq.(13)*

where \( \Delta \) is a real number and \( x=(x_1, x_2, \ldots, x_k) \) is any point of the sample space, \( \Omega \), such that \((x_1, x_2, \ldots, x_{i-1}, x_i+\Delta, x_{i+1}, \ldots, x_k)\) is still in \( \Omega \).

This implies working on a discretized version of the above presented concepts. Following Morris’ original strategy, each \( x_i \) must be scaled to take values in the interval [0, 1], i.e., \( \Omega \) is the unit hypercube.
A discretized approach to a regular k-dimensional p-level grid, named region of interest and denoted by $\omega$, must be performed. Then, each $x_i$ may take on values from the set \{0, 1/(p-1), 2/(p-1), ..., 1\}. As is justified in Morris (1991), it is convenient to select an even value for $p$ and $\Delta = p/[2(p-1)]$.

As cited in Patelli et al. (2010), since the variables are usually of different scale, taking into account only $\partial y/\partial x_i$ might not be very informative, so an available practice is the normalization of the derivatives by the standard deviation of factors, $\sigma_i$, divided by the standard deviation of response, $\sigma_y$:

$$EE_i^N = \frac{\sigma_i}{\sigma_y} EE_i$$

Eq.(14)

where superscript N denotes ‘normalized value’.

The traditional OAT SA design (shown in Figure 7a) tests each input individually modifying it between two values (two model simulations) which are considered as one experiment. This traditional OAT SA design requires $2k$ model simulations to determine an EE for each of the $k$ input variables to obtain a r-size sample of the $k$ factors.
However, less computationally expensive sampling plans have been developed for this purpose (Campolongo et al., 2007, 2011; Morris, 1991).

This strategy has the advantage that, although it does not guarantee equal-probability sampling from each $F_i$, at least a certain symmetric treatment of inputs is ensured, which may be desirable in the analysis of complex simulation models (Campolongo et al., 2011; Morris, 1991).

The Morris sampling plan (Figure 7b) is based on the concept of trajectory which is a set of $k+1$ points of $\omega$ such that two consecutive points of the trajectory differ only in their $i$-th component and allow to obtain a $r$-size sample of the $k$ factors at a total cost of $r(k+1)$ model runs.
2.4. IN-STREAM PROCESSES FOR NUTRIENT RETENTION MODELS

Due to the complexity of river networks at the watershed scale, modeling tools are necessary in the determination of the sources and processes of nutrients and their transport. However, it is necessary to strike a balance between approximation and realism in this kind of models.

On one hand, it is often impractical to account for all processes in a model and it is impractical to apply because of the huge amount of data needed, which is also difficult to collect. On the other hand, many simplifications could give biased results and unreliable conclusions.

As mentioned earlier, InVEST modeling tools consist of a suite of models that estimate the levels and economic values of ecosystem services. InVEST runs on a regular grid map in ArcGIS, so the area is discretized in squares (pixels). Working with the InVEST nutrient retention model, only the terrestrial processes is taken into account. Currently, InVEST assumes that pollutants arriving at the stream network of the watershed from the surrounding area are simply transported to the outlet without any in-stream processing or retention. The different land use pixels have different retention coefficients. Each land use retains some of the nutrients that arrive at them, therefore, in the current InVEST version the river pixels do not retain anything of the nutrients arriving at them. The InVEST User’s Manual suggests that ‘the user should consider the likely impact of in-stream processes in any calibration work as this model does not include in-stream processes’ (Tallis et al., 2013).
In the specific case of the water purification service (nutrient retention), the contribution of the river and stream network to removal of nutrient pollutants from runoff water has been recently studied in Europe (Maes et al., 2011). This study shows the important function of rivers, streams and lakes in nutrient retention using different models and scales to this aim (i.e. for Europe the river and stream network is estimated to remove 1.5 million ton of nitrogen from surface waters).

Several models can be used in order to model the nutrient removal by aquatic ecosystems, most of them using mass balance. Thus, the in-stream retention (the total amount of nutrients removed by surface water and the associated changes in water quality) can also be mapped using mass balance. Some examples where models are applied in order to estimates nutrient in-stream retention are the GREEN model (Geospatial Regression Equation for European Nutrient losses) (Grizzetti et al. 2005; Grizzetti and Bouraoui 2006; Grizzetti et al., 2008; Bouraoui et al., 2009), an empirical model based on a world-wide database of observations (Seitzinger et al. 2002), or an integrated model that consists of MONERIS, GREAT-ER and LFBilanz (Berlekamp et al. 2007; Lautenbach et al. 2009).

Specifically in the Llobregat River Basin, recent studies confirm that stream networks play an important role in material export from watersheds (Aguilera et al., 2012; 2013). In those studies a hybrid process-based and statistical model (SPARROW, Spatially Referenced Regression on Watershed Attributes) is applied in the Llobregat River Basin (a highly impaired, high stream order watershed) in order to estimate the loads of nitrogen and phosphorus that reach
the drainage network per year. The contribution of in-stream processes in nutrient transport and retention is highlighted by the SPARROW model.

Therefore, an attempt to include the nutrient retention in the river and stream network in InVEST, and test the results obtained, is justifiable. Developing a methodology for testing the role of the in-stream processes should be the first step. This methodology should be devised in order to test the results by including in-stream processes but minimizing the changes in the InVEST source code.

This methodology would save time because it re-uses the code already developed, simultaneously avoiding wasting time in making expensive code changes that turn out to be unnecessary and need to be reverted.

With the addition of in-stream retention in the InVEST nutrient retention model, not only the overestimation of the terrestrial retention could be compensated, but the benefits of the ecosystem services of the aquatic area could also be evaluated, increasing the model usefulness for watershed management.
Chapter 3

3. STUDY AREA

3.1. MEDITERRANEAN REGIONS

The delivery of hydrological ecosystem services is highly dependent on the characteristics of the watersheds. Climate, LULC and topography have prominently been referenced in guidelines for the provision of services (Brauman et al., 2007). One of the most influential factors in Mediterranean basins is climate, which presents larger extremes than in more humid areas.

Mean annual and extreme climatic conditions are relevant indicators of the hydrological regime in Mediterranean regions, and consequently, both are required to be considered when assessing services at the basin scale. In addition, the Mediterranean region has been globally identified as one of the most vulnerable to global change (Schröter et al., 2005).

Particularly climate change projections in Mediterranean regions are associated with more frequent extreme climatic conditions, which could alter water
availability and impact the delivery of ecosystem services. Different potential impacts are projected, including increased temperatures and reduced vegetation. The hydrological cycle in the Mediterranean areas will intensify through increase in temperatures, rainfall concentration in shorter periods of the year, and more extended droughts (Hisdal et al., 2001).

Associated human impacts through changes in ecosystem services could include drinking water shortages, increased risk of forest fires, shifts in the distribution of species, and agricultural losses, among others (Schröter et al., 2005).

Mediterranean-type ecosystems are found in five regions of the globe (Figure 8): California, central Chile, Mediterranean Basin, southern Cape region and south western and southern Australia, between 40º and 32º latitude, and where land is influenced by cold offshore ocean currents (Cody and Mooney, 1978).

The Mediterranean Basin and California are the most populated, and thus impacted, areas with Mediterranean climate (Ochoa-Hueso et al., 2011). In this context, the Llobregat River basin is an excellent candidate as a case study, since its location in the Mediterranean Basin, the sufficient amount of available data, and the number of previous studies conducted in this basin contributes to make it one of the best places to frame the work.
3.2. STUDY CASE: THE LLOBREGAT RIVER BASIN

The Llobregat River basin is located in the NE of the Iberian Peninsula (Catalonia, NE Spain, Figure 9) and drains an area of 4950 km². The river, which is 156 km long, arises in the Pyrenean Mountains and flows southward into the Mediterranean Sea near the city of Barcelona. The Llobregat River is an important water source for Barcelona and its metropolitan area (over 3 million people).
Figure 9. The Llobregat River basin is a Mediterranean river located in the NE of the Iberian Peninsula (Catalonia, NE Spain).
The basin is dominated by forest cover (38.2%) and by a mixture of grass and shrub land (31.6%). The third land use in importance is cultivated land, where the fraction that is non-irrigated (23.6%) is clearly the most important (with a 0.4% of irrigated cultivated land). The rest of the watershed area (6.2%) is covered by urban landscapes (Figure 10).

![5 LAND USES](image)

**Figure 10. The 5-land uses map of the Llobregat River basin**

The climate is Mediterranean, with strong seasonal fluctuations in temperature and rainfall, and peak rainfall events in spring (March-June) and autumn (September-December). Annual rainfall ranges widely within the river basin, from 400 mm per year in the mid-section to 1000 mm in the upper segments, with 716 mm of annual mean rainfall (Mujeriego, 2006).
There are three reservoirs located in the headwaters of the river basin: La Baells ($115 \times 10^6$ m$^3$), Sant Ponç ($24 \times 10^6$ m$^3$), and La Llosa del Cavall ($80 \times 10^6$ m$^3$). There are several drinking water treatments plants, the largest being located near the river mouth (Figure 11).

*Figure 11. Llobregat river basin: reservoirs (demarcated by points a–c, located in the upper region of the basin), confluences of tributaries, and water treatment plant (located at the outlet of the basin).*
3.2.1. **Study area for the Sensitivity Analysis of the InVEST water provision model**

The input data needed to perform the sensitivity analysis of the InVEST water provision model in the study area on ArcGIS are the precipitation raster, the evapotranspiration, the $Z$ coefficient, and the individual sub-watersheds, which are used in conjunction with the whole watershed to aggregate some of the results.

All the model calculations were made at a pixel resolution of $200 \times 200$ m to capture the spatial heterogeneity of key driving factors such as soil type, rainfall and vegetation type.

The average conditions of precipitation and evapotranspiration in the Llobregat basin were obtained from the information of Ninyerola et al. (2000) modelled for the period of 1951-2000, where a map of their results are shown in Figure 12. Ninyerola et al. (2000) developed and applied in Catalonia (northeast Spain) an empirical methodology for modelling and mapping the air temperature and total precipitation by means of geographical information systems (GIS) techniques.
Figure 12. Map of average precipitation distribution (a), average evapotranspiration distribution (b) for the period [1951–2000] in Llobregat River basin, and (c) the 154 sub-basins and the localization of the reservoirs and the water plant treatment.
A limitation inherent to InVEST is the fact that it models precipitation as an annual average, however, precipitation regime in the Mediterranean area is characterized by high variability in both the spatial and temporal domains (Esteban-Parra et al., 1998; Trigo and Palutikof, 2001), hence, a significant portion of information is lost by averaging over a year.

For the case of the sensitivity analysis conducted in the Llobregat river basin, those inputs were modified by multiplying each map by a coefficient, which varies within a range between 0.1 and 2. In this way, the precipitation and evapotranspiration in the case study can be modified from a decrease of 90% to an increase of 100%. This range of values for precipitation and evapotranspiration are realistic for this study area.

The Z coefficient is a seasonality factor that presents the seasonal rainfall distribution and rainfall depths (Milly, 1994; Zhang and McFarlane, 1995; Zhang et al., 2001; 2004) with values between 1 and 10.

The Z coefficient will approach in areas of winter rains 10, whereas in humid areas with rain events distributed throughout the year or regions with summer rains, it will approach 1. The InVEST User’s guide indicates that previous testing of the model in different watersheds and different eco-regions worldwide shows that this factor is around 4 in tropical watersheds, 9 in temperate watersheds and 1 in monsoon watersheds (Tallis et al., 2011).
Specifically, in the Llobregat river basin, the Z coefficient value is estimated to be between 7 and 9 (Piñol et al., 1991), and this is the range of variation within which this input was modified for the sensitivity analysis (Table 1).

<table>
<thead>
<tr>
<th></th>
<th>Z coefficient</th>
<th>Precipitation multiply by (prec_coef)</th>
<th>Evapotranspiration multiply by (eto_coef)</th>
</tr>
</thead>
<tbody>
<tr>
<td>maximum value</td>
<td>7.00</td>
<td>0.10</td>
<td>0.10</td>
</tr>
<tr>
<td>minimum value</td>
<td>9.00</td>
<td>2.00</td>
<td>2.00</td>
</tr>
</tbody>
</table>

*Table 1. Factors’ range of variation for the sensitivity analysis of the InVEST water provisioning model*

The model results are going to be reported on sub-watersheds. For the sensitivity analysis of the InVEST water provision model, the watershed has been divided at the water body scale as defined by the local water agency, the Catalan Water Agency (ACA). In this way, dividing the watershed in water bodies, based on the specifications of the Water Framework Directive (WFD 2000), the number of sub-watershed obtained were 154 (Figure 12, c). This high number of results shows a quite fine discretization of this watershed whereby the sensitivity of this model can be quantified in relatively small areas (each sub-watershed).
3.2.2. Study area for the Sensitivity Analysis of the InVEST erosion control model

The input data needed to perform the sensitivity analysis of the InVEST erosion control model in the study area on ArcGIS are the following factors of the Universal Soil Loss Equation (USLE): rainfall erosivity (R), soil erodibility (K), cover management factor (C) for each land use/land cover (LULC), and support practice factor (P) for each LULC (Figure 13).

Figure 13. The factors of the Universal Soil Loss Equation (USLE) for the sensitivity analysis of the InVEST erosion control model in the study area on ArcGIS are the following: rainfall erosivity (R), soil erodibility (K), cover management factor (C) for each land use/land cover (LULC), and support practice factor (P) for each LULC.
The sediment retention efficiency of each LULC is also studied in terms of sensitivity analysis. All these inputs (16) are the ones studied for the benefit of avoided reservoir sedimentation of erosion control service. The calibrated values of all these inputs are based on a previous work made in the Llobregat river basin (Terrado et al., 2014).

For the benefit of higher water quality, apart from the inputs cited above, a seventeenth input is studied, which is the allowed annual sediment load. In this model, all the calculations were made at a pixel resolution of 200 × 200 m to capture the spatial heterogeneity of key driving factors.

However, the model simulates hydrologic processes that are best interpreted at the sub-watershed (and watershed) scale, so the pixel values are aggregated (summed and/or averaged) to match these scales to allow result interpretation and model validation.

The ranges of variation for all of these components are showed in Table 2. For the two raster inputs (the rainfall erosivity, R, and soil erodibility, K), the coefficient that multiplies R varies within a range between 0.1 and 2; however, the coefficient that multiplies K varies within 0.001 and 2.

These ranges for each input are realistic for this study area. The other inputs are mentioned in the InVEST interface as numbers, thus the appropriate range of variation in this case study follows the recommendation of Wischmeier and Smith (1978).

Similarly for the second benefit (higher water quality), the annual load of sediment that arrives at a drinking water treatment plant was fed as an input for the sensitivity analysis. To calculate this particular benefit, the
model compares it to a threshold value of total suspended solids allowed in drinking water.

Due to the lack of such threshold for total suspended solids in the European legislation (DWD, 1998), the range of this threshold value was based on the allowed limits from the US EPA (2009).

<table>
<thead>
<tr>
<th>INPUT</th>
<th>LULC</th>
<th>NUM. INPUT</th>
<th>MIN.</th>
<th>MAX.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rainfall erosivity coefficient (by which the factor is multiplied)</td>
<td></td>
<td>1</td>
<td>0.100</td>
<td>2</td>
</tr>
<tr>
<td>Soil erodibility coefficient (by which the factor is multiplied)</td>
<td></td>
<td>2</td>
<td>0.001</td>
<td>2</td>
</tr>
<tr>
<td>Cover management factor a</td>
<td></td>
<td>3</td>
<td>0.001</td>
<td>0.010</td>
</tr>
<tr>
<td>for urban land</td>
<td></td>
<td>4</td>
<td>0.220</td>
<td>0.770</td>
</tr>
<tr>
<td>for non-irrigated cultivated land</td>
<td></td>
<td>5</td>
<td>0.250</td>
<td>0.500</td>
</tr>
<tr>
<td>for irrigated cultivated land</td>
<td></td>
<td>6</td>
<td>0.011</td>
<td>0.230</td>
</tr>
<tr>
<td>for grass and shrub land</td>
<td></td>
<td>7</td>
<td>0.001</td>
<td>0.028</td>
</tr>
<tr>
<td>for forest land</td>
<td></td>
<td>8</td>
<td>0.250</td>
<td>0.900</td>
</tr>
<tr>
<td>for non-irrigated cultivated land</td>
<td></td>
<td>9</td>
<td>0.250</td>
<td>0.900</td>
</tr>
<tr>
<td>for irrigated cultivated land</td>
<td></td>
<td>10</td>
<td>0.250</td>
<td>0.900</td>
</tr>
<tr>
<td>for grass and shrub land</td>
<td></td>
<td>11</td>
<td>0.250</td>
<td>0.900</td>
</tr>
<tr>
<td>for forest land</td>
<td></td>
<td>12</td>
<td>0</td>
<td>10</td>
</tr>
<tr>
<td>Support practice factor b</td>
<td></td>
<td>13</td>
<td>25</td>
<td>40</td>
</tr>
<tr>
<td>for non-irrigated cultivated land</td>
<td></td>
<td>14</td>
<td>25</td>
<td>40</td>
</tr>
<tr>
<td>for irrigated cultivated land</td>
<td></td>
<td>15</td>
<td>40</td>
<td>55</td>
</tr>
<tr>
<td>for grass and shrub land</td>
<td></td>
<td>16</td>
<td>50</td>
<td>90</td>
</tr>
<tr>
<td>for forest land</td>
<td></td>
<td>17</td>
<td>181800</td>
<td>424200</td>
</tr>
</tbody>
</table>

Table 2. Inputs and ranges of variation for the sensitivity analysis of both benefits of the InVEST erosion control model (avoided reservoir sedimentation and higher water quality).

Wischmeier and Smith, 1978
US EPA, 2009
Terrado et al., 2014
In order to analyse spatial sensitivity for the InVEST erosion control model, the river basin was divided into 7 sub-basins. Sub-basins were randomly numbered from 1 to 7, with 1 being the outlet of the basin, where the water treatment plant is located. Reservoirs, demarcated by points a-c, are located in the upper region of the basin and they delimited three sub-basins (2, 3 and 4).

Confluence of tributaries, represented by two points, divided the rest of the basin in four sub-basins (1, 5, 6 and 7), three sub-basins upstream of the three reservoirs (sub-basins numbers 2, 3 and 4, Figure 13), and the other four covering the rest of the basin (sub-basins numbers 1, 5, 6 and 7, Figure 13).

The characteristics of each of the 7 sub-watersheds are shown in Table 3. The differences between the sub-watersheds are mostly related to their area, their slope and the proportion of each LULC in the sub-basin. Regarding their characteristics, the two sub-basins most different are number 2 and 6.

The sub-basin 2 is a high mountain sub-basin composed of mostly forest (63.2%), with a wide area of shrubs and grass cover (35.1%), it contains a reservoir at the outlet (La Baells), and has the steepest slope (both maximum and mean).

Sub-basin 6, in contrast, is mainly composed of non-irrigated cultivated land (43.8 %), with less grass and shrub land (25.5 %) and forest (24.9 %), it has the shallowest slope (both maximum and mean).
Table 3. Characteristics of each sub-watershed for the sensitivity analysis of the InVEST erosion control model (from 1 to 7)

<table>
<thead>
<tr>
<th>% LULC</th>
<th>Sub-basin</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urban land (0)</td>
<td>Mean</td>
<td>24.0</td>
<td>0.5</td>
<td>0.4</td>
<td>0.3</td>
<td>2.4</td>
<td>5.7</td>
<td>2.7</td>
</tr>
<tr>
<td>Non-irrigated</td>
<td>Mean</td>
<td>12.0</td>
<td>1.2</td>
<td>1.5</td>
<td>22.1</td>
<td>28.7</td>
<td>43.8</td>
<td>25.6</td>
</tr>
<tr>
<td>Irrigated land</td>
<td>Mean</td>
<td>1.5</td>
<td>0.0</td>
<td>0.1</td>
<td>0.0</td>
<td>0.3</td>
<td>0.1</td>
<td>0.3</td>
</tr>
<tr>
<td>Shrub &amp; grass</td>
<td>Mean</td>
<td>34.6</td>
<td>35.1</td>
<td>40.2</td>
<td>14.4</td>
<td>38.5</td>
<td>25.5</td>
<td>26.9</td>
</tr>
<tr>
<td>Forest land (4)</td>
<td>Mean</td>
<td>27.9</td>
<td>63.2</td>
<td>57.8</td>
<td>63.3</td>
<td>30.2</td>
<td>25.0</td>
<td>44.5</td>
</tr>
</tbody>
</table>

3.2.3. Study area for testing in-stream processes in the InVEST nutrient retention model

For studying the role of in-stream retention in the InVEST nutrient retention model, it was not possible to divide the river basin into sub-watersheds, so it was used in its entirety.

This is because the InVEST model uses a threshold flow accumulation value as an input, which determines the number of upstream cells that must flow into a cell before it is considered part of a stream. If the threshold flow accumulation doesn’t yield a river network in a sub-watershed, the model is not able to give a result in this sub-watershed.
For the tests made in this section, the threshold flow accumulation value was adjusted so that it gives an only 1-pixel wide river network, at the end of the watershed (Figure 14).

Thus, if any split of the watershed were made to create sub-watersheds, any of them would contain a river network within them, where the model could not give a result in these hypothetical sub-watershed division (except the last one). This lack of results at the sub-watershed is the reason for working at the watershed scale.

Figure 14. River network, using a threshold flow accumulation that produces a river network consistent with the map provided by the Catalan Water Agency – ACA (a), and using a threshold flow accumulation that produces a 1-pixel wide river network (b). The latest is used in the in-stream methodology performed.
The data used as inputs were the same as those used in a previous study of the calibration and application of the InVEST model in the Llobregat river basin (Terrado et al., 2014) in order to create a base scenario (one for each pollutant studied: nitrogen and phosphorus).

Both base scenarios were calibrated with the loading information at the basin outlet from Aguilera et al. (2013). In the work cited, nitrate and phosphate daily loads were calculated by the software Load Estimator (LOADEST) (Runkel et al., 2004) with the concentration and daily discharge data obtained from 24 locations monitored by the Catalan Water Agency (ACA).

The sampling data for every station were collected within a period ranging from 2000 to 2006. Those loads were averaged for each of the 7 years and converted to kg TN/year or kg TP/year (depending on the nutrient chosen to run the model) to compare them with those from the InVEST nutrient retention model (that are in the same units) using the ratios N-NO3/TN = 0.75, and P-PO4/TP = 0.48 (Ludwig et al., 2009).

The outlet of the watershed studied in InVEST was between two of the 24 stations monitored (Sant Joan Despi and Barcelona), therefore the value for comparing the base scenarios was interpolated between both values (Figure 15).

This interpolated value for TN is 1,457,603 kg/ws year (Aguilera et al., 2012; 2013) which is close to the one reported by Ludwig et al. (2009) 1,359,000 kg TN/ws year, and for TP is 263,362,96 kg/ws year (Aguilera et al., 2012; 2013).
Figure 15. The 24 stations monitored in the Llobregat river basin, and the outlet of the basin is between two of these ones (Sant Joan Despí and Barcelona)

The LULC map use in this part of the work is an 8-land use map instead of the 5-land use map used for the sensitivity analyses. The 8-land use map was made by adding 3 ‘land uses’ to the 5 LULC existing (Figure 16).

Those 3 new ‘land uses’ (dam, reservoir and river) are not land uses per se because they represent the aquatic part of the watershed, but they work as land uses in the methodology created for studying the influence of the in-stream processes explained in Chapter 4.2 of the present work.
The characteristics of the watershed studied are explained in Section 3.2 of the present work. One of the most relevant aspect to take into account is the huge difference between the terrestrial surface with respect to the aquatic surface in this sub-watershed (96% terrestrial, 4% aquatic).

![Pie chart showing 96% terrestrial and 4% aquatic](image)

*Figure 17. Proportion of terrestrial (96% of the pixels are terrestrial) and aquatic Surface in the Llobregat river basin (4% of the pixels are aquatic)*
Chapter 4

4. METHODOLOGY

4.1. METHODOLOGY: SENSITIVITY ANALYSIS USED WITH THE InVEST MODEL IN THE LLOBREGAT RIVER BASIN

As explained in the Chapters before, InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) model is a specific tool to estimate the amount of ecosystem services and their economic value.

InVEST has been successfully applied to improve the knowledge about managing ecosystems and understanding the behaviour of ecosystems response changing the conditions in different scenarios.

Two of the InVEST models, which are the water provisioning model and the erosion control model, were applied to the basin of the second longest river in Catalonia (NE Spain), the Llobregat River as an example of a Mediterranean river basin.
Once those InVEST models have been tailored to the Mediterranean basin (Terrado et al., 2014) a number of possible methodologies can be employed for the calculation of the inputs sensitivity. There is one particular screening method for sensitivity analysis that could be considered the state of the art and particularly efficient computationally: The Morris method (Morris, 1991).

The typical application for this tool is where many parameters and available resources do not allow specifying probability density functions for a full Monte Carlo analysis.

Moreover, if the Morris method indicates that the parameters are independent, it is unnecessary to use methods that take into account the interaction between multiple parameters such as the Monte Carlo analysis.

The Morris method was preferred for its computational efficiency and effectiveness in screening a small subset of important factors among a much larger set. In this regard, it is worth noting that each experiment performed in this work (350 model runs) amounted to about 100 CPU hours. Other SA methods, which require more than 5000 models runs demand computation times that are very difficult to satisfy in practice.

As with all sensitivity analysis methods, the goal of the Morris method is to determine the relative influence of each input variable or factor \((x_i)\) on the output or response \((y)\) through a series of model runs.

To achieve this, the method determines the mean \((\mu)\) and standard deviation \((\sigma)\) of \(\frac{\partial y}{\partial x_i}\) at several points in the sample space employing an OAT calculation scheme that fixes at each step all input variables except the one whose effect is being calculated.
The original Morris approach could be extended by applying the 'mean of absolute values' ($\mu^*$) defined by Campolongo et al. (2007), to avoid the effect of cancellation of terms of opposite signs but similar absolute values.

