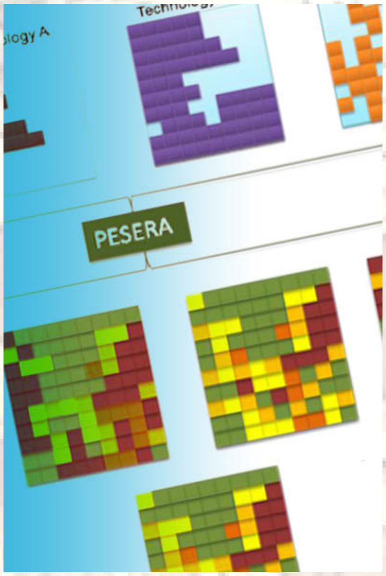
DESIRE REPORT series



A model that integrates the main bio-physical and socio-economic processes interacting within an agro-ecosystem.

WP 5.1 Coarse scale model development

Author: L.Fleskens, B. Irvine, M.K. Kirkby, D. Nainggolan, M.S. Reed, M. Termansen.

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Summary

The aim of Deliverable 5.1.1 is to "develop a model for the main bio-physical and socio-economic processes interacting within an agroecosystem, building on existing experience in combination with results generated within WBs 1-4". The inter-linked models described in this report are designed to evaluate the likely biophysical and socio-economic effects of applying remediation strategies selected by stakeholders in WB3 at a regional scale, by scaling up results from field trials and secondary data. These models will be applied in all study areas for which there is sufficient data. Additional socio-economic models have been developed for application in a single study site to further explore factors influencing the adoption of remediation strategies by land managers and the wider effects of adoption on the regional economy.

The report starts by introducing the modeling approaches that are used and explains how these move beyond the state-of-the-art. Next, it shows how the biophysical model proposed for the DESIRE project builds on and extends the PESERA model. It also describes a database that is used to assess the effects of soil and water conservation measures on runoff, soil loss and sediment yield at catchment scales. In order to determine which remediation strategies should be implemented where to achieve desertification policy targets at least cost, and to make an investment analysis of these strategies for both land users and societies, PESERA outputs are linked to the implementation costs of remediation strategies in each study area to produce a cross-site cost-effectiveness analysis and financial cost-benefit analysis – the latter also extended with some selected wider economic effects. Finally, the report describes models that have been developed for application in Spain to investigate the regional economic effects of adopting different remediation strategies (using regional economic modeling), and determine what factors influence land managers to adopt different remediation strategies and change land use under different future scenarios (using Agent-Based Modeling).

The proposed modeling approach contains a number of important novelties. For example, the approach overcomes a number of challenges to incorporate inputs from multiple stakeholders in very different contexts into the modeling process, in order to enhance both the realism and relevance of outputs for policy and practice. A number of modelling approaches are being applied to the mitigation of land degradation for the first time to provide novel insights. For example, site-selection modelling is being applied to land degradation mitigation for the first time to enable landscape-scale assessments of the most economically optimal way to attain of environmental targets. There have been few attempts to use Cost-Benefit Analysis to investigate the spatial variability of the profitability of SWC measures, which may have important implications for the adoption of measures across landscapes and their consequent environmental effects. For the first time, regional (input-output) economic models are also being used to consider the effects of land degradation mitigation on the regional economy. By linking (Agent-Based) models of human behaviour to models that describe the wider regional economic and biophysical implications of people's actions, it may be possible to better understand how people are likely to respond to environmental change, and how their responses in turn are likely to influence their environment. Such models may offer us the opportunity to explore how land managers might react to different future policy options and provide ways to make refinements to policy design that can more effectively achieve

environmental sustainability goals. The fact that much of this is being done for multiple study areas based on data gathered by a collective effort between researchers and local stakeholders makes the approach truly unique. Cross-site scaling-up of the model will for the first time be able to provide estimates of the global impact of land degradation mitigation, built on local realities.

Acronyms

ABM	Agent Based Model
CBA	Cost Benefit Analysis
DEM	Digital Elevation Model
IEEA	Integrated Environmental and Economic Accounting
INE	Instituto Nacional de Estadística (Spanish National statistics Institute)
I/O	Input-Output
IRR	Internal Rate of Return
GIS	Geographical Information System
NPV	Net Present Value
PESERA	Pan European Soil Erosion Risk Assessment
SDR	Sediment Delivery Ratio
SLM	Sustainable Land Management
SWC	Soil and Water Conservation
SY	Sediment Yield
QT	WOCAT Technology Questionnaire
QA	WOCAT Approach Questionnaire
WOCAT	World Overview of Conservation Approaches and Technologies

1 Introduction

The aim of Deliverable 5.1.1 is to "develop a model for the main bio-physical and socio-economic processes interacting within an agroecosystem, building on existing experience in combination with results generated within WBs 1-4". The inter-linked models described in this report are designed to evaluate the likely biophysical and socio-economic effects of applying remediation strategies selected by stakeholders in WB3 at a regional scale, by scaling up results from field trials and secondary data. These models will be applied in all study areas for which there is sufficient data. Additional socio-economic models have been developed for application in a single study site to further explore factors influencing the adoption of remediation strategies by land managers and the wider effects of adoption on the regional economy.

Increasingly sophisticated models are being used to represent land degradation processes in highly complex environmental, economic and social systems. Modelling has primarily been used by natural scientists as a means of capturing and predicting aspects of these systems, usually within disciplinary boundaries (e.g. hydrology, soil or atmospheric models). However, many of these models bear little or no relation to physical reality (Prell *et al.*, 2007). In contrast, the (relatively recent) development of 'theoretical' models, which possess some physical basis, allows the real possibility of application over a wide range of conditions and locations, as well as aiding our understanding of natural processes and systems (Anderson and Burt, 1985). Economists also have a fairly long tradition of modelling components of socio-ecological systems, especially human-environment interactions (e.g. Ciriacy-Wantrup, 1952; Clark, 1976; Bergh and Straaten, 1997). However, many of these models also bear little or no relation to physical reality, often being based on stringent assumptions such as perfect information, optimal behaviour, and rational choice (e.g. Simon, 1955). Partly in response to these limitations, models are now increasingly being informed by inputs from stakeholders. The importance of participatory modelling, especially in land degradation and rehabilitation, derives from the awareness of the inadequacy of traditional, engineering approaches to dealing with 'complex and ill-structured problems' (Giordano *et al.*, 2007). It has become increasingly obvious that traditional modelling approaches based on optimization have to be combined with inputs from stakeholders, if their outputs are to feed effectively into policy design and implementation (Giordano *et al.*, 2007; Funtowicz and Ravetz, 1990, 1994; Funtowicz *et al.*, 1998).

Although there are now approaches that can incorporate inputs from stakeholders into model development, many limitations remain. Firstly, stakeholder knowledge tends to be restricted to local contexts, so input to models with regional or global coverage is difficult (Wohling, 2009). Second, there are many (often competing) stakeholder interests in land degradation and rehabilitation, with different knowledges and priorities over the processes and potential solutions that should be modelled (Raymond *et al.*, under review). Finally, although there have been many separate attempts to incorporate stakeholder inputs into models of biophysical systems, human behaviour and the local or regional economy, there have been no attempts to do this for combined social, economic and/or environmental systems (Prell *et al.*, 2007; Hubacek and Reed, 2009). In response to these challenges, the modelling approach described in this report incorporates various inputs from stakeholders to enhance both the realism and relevance of outputs for application in policy and practice:

- The DESIRE project collaborates with stakeholders to define the most important land degradation processes (WB1) and potential solutions to model in WB5. Stakeholder analysis is used to ensure a cross-section of stakeholders with different knowledge are represented and decision support tools are used to negotiate differing stakeholder priorities (WB3);
- Information collected from stakeholders in WB3 provides the basis for assessing the cost-effectiveness of remediation options across environmental and socio-economic gradients;
- Environmental effects of selected remediation options are evaluated using the PESERA model;
- The resulting linked models have the potential to be applied around the world through the case study approach of the DESIRE project, whilst retaining and building on inputs based on local knowledge;
- In one study site, this is expanded by incorporating stakeholder inputs into (Agent Based) models of human behaviour using data from structured questionnaires and combining this with a (Input-Output) regional economic model.

By linking these models of human behaviour to models that describe the wider regional economic and biophysical implications of people's actions, it may be possible to better understand how people are likely to respond to environmental change, and how their responses in turn are likely to influence their environment. Such models[1] may offer us the opportunity to explore how land managers might react to different future policy options and provide ways to make refinements to policy design that can more effectively achieve stated goals.

Site-selection modelling for optimisation of conservation efforts is a well established research area on biodiversity conservation (e.g. Camm *et al.*, 1996; Crossman *et al.*, 2007), but has so far not been applied to the mitigation of land degradation. This research will enable landscape-scale assessments of the most economically optimal ways to attain environmental targets. Furthermore, although Cost-Benefit Analysis is an established method in evaluating soil and water conservation measures, from individual measures (de Graaff, 1996; Ludi, 2004; Posthumus and de Graaff, 2005; Fleskens *et al.* 2005, 2007) to projects (de Graaff, 1996; Ninan and Lakshmikanthamma, 2001) to continental and global scales (Pimentel *et al.*, 2005; Kuhlman *et al.*, in press), so far the spatial variability of the profitability of SWC measures has received little attention[2]. The model described in this report offers a method which considers the perspective of both individual land users and policy makers, and can scale up results from the field to the region and beyond.

Linking environmental and socio-economic models not only facilitates a spatially explicit evaluation of mitigation strategies, but vice-versa, the biophysical effects simulated by the environmental model can be attributed real meaning as the spatial

[1] This Agent-Based Model is derived from spatially explicit data collected from land managers in Guadalentin, Spain, using a structured questionnaires

[2] Heidkamp (2008), in a broader context, argues that 'the environment has been largely ignored beyond its treatment as a more or less passive location condition or resource factor input'.

configuration of the adoption of mitigation strategies by individual land users is based on economic analysis of available alternative options. The coupled models can be used to model environmental (e.g. climate change) as well as socio-economic (e.g. policy) scenarios. The fact that this is done for multiple study areas based on data gathered by a collective effort between researchers and local stakeholders makes the approach truly unique. Cross-site scaling-up of the model will for the first time be able to provide estimates of global impact of land degradation mitigation, built on local realities.

Another innovative aspect of the approach is that it considers the effect of land degradation mitigation on a regional economy. Regional economic modelling using input/output modelling is a long-established discipline (Miller and Blair, 1985) that has to our knowledge never been used to consider the effects of desertification. Although environmental effects have been considered in such models (e.g. water use, Duarte *et al*., 2002; Guan and Hubacek, 2008), this is one of the very first models that consider soil erosion.

Section 2 shows how the biophysical model proposed for the DESIRE project builds on and extends the PESERA model (Kirkby *et al.*, 2008), originally developed for Pan-European Soil Erosion Risk Assessment within a dedicated EU (FP5) project (section 2.1). It also describes a database that is used to assess the effects of soil and water conservation measures on runoff, soil loss and sediment yield at catchment scales (section 2.2). In order to determine which remediation strategies should be implemented where to achieve desertification policy targets at least cost, and to make an investment analysis of these strategies for both land users and societies, section 3 links PESERA outputs to the implementation costs of remediation strategies in each study area (identified in WB3) to produce a cross-site cost-effectiveness analysis, and financial and economic cost-benefit analyses. Finally, section 4 describes models that have been developed to investigate the regional economic effects of adopting different remediation strategies (using input-output analysis for regional economic modeling), and determine what factors influence land managers to adopt different remediation strategies and change land use under different future scenarios (using Agent-Based Modeling), which will be applied in Spain. Figure 1.1 shows how the different models are interrelated.

This report is the first of a series of deliverable reports from WB5. The PESERA model described in this report is being extended to capture the role of grazing, fire and wind erosion more effectively, and enhance pedotransfer functions (to be reported in Deliverable 5.2.1). The model is being adapted to each study area to reflect indicators and land degradation drivers identified in WBs 1 & 2 as closely as possible. We will use this model to look at the biophysical effects of different remediation options that we have trialed in study areas at a regional or perhaps national scale (Deliverable 5.3.1). These results will be integrated with field trial results in all study areas, and will form the basis of a final stakeholder workshop, in which we will discuss recommendations for policy-makers and extension services (Deliverable 5.4.1). Locally calibrated application of the fine-scale PESERA and/or alternative models (WP 5.2 below) will then be used to extend the results of pilot area studies to a larger hinterland, in order to evaluate the impact of recommended conservation measures (from WB 3) for the surrounding area. The extent of this wider hinterland will be constrained by broad similarities of environment (guided by WB 2) and the availability of coarse (1km) resolution data, although reference data is already available at this scale for much of Europe.

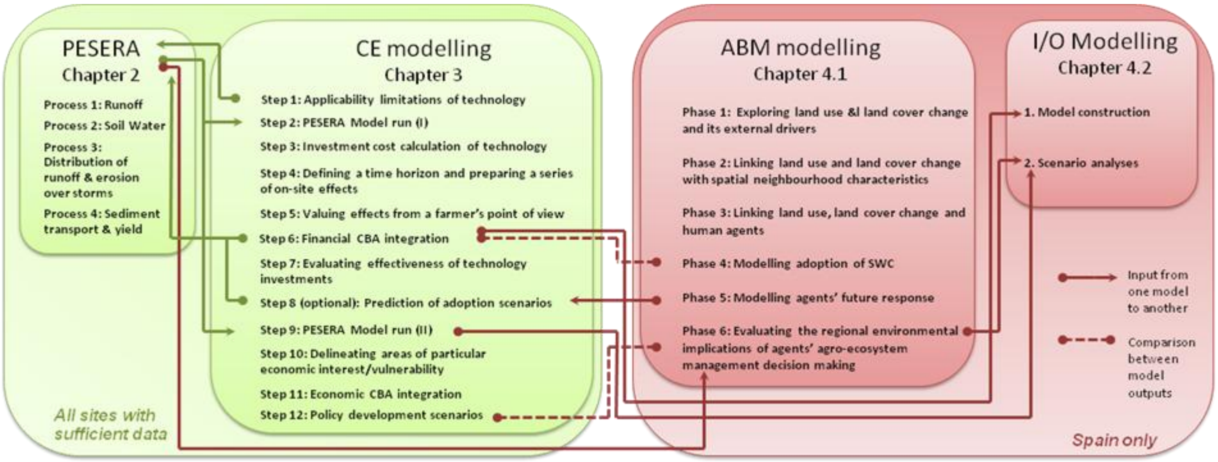

Figure 1.1: Schematic overview of model interrelations within WB5

2 Coarse-scale model description

2.1 Introduction

PESERA is currently being used to predict runoff and erosion across Europe, and now provides the core bio-physical platform into which additional elements can be built to reflect drivers and indicators identified in WBs 1 & 2 for each site. The core of the model is a physically based model for partitioning precipitation between surface runoff (driving soil erosion), subsurface runoff (providing long term drainage) and soil moisture driving plant growth. The model was originally designed to integrate soil erosion loss for 1 km^2 grid squares, and for areas up to the whole of Europe. Runoff and erosion respond to climate, topography, land use and soil type, as well as dynamically interacting with soil moisture to estimate vegetation performance in a given year or climate. The expressions used are compatible with finer scale models, and the model has also been used for smaller grid squares, down to a limit of 100m x 100m.

The particular value of PESERA as a core model is that it is implemented within ARC-GIS. The use of PESERA as the model platform also enables us to ensure that information is available in a spatially-distributed way with topography and other factors incorporated. It is possible to show how the same management in one part of catchment will have a different impact/risk associated with it compared to that management occurring in another part of the catchment. This facilitates the development of distributed decision-making in a move away from simple blanket policies.

The following text describes PESERA (sections 2.2-2.4), explains how it has been extended through the DESIRE project and other related IPs (section 2.5), and then summarises how it has been adapted to assess SLM strategies (section 2.6). Finally, it summarises an approach being developed to assess the effects of soil and water conservation measures on runoff, soil loss and sediment yield at catchment scales, which will be reported fully in Deliverable 5.3.1 (section 2.7).