The methodology developed for applying the Morris method with the two InVEST models cited above consists in the following five steps:

1. Examine the calibrated InVEST model to select the input variables that are likely to be affected by global change and define the ranges within which they will be modified during each test.

2. Implement the extended Morris approach and calculate $\frac{\partial y}{\partial x_i}$ employing the OAT type calculation scheme.

3. Obtain the set of combinations of input variables.

4. Run the calibrated InVEST model for each combination of input variables of factors, obtaining a set of model responses.

5. Calculate $\mu^*$ and $\sigma$ for each sub-watershed and for the whole basin.

The InVEST source code is written in Python, and had to be modified in order to run the experiments that correspond to each input combination. The model runs for different combinations of input variables and the calculation of the Morris indices were executed using code written in the FORTRAN programming language.

In the first stage of the study, a new calibration phase of the model was developed in order to ensure that the correction brought to the two InVEST models used worked correctly once the changes in the code were made.

This calibration was performed to verify that the results were comparable to those previously obtained by (Terrado et al., 2014).
4.1.1. The Morris methodology used with the InVEST water provisioning model applied in the Llobregat River basin

As indicated above, the first step of the methodology is to examine the calibrated InVEST model and to select the inputs being studied and also the outputs for which the sensitivity is going to be calculated.

The InVEST water provisioning model generates various outputs at each step. As stated in Chapter 2, we have studied the first two steps in terms of sensitivity analysis.

For those steps, the outputs obtained were water yield (\textit{wyield\_vol}, \textit{wyield\_ha} and \textit{wyield\_mn}), and water scarcity, in two ways (\textit{rsupply\_vol} and \textit{rsupply\_mn}).

With respect to water yield outputs, \textit{wyield\_vol} is the total water yield per sub-watershed, namely the approximate absolute annual water yield across the landscape, calculated as the difference between precipitation and actual evapotranspiration on each land parcel, given in m$^3$; \textit{wyield\_ha} is the water yield volume per hectare per sub-watershed, given in m$^3$/ha and \textit{wyield\_mn} is the mean water yield per sub-watershed, given in mm.

With regard to water scarcity outputs, \textit{rsupply\_vol} is the realized water supply volume (water yield – consumption) for each sub-watershed, given in m$^3$ and \textit{rsupply\_mn} is the mean realized water supply per sub-watershed (water yield – consumption), given in m$^3$/ha.
Sensitivity analysis was carried out only for the `wyield_vol` model output, since the other outputs could be calculated from `wyield_vol` (multiplying by a scalar for `wyield_ha` and `wyield_mn`, or/and subtracting a scalar for `rsupply_vol` and `rsupply_mn`), and consequently the sensitivity analysis results would yield identical results.

Having studied the InVEST model, there are four inputs that may be subject to variation for sensitivity analysis: precipitation, evapotranspiration, the Z coefficient, and plant evaporation coefficient for each LULC class.

The first three inputs are non-varied land use parameters, however, the last one vary when changes in land use are made. The plant evapotranspiration coefficient for each LULC class parameter is used to obtain potential evapotranspiration by using plant energy/transpiration characteristics to modify the reference evapotranspiration, which is based on alfalfa.

In this work, this parameter was not selected for the sensitivity analysis because it requires a double point of view: the sensitivity analysis of the parameter within a range of variation but also the sensitivity analysis of the parameter with changes in the land uses.

After the model inputs have been selected, it is necessary to specify their range of variation. The precipitation and evapotranspiration raster maps had to be rescaled by a coefficient (precipitation coefficient and evapotranspiration coefficient).
These variations in the ranges have been explained in more detail in Chapter 3 and are shown in the Table 1 in this Chapter.

Before using different combinations of preselected inputs for sensitivity analysis (Z, precipitation, and evapotranspiration coefficients), several changes were made to InVEST, in order to adapt the simulation tool to the subsequent sensitivity analysis.

These alterations are basically focused on modifying the Python script so that InVEST accepts a new input table along with the three preselected input variables: Z, precipitation and evapotranspiration coefficients), and produces different output tables for each selected input parameter combination.

These changes have resulted in loops for each group of inputs at every simulation run. The algorithm that combines the different input variables was implemented in FORTRAN code. This algorithm generates the Morris' trajectories (Morris, 1991) by taking into account the boundaries of each factor.

Once these modifications were performed, the model was run for each of the preselected input combinations. Thereafter, the InVEST water provisioning model was run for each combination of parameters. The Morris indices $\mu^*$ and $\sigma$ of each model response to each input for each sub-watershed were subsequently calculated, analysed and compared.
4.1.2. The Morris method used with the InVEST erosion control model applied in the Llobregat River basin

As stated earlier, it is mandatory to first examine the calibrated InVEST model before applying SA. The model was calibrated by Terrado et al. (2014) at La Baells reservoir (outlet of sub-basin 2) using temporal changes in the reservoir’s bathymetry (Catari et al., 2009).

Results of the model were then validated with data reported by Liquete et al. (2009). Despite only accounting for the sheet wash process in the sediment budget (gully or stream bank erosion are ignored), a reasonable estimate of sediment yield was obtained at the basin scale with physical parameters in reasonable ranges (relative error of 7%; Table 3; Liquete et al., 2009).

However, at sub-basin scale in the Upper Llobregat, above La Baells reservoir, the relative error was higher (25%—see Table 3; Catari et al., 2009). This could be due to additional sediment sources in this particular area that are not accounted for in the model (e.g. bank erosion, gullies), as suggested by Catari et al. (2009).

The large uncertainties about the relative contribution from sediment sources (sheetwash vs. gullies and bank erosion) could affect the validation performed by Terrado et al. (2014).

However, these uncertainties have a limited impact on the sensitivity analysis presented here since results are interpreted as representing the contribution from sheetwash erosion only.
All the calculations are made at a pixel resolution of 200 × 200 m to capture the spatial heterogeneity of key driving factors such as soil type, rainfall and vegetation type.

However, the model simulates hydrologic processes that are best interpreted at the sub-basin or basin scale, so the pixel values are aggregated (summed and/or averaged) to match these scales for result interpretation and model validation.

After the examination of the calibrated InVEST model, the selection of the outputs and inputs for the sensitivity analysis is made. With respect to the outputs, the InVEST erosion control model generates three outputs:

1) total sediment exported,
2) fraction of total sediment retained which contributes to the higher water quality benefit, and
3) fraction of total sediment retained to the avoided reservoir sedimentation benefit. The sensitivity analysis of the inputs was made three times, one for each of these outputs.

Using the equations that describe this model in InVEST, the inputs inspected in the sensitivity analysis included: rainfall erosivity (R); soil erodibility (K); and land use/land cover (LULC), with five LU categories (urban land, non-irrigated cultivated land, irrigated cultivated land, grass and shrub land, and forest land).

Three inputs were linked to land use type: cover management factor (C); the support practice factor (P); and a sediment retention value that
identifies the capacity of vegetation to retain sediment as a percentage of the amount of sediment flowing into a cell from upslope.

There are two additional inputs that could be included in the sensitivity analysis: the threshold flow accumulation and slope threshold that both depend on the digital elevation model (DEM).

Because errors in the DEM were not a focus of this analysis, the values from a former study were used (Terrado et al. 2014).

The threshold flow accumulation, set to 1000, produced outputs consistent with the stream network map provided by the Catalan Water Agency (ACA) and is not included in the analysis either.

The ranges of variation for all the inputs were obtained from the literature (Wischmeier and Smith, 1978; US EPA, 2009; Terrado et al., 2014). The values $\frac{\partial y}{\partial x_i}$ were normalized by the response’s standard deviations to avoid scaling artifacts (Patelli et al., 2010). These ranges of variation of the inputs selected for sensitivity analysis are explained in detail in Chapter 3 and summarized in Table 2.

The sensitivity analysis of the model was then performed following the Morris method. As already explained in detail, the Morris method (Morris, 1991) is a one-at-a-time (OAT) sensitivity analysis and aims at isolating the influential parameters from a large number of input variables or factors.

This method has several advantages over other methods. It is easy to understand, does not depend on assumptions about the model, is
computationally inexpensive, the parametric space is covered efficiently, and it shows input interactions and non-linear effects. However, this method cannot quantify the contribution of a parameter to the variability of the output in a highly non-linear model (Wu et al., 2013).

The sensitivity analysis parameters $\mu^*$ and $\sigma$ were calculated for each factor with respect to the 3 final outputs:

1) total sediment exported,

2) total sediment retained for avoided reservoir sedimentation benefit, and

3) total sediment retained for higher water quality benefit.

These parameters were calculated at both sub-basin and basin scales. A high value of $\sigma$ suggests the existence of nonlinear effects and/or interaction with other input factors, while a low $\sigma$ indicates independence (Morris, 1991).

The value of $\mu^*$ is an indication of the importance of the factor, with higher values attributed to factors to which the output of the model is most sensitive.

The relevant parameters obtained for total sediment exported were then correlated with the characteristics of each sub-basin and with the whole river basin using the Spearman’s rank correlation matrix due to the small sample size and the presence of outliers in the data, invalidating the assumption of normal distribution. To cross-check the results, we also computed the Pearson’s $r$ correlation coefficients, which proved to be similar.
4.2. METHODOLOGY: TESTING THE RELEVANCE OF IN-STREAM PROCESSES FOR InVEST NUTRIENT RETENTION MODEL IN THE LLOBREGAT RIVER BASIN

The InVEST nutrient retention model (Tallis et al., 2013) estimates the water purification service for two pollutants (total nitrogen, TN, and total phosphorus, TP), i.e. the quantity of pollutants retained from the landscape and the benefits from this service.

InVEST models only take into account the terrestrial processes. In particular, in this model, InVEST assumes that pollutants arriving at the stream network of the watershed from the surrounding area are simply transported to the outlet without any in-stream processing or retention.

However, recent knowledge suggests that stream networks play an important role in material export from watersheds, specifically in the Llobregat River Basin (Aguilera et al., 2012; 2013). In addition, the InVEST User’s Manual suggests that ‘the user should consider the likely impact of in-stream processes in any calibration work as this model does not include in-stream processes’ (Tallis et al., 2013).

Therefore, to test if this new knowledge is worth being included in InVEST is proposed here. Developing a methodology for testing the role of the in-stream processes should be the first step. This methodology should be devised in order to test the results by including in-stream processes but without the changing InVEST source code.

The strategy for all the tests was to add the river and reservoirs as new ‘land uses’ in the land use map, and to apply a retention coefficient for these new
land uses. This way, the 5-land use map input was changed to an 8-land use map by adding 3 ‘land uses’: dam, reservoir and river. In the subsequent tests, we assume the same retention for the three new ‘land uses’.

As noted above, this strategy has the advantage of avoiding changing the model’s source code, reducing time and possible errors from modifying the original code.

*Figure 18. The 5-land use map input (a) used in the sensitivity analysis, and the 8-land use map input (b) used in the in-stream tests by adding 3 ‘land uses’: dam, reservoir and river.*

One challenge in implementing this strategy was that the code does not allow any kind of retention on the pixels in the river network. A river network with only one pixel, at the outlet of the watershed, is needed to successfully implement the proposed strategy. To this aim, it was necessary to make several tests in order to determine the threshold flow accumulation value which gives a 1-pixel river network.
The threshold flow accumulation value is an input introduced in the model, which determines the number of upstream cells that must flow into a cell before it is considered part of a stream. This threshold expresses where retention stops and the remaining pollutant will be exported to the stream (Tallis et al., 2013).

The strategy adopted has to be applied at watershed scale because applying it at sub-watershed scale, a 1-pixel river network does not yield a river network in the sub-watersheds upstream the basin outlet, and in this case the model is not able to give reliable results for those sub-watersheds. Conversely, if the threshold flow accumulation yields a river network with more than 1 pixel wide in order to have results in all of the sub-watersheds, the new ‘land uses’ pixels would not work as land use, which is required in this work.

The data used as inputs were the same of those used in a previous study of the application of the InVEST models in the Llobregat river basin (Terrado et al., 2014). All scenarios were calibrated with the loading information at the basin outlet for both nitrogen and phosphorus with a relative error within ± 2%. In this case, the relative error was calculated as the ratio of the absolute error to the measured value. The absolute error is the difference between the measured value and the modeled result. The relative error gives an indication of how good a measurement is relative to the size of the thing being measured.

The loading information at the basin outlet for both nutrients (nitrogen and phosphorus) was obtained from Aguilera et al. (2013). In this work, nitrate and phosphate daily loads were calculated by the software Load Estimator.
(LOADEST) (Runkel et al., 2004) with the concentration and daily discharge data obtained from 24 locations monitored by the Catalan Water Agency (ACA).

The sampling data for every station were collected within a period ranging from 2000 to 2006. Those loads were averaged for each of the 7 years and converted to kg TN/year or kg TP/year (depending on the nutrient chosen to run the model) to compare them with those from the InVEST nutrient retention model (that are in the same units) using the ratios N-NO$_3$/TN = 0.75, and P-PO$_4$/TP = 0.48 (Ludwig et al., 2009).

The outlet of the watershed (ws) studied in InVEST is allocated between two of the 24 stations monitored (Sant Joan Despi and Barcelona), therefore the value for comparing the scenarios was interpolated between the values of both stations monitored (points 23 and 24, Figure 19; more detailed in Figure 15 Section 3.2.3.). This interpolated value for TN is 1.457.603 kg/ws year (Aguilera et al., 2012; 2013) which is within close to the one reported by Ludwig et al. (2009) 1.359.000 kg TN/ws year, and for TP is 263.362,96 kg/ws year (Aguilera et al., 2012; 2013).

During the calibration process, the model parameter that was adjusted was the per-pixel vegetation filtering value for each terrestrial LULC class (or retention coefficient). This parameter describes the capacity of vegetation to retain TN and TP as a percentage of the amount of nutrient flowing into a cell from upslope.

Note that nutrient filtration efficiency was assumed to be constant when working with annual averages. However, at the event scale, this parameter would depend on the load, meaning that with higher loads the filtration efficiency of
vegetation would be much lower than with events of smaller size. Since the parameter is related to vegetation, this parameter is set to 0% for urban LULC. The calibration was made for the base scenario of each nutrient that the model can estimate (total nitrogen and total phosphorus).

![Figure 19. The 24 stations monitored in the Llobregat river basin (names and localizations)](image)

Since the code in the model allowed integer values for the nutrient filtration efficiency parameter but the previous tests shown that this value might be between 0 and 1% for rivers, the code was changed to handle a real-valued parameter.

The procedure followed was the same for both nutrients estimated by the InVEST nutrient retention model: total nitrogen (TN) and total phosphorus (TP). The procedure was to first create a base scenario (calibrated with the monitoring export value), and then create different scenarios by changing the
Nitrogen

With respect to the TN scenarios, calibration of the amount exported at the outlet basin is not possible by solely changing the nutrient filtration efficiency parameter for the terrestrial LULC. In order to calibrate the base scenario with 0% retention coefficient in water LULC pixels, it is necessary to adjust loads over terrestrial LULC pixels. This option was discarded because introducing changes in other parameters (such as the load parameter) might add more uncertainty. In addition, the target of this analysis is to study the repercussion of retention on the landscape parameter by introducing river retention.

Therefore, calibration was only made with the retention parameters. Thus, base scenario was calibrated with a retention coefficient of 0.5% for aquatic LULC (and varying for calibration the terrestrial LULC retention parameter), whilst the others scenarios were calibrated after increasing or decreasing this retention value for aquatic LULC; in other words, the model calibration was done with a single factor modifying all the terrestrial LULC retention parameters uniformly.

The scenarios performed and calibrated as it was explained above were the following:

- **TN_BASE**: base scenario with a retention coefficient of 0.5% for aquatic LULC and a vegetation filtering value of 25% for cultivated land areas (both irrigated and non-irrigated) and 50% for natural land areas (grass/shrub and forest). Those values are consistent with the values recommended by the InVEST 2.6.0 User’s Guide (Tallis et al., 2013):
'high values (60 to 80) may be assigned to all natural vegetation types and intermediary value also may be assigned to features such as contour buffers'. The result of this scenario has a relative error with respect to the reference value of -1.6%.

- **TN+1**: In this scenario, the retention coefficient of the aquatic LULC pixels was 0.4%, which means a 0.1% decrease compared to the base scenario (TN_BASE). The calibration of the result was obtained with an increase in the terrestrial LULC retention coefficient of 30% (relative error of 1.6%).

- **TN+2**: For this scenario, the retention coefficient for the aquatic LULC pixels was set to 0.3%, which corresponds to a decrease of 0.2% with respect to the base scenario (TN_BASE). It was necessary to increase the terrestrial LULC retention coefficient by 85% for the new calibration (relative error of 1.0%).

- **TN-1**: The retention coefficient was increased by 0.1% with respect to the base scenario (TN_BASE), or equivalently, the retention coefficient was set to 0.6% on the aquatic LULC pixels. The terrestrial LULC retention coefficient was decreased by 30% for the new calibration (relative error value of -0.2%).

- **TN-2**: The retention coefficient for the aquatic LULC pixels was set to 0.7%, i.e. an increase of 0.2% with respect to the base scenario
(TN_BASE). For the new calibration, the terrestrial LULC retention coefficients were decreased by 50% (relative error of 0.3%).

- **TN-3**: The retention coefficient was set to 0.8% for the aquatic LULC pixels, what means an increase of 0.3% with respect to the scenario base (TN_BASE). It is necessary to decrease the terrestrial LULC retention coefficient by 65% for the new calibration to be within a relative error of 1.2%.

The calibration of other scenarios (less than 0.3% or more than 0.8% in retention coefficient for aquatic LULC pixels) against the monitoring export value was impossible to achieve within ± 2% in relative error. Thus, the scenarios studied for Total Nitrogen exported and retained were the following:

<table>
<thead>
<tr>
<th>Nitrogen Scenarios</th>
<th>In-stream: Retention coefficient (%)</th>
<th>Land: Increase or decrease (-) in Retention coefficient with respect to Base Scenario (%)</th>
<th>Error in calibration (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TN+2</td>
<td>0.3</td>
<td>85</td>
<td>1.0</td>
</tr>
<tr>
<td>TN+1</td>
<td>0.4</td>
<td>30</td>
<td>1.6</td>
</tr>
<tr>
<td>TN_BASE</td>
<td>0.5</td>
<td>0</td>
<td>-1.6</td>
</tr>
<tr>
<td>TN-1</td>
<td>0.6</td>
<td>-30</td>
<td>-0.2</td>
</tr>
<tr>
<td>TN-2</td>
<td>0.7</td>
<td>-50</td>
<td>0.3</td>
</tr>
<tr>
<td>TN-3</td>
<td>0.8</td>
<td>-65</td>
<td>1.2</td>
</tr>
</tbody>
</table>

*Table 4. The 6 scenarios performed and calibrated for Total Nitrogen*
Phosphorus

Regarding TP scenarios, the amount exported at the outlet basin was calibrated with a constant retention coefficient of 0% for water LULC while varying the retention coefficient of the terrestrial LULC away from the base scenario.

Each successive scenario differed from the previous one by discrete increases of 0.1% in retention coefficient for water LULC, and by modifying the retention coefficient of the terrestrial LULC. The resulting scenarios, with a relative error within ± 2%, are described below:

- **TP BASE**: the base scenario had a 0% retention coefficient for aquatic LULC and a vegetation filtering value of 45% for cultivated land (both irrigated and non-irrigated), and 90% for natural land areas (grass/shrub and forest). Those values correspond to those recommended in the InVEST 2.6.0 User's Guide (Tallis et al., 2013): ‘high values (60 to 80) may be assigned to all natural vegetation types and intermediary value also may be assigned to features such as contour buffers’. The result of this scenario has a relative error with respect to the reference value of -0.9%.

- **TP-1**: For this scenario, 0.1% was the value used in the retention coefficient of the aquatic LULC pixels, which is an increase of 0.1% with respect to the base scenario (TN_BASE). It was necessary a decrease the terrestrial LULC retention coefficient by of 35% for the new calibration (relative error of 1.5%).
• TP-2: Increasing the retention coefficient of the aquatic LULC pixels by 0.1% with respect to the scenario TP-1, or equivalently, using a retention coefficient of 0.2% for those pixels. The terrestrial LULC retention coefficient was decreased by 55% for the new calibration (relative error of -0.5%).

• TP-3: Increasing the retention coefficient of the aquatic LULC pixels by 0.1% as in the previous scenario, which translates to a retention coefficient of 0.3%. The terrestrial LULC retention coefficients were decreased by 70% for the new calibration (relative error of -1.0%).

• TP-4: The retention coefficient for the aquatic LULC pixels was 0.1% higher than in TP-3, which corresponds a 0.4%. A decrease of 82% in the terrestrial LULC retention coefficient was necessary for the new calibration to be within a relative error of 1.2%.

• TP-5: When the retention coefficient of the aquatic LULC pixels is 0.5%, i.e. an increasing of 0.1% with respect to the TP-4 scenario. The terrestrial LULC retention coefficient was decreased by 88% (relative error of -1.1%).

• TP-6: The retention coefficient of the aquatic LULC pixels used for this scenario was 0.6%, i.e. an increasing in 0.1% with respect to the before scenario. For the calibration of this scenario the terrestrial LULC retention coefficients were decreased in 94% (relative error of 0.8%).
- **TP-7**: In this scenario, the retention coefficients of the aquatic LULC pixels used were 0.7% (increasing 0.1% the retention coefficients of the scenario above). The terrestrial LULC retention coefficients were decreased in 98% for the calibration of this scenario (relative error of 1.5%).

<table>
<thead>
<tr>
<th>Phosphorus Scenarios</th>
<th>In-stream: Retention coefficients (%)</th>
<th>Land: Decrease (-) in Retention coefficients with respect to Base Scenario (%)</th>
<th>Error in calibration (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP_BASE</td>
<td>0.0</td>
<td>0</td>
<td>0.9</td>
</tr>
<tr>
<td>TP-1</td>
<td>0.1</td>
<td>-35</td>
<td>1.5</td>
</tr>
<tr>
<td>TP-2</td>
<td>0.2</td>
<td>-55</td>
<td>-0.5</td>
</tr>
<tr>
<td>TP-3</td>
<td>0.3</td>
<td>-70</td>
<td>-1.0</td>
</tr>
<tr>
<td>TP-4</td>
<td>0.4</td>
<td>-82</td>
<td>1.2</td>
</tr>
<tr>
<td>TP-5</td>
<td>0.5</td>
<td>-88</td>
<td>-1.1</td>
</tr>
<tr>
<td>TP-6</td>
<td>0.6</td>
<td>-94</td>
<td>0.8</td>
</tr>
<tr>
<td>TP-7</td>
<td>0.7</td>
<td>-98</td>
<td>1.5</td>
</tr>
</tbody>
</table>

*Table 5. The 8 scenarios performed and calibrated for Total Phosphorus*

In summary, the results of exported nutrient were estimated from the InVEST model and were used to build different scenarios, for both TN and TP, as described above.

In addition, the model also estimates the nutrient retention of the benefits chosen of a service for all these scenarios, as it was explained in Chapter 2. However, the nutrient retention needed for the present study was the real one.
The real retention is the one calculated before the extraction of the total allowed annual load for the pollutant of interest. The rationale for using the real retention of the nutrient of interest instead of the retention of the benefit selected, is because the purpose of the study is the real share of the river and the terrestrial in the retention of both nutrients, and not the relative to the benefit.
Chapter 5

5. SENSITIVITY ANALYSIS FOR THE INVEST WATER PROVISIONING MODEL IN THE LLOBREGAT RIVER BASIN

The present Chapter is entirely based on the published paper Sánchez-Canales et al. (2012).

5.1. INTRODUCTION

In recent years, the need to understand the behaviour of many types of very complex ecosystem processes has led to an extraordinary increase in the development of suitable mathematical models, which are of capital importance in order to estimate how an ecosystem behaves and to predict its behaviour in the future.

These models are typically computationally expensive because of the large number of variables, parameters and mathematical equations that constitute
them. The problem is that not all processes can accurately be described with simplistic equations, and many of them are modelled as approximations, when not completely avoided altogether.

Moreover, not all model input parameters are easily measured and often need to be estimated using current, limited knowledge about their underlying processes. These uncertainties in the input parameters lead to uncertainties in the model outputs that need to be quantified in order to gain a better confidence about model response when it is used during the decision making process.

Sensitivity analysis (SA) techniques have been widely used to predict and analyse how the model output (response) depends on the variations in the input parameters (factors). Many applications of SA techniques in ecosystem science can be found in recent papers (Campolongo and Saltelli 1997; Confalonieri et al., 2010a; 2010b; Yang, 2011).

It is particularly useful for complex simulation tools like regional hydrological models that require a large number of factors with uncertainty where it is crucial to identify either the most important or the least relevant parameters.

A large number of previous studies have shown that land-use influences the hydrologic regime and water quality of streams draining watersheds. Extensive literature exists concerning hydrologic response associated with land-use changes such as forest harvesting, conversions of vegetation on the landscape, and draining of agricultural lands (Bosch and Hewlett, 1982; Robinson and Rycroft, 1999).

Water provisioning is an ecosystem service that contributes to the well-being of society, on one hand by satisfying human needs for freshwater provisioning
services for irrigation, domestic water, and power generation, among others, and on the other hand, it contributes to human well-being through recreation, scenic quality, maintenance of fisheries and biodiversity, and ecosystem function.

The InVEST water provisioning model (version 2.2.2; Tallis et al. 2011) informs about the total amount of water available in a basin. The amount of water provisioned from each cell in the landscape (water yield) is calculated as the annual amount of rainfall that does not evapotranspirate, determined by the cell vegetation characteristics (Canadell et al., 1996). Water demands for consumptive uses other than those evaluated are removed from the total yield before assessing the benefit.

The Mediterranean region has been globally identified as one of the most vulnerable to global change (Schröter et al., 2005). Different potential impacts are projected, including increased temperatures and reduced vegetation. The hydrological cycle in the Mediterranean areas will intensify through increases in temperature, rainfall concentration in shorter periods of the year, and more extended droughts (Hisdal et al., 2001).

Associated human impacts through changes in ecosystem services could include drinking water shortages, increased risk of forest fires, shifts in the distribution of species, and agricultural losses, among others (Schröter et al., 2005).

Taking into account all of this, the InVEST water provisioning model was applied to a stakeholder-defined scenario of LU/LC change in a Mediterranean River basin (Llobregat River, Catalonia, Spain). After model calibration for this
basin, a sensitivity analysis was carried out in order to determine the variability of the model response for three of its main coefficients: Z (seasonal precipitation distribution), precipitation (annual) and evapotranspiration (annual).

The SA technique used here is a One-At-a-Time (OAT) screening method known as the Morris method, applied over each one of the 154 sub-watersheds that constitute the Llobregat River basin. As a result, this method provides sensitivity indices for each one of the sub-watersheds, which are mapped to study how they are spatially distributed.

Chapter 2 gives a detailed explanation of the concept of ecosystem services, of the InVEST water provisioning model, and the sensitivity analysis techniques, with an emphasis on the Morris method. In Chapter 3, the case study is thoroughly explained. Chapter 4 presents an extensive explanation of the methodology used for the purpose of the present research.

5.2. RESULTS AND DISCUSSION

As a result of applying the Morris method in the InVEST water provisioning model in the Llobregat river basin, the SA indices $\mu^*$ and $\sigma$ for each sub-watershed and for each factor have been obtained, that correspond to the model response and the three factors mentioned above.

In Figure 20, the SA index $\mu^*$ obtained for the total water yield per sub-watershed with regards to the three selected input parameters is mapped. In these maps, the more intense the red colour of the sub-watershed is, the highest $\mu^*$ is, and as a result, this input parameter has a greater influence on the output.
Therefore, in terms of $\mu^*$, the greatest impact input parameter is precipitation and the least are the Z coefficient and evapotranspiration. Regarding the Z coefficient, the observed sensitivity indices in the range of study [7, 9] indicate that this factor does not influence water yield in this interval.

For precipitation, the results indicate that the most sensitive sub-watersheds are found further north and southeast. In the northern part of the basin, the precipitation is high (between 800 and 1000 mm) and evapotranspiration is quite low (around 400 mm), so the influence and importance of evapotranspiration are lower, though the model is very sensitive to precipitation and large changes in this input cause significant changes in the total water yield per sub-watershed.

In the south-eastern part of the basin, evapotranspiration is quite high (between 800 and 1000 mm) and rainfall is lower (around 600 mm), so variations in precipitation can cause significant changes in the total water yield per sub-watershed.

The influence of evapotranspiration over the output is quite low, and irrelevant for sub-watersheds where precipitation has a great impact on the output (north and southeast sub-watersheds). This relationship observed between the behaviour of precipitation and evapotranspiration may also be explained by the standard deviation ($\sigma$).
Figure 20. For total water yield per sub-watershed, the sensitivity index $\mu^*$ (mean of absolute values) for the Z coefficient (a), for precipitation (b), and for evapotranspiration (c)

In terms of standard deviation, the results obtained for total water yield per sub-watershed for each of the three input parameters considered are represented in the maps shown in Figure 21.

The higher the standard deviation is, the greater the coupling between variables. In this case there is no coupling of the Z coefficient with the other two variables. However, we observed that a coupling exists between evapotranspiration and precipitation, with their correlation being stronger in the central region of the watershed.

In this region, precipitation and evapotranspiration have similar orders of magnitude and small variations in either input parameters have a significant influence on the output.
Furthermore, it can be derived from the $\sigma$ vs. $\mu^*$ graph (Figure 22, Figure 23) that total water yield per sub-watershed is strongly influenced by precipitation and, to a lesser extent, by evapotranspiration. In this context, the effect of the $Z$ coefficient seems to be entirely negligible.

The high values of the standard deviation for evapotranspiration and precipitation, especially in the sub-watersheds located in the central part of the Llobregat River basin, suggest a relatively strong importance of the interactions between these two factors that should be the subject for further research.
5.3. CONCLUSIONS

InVEST is a spatially explicit simulation tool consisting of a suite of models, some of which using land use and land cover patterns to estimate the levels and economic values of ecosystem services. The InVEST water provisioning model runs in a gridded map at an annual time step, and results can be reported in either biophysical or monetary terms, depending on the needs and the availability of information.

The biophysical components of the terrestrial InVEST models calculate the relative contribution of the different parts of the landscape to the provision of
each service. Thus, for the water provisioning service, the amount of water provisioned from each cell in the landscape (water yield) is calculated as the annual amount of rainfall that does not evapotranspirate, determined by the cell’s vegetation characteristics (Canadell et al., 1996).

Water demands for consumptive uses other than those evaluated in this model are removed from the total yield before assessing the benefit.

In the present work, once an InVEST model had been adapted to assess the water provisioning service in the Llobregat River basin, a method for sensitivity analysis was applied to identify the most sensitive input parameters for the water provisioning service that require more precision and need to be monitored.

The results obtained indicate that the Z coefficient in the geographical area is not a sensitive factor for the range of values studied. However, precipitation is very sensitive, especially in the more humid areas of the watershed. Generally, evapotranspiration is less sensitive than precipitation but there are some sub-watersheds where its sensitivity is significant.

In conclusion, in order to apply the InVEST water provisioning model on watersheds similar to that under study, it is strongly advised to follow the next steps:

1) use the precipitation map that best suits the region under consideration, due to the fact that this input is crucial for the purpose of sensitivity analysis. Note that poor data for this input (that yield a poor fit) would imply non-realistic results;
2) fit precisely the evapotranspiration map, as the interaction of this factor with precipitation could have a considerable importance in some particular areas;

3) select a range of values for the Z coefficient so that it is consistent with the characteristics in the distribution of rain for the region of interest and choose any value within this range, as it will give the same output.

The SA methodology implemented here could be extended to all other ecosystem service modules of InVEST. Furthermore, this methodology can be applied in all the basins of the world.
6. SENSITIVITY ANALYSIS FOR THE INVEST EROSION CONTROL MODEL IN THE LLOBREGAT RIVER BASIN

The present Chapter is entirely based on the published paper Sánchez-Canales et al. (2015).

6.1. INTRODUCTION

Sediment dynamics in river basins are severely influenced by changes in rainfall and land-use patterns (Walling, 2008). Recent studies suggest that this influence is particularly evident in scenarios of environmental land use conflicts, where actual land uses deviate from natural uses determined by soil characteristics (Pacheco et al., 2014; Valle Junior et al., 2014).

Consequently, erosion and its impacts receive increasing attention from local, national, European and global policy makers (e. g., European Commission (EC))
as a result of its sensitivity to global change and the environmental and economic relevance of sediment dynamics.

From drinking water to hydro-power or irrigation canals, there is a growing interest in assessing the erosion control service provided by natural landscapes, to adopt watershed management measures that enhance this service (Clark, 1985; MA, 2005; CIRIA, 2013).

Policy makers require reliable predictions about how global change will affect erosion and retention processes in order to design effective mitigation and adaptation measures. Therefore, it is crucial to evaluate and minimize model uncertainty to avoid bias in decision making (Chavas, 2000; National Research Council, 2005).

Sediment production and transport in river basins is controlled by many factors, including rainfall patterns, soil characteristics, hill-slope steepness, and vegetation cover type. Generally, the erosion of a land with a permanent vegetation cover (shrubs, grassland and forest) is much lower than those on cultivated land (Cerdan et al., 2010).

Soil loss may therefore be mitigated through agricultural best management practices (Bakker et al., 2008), but broader and more dramatic land-use changes such as a transition from vegetated to urban areas may have larger effects on the capacity of a landscape to retain sediments from upslope areas.

The risks posed by altered sediment dynamics are particularly evident in Mediterranean and others semi-arid regions, which are among the most vulnerable areas to global change (Schröter et al., 2005).
Sediment dynamics models vary widely in complexity but typically involve a large number of parameters requiring calibration (Merritt et al., 2003). Irrespective of the model complexity, accurate characterization of all physical processes may not always be possible; input parameters are often difficult to measure and therefore need to be estimated.

The uncertainties related to parameter selection may therefore be significant and need to be quantified to improve model interpretation for watershed management.

In this study, a sensitivity analysis of a simple erosion control model calibrated in a Mediterranean basin (Llobregat river basin) was performed, in order to determine the parameters that had the greatest influence on the model outputs, and thus have particular importance for model calibration and interpretation.

The model was calibrated and validated in the case of study with a thoroughly explanation in Chapter 3. Despite only accounting for the sheetwash process in the sediment budget (gully or stream bank erosion are ignored), a reasonable estimate of sediment yield was obtained at the basin scale with physical parameters in reasonable ranges.

Due to the fact that complex models with a large number of parameters may have inherently large uncertainty, the InVEST erosion control model (version 2.4.4; Tallis et al., 2013), which is considered a relatively simple model, was chosen for the sensitivity analysis. The InVEST erosion control model can quantify the ecosystem services of erosion control, which is the relative contribution of different parts of the landscape to sediment retention.
Specifically, the model considers two different benefits derived from erosion control service: avoided reservoir sedimentation and higher water quality. For this purpose, the model employs a method based on the Universal Soil Loss Equation (USLE).

The sensitivity analysis used was the Morris method (Morris, 1991), which is a one-at-a-time (OAT) sensitivity analysis that aims at isolating the influential parameters from a large number of input factors determining the mean (μ) and standard deviation (σ) of ∂y/∂xi at several points, as well as the ‘mean of absolute values’ (μ*) defined by Campolongo et al. (2007), to avoid the effect of cancellation of terms with similar absolute values but different signs.

Chapter 2 gives a detailed explanation of the concept of ecosystem services, of the InVEST erosion control model, and the sensitivity analysis techniques, with an emphasis on the Morris method.

The InVEST erosion control model’s inputs for the sensitivity analysis included: rainfall erosivity (R); soil erodibility (K); and land use/land cover (LULC), with five LU categories (urban land, non-irrigated cultivated land, irrigated cultivated land, grass and shrub land, and forested land).

Three inputs were linked to land use type: cover management factor (C); the support practice factor (P); and a sediment retention value that identifies the capacity of vegetation to retain sediment as a percentage of the amount of sediment flowing into a cell from upslope. These inputs were varied within the ranges in Table 2, obtained from the literature (Wischmeier and Smith, 1978; US EPA, 2009; Terrado et al., 2014).
The SA indices, $\mu^*$ and $\sigma$, were calculated for each factor with respect to the 3 final outputs: (1) total sediment exported, (2) total sediment retained for the avoided reservoir sedimentation benefit, and (3) for the higher water quality benefit.

These parameters were calculated at both sub-basin and basin scales. The value of $\mu^*$ is an indication of the importance of the factor, with higher values attributed to factors to which the output of the model is most sensitive. The cut-off threshold for Morris method has to be defined. This threshold allows distinguishing between important factors and non-influential factors. A common threshold used is when $\mu^* < 0.1$ (Vanrolleghem et al., 2015).

A high value of $\sigma$ suggests the existence of nonlinear effects and/or interaction with other input factors, while a low $\sigma$ indicates independence. In particular, the Morris method (Morris, 1991) as modified by Campolongo et al. (2007) defines an area with values of $\mu^*$ higher than the cut-off threshold, and values of $\sigma$ higher than an oblique line. The oblique line is defined as $\sigma = \mu^* \sqrt{r / 2}$. The factors include in this area are interacting factors (Figure 23).
Figure 23. Representation of the Morris indices in a graph. A high value of $\mu^*$ is an indication of the importance of the factor. A high value of $\sigma$ suggests the existence of non-linear effects and/or interaction with other input factors, while a low $\sigma$ indicates independence (based on Morris, 1991; Campolongo et al., 2007; Vanrolleghem et al., 2015).

The relevant parameters obtained for total sediment exported were correlated with the characteristics of each sub-basin and with the whole river basin using the Spearman’s rank correlation matrix due to the small sample size and the presence of outliers in the data, invalidating the assumption of normal distribution.

To cross-check the results, we also computed the Pearson’s $r$ correlation coefficients, which proved to be similar. Chapter 4 presents an extensive explanation of the methodology used for the purpose of the present research.
6.2. RESULTS AND DISCUSSION

The results obtained for applying the Morris method in the InVEST water provisioning model in the Llobregat river basin are discussed below for each output.

6.2.1. Total sediment exported

Total sediment exported was calculated for the seven sub-basins as well as for the whole river basin (see Figure 11, Section 3.2). The sensitivity of this output with respect to the physical factors R (rainfall erosivity) and K (soil erodibility) was found to be similar for individual sub-basins and for the whole basin. Overall, the greatest influence was found for rainfall erosivity, closely followed by soil erodibility.

For the rest of the model inputs, differences in sensitivity were apparent, which we hypothesized to be due to differences in sub-basin characteristics. Figure 24 shows the results of the sensitivity analysis performed on two of the most different sub-basins (sub-basin 2 and 6), as well as for the whole river basin.

The entire basin has mostly natural vegetation with 38.2% of forest and 31.6% grass and shrub land. However, it also has a significant proportion of (non-irrigated) cropland (23.6%).
The results of the sensitivity analysis highlight that the support practice factor (P) for non-irrigated cultivated land and the cover management factor (C) for grass and shrub land have the greatest influence on sediment export, after the R and K factors (Figure 24a).

Similarly, in sub-basin 2 the most influential factors (apart from R and K) are C and P for grass and shrub land (Figure 24b) that reach values of $\mu^*$.
as high as those for K. This is a high mountain sub-basin composed of mostly forest (63.2%) with a wide area of shrubs and grass covers (35.1%), and a reservoir at the outlet (La Baells).

Sub-basin 6, in contrast, is mainly composed of non-irrigated cultivated land (43.8%), with less grass and shrub land (25.5%) and forest (24.9%). The most important factor after R and K for this sub-basin is P for non-irrigated cultivated land (Figure 24c).

These observations were confirmed by the correlation analyses. The Spearman’s rank correlation matrix shows the influence of the characteristics of each sub-basin, in particular the percentage of each land use, on the sensitivity of each parameter. In general, the proportion of non-irrigated cultivated land use and forests were significantly correlated with the sensitivity of the factors C and P.

Table 6. Spearman’s $\rho$ correlation matrix between sensitivity of model inputs, and river basin characteristics for total sediment export output. Sensitivity of model inputs are evaluated as $\mu^*$ and the variation in that $\sigma$. Only inputs with a significant sensitivity index value ($\mu^* N 0.1$) are included here. Greyed values are those where $|\rho| N 0.6$, indicating a strong correlation (null hypothesis rejected with a p-value $b 0.05$).
Additional observations can be made from Table 6. It seems that the higher the mean slope or the mean rainfall, the most important are the C and P factors for the land use of grass and shrub land, and also K is directly correlated with mean slope in the Llobregat river basin.

In contrast, the higher the mean slope or the mean rainfall, the less influential the C and P factors were for non-irrigated cultivated land. Interestingly, although input factors related to forested land were not significantly correlated with basin characteristics, the fraction of this LULC in the study area affects the relevance of others important inputs. In fact, the higher the fraction of forested land, the less important C and P become for non-irrigated cultivated land, as well as R and K.

To provide some perspective to the qualitative sensitivity analyses, we also provide quantitative insights into the effect of parameter errors. Table 7 reports the results of several pairs of runs from the sensitivity analyses, showing the relative difference in input parameters and the resulting difference in sediment export. For the whole basin, the first example shows the difference in sediment export (~49%) due to the change only in C for non-irrigated land use (a decrease of 64%).
This result confirms that the model is less sensitive to C parameters than to R and K parameters, to which the model responds proportionally, according to the analysis of the model structure. The three other pairs of runs illustrate how differences in parameters may compensate each other.

For example, for the whole basin, we show that an increase of 63% in R may be compensated by high decreases in P for non-irrigated and C for forest land uses (the difference in sediment export is only 0.2%).

Similarly, in sub-basin 2, an increase in both physical factors (46% for R and 33% for K, respectively) and high decreases in C and P factors result in comparable sediment export (1% difference in sediment export).
In sub-basin 6, similar results were obtained with an increase of 46% in R and a high decrease in the most sensitive inputs of human-related factors (C and P).

These examples illustrate a classic case of equifinality in environmental modelling, which discussion is outside the scope of this work. In practical terms, they suggest that changes in some parameters (e.g. rainfall erosivity) may be compensated by others (e.g. management factors), as discussed below.

6.2.2. **Total sediment retained**

As explained before, the model assesses the erosion control service as the total sediment retained in the basin, i.e. the amount of soil retained by the landscape. The two different benefits calculated for this ecosystem service are higher water quality and avoided reservoir sedimentation. In the basin under study, the total sediment retained that could contribute to higher water quality was calculated for the whole basin, since the drinking water treatment plant is located at its outlet.

However, the avoided reservoir sedimentation was calculated for sub-basins 2, 3, and 4, each of which has a reservoir at its outlet (see Figure 13 for the location of these basins). In contrast to the complex dependencies seen for sediment export (above), the model for total sediment retention (for both avoided reservoir sedimentation and improved water quality) only appears to be strongly sensitive to the physical factors, R and K.
Since the results from the sensitivity analysis shows that the provision of both benefits in all the sub-basins and the whole basin are the same, avoided reservoir sedimentation and improved water quality are independent of the basins’ characteristics. These results are best interpreted in light of the model structure.

The total sediment retained is the sum of sediment retention on each pixel, which is itself based on a simple comparison between the current LULC and bare soil (see Methods). This suggesting that factors affecting all pixels (such as R and K) will have a stronger effect than LULC-specific factors that only affect a proportion of the pixels.

### 6.2.3. Management implications

The sensitivity analysis of the InVEST erosion control model suggests that R and K (rainfall erosivity and soil erodibility) are the most influential inputs in the Mediterranean river basin under study. It is worth noting that both of these factors are potentially climate-driven; R directly and K through the effects of temperature and moisture on the soil’s organic content.

The soil erodibility (K) is related to soil organic matter content, which varies with temperature and moisture in a non-linear way, although with high spatial heterogeneity due to its relationship with vegetation cover patterns (Lavee et al., 1998; Sarah, 2006). This implies that in general, the model has greater sensitivity to climate change than anthropogenic land-use change.
However, despite the prominent influence of the climate-driven factors, other factors directly related to land management played a significant role for total sediment exported. Cover management and support practice factors were relevant for some LULC, depending on the proportion of each LULC type and sub-basin slopes.

Regional differences in model sensitivity to anthropogenic factors were observed for sub-basins differing in LULC composition and mean slope. In high mountain sub-basins such as basin number 2 (Figure 13b), since the cover management factor for grass and shrubland are as relevant as climate driven factors, it could be possible to mitigate erosion from climate-driven changes by promoting species with wider ground cover on this type of land.

The result obtained in the sub-basin 2 is in agreement with the results reported by Catari’s Thesis dissertation (2007) in the same area (Upper Llobregat, above La Baells reservoir). This work reports that there is an increase in erosion rates for all the scenarios studied incorporating global change, and concludes that the two most sensitive inputs for all these scenarios are both rainfall erosivity and cover management-related factors.

In contrast, in truly Mediterranean climates such as sub-basin number 6 (Figure 13c), the support practice factor in non-irrigated cultivated land is the most relevant factor after the climate-driven ones, suggesting that the most effective strategy to enhance the retention service would be
expanding management practices such as contouring, strip-cropping or terracing.

Therefore, different mitigation or adaptation management actions may be optimally beneficial in different regions in the same river basin. As suggested in Table 7 a 63% increase in rainfall erosivity (R) for the whole basin could be compensated with changes in the cover management factor (C) for forest land, and in the support practice factor (P) for non-irrigated land use. For sub-watersheds 2 and 6, the increase in climate-driven factors could be possibly mitigated with changes in cover management.

Finally, despite the model's lack of sensitivity to forested LULC factors, it is interesting to note that the proportion of forest cover in the basins is correlated with the sensitivity of the factors C and P on the others land-uses; in sub-basins with lower forest cover, the model was more sensitive to C and P on agricultural land-uses but less so to grass and shrub land covers. While these results may seem counter-intuitive, they can be explained by the high retention capacity of forested LULC.

Notice that this analysis does not take into account spatial distribution of the LULC, and that patches closer to the stream would likely have a greater impact on sediment retention and export. In addition, since the results of the correlation analyses are limited to a small sample size (7 watersheds), the robustness of the result should be interpreted with caution, especially if they are used in future studies.
6.3. CONCLUSIONS

The sensitivity analysis of the InVEST sediment model identifies the climate-driven model parameters (rainfall erosivity and soil erodibility factors) as the most influential ones for sediment export and retention, and other parameters such as cover management or support practice as playing a secondary but significant role.

Accordingly, small changes in the magnitude and frequency of extreme rainfall events may cause major changes in sediment dynamics, highlighting the susceptibility of erosion control service to climate change in Mediterranean basins.

Moreover, the results from this sensitivity analysis further suggest that it is feasible to compensate the likely effects of climate change on sediment export in some areas by introducing management practices such as contouring, strip-cropping or terracing, as well as increasing the percentage of soil coverage by vegetation.

Implications from such analyses should, however, be put in perspective of the scope of the model: structural errors in the modelling of sheetwash erosion and the omission of additional sources of sediment may also impact management decisions for erosion control.

This work focused on a simple one-parameter-at-a-time method (Morris method) for the sensitivity analysis because the results reveals that the parameters are independent (low $\sigma$ for all the parameters studied, less than 1/3 of $\mu$). As it is explaining in Chapter 4, a high value of $\sigma$ suggests the existence
of nonlinear effects and/or interaction with other input factors, while a low $\sigma$ indicates independence (Morris, 1991).

The Morris method proved to be sufficiently reliable; hence, it is not necessary to consider other sensitivity analysis techniques that take into account the interactions between multiple parameters.
Chapter 7

7. IN-STREAM PROCESSES FOR THE INVEST NUTRIENT RETENTION MODEL

7.1. INTRODUCTION

Nitrogen (N) and phosphorus (P) play an important role in controlling the trophic levels of surface waters. Consequently, modeling those nutrients can help to evaluate possible impacts of policies aimed at reducing or managing N and P in river systems (Jarvie et al., 2002).

To this end, modeling tools and techniques have been used to detect nutrient resources and relevant nutrient processes at the watershed scale (Smith and Alexander, 2000; Grizzeti et al., 2005; Aguilera et al., 2012). Models can prove valuable for catchment management and policy decision-making by simulating these nutrients sources, and their delivery and transport processes.
However a compromise between model complexity and data requirement is necessary to identify the most suitable modeling tool for the intended application (Paudel and Jawitz, 2012). As the structure of ecological models grows increasingly complex, it becomes necessary to identify the optimal level of complexity that most reliably describes and predicts the processes under study.

In this way, there is an ongoing effort for adjusting model structure to most appropriately represent nutrient cycling dynamics and establish a rigorous relationship with field data to provide reliable predictions (Paudel and Jawitz, 2012).

The choice of an appropriate model structure should be based on the requirements of a given research or management objective. A complex model may be an appropriate option as a tool for more fundamental scientific inquiry, such as testing hypotheses or gaining understanding the system components and interactions. More complex models comprise more components and processes, and generally represent complex systems more comprehensively than their more simple counterparts.

As Beven (2001) emphasizes, the processes perceived to have an effect in the real system should be included in a model. However, if the models are intended for management purposes, simple alternatives may also satisfy the modeling needs because data scarcity might be more important than accuracy, and they are relatively easy to analyze due to the relatively lower number of parameters they comprise. Simple models can serve as applied tools for managers to
evaluate basin performance as well as conducting multiple diagnostic tests of management alternatives.

The InVEST software consists of a suite of tools that are specifically designed for modeling ecosystem services and their benefits. Particularly, the InVEST nutrient retention model (version 2.6.0; Tallis et al., 2013) estimates the service derived from the quantity of pollutants retained for water purification from the landscape and the benefits of water purification are assessed for two pollutants (total nitrogen, TN, and total phosphorus, TP). Chapter 2 gives a detailed explanation of the concept of ecosystem services and the InVEST nutrient retention model. An important aspect of the model for the present work is that the model only represents the contribution of the landscape to the watershed.

The InVEST model first computes the amount of nutrient retained by the landscape and then assess the benefit this retention service. Therefore, the current version of the InVEST nutrient retention model only allows the user to calculate benefits derived from terrestrial ecosystem services. In-stream processes are not currently implemented in the model, which may result in an overestimation of the landscape retention when users calibrate the model with measured data.

Streams and rivers help control exported nutrients they carry from terrestrial and upstream sources (Ensign et al., 2006). In order to obtain better modeling accuracy, and consequently a better management with the model results, it is vital to understand the connection between terrestrial nutrient sources and the aquatic nutrient transformation, storage, removal, and transport within a watershed (Ensign and Doyle, 2006; Wollheim et al., 2006).
In this line, two recent studies have demonstrated the potential role of in-stream processing in limiting nutrient export in the Llobregat River basin (Aguilera et al., 2012; 2013).

Based on these findings and following the recommendation of Paudel and Jawitz (2012) to evaluate models across varying levels of complexity, several tests were made in order to determine the role of in-stream processes in the watershed in order to assess the need to include in-stream retention in the InVEST nutrient retention model.

Including in-stream water purification processes in InVEST might be necessary for two main reasons:

- to be able to assess the ecosystem services supplied by the stream network itself;
- to avoid potential biases in services estimated for terrestrial ecosystems caused by ignoring river function.

In an attempt to test if this new knowledge is worth being included in the InVEST, a methodology for adding the in-stream processes in the InVEST nutrient model was designed. This methodology was devised in order to test nutrient retention of the stream without the need to change the source code in InVEST.