2.2 The PESERA model: Summary

PESERA is a process-based model that is designed to estimate long term average erosion rates at 1 km resolution and has, to date, been applied to most of Europe and parts of North and West Africa. . The model is built around a partition of precipitation into components for overland flow (infiltration excess, saturation excess and snowmelt), evapo-transpiration and changes in soil moisture storage. Transpiration is used to drive a generic plant growth model for biomass, constrained as necessary by land use decisions, primarily on a monthly time step. Leaf fall, with corrections for cropping, grazing etc, also drives a simple model for soil organic matter. The runoff threshold for infiltration excess overland flow depends dynamically on vegetation cover, organic matter and soil properties, varying over the year. The distribution of daily rainfall totals has been fitted to a Gamma distribution for each month, and drives overland flow and sediment transport (proportional to the sum of overland flow squared) by summing over this distribution. Total erosion is driven by erodibility, derived from soil properties, squared overland flow discharge and gradient; it is assessed at the slope base to estimate total loss from the land, and delivered to stream channels.

PESERA is attempting to make best use of advances in process understanding, while allowing application across a large region, e.g. at 1-km resolution across most of Europe. Although it is recognised that limitations of resolution in time and space, determined by ready availability of suitable data, must partially compromise the accuracy of any such forecast, an objective comparison tool of this type has proved valualbe, for example in developing the European Soil Protection Policy. The PESERA model, providing explicit dependence on climate and vegetation, both retains essential features of more detailed process models and shows a clear response, in the appropriate direction, to the components that drive other assessments, such as USLE and CORINE, combining these and other driving factors within a consistent process-based rationale.

A number of factors contribute to the risk of erosion and, in PESERA, they are combined, in a physically meaningful way, with the intention of making the best possible estimate of long-term average erosion rates. The current version of the model was developed within the structure of the PESERA project, and partly based on previous funded and un-funded research (Kirkby & Neale, 1987; de Ploey *et al.*, 1991; Kirkby & Cox, 1995; Kirkby *et al.*, 2000). The PESERA model combines the effect of topography, climate, vegetation cover and soil into a single integrated forecast of runoff and soil erosion. It is recognised that data for validating estimates of soil loss from erosion models are sparse, and that current models generally forecast runoff as a necessary intermediate to forecasting sediment transport. Since runoff processes are also better understood than sediment transport, particularly on hillsides, it has seemed sensible to build PESERA on a hydrological core.

Erosion by running water has been identified as the most severe hazard threatening the protection of soil in Europe (European Commission, 2006), and is one of the major forms of desertification. By removing the most fertile topsoil, erosion reduces soil productivity leading, where soils are shallow, to a progressive and ultimately irreversible loss of natural farmland, and in vulnerable areas, is one major process of desertification. Severe erosion is commonly associated with the development of temporary or permanently eroded channels or gullies which can fragment farmland. Much of the soil and runoff removed from the land during a large storm generally accumulates below the eroded areas, with some sediment spilling offsite and in severe cases blocking roadways or channels and inundating buildings. Erosion rate is sensitive to both climate and land use, as well as to detailed conservation practice at farm level, as documented by the US National Resources Conservation Service and similar organisations worldwide. In a period of rapid changes in both climate and land use, resulting from revised agricultural policies in reaction to global warming and international markets, it is valuable to be able to assess the state of soil erosion at a regional scale. This needs an objective methodology operating on standard data sets, which allows the assessment to be repeated as conditions, pressures and drivers, change, or to explore the broad scale implications of prospective global or Europe-wide changes. This provides a sound basis for estimating the overall costs attributable to erosion under present and changed conditions, and objectively suggests areas for more detailed study and possible remedial action.

The PESERA model provides such an objective harmonised estimate of current rates of soil erosion, averaged over a series of years with current climate and land use. European estimates have been made at a resolution of 1 km, and indicate the rate of loss of material from hillslopes. Sediment delivery through the river system is explicitly not taken into account, and much of the eroded material generally remains close to its source, with significant off-site effects generally confined to a local area, and strong de-coupling between slope and channel sediment transport (e.g. Trimble, 1981, Govers, 1987)

2.3 Alternative approaches to assessing soil erosion

There are a number of possible methodologies for creating a coarse scale erosion map (Gobin *et al.*, 2004). Some of these are based on the collection of distributed field observations, others on an assessment of factors, and their combination, which influence erosion rates, and others primarily on a modelling approach. All of these methods require calibration and validation, although the type of validation needed is different for each category. There are also differences in the extent to which the assessment methods identify past erosion of an already degraded soil resource, as opposed to risks of future erosion, under present climate and land use, or under scenarios of global change.

Process models have the potential to respond explicitly and in accordance with experience to changes in climate or land use, and so have great promise for developing scenarios of change and 'what-if' analyses of policy or economic options. Set against this advantage, process models generally make no assessment of degradation up to the present time, and can only incorporate the impact of past erosion where this is recorded in other data, such as soil data bases. Models also generally simplify the set of processes operating, so that they may not be appropriate under particular local circumstances. Although the USLE and RUSLE are the models most widely applied in Europe (e.g. van der Knijff *et al*, 2000), the USLE-approach is now widely considered to be conceptually flawed in that it fails to properly distinguish between soil and climatic conditions in the infiltration process. The other models that are now emerging are based on runoff thresholds (e.g. Kirkby *et al*, 2000) or the MIR (Minimum Information Requirement) approach (Brazier *et al*, 2001) applied to the more complex USDA WEPP model (Nearing *et al*, 1989).

The application of a process model has been preferred here for three main reasons:

1. It applies the same objective criteria to all areas, and so can be applied throughout a region, subject to the availability of suitable generic data;
2. It provides a quantitative estimate of erosion rate which can be compared with long term averages for tolerable erosion; and
3. The methodology can be re-applied with equal consistency as available data sources are improved, and for past and present scenarios of changed climate and land use.

Nevertheless a process model, and particularly a coarse-scale model such as PESERA, has a number of inherent disadvantages compared to simpler models, including:

1. The need for input data which may not be freely or readily available. For example it has not been possible to access Europe-wide climate data at better than daily resolution, or better than 50km or 0.5 degree spatial resolution, even though such data could be purchased from national databases (but at prohibitive cost). It is well known (e.g. Wainwright and Parsons, 2002) that there is a inter-dependence between temporal resolution and erosion estimates;
2. The need to rely on spatial soil data that have been collected nationally, using criteria that differ from country to country, combined into soil types that are not completely uniform, and only partially harmonised when compiled in the European Soil Map (King *et al.*, 1994) and incorporated into the European Soil Database (King *et al.*, 1995); and
3. An inevitable concentration on the relevant dominant processes that are most widespread, in this case infiltration *excess overland flow*, so that erosion by *saturation overland flow* and snowmelt, for example, are less well estimated.

There are many pitfalls and alternative approaches to the issues of scale. Here we focus primarily on a single spatial scale, even though this scale is applied over a large spatial extent. Some of the particular problems identified by Zhang *et al* (2002) are thus minimized and the approaches used have been deliberately selected to reduce the impact of scale. For example the use of relief has been found empirically to be much less sensitive to DEM resolution than estimated gradients, provided that relief is measured within the same radius around each point. Similarly cover is generally defined, via land use, at the field scale rather than from the scale-dependent estimates derived from remote sensing.

It is recognised that the interdependence between infiltration parameters and the temporal resolution of rainfall discussed by Wainwright & Parsons (2002) remains a potential problem as spatial scale changes. In practice, this means that the effective storage capacities vary with spatial resolution. In the model as described here, however, spatial scale remains fixed at 1 km, so that the runoff thresholds used are implicitly linked to this scale, and might need to change only when the spatial scale is altered.

2.4 Process representation in PESERA

I: Runoff

PESERA represents a fundamental advance on previous models of comparable simplicity, most notably the USLE and its derivatives by explicitly separating hydrology from sediment transport. That is to say that it first estimates storm overland flow runoff, and then uses the runoff to estimate sediment transport. Soil properties therefore enter separately into these two stages, replacing the USLE erosivity, a climatic property, and erodibility, a soil property.

At the same time, the PESERA model has been designed to provide an estimate of long term erosion and must therefore scale up from our knowledge of instantaneous sediment transport, as a function of shear stress or flow power, to firstly an aggregate relationship between event discharge and event sediment discharge, and secondly from single events to the aggregate of storm events across the relevant distribution of storms. This temporal up-scaling provides the essential link between climate, defined by the distribution of rainfall events, and long term sediment transport. Although this scaling up has been discussed and partially implemented in previous models (Kirkby *et al.*, 1996; Kirkby, 1998), it has not previously been applied within a comprehensive soil erosion model.

Precipitation is divided into daily storm events, expressed as a frequency distribution, that drive infiltration excess overland flow and soil erosion, and monthly precipitation, some of which may be as snow, driving saturation levels in the soil. Infiltration excess overland flow runoff is estimated from storm rainfall and soil moisture. Sediment transport is then estimated from overland flow and routed, in principle, downslope. To obtain long term estimates of soil erosion these estimates must then be scaled up by integrating over time. This process of scaling up has two stages, first from momentary to event-integrated dependence, and secondly from events to long term averages via the frequency distribution. For the first stage, if instantaneous sediment discharge can be expressed as a power law dependent on instantaneous water discharge, for example through the Yalin Equation (Finkner *et al.*, 1989). The relationship between event total sediment discharge and event total discharge will, in general, also be a power law, but the exponent will differ according to how hydrograph forms change with flood volume. In this respect, it is intermediate between very short interval sediment transport models and the much coarser time resolution in long term landscape evolution models (e.g. Tucker *et al.*, 2001; Coulthard *et al.*, 2005).

In the second stage of scaling up, individual storm totals are integrated over the frequency distribution of storms. Two assumptions are normally made, first that the distribution of storms can be replaced by the distribution of daily rainfalls, and second that overland flow can be estimated on the basis of monthly average soil moisture conditions. The first of these assumptions avoids the discussion of how rainfall is divided, more or less arbitrarily, into storm events. The use of a daily unit is both convenient, in that daily rainfall data is relatively widely available, and appropriate because bursts of rainfall within a single day are significantly influenced by raised soil moisture levels from previous bursts, whereas for longer periods there may be significant drying between bursts. Similarly, monthly updating of soil moisture is sufficient to reflect important seasonal differences in weather, to respond to seasonal differences in land cover and to make use of widely available meteorological data. These assumptions are, however, a compromise, attempting to simplify the estimation of storm runoff while retaining the frequency signature of storms (daily) and soil moisture (monthly).

This approach can be applied using either an historic (or simulated historic) sequence of daily rainfalls, or by summing over a frequency distribution of daily rainfall events for each month. The former approach is preferable for comparison with observed data, whereas the latter is more suitable for estimating long term average rates: it has the disadvantage that it does not respond to inter-annual differences or to the timing of consecutive storms within a month. These methods thus provide an explicit link to available climatic data, providing an improved physical basis for comparisons across large regions, and with climate scenarios or historic data.

There are a number of simple methods for estimating storm runoff from storm rainfall. Implicitly, these are all based on an understanding of i) the infiltration process, and ii) that erosive overland flow can generally be represented as an infiltration excess, or Hortonian, process. The effect of subsurface flow, where and when it is important, may then be used to modify potential rates of infiltration with lower infiltration under wet conditions. Similarly, the role of vegetation and soil organic matter can modify the infiltration rates through changes in soil structure and/or the development over time of surface or near-surface crusting. Three models are coupled to provide the dynamics of these responses; i) an 'at-a-point' hydrological balance, which partitions precipitation between evapo-transpiration, overland flow, subsurface flow and changes in soil moisture; ii) a vegetation growth model, which budgets living biomass and organic matter subject to the constraints of land use and cultivation choices; and iii) a soil model, which estimates the required hydrological variables from moisture, vegetation and seasonal rainfall history.

'At-a-point' soil hydrology can be described through the Richards' equation, although with reservations where both matrix and macropore flow are active. Solutions may be approximated through the use of infiltration equations, such as the Green-Ampt (1911) or Philip (1957) formulations. However, these approaches are not compatible with the use of daily time steps, within which the detail of storm profiles is lost, and it is impracticable to provide better estimates of runoff than those from the SCS (Soil Conservation Service) curve number (Yuan *et al.*, 2001) or a simple bucket model. Here the bucket model is preferred, which offers a simple conceptual insight into the volume of infiltration before runoff occurs, and can be linked directly to concepts of soil moisture storage, as it varies within and between sites. In the bucket model, runoff $\boldsymbol{r}$ is given by:

$$r = p(R - R_0) \quad (1),$$

In which $\boldsymbol{R}$ is total storm rainfall, $\boldsymbol{R_0}$ is the runoff threshold or bucket storage capacity and $\boldsymbol{p}$ is the dimensionless proportion of subsequent rainfall that runs off. All values are normally expressed in mm.

Although there is substantial scatter in relationships between observed total rainfall and runoff, and the bucket model (equation 1) has been adopted in PESERA, as the simplest model and one that is broadly consistent with these data, in which storms are treated as independent random events. Comparison with a more detailed model, based on the Green-Ampt equation, shows a similar scatter for daily rainfall totals over a set of storm events taken from a continuous record for a semi-arid area in SE Spain. Equation (1) has been freely fitted to the data and it can be seen that, without a more detailed knowledge of storm profiles than can be derived from the daily record, it is both impracticable to apply a more sophisticated model, and unwise to make runoff forecasts for any individual storm.

II: Soil Water

Water infiltrating into the soil is limited by the runoff threshold, which is conceptualized as an available near-surface water store. The upper limit for this store is constrained by soil properties, and is currently estimated from mapped soil classes in the European Soil Database (King *et al.*, 1995). This store may be decreased where the soil is crusted, and/or if subsurface flow brings saturated conditions close to the surface. Additional considerations apply where the soil is frozen or snow covered. Both sub-surface flow and the near-surface water store are available for evaporation and for evapo-transpiration linked to plant growth. Soil properties have, necessarily, been taken from existing soil maps, since details of the primary properties required are not available at a European scale. Data have been derived from the European Soil Database, giving estimates of available water storage capacity, crustability and erodibility (as defined by k in equation 8 below). The pedo-transfer rules used have been closely modelled on work by Le Bissonnais *et al.* (2002, 2005) and Cerdan *et al.* (2002), with modification described by Gobin *et al.* (2004) and Jones *et al.* (2000). It is recognised that soil maps are an imperfect source of data, but are unlikely to be superseded in the near future.

After allowing for interception, evapo-transpiration is partitioned between the vegetated and un-vegetated fractions of the surface according to the proportional vegetative crown cover. Interception is calculated as a fraction of rainfall rather than a fixed capacity, and this fraction increases with vegetation biomass (Llorens *et al.*, 1997). Each evapo-transpiration component is associated with a rooting depth (e.g. Shah *et al.*, 2007) according to the land cover type for the vegetated area and normally set at 10mm for the bare soil. For each component, potential evaporation (PE), after subtraction of interception, is then reduced exponentially to an actual rate (AE) of:

$$AE = WUE.PE.\exp(-D/h_R) \quad (2)$$

Where WUE = dimensionless water use efficiency for stage of plant growth (or 1.0 for bare soil)

D is saturated subsurface deficit (mm)

and h_R is the rooting depth (mm of water) for each partition.

Contributions to evaporation (in mm per measurement period) are weighted for the fractional plant cover to give a combined estimate.

Subsurface flow is estimated using TopModel (Beven & Kirkby, 1979), with topographic properties estimated from local relief (from DEM) and soil characteristics (saturated hydraulic conductivity and TopModel soil parameter, m) from the soil type, based on field experience (e.g. Beven *et al.*, 1984). The average saturated deficit is estimated in monthly steps to provide the background hydrological conditions and, in particular, the saturation constraint on the runoff threshold which controls overland flow runoff in each storm. Deficit is updated monthly from the TopModel expression:

$$D = D_0 + m\ln\left\{\frac{j_*}{i}\exp\left(-\frac{D_0}{m}\right) + \left[1 - \frac{j_*}{i}\exp\left(-\frac{D_0}{m}\right)\right]\exp\left(-\frac{it}{m}\right)\right\} \mathit{for}\ i \neq 0$$

$$D = D_0 + m\ln\left[1 + \frac{j_* t}{m}\exp\left(-\frac{D_0}{m}\right)\right] \ \mathit{for}\ i = 0 \quad (3)$$

where D is the deficit after time t (as in equation 2)

D_0 is the initial deficit (mm),

i is the net rainfall intensity (mm/month)

m is the TopModel soil parameter (mm),

and j_* is the average saturated runoff rate (mm/month)

This expression also estimates the net subsurface runoff over the month as

$$D - D_0 + it = m \ln\left[1 - \frac{j_*}{i}\exp\left(-\frac{D_0}{m}\right) + \frac{j_*}{i}\exp\left(\frac{it - D_0}{m}\right)\right] \quad (4)$$

In these calculations the total net rainfall is used, corrected for the overland flow runoff where this is a significant fraction is used. Where the deficit falls below zero, the negative deficit is re-calculated as saturation overland flow.