The methodology is thoroughly explained in Chapter 4, which consist in adding the river and reservoirs as new land uses in all the tests performed, and a retention coefficient was applied for these new land uses. This strategy has the
advantage of avoiding changing the model’s source code, reducing time and possible errors from modifying the original code.

The 3 land use classes added to the land use map were: dam, reservoir and river. In the subsequent tests, the same retention for the three new land use classes was assumed. However, the code does not allow any kind of retention on the pixels in the river network; therefore a river network of only one pixel, at the watershed outlet, was created to successfully implement the proposed strategy. This was done by testing the threshold flow accumulation up to find the value that causes the model generates a one-pixel river network.

The results of exported nutrients were estimated from the InVEST retention model and were used to build different scenarios, for both TN and TP (described in detail in Chapter 4, and summarised in Table 4 and Table 5).

For Total Nitrogen, the scenarios performed and calibrated were the following:

- **TN_BASE**: base scenario with a retention coefficient of 0.5% for aquatic LULC and a vegetation filtering value of 25% for cultivated land areas (both irrigated and non-irrigated) and 50% for natural land areas (grass/shrub and forest). The result of this scenario has a relative error with respect to the reference value of -1.6%.

- **TN+1**: In this scenario, the retention coefficient of the aquatic LULC pixels was 0.4%, which means a 0.1% decrease compared to the base scenario (TN_BASE). The calibration of the result was obtained with an increase in the terrestrial LULC retention coefficient of 30% (relative error of 1.6%).
- **TN+2**: For this scenario, the retention coefficient for the aquatic LULC pixels was set to 0.3%, which corresponds to a decrease of 0.2% with respect to the base scenario (TN_BASE). It was necessary to increase the terrestrial LULC retention coefficient by 85% for the new calibration (relative error of 1.0%).

- **TN-1**: The retention coefficient was increased by 0.1% with respect to the base scenario (TN_BASE), or equivalently, the retention coefficient was set to 0.6% on the aquatic LULC pixels. The terrestrial LULC retention coefficient was decreased by 30% for the new calibration (relative error value of -0.2%).

- **TN-2**: The retention coefficient for the aquatic LULC pixels was set to 0.7%, i.e. an increase of 0.2% with respect to the base scenario (TN_BASE). For the new calibration, the terrestrial LULC retention coefficients were decreased by 50% (relative error of 0.3%).

- **TN-3**: The retention coefficient was set to 0.8% for the aquatic LULC pixels, what means an increase of 0.3% with respect to the scenario base (TN_BASE). It is necessary to decrease the terrestrial LULC retention coefficient by 65% for the new calibration to be within a relative error of 1.2%. 
For Total Phosphorus, the scenarios performed and calibrated were the following:

- **TP BASE**: the base scenario had a 0% retention coefficient for aquatic LULC and a vegetation filtering value of 45% for cultivated land (both irrigated and non-irrigated), and 90% for natural land areas (grass/shrub and forest). The result of this scenario has a relative error with respect to the reference value of -0.9%.

- **TP-1**: For this scenario, 0.1% was the value used in the retention coefficient of the aquatic LULC pixels, which is an increase of 0.1% with respect to the base scenario (TN_BASE). It was necessary a decrease the terrestrial LULC retention coefficient by of 35% for the new calibration (relative error of 1.5%).

- **TP-2**: Increasing the retention coefficient of the aquatic LULC pixels by 0.1% with respect to the scenario TP-1, or equivalently, using a retention coefficient of 0.2% for those pixels. The terrestrial LULC retention coefficient was decreased by 55% for the new calibration (relative error of -0.5%).

- **TP-3**: Increasing the retention coefficient of the aquatic LULC pixels by 0.1% as in the previous scenario, which translates to a retention coefficient of 0.3%. The terrestrial LULC retention coefficients were decreased by 70% for the new calibration (relative error of -1.0%).
- **TP-4**: the retention coefficient for the aquatic LULC pixels was 0.1% higher than in TP-3, which corresponds a 0.4%. A decrease of 82% in the terrestrial LULC retention coefficient was necessary for the new calibration to be within a relative error of 1.2%.

- **TP-5**: When the retention coefficient of the aquatic LULC pixels is 0.5%, i.e. an increasing of 0.1% with respect to the TP-4 scenario. The terrestrial LULC retention coefficient was decreased by 88% (relative error of -1.1%).

- **TP-6**: The retention coefficient of the aquatic LULC pixels used for this scenario was 0.6%, i.e. an increasing in 0.1% with respect to the before scenario. For the calibration of this scenario the terrestrial LULC retention coefficients were decreased in 94% (relative error of 0.8%).

- **TP-7**: In this scenario, the retention coefficients of the aquatic LULC pixels used were 0.7% (increasing 0.1% the retention coefficients of the scenario above). The terrestrial LULC retention coefficients were decreased in 98% for the calibration of this scenario (relative error of 1.5%).

The calibration of all the scenarios was made against the monitoring export value to achieve a relative error within ± 2%. This error margin was achievable for scenarios within a range of variation in retention coefficient for aquatic LULC pixels between 0.3% and 0.8% for TN, and between 0% and 0.7% for TP.
In addition, the model also estimates the nutrient retention of the benefits chosen of a service for all these scenarios, as it was explained in Chapter 2, however, the nutrient retention needed for the present study was the real one.

The real retention is the one calculated before the extraction of the total allowed annual load for the pollutant of interest. The reason to use the real retention of the nutrient of interest instead of the retention of the benefit selected is because the purpose of the study is the real share of the river and the terrestrial pixels in the retention of both nutrients, and not the relative to benefit.

7.2. RESULTS AND DISCUSSION

All the calculations were made for both nutrients (TN and TP) and the results obtained for each one were the following:

**Nitrogen**

The real terrestrial nitrogen retention for each of the TN scenario is shown in Figure 25. The retention coefficients in the aquatic land use pixels range between 0.3 and 0.8%. The variation/adjustment in retention in the terrestrial land use pixels is represented as a retention coefficients decrease or increase in percent with respect to the base scenario (TN_BASE).

The graph (Figure 25) shows that as the retention coefficients in river pixels increase, the retention coefficients decrease more slowly for higher river retention values in a non-linear fashion. This relationship can be explained by the higher retention efficiency of the river pixels, as the river carries all the nutrient load exported from land, however, the land pixels only have to retain
the loads from the land pixels above them and the most efficient terrestrial pixels are those closest to the stream.

![Graph showing percent increased or decreased in terrestrial retention coefficient with respect to the base scenario for each of the 6 scenarios performed and calibrated for Total Nitrogen.](image)

*Figure 25. Percent increased or decreased in terrestrial retention coefficient (with respect to the base scenario) for each of the 6 scenarios performed and calibrated for Total Nitrogen.*

The real TN retention was calculated and saved as a raster map for all the scenarios. Then two raster maps were further derived for each scenario. The first one is the real TN retained for one of the terrestrial LULC pixels (in this case, the forest land use was mapped as an example of terrestrial land use), and the second was the real TN retained for water pixels. An example of both raster maps for the TN+2 scenario (a 0.3% retention coefficient for aquatic LULC pixels) is shown in Figure 26. The raster map on the left is the real TN retained for each pixel of forest in the TN+2 scenario, and the raster map on the
right is the real TN retained for each pixel of aquatic land, both in kg of terrestrial total nitrogen retained per pixel per year.

Figure 26. The real TN retained for each pixel of the forest land (a) and for each pixel of aquatic (b), both for the TN+2 scenario (0.3% retention coefficient for aquatic LULC pixels) in Kg TN/pixel year.

The real retention of each aquatic land use pixel was plotted for three of the scenarios (TN+2, TN_BASE and TN-2) in Figure 27. The ranges of the most probable values of the rivers in the world obtained from the literature (references used to compile the bibliographical data are in Appendix A; source: Aguilera et al., 2013) are plotted in Figure 28 with a range between 65 and 1165
kg of TN per ha per year (1st and 3rd quartiles resp.). By comparing the results of the nutrient retention values of all the pixels with the most probable ranges (highlighted between two horizontal lines in Figure 27) obtained from the literature (references used to compile the bibliographical data are in Appendix A; source: Aguilera et al., 2013), the scenario with most pixels (63% in Table 8) in this range is the TN-2 scenario.

Figure 27. Distribution of the real TN retained in all the aquatic pixels the TN+2 scenario (a), TN_BASE scenario (b), and TN-2 scenario (c). The values of these pixels are sorted from lowest to highest. The percent of aquatic pixels with a real TN retained included in the most probably range of retention in rivers of the word are 36%, 47% and 63% respectively.
Figure 28. Distribution of the real TN retained in rivers of the world. Source: Aguilera et al. 2013. References used to compile the bibliographic data are in Appendix A.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>In-stream: Retention coefficients (%)</th>
<th>River Pixels within the most probably range of retention in rivers of the world (between 1st and 3rd quartile)</th>
<th>Total pixel of river in the watershed studied</th>
<th>% pixels of river within the most probably range of retention in rivers of the world</th>
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</thead>
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<tr>
<td>TN+2</td>
<td>0.3%</td>
<td>1624</td>
<td>4563</td>
<td>36%</td>
</tr>
<tr>
<td>TN_BASE</td>
<td>0.5%</td>
<td>2150</td>
<td>4563</td>
<td>47%</td>
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<tr>
<td>TN-2</td>
<td>0.7%</td>
<td>2855</td>
<td>4563</td>
<td>63%</td>
</tr>
</tbody>
</table>

Table 8. The percent of aquatic pixels with a real TN retained included in the most probably range of retention in rivers of the world for the TN-2 scenario, TN_BASE scenario, and TN+2 scenario.

The real retention of each forest pixel for three of the scenarios (TN+2, TN_BASE and TN-2) was also plotted in Figure 29, however, the values most likely can’t be compared because there are no consistent references in the literature about them, and the present work only focused on river.
Figure 29. Distribution of the real TN retained in all the forest pixels the TN+2 scenario (a1), TN_BASE scenario (b1), and TN-2 scenario (c1). The values of these pixels are sorted from lowest to highest (zoom in a2, b2, and c2).

The real TN retention calculated for each pixel in both land uses referred above (water and forest land uses) was also plotted in a boxplot (Figure 30) for a better visualize the distribution for three sample scenarios (TN+2, TN_BASE and TN-2 scenarios).

In Figure 30, the most likely results of the scenario TN-2 are between the first and third quartiles (90-690 Kg TN/pixel year). This range of values is entirely
within the range of the most probable results from the literature about retention in rivers of the world (between 65 and 1165 kg TN/ha year, 1st and 3rd quartiles resp. in Figure 28), suggesting this might be the most likely scenario among the tested scenarios.

![Box plot showing distribution of retention in pixels of river](image)

*Figure 30. Distribution of the real TN retained obtained for each aquatic pixel in three of the scenarios studied (TN+2, TN_BASE, and TN-2).*

In Figure 31, the real TN retention in the pixels of forest for the three scenarios represented, the range of the values more probably are very similar, 21-163 Kg TN/pixel year for TN+2 scenario, 15-157 Kg TN/pixel year for TN_BASE scenario and 8-131 Kg TN/pixel year for TN-2 scenario. Those results show that the variation between different scenarios of the real retention in forest pixels is low (much more lower than in water pixels).
Figure 31. Distribution of the real TN retained obtained for each forest pixel in three of the scenarios studied (TN+2, TN_BASE, and TN-2).

Similar, the Figure 32 shows the rate of the real retention per ha for the aquatic and terrestrial pixels for each of the 6 scenarios with a possible calibration. The retention rate for terrestrial pixels is almost constant while that of the aquatic pixels shows a steep slope, which means that the aquatic pixel retention increases rapidly when the retention coefficient increases.
Figure 32. Ratio of the real retention per ha for the aquatic pixels (with black points), and for terrestrial pixels (with white points) all of the 6 scenarios for TN calibrated.

In this figure (Figure 32), the six scenarios studied (aquatic pixels retention coefficient between 0.3% and 0.8%) are reasonable, because the values of their retention coefficients are within the most probable retention coefficients (1st and 3rd quartiles) calculated by Aguilera et al. (2013) that correspond to retention coefficients in aquatic pixels between 0.27% and 0.89% for the Llobregat river basin (Table 9).
Table 9. Distribution of the aquatic retention coefficients (TN retained coefficient in river, in %) calculated in the Llobregat river basin by Aguilera et al. (2013).

<table>
<thead>
<tr>
<th></th>
<th>% retention of TN (Llobregat river basin)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>maximum</strong></td>
<td>7.09</td>
</tr>
<tr>
<td><strong>3rd quartile</strong></td>
<td>0.89</td>
</tr>
<tr>
<td><strong>median</strong></td>
<td>0.46</td>
</tr>
<tr>
<td><strong>1st quartile</strong></td>
<td>0.27</td>
</tr>
<tr>
<td><strong>minimum</strong></td>
<td>0.03</td>
</tr>
</tbody>
</table>

**Phosphorus**

Figure 33 shows the real terrestrial phosphorus retention for each of the TP scenarios. The pixels of aquatic are within 0.0 and 0.7%. The base scenario (TP_BASE) is the one without retention in aquatic while the rest of the scenarios are built from if by increasing aquatic retention and adjusting retention for the terrestrial land use pixels.

As for the case of phosphorus retention, the graph shows that as the retention coefficients in river pixels increase, the retention coefficients decrease more slowly for higher river retention values in a non-linear fashion. This relationship can be explained by the higher retention efficiency of the river pixels, as the river carries the entire nutrient load exported from land, however, the land pixels only have to retain the loads from the land pixels upstream them. The most efficient terrestrial pixels are those closest to the stream.
Figure 33. Percent decrease in terrestrial retention coefficient (with respect to the base scenario) for each of the 8 scenarios performed and calibrated for Total Phosphorus.

The real TP retention was calculated and mapped for all the scenarios. Then, two raster maps were derived. Those maps are the real TP retained for each terrestrial LULC pixels, and the real TP retained for aquatic pixels. As an example, the first map is shown with only the forest land use selected. An example of both raster maps for the TP-2 scenario (a 0.2% retention coefficient for aquatic LULC pixels) is shown in Figure 34. The raster map on the left is the real TP retained for each pixel of forest in the TP-2 scenario, and the raster map in the right is the real TP retained for each pixel of aquatic land, both in kg of terrestrial total phosphorus retained per pixel per year.
Figure 34. The real TP retained for each pixel of the forest land (a) and for each pixel of aquatic (b), both for the TP-2 scenario (0.2% retention coefficient for aquatic LULC pixels) in Kg TP/pixel year.

The real retention of each aquatic land use pixel was plotted for three of the scenarios (TP-2, TP-4 and TP-6) in Figure 35. The ranges of the most probable values of the rivers in the world getting from the literature (references used to compile the bibliographical data are in Appendix B; source: Aguilera et al., 2013) are plotted in Figure 36 with a range between 54 and 292 kg of TP per ha per year (1st and 3rd quartiles resp.). By comparing the results of the nutrient retention values of all the pixels with the most probable ranges (highlighted between two horizontal lines in Figure 35) obtained from the literature (references used to compile the bibliographical data are in Appendix B; source: Aguilera et al., 2013), the scenario with most pixels (19% in Table 10) in this range is the TP-6 scenario.
Figure 35. Distribution of the real TP retained in all the aquatic pixels the TP-2 scenario (a), TP-4 scenario (b), and TP-6 scenario (c). The values of these pixels are sorted from lowest to highest. The percent of aquatic pixels with a real TN retained included in the most probably range of retention in rivers of the word are 10%, 16% and 19% respectively.

Figure 36. Distribution of the real TP retained in rivers of the world. Source: Aguilera et al. 2013. References used to compile the bibliographic data are in Appendix B.
<table>
<thead>
<tr>
<th>Scenario</th>
<th>In-stream: Retention coefficients (%)</th>
<th>River Pixels within the most probably range of retention in rivers of the word (between 1st and 3rd quartile)</th>
<th>Total pixel of river in the watershed studied</th>
<th>% pixels of river within the most probably range of retention in rivers of the word</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP-2</td>
<td>0.2%</td>
<td>448</td>
<td>4563</td>
<td>10%</td>
</tr>
<tr>
<td>TP-4</td>
<td>0.4%</td>
<td>732</td>
<td>4563</td>
<td>16%</td>
</tr>
<tr>
<td>TP-6</td>
<td>0.6%</td>
<td>866</td>
<td>4563</td>
<td>19%</td>
</tr>
</tbody>
</table>

Table 10. The percent of aquatic pixels with a real TP retained included in the most probably range of retention in rivers of the word for the TP-2 scenario, TP-4 scenario, and TP-6 scenario.

For three of the scenarios (TP-2, TP-4 and TP-6) the real retention of each forest pixel was also plotted in Figure 37, however, the values most likely can’t be because there are no consistent references in the literature about them, and this work only focused on river.

Figure 37. Distribution of real TP retained in all forest pixels TP-2 scenario (a1), TP-4 scenario (b1), and TP-6 scenario (c1). The values of these pixels are sorted from lowest to highest (zoom in a2, b2, and c2).
The real TP retention calculated for each pixel in both land uses referred above (water and forest land uses) were also plotted in a Boxplot to better visualize the distribution of three sample scenarios (TP-2, TP-4 and TP-6 scenarios).

In Figure 38, the most likely values for the scenario TP-6 are between the first and third quartiles (9-72 Kg TP/pixel year). This is the scenario with most of its values within the range of the most probable range in the literature about retention in rivers of the world (between 54 and 292 kg TP/ha year, 1st and 3rd quartiles resp., in Figure 36), suggesting this might be the most likely scenario among the ones tested.

![Boxplot of TP retention in pixels of the river](image)

*Figure 38. Distribution of the real TP retained obtained for each aquatic pixel in three of the scenarios studied (TP-2, TP-4, and TP-6).*
In Figure 39, the real TP retention from forest pixels in the three scenarios represented above, the most probable range of values is very small, 0.4-4.8 Kg TP/pixel year for the TP-2 scenario, 0.2-3.5 Kg TP/pixel year for the TP-4 scenario and 0.1-1.4 Kg TP/pixel year for the TP-6 scenario. Those results show that the variation between different scenarios of real retention in forest pixels is low (much more lower than in water pixels), however, the scenario with most variability is the one with little retention in aquatic pixels (TP-2).

![Figure 39. Distribution of the real TP retained obtained for each forest pixel in three of the scenarios studied (TP-2, TP-4, and TP-6).](image)

Similar, the Figure 40 shows the rate of real retention per ha of aquatic and terrestrial pixels for each of the 8 scenarios with the most plausible calibration. The rate for terrestrial pixels is almost constant while that of the water pixels has a steep slope, which means that water pixel retention increases rapidly when the retention coefficient increases.
Figure 40. Ratio of the real retention per ha for the aquatic pixels (with black points), and for terrestrial pixels (with white points) all of the 8 scenarios for TP calibrated.

In this figure, only two scenarios studied are reasonable (those ones with aquatic pixels retention coefficient of 0.6% and 0.7%, respectively), because the values of their retention coefficients are within the most probable retention coefficients (1st and 3rd quartiles) calculated by Aguilera et al. (2013) that correspond to retention coefficients in aquatic pixels between 0.59% and 1.82% for the Llobregat river basin (Table 11).
Table 11. Distribution of the aquatic retention coefficients (TP retained coefficient in river, in %) calculated in the Llobregat river basin by Aguilera et al., (2013).

<table>
<thead>
<tr>
<th>% retention of TP</th>
<th>Llobregat river basin</th>
</tr>
</thead>
<tbody>
<tr>
<td>maximum</td>
<td>10.20</td>
</tr>
<tr>
<td>3rd quartile</td>
<td>1.82</td>
</tr>
<tr>
<td>median</td>
<td>1.06</td>
</tr>
<tr>
<td>1st quartile</td>
<td>0.59</td>
</tr>
<tr>
<td>minimum</td>
<td>0.07</td>
</tr>
</tbody>
</table>

7.3. CONCLUSIONS

The proposed methodology in order to include the in-stream processes in the InVEST nutrient retention model allows to estimate the retention of total nitrogen (TN), and total phosphorus (TP) from the stream in the basin studied.

The results obtained for TN as well as for TP have the same tendency, a small increase in the retention coefficient assigned to the aquatic pixels leads to a large decreases in the terrestrial retention coefficients (i.e. + 0.1% in aquatic retention between TP_BASE and TP-1 means a -35% in terrestrial retention).

Those significant changes are more relevant when there is a substantial difference in the watershed between the terrestrial and the aquatic surfaces. The proportions of terrestrial and aquatic surfaces are 96% and 4% respectively (Figure 17).

In addition, the methodology developed in order to estimate in-stream retention shows that it is possible to include retention in streams in a way that is similar to
how the model currently estimates the retention in land, with a constant retention coefficient in the aquatic pixel.

The results for retention in the aquatic pixels in the Llobregat river basin obtained with this method are in line with those obtained in recent studies for the same study area (Aguilera et al., 2012; 2013). This simple way of incorporating retention for aquatic pixels in the model is useful as a first approach, when the user has little information and/or data. However, deeper studied approaches could be recommended for inclusion. One of these approaches would be to distinguish between stream sizes based on a hierarchy of tributaries with the Strahler stream order (Strahler, 1957) and to assign different values of retention coefficient to each stream category. In this way, the pixels for dams or reservoirs should also have different retention coefficients, because retention in those pixels is much higher than that of the other aquatic pixels.

Figure 41. The Strahler stream order, which assigns each headwater perennial stream an order of 1, and then at the confluence of two 1st-orders Streams assigns the downstream reach an order of 2, and so forth (based on Strahler, 1957).
In addition, including in-stream retention in the InVEST nutrient retention model, not only the overestimation of the terrestrial retention could be compensated, but the benefits of the ecosystem services of the streams could also be evaluated, increasing the model usefulness for watershed management.
Chapter 8

8. GENERAL SUMMARY, CONCLUSIONS AND SUGGESTIONS FOR FURTHER RESEARCH

8.1. GENERAL SUMMARY AND CONCLUSIONS

During the modeling process, real-world situations often have to be idealized through simplifications in order to obtain a model. Applying a model is useful in understanding the behavior of a particular system, causes of possible changes, and how sensitive it is to some of these changes.

In addition, in the field of environmental management, decision-making is often supported by models because of the need to predict future events.

However, model estimations lead to uncertainties resulting from input data and parameter variability, as well as from model structure. Often, significant management decisions are taken based on model results, so model uncertainty
should be a key concern. Models are powerful tools and improving them will allow users to reduce uncertainty and increase the confidence in the results obtained. Moreover, those model uncertainties have to be evaluated formally if the model is going to be used as a decision-support tool.

The role of the sensitivity analysis (SA) is to assign model output uncertainty based on the inputs and can increase confidence in model predictions by providing deeper understanding of model behavior.

However, sensitivity analysis is often omitted in modelling, usually as a result of the growing effort it involves as the complexity and size of the models increase. The application of sensitivity methods for formal evaluation of models is still uncommon in spite of its importance (Zajac, 2010).

In addition, due to the fact that a model is always a simplification of reality, it should not be more complex than what is absolutely necessary to describe the required outputs. However, it is not easy to assess the balance between accuracy and simplicity.

For this reason, when a model is developed, it is necessary to test it in order to understand its behavior and to include, if necessary, more complexity to get a better response.

The objective of this research is two-fold: first, assess the application of a sensitivity analysis method on a specific ecosystem service model, and second, develop a methodology with the addition of the in-stream processes that are not currently modeled in the cited model.
Ecosystem services are the conditions and processes through which natural ecosystems, and their constituent species, sustain and fulfill human life. Natural ecosystems are very important to societies because ecological systems services and the natural capital stocks that produce them are critical to the functioning of Earth’s life-support system.

The relevance of ecosystem services and the need to better manage them and their associated benefits have stimulated the emergence of models and tools to measure them.

Ecosystem services tools represent a new kind of dedicated tools, focusing mainly on end services and their visualization across a landscape.

InVEST, Integrated Valuation of Ecosystem Services and Tradoffs, is one of these ecosystem services-specific tools, and is widely accepted by scientists because it uses biophysical relationships that are good estimates of physical reality, despite its limits and assumptions.

InVEST is developed by the Natural Capital Project. It is a platform that maps and quantifies ecosystem services and how they benefit society. It is a free and open-source software suite to inform and improve natural resource management and investment decisions (www.naturalcapitalproject.org).

In addition, the versions of this tool used in the present work have the advantage of being implemented using relatively transparent code that can be modified if needed.

As a result of the growing interest in measuring ecosystem services, the use of InVEST is anticipated to grow exponentially in the coming years. However,
apart from model development, making a model involves other crucial stages such as its evaluation and application in order to validate estimations.

The results of this Thesis will contribute to the understanding of the uncertainties involved in the modeling process. It will also illustrate the need to check the behavior of every model developed before putting them in production and illustrate the importance to understand their behavior in terms of correctly interpreting the results obtained in light of uncertainty.

The work in this Thesis will contribute to improve the InVEST platform, which is an important tool in the field of ecosystem services. Such work will benefit future users, whether they are researchers (in their future research), or technicians (in their future work in ecosystem conservation or management decisions).

The work developed in this Thesis tries to help in the relevant and imperative phase of the modeling process, and does so in two different ways.

The first one is to conduct a sensitivity analysis of the model, which consists in choosing and applying a methodology in an area and analyzing the results obtained.

The second is related to the in-stream processes that are not modeled in the current model, and consists in creating and applying a methodology for testing the role of streams in nutrient retention in a case study and analyzing the results obtained.

The most relevant conclusions obtained in this Thesis are following summarized:
The Morris method for sensitivity analysis was applied for two InVEST models (water provisioning and erosion control) in the Llobregat River basin to identify the most sensitive input parameters for each model. The Morris method proved to be sufficiently reliable; hence, it is not necessary to consider other sensitivity analysis techniques that take into account the interactions between multiple parameters.

The results obtained for the InVEST water provisioning model indicate that the seasonality factor that presents the seasonal rainfall distribution and rainfall depths during the year (Z coefficient) is not a sensitive factor for the range of values that has sense in the geographical area studied. However, precipitation is very sensitive factor, especially in the more humid areas of the watershed. Generally, evapotranspiration is less sensitive than precipitation but there are some sub-watersheds where its sensitivity is significant, because of the correlation between precipitation and evapotranspiration (revealed by the sensitivity analysis).

Due to the fact that climate change projections in Mediterranean regions like the basin studied, are associated with more extreme climatic conditions, the results from this sensitivity analysis further suggest that this changes in climate projected will affect to the water provisioning service and its benefits associated.

The sensitivity analysis of the InVEST erosion control model identifies the climate-driven model parameters (rainfall erosivity and soil erodibility factors) as the most influential ones for sediment export and retention, and other
parameters such as cover management or support practice as playing a secondary but significant role.

Accordingly, small changes in the magnitude and frequency of extreme rainfall events may cause major changes in sediment dynamics, highlighting the susceptibility of erosion control service to climate change in Mediterranean basins.

Moreover, the results from this sensitivity analysis further suggest that it is feasible to compensate the likely effects of climate change on sediment export in some areas by introducing management practices such as contouring, strip-cropping or terracing, as well as increasing the percentage of soil coverage by vegetation.

Implications from such analyses should, however, be put in perspective of the scope of the model: structural errors in the modelling of sheetwash erosion and the omission of additional sources of sediment may also impact management decisions for erosion control service.

**In-stream processes**

The proposed methodology in order to include the in-stream processes in the InVEST nutrient retention model allows to estimate the retention of total nitrogen (TN), and total phosphorus (TP) from the stream in the basin studied.

In addition, the methodology developed in order to estimate in-stream retention shows that it is possible to include retention in rivers in a way that is similar to how the model currently estimates the retention in land, with a constant retention coefficient in the aquatic pixels.
The results obtained for TN as well as for TP have the same tendency, a small increase in the retention coefficient assigned to the aquatic pixels leads to large decreases in the terrestrial retention coefficients.

Those significant changes are more relevant when there is a substantial difference in the watershed between the terrestrial and the aquatic areas (96% and 4% respectively).

The results for retention in the aquatic pixels in the Llobregat river basin obtained with this methodology are in line with those obtained in recent studies for the same study area (Aguilera et al., 2012; 2013). This simple way of incorporating retention for aquatic pixels in the model is useful as a first approach, when the user has little information and/or data.

In addition, with the inclusion of in-stream retention in the InVEST nutrient retention model, not only the overestimation of the terrestrial retention could be compensated, but the benefits of the ecosystem services of the aquatic area could also be evaluated, increasing the model usefulness for watershed management.
8.2. RECOMMENDATIONS FOR FURTHER RESEARCH

8.2.1. Sensitivity Analysis in InVEST model

The Morris method was chosen to conduct the sensitivity analysis in this work because of its computational efficiency with respect to other methods. All the versions used during the Thesis period had to run in ArcGIS, however, the next generation platform of InVEST is currently being developed (standalone): version 3.0.0 and later versions (Tallis et al., 2013; Sharp et al., 2014). This new generation of InVEST models runs without ArcGIS and offer runtime performances improvements to for all the models. These faster models open the possibility of using other methods for sensitivity analysis impossible to use when InVEST runs in ArcGIS.

The sensitivity analysis performance in this work implied modifications of the InVEST source code (written in Python) in order to select automatically the thousands of input combinations needed for the sensitivity analysis. However, the calculation of those different combinations of input variables and the calculation of the Morris indices were executed using code written in the FORTRAN programming language. The implementation of a module for calculating in InVEST all of it allows an optimal sensitivity analysis for the tool users.

Particularly, in the sensitivity analysis of the InVEST erosion control model, the model's structural errors were not investigated. Further studies may thus be necessary to gain confidence in the implications
drawn from our sensitivity analyses. For example, the implications for practice implicitly assume that the model outputs are the sediment exported as observed in the stream. As highlighted in the methods, the contribution from additional sources of sediments such as gullies and bank erosion may have an effect on the results presented here.

Notice that the performance of sensitivity analysis was assessed without taking into account spatial distribution of the LULC.

For the InVEST water provisioning model, the plant evapotranspiration coefficient for each LULC class parameter should be taken into account in the sensitivity analysis, studying in this way the spatial sensitivity, evaluating the sensitivity of the parameter with changes in the land uses.

For the InVEST erosion control model, the model structure implies that most of the exported sediment is mainly from the riparian zone.

While this observation is common in the literature, the relative contribution of the riparian zone LULC may be under- or over-estimated, which would affect the land-use dependent relationships derived in our study. Preliminary investigations were conducted to assess the relationship between sensitivity and proportion of LULC in the riparian zone, but did not result in higher correlations than those reported here. Although it was not in the scope of this study, further analysis of the assumptions underlying the erosion control model should help to improve model’s interpretation.

In addition, the sensitivity analysis in this study was carried out with inputs for current conditions; however, these results may be different
under different conditions. In this way, the InVEST water provisioning model and the InVEST erosion control model were evaluated in the Llobregat river basin under different future climate change scenarios and the impacts of climate change those ecosystem services was estimated by Bangash et al. (2013). Therefore, a further research could be to evaluate the sensitivity analysis in different future global change scenarios.

8.2.2. In-stream processes in InVEST model

The results of the tests performed with the InVEST erosion control model are consistent with the relevance of in-stream processes in the retention and transport of nutrients highlighted in recent studies in the Llobregat river basin.

However, these tests are made with scenarios calibrated only for a single point (at the basin outlet). Therefore, after obtaining results with these tests, scenarios generated with a multipoint calibration is recommended in sensitivity analysis performed in this work required modifications of the InVEST source code in order to get more confident results.

The tests could benefit from further improvements to check different retention coefficients for the different aquatic ‘land uses’ (dam, reservoir and river), furthermore, a differentiation between stream sizes based on a hierarchy of tributaries with the Strahler stream order (Strahler, 1957), and an assignation of different value to each stream order retention coefficient.
9. BIBLIOGRAPHY


10. APPENDICES
APPENDIX A:
REFERENCES USED TO COMPILE THE BIBLIOGRAPHICAL DATA FOR RETENTION IN RIVER (TOTAL NITROGEN). SOURCE AGUILERA ET AL., 2013


APPENDIX B:
REFERENCES USED TO COMPILE THE BIBLIOGRAPHICAL DATA FOR RETENTION IN RIVER (TOTAL PHOSPHORUS). SOURCE AGUILERA ET AL., 2013


APPENDIX C:

PAPERS PUBLISHED DURING THE THESIS PERIOD


Sensitivity analysis of ecosystem service valuation in a Mediterranean watershed

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HIGHLIGHTS

► The Morris sensitivity indexes of InVEST model applied in a Mediterranean basin are quantified.
► The water provisioning spatial distribution sensitivities to main input parameters have been obtained.
► This service is very sensitive to higher precipitation and lower evapotranspiration, and vice versa.
► The seasonal precipitation distribution parameter is much less important than the other input parameters for this area.
► It is noticed that the plant evapotranspiration coefficient for each LULC class is a potentially very sensitive parameter.

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ABSTRACT

The services of natural ecosystems are clearly very important to our societies. In the last years, efforts to conserve and value ecosystem services have been fomented. By way of illustration, the Natural Capital Project integrates ecosystem services into everyday decision making around the world. This project has developed InVEST (a system for Integrated Valuation of Ecosystem Services and Tradeoffs). The InVEST model is a spatially integrated modelling tool that allows us to predict changes in ecosystem services, biodiversity conservation and commodity production levels. Here, InVEST model is applied to a stakeholder defined scenario of land use/land cover change in a Mediterranean region basin (the Llobregat basin, Catalonia, Spain). Of all InVEST modules and sub modules, only the behaviour of the water provisioning one is investigated in this article. The main novel aspect of this work is the sensitivity analysis (SA) carried out to the InVEST model in order to determine the variability of the model response when the values of three of its main coefficients: Z (seasonal precipitation distribution), prec (annual precipitation) and eto (annual evapotranspiration), change. The SA technique used here is a One At a Time (OAT) screening method known as Morris method, applied over each one of the one hundred and fifty four sub watersheds in which the Llobregat River basin is divided. As a result, this method provides three sensitivity indices for each one of the sub watersheds under consideration, which are mapped to study how they are spatially distributed. From their analysis, the study shows that, in the case under consideration and between the limits considered for each factor, the effect of the Z coefficient on the model response is negligible, while the other two need to be accurately determined in order to obtain precise output variables. The results of this study will be applicable to the others watersheds assessed in the Consolider Scarce Project.

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1. Introduction

The services of natural ecosystems are clearly very important to our societies: we probably could not live without them (see Daily, 1997 and Costanza et al., 1997 for a review of the importance of ecosystem services). In recent years, many efforts to preserve and value ecosystem services have been fomented. Theoretically, if the value of nature were widely recognized by the population and institutions, then this would imply an increase investment in conservation and, as a result, an improvement in human welfare. Practically, there is a great interest in developing the scientific basis, the policy and finance mechanisms, for incorporating natural capital into resource and land use decisions on a large scale. In this regard, the Natural Capital Project integrates ecosystem services into everyday decision making around the world. This project is a partner ship between Stanford University, The Nature Conservancy, and World Wildlife Fund (www.naturalcapitalproject.org) and its main challenge remains often poorly understood of assets included in ecosystems relative...
to other capital forms extensively studied (Heal, 2000; MA, 2005; Mäler et al., 2008). The science of ecosystem services needs to develop fast in order to sustain the efforts that are currently in progress, to provide eco-

system services influence in decision making. In addition, the scientific

community should provide the knowledge and tools to estimate and quantify the necessary investments in the nature. To help focus on this challenge, the Natural Capital Project has developed a computer tool for Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) (Kareiva et al., 2011; Tallis and Polasky, 2011).

In recent years, the need to understand the behaviour of many types of very complex ecosystem processes has led to an extraordinary increase in the development of suitable mathematical models, which are of capital importance in order to estimate how an ecosystem behaves and to predict how it might behave in the future. Commonly, these models are computationally expensive because of the involvement of a large number of variables, parameters and mathematical equations. The problem is that not all processes can accurately describe by simplistic equations, and many of them are modelled only in an approximate way or, have been completely removed. Moreover, not all model input parameters are easily measured and need to be estimated using current knowledge of system processes. These uncertainties in the input parameters lead to uncer-
tainties in the model outputs that need to be quantified in order to inte-
grate them in the decision making process.

Sensitivity analysis (SA) techniques have been widely used to pre-
dict and analyse how the model output (response) depends on the variations in the input parameters (factors). Many ecosystem applica-
tions of SA techniques can be seen in recent papers (see Campolongo and Saltelli, 1997; Confalonieri et al., 2010a, 2010b; Yang, 2011). It is particularly useful for complex simulation tools like regional hydrologic models that require a large number of usually uncertain fac-
tors where it is crucial to identify either the most important or the least relevant parameters.

A large number of previous studies have shown that land use in flu-
cees the hydrologic regime and water quality of streams draining these watersheds. Extensive literature exists concerning hydrologic response associated with land use changes such as forest harvesting, conversions of vegetation on the landscape, and draining of agricul-
tural lands (see Bosch and Hewlett, 1982; Ice and Stednick, 2004; Robinson and Rycroft, 1999).

The water supply is an ecosystem service that contributes to the well being of the society, on one side by satisfying human needs for freshwater provisioning services of irrigation, domestic water, and power generation, among others, and on other aspect, it contributes to human well being through recreation, scenic values, maintenance of fisheries and biodiversity, and ecosystem function. Because of all this, it is necessary to have a deep and detailed knowledge of how land uses and landscape contribute to water provisioning for a better man-
agement of this resource and as a tool in making decisions of landscape changes. Land use/land cover (LULC) can modify hydrologic cycles, influencing regime of evapotranspiration, infiltration and water reten-
tion, and the amount of water available in rivers, streams and ground water sources (Ennaanay, 2006; World Commission on Dams, 2000).

The delivery of hydrological ecosystems services is highly dependent on the characteristics of the watersheds. Climate, LULC and topography, have ruling the guidelines for the provision of services (Brauman et al., 2007). One of the most important shaping factors in Mediterranean ba-
sins is climate, which presents larger extremes than more humid areas.

Increased temperatures and reduced vegetation. The hydrological cycle in the Mediterranean areas will intensify through increase in tempera-
tures, rainfall concentration in shorter periods of the year, and more ex-
tended droughts (Hisdal et al., 2001). Associated human impacts through changes in ecosystem services could include drinking water shortages, increased risk of forest fires, shifts in the distribution of spe-
cies, and agricultural losses, among others (Schöter et al., 2005).

InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) model is a powerful tool to estimate the levels and economic values of these ecosystem services and it has been applied to improve the knowledge and understanding the behaviour of several ecosystems. An illustration of this growing interest is the huge number of studies that apply InVEST to assess different ecosystem services (see Guerry et al., 2012; Haines Young, in press; Nelson et al., 2009 and Polasky et al., 2011). However, there are no details in the literature to the author's knowledge on the relative importance of each input parameter in the different ecosystems studied given by performing a sensitivity analysis on the model. This constitutes a novel aspect of this work. As the Consolider Scarc Project is interested in applying the InVEST model to main Mediterranean basins of the Iberian Peninsula, a careful determi-
nation, not only of the relationships among the model response and its parameters, but also of their spatial variations, is of capital importance.

The InVEST model was applied to the basin of the second longest river in Catalonia (NE Spain), the Llobregat River. This river is one of the main sources of Barcelona drinking water, with a basin that receives large discharges of municipal and industrial wastewater and surface runoff from forest and agricultural areas.

Once InVEST has been tailored to the Mediterranean basin (Terrado et al., in revision), its sensitivity analysis was performed using the Morris method. This is done to assess the importance of each input parameter. The Morris method was selected since it is computationally efficient and very effective in screening a small subset of important fac-
tors among the large number contained in a model.

The integration of a sensitivity analysis module in the InVEST soft-
ware will allow the identification of the most important variables of the model and it will provide insight and guidelines into the potential applicability of the model to other Mediterranean river basins.

2. Background

2.1. The InVEST model

Accoding to recent studies, (Vigerstol and Aukema, 2011) hydrologic tools and ecosystem service tools might be successful for model-
ing freshwater ecosystem services. The first type is more detailed and focuses more on ecosystem service drivers, while the second type provides a more general overview of ecosystem services and is easier for non expert user. In this work, we used an InVEST model ecosys-
tem service tool.

InVEST is the acronym of Integrated Valuation of Ecosystem Services and Tradeoffs and is a modelling tool developed by the Natural Capital Project (Kareiva et al., 2011; Tallis and Polasky, 2011). It consists of a suite of models that use land use (LU) and land cover (LC) patterns to es-
timate the levels and economic values of ecosystem services. The model runs on a gridded map at an annual average time step, and its results can be reported in either biophysical or monetary terms, depending on the needs and the availability of information. InVEST includes models for quantifying, mapping, and valuing the benefits provided by terrestrial, freshwater, and marine systems. This work focuses only on the water pro-
visioning service, included in the InVEST freshwater module. This service brings information about the total amount of water available in a basin, taking into account the inputs and outputs of freshwater, and other phenomena like evapotranspiration.

The biophysical models of InVEST calculate the relative contribution of the different parts of the landscape to the provision of services. Thus, for the water provisioning service, the amount of water provisioned
from each cell in the landscape (water yield) is calculated as the annual amount of rainfall that does not evaporate, determined by the cell vegetation characteristics (Canadell et al., 1996). Water demands for consumptive uses other than those evaluated are removed from the total yield before assessing the benefit.

InVEST 2.2.2. Beta used here was the latest version available during the investigation period. In this version of the InVEST model, the water provisioning service has three steps referred to as: water yield, water scarcity, and valuation in terms of hydropower energy. The first step (water yield) uses data on average annual precipitation, annual reference evapotranspiration and a correction factor for vegetation type, soil depth, plant available water content, land use and land cover, root depth and elevation. The second step adds saturated hydraulic conductivity and consumptive water use to determine water scarcity. The valuation model (third step) uses data on hydropower market value and production costs, the remaining lifetime of the reservoir, and a discount rate. The biophysical models do not consider surface ground water interactions or the temporal dimension of water supply. In terms of sensitivity analysis, we have studied the first two steps (water yield and water scarcity) (Table 1).

In the InVEST model version used, the annual water yield ($Y_{xj}$) for each pixel on the landscape ($x=1,2,...,X$) is calculated as it is shown below:

$$Y_{xj} = \left(1 - \frac{AET_{xj}}{P_x}\right)P_x \quad (1.1)$$

where $AET_{xj}$ is the actual evapotranspiration (annual) on pixel x for LULC j (LULC class code; e.g., 1 for forest, 3 for grassland, etc.), and $P_x$ is the annual precipitation on pixel x.

The approximation of the Budyko curve developed by Zhang et al. (2001) is used to calculate the evapotranspiration partition of the water balance ($\frac{AET_{xj}}{P_x}$) as follows:

$$\frac{AET_{xj}}{P_x} = \frac{1 + w_xR_{xj}}{1 + w_xR_{xj} + \frac{1}{R_{xj}}} \quad (1.2)$$

$R_{xj}$ is the Budyko dryness index on pixel x for LULC j (Budyko, 1974) and values greater than 1 mean pixels that are potentially arid (Arora, 2001; Budyko, 1974). This dimensionless index is defined by:

$$R_{xj} = \frac{k_{xj}ETo_x}{P_x} \quad (1.3)$$

where ETo$_x$ is the reference evapotranspiration from pixel x and $k_{xj}$ is the plant (vegetation) evapotranspiration coefficient associated with the LULC$_j$ on pixel x. ETo$_x$ represents an index of climatic demand and $k_{xj}$ is determined by the characteristics of vegetative characteristics in each x (Allen et al., 1998).

$w_x$ is the plant available water coefficient on pixel x. This dimensionless coefficient represents the relative difference in the way plants use soil water for transpiration (Zhang et al., 2001). It can be estimated as:

$$w_x = Z \frac{AWC_x}{P_x} \quad (1.4)$$

where Z coefficient is a seasonality factor that presents the seasonal rainfall distribution and rainfall depths (see Zhang and McFarlane, 1995; Zhang et al., 2001, 2004; Milly, 1994) with values between 1 and 10. The z coef will approach in areas of winter rains 10, whereas in humid areas with rain events distributed throughout the year or regions with summer rains it will approach 1. AWC$_x$ is the volumetric (mm) plant available water content on pixel x.

Afterwards, the water scarcity value is calculated based on water yield and water consumptive use in the watershed of interest, as their difference.

2.2. Sensitivity analysis (SA) methodology

In this section a large and/or computationally expensive numerical model will be considered:

$$y = f(x_1, x_2, ..., x_k) \quad (2.1)$$

In Eq. (2.1), the $x$s (input variables or factors) are defined in the region $\Omega$ of $\mathbb{R}^n$ and $y$ is named the model response. The objective of sensitivity analysis (SA) technique is to quantify the influence of input factors on the model response. Normally, sensitivity analysis techniques enable the exploration of the entire interval of definition for each input factor and do not require any assumptions on the model’s nature (such as linearity or additivity). This study presents the application of a sensitivity analysis method to the InVEST model tailored to the conditions of a Mediterranean basin in the Iberian Peninsula.

As all SA methods, the goal of the Morris screening method is to determine by means of a number r, as small as possible, of model runs the relative influence of each $x_i$ on $y$. The main idea of this method (see Morris, 1991) is to use for this purpose the value of $\frac{\partial y}{\partial x_i}$, what is called i th elemental effect (EE$_i$), employing an OAT (One factor At a
Time) type calculation scheme that fixes at each step all input variables except the one which effect is being calculated.

As cited in Patelli et al. (2010), as the variables are usually differently scaled, taking into account only the $\frac{\partial y}{\partial x_i}$ value could not be very informative, so an available practice is the normalization of the derivatives by the standard deviation of factors, $\sigma_i$, divided by the standard deviation of response, $\sigma_y$:

$$EE_i^N = \frac{\sigma_i}{\sigma_y} EE_i$$  \hspace{1cm} (2.2)

where superscript $N$ denotes 'normalized value'.

If $x$ is considered to be a random variable whose sample space is $\Omega$, each elemental effect, $EE_i$, will be a random variable whose probability distribution function is denoted by $F_i$ and $\mu_i$ and $\sigma_i$ will be the mean and standard deviation of $F_i$ respectively. If $\mu_i$ is small, then the influence of $x_i$ on $y$ will be small too, while high values of $\sigma_i$ suggest the existence of nonlinear effects and/or interaction with other input factors (Morris, 1991).

The practical problem is that, due to the impossibility (or great complexity) to obtain analytically the $\frac{\partial y}{\partial x_i}$ value, the determination of each $EE_i$ must be performed using a numerical approximation:

$$EE_i = \frac{f(x_1,x_2,\ldots,x_{i-1},x_i + \Delta,x_{i+1},\ldots,x_n) - f(x_1,x_2,\ldots,x_n)}{\Delta}$$ \hspace{1cm} (2.3)

Fig. 1. Map of the Llobregat catchment in northeast of Spain, with Llobregat River, urban land and rural land areas.
where $\Delta$ is a real number and $x=(x_1, x_2, \ldots, x_k)$ is any point of the sample space, $\Omega$, such that $(x_1, x_2, \ldots, x_{i-1}, x_i+\Delta, x_{i+1}, \ldots, x_k)$ is still in $\Omega$. This involves working on a discretized version of the above presented concepts. Following the original Morris strategy (Morris, 1991), it must be considered that each $x_i$ is scaled to take values in the interval $[0, 1]$, i.e., $\Omega$ is the unit hypercube, and a discretized approach to a regular k dimensional $p$ level grid, named region of interest and denoted by $\omega$, must be performed. Then, each $x_i$ may take on values from the set $\{0,1/(p-1),2/(p-1),\ldots,1\}$. As is justified in the cited article, it is convenient to select an even value for $p$ and $\Delta=p/[2(p-1)]$.

Obviously, in the simplest form, to obtain a $r$ size sample of the $k$ factors it will be necessary to perform $2kr$ model runs. However, less computationally expensive sampling plans have been developed for this purpose (Campolongo et al., 2007; 2011; Morris, 1991). In this article the original Morris strategy (Morris, 1991) will be used. This strategy has the advantage that, although it does not guarantee equal probability sampling from each $\Omega$, at least a certain symmetric treatment of inputs is ensured, which may be desirable in the analysis of complex simulation models (Campolongo et al., 2011; Morris, 1991). The Morris sampling plan is based on the concept of trajectory which is a set of $k+1$ points of $\omega$ such that two consecutive points of the trajectory differ only in their $i$th component and allow to obtain a $r$ size sample of the $k$ factors at a total cost of $r(k+1)$ model runs.

In Campolongo et al. (2007), a different sensitivity measure $\mu_r$ was defined:

$$
\mu_r = \frac{1}{r} \sum_{i=1}^{r} |E_i|
$$

(2.4)

![Fig. 2. Map of average precipitation distribution (1951–2000) in Llobregat River basin (northeast of Spain).](image)
The use of this 'mean of absolute values' instead of the 'conventional media' is justified in order to avoid the effect of cancellation of sum mands with similar absolute values but different signs that can occur when using the simple average as a measure of sensitivity. The drawback is, of course, the loss of information on the sign of the effect.

3. Application of the Morris method to the water provisioning service of InVEST model

3.1. Case study

The case study is the Llobregat River basin situated in the northeast of Spain. The Llobregat River basin (Catalonia, NE Spain) covers an area of 4950 km². The river, which is 156.5 km long, has its headwaters in the Pyrenees Mountains and flows southward into the Mediterranean Sea near the city of Barcelona. It is the main water source for Barcelona and its metropolitan area (more than 3 million people). Climate in the Llobregat River basin is Mediterranean with strong seasonal fluctuations in temperature and rainfall, presenting peak rainfall events in spring (March-June) and autumn (September-December). Annual rainfall varies substantially within the basin from more than 1000 mm in the Pyrenees to less than 600 mm near the coast. Three large reservoirs are located in the upper part of the basin: La Baells (115 × 10⁶ m³), Sant Ponç (24 × 10⁶ m³), and La Llosa del Cavall (80 × 10⁶ m³). For this analysis, the watershed has been divided in 154 sub-basins that were made at water body scale, as defined by ACA, the local water agency, based on

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**Fig. 3.** Map of average evapotranspiration distribution [1951–2000] in Llobregat River basin (northeast of Spain).
Table 2: Factors' range of variation.

<table>
<thead>
<tr>
<th></th>
<th>$z_{cof}$</th>
<th>prec_coef</th>
<th>eto_coef</th>
</tr>
</thead>
<tbody>
<tr>
<td>Minimum value</td>
<td>7.00</td>
<td>0.10</td>
<td>0.10</td>
</tr>
<tr>
<td>Maximum value</td>
<td>9.00</td>
<td>2.00</td>
<td>2.00</td>
</tr>
</tbody>
</table>

the specifications of the Water Framework Directive (WFD 2000/60/EC) (Fig. 1).

Ninyerola et al. (2000) developed and applied in Catalonia (north east Spain) an empirical methodology for modelling and mapping the air temperature and total precipitation by means of geographical information systems (GIS) techniques. Average conditions of precipitation and evapotranspiration in the Llobregat basin were obtained from Ninyerola et al.'s (2000) model for the period 1951–2000, and mapping results are shown in Figs. 2 and 3.

A limitation detected is that the InVEST makes the modelling with an annual average time step, however, precipitation regime in the case study is characterized by high variability in both the spatial and temporal domains (Esteban Farra et al., 1998; Trigo and Palutikof, 2001), so that making a generalization for one year entails loss of information.