This combination of an infiltration excess mechanism, represented by the bucket model, with a saturation excess mechanism, represented by TopModel, provides a robust hydrological sub-model which provides an adequate response across the humid to semi-arid continuum. As shown below, the evapo-transpiration stream is also used to drive a simple plant growth model, which is also responsive to this range of conditions.

The runoff threshold for infiltration excess overland flow is estimated as an area-weighted average of the thresholds under vegetation and in the bare gaps between. Under vegetation, rainfall is lost to interception, and the runoff threshold is calculated as the lesser of two values:

(1) available near-surface water storage capacity (depending on soil textural properties), or
(2) the sub-surface saturation deficit (from the TopModel estimate described above)

In arable areas, surface roughness represents the full storage capacity of furrows immediately after ploughing, and this decays exponentially with time in the subsequent period, eventually falling to a minimum value representing the textural roughness of the surface (Darboux *et al*, 2002; le Bissonnais *et al.*, 2005. Naturally vegetated areas are also assumed to present this minimum roughness.

Bare areas are also considered to be subject to crusting, with a tendency to crusting referred to mapped soil classes, largely interpreted in textural terms as a minimum runoff threshold for a fully crusted surface (Le Bissonnais *et al.*, 2002, 2005). For arable areas, the runoff threshold for a bare area is re-calculated as beneath vegetation immediately after tillage, and this decays exponentially towards the minimum for each soil type with accumulated monthly rainfalls.

This formulation provides a seasonal response in runoff thresholds, and therefore in infiltration excess overland flow. For a conventionally ploughed annual crop, for example, thresholds are high on first planting, but fall very rapidly immediately afterwards, particularly if there is rain, as crusting develops while the crop provides little cover. As the crop grows, the runoff threshold recovers, reaching high values as the crop matures. After harvest these high values fall again, depending on how or whether the surface is protected. Under natural vegetation there is much less annual variation, with runoff thresholds responding to the seasonality of cover.

III. The distribution of runoff and erosion over storms

Storm rainfalls are considered as independent random events, defined by a frequency distribution for each month of the year. The autocorrelation between successive events is weakly represented by the seasonal variations in soil moisture, but there is some loss of information by using this approach. This represents a trade-off between simplicity and accuracy, with the least impact on estimates for the semi-arid areas where soil erosion is generally considered to be most severe, because soils normally dry out between major events.

As noted above, daily rainfall totals have been used as the basis for analysis because, while recognizing the limitations of this approach, it allows the use of the widespread daily precipitation data. On a month by month basis, daily rainfall is analyzed to give monthly total, mean rain per rain-day and the standard deviation of rainfalls on rain-days. These statistical moments allow fitting most observed data for daily rainfalls to the probability density function for a Gamma distribution as follows:

$$pd(R) = \frac{\alpha}{\overline{R}} \frac{\left(\alpha R/\overline{R}\right)^{\alpha-1}}{\Gamma(\alpha)} \exp\left(-\alpha R/\overline{R}\right)$$

where $\overline{\mathrm{R}}$ is the mean rain in mm per rainday

and $\alpha = (1/\mathrm{CV})^2$ (dimensionless)

where CV is the coefficient of variation $= \sigma / \overline{R}$ (5)

The gamma distribution provides a robust fit (e.g. McSweeney, 2007), giving a good balance between small and large events. The CV is generally between zero and unity, so that the probability density distributions peak at zero rainfall.

Infiltration excess overland flow for a storm of rainfall R is then given by equation (1) above, and the total overland flow runoff for the month integrated numerically as:

$$\sum r = \int_{R_0}^{\infty} (R - R_0) \frac{\alpha}{\overline{R}} \frac{\left(\alpha R/\overline{R}\right)^{\alpha-1}}{\Gamma(\alpha)} \exp\left(-\alpha R/\overline{R}\right) dR \qquad (6)$$

This is used directly as a component of the water balance, but it will be seen below that a power of event runoff is used to estimate sediment transport. For a power law of 2.0, the corresponding summation of $(\mathrm{Runoff})^2$ then takes the form:

$$\sum r^2 = \int_{R_0}^{\infty} (R - R_0)^2 \frac{\alpha}{\overline{R}} \frac{\left(\alpha R/\overline{R}\right)^{\alpha-1}}{\Gamma(\alpha)} \exp\left(-\alpha R/\overline{R}\right) dR \qquad (7),$$

Comparable expressions are required if other integral powers are used. This then gives the correct strong weighting to the largest events in the accumulated total.

IV. Land use and vegetation cover

The hydrological components of the model, as described above, are strongly dependent on vegetation cover, which is understood to be a major control on both runoff and erosion. For Mississippi Loess soils, measured runoff on bare soil exceeds 80%, and falls to 2% under a dense vegetation cover, and this 40-fold difference in runoff gives a 2000-fold difference in sediment loss (Meginnis, 1935). Other experiments (e.g. Hudson and Jackson, 1959) have shown that fine netting stretched above the surface of an agricultural field has almost as strong an effect as dense vegetation in reducing runoff and erosion. Thus the importance of crown cover for both runoff and erosion is extremely strong, although it is recognised that root and soil organic matter effects are also important for uncultivated areas (e.g. Kirkby and Morgan, 1978).

Input of land cover data has been approached in the model through two alternative strategies, each of which has its advantages: first through direct remote sensing of land cover and second through modelling vegetation growth. Geomatic data has the advantage that it provides a direct measure of real vegetation abundance, which is now available monthly for a period of over twenty years, through AVHRR and LANDSAT images. This integrates the effects of all impacts on the cover in an unambiguous historical record. It therefore includes the impacts of factors which may not all be fully incorporated in a model. However, the analysis is based on the best of three monthly satellite passes, and suffers from the persistence of cloud cover in Northern Europe and other humid areas. It also lacks any direct forecasting potential, and therefore has limited applicability for analyses of scenarios for land use and/or climate change.

Vegetation growth models are well established, with both generic and crop-specific models (e.g. White *et al.*, 2005). The models applied here have been based on a biomass carbon balance for both living vegetation and soil organic matter. Such models may be insufficiently parameterized to cover the full range of functional types, and are commonly limited by absence or inadequate representation of some processes. Fire and grazing are, for example, not directly represented in the models that have been used to date with PESERA. As a result, the vegetation cover is more a 'potential' than actual cover, with only indirect parameterization of some relevant influences. However, growth models respond directly to changes in land use or climate drivers, and so have greater scenario testing potential.

Analysis of RS images can be based directly on NDVI, but improved results have been obtained using the satellite-derived surface temperature to correct for water content, linearly un-mixing in a phase-space triangle between water, vegetation and soil. This gives a measure of vegetation abundance, which can be empirically related to cover and/or above ground biomass, and from which some land use classes can be interpreted from the seasonal cover cycle. (Haboudane *et al.*, 2002).

The generic vegetation model estimates gross primary productivity (GPP) as being proportional to the actual transpiration from the plant. This is offset by respiration, at a rate increasing exponentially with temperature and proportional to biomass. Leaf fall fraction is a decreasing function of biomass, to allow for a larger structural component in large plants. Where respiration is greater than GPP, a 'deciduous' response increases an additional leaf fall at a rate that increases with temperature. Finally the modelled vegetation biomass is allowed to lose a fraction to grazing or plant gathering activities. The vegetation is protected from complete elimination by allowing only a fraction to be consumed, and this also relates grazing consumption to availability.

Soil organic matter is increased by leaf fall, except where crops are harvested, and decomposes as a single linear store at a rate that increases with temperature.

Cover is calculated independently, with reference to an equilibrium cover defined as the ratio of plant transpiration to potential evapo-transpiration rate. Cover converges on this (changing) equilibrium value at a rate which is larger where biomass is small, and is the variable which drives the seasonal partition of runoff threshold between vegetated and bare areas. This generic model has been calibrated against global distributions of biomass (Kirkby and Neale, 1987). Crop models are variants of this generic model, with additional controls through data on regional patterns of planting and harvest dates, and with an evolution of water use efficiency through the life cycle of the crop (Gobin and Govers, 2003).

V. Sediment transport and sediment yield

Runoff generated locally may not reach the base of the slope to deliver sediment to a channel, and the runoff coefficient for infiltration excess overland flow has therefore generally been observed to decrease with distance or area downslope. Summed over the distribution of storm sizes described above, these factors lead to a less then linear increase of discharge with distance downslope, and this has generally been represented as a logarithmic or power law (with exponent ~ 2/3) relationship (Kirkby *et al.*, 2005).

Estimates of sediment transport are based on infiltration excess overland flow discharge which has been discussed above. Most sediment transport equations are based on considerations of tractive stress or flow power, and commonly generalized into a power law in discharge and gradient, thus avoiding a more detailed analysis of flow thread geometry. The commonest formulations (e.g. Musgrave 1947) assume that there is an ample sediment supply,and that sediment is everywhere transported by soil erosion at its transporting capacity per unit flow width C (kg.m^{-1}day^{-1}), expressed in the form:

$$C = kq^m \Lambda^n \qquad (8)$$

where k is the soil erodibility,

q is the overland flow discharge per unit flow width (l.m^{-1}day^{-1})

Λ is the local slope gradient (dimensionless),

and m, n are empirical exponents, generally in the ranges m = 1.5-3; n = 1-2.

The units for erodibility depend on the exponent m, for example being kg.l^{-2}.m.day for m =2. In such expressions, discharge is generally associated with distance from the divide, possibly with a change in the exponent m. It has generally been found that the performance of erosion models is remarkably insensitive to the choice of exponents, largely because slope and distance tend to change together, particularly along the upper concavity of a slope profile.

Evaluation of appropriate exponents may be made at a range of time and space scales (e.g. Kirkby *et al.*, 2003). The most direct approach is through soil erosion plots, but these are often not corrected for the frequency distribution of storms to provide meaningful long term averages. A second approach is to look at the critical areas required to support an ephemeral gully formed in a particular storm. This approach requires an analysis of the stability of small depressions, as a balance is reached between infilling by diffusive processes, primarily rainsplash in relevant contexts and their enlargement by soil erosion (rillwash) processes. A third approach is by back analysis of hillslope profile form, which is formed over a period in response to the full distribution of events. The difficulty with this latter approach lies in uncertainty about whether the observed landscape form has developed under process conditions that are still current, or are inherited from conditions of different climate and/or land cover.

Exponent values of m = 2, n = 1 have been adopted here, with computational advantages that are evident below. These values lie within the empirical range, and facilitate the creation of a consistent coarse scale model. Hence for transporting capacity C, it is proposed that:

$$C = k(rx)^2 \Lambda \quad (9)$$

where r is the local runoff in mm for each event, from equation (1) above,

and x is the distance from the divide (m), so that the term rx is replacing discharge, q in equation (8) above.

Summing over the frequency distribution of daily storm events in any month, the mean total sediment transport takes the form:

$$\sum C = kx^2 \Lambda . \sum r^2 \quad (10)$$

in which the final term may be taken from equation (7) above.

Alternatives to this composite power law approach can simulate selective transportation of different grain sizes, for example by defining transport capacity as the product of detachment rate and travel distance. This approach has the advantage of allowing a spectrum of responses, from a strictly transport limited approach for the coarser soil fractions, to a detachment or supply limited approach for the finest material. Although this latter approach has merit, there are not sufficient data to properly parameterize it for the proposed coarse scale model. In practice this means that the erodibility of fine soils must implicitly be reduced to allow for the limited rate of supply, whether through hydraulic erosion or through removal of previously detached material, and that, for rangeland, selective transportation creates an armour layer over time that reduces erosion rates.

In the PESERA model, sediment transport is interpreted as the mean sediment yield delivered to stream channels and includes no allowance for downstream routing within the channel network. Sediment Yield Y (kg.m^{-2}a^{-1}) is the sediment transported to the slope base, averaged over the slope length, that is:

$$Y = \frac{\sum C_B}{L} = k \frac{L^2 \Lambda_B}{L} \sum r^2 = kL\Lambda_B \sum r^2 \quad (11)$$

where the suffix B indicates evaluation at the slope base, the summation is taken over the frequency distribution of daily events in an average year and $L = x_B$ is the total slope length (m).

The term $L\Lambda_B$ can be expressed, in terms of the total slope relief in metres, $H = L\overline{\Lambda}$, where $\overline{\Lambda}$ is the average slope gradient from crest to base, giving:

$$Y = \varsigma kH \sum r^2 \quad (12)$$

Where $\varsigma = \Lambda_B / \overline{\Lambda}$ is the ratio of slope base to average gradient, a number that generally lies between 0.5 and 1.0 for typical convexo-concave slopes. This correction term can be included where available, but generally defaults to a slight correction in the empirical value for erodibility, k.

Equations (11) and (12) are taken as the final form of the expression used in the PESERA model which includes three terms:

1. Soil erodibility, which is derived from soil classification data, primarily interpreted as texture (Le Bissonnais *et al.*, 2002).

2. Local relief, which is derived from DEM data as the standard deviation of elevation within a defined radius around each point.
3. An estimate of accumulated (runoff)[2], which is derived from a biophysical model that combines the frequency of daily storm sizes with an assessment of runoff thresholds based on seasonal water deficit and vegetation growth.

2.5 Extensions of the PESERA model in DESIRE and other projects

The fundamental outline of the PESERA model was established within the PESERA EU project, and provided the best currently available level of calibration, so that the model could be applied broadly within Europe. To take advantage of this prior work, it is essential to retain the fundamental framework of the model, and to add refinements to enhance its range of application, providing representation for additional processes in a way that is compatible with the overall framework.

The main part of this further model development work took place in the DeSurvey project, including integration within the DeSurvey Integrated Assessment Model (in collaboration with other partners), as well as application to selected areas. This work primarily included the incorporation of model components for explicit treatment of wind erosion and grazing, which were developed making using of knowledge and data from Tunisia and Senegal in particular, thus extending the application of the model for the first time into North and West Africa.

Within the DESIRE project, it was planned to incorporate the effects of fire in a more explicit way, and to incorporate fine scale features within PESERA to adapt it for estimation of the erosional and hydrological impacts of the mitigation and remediation strategies described in the WOCAT questionnaires and selected in participatory workshops for each site area. This work built upon the existing model and its developments to provide a secure foundation for the additional work. These developments are summarised in the Table below.

The PESERA model is being used as one of the fundamental building blocks within the development of integrated models for both the Desire (2007-2012) and DeSurvey (2005-2010) IPs. The inclusion of a relatively well established model (Kirkby *et al.* 2004, 2008; Licciardello *et al.* 2009, Tsara *et al.*, 2005) provides a solid basis for development of the special features relevant to each project. It allows Desire to make use of advances made within DeSurvey, and offers mutual reinforcement of complementary work in the two projects. This allows us to concentrate our evaluation on other, less well tested components of the integrated model. Continuing support for work with PESERA also helps to ensure the continued availability of the model and support for applying it in these and other projects. Table 2.1 summarises the main commonalities and differences in the use of PESERA within the two projects.

The main socio-economic model used in DeSurvey has been developed by RIKS (NL) and is largely a spatially distributed economic model, based on probabilities of change within a cellular automaton framework. In contrast, the innovative model being developed in DESIRE for the Guadalentin is an Agent-Based Model making use of questionnaires addressed to stakeholders in the field (see section 4.1 for details). It will be highly instructive to compare the outcomes of these contrasting approaches, and it is particularly valuable that both are being applied within the Guadalentin catchment. At the present state of the art, the process basis of biophysical models is relatively secure, whereas effective integration with socio-economic models is an area of recognised importance but without a definitive preferred methodology. This complementarity between the DeSurvey and Desire projects is therefore of considerable scientific interest and importance.

Table 2.1: Key commonalities and differences between the use of PESERA in DESIRE and DeSurvey

Common ground between DeSurvey and Desire	**Features developed within DESIRE**
Application to the Guadalentin and in Tunisia	Application to many other distinct areas (>10)
Use of core model together with additional relevant features developed in DeSurvey, particularly with respect to wind erosion and grazing intensity.	Responsiveness to the mitigation/ remediation strategies described in WB3. And selected for each are in participatory workshops. Incorporation of additional features with respect to the impact of fire.
Capacity building, providing training and some ownership of the model by groups from European and extra-European countries.	Additional training, application and capacity building in a wider range of countries, building on previous experience.
Integration with a socio-economic model	The socio-economic models in Desire have a very different methodological and conceptual basis from those used in DeSurvey.