All the other data used in this work are based on Terrado et al. (in revision) that implemented the InVEST model to the Llobregat River basin in order to study the impact of the climatic extremes on hydrological ecosystem services in a strongly anthropized Mediterranean basin.

In the first stage of the study, a new calibration phase of the model was developed in order to check that the adaptation of InVEST model
worked correctly once the changes in the code, explained in Sections 2.1 and 3.2, were made. This calibration was performed to verify that the results were comparable to those obtained before by Terrado et al. (in revision).

3.2. Overview of SA Morris method application

The sensitivity analysis methodology is explained in Section 2.2. As stated in that section, the Morris method is used, and its application is based on multiple steps to be taken as given below:

1. Selecting the output parameter(s) that should be analysed. Determine the input parameters that we expect to affect this output(s) and define their ranges. Morris method then samples the joint input space to provide a set of n input value combinations.

2. The model is run for each of these n combinations, and the output values for each combination are obtained.
3. The 'mean of absolute values' (µ*) of Campolongo et al. (2007) and the standard deviation (σ) are calculated for each output.
4. For each output, analyse the results achieved with this SA method.

The work that has been done to implement this methodology in the InVEST for our case study is described below (Section 3.3).

3.3. Implementation of the SA in the InVEST model

The water provisioning service in the InVEST model generates various outputs in each of the steps that form it. As stated above (in Section 2.1) we have studied the first two steps in terms of sensitivity analysis. For those steps, the main outputs obtained were water yield,
in three different ways (wyield_vol, wyield_ha and wyield_mm), and water scarcity, in two ways (rsupply_vol and rsupply_mm).

In respect of water yield outputs, wyield_vol is the total water yield per sub watershed, namely the approximate absolute annual water yield across the landscape, calculated as the difference between precipitation and actual evapotranspiration on each land parcel, given in m³; wyield_ha is the water yield volume per hectare per sub watershed, given in m³/ha and wyield_mm is the mean water yield per sub watershed, given in mm.

With regard to water scarcity outputs, rsupply_vol is the realized water supply (water yield—consumption) volume for each sub watershed, given in m³ and rsupply_mm is the mean realized water supply (water yield—consumption) per sub watershed, given in m³/ha.

As for the different outputs, the sensitivity analysis for the model was carried out with respect to wyield_vol results. It is unnecessary to perform this analysis with wyield_ha and wyield_mm outputs since both of them could be calculated from wyield_vol, and consequently the sensitivity analysis results would be the same as for wyield_vol.

Having studied the InVEST model, there are four inputs that may be subject to variation for sensitivity analysis, they are z_coef, prec_coef, eto_coef and etk_coef. In this paper we focus on the non-variated landuse parameters, they are the first three parameters. The last one etk_coef is the plant evapotranspiration coefficient for each LULC class, used to obtain potential evapotranspiration by using plant energy/transpiration characteristics to modify the reference...
evapotranspiration, which is based on alfalfa. It requires a double point of view and it will not be analysed in this study.

After the model inputs have been selected, it is necessary to specify their range of variation, as can be shown in Table 2. These variations in the ranges have been determined in previous InVEST model applications in the study area (Terrado et al., in revision).

In the area of study, z_coef value is estimated to be between 7 and 9 (Féoli et al., 1991). And the precipitation (prec_coef) and evapotranspiration (eto_coef) coefficients will be used to change the inputs of precipitation and evapotranspiration by multiplying, respectively (prec_coef and eto_coef values were varied between 0.1 and 2).

Before using the different combinations of z_coef, prec_coef and eto_coef, several changes were realized in the InVEST version used, to adapt the simulation tool for the subsequent sensitivity analysis. These adaptations are basically focused on modifying the Python script so that InVEST accepts a new input table with these three input variables: z, precipitation and evapotranspiration coefficients, and also it obtains the different output tables for each selected input parameter combinations. These changes have resulted in automatic loops for each group of inputs at every simulation run made by the model.

The algorithm that performs the different input variable combinations was implemented in a FORTRAN code. This algorithm generates the Morris’ trajectories having into account the boundaries of the factors. Once these modifications were performed, the model was run for each of the z_coef, prec_coef and eto_coef combinations. Thereafter, the InVEST model was run for each combination of parameters being.

![Map of the study area with Standard Deviation and Morris index of wyield_vol over z_coef](Fig. 7. Sensitivity index σ of z_coef for wyield_vol.)
obtained at the end of the Morris indices $\mu^*$ and $\sigma$, of each model response with respect to each input by sub watershed in the study area and those results achieved were subsequently analysed and compared.

4. Results and discussion

In order to measure the sensitivity of InVEST model responses to the factors, we have studied the Morris sensitivity indices obtained for each sub watershed, corresponding to the responses of interest and the three factors above mentioned. As a result, the Morris indices for each sub watershed and for each factor have been obtained: $\mu^*$ and $\sigma$.

In Figs. 4, 5 and 6, sensitivity index $\mu^*$ obtained for wyield_vol with regard to the three selected input parameters is mapped. In these maps, the more intense the red colour of the sub watershed is, the highest index $\mu^*$, and as a result, this input parameter has a greater influence in the output analysed. Therefore, in terms of $\mu^*$, the greatest impact input parameter is prec_coeff (precipitation) and the least are z_coef and eto_coef (evapotranspiration). Regarding z_coef, in the range of study [7, 9] the observed sensitivity indices indicate that this factor does not influence water yield in the interval [7, 9] tested in this area.

For prec_coeff, Fig. 5 indicates that the most sensitive sub watersheds are found farther north and southeast. In the northern part of the basin, their precipitation is high (between 800 and 1000 mm) and evapotranspiration is quite low (around 400 mm), so the influence and importance of eto_coef are lower, though the model is very sensitive to prec_coeff and high changes in this input cause significant changes in
wyield_vol. In the southeastern part the evapotranspiration is quite high (between 800 and 1000 mm) and rainfall is less than these values (around 600 mm), so variations in precipitation can cause significant changes in the wyield_vol response.

With respect to eto_coef, its influence over the output is quite low, and irrelevant for sub watersheds where prec_coef has a great impact on the output (north and southeast sub watersheds). This relationship observed between the behaviour of prec_coef and eto_coef may also be explained by the standard deviation (σ).

In terms of standard deviation, the results obtained for wyield_vol for to each of the three input parameters considered, and per sub watershed, are represented in the maps shown in Figs. 7, 8 and 9. The higher the standard deviation, the greater the coupling between the variables. In this case there is no coupling of z_coef with the other two variables. However, we observed that a coupling exists between eto_coef and prec_coef, with their correlation stronger in the central region of the watershed. In this region precipitation and evapotranspiration have similar orders of magnitude and small variations in either input parameter have a significant influence on the output.

Furthermore, it can be derived from the $\sigma$ vs. $\mu^*$ graph (see Fig. 10) that wyield_vol is strongly influenced by prec_coef and, to a lesser extent, by eto_coef. In this context, the effect of z_coef seems to be entirely negligible. The high values of eto_coef and prec_coef standard deviation, especially in the sub watersheds located in the central part of the Llobregat River basin, suggest the relatively strong importance of the interactions between these two factors that should be the subject of further research.

The results obtained in this study were carried out with inputs for current conditions (temperature and evapotranspiration average of
the last 50 years) but these results may be different in other conditions (future climate change scenarios) therefore it is recommended to be studied.

5. Conclusions

InVEST is a spatially explicit simulation tool consisting of a suite of models that use land use and land cover parameters to estimate the levels and economic values of ecosystem services. The InVEST model runs in a gridded map at an annual average time step, and results can be reported in either biological or monetary terms, depending on the needs and the availability of information. The biophysical models in InVEST calculate the relative contribution of the different parts of the landscape to the provision of each service. Thus, for the water provisioning service, the amount of water provisioned from each cell in the landscape (water yield) is calculated as the annual amount of rainfall that does not evaportranspire, determined by the cell vegetation characteristics (Canadell et al., 1996). Water demands for consumptive uses other than the evaluated are removed from the total yield before assessing the benefit.

In the present work, since an InVEST model had been developed to assess the water provisioning service in the Llobregat River basin a sensitivity analysis methodology was applied to determine which input parameters are more important for the water provisioning service results, all that for the purpose of stressing what model inputs require more precision and need to be monitored.

The obtained results indicate that the $z_{\text{coef}}$ in the geographical area, is not an important factor. However, the precipitation has a high importance (is very influential) especially in the more humid areas of the watershed. Generally, evapotranspiration is less influential than precipitation but there are some sub-watersheds in which it has a higher importance.

In conclusion, to use InVEST model for the watersheds similar to that under study it is strongly advised to follow the steps: 1) use the precipitation map that best suits the region under consideration, due to the fact that this input becomes crucial as a result of the sensitivity analysis performed. Note that a poor fit in this input would imply non realistic results; 2) fit precisely the evapotranspiration map, as the interaction of this factor with precipitation could have a lot of importance in some particular areas; and 3) select a range of values for $z_{\text{coef}}$ so that it makes sense with the characteristics in the distribution of rain for the region of interest and choose any value within this range, as it will give the same output.

Finally, it needs to be noted that the present work is included inside the SCARCE project (Consolider Ingenio 2010 CSD2009 00065). As a result, the SA methodology here implemented would be extended to all other ecosystem service modules of the InVEST model. And indeed this know how can be used in all the basins submitted on the SCARCE project as well as could be carried out in Mediterranean basin anywhere else.

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References


Haines-Young, R, Poschkin, M, And Kienast, F. Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs. Ecological Indicators in press.


Polasky S, Nelson E, Pennington D, Johnson KA. The impact of land-use change on ecosystem services, biodiversity and returns to landowners: a case study in the State of Minnesota. Environ Resour Econ 2011;48:219–42.


Ecosystem services in Mediterranean river basin: Climate change impact on water provisioning and erosion control

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HIGHLIGHTS

• Mediterranean hydrological ecosystem services (HES) are threatened by climate change.
• Provisioning (water) services are expected to decrease between 3 and 49%.
• Regulating (erosion control) services are expected to decrease between 5 and 43%.
• Pyrenees mountains are a significant contributor in Llobregat basin’s water yield.
• The mean sediment retention is decreasing from upper to lower part of the basin.

ABSTRACT

The Mediterranean basin is considered one of the most vulnerable regions of the world to climate change and such changes impact the capacity of ecosystems to provide goods and services to human society. The predicted future scenarios for this region present an increased frequency of floods and extended droughts, especially at the Iberian Peninsula. This paper evaluates the impacts of climate change on the water provisioning and erosion control services in the densely populated Mediterranean Llobregat river basin of. The assessment of ecosystem services and their mapping at the basin scale identify the current pressures on the river basin including the source area in the Pyrenees Mountains. Drinking water provisioning is expected to decrease between 3 and 49%, while total hydropower production will decrease between 5 and 43%. Erosion control will be reduced by up to 23%, indicating that costs for dredging the reservoirs as well as for treating drinking water will also increase. Based on these data, the concept for an appropriate quantification and related spatial visualization of ecosystem service is elaborated and discussed.

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1. Introduction

Climate change has the potential to cause negative changes in ecosystems and their associated ecosystem services. Climate change is expected to bring several changes in global temperature and rainfall patterns, which are likely to present deep impacts on water availability and quality (Marcé et al., 2010; López Moreno et al., 2011). Indeed, the Mediterranean basin is considered one of the most vulnerable regions of the world to climate change (Schröter et al., 2005) and with a high potential to present important problems in water scarcity in the next few years. Under climate change, rainfall and runoff are expected to decrease in the Mediterranean area (Milly et al., 2005), as well as river flow (López Moreno et al., 2011). Many research projects and several environmental assessments are currently addressing these concerns at all relevant scales, frequently in multidisciplinary collaborations. However, integrating this wealth of information across disciplines remains a considerable challenge (Millennium Ecosystem Assessment, 2003).

Ecosystem services are the benefits that people derive from nature. These include provisioning services such as food and water, regulating services such as flood and disease control, cultural services such as spiritual, recreational, and cultural benefits, and supporting services, such as nutrient cycling, that maintain the conditions for life on Earth (MEA, 2003). Many ecosystem services are derived from freshwater and are commonly referred to as hydrological ecosystem services (HES). These benefits provided by ecosystems in clude provisioning services such as water supply for drinking, power
production, industrial use and irrigation, as well as regulating services such as water purification and erosion control (de Groot et al., 2010). The provision of HES in the Mediterranean basin is likely to be impacted, as climate is one of the major shaping factors in semi-arid basins, which present larger extremes than more humid areas. Previous studies on the Llobregat basin (Catalonia, NE Spain) indicate that impacts on the delivery of services are especially important during dry conditions (Terrado et al., in press). Hence, the application of future climate change predictions that consider the changes to rainfall and temperature patterns is essential to indicate the possible impacts on ecosystem services provision at the Mediterranean basin.

As a consequence of acute water shortages in the Mediterranean basin, competition for water and water stress are likely to increase, especially in summer (Falloon and Betts, 2010). Changes in sediment retention are also expected. According to the IPCC (2007), in southern Europe, runoff will decrease by up to 23% by the 2020s and from 6 to 36% by the 2070s. The projected changes in annual river basin discharge by the 2020s are likely to be affected by climate change. These estimations are based on global rather than regional climate models and a high uncertainty is related to those models. However, regionalized GCMs developed by CEDEX (2011) also projected runoff reduction (below 15%) in Mediterranean river basins.

In this work we used a conceptual framework that focuses on quantifying the benefits associated with changes in ecosystem services as a result of climate change, through a comparison of two climate change scenarios against a base scenario. This approach is in line with the emerging consensus about the importance of comparing alternative scenarios rather than a static analysis of current service provision (Nelson et al., 2009; Tallis et al., 2009). Another key feature of the approach adopted here is that it is spatially explicit, reflecting the fact that both the production and value of ecosystem services vary spatially (Tallis et al., 2009; Birch et al., 2010). Relatively few previous attempts have been made to analyze the spatial dynamics of ecosystem services in relation to climate change scenarios, although recent progress has been made by the Natural Capital Project and others (Nelson et al., 2009; Terrado et al., in press).

In this paper we evaluated the impacts of climate change on provisioning (water) and regulating (erosion control) services, at the Mediterranean Llobregat river basin. When the river basin faces pressures from a high water demand while located in the semi-arid Mediterranean area, it is important to evaluate future scenarios and their effect on these services. Water scarcity can restrict activities dependent on water use, such as human and industrial consumption, electricity production, and agricultural irrigation. Water is already scarce in the region because of its extractive use for industry, human consumption, and agriculture; these activities have also contributed to water quality degradation (Sabater et al., 1987; Terrado et al., 2009; López Doval et al., 2010). We applied Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST), a spatially explicit modeling tool for ecosystem services estimation (Tallis and Polasky, 2011). The model, integrated in a GIS platform, is applied to different future climate scenarios (ranging from 2001 to 2100) and further compared to a base scenario (1971-2000). We hypothesize that the provision of water for different uses, as well as the total sediment exported by the basin will decrease as a consequence of climate change, and the manuscript aims to determine the degree of change in these services, as well as to identify the areas of the basin that are most impacted by these changes. A limitation of this study is that the land use/land cover is considered constant over the whole period of analysis (2001-2100).

2. Methodology

2.1. Study area

The Llobregat basin (NE Spain) drains an area of 4957 km², and is characterized as a typical Mediterranean basin, with a highly variable flow as a result of seasonal rainfall differences. The length of the river is 157 km and has three main reservoirs, Sant Ponç (24 hm³), Llosa de Cavall (80 hm³), and Baells (115 hm³). It is one of the richest and most rapidly developing regions in Spain. The basin is heavily populated (more than 3 million inhabitants) and among one of Barcelona’s major drinking water resources. Heavy anthropogenic pressures, characterized by extensive urban and industrial wastewater discharge as well as diffuse contamination from agricultural areas, are observed at the basin. Main watercourses are regulated by three large dams that impound around 35% of the basin’s mean annual runoff.

2.2. Climate change scenarios data

The data applied in this study is acquired after a downsampling exercise performed by the Catalan Meteorological Service (2012). The typical resolution of general circulation models (between 100 and 300 km) is unsuitable for an evaluation at the basin scale, especially for Mediterranean regions with a complex orography such as Catalonia, which is influenced by polar and tropical air masses. The data provided for mean daily precipitation and temperature, compares the predictions of these parameters for the beginning (2001 2030), middle (2031 2070) and end (2071 2100) of the 21st century, with a base scenario (1971 2000). Specifically for the climate change scenarios, the raster resolution was 7.5 km 7.5 km. The IPCC scenarios selected for evaluation of Llobregat basin are A2 and B1 (IPCC, 2000). The A2 scenario describes a very heterogeneous world, with a continuously increasing global population. Under this scenario economic development is primarily regionally oriented and per capita economic growth and technological changes are more fragmented and slower than in other storylines. The B1 storyline and scenario describes a convergent world with the same global population that peaks in mid-century and declines thereafter, with rapid changes in economic structures toward a service and information economy, with reductions in material intensity, and the introduction of clean and resource efficient technologies. The emphasis is on global solutions to economic, social, and environmental sustainability, including improved equity, but without additional climate initiatives. Therefore, scenario A2 considers more severe changes, while scenario B1 considers moderate changes in global climate. These scenarios are calculated for three time spans and compared to a base scenario, as shows Table 1.

2.3. Ecosystem services modelling and mapping

InVEST is a spatially explicit tool to model and map a suite of ecosystem services caused by land cover changes or climate change impacts (Tallis and Polasky, 2011). Model results can be reported in either biophysical or monetary terms, depending on the needs and the availability of information.

The ecosystem services of water provisioning (hydropower production and drinking water availability), and erosion control (dredging and water quality) are evaluated at the annual scale for the Llobregat basin, with a raster resolution of 200 m 200 m. The definition of the subcatchments was made at the water body scale, as defined by the local environmental agency, based on the specifications of the Water Framework Directive (European Council, 2000). Meteorological data are used to calculate annual evapotranspiration (ETo) and the rainfall erosivity index (R) for each scenario. Evapotranspiration was calculated based on the Hamon’s equation (Tallis et al., 2011), as a function of temperature, and then calibrated with values from 2 stations at the Llobregat basin. The rainfall erosivity index (R) was calculated based on the work of Catari and Gallart (2010) that describes the erosivity at Llobregat basin as a function of mean precipitation (P) in summer months and in the other months of the year. The input data required for each model varies and depends on
Table 1: Input raster maps for the evaluated climate change scenarios.

<table>
<thead>
<tr>
<th>Raster</th>
<th>Source</th>
<th>Time span</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature</td>
<td>Elaborated based on data from Catalan Meteorological Service (2012)</td>
<td>IPCC A2 and B1 scenarios</td>
</tr>
<tr>
<td>Rainfall</td>
<td>Elaborated based on data from Catalan Meteorological Service (2012)</td>
<td>IPCC A2 and B1 scenarios</td>
</tr>
<tr>
<td>ETo</td>
<td>Own elaboration, as a function of temperature</td>
<td>IPCC A2 and B1 scenarios</td>
</tr>
<tr>
<td>Erosivity (R)</td>
<td>Own elaboration, as a function of rainfall</td>
<td>IPCC A2 and B1 scenarios</td>
</tr>
</tbody>
</table>

The service to evaluate. Most of the data formats are GIS raster grids, shapefiles and database tables. Input data requirements and outputs of the model for the selected services are given in Table 2. This study evaluates the changes in ecosystem services provision due to variation in the supply that are likely to be affected by climate change. However, increased water demand and population growth are not considered in scenario development.

2.3.1. Water provisioning

InVEST biophysical models calculate the relative contribution of the different parts of the landscape to the provision of services. For the water provisioning service, the amount of water provisioned from each cell in the landscape (water yield) is calculated as the annual amount of rainfall that does not evapotranspire, and determined by the cell vegetation characteristics (Canadell et al., 1996).

The water yield model is based on the Budyko curve (Budyko, 1974) and annual average precipitation. Annual water yield \( Y_{ijx} \) is determined for each pixel on the landscape (indexed by \( x = 1, 2, \ldots \)) as follows:

\[
Y_{ijx} = \left( 1 - \frac{\text{AET}_{ij}}{P_x} \right) \cdot P_x
\]

Where, \( \text{AET}_{ij} \) is the actual evapotranspiration (annual) on pixel \( x \). The evapotranspiration partition of the water balance, \( 1 - \frac{\text{AET}_{ij}}{P_x} \), is an approximation of the Budyko curve developed by Zhang et al. (2001):

\[
\frac{\text{AET}_{ij}}{P_x} = \frac{1 + w_x R_{xj}}{1 + w_x R_{xj} + \text{PAWC}}
\]

where, \( R_{xj} \) is the dimensionless Budyko Dryness index on pixel \( x \) with \( LULC_j \), defined as the ratio of potential evapotranspiration to precipitation (Budyko, 1974) and \( w_x \) is a modified dimensionless ratio of plant accessible water storage to expected precipitation during the year (Zhang et al., 2001). Finally, we define the Budyko dryness index, where \( R_{xj} \) values that are greater than one denote pixels that are potentially arid (Budyko, 1974), as follows:

\[
R_{xj} = \frac{\text{ET}_{xj} \cdot ETo_x}{P_x}
\]

where, \( \text{ET}_{xj} \) is the reference evapotranspiration from pixel \( x \) and \( k_{xj} \) is the plant (vegetation) evapotranspiration coefficient associated with the LULC \( j \) on pixel \( x \). \( ETo_x \) represents an index of climatic demand while \( k_{xj} \) is largely determined by \( x \)’s vegetative characteristics (Allen Wardell et al., 1998). The water yield model script generates outputs in form of total and average water yield at the sub-watershed level. The input data required for such calculations is described in Table 2.

Table 2: Data requirements and outputs for the selected ecosystem services (ES).

<table>
<thead>
<tr>
<th>ES</th>
<th>Step</th>
<th>Data requirements</th>
<th>Process</th>
<th>Output</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water provisioning</td>
<td>Water yield</td>
<td>DEM (m); Land use/land cover (LULC); Effective soil depth (mm); Average annual rainfall (mm); Average annual reference evapotranspiration (mm); Plant available water content; PAWC (fraction [0,1]); Maximum root depth (mm); Evapotranspiration coefficient; Zhang coefficient [0.0, 1]</td>
<td>Calculates cell level yield as difference between rainfall and evapotranspiration</td>
<td>Annual average water yield (mm yr (^{-1}))</td>
</tr>
<tr>
<td>Water scarcity</td>
<td></td>
<td>Consumptive use by LULC (m(^{3}) yr (^{-1}))</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Erosion control</td>
<td>Valuation</td>
<td>Turbine efficiency (fraction [0.01])</td>
<td>Estimates power generated by water available for hydropower</td>
<td>Energy production (KWh yr (^{-1}))</td>
</tr>
<tr>
<td></td>
<td>Soil loss</td>
<td>DEM (m); Land use/land cover (LULC); Rainfall erosivity (R) (MJ mm ha (^{-1}) yr (^{-1})); Soil erodibility (K) (Mg ha (^{-1}) yr (^{-1}))</td>
<td>Calculates sediment retention at each cell using USLE and routing</td>
<td>Annual average sediment retention (Mg yr (^{-1}))</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Sediment retention efficiency (%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Slope threshold (%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Threshold flow accumulation</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Valuation</td>
<td>Reservoir dead volume (m(^{3}))</td>
<td>Subtracts sediment equal to dead volume</td>
<td>Annual average sediment retention to reservoirs (Mg yr (^{-1}))</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Allowed level of suspended solids pollution (kg yr (^{-1}))</td>
<td>Subtracts retention equal to amount of maximum allowed level</td>
<td>Sediment retention for water quality (kg yr (^{-1}))</td>
</tr>
</tbody>
</table>
Water demands for other consumptive uses (agricultural, industrial, forest) are removed from the total yield before assessing the benefit. The amount of water that actually reaches the reservoir for dam d (realized supply) is defined as the difference between total water yield from the watershed and total consumptive use in the watershed.

\[ V_{in} - Y = ud \]  

(4)

where \( ud \) is the total volume of water consumed in the watershed upstream of dam d and \( Y \) is the total water yield from the watershed upstream of dam d.

Hydropower production and available drinking water are the benefits assessed for the water provisioning services. Drinking water constitutes the most important annual consumptive demand of water resources in the Llobregat basin (65%), followed by industry (25%), agriculture (8%) and livestock (2%) (Catalan Water Agency, 2002). The fraction of water available for drinking purposes is calculated as the remaining water fraction after the removal of the demand for other consumptive uses (Catalan Water Agency, 2002) and the regulated environmental flow allocated at the basin outlet (Catalan Government, 2006). To calculate the amount of energy produced by the three assessed hydropower stations, we performed a similar balance, discounting all the consumptive uses at the upstream of reservoirs. Although more than 100 small hydropower plants exist in the Llobregat basin, the lack of information about diversion concessions forced us to use only power stations located in reservoir systems.

2.3.2. Erosion control

The service of erosion control, which is the relative contribution of the different parts of the landscape to sediment retention, is estimated considering the land use patterns that affect sedimentation into down stream reservoirs, which can affect their water capacity and functioning for hydropower generation. Sediment retention service is calculated as the difference from received (from upstream cells) and exported sediment. Eroded soil from each cell is estimated using the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978), while the retained amount of sediment by each cell is a function of the retention coefficients associated to vegetation covers. Only sheet wash erosion was included in the model (no rill inter rill, gully or stream bank erosion were considered), and the benefits of erosion control are calculated upstream of reservoirs for avoided reservoir sedimentation as the evaluated reservoirs are located in the upstream of the river Llobregat, while for water quality the whole basin is considered. We assumed that the landscape of upstream reservoir has a maximum export allowance, the reservoir dead volume, or the volume that, when filled with sediment, affects the reservoir's function. Given that any retention of sediment when fluxes are lower than the dead volume does not provide a reduction in dredging costs, infrastructure maintenance or production potential, this service was evaluated as the difference between total sediment production and maximum soil export allowance.

The InVEST model also calculates the total amount of sediment reaching a point of interest (water quality). In this study, the point of interest was a drinking water treatment plant near the outlet of the river basin. A threshold of total dissolved solids (US EPA, 2009) in drinking water is compared to the annual load of sediments that would arrive to the drinking water treatment plant near the outlet of the basin. These thresholds are applied where we considered that sediment retention is below the drinking water quality standard and does not provide human benefit.

Sensitivity analysis of the input parameters shows that the Zhang coefficient, a floating point value between 1 and 10, corresponding to the seasonal distribution of precipitation, is not an important factor in the geographical area. However, the precipitation is significant, especially in the more humid areas of the watershed (Sánchez Canales et al., 2012). The main idea of the scenario approach is to analyze climate change impacts on selected ecosystem services with the constant land use maps of the river basin. Moreover, the sub watersheds is considered as the spatial unit of analysis to assess the performance of each ecosystem service. Model calibration for the Llobregat basin was performed as described by Terrado et al. (in press).

3. Results and discussion

3.1. Water provisioning

3.1.1. Water yield

Water yield represents the difference between the precipitation and the actual evapotranspiration (AET) in each land parcel. The contribution of each sub watershed to the basin’s total water yield varies along the territory (Fig. 1), with the major contribution from the North of the river basin. The yield of each sub watershed varies between less than 500 m³ ha⁻¹ yr⁻¹ (areas in dark red in Fig. 1) to more than 5000 m³ ha⁻¹ yr⁻¹ (areas in dark blue in Fig. 1), in the mountainous areas located in the Pyrenees (Northern). The Pyrenees area is the most important contributor to the basin’s water yield (Fig. 1). These mountainous areas are functions as regional “water towers” (Viviroli et al., 2007). This area presents hydrological changes that are already affecting inflows into the Pyrenees reservoir in two ways: (i) a reduction in the annual incoming water volume; and (ii) changes in the seasonal distribution of inflow, with a reduction in spring discharge and the earlier occurrence of the annual maximum monthly flow (López Moreno et al., 2008). Changes in the hydrological cycle in the Pyrenees may have significant implications in water availability for different uses in the whole basin. From an ecosystem perspective, the hydrological changes linked to the domestication of the Mediterranean forests have influenced the capacity of the study area to supply hydrologic ecosystem services at a wide range of scales (Williams et al., 2012).

Water yield in the Llobregat basin is expected to be affected at different extent by climate change (Fig. 2). Modest yield improvement is only expected in the southern part of the basin for the scenario B1 (2031 2070). The results of the remaining scenarios show that water yield values per sub watershed are likely to be reduced by up to 60%. The northern part of the basin, which has the most significant contribution to the provision of freshwater to downstream areas, will be highly affected in the two scenarios (A2 and B1), especially for the time span 2071 2100.

The total annual water volume at the outlet of the basin is represented by the difference between the yield (hm³) and consumptive uses (hm³) of water and expected to decrease in all evaluated scenarios (Table 3). Significant reductions (between 42 and 69%) are observed for the scenarios of time span 2071 2100. These changes are a consequence of a reduced capacity of the river basin in terms of water yield. To increase the water yield, small reservoirs can significantly impact the hydrological regime of river basins (Wisser et al., 2010). Rainfall and AET are the main drivers of change in water yield along with land use changes. But the land use changes are not evaluated in this work.

The different degrees of rainfall reductions in the future scenarios are directly related to the reduced water yields for the respective scenarios. In a similar way, changes in AET are directly linked to the changes in mean annual temperature. In this case, an increase is observed for 3 scenarios: A2 (2031 2070) and B1 (2031 2070 and 2071 2100). Rainfall decreases and AET increases are likely to impact negatively on water yield in the river basin. Another study of the basin (Sánchez Canales et al., 2012) also observed that coupling exists between evapotranspiration coefficient and precipitation coefficient, with their correlation stronger in the central region of the watershed. Water yields were found to be most affected by Scenario A2 (2071 2100), with a yield reduction of 42%. Although reductions are more severe for the A2 scenario, a notable decrease in water yield is also expected for the moderate scenario (B1), with an annual yield reduction between 3 and 26%.
An increase in forest area is observed around 14% between 1957 and 1993, as a consequence of improvement of forest from sparse to dense covers, and the change of land use from agriculture to pasture and forestry (Gallart et al., 2011). However, forest management is considered far less effective for water resource management than dam reservoir development (Komatsu et al., 2010). The management...
of land cover with the aim of improving the availability of water resources is a complex task, as the goal is to obtain higher runoff coefficients while maintaining slope stability, ensuring low erosion rates, reducing reservoir siltation, and mitigating the risks associated with flood events (Beguería et al., 2006; López Moreno et al., 2006). Water demand in the Llobregat region is increasing over time, especially in the low lands for agricultural purposes. To deal with this problem, an appropriate understanding of the importance of land cover management for water resource availability is needed, as has been confirmed in other studies of the basin (Beguería et al., 2003; López Moreno et al., 2006). Trade-off analysis is needed to optimize river basin management, assuring the provision of key ecosystem services.

3.1.2. Water scarcity

Water yield reduction impacts negatively the basin capacity to provide drinking water and energy through hydropower production (Tables 3 and 4). Under climate change analysis, a reduction between 17 and 49% is observed for the drinking water service (Table 3), with the exception of scenario B1 (2031–2070), where an increase of 2% in rainfall leads to a reduction of only 3% in drinking water volume. For the base scenario, the Llobregat basin supplies 842 hm³ yr⁻¹ of water that could be used for drinking purposes, while the real drinking water demand in the basin is around 300 hm³ yr⁻¹ (Catalan Water Agency, 2002). The drinking water demand may also increase in the future as a consequence of land use change and population increase (Pouget et al., 2012), the factors that were not considered in this study. Differences encountered in the hydrologic ecosystem services provisioning capacity are intimately linked with the vegetation cover and management of the territory (Willaarts et al., 2012). However, the globe’s most vulnerable regions are in need of more detailed analysis and the relative importance of population growth versus climate change in altering future freshwater supplies and future per capita water availability may be more a function of population change than climate change (Parish et al., 2012). A 2003 study by the World Resources Institute (WRI, 2003) concluded that 48% of the world’s projected population (~3.5 billion people) will live in water-stressed river basins by 2025.

Water provisioning for drinking use is already an environmental problem in the Llobregat river basin, and water shortages are observed during drought periods. These water shortage periods are difficult to evaluate at an annual basis, and especially when the evaluation is based on the mean annual climate parameters for large time spans (between 30 and 40 years). More detailed studies are needed to evaluate the probability of having important impacts on water provision as a consequence of extended droughts, as the difficulty of supporting a prolonged drought period with the present consumption rates. Application of efficient water use systems is an important way to reduce water demand in a water scarce region (Zhang et al., 2010). Due to the seasonal variations of the rainfall, efficient capture and retention of precipitation during rainy season and its recycling to water shortage periods as per the water requirements may be one of the best options to increase water availability during dry seasons.

One of the most important issues related to climate change in the basin is water security. Interactions between climate change and population growth are expected to increase water demand, and arid and semiarid regions will face additional challenges of absolute water scarcity (Vörösmarty et al., 2000). Water is a primary input to all goods and services either directly or indirectly. The available water quantity and quality can affect the production of goods and services and thus influence the level of economic activities, especially in quickly transforming societies, from agricultural based towards industrialized and modernizing economies (Guan and Hubacek, 2008). A recent study on the Llobregat basin showed that the total annual water volume had experienced an 80% decrease in dry conditions, while an increase of 160% was observed in wet conditions for the period 1970–2000. Drinking water demand remains approximately constant in the basin unless exceptional water demand restrictions are enforced during prolonged droughts (Trenado et al., in press). These changes could lead to a reduction in drinking water availability to levels much lower than those observed in Table 3. According to these results, future challenges to water infrastructures and associated water challenges will potentially lead to important economic costs in the implementation of response alternatives. The imbalance between water demand and resources induces the pressure and degradation of the water quality. In such a case, the artificial recharge of water table aquifers by water from dams is a credible alternative to improve the hydrodynamic and physicochemical conditions of the groundwater (Bouin and Dhia, 2010). However, the INVEST model is unable to account for deep groundwater recharge and water resource infrastructure that redistributes water flow (Vigerstol and Aukema, 2011).

3.1.3. Hydropower production

Similarly to drinking water availability, a decrease in hydropower production is observed in future scenarios developed for 3 reservoirs (Table 4). This agrees with the predicted reduction of hydropower potentials in Southern and South Eastern Europe (Lehner et al., 2005). For each future scenario, the degree of reduction is computed for all three reservoirs. Two reservoirs (La Llosa del Cavall and Sant Ponç, Fig. 1) are located consecutively in the river basin and provide higher benefits to the region in terms of energy produced. It should be noted that the values reported in Table 4 correspond to the potential energy produced if all the available water is utilized for hydropower generation. However, the power stations are not working continuously and the actual amount of electricity produced is lower. The reduction in hydropower potential combined

<table>
<thead>
<tr>
<th>Period</th>
<th>Scenario</th>
<th>Rainfall (mm yr⁻¹)</th>
<th>Rainfall reduction (%)</th>
<th>AET²</th>
<th>AET reduction ($)</th>
<th>Water yield (hm³ yr⁻¹)</th>
<th>Water yield reduction (%)</th>
<th>Drinking water³</th>
<th>Drinking water reduction ($)</th>
<th>Water volume in the outlet (hm³ yr⁻¹)</th>
<th>Water volume reduction ($)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1971–2000 Base</td>
<td>938.69</td>
<td>-</td>
<td>595.02</td>
<td>-</td>
<td>978.00</td>
<td>-</td>
<td>841.71</td>
<td>-</td>
<td>602.92</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>2001–2030 A1</td>
<td>879.81</td>
<td>6%</td>
<td>587.65</td>
<td>1.30%</td>
<td>833.52</td>
<td>15%</td>
<td>697.23</td>
<td>17%</td>
<td>458.44</td>
<td>24%</td>
<td>-</td>
</tr>
<tr>
<td>2031–2070 A2</td>
<td>878.38</td>
<td>8%</td>
<td>605.49</td>
<td>1.61%</td>
<td>778.01</td>
<td>20%</td>
<td>641.72</td>
<td>24%</td>
<td>402.92</td>
<td>33%</td>
<td>-</td>
</tr>
<tr>
<td>2071–2100 B1</td>
<td>869.12</td>
<td>16%</td>
<td>984.45</td>
<td>0.25%</td>
<td>563.09</td>
<td>42%</td>
<td>426.89</td>
<td>49%</td>
<td>188.00</td>
<td>65%</td>
<td>-</td>
</tr>
<tr>
<td>2071–2100 B1</td>
<td>958.15</td>
<td>3%</td>
<td>625.65</td>
<td>4.99%</td>
<td>809.75</td>
<td>7%</td>
<td>89.72</td>
<td>17%</td>
<td>373.67</td>
<td>5%</td>
<td>315.02</td>
</tr>
</tbody>
</table>

² AET: actual evapotranspiration.
³ Based on the assumption of a constant demand of 136.29 hm³/yr.
with the expected population growth in the basin indicates that either alternative energy sources should be considered, or to construct more dams in the studied area to meet the future demands. However, fact could also lead to different impacts, such as fish biomass and biodiversity losses as a consequence of the barrier to fish migration routes (Ziv et al., 2012). In this regard, strategic analysis is needed to assess which should be the most suitable energy sources for this specific case, avoiding unnecessary risks to ecosystems and environmental services.

3.2. Erosion control

Total sediment retained by the basin was evaluated with two different approaches: (i) considering the sediment thresholds of each reservoir (avoided reservoir sedimentation), to calculate the avoided need of dredging reservoirs; (ii) considering the threshold for suspended solids in the water treatment plant (water quality), to represent the avoided need to remove these solids from drinking water.

It is observed that the benefits from erosion control for water quality range between 1070 and 1390 ton yr$^{-1}$. The higher values of mean sediment retention are observed in the upper part of the basin (Pyrenees mountains area, Fig. 3), and very low levels are observed close to the basin outlet (point 4 in Fig. 3). The estimated sediment export value of base scenario for the Llobregat basin is 1.37 10$^6$ ton yr$^{-1}$ (Table 5). These values are very similar to the levels reported in another study for the same basin (Liquete et al., 2009). Similar trends are observed for the mean sediment retention for dredging service. However, higher sediment retention amounts are observed at La Baells reservoir as compared to other two reservoirs in the basin (Fig. 3).

Similar to water provisioning, the effect of climatic change on soil erosion is also not homogeneous throughout the basin. The analysis of climate change scenarios showed that the sediment retention at upstream of the reservoirs will be reduced in all cases (Fig. 4) but more drastic reductions are expected for the period 2071–2100. Reduction of sediment retention could also affect the water quality at the basin outlet (Fig. 5). Significant impacts are observed in the basin.

![Fig. 3. Mean sediment retained per year on each subwatershed, including sediment retained that originates from upslope subwatersheds as well as sediment that originates on the subwatershed itself. The results represent the sediment retention service for avoided dredging (hydropower plants) and water quality (water treatment plant) regulation.](image-url)

<table>
<thead>
<tr>
<th>Period</th>
<th>Scenario</th>
<th>La Baells</th>
<th>La Llosa del Cavall</th>
<th>SantPonç</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>1971–2000</td>
<td>Base</td>
<td>39,657</td>
<td>17,910</td>
<td>21,412</td>
<td>78,979</td>
</tr>
<tr>
<td>2001–2030</td>
<td>A2</td>
<td>33,997</td>
<td>15,366</td>
<td>18,272</td>
<td>67,635</td>
</tr>
<tr>
<td>2031–2070</td>
<td>A2</td>
<td>32,422</td>
<td>14,325</td>
<td>16,929</td>
<td>63,675</td>
</tr>
<tr>
<td>2071–2100</td>
<td>A2</td>
<td>22,891</td>
<td>10,319</td>
<td>12,130</td>
<td>45,340</td>
</tr>
<tr>
<td>2001–2030</td>
<td>B1</td>
<td>35,008</td>
<td>15,987</td>
<td>18,919</td>
<td>69,914</td>
</tr>
<tr>
<td>2031–2070</td>
<td>B1</td>
<td>37,490</td>
<td>17,237</td>
<td>20,569</td>
<td>75,296</td>
</tr>
<tr>
<td>2071–2100</td>
<td>B1</td>
<td>30,921</td>
<td>13,660</td>
<td>16,082</td>
<td>60,663</td>
</tr>
</tbody>
</table>
The amount of sediments exported (Table 5) is also expected to decrease, agreeing with the predictions reported by Milly et al. (2005) for the same period in the region. For this reason, erosion control results are interpreted considering both sediment retention and exports (Table 5). For all the scenarios, the amount of sediment retained is two orders of magnitude higher than that exported, and most of the sediment produced in the basin is retained by existing vegetation. Results of this study illustrates the same trends as reported by Terrado et al. (in press), and higher export values are observed close to the main stream, while higher retention values are observed far from the stream. As a result of the lower erosion values, sediment retention also decreased in the evaluated periods. The same tendency is observed by López Moreno et al. (2008), revealing that climate trends are leading to more restrictive conditions for runoff generation due to an increase in evapotranspiration and a decrease in rainfall during certain periods of the year. Catalan Water Agency (ACA) has planned several interventions in the basin to ensure the sustainable water supply and improve water quality. For example, river bank plantation at lower and upper parts of the river will minimize the erosion values along with other environmental objectives. However, a quantified analysis of each intervention is required with the perspective of ecosystem services.

At the reservoir level, dam managers are assumed to generate their profits by providing irrigation water and hydroelectric power, which are dependent on the reservoir storage capacity (Lee et al., 2011). The life of a reservoir is reduced significantly when there is high sediment deposition, and periodic sediment removal can recover reservoir storage capacity. The primary controls on temporal change in reservoir sedimentation rates include climate, land use or land cover, geologic materials and internal fluvial system operations (Graf et al., 2010). A study (Shi et al., 2012) shows that soil conservation measures taken in fields effectively reduce on site soil loss and sediment yield. However, off site sediment control measures appear to be much less effective at reducing sediment yield. The sediment delivery processes are complex in nature, and inputs of reservoir sediment may have periods of very different rates of delivery separated by short periods of rapid change. In the case of the Llobregat basin, temporal variation of sedimentation is related to seasonal variations in the rainfall erosivity index, which in turn is a function of the variability in rainfall within a one year period. Rainfall seasonality is considered in the study only to differentiate mean rainfall values for summer months and the other months of the year. As the study is based on yearly data, with mean values for large periods, the existence of isolated rainfall events is not assessed by the model. An important drawback of the model used in the present study is that it runs on an annual basis and seasonal climate variations vary significantly within a year.

### Table 5
Comparison between the amounts of sediment exported and retained per year at the Llobregat basin.

<table>
<thead>
<tr>
<th>Period</th>
<th>Scenario</th>
<th>Sediment exported (ton yr^-1)</th>
<th>Export reduction</th>
<th>Sediment retained (ton yr^-1)</th>
<th>Retention reduction</th>
</tr>
</thead>
<tbody>
<tr>
<td>1971–2000</td>
<td>Base</td>
<td>1.37E + 06</td>
<td>-</td>
<td>2.04E + 08</td>
<td>-</td>
</tr>
<tr>
<td>2001–2030</td>
<td>A2</td>
<td>1.34E + 06</td>
<td>2%</td>
<td>1.98E + 08</td>
<td>3%</td>
</tr>
<tr>
<td>2031–2070</td>
<td>A2</td>
<td>1.27E + 06</td>
<td>8%</td>
<td>1.87E + 08</td>
<td>8%</td>
</tr>
<tr>
<td>2071–2100</td>
<td>A2</td>
<td>1.07E + 06</td>
<td>22%</td>
<td>1.57E + 08</td>
<td>23%</td>
</tr>
<tr>
<td>2001–2030</td>
<td>B1</td>
<td>1.28E + 06</td>
<td>7%</td>
<td>1.90E + 08</td>
<td>7%</td>
</tr>
<tr>
<td>2031–2070</td>
<td>B1</td>
<td>1.39E + 06</td>
<td>1%</td>
<td>2.03E + 08</td>
<td>0%</td>
</tr>
<tr>
<td>2071–2100</td>
<td>B1</td>
<td>1.19E + 06</td>
<td>13%</td>
<td>1.77E + 08</td>
<td>13%</td>
</tr>
</tbody>
</table>

Fig. 4. Climate change impacts on avoided reservoir sedimentation per reservoir for the evaluated future scenarios.
are not assessed. Further studies will consider evaluating these climate change scenarios for the wettest and driest year of each period. These studies would allow evaluating the real impact of extreme future climatic conditions.

4. Conclusions

The evaluation of climate change impacts on ecosystem services provision shows that water provisioning and erosion control are highly sensitive to climate change in the Llobregat river basin. A review study (Gosling, 2013) found a proportionally larger amount of evidence to suggest that ecosystem services are vulnerable to changes in the large scale climate earth system in the Mediterranean region. Services supply and delivery are likely to reduce by significant amounts, indicating that urgent measures must be taken to avoid future water stress in the basin. The sub watersheds from the Pyrenees region are responsible for most of the services provision, and also the most impacted areas regarding climate change. Interventions to enhance the provision of regulating services should focus in certain areas where obtained benefits per surface area are estimated to be the highest. For the protection of these areas, interventions such as restoration and measures suitable for increasing or maintaining resilience in rivers are essential to assure future water use in the basin. The groundwater surface water interplay and the temporal nature of water demands in the Mediterranean region also lend complexity to the system (Bangash et al., 2012). The aim of the study was the detection of change in trends over time and the quantification of ecosystem service provisioning under climate change impact. The results show clear trends over time, with decreases in water yield and the amounts of sediment retained being two orders of magnitude higher than that exported. Climate change the only variable and driving force considered in this study and other important drivers of change, land use and land cover and the increase in demand by different sectors are considered constant for the whole catchment. However it is obvious that, the protection of water resources is not sufficient if the levels of consumption continue to increase in the future. Proactive management of basin should be implemented for adapting to climate change as mitigating measures taken in the present may avoid long term future consequences.

Acknowledgments

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References


Sensitivity analysis of a sediment dynamics model applied in a Mediterranean river basin: Global change and management implications

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HIGHLIGHT

• A sensitivity analysis was performed for a sediment model in a Mediterranean basin.
• For sediment retention benefits, the model was sensitive to erosivity and erodibility.
• For sediment export, it was also sensitive to land management factors.
• Management practices may mitigate the impact of climate change on sediment export.
• Sensitivity to physical or management factors varied with the forest cover proportion.

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ABSTRACT

Climate change and land use change are major factors influencing sediment dynamics. Models can be used to better understand sediment production and retention by the landscape, although their interpretation is limited by large uncertainties, including model parameter uncertainties. The uncertainties related to parameter selection may be significant and need to be quantified to improve model interpretation for watershed management. In this study, we performed a sensitivity analysis of the InVEST (Integrated Valuation of Environmental Services and Tradeoffs) sediment retention model in order to determine which model parameters had the greatest influence on model outputs, and therefore require special attention during calibration. The estimation of the sediment loads in this model is based on the Universal Soil Loss Equation (USLE). The sensitivity analysis was performed in the Llobregat basin (NE Iberian Peninsula) for exported and retained sediment, which support two different ecosystem service benefits (avoided reservoir sedimentation and improved water quality). Our analysis identified the model parameters related to the natural environment as the most influential for sediment export and retention. Accordingly, small changes in variables such as the magnitude and frequency of extreme rainfall events could cause major changes in sediment dynamics, demonstrating the sensitivity of these dynamics to climate change in Mediterranean basins. Parameters directly related to human activities and decisions (such as cover management factor, C) were also influential, especially for sediment exported. The importance of these human related parameters in the sediment export process suggests that mitigation measures have the potential to at least partially ameliorate climate change driven changes in sediment exportation.

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1. Introduction

Changes in rainfall and land use patterns are severely influencing sediment dynamics (e.g., erosion and retention processes) in river basins (Walling, 2008). Recent studies suggest that this influence is particularly evident in scenarios of environmental land use conflicts, where actual land uses deviate from natural uses determined by soil characteristics (Pacheco et al., 2014; Valle Junior et al., 2014). Because of its sensitivity to global change and the environmental and economic relevance of sediment dynamics, erosion and its impacts receive increasing attention from local, national, European and global policy makers (e.g., European Commission, 2002; COST634, 2005; MA, Millennium Ecosystem Assessment, 2005). From drinking water to hydro power or irrigation canals, there is a growing interest in assessing the sediment retention service provided by natural landscape, to adopt

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watershed management measures that enhance this service (Clark, 1985; MA, Millennium Ecosystem Assessment, 2005; CIRIA, 2013). Policy makers require reliable predictions of how global change will affect erosion and retention processes in order to design effective mitigation or adaptation measures. Therefore, it is crucial to evaluate and minimize uncertainty of the models to avoid bias in decision making (Chavas, 2000; National Research Council, 2005). Russel (1949) has illuminated these circumstances with more optimism: ‘When one admits that nothing is certain one must, I think, also admit that some things are much more nearly certain than others.’ That is an encouraging statement especially for scientists whose basic vocation is to characterize and reduce uncertainty where possible.

Sediment production and transport in river basins is controlled by many factors, including rainfall patterns, soil characteristics, steepness of the hill slope, and the type of vegetation cover. Different types of vegetation decrease soil loss and retain sediments from upslope areas to different degrees (Tallis et al., 2011). In Europe, in particular, the analysis of erosion rates under natural rainfall confirmed the dominant influence of land use and land cover. Generally, the erosion of a land with a permanent vegetation cover (shrubs, grassland and forest) is much lower than those on cultivated land (Cerdan et al., 2010). Soil loss may therefore be mitigated through agricultural best management practices (Bakker et al., 2008), but broader and more dramatic land use changes such as a transition from vegetated to urban areas may have larger effects on the capacity of a landscape to retain sediments from upslope areas. The risks posed by altered sediment dynamics are particularly evident in Mediterranean and other semiarid regions, which are among the most vulnerable areas to global change (Schröter et al., 2005). Since the 1980s, a substantial body of work appeared about the erosion and transport of sediments in different parts of the Llobregat basin located in the northeastern of the Iberian Peninsula (Cloret et al., 1983; Cloret, 1984; Cloret and Gallart, 1988; Llorens et al., 1998; Regiés et al., 2000; Regiés and Gallart, 2004; Farguell and Sala, 2005; Gallart et al., 2005; Catarí, 2007) but none have targeted the basin in its entirety. Sediment dynamics models vary widely in complexity but typically involve a large number of parameters requiring calibration (Merritt et al., 2000; Regués and Gallart, 2004; Farguell and Sala, 2005; Gallart et al., 2005; Catarí, 2007) but none have targeted the basin in its entirety. Sediment dynamics models vary widely in complexity but typically involve a large number of parameters requiring calibration (Merritt et al., 2003). Irrespective of the model complexity, accurate characterization of all physical processes may not always be possible; input parameters are often difficult to measure and therefore need to be estimated. The uncertainties related to parameter selection may therefore be significant and need to be quantified to improve model interpretation for watershed management.

In this study we performed a sensitivity analysis of a simple sediment retention calibrated in a Mediterranean basin, in order to determine the parameters that had the greatest influence on the model outputs, and thus have particular importance for model calibration and interpretation. Such sensitivity analysis techniques have been widely used to reveal the relative importance of different factors in spatially distributed models (Conflaloneri et al., 2010a, 2010b; Yang, 2011; Sánchez Canales et al., 2012). The sensitivity analysis described here was performed for two separate components of the model, predicting exported and retained sediment, respectively, and reflecting two different ecosystem benefits (avoided reservoir sedimentation and higher water quality) (de Groot et al., 2010).