2.6 Adapting PESERA for assessing SLM strategies

To meet the needs of the integrated model as proposed in this report, the PESERA model needs to be run first to equilibrium, in order to establish average values of runoff, erosion and productivity under current conditions and to establish initial conditions for runs with explicit time series drawn as realisations of future climatic conditions. Using the same time series for climate in each site, the model can then be run again, applying alternative proposed technologies either as a step-change or through gradual adoption over time. These runs can then be used to assess the expected responses of land managers to the changing performance and its economic consequences. In order to do this, PESERA will be developed to ensure that model output responds appropriately to the remedial technologies that are being proposed within the project through WB3. PESERA will itself deal exclusively with the technologies[3] involved in SLM strategies; strategies, however,

[3] SLM technologies are the agronomic, vegetative, structural and management measures that control land degradation and enhance productivity in the field (Schwilch *et al.*, 2009).

also include approaches[4]. The impacts of SLM approaches will be incorporated in the cost-effectiveness modelling and agent-based modelling (chapters 3 and 4)[5].

There are a number of parameters and methods that can be adapted to represent the impact of the various SLM technologies proposed. Among the more relevant SLM technologies are those described in Table 2.2.

Table 2.2: Parameters and methods from PESERA that can be adapted to represent the impact of different SLM technologies proposed in DESIRE

Remedial Measures	**Examples from the WOCAT database on technologies (from Del. 3.2.1)**	**Model manipulation**	**Details**
• Mulching and/or maintaining ground cover vegetation within tree crops (vines, nuts, olives...) • Crop or fallowing rotation • Changes of land use (e.g. tree addition/ removal) • Zero or reduced tillage	SPA03 (Spain); MOR14 (Morocco) MOR11, MOR12 (Morocco); TUR04 (Turkey) CPV03 (Cape Verde); MOR013 (Morocco) CHL01 (Chile); GRE01, GRE03 (Greece);	Change of month-by-month ground cover	Reduces surface crusting and therefore runoff and erosion. Better water retention favours vegetation growth etc.
• Retention of crop residues as litter layer at harvesting of arable and other crops • Zero or reduced tillage	CHL01 (Chile); GRE01, GRE03 (Greece);	Modifies biomass balances and cover	Affects surface properties as above and feeding slowly into soil organic matter that further enhances water retention etc.
Irrigation	GRE02 (Greece); GRE05 (Greece); RUS01 (Russia); TUR03 (Turkey)	Added water for greater growth of crops	Expressed as a proportion of irrigation demand met after using rainfall to the full. Output as total water

[4] SLM Approaches are ways and means of support that help to introduce, implement, adapt and apply SLM technologies on the ground. An SLM approach consists of all participants (policy-makers,administrators, experts, technicians, land users, i.e. actors at all levels), inputs and means (financial, material, legislative, etc.), and know-how (technical, scientific, practical) (Schwilch *et al.*, 2009).

[5] The integrated models described in this deliverable will thus be able to assess the impact of SLM strategies; the reader should keep in mind the place in which technologies and approaches will be addressed when reference is made to strategies.

			required as well as improved crop yields etc
Water harvesting	BOT04 (Botswana); CPV01 (Cape Verde); SPA04 (Spain); TUN09, TUN12, TUN13 (Tunisia)	Added water for greater growth of crops. Reduced area available for crop growth. Requires suitably compact collecting areas or diversion from ephemeral streams. Cisterns/ storage reservoirs allow displacement of irrigation over time.	Expressed as a multiplier representing ratio of collecting area to irrigation area, allowing for efficiency of collection (i.e. measures to enhance runoff from collecting area). Upper thresholds set by spillway design and associated erosion risks.
• Changing intensity of grazing • Changes in fuel wood harvesting • Removal of unpalatable species • Game ranching	ITA01 (Italy); TUN11 (Tunisia); TUR01 (Turkey) BOT05, BOT06 (Botswana) BOT07 (Botswana)	Expressed as fraction of available biomass growth removed by animals or people.	Grazing intensity needs to recognise contribution of supplementary fodder. Relevant for biogas or solar cookers
• Terracing with vegetated, earth or stone strips/banks • Strip cropping • Contour .v. downslope cultivations • Novel cultivation patterns	CHN51, CHN52, CHN53 (China), CPV02, CPV04 (Cape Verde), GRE04 (Greece), SPA02 (Spain); TUN10 (Tunisia) CPV05, CPV06 (Cape Verde), POR01 (Portugal) SPA01 (Spain) SPA05 (Spain)	Sub-grid modelling (Finer scale model to parameterise impacts of treatments that have a finer texture than the 100m or 1 km cell)	Details vary with treatment. Sub-model resolution 1-10m. Output as a correction factor for main PESERA model (hopefully with appropriate scale dependence)
Use of nitrogen fixing crops in rotations	MOR11, MOR12 (Morocco); TUR04 (Turkey)	Enable nitrogen and carbon budget components of PESERA	Show effect of fertilisation in enhanced crop yields etc.
Plastic sheeting/ greenhouses		Manage irrigated water use and increase winter temperatures. Suppress weeds.	May require increased pesticide use, and replacement of topsoil. Increased yield, especially of winter crops.

2.7 Assessment of the effects of soil and water conservation measures on runoff, soil loss and sediment yield at catchment scales

Work is ongoing to develop ways to calibrate/validate/evaluate PESERA within the DESIRE project. To do this, K.U. Leuven (Partner 2) developed a database with sediment export rates from river catchments in Europe, the Mediterranean World and the regions of the DESIRE hotspot areas outside Europe. The general objective of this sediment yield (SY) database in WB5 is to allow the calibration and validation of the (adapted) PESERA model and provide a framework to evaluate mitigation strategies at the catchment scale, considering their effects on the total sediment export. The established sediment export database allows for comparison of erosion rates, predicted by the PESERA model, with actual sediment export rates. This comparison will serve as a basis indication where eventual other sediment sources are important and where additional attention needs to be given to the PESERA model. This work will be reported fully in Deliverable 5.3.1.

3 Cost-effectiveness modelling

3.1 Introduction

This section outlines the socio-economic modelling approach developed for application across all DESIRE study areas. The approach is developed to integrate with the PESERA model, which is used to evaluate the biophysical consequences of alternative remediation strategies. According to the WOCAT terminology applied in WB3, remediation strategies consist of technologies and approaches[6]. A technology can consist of a single or multiple of four types of measures: structural, vegetative, agronomic and management measures, respectively (WOCAT, 2007). The cost-effectiveness modelling methodology consists of twelve steps which will be described below. These twelve steps form the logical modelling sequence and include both steps delivering relevant intermediate output and technical steps to allow progression to subsequent analyses. The intermediate outputs correspond to the following topics and methodologies:

1. Applicability limitations and spatial variation of investment costs (steps 1 and 3)
2. Evaluating effectiveness of technology investments using cost-effectiveness and financial cost-benefit analysis (steps 4-7)
3. Adoption of technologies and diffusion of innovations (step 8)
4. Economic (including wider economic effects) Cost-Benefit Analysis (steps 10-11)
5. Policy scenario analysis (step 12)

Before describing the twelve steps, we will first touch upon some key-issues of the above points. The first point deals with the planning and design of conservation technologies. All technologies, whether based on indigenous knowledge and dating back centuries or the result of recent scientific experimentation, are designed for specific environmental and socio-economic conditions. The design of conservation technologies is due to its practical aspect probably the most studied aspect of land management. Much of the design literature can be associated with large-scale government-led project interventions initiated to tackle land degradation problems, as guidelines and manuals needed to be prepared for training field technicians and providing them with rules on how to lay out selected measures (Wenner, 1981; US Bureau of Reclamation, 1987; Alaya *et al.*, 1993; WDLUD, 1995). Of later date (except perhaps early anthropological and historical accounts) are contributions documenting indigenous land management practices (e.g. Reij *et al.*, 1996). Many of those studies were inspired by the apparent lack of effectiveness of large-scale soil and water conservation campaigns that were rolled out across the developing world – one reason for which was concluded to be a lack of fit of the imposed ‘solutions’ to the realities of the farmers on who’s land they were implemented (Hudson, 1991). Of recent origin is the idea to share success stories in

[6] The cost-effectiveness modelling focuses on the technologies. Approaches are nevertheless also considered – in a similar fashion as policies, i.e. in valuation and/or adoption scenarios.

conservation across the globe to increase chances of cross-pollination (matching tradition with innovation). The WOCAT initiative (www.wocat.net) is the most well-known of such network efforts, and its approach was also adopted by the DESIRE project (Schwilch *et al.*, 2009).

A first step in tailoring conservation technologies to a specific environment is to establish the preconditions necessary for their implementation. WOCAT uses lists of environmental and socio-economic variables to label characteristics of localities where technologies and approaches followed for their dissemination have been effective. As the WOCAT methodology is designed to facilitate knowledge exchange, such relatively broad labels suffice. However, in spatially explicit modelling, a further refinement of applicability is necessary.

Cost-effectiveness analysis and cost-benefit analysis are economic evaluation methods used to select the best among several alternatives. In the case of cost-effectiveness analysis, (biophysical) effects resulting from the alternatives considered are evaluated as is, while cost-benefit analysis implies that all effects are translated in monetary units. They are distinct methods in that cost-effectiveness analysis needs an explicit (policy) objective against which to evaluate performance of alternatives, whereas cost-benefit analysis will select the best alternative given a series of cash flows of monetary costs and benefits for each of the alternatives and a discount factor. Cost-effectiveness analysis has been criticized for being arbitrary with regard to the subjective element of setting targets (de Graaff, 1996), while cost-benefit analysis has generated discussion over the possibility and desirability of attributing monetary value to all impacts of any government initiated project (e.g. intangible effects on biodiversity, human lives saved, etc.) and philosophical and technical discussions over what discount rate should be applied (e.g. Pearce and Turner, 1990; Arrow *et al.*, 1996; Almansa Sáez and Calatrava Requena, 2007). The latter is especially relevant when a societal perspective is taken (economic Cost-Benefit Analysis (CBA), as opposed to financial CBA), and particularly when decisions have to be made about environmental sustainability (discounting is essentially incompatible with long-term decision-making, leading to discussions over inter-generational equitability).

Even for the relatively straight-forward application of financial CBA, more frequently than not, ex-post analyses have shown that predicted rational adoption behaviour (based on profit maximising) has more often than not poorly been correlated to actual land user's behaviour. Land users face several challenges that are either difficult to incorporate or have often been neglected in financial CBA. Among such challenges are elements of risk (e.g. land tenure, climate, pests and diseases, price fluctuations), lack of access to knowledge, labour and/or capital resources, and socio-cultural and psychological factors (e.g. ineffective decision-making structures, power relations, inappropriate technology, cultural norms and values). There is a large body of research on the factors influencing adoption of soil and water conservation measures, which appears to come up with context-specific determining factors (Lapar and Pandey, 1999; Shiferaw and Holden, 2001; Tenge *et al.*, 2004). Several studies also indicate differences in factors determining initial and sustained adoption (Paudel and Thapa, 2004; Amsalu and de Graaff, 2007). Apart from the question *what* determines adoption, the issue *how* adoption processes take place is a pertinent one, dealt with broadly in the social theory of diffusion of innovations (Rogers, 2003). If we consider a sustainable land management technology to be an innovation, how will it, after its introduction, disperse among agents, in time and in space? Rural sociologists have since the 1940s extensively studied the diffusion of innovations in agriculture (including the pioneering study by Ryan and Gross, 1943), but its integration with GIS offers the potential to put much more emphasis on the spatial dimension then has hitherto been possible.

Notwithstanding the difficulties associated with the methods introduced above, policy-makers face an acute level of urgency in dealing with land degradation. Important questions are how to motivate land users to adopt more sustainable production methods, what policy instruments to use and where to focus attention. These complex issues can only be resolved by making assumptions and simulate decisions in scenario studies using modelling approaches. Hence, we depart from the assumption that although other factors may limit adoption, a positive expected return to investment (as calculated with CBA) is a precondition for a technology to be taken up by land users. By applying CBA, an upper boundary for potential adoption can thus be inferred. More realistic adoption dynamics can subsequently be included by using additional models (e.g. ABM, Section 4.1). Policy-makers may employ a range of instruments to stimulate adoption; however, their criteria to do so will depend on various measures of cost-effectiveness. By combining cost-effectiveness analysis, CBA and ABM in one model (or, as in our case, several coupled models), it becomes possible to evaluate a multitude of 'what if' options. As validation of scenario studies is notoriously difficult, the possibility to evaluate the effect of assumptions made is an essential feature of complex models. Moreover, as our models are embedded in a participatory approach (information will be taken up and results evaluated in workshops with stakeholders), they will receive an additional validation or reality check.

The following text describes each of the 12 steps in the socio-economic modelling approach developed for application across all DESIRE study areas. For clarity, a summary is provided at the end of each step, showing the required data inputs and intermediary outputs that are used in other steps. Figure 3.1 presents a graphical representation of the cost-effectiveness model and its interrelations with other models.

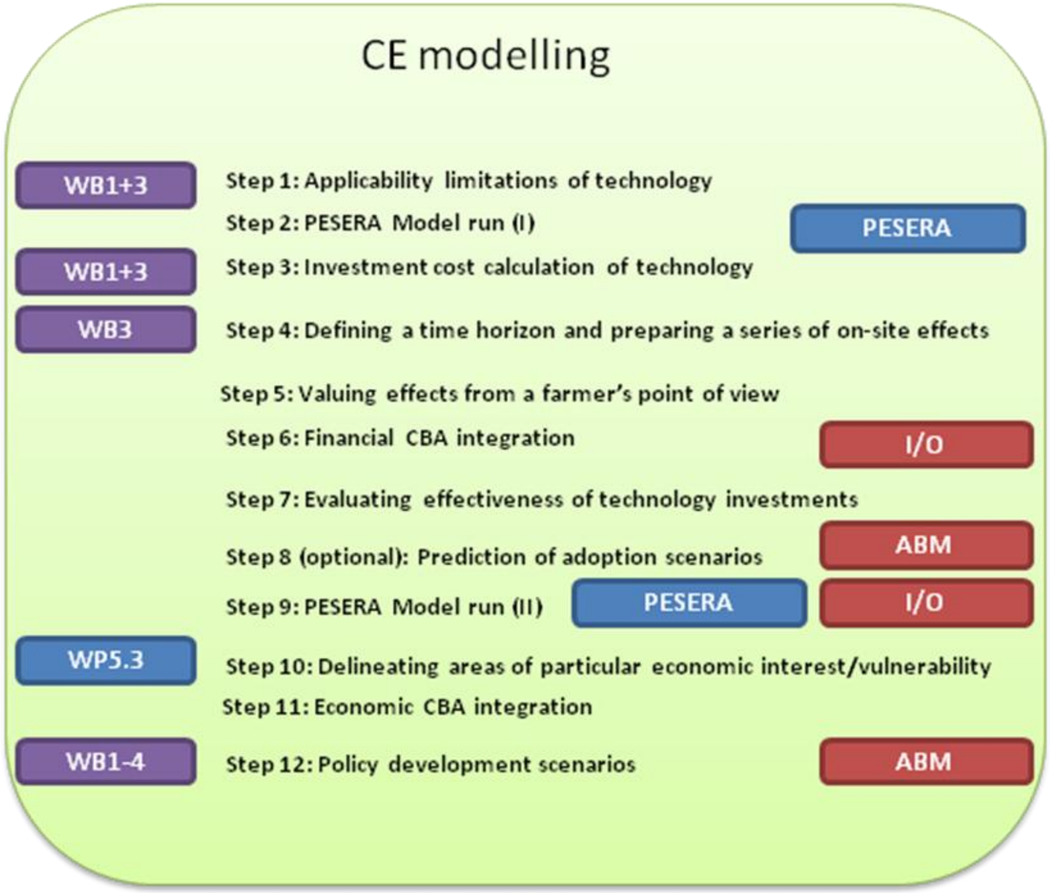

Figure 3.1: The twelve steps of cost-effectiveness modelling and interrelations with other models and project work blocks

3.2 Step 1: Defining applicability limitations of each technology

The WOCAT Technology Database (based on the Technology Questionnaires completed in WB3) presents data on the technologies that can be considered for each study area[7]. A technology may have several basic requirements that must be met for it to be implemented. The purpose of this phase is to rule out any part of a study area that is not suitable for the application of the technology being evaluated. Limitations as meant here are physical constraints, rather than factors reducing expectations that the technology will be cost-efficient[8]. For example, it is not possible to construct terraces on steep slopes with shallow soils. Only when all applicability limitations of a technology are satisfied can the

[7] The WOCAT Technology database also allows considering technologies documented for other study sites to be considered along with technologies reported by individual study sites
[8] These socio-economic factors will be addressed in the CBA

technology be applied in a certain area. Currently, the following options are available to define applicability limitations[9]:

- Soil depth interval[10]: the rootdepth input layer for PESERA is used as a proxy)
- Slope gradient interval: this can be extracted using the 'Slope' tool from Arctoolbox to extract slopes from a Digital Elevation Model (DEM)[11]
- Landform: landform can be defined based on existing landform maps, or alternatively by using the 'Curvature' tool from Arctoolbox which extracts curvature from the DEM. Convex areas are shoulder areas whereas concave areas are footslopes. A subsequent operation is needed to distinguish between plateaus and valley bottoms, which both are characterised by low or zero curvature values
- Land use: the model uses the simplified legend from the CORINE land cover mapping project proposed for the PESERA land use input data layer (Irvine and Kosmas, 2003). Where necessary the more specific original categories of CORINE[12] can be used. Even more detailed local land use categories may be used as long as they are coded as further subclasses of the CORINE land cover classes. The maps to be constructed under WB1 are the preferred source, although a conversion process to match CORINE land use legends may be necessary.
- Temperature: currently an interval based on average annual temperature (constructed as the average mean monthly temperature of the twelve monthly rasters used as input for PESERA) can be specified
- Precipitation: currently an interval based on total annual precipitation (constructed as the sum of mean monthly precipitation of the twelve monthly rasters used as input for PESERA) can be specified
- Distance to stream: from the DEM, a drainage network can be constructed using the 'Flowdirection' and 'Flowaccumulation' tools from the Arctoolbox and subsequent distillation using Map algebra. To improve the quality (hydrological correctness) of the DEM, a "fill sinks" operation can be applied. Alternatively, a drainage network digitized from maps or aerial photos can also be used. Either way, a buffer operation can be applied to specify the maximum distance to streams required by a certain technology
- Depth of groundwater: currently not implemented due to lack of data, but if an input layer is available this criterion can be considered as well
- Combinations of the above criteria can be considered by querying the relevant layers (optional)

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[9] Additional applicability limitations might be defined, e.g. considering soil texture for water storage or stability, contributing runoff area for water harvesting, or altitude to replace climate when only coarse scale data are available

[10] The concept of 'interval' in the limitations identifies the range of conditions of a property within which a technology can be applied. It allows for setting minimum values, maximum values or both.

[11] A possible alternative approach is to relate slope gradient to standard deviation of elevation, which is used as an input raster for PESERA.

[12] http://etc-lusi.eionet.europa.eu/CLC2000/

For each technology, each of the above criteria will result in an output map showing the applicability in a dichotomous fashion. Only raster cells where all conditions are met will finally be classified as the applicability area for the technology considered. Where maps of conservation areas (WB1) have been prepared that identify areas already treated with specific technologies, those can be used to refine the potential applicability area.

Data sources: WOCAT Technology Questionnaire (TQ); PESERA model input data; DEM of the study area (e.g. SRTM 90m); WB1 maps of conservation efforts; optional additional input data.

Intermediate product: Raster layer showing the potential applicability area of each technology (for a schematic example, see figure 3.2).

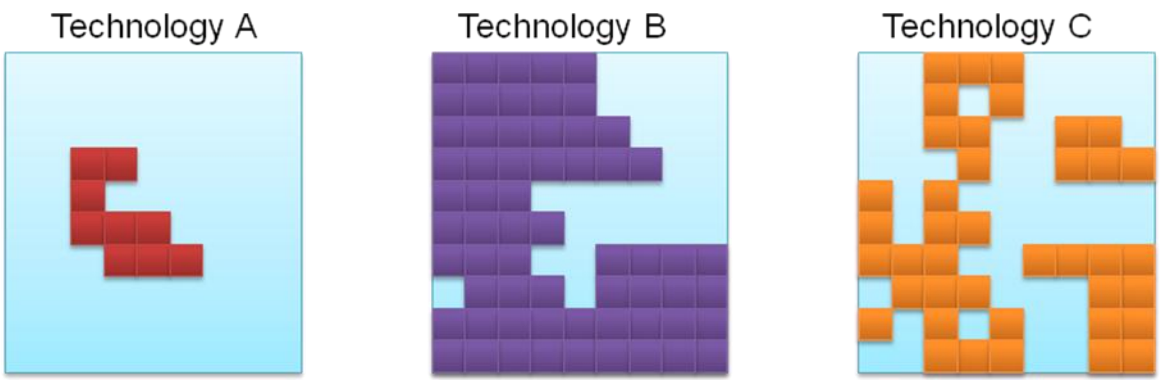

Figure 3.2: Hypothetical applicability areas for three technologies

3.3 Step 2: PESERA Model run 1

The physical effects of implementing the technology can now be evaluated using the PESERA model. This should be done separately for each technology, taking into account its potential applicability area (step 1). The resulting output should be compared to the situation without implementing any (additional) technology. This output, which can be considered the technically feasible mitigation of land degradation, provides a meaningful PESERA output, but is not yet based on socio-economic variables. From PESERA runs, which include stochastic climate events, temporal trend series will be constructed that will simulate the gradual effects that many technologies may have (e.g. gradual build-up of soil organic matter, resulting in gradually declining erosion levels and gradually increasing yield levels). Timing of biophysical effects is crucial for subsequent economic valuation (steps 6 and 11). Thus, this step will produce annual output maps for a time series of 20 years (assuming no technology will have an economic lifetime longer than that, see step 4).

The PESERA model, adapted to evaluate the biophysical effects of land degradation remediation technologies (section 2) is a grid-based model that can be applied at a grid cell size of 0.01 – 1 km^2. Table 3.1 shows a list of typical output variables generated by PESERA. The cost-effectiveness model will produce output at the same resolution as PESERA. However, some intermediate results may be available at finer (or coarser) resolution. Where this is the case subgrids (or supergrids) will be defined that can be generalized in the final resolution.

Table 3.1: Typical output variables for each cell in the PESERA model (Kirkby *et al.*, 2008)

Output parameters PESERA	Unit
Erosion (monthly)	tons/ha
Overland flow runoff (monthly)	Mm
Soil water deficit (monthly)	Mm
Percentage interception (monthly)	%
Vegetation biomass (monthly)	kg/m^2
Cover monthly (if not pre-set by land use)	%
Soil organic matter biomass (monthly)	kg/m^2

Data sources: 1) PESERA input data; 2) raster layer showing potential applicability of each technology (step 1)

Intermediate products: annual maps of PESERA output values for with and without situations for each relevant technology in each site (Figure 3.3)

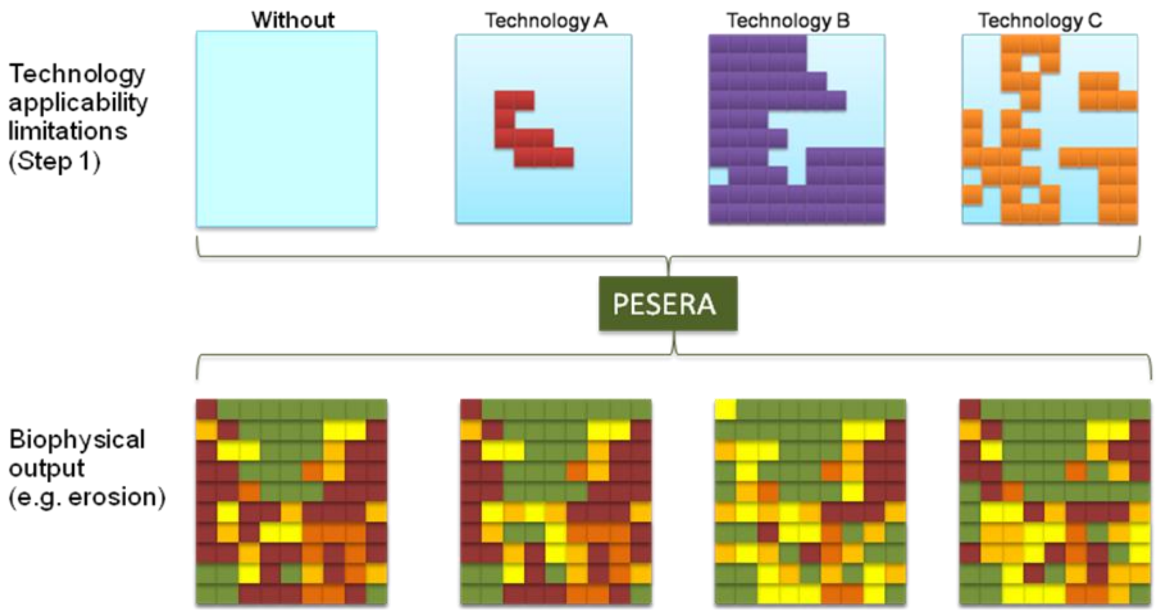

Figure 3.3: Example output maps for the first PESERA run for different with and without situations

3.4 Step 3: Investment cost calculation of the technology

The WOCAT QT in most cases presents a cost estimate of the technology. However, this estimate is made for its most common application area or is an informed estimate of average costs across several local application areas. In reality, construction costs will differ based on environmental factors (e.g. slope) and socio-economic factors (e.g. distance to market). The same holds for maintenance costs. In this phase, investment and maintenance costs will be made spatially explicit by considering both types of factors. For each type of input the location of source areas (markets) needs to be identified and transportation costs defined. The following points present the pathway to arrive at spatially-explicit investment and maintenance costs:

- Data preparation: the data contained in the WOCAT database will be stored in a modified database to allow better querying. This also involves the inclusion of rules of how inputs would vary with environmental conditions. Data on source areas (markets), costs of inputs in those source areas, transport methods and associated costs, and infrastructure will need to be prepared. Further details about this will be sent to study site coordinators in due course. Quantities and unit costs will be assessed separately;
- Variation of input requirements with varying environmental conditions: a cost breakdown for the standard situation is provided in the database (WOCAT QT data). Through a query, a single line of total quantity per input category per technology will be extracted. This is the quantity for a standard situation. By using technology-specific rules, the standard quantities per input category will be linked to the environmental conditions in each grid cell;
- Calculation of the cost of inputs at the destination area: network analysis will be used to specify the price of inputs in nodes of an infrastructure (road) network, assuming the cheapest transport path. From there, the costs of transport to individual grid cells will be added;
- Labour: this is a special case as it entails multiple return journeys. This does not only affect the labour (opportunity) cost, but also the total amount of labour needed. Hence, for the case of labour, distance-related travel time will be considered to reduce the effective 'on-site' person day from 1 with a proportionate fraction. If this effective person day is e.g. 0.8, a total 'on-site' labour input of 40 will then require 50 person days (possible transport costs for 50 return journeys would still need to be added when the journey is not made on foot);
- Multiplying spatially-explicit inputs with their respective spatially-explicit costs gives the total investment or annual maintenance cost.

While the above presents a general methodology, some deviations may be necessary. For example, some inputs may be assumed to be present everywhere (e.g. earth) without the need to consider transport costs. Some, like wood or stones, could be linked to input map layers to consider patchy availability across a landscape. The source area for labour may be hard to define, as where farmers live and where their fields are is very likely unknown. One solution to this could be to assume an average estimated distance from farm to field.

Alternatively, in cases where population lives in village centres (and those are digitized as a GIS layer), the distance from farm to field can be assumed to be that of the closest distance village centre – grid cell.

The above refers to investment cost calculation for new technologies. Where SLM technologies are already existing and their extent is mapped (WB1), treated areas can either be considered as areas where no investment is needed (taking only maintenance costs into account), or as areas where upgrading is necessary to reach the design standards of newly constructed technologies (where upgrading can be a variable percentage of the standard construction costs; maintenance costs are fully accounted for).

Data sources:

1. WOCAT QT – further cost breakdown may need to be requested from study sites
2. Estimates of the variation of implementation costs for each technology with the most important environmental factors need to be requested from the study sites and complemented by literature research (an important source may be construction manuals from extension services if it concerns technologies that are already being implemented)
3. Additional input data required for distance-related costs are:
    a. Source point/area vector data needed for all input types (or assumed to be available everywhere)
    b. Price data for all input types (prices in source area)
    c. Infrastructure data needed and their accessibility for different transport categories
    d. Table of cost, time (labour), and capacity per transport type and distance unit
4. Maps of existing SLM technologies (WB1) and data of their current quality/maintenance status or age

Intermediate product: Raster layers showing the spatially explicit investment and maintenance costs of implementing a technology (for an example see Figure 3.4).

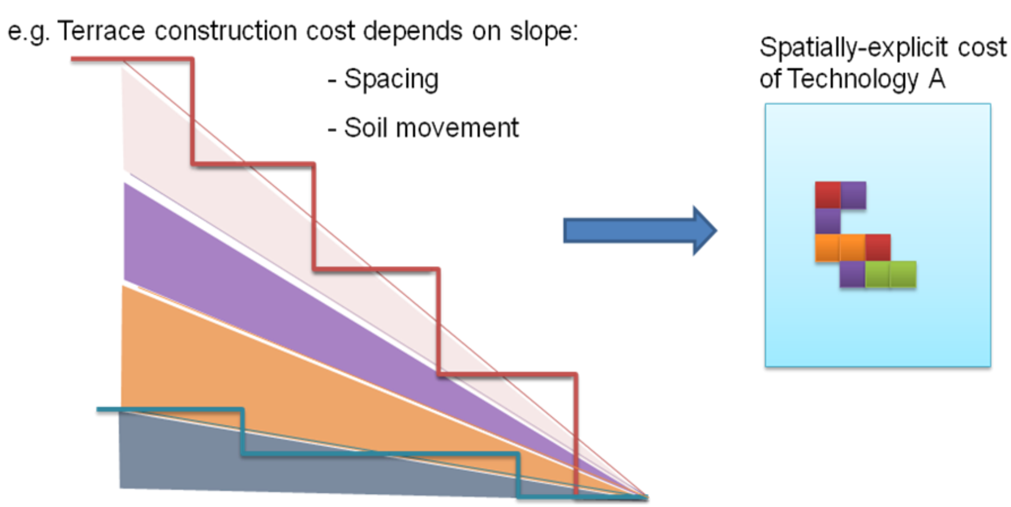

Figure 3.4: A schematic example of calculating spatially-explicit investment costs of a technology

3.5 Step 4: Defining a time horizon and preparing a series of on-site effects

This step first of all compares the lifetimes of the technologies considered. For the subsequent cost-benefit analyses (step 6 and 11), it is important that technologies are considered over the same time horizon. However, individual technologies may differ widely in their lifetimes. PESERA can be run for 20 years for all technologies: few technologies are expected to have a lifetime longer than this, and even if they had, the use of discounting (see step 7) makes the contribution of effects distant from now increasingly insignificant. However, if technologies are compared that have a lifetime of e.g. 1, 3, 5 and 15 years, the appropriate time horizon would be 15 years for the last technology, and several cycles of re-investment (respectively 15, 5 and 3) would be assumed for the other technologies. The assumption of re-investment has a drawback: as conditions change over time (as evidenced by the annual output of PESERA), the investment decision is actually not the same each time. A technology may become either more or less attractive with each cycle. However, for the sake of keeping the model simple a single cost-benefit analysis will be performed for the potential multiple cycles of re-investment. On the other hand, it is often justifiable to assume that effects will only gradually develop, and several cycles of re-investment of short-lived technologies might be necessary to realise these effects. For example, a single year of applying no-tillage will not lead to significant build up of soil organic matter; it is the sustained adoption (annual re-investment) that delivers the desired effect. Thus, even if all technologies considered would be short-lived, it would still be sensible to consider a minimum re-investment cycle. If multiple re-investment cycles can be accommodated within the 20 year time-span, the number of cycles resulting in the highest Net Present Values (NPV) in the CBA should be preferred. Where appropriate, a distinction can then be made between initial and sustained adoption of the technology.