2. Material and methods

2.1. Study region

The Llobregat River basin is located in the NE of the Iberian Peninsula and drains an area of 4950 km². The river, which is 156 km long, arises in the Pyrenean Mountains and flows southward into the Mediterranean Sea near the city of Barcelona. This river is an important water source for Barcelona and its metropolitan area (over 3 million people). The basin is dominated by forest cover (38.2%) and by a mixture of grass and shrubland (31.6%, for more details, see Table 1). The climate is Mediterranean, with strong seasonal fluctuations in temperature and rainfall, and peak rainfall events in spring (March-June) and autumn (September-December). Annual rainfall ranges widely within the river basin, from 400 mm per year in the mid section to 1000 mm in the upper segments (Mujerio, 2006). There are three reservoirs located in the headwaters of the river basin: La Baells (115 · 10⁶ m³), Sant Pons (24 · 10⁶ m³), and La Llosa del Cavall (80 · 10⁶ m³). There are several drinking water treatments plants, the largest being located near the river mouth. In order to analyse spatial sensitivity, the river basin was divided into 7 sub basins, three upstream of the three reservoirs (sub basins numbers 2, 3 and 4, Fig. 1), and the other four covering the rest of the basin (sub basins numbers 1, 5, 6 and 7, Fig. 1).

2.2. Modeling approach

Due to the fact that complex model with more parameters may have even greater uncertainty, we chosen for the sensitivity analysis a simple one as the InVEST sediment retention model.

The InVEST sediment retention model produces spatially explicit outputs at an annual average time scale, see Table 2 for details (Tallis et al., 2011; version 2.4). This model computes the total amount of sediment exported by estimating the average annual sediment generated by each parcel of land, employing a method based on the Universal Soil Loss Equation (USLE) (Tallis et al., 2011; Wischmeier and Smith, 1978) at the pixel scale. The USLE integrates information on land use/
land cover (LULC) patterns, soil properties, topography, rainfall and climate:

\[ \text{USLE} = R \times K \times LS \times C \times P \]

where R is the rainfall erosivity factor (a measure of the intensity and duration of rainfall events), K is the soil erodibility factor (which measures the susceptibility of soil particles to detachment and transport by rainfall and runoff), LS is the slope length gradient factor (determined by the local topography, calculated according to Moore and Burch, 1986), C is the cover management factor (representing the specified crop and management relative to tilled continuous fallow) and P is the support practice factor (representing the effects of contouring or terracing). Among these factors, R, K, and LS are physical factors, while C and P are mainly human driven, depending on local agricultural and erosion control practices. Each pixel’s contribution to the process is calculated using the Universal Soil Loss Equation (USLE) and routing.

**Table 2**
Components and description of the biophysical processes included in the InVEST model of erosion control (Tallis et al., 2011).

<table>
<thead>
<tr>
<th>Step</th>
<th>Inputs</th>
<th>Process</th>
<th>Outputs</th>
</tr>
</thead>
<tbody>
<tr>
<td>Supply</td>
<td>Land use/land cover (LU/LC)</td>
<td>Calculates sediment retention using USLE and routing</td>
<td>Annual average sediment retention</td>
</tr>
<tr>
<td></td>
<td>Rainfall erosivity factor (R)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Soil erodibility factor (K)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Slope threshold (LS)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cover management factor (C)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Support practice factor (P)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>DEM</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Reservoir</td>
<td>Reservoir dead volume</td>
<td>Subtracts sediment equal to dead volume</td>
<td>Annual total sediment retention of value to reservoirs (sed_ret_dr)</td>
</tr>
<tr>
<td>Service</td>
<td>Watersheds above points of interest</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Treatment Plant Service</td>
<td>Allowed sediment load in rivers</td>
<td>Subtracts sediment equal to allowed load</td>
<td>Annual total sediment retention of value to water treatment plants (sed_ret_wq)</td>
</tr>
<tr>
<td></td>
<td>Watersheds above points of interest</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
load of suspended sediment in the river is estimated by first calculating erosion with USLE, and then routing eroded soil down the flowpath and allowing retention by present vegetation.

The model quantifies the total amount of sediment retained by estimating how much soil is retained by vegetation on each pixel, both in terms of the total amount eroded and in terms of captured sediment from upstream. Stream surface water running over a pixel. In other words, the model first computes the amount of sediment exported by a hypothetical bare soil and the actual land cover on that pixel, equivalent to \( RKLS \times (1 - CP) \) (Step Supply, Table 2). Next, the model adds the amount of sediment eroded on all upstream sources trapped by that pixel.

Within this modelling framework, InVEST can quantify the ecosystem services of erosion control, which is the relative contribution of different parts of the landscape to sediment retention. Specifically, the model considers two of the different benefits derived from erosion control: avoided reservoir sedimentation and higher water quality. For the former, we assume that the landscape upstream the reservoirs has a maximum export allowance. The reservoir dead volume represents the volume of sediment that impacts the reservoir function. For the second benefit (higher water quality), we consider the annual load of sediment that would arrive to a drinking water treatment plant and compare it to a threshold value of total suspended solids allowed in drinking water. Due to the lack of a threshold value of total suspended solids in the European legislation (DWD, 1998), we use the threshold value given by US EPA, 2009.

All the calculations are made at a pixel resolution of 200 x 200 m to capture the spatial heterogeneity of key driving factors such as soil type, rainfall and vegetation type. However, the model simulates by drologic processes that are best interpreted at the sub-basin or basin scale, so the pixel values are aggregated (summed and/or averaged) to match these scales for result interpretation and model validation. The model was calibrated by Terrado et al. (2014) at La Baells reservoir (outlet of sub-basin 2) using temporal changes in the reservoir’s bathymetry (Catari et al., 2009). Results of the model were then validated with data reported by Liquete et al. (2009). Despite only accounting for the sheetwash process in the sediment budget (gully or stream bank erosion are ignored), a reasonable estimate of sediment yield was obtained at the basin scale with physical parameters in reasonable ranges (relative error of 7%; see Table 3; Liquete et al., 2009). However, at sub-basin scale in the Upper Llobregat, above La Baells reservoir, the relative error (25%) was higher (see Table 3; Terrado et al., 2009). Despite the lack of a threshold value of total suspended solids in the European legislation (DWD, 1998), we use the threshold value given by US EPA, 2009.

Our sensitivity analysis focuses on three different outputs given by the model: 1) total sediment exported, 2) fraction of total sediment retained which is contributing to higher water quality benefit, and 3) fraction of total sediment retained for avoided reservoir sedimentation benefit. We performed the sensitivity analysis on the model using the initial parameter values from Terrado et al. (2014). We employed the Morris method (Morris, 1991), which is a one at a time (OAT) sensitivity analysis that aims at isolating the influential parameters from a large number of input variables or factors. This method has several advantages over other methods. It is easy to understand, does not depend on assumptions about the model, is computationally inexpensive, the parametric space is covered efficiently, and it shows input interactions and non linear effects. However, this method cannot quantify the contribution of a parameter to the variability of the output in a highly non linear model (Wu et al., 2013). As with all sensitivity analysis methods, the goal of the Morris screening method is to determine the relative influence of each input variable or factor \( \left( x_i \right) \) on the output or response \( \left( y \right) \) through a series of model runs. To achieve this, the method determines the mean \( \left( \mu \right) \) and standard deviation \( \left( \sigma \right) \) of \( \partial y / \partial x_i \) at several points in the sample space employing an OAT type calculation scheme that fixes at each step all input variables except the one whose effect is being calculated. We extended the original Morris approach by applying the ‘mean of absolute values’ \( \left( \mu^* \right) \) defined by Campolongo et al., 2007, to avoid the effect of cancellation of terms with similar absolute values but different signs (see Sánchez Canales et al., 2012 for more details).

Our application of the Morris methodology consists in five steps: (i) examining the calibrated InVEST model to select the inputs that are likely to be affected by global change and define their ranges to modify in each test (Table 4); (ii) implementing the extended Morris approach and calculate the value of \( \partial y / \partial x_i \) employing the OAT type calculation scheme; (iii) obtaining a set of combinations of input variables; (iv) running the calibrated InVEST model for each combination of input variables of factors, obtaining a set of model responses; and (v), calculating \( \mu^* \) and \( \sigma^* \) for each sub-basin and for the whole river basin. The sensitivity analysis algorithm was implemented in Fortran 2003.

Using the equations that describe the InVEST sediment retention model, the inputs inspected in the sensitivity analysis included: rainfall erosivity \( \left( R_e \right) \); soil erodibility \( \left( K \right) \); and land use/land cover (LU/LC), with five LU categories (urban land, non irrigated cultivated land, irrigated cultivated land, grass and shrub land, and forest land). Three inputs were linked to land use type: cover management factor \( \left( C \right) \); the support practice factor \( \left( P \right) \); and a sediment retention value that identifies the capacity of vegetation to retain sediment as a percentage of the amount of sediment flowing into a cell from upslope. We noted that the model requires two additional inputs: the threshold flow accumulation and slope threshold that both depend on the digital elevation model (DEM). Because errors in the DEM were not a focus of this analysis, we used the values from a former study (Terrado et al., 2014). The threshold flow accumulation, set to 1000, produced output consistent with the stream network map provided by the Catalan Water Agency (ACA).

Table 4 contains the ranges of variation of the inputs. The ranges for all the inputs were obtained from the literature (Wischmeier and Smith, 1978; US EPA, 2009; Terrado et al., 2014). The values \( \partial y / \partial x_i \) were normalized by the response standard deviations to avoid scaling artifacts (Patelli et al., 2010). The Morris sampling plan takes an r-sized

---

### Table 3

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Point of interest</th>
<th>Region</th>
<th>Observed</th>
<th>Predicted (Mean)</th>
<th>Relative error (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sediment export</td>
<td>La Baells reservoir</td>
<td>Sub-basin 2</td>
<td>200</td>
<td>150</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>The basin outlet</td>
<td>Whole basin</td>
<td>1428</td>
<td>1535</td>
<td>7</td>
</tr>
</tbody>
</table>

Data sources:

1. Catari et al., 2009.
2. Liquete et al., 2009.
sample of the $k$ factors to be tested, and costs $r(k + 1)$ model runs, if considering a set of $k + 1$ points on the region of interest such that two consecutive points of the sampling differ only by one component.

The sensitivity analysis parameters $\mu$ and $\sigma$ were calculated for each factor with respect to the 3 final outputs (total sediment exported and total sediment retained for both avoided reservoir sedimentation and higher water quality benefits). These parameters were calculated at both sub basin and basin scales. A high value of $\sigma$ suggests the existence of nonlinear effects and/or interaction with other input factors, while a low $\sigma$ indicates independence (Morris, 1991). The value of $\mu$ is an indication of the importance of the factor, with higher values attributed to factors to which the output of the model is most sensitive (Fig. 2).

The relevant parameters obtained for total sediment exported were correlated with the characteristics of each sub basin and with the whole river basin using the Spearman’s rank correlation matrix due to the small sample size and the presence of outliers in the data, invalidating the assumption of normal distribution. To cross check the results, we also computed the Pearson’s $r$ correlation coefficients, which proved to be similar.

![Fig. 2. Sensitivity analysis indices ($\mu$ and $\sigma$) calculated for each factor with respect to the total sediment exported. These indices are calculated for basin (a) and sub-basin scale, plotted results more different, sub-basin 2 (b), and by sub-basin 6 (c). The value of $\mu$ is an indication of the importance of the factor, with higher values attributed to factors to which the output of the model is most sensitive. A high value of $\sigma$ suggests the existence of nonlinear effects and/or interaction with other input factors, while a low $\sigma$ indicates that the factor is almost independent of other factors.](image-url)
3. Results

3.1. Total sediment exported

The total sediment exported was calculated for the seven sub basins as well as for the whole river basin (Fig. 1). The sensitivity of this output with respect to the physical factors R (rainfall erosivity) and K (soil erodibility) was found to be similar for individual sub basins and for the whole basin. Overall, the greatest influence was found for rainfall erosivity, closely followed by soil erodibility. For the rest of the model inputs, differences in sensitivity were apparent, which we hypothesized to be due to differences in sub basin characteristics. Fig. 2 shows the results of the sensitivity analysis performed on two of the most different sub basins (sub basin 2 and 6), as well as for the whole river basin.

The entire basin has mostly natural vegetation with 38.2% of forest and 31.6% grass and shrub land. However, it also has a significant proportion of (non irrigated) cropland (23.6%). The results of the sensitivity analysis highlight that the support practice factor (P) for non irrigated cultivated land and the cover management factor (C) for grass and shrub land have the greatest influence on sediment export, after the R and K factors (Fig. 2a).

Similarly, in sub basin 2 the most influential factors (apart from R and K) are C and P for grass and shrubland (Fig. 2b) that reach values of μ* as high as those for K. This is a high mountain sub basin composed of mostly forest (63.2%), with a wide area of shrubs and grass covers (35.1%), and a reservoir at the outlet (La Baells). Sub basin 6, in contrast, is mainly composed of non irrigated cultivated land (43.8%), with less grass and shrubland (25.5%) and forest (24.9%). The most important factor after R and K for this sub basin is P for non irrigated cultivated land (Fig. 2c).

These observations were confirmed by the correlation analyses. The Spearman’s rank correlation matrix (Table 5) shows the influence of the characteristics of each sub basin, in particular the percentage of land use, on the sensitivity of each parameter. In general, the proportion of non irrigated cultivated land use and forests were significantly correlated with the sensitivity of the C and P factors.

Additional observations can be made from Table 5. In the Llobregat basin, it seems that the higher the mean slope or the mean rainfall, the more important were K, and the C and P factors for grass and shrub land respectively. In contrast, the higher the mean slope or the mean rainfall, the less influential were the management and P factors for non irrigated cultivated land. Interestingly, although input factors related to forested land were not significantly correlated with basin characteristics, the fraction of this LULC in the study area affects the relevance of others important inputs. In fact, the higher the fraction of forested land, the less important C and P become for non irrigated cultivated land, as well as R and K.

To provide some perspective to the qualitative sensitivity analyses, we also provide quantitative insights into the effect of parameter errors. Table 6 reports the results of several pairs of runs from the sensitivity analyses, showing the relative difference in input parameters and the resulting difference in sediment export. For the whole basin, the first example shows the difference in sediment export (−49%) due to the change only in C for non irrigated land use (a decrease of 64%). This result confirms that the model is less sensitive to C parameters than to R and K parameters, to which the model responds proportionally, according to the analysis of the model structure. The three other pairs of runs illustrate how differences in parameters may compensate each other. For example, for the whole basin, we show that an increase of 63% in R may be compensated by high decreases in P for non irrigated and forest land uses (the difference in sediment export is only 0.2%). Similarly, in subbasin 2, an increase in both physical factors (46% for R and 33% for K, respectively) and high decreases in P and C factors result in comparable sediment export (1% difference in sediment export). In subbasin 6, similar results were obtained with an increase of 46% in R and also a high decrease in the most sensitive inputs of human related factors (C and P). These examples illustrate a classic case of equifinality in environmental modelling, whose discussion in outside the scope of this paper. In practical terms, they suggest that changes in some parameters (e.g. rainfall erosivity) may be compensated by others (e.g. management factors), as discussed below.

Table 5

<table>
<thead>
<tr>
<th>Sensitivity analysis indexes of total sediment exported with respect to</th>
<th>Rainfall erosivity factor (R)</th>
<th>Soil erodibility factor (K)</th>
<th>Cover management factor (C)</th>
<th>Support practice factor (P)</th>
<th>Cover management factor (C)</th>
<th>Support practice factor (P)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cultivated non–irrigated land</td>
<td>Grass and shrub land</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Watershed characteristics</td>
<td>μ*</td>
<td>σ</td>
<td>μ*</td>
<td>σ</td>
<td>μ*</td>
<td>σ</td>
</tr>
<tr>
<td>Maximum slope</td>
<td>0.08</td>
<td>0.02</td>
<td>0.50</td>
<td>0.57</td>
<td>0.52</td>
<td>0.52</td>
</tr>
<tr>
<td>Mean slope</td>
<td>−0.37</td>
<td>−0.52</td>
<td>0.64</td>
<td>0.73</td>
<td>−0.83</td>
<td>−0.83</td>
</tr>
<tr>
<td>% urban land (6)</td>
<td>−0.25</td>
<td>0.63</td>
<td>−0.26</td>
<td>−0.39</td>
<td>0.52</td>
<td>0.52</td>
</tr>
<tr>
<td>% cultivated non–irrigated land (1)</td>
<td>0.39</td>
<td>0.73</td>
<td>−0.90</td>
<td>−0.95</td>
<td>0.95</td>
<td>0.95</td>
</tr>
<tr>
<td>% cultivated irrigated land (2)</td>
<td>0.95</td>
<td>0.51</td>
<td>0.95</td>
<td>0.95</td>
<td>0.36</td>
<td>0.36</td>
</tr>
<tr>
<td>% grass and shrub land (3)</td>
<td>−0.08</td>
<td>−0.23</td>
<td>−0.08</td>
<td>0.57</td>
<td>−0.43</td>
<td>−0.43</td>
</tr>
<tr>
<td>% forested land (4)</td>
<td>−0.67</td>
<td>−0.76</td>
<td>−0.67</td>
<td>0.57</td>
<td>−0.71</td>
<td>−0.71</td>
</tr>
<tr>
<td>River length (km)</td>
<td>0.76</td>
<td>0.74</td>
<td>0.76</td>
<td>0.52</td>
<td>0.60</td>
<td>0.60</td>
</tr>
<tr>
<td>River length per surface area</td>
<td>0.68</td>
<td>0.83</td>
<td>−0.62</td>
<td>−0.64</td>
<td>0.69</td>
<td>0.69</td>
</tr>
<tr>
<td>River length per surface area</td>
<td>0.28</td>
<td>0.67</td>
<td>0.28</td>
<td>0.77</td>
<td>0.68</td>
<td>0.68</td>
</tr>
<tr>
<td>Mean rainfall</td>
<td>−0.58</td>
<td>−0.68</td>
<td>0.67</td>
<td>0.78</td>
<td>−0.88</td>
<td>−0.88</td>
</tr>
<tr>
<td>Maximum erosivity</td>
<td>−0.80</td>
<td>−0.38</td>
<td>0.33</td>
<td>0.49</td>
<td>−0.57</td>
<td>−0.57</td>
</tr>
<tr>
<td>Mean erosivity</td>
<td>−0.83</td>
<td>−0.38</td>
<td>0.24</td>
<td>0.38</td>
<td>−0.43</td>
<td>−0.43</td>
</tr>
<tr>
<td>Maximum erodibility</td>
<td>0.12</td>
<td>0.70</td>
<td>−0.57</td>
<td>−0.38</td>
<td>0.43</td>
<td>0.43</td>
</tr>
<tr>
<td>Mean erodibility</td>
<td>−0.23</td>
<td>−0.04</td>
<td>−0.52</td>
<td>−0.30</td>
<td>0.24</td>
<td>0.24</td>
</tr>
</tbody>
</table>
3.2. Total sediment retained

As explained before, the model assesses the erosion control service as the total sediment retained in the basin, i.e. the amount of soil retained by the landscape. The two different benefits calculated for this ecosystem service are higher water quality and avoided reservoir sedimentation. In our study basin, the total sediment retained that could contribute to higher water quality was calculated for the whole basin, since the drinking water treatment plant is located at its outlet. However, the avoided reservoir sedimentation was calculated for sub basins 2, 3, and 4, each of which has a reservoir at its outlet (see Fig. 1 for location of these basins).

In contrast to the complex dependencies seen for sediment export (above), the model for total sediment retention (for both avoided reservoir sedimentation and improved water quality) only appears to be strongly sensitive to the physical factors, R and K. Since the results from the sensitivity analysis shows that the provision of both benefits in all the sub basins and the whole basin are the same, avoided reservoir sedimentation and improved water quality are independent of the ba


4. Discussion

4.1. Management implications

The sensitivity analysis of the InVEST sediment retention model suggests that R and K (rainfall erosivity and soil erodibility) are the most in influential inputs in the Mediterranean river basin under study. It is worth noting that both of these factors are potentially climate driven; R directly and K through the effects of temperature and moisture on the soil’s organic content. The soil erodibility (K) is related to soil organic matter content, which varies with temperature and moisture in a non-linear way, although with high spatial heterogeneity due to its relationship with vegetation cover patterns (Lavee et al., 1998; Sarah, 2006). This implies that in general, the model has greater sensitivity to climate change than anthropogenic land use change.

However, despite the prominent influence of the climate driven factors, other factors directly related to land management played a significant role for total sediment exported. Cover management and support practice factors were relevant for some LULC, depending on the proportion of each LULC type and sub basin slopes. Regional differences in model sensitivity to anthropogenic factors were observed for sub basins differing in LULC composition and mean slope. In high mountain sub basins such as number 2 (Fig. 2b), since the cover management factor for grass and shrublands are as relevant as climate driven factors, it could be possible to mitigate erosion from climate driven changes by promoting species with higher ground cover on this type of land. The results obtained in the sub basin 2 is in agreement with the results reported by Catari, 2007 in the same area (Upper Llobregat, above La Baells reservoir). This thesis work reports that there is an increase in erosion rates for all the scenarios studied incorporating global change, and concludes that the two most sensitive inputs for all these scenarios are both rainfall erosivity and cover management related factors. In contrast, in truly Mediterranean climates such as sub basin number 6 (Fig. 2c), the support practice factor in non irrigated cultivated land is the most relevant factor after the climate driven ones, suggesting that the most effective strategy to enhance the retention service would be expanding management practices such as contouring, strip cropping or terracing. Therefore, different mitigation or adaptation management actions may be more beneficial in different regions in the same river basin. As suggested in Table 6, a 63% increase in rainfall erosivity (R) for the whole basin could be compensated with changes in the cover management factor (C) for forest land, and in the support practice factor (P) for non irrigated land use. For subwatersheds 2 and 6, the increase in climate driven factors could be possibly mitigated with changes in cover management.

Finally, despite the model’s lack of sensitivity to forested LULC factors, it is interesting to note that the proportion of forest cover in the basins is correlated with the sensitivity of other land uses to the factors C and P; in sub basins with lower forest cover, the model was more sensitive to C and P on agricultural land uses but less so to grass and shrub land covers. While these results may seem counter intuitive, they can be explained by the high retention capacity of forested LULC. In comparison, factors like C and P are relatively less influential since most of the sediments from the watershed are retained by the forest patches, making the model relatively insensitive to other LULC’s characteristics. We note that this analysis does not take into account spatial distribution of the LULC, and that patches closer to the stream would likely have a greater impact on sediment retention and export. In
addition, since the results of the correlation analyses are limited by a small sample size (7 watersheds), their interpretation for practice remains tentative.

4.2. Recommendations for future analyses

This study illustrates a straightforward method to quantify the InVEST sediment retention model’s sensitivity to input parameters. The result of such analysis enables an informed use of the model by narrowing down parameter selection of the key inputs and facilitating model calibration. To guide future studies, two main limitations of such simple analyses are highlighted here.

First, we note that the Morris indices are relative measures that do not directly translate into errors in outputs. For an end user, further analyses are thus needed to characterize the relationship between input and output errors. In the Llobregat basin, we undertook succinct analyses and observed distinct responses to input errors: as expected from the model structure, exported sediments responded linearly to a change in R and K factors. Conversely, non linearity was observed for C and P, since these factors depend on the proportion and relative location of LULC in the area studied (in the whole basin, with a 23.6% of non irrigated land use, a 64% decrement in C for this LULC results in a 49% decrease in sediment exported, see Table 6). These results suggest that additional analyses are necessary to interpret quantitatively the effects of input parameter uncertainties.

Second, our methodology did not investigate the model’s structural errors. Further studies may thus be necessary to gain confidence in the implications drawn from our sensitivity analyses. For example, the implications for practice implicitly assume that the model outputs are the sediment exported as observed in the stream. As highlighted in the methods, the contribution from additional sources of sediments such as gullies and bank erosion may have an effect on the results presented here. In addition, the structure of the InVEST model implies that most of the exported sediment is mainly from the riparian zone. While this observation is common in the literature, the relative contribution of the riparian zone LULC may be under or over estimated, which would affect the land use dependent relationships derived in our study. Preliminary investigations were conducted to investigate the relationship between sensitivity and proportion of LULC in the riparian zone, but did not result in higher correlations than those reported here. Although it was not in the scope of this study, further analyses of the assumptions underlying the sediment retention model should help improve model’s interpretation. Such work is currently conducted by the Natural Capital Project and should result in additional insights into the modeling of sediment retention services.

5. Conclusions

This sensitivity analysis of the InVEST sediment model identified the climate driven model parameters (rainfall erosivity and soil erodibility factors) as most influential for sediment export and retention, and other parameters such as cover management or support practice as playing a secondary but significant role. Accordingly, small changes in the magnitude and frequency of extreme rainfall events may cause major changes in sediment dynamics, highlighting the susceptibility of sediment retention service to climate change in Mediterranean basins. Moreover, the results from our sensitivity analysis further suggest that it is feasible to compensate the likely effects of climate change on sediment export in some areas by introducing management practices such as contouring, strip cropping or terracing, as well as increasing the per centage of soil coverage by vegetation. Implications from such analyses should, however, be put in perspective of the model’s scope: structural errors in the modelling of sheetwash erosion and the omission of additional sources of sediment may also impact management decisions for erosion control.

Acknowledgements

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References

Cloret N, Gallart F. Sediment yield in a mountainous basin under high Mediterranean clime.
Geomorph N Fl 1986;60:205–16.
COST03. Off- and on-site environmental impacts of runoff and erosion, European coopera-
de Groot R, Alkemade R, Braat L, Hein I, Willemsen L. Challenges in integrating the con-
cept of ecosystem services and values in landscape planning, management and deci-
Lavee H, Imeson AC, Sarah P. The impact of climate change on geomorphology and desert-
MA (Millennium Ecosystem Assessment). Ecosystems and human well-being: the assess-


Russell B. Authority and the individual. In: Allen Unwin, editor. Michigan University; 1949. [125 pgs.]