When a time horizon has been defined, the data need to be prepared for subsequent valuation (step 6). First of all, investment costs are entered in the relevant years they (re-)occur. Secondly, maintenance costs are added for the years between (re-)investments. In addition to investment and maintenance costs, production costs will be considered. This is important as: 1) adopting a remediation technology may imply a change of land use; and 2) production costs with or without a remediation technology may be different (in some cases the technology modifies the use of inputs, in many cases the technology will lead to increased yields, which require higher labour input for harvesting). In cases where no or reduced investment costs were considered for existing technologies (Step 3), this only holds for the first investment cycle; in all potential re-investment cycles, full investment costs will be attributed.

Data sources: 1) WOCAT QT data; 2) production costs in the with and without case (see also step 3); 3) PESERA output data

Intermediate products: 1) Data series for on-site effects for the time horizon for all pairs of with/without situations;

3.6 Step 5: Valuing effects of the technology from a farmer's point of view

Once the data series for on-site effects are prepared (step 4) and PESERA output data are available (step 2), the effects can be priced (for input costs this has already been done in step 4). Although most effects are expected to be regarded as benefits, some costs may result as well. A common cost is for example the reduction of cultivable area that results from many structural measures. The most straightforward benefit would be a yield increase, which could be valued at the local market price for the crop grown. If the technology implies a land use change, the gross benefit consists of the difference between the crop return in the with/without situation. Besides valuing the outputs of the PESERA model, a farmer may value effects of the technology that are not considered in the model. In the WB3 workshops, indicators were suggested by stakeholders that are not simulated by the PESERA model. These (local) indicators should where possible be valued separately and added to the benefits that are derived from the PESERA model[13]. Also, the effect of policies need to be taken into account (possible scenarios can be fed back to this step from step 12).

Data sources: 1) Data series for on-site effects for the time horizon for all pairs of with/without situations (step 4); 2) valuation of local indicators (optional); 3) data on policies affecting the farmer valuation (subsidies).

Intermediate product: annual cashflow series[14] for farm finances (Figure 3.5)

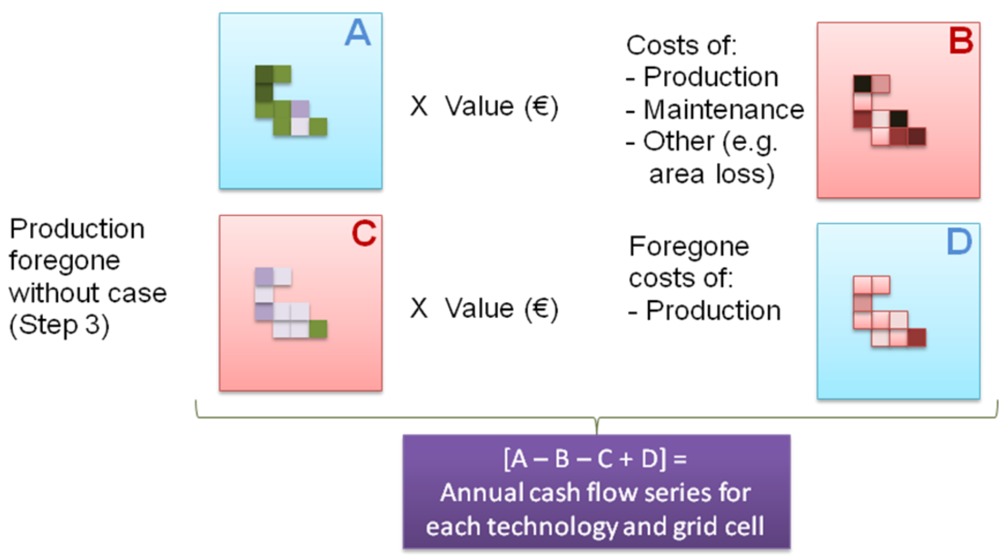

Figure 3.5: Procedure to calculate annual cashflow series

[13] In order to be able to put a value on local indicators, further research may be required. In some cases these indicators will be used to monitor the effect of mitigation strategies in WB4 trials. Where this is the case, the results from these trials can be used

[14] Annual cashflows are the net result of the sum of benefits obtained from implementing the technology minus the sum of costs incurred. Both benefits and costs should be considered comparative to the without case

3.7 Step 6: Financial CBA integration

The steps 1-5 allow quantification and valuation of all effects considered, for each technology separately. In this subsequent step, a formal decision based on farmer profit maximisation will be made for each grid cell. This entails applying a discount rate[15] to the annual cashflows generated by each technology, and determining Net Present Values (NPV) and Internal Rates of Return (IRR)[16]. NPV is obtained by summing discounted cashflows. The IRR is the discount factor at which the investment becomes attractive (i.e. at which the NPV is 0). The discount factor can be established from the cost farmers incur to borrow money. The rationale behind this is that borrowing money to invest should at least yield the sum borrowed plus interest and transaction costs[17]. Adoption of the most profitable technology (based on NPV and/or IRR) will be assumed (this may also include non-adoption, if none of the technologies evaluated result in sufficient tangible benefits for the farmer). For each grid cell, one of the following three possible outcomes will therefore apply:

a. The technology with highest NPV will be selected (when positive);
b. No technology will be selected if all NPV's are negative; and
c. No technology will be selected if no technology is applicable in the area.

The above standard method evaluates the profitability of investment alternatives when capital and land are the scarcest factors. This may not always be the case. The following additional analyses can therefore be undertaken:

1. For many farmers labour may be a critical factor, and return to labour rather than return to capital investment is important. Especially where off-farm work opportunities exist, return to labour should be considered in relation to the opportunity cost of labour. If this comparison turns out negative, then investing in the land is doubtful, and abandonment would be a sensible strategy
2. Labour itself may also be limiting. In areas with ample land availability and characterised by the absence of a labour market (especially when land is held as a common property resource), it could be assumed that technologies would be adopted in the areas of highest profitability until available labour resources are exhausted. The model can handle this by attributing finite labour pools to source areas (villages) which will impose a limit on the adoption of technologies

These additional analyses can be performed optionally after the general analysis has been completed.

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[15] The function of a discount rate is to reflect the cost of capital. It results in time preference: when comparing equally priced and equally effective alternative remediation strategies, the choice clearly falls on the technology that delivers those benefits sooner.

[16] When investment costs of alternative technologies are of the same order of magnitude, the NPV is a good economic indicator. When they are very different, the IRR is a better indicator.

[17] For example, Nelson *et al.* (1998) using this method for a Philippines case study apply a discount rate of 25%, which could be reduced by appropriate policies to an estimated 10%.

Data sources: 1) annual cashflow series (step 5); 2) discount factor; 3) (optional) additional data for labour opportunity costs or labour availability.

Intermediate products: 1) raster map with spatially explicit IRR/NPV of each technology (Figure 3.6); 2) raster map with combined constructed adoption of all technologies (Figure 3.6); 3) raster map with spatially explicit IRR/NPV of adopted technologies.

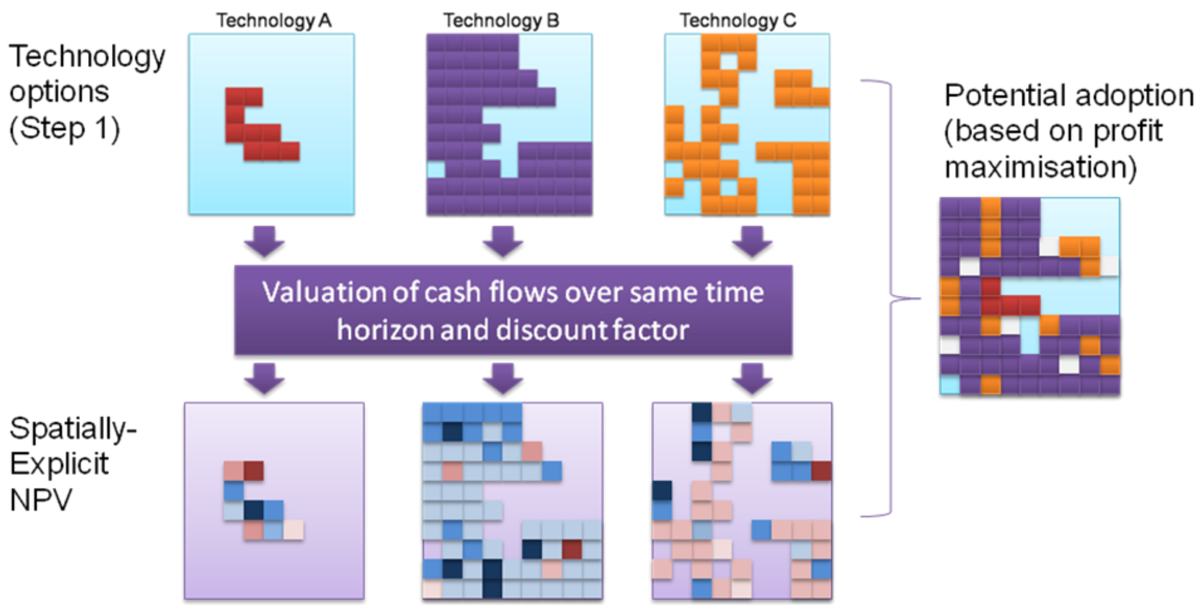

Figure 3.6: Construction of maps with spatially-explicit NPV of each technology (bottom) and resulting adoption map (right)

3.8 Step 7: Evaluating effectiveness of technology investments

In this step, the socio-economic analyses done so far will be contrasted against the biophysical impact of technologies according to PESERA. The first method to do this is cost-effectiveness analysis. This analysis allows for an assessment of biophysical effects before valuation of technologies from either a farmer or societal point of view. The analysis may be repeated to capture two relevant criteria:

1. The investment costs of a technology relative to its effectiveness to reduce land degradation problems. The investment required may limit a land user or group of land users' capacity to apply a technology. The height of such an initial investment relative to the expected mitigation of land degradation is a policy-relevant indicator. As shown in the next criterion such costs should not necessarily be borne totally by public investment, but could form an essential step in designing financing mechanisms to facilitate adoption; and
2. The total non-discounted cashflow of a technology relative to its effectiveness in reducing land degradation problems. Technologies will usually only be adopted if

the land user can expect a return to his investment[18]. From a policy perspective, this return (e.g. increased crop yields) is a private gain that in itself can stimulate uptake of the technology without a need to spend public resources. The real cost to society of undertaking the investment should thus subtract private returns to investment. The justification for a potential contribution by society to investing in land degradation mitigation technologies should offset costs against effect.

The effectiveness of technology investments can also be evaluated using the NPV values resulting from financial CBA (step 6). In this case the farmer valuation of a technology is compared with its effectiveness to mitigate land degradation. Negative NPV's signal cases in which policy incentives could be needed to motivate land users to adopt the technology. When the previous cost-effectiveness analysis using the total non-discounted cashflow is positive but the repeated analysis with NPV is negative, it can be concluded that the land user perceives benefits too late relative to the cost incurred. It is important for policy-makers to be aware of such a potential time lag.

In addition to the above, cost-effectiveness analysis provides a quick overview of the technical possibilities of remediating land degradation and the costs involved. A meaningful comparative cost-effectiveness analysis requires a (soil conservation policy) target to be set. Two different approaches can be taken:

1. Setting a general target: when a general target is set, remedial action should be taken in all grid cells until they meet the target. For some grid cells the target may be reached at limited cost, but in order to reach the target in the last cells, very costly interventions may be necessary (and some cells might even then not reach the target, or perhaps none of the technologies considered is applicable). For this analysis, the cheapest investment meeting the target or the investment best approaching the target will be selected; and
2. Setting an aggregate-level target: in this case, e.g. a global reduction target of 30% of soil erosion is specified. A map algorithm will be used to reach the target at minimum cost.

The above approaches will likely yield distinct conservation strategies. In the first case, 'hotspot' areas will have to be treated at high cost. In the second, those areas in an advanced state of degradation will be sacrificed and remediation efforts concentrated on less expensive moderately affected areas. The policy targets set can be part of a scenario study (step 12).

Data sources: 2) PESERA output data (step 2); 1) Investment costs, total non-discounted cashflows and NPV (steps 3-6); 3) Target levels for (selected) PESERA output variables.

Intermediate products: 1) raster maps of cost-effectiveness of each technology for biophysical effects (see Table 3.1), e.g. expressed in monetary units spend (on a remediation technology investment) per unit of soil loss prevented (comparing with/without situation, see example in Figure 3.7); 2) raster maps of the spatial

[18] Assuming that profit maximisation is the aim and if the land users are not forced by authorities to implement technologies.

configuration of selected technologies to meet general or aggregate-level (policy) targets; 3) graphs showing the marginal cost vs. target fulfilment relationship.

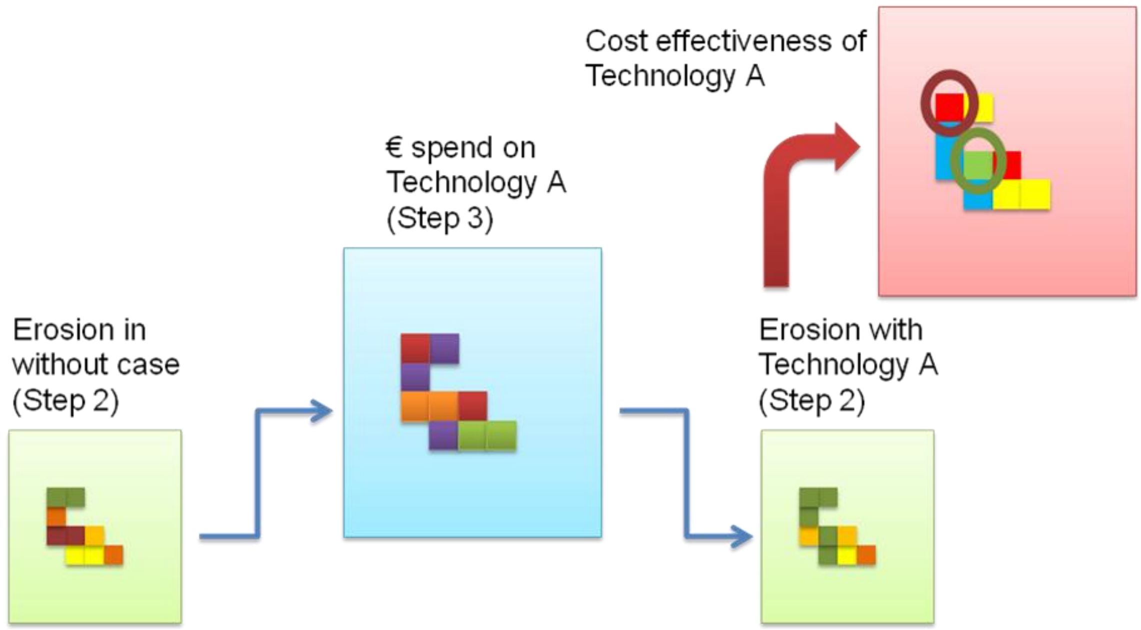

Figure 3.7: Hypothetical example of calculating cost-effectiveness of a technology

3.9 Step 8 (optional): Prediction of adoption scenarios

The assumed adoption raster map of step 6 will be fed into PESERA again for a cost-benefit analysis including selected wider economic effects. Adoption in step 6 is however crudely defined by assuming that: 1) farmers will adopt the most profitable technology regardless of any constraints/reservations they might have; and 2) they will do so unanimously and at once. This optional step allows for the delineation of the temporal and spatial trends of adoption. The simplest form would be to estimate a likelihood of adoption (e.g. as a function of IRR of the investment), and apply this as a random process of allotting technologies to grid cells – potentially including a time line (based on evidence from the literature) to reach the estimated level of adoption. More elaborate adoption scenarios could include social factors such as how innovations spread through networks. This is also the step in which the effect of policies (or approaches – data from WOCAT Approach Questionnaire) on adoption can be assessed (step 12), i.e. how changes in policies/approaches will affect adoption patterns. Since this step is additional to the work that was originally proposed, it will only be done if time and resources permit. If it is not possible to perform this step for additional study sites, we will only conduct this step in the Spanish site, using an Agent-Based Modelling approach. Adoption of mitigation measures as defined in step 7 is solely based on profit maximization. Land users may in reality have more varied reasons to adopt or not to adopt SWC measures. For the Spanish case study site, we will do a detailed study of spatially explicit adoption. In this case study, a questionnaire will yield the necessary data for developing an agent-based model integrated with GIS (see section 4.1). The results of the ABM model can be entered here in lieu of the raster map showing the spatial configuration of technologies produced by step 7.

Data sources: ABM model output (section 4.1) or rules based on economic indicators

Intermediate product: (revised) map with spatially explicit adoption of technologies

3.10 Step 9: PESERA Model run 2

When a technology adoption map (step 6 or 8) is ready, this can be fed back into PESERA to evaluate the impact on output variables. In this second PESERA model run, grid cells will either feature the technology with the highest NPV (if the technology adoption map from step 6 is used) or the technology as predicted according to the adoption pattern simulated in step 8[19]. This PESERA model run thus differs from the first run in that it will be based on a configured technology adoption map instead of separate runs for each technology taking into account its applicability limitations. It will be interesting to map PESERA output data for their own sake, either by visualizing the final physical impact of land mitigation strategies or as a reduction relative to the without situation. Such output maps can be used to inform policy development scenarios (step 12) and to integrate these effects in a CBA including selected wider economic effects (step 11). Apart from maps, also aggregate accounts for each of the output variables can be elaborated.

Data sources: 1) PESERA input data; 2) map with spatially explicit adoption of technologies (step 6/8)

Intermediate products: annual maps of PESERA output values (Figure 3.8)

[19] Note that both step 6 and 8 may produce technology adoption maps with no feasible technologies for particular grid cells.

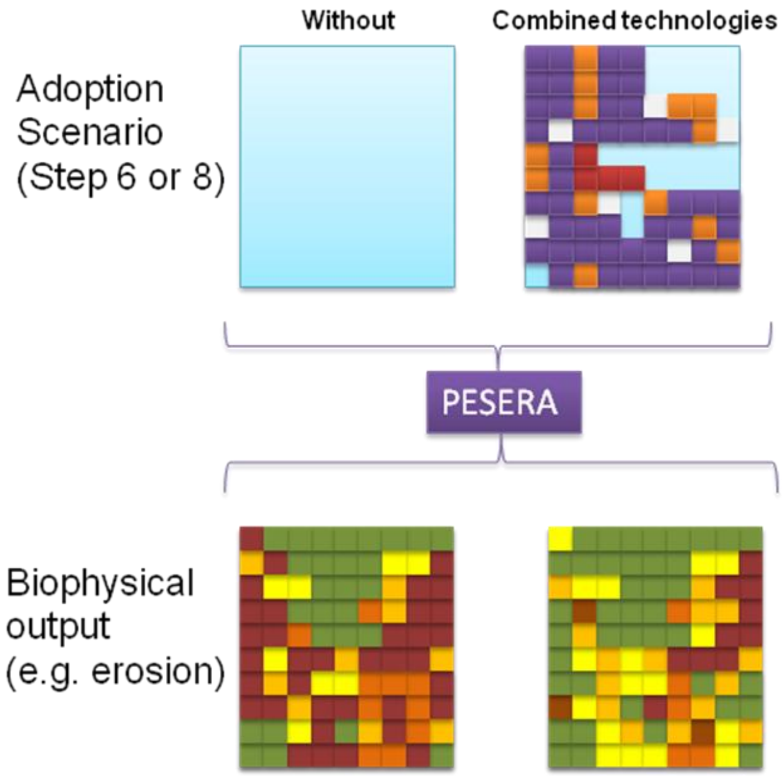

Figure 3.8: Example output maps for the second PESERA run for combined technologies and without situation

3.11 Step 10: Delineating areas of particular economic interest/vulnerability to off-site effects

Besides on-site effects, remediation technologies may also mitigate off-site or downstream effects of land degradation. Some of these off-site effects will be particularly relevant for confined areas. As PESERA has its limitations in that it treats grid cells as independent units, it cannot be used directly to assess off-site effects. This step details how PESERA can still be used to assess off-site effects in an indirect fashion. Thereto, first for each effect the area of interest will be delineated. Delineation of such areas should be done both for a with and without situation to allow for the subsequent valuation of differences in economic terms (step 11). The most relevant (and currently the only) off-site effect which can be considered for soil erosion is reservoir sedimentation. Use will be made of the sedimentation database developed by KU Leuven (Partner 2), of which some details are included in section 2.2[20]. The remainder of this step explains the method for doing this.

A map layer with reservoirs should be created, and its attribute table be completed with details of construction costs and reservoir volume. For each reservoir the contributing catchment area will subsequently be calculated. PESERA output (step 9) includes soil erosion values for each grid cell. These should be summed for the catchment

[20] A full description of the sedimentation database will be given in the forthcoming Deliverable 5.3.1.

areas. The relation between the soil loss from grid cells and reservoir sedimentation is formed by a Sediment Delivery Ratio (SDR) (i.e. only a certain fraction of sediments leaving the area end up in the reservoir). Section 2.2 briefly reports on ongoing work to construct a database of SDR values. Rules will be applied to select the appropriate SDR for each reservoir (catchment area). One way of doing this is to establish a SDR based on the distance from a grid cell, through the drainage network, to the reservoir. Multiplying soil loss with SDR gives a cell's contribution to annual reservoir sedimentation. Summing all individual cell contributions and multiplying sedimentation with specific density of sediments, the reservoir volume capacity lost can be calculated, and (in step 11) valued.

Data sources: 1) reservoir map layer with relevant attributes; 2) relevant PESERA output maps (soil erosion)

Intermediate products: 1) maps with areas of particular economic interest/ vulnerability for both with and without situation to off-site effects; 2) maps/data of (areas with) calculated actual damage (i.e. reservoir volume capacity loss)

3.12 Step 11: Including wider economic effects in CBA

From the societal point of view, undertaking remediation measures may be valued differently than for individual land users. In this step we will include selected off-site effects (e.g. reservoir sedimentation) and, where appropriate (e.g. for sites that have reported reforestation as a technology), extend the time horizon of the CBA to account for effects beyond the immediate interest of the land user. A (reduced) social discount rate would need to be adopted as well. Off-site effects can be considered by valuing effects on areas of particular economic interest/vulnerability, currently limited to reservoir sedimentation. To account for reservoir sedimentation in the CBA, a valuation based on estimated costs/benefits needs to be made, evaluating the combined effect of technologies adopted on the sedimentation rate compared to the without case. The without case situation defines the actual lifetime of the reservoir. We will assume that re-investment will be necessary once the reservoir capacity is reduced (e.g. by more than halve). The combined technologies in case of adoption may prolong the time before re-investment is needed. Entering the re-investment cost values in the cashflow in the concerning year will make a contribution to the NPV[21]. Besides extending the CBA with the above effects, economic prices of inputs and outputs will have to be used instead of farm gate prices (i.e. possible subsidies to land owners should be subtracted).

Data sources: 1) output from previous steps (2, 6, 9, 10); 2) economic prices of (farm) inputs; 3) economic prices of off-site and next generation effects.

Intermediate product: a CBA including selected wider economic effects for the study area.

[21] In the unlikely case that the use-value of the water is known, the effect of a higher water volume available can be used as well.

3.13 Step 12: Policy development scenarios

The previous steps 1-11 allow for an analysis based on individual land user decisions (based on assumed profit maximisation). For decision-making at higher hierarchical levels (i.e. regional, national authorities), step 7 introduced the possibility of setting environmental targets to evaluate the effectiveness of technology investment. However, doing this has no repercussions on land users' individual decision-making. In this step, the feedback of different policies (either in an implementation or design phase) on the valuation of effects from a farmer's point of view (step 5) can be considered, potentially leading to a different adoption pattern of technologies (step 6/8), with implications for wider economic effects (steps 9-11). Policies can for this purpose conceptually be defined as: 1) subsidies for specific (combinations of) technologies/landscape characteristics; 2) penalties for specific (combinations of) technologies/landscape characteristics; 3) subsidies/penalties for specific on-site environmental targets; and 4) measures altering prices for individual land users. The basic decision-variable assumed by the model remains the financial attractiveness of technologies to individual land users. If technologies cease to be attractive, farmers will be assumed simply not to adopt the measure[22].

Given the preceding possibilities, policy development scenarios can be fed into the model to evaluate their effects on financial attractiveness, adoption rates, and the environment. Any policy scenario of the above types 1-4 could be addressed. In order to perform meaningful analysis, we will link this step to policies or policy recommendations produced in other deliverables of the DESIRE project. These policy scenarios include the most relevant site specific policies identified in WB1 and policy-relevant scenarios (e.g. dealing with climate change and land use change).

Apart from policy scenarios, the WOCAT Approach Questionnaire reveals how in a certain study site, remediation technologies are being propagated. Where incentive structures of (project-based) approaches resemble policy types 1-4, their effect can be included. Where appropriate, the current step will at least be applied to evaluate the locally adopted approach.

Where off-site policy targets are involved, automating calculations would get extremely complex and therefore not feasible to implement. A work-around should this occur is to translate off-site targets into assumed on-site targets. For instance, if an off-site target would be to decrease reservoir sedimentation by 50%, this could be translated in an on-site target of say decreasing erosion by 30% (the model here described could be used to calibrate the translation terms).

Data sources: for the standard analysis: data provided by the WOCAT Approach Questionnaire; for additional analyses: selected policy scenarios with explicit financial data from other DESIRE deliverables (produced in WB1-4) or external sources.

[22] If for example, legal requirements need to be addressed, policies enforcing adoption could be entered in the model from step 8 onward and may then possibly imply a negative return to the complying land user. It would even be possible that such policies enforce adoption of measures outside the applicability limits of step 1 – potentially leading to environmental risks.

Products: all relevant intermediate products; policy-specific tailored output maps (e.g. Figure 3.9)

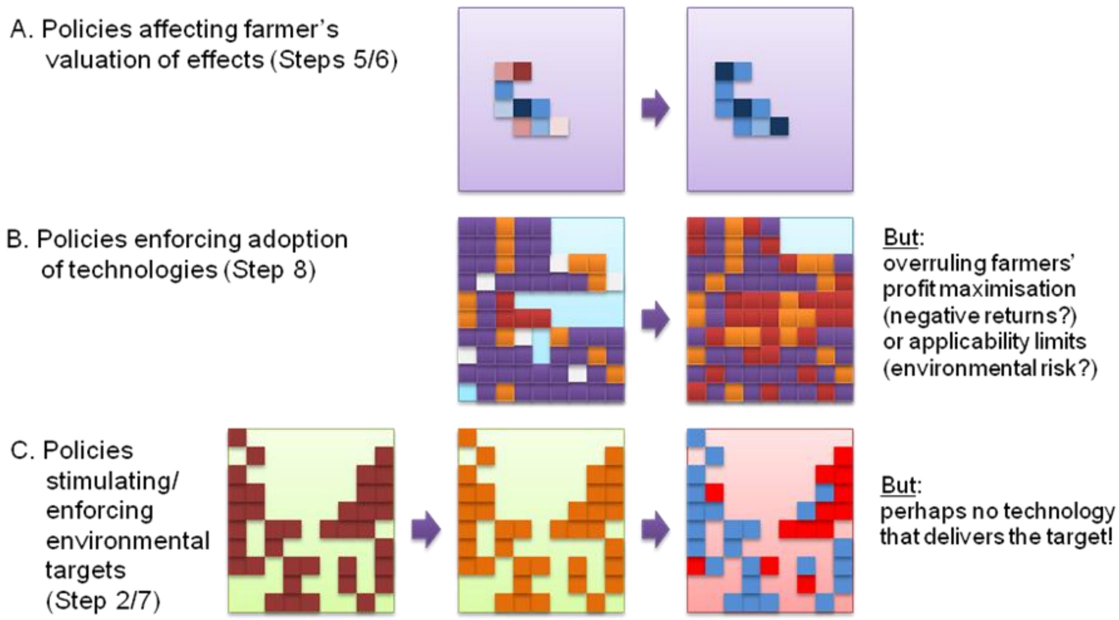

Figure 3.9: Example policy scenarios, including steps in which changes will occur and potential results

4 Agent-Based and Input-Output modelling in Spain

4.1 Agent-Based Model

Using the Torrealvilla catchment, extending over an area of 250 km^2, of the Guadalentin basin (Spain) as a case study, a model is being constructed to determine what factors influence the decisions of land managers to adopt different remediation strategies and to change land uses under different future scenarios using Agent-Based Modelling. By investigating the effects of different policy scenarios on these decisions, it will be possible to evaluate how different policy options may affect adoption of different remediation strategies and land uses across landscapes, and evaluate the biophysical consequences of such changes. The following text describes the modelling framework that has been developed, and on going works involved to implement it.

4.1.1 Phase 1: Exploring land use change and its external drivers

The first analytical phase employs land use change analysis to unravel the historical dynamics of agro-ecosystem management in the study area. To allow the analysis, aerial photographs covering the study area (Torrealvilla catchment) for five different moments in time (1956, 1986, 2004, and 2008) are being digitised using the ArcMap application in ArcGIS. So far, the digitisation and classification of the 2004 land use map for the study area has been completed. The photographs of 1956, 1986, and 2004 are available in orthorectified, digital format and are sourced from the Regional Ministry of Agriculture, Water and Environment of Murcia. The 2008 images are accessed online via http://www.murcianatural.com/natmur08/. The work is being carried out in a close collaboration with the Desertification and Geo-Ecology Department of the Estación Experimental de Zonas Áridas (EEZA-CSIC) (Partner 6) and Centro De Edafología Y Biología Aplicada Del Segura of Consejo Superior de Investigaciones Científicas (CEBAS-CSIC) in Murcia, Spain.

Prior to land use digitisation and classification, two site visits, one during winter and one during summer of 2008, were carried out. The purpose was to become familiar with the range of land uses in the study area and be able to recognise and differentiate one land use type from the others on the aerial images. This included the identifying locations of dominant land uses, different land surface formations (e.g. characteristics of gullies), patch/assemblage size, distinctive shapes and configuration arrangement and density of components that make up a particular land use, etc. For example, dryland (wine) grapes are normally sparsely planted and are found mostly in hilly areas while irrigated (table/eating) grapes are denser, covered with nets, and are grown in valley areas not far from settlements. Confidence in recognising the distinctive features of the different types of land uses within the study area is crucial and contributes to consistency in digitising and assigning land use classes.

Following this, another site visit was arranged to assess the accuracy of the digitisation and classification of land uses to date (based on 2004 imagery). This process, also known as “ground truthing”, was done by taking a sample list of patches with their respective geo-references (representative of all land use classes and spatial heterogeneity) to the field to assess whether the land use classes that have been assigned were correct. Ten land use classes were distinguished; namely cereals, almond/olive orchards, almond/olive orchards with cereals understorey, intensive irrigated agriculture (horticulture), grapes orchards, open pine forest, dense pine forest, shrubs (mattoral), rangeland and abandoned fields, pig farms (Figure 4.1). Following ground truthing, a field check confirmed 98% accuracy in digitisation based on 2004 land use mapping. Work is currently on-going to digitise and subsequently classify the land use maps for the other three moments in time.

This first phase also explores how various (external) socio-economic factors at the macro level (such as policy, market, population growth and compositional change, etc) have shaped agricultural land uses within the study area. Insights from this part of the analysis provide the wider socio-economic context of land use change that takes place in the study area. A triangulation approach is employed to identify the socio-economic insights which will further explain trends emerging from the spatial analysis. The triangulation method involves literature review, semi-structured interviews with household farmers, oral histories and key informant interviews. Multi-temporal land use maps produced at the beginning of this phase are used as visual aids for the interviews. This phase also investigates land managers/farmers’ attitudes and behaviours to: agro-ecosystem management; land degradation/ desertification; land and agricultural markets; population growth and compositional changes; regional development; investment opportunities (short term versus long term); new land use opportunities; risk and uncertainty; policies related to agriculture, environment, and water management.

About 30 household farmers/land managers have already been interviewed for this purpose; these farmers are a sub-sample of those already taking part in ABM Interviews (see section 4.1.3). Stratified sampling, based on the dominant land uses observed from the results of the 2004 land use map digitisation, has been adopted for these interviews. Consequently, an even proportion of interviewees are expected, equally representing each of the present different land uses.

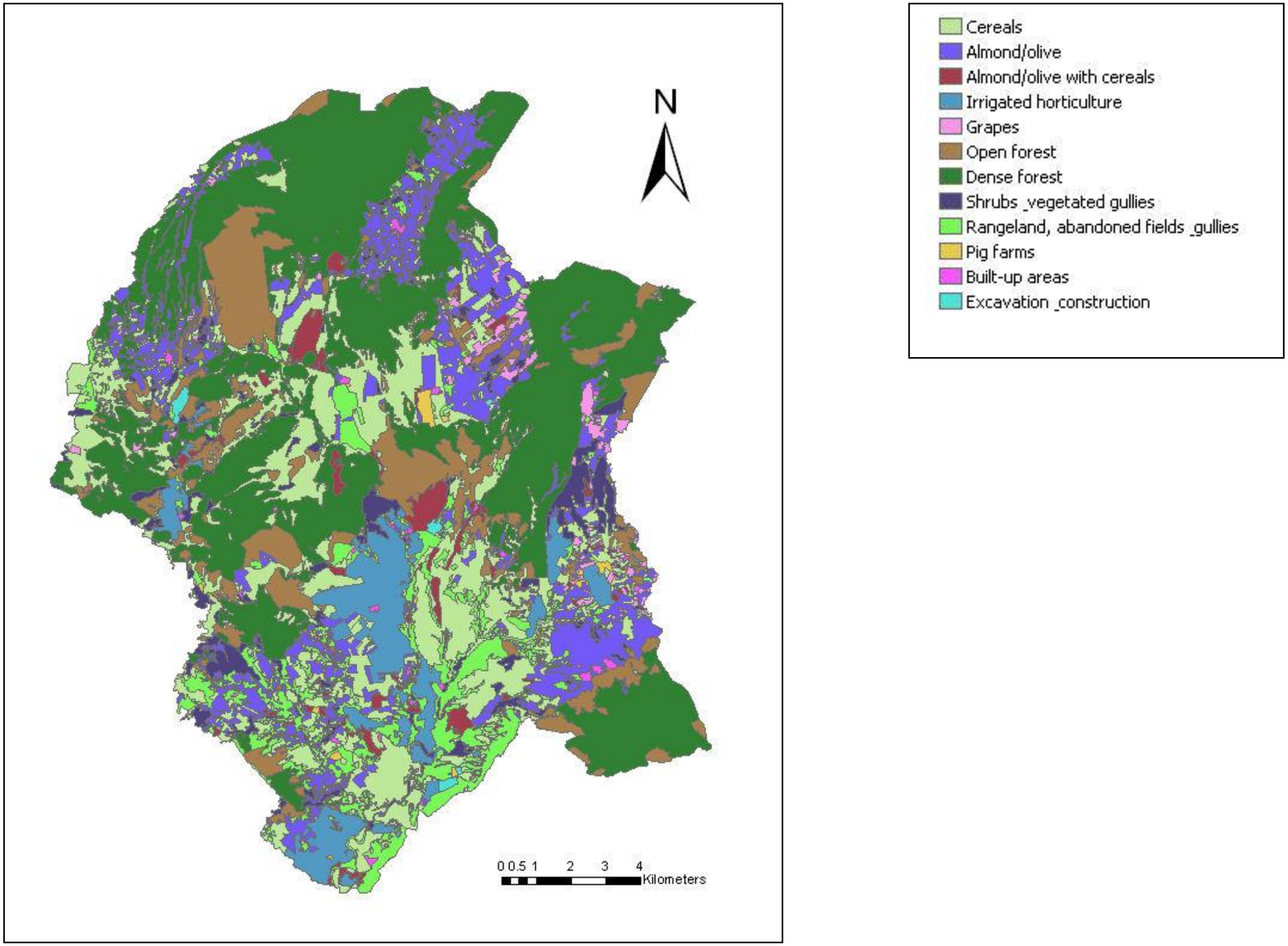

Figure 4.1: 2004 Land use map of Torrealvilla subcatchment

4.1.2 Phase 2: Linking land use change with spatial neighbourhood characteristics

This part of the modelling framework examines the extent to which the observed land uses have been shaped by the biophysical properties that characterise the agro-ecosystem being studied. The purpose in particular is to model the likelihood of transition from one type of land use to another given the existing biophysical features of the landscape. The multi-temporal land use maps produced from the first analytical phase serve as the main input for this analysis. From this process, the biophysical features that significantly explain the land uses in the study area (observed in Phase 1) can be identified. In other words, this phase explores variation across space that influences land uses. Below is a list of potential spatial explanatory variables:

1. Dominant land use types surrounding a given unit of analysis (determined using Euclidean distance and neighbourhood statistics in ArcGIS);
2. Slope (derived from Digital Elevation Model – DEM);
3. Soil characteristics (soil map);
4. Distance to main road (derived using road network map and distance measuring tool in ArcGIS; road network in the study area has been fully digitised) (Figure 4.2);
5. Distance to nearby settlement (derived using distance measuring tool in ArcGIS; settlements existing in the study area have also been digitised) (Figure 4.2); and
6. Distance to market (derived using distance measuring tool in ArcGIS) by using towns and cities within and nearby the study area, including the capital city of the Region of Murcia.

The last three variables in the list above can be considered as socio-spatial features as they are defined in relation to human infrastructure as opposed to the first three which are purely biophysical states. This phase of the modelling is expected to be straightforward once the digitisation of all the land use maps has been completed.

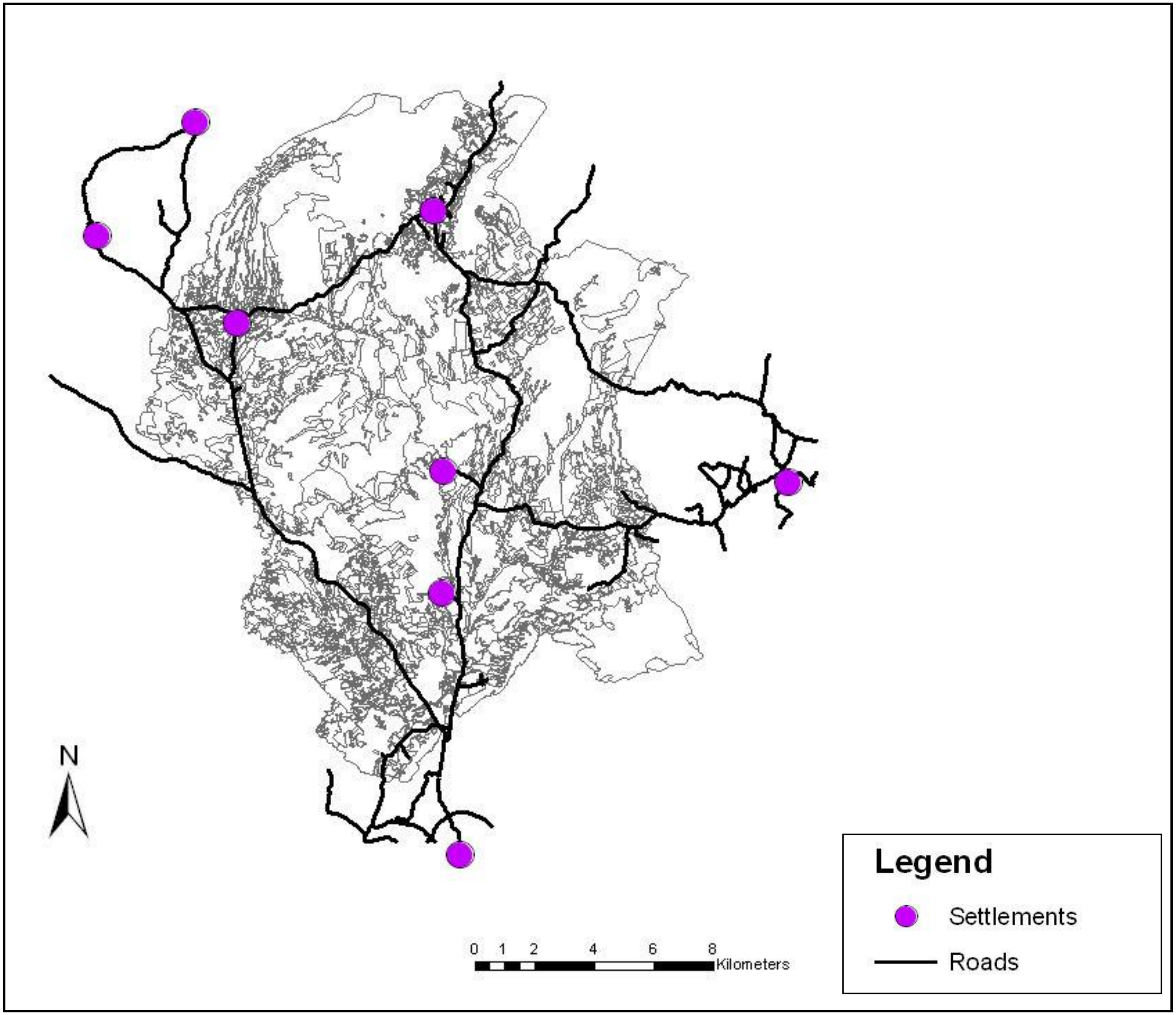

Figure 4.2: Roads and settlements within Torealvilla subcatchment

4.1.3 Phase 3: Linking land use and human agents

This part of the modelling framework links long-term trends in the management of agro-ecosystems, as indicated by the observed land uses, with those who are managing land within this system. It aims to capture interactions between human agents and their environment. As such, this part of the analysis will shed light on how the socio-economic characteristics of the land managers/users have constrained their land use decision making, which in turn has shaped the overall pattern of agro-ecosystem management over the years. This kind of understanding has the potential to form a basis for making forecasts about future trends in agro-ecosystem management in the area.

In principle, the approach used in this phase follows the preceding phase in that land use maps produced in phase 1 provide one of the main inputs for the analysis. However, in this phase it is not the spatial but the socio-economic characteristics of the agents that are treated as the explanatory variables. Two forms of 'relationship' will be investigated between the characteristics of the agents and their: 1) dominant land uses; and 2) land use diversity (captured by standard diversity indices such as Shannon's diversity index) across the land (s) managed by these agents. The dichotomy arose from preliminary site visit observations that a land manager/farmer could have more than one agricultural land use. To obtain the socioeconomic data of the human agents for this modelling, semi structured interviews are presently being carried out targeting up to 100 farmers, including those 30 farmers/land managers participating in the interviews in Phase 1. The observed spatial land use patterns, historical (result of phase 1) and current (with the addition of solar panel for example), forms part of the basis for selecting these interviewees. The primary target land managers/farmers of the study are those who have been managing land or farming in the area from at least 2004; although the available land use map stretches back to as early as 1956.

A semi-structured questionnaire has been devised and was trialled towards the end of 2008 in the study area. The resulting questionnaire is currently being administered by a professional interviewer. To date, 30 interviews have been completed with good data coverage. The interview captures the following information:

1. The characteristics of land tenure, employment status and off-farm income of the interviewees;
2. Data on land use decisions on individual parcels of land including reported past and recent changes as applicable;
3. Economic data relating to land management;
4. Data on actual and potential application of soil and/or water conservation measures on the lands that the interviewees own and/or manage. The purpose of this section is to find out an array of factors that may encourage or otherwise deter farmers from adopting particular soil and water conservation measures. As part of this exercise, the interviewees were also presented with a set of hypothetical financial incentives to see how future external assistance may increase the likelihood of certain measures to be adopted especially for those that were considered by the interviewees for being too costly to install and/or to maintain;
5. Water access for agricultural production;

6. Data on labour requirements; and
7. Demographic information about the interviewees and their household: age, gender, education, household composition, and the availability of a successor to take over the management of the land when the interviewees retire.

One section of the interview is designed to explore how land managers/owners/farmers would respond to future scenarios that are deemed relevant for future policy making. This is further elaborated in section 4.1.5. In brief, these scenarios are contained in the following three broad categories:

1. Change in EU subsidy scheme under the CAP;
2. Regulation of water access & water pricing; and
3. Environmental change.

These selected scenarios were informed by literature and concerns that farmers raised during the two stakeholder workshops organised as part of WB3 in Spain during February and June 2008. Based on this set of scenarios, future change in land use patterns within the study area can be empirically and realistically unravelled.

A behavioural choice model will be estimated using data from the questionnaires and used to characterise land use decision making. The model outcome shall highlight a range of the characteristics that have a significant effect on their respective agro-ecosystem management decision making. These significant characteristics form the building blocks of the algorithms for decision making rules for the Agent Based Modelling (ABM). Figure 4.3a provides an overview of land use decision making as influenced by an array of factors while Figure 4.3b depicts a simplified model framework for the ABM. Using the spatial information of the land being managed by each of the land managers/farmers, the ABM is made spatially explicit taking advantage of the GIS extension available in NetLogo[23].

[23] http://ccl.northwestern.edu/netlogo/. The NetLogo environment enables the construction of models of complex phenomena in the natural and social worlds. Users can give simple rules to individual "agents" in a simulation and observe the collective result of all the agents' behaviour

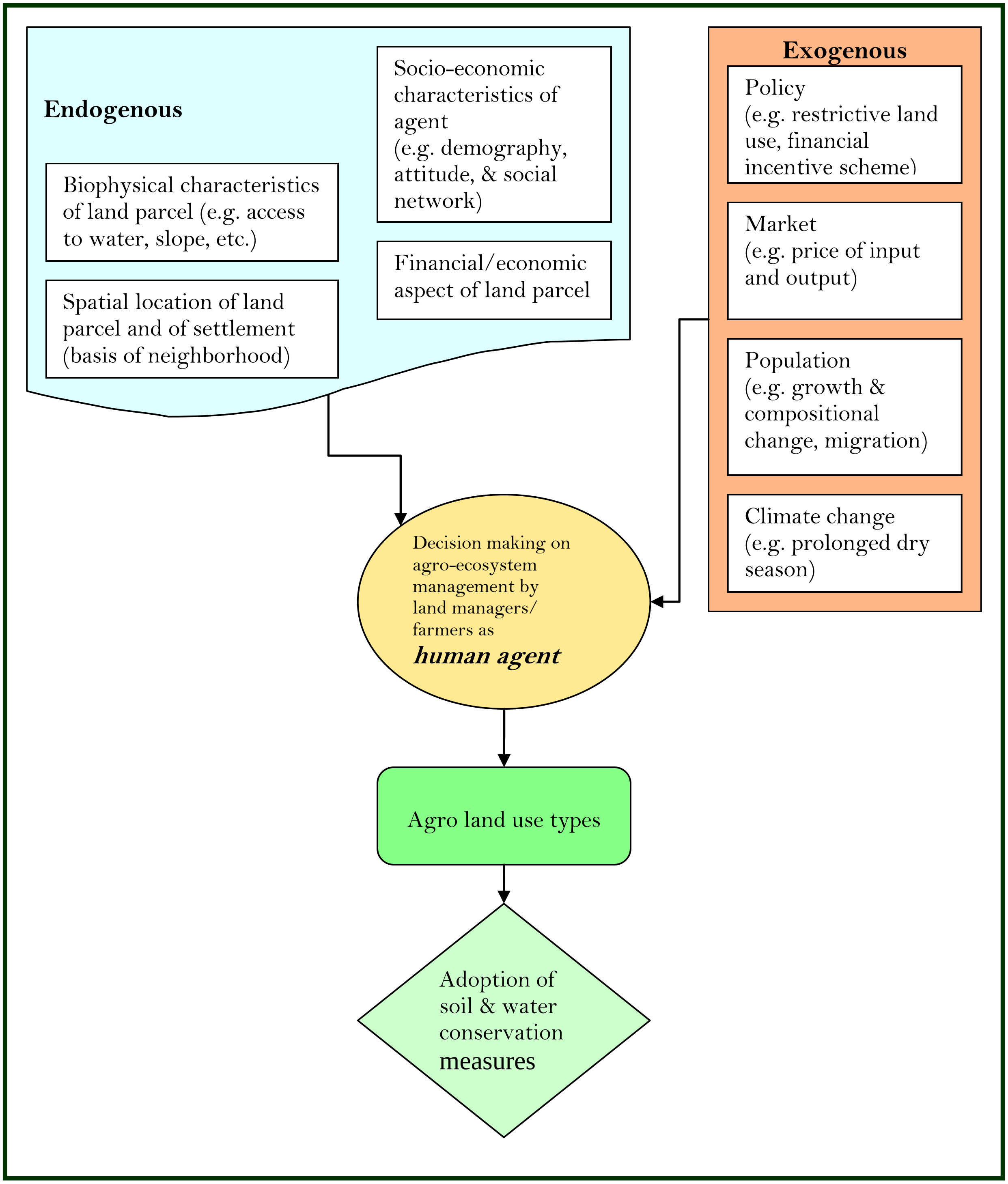

Figure 4.3a: The dynamics of land use decision making in dryland agro-ecosystem management in south-eastern Spain

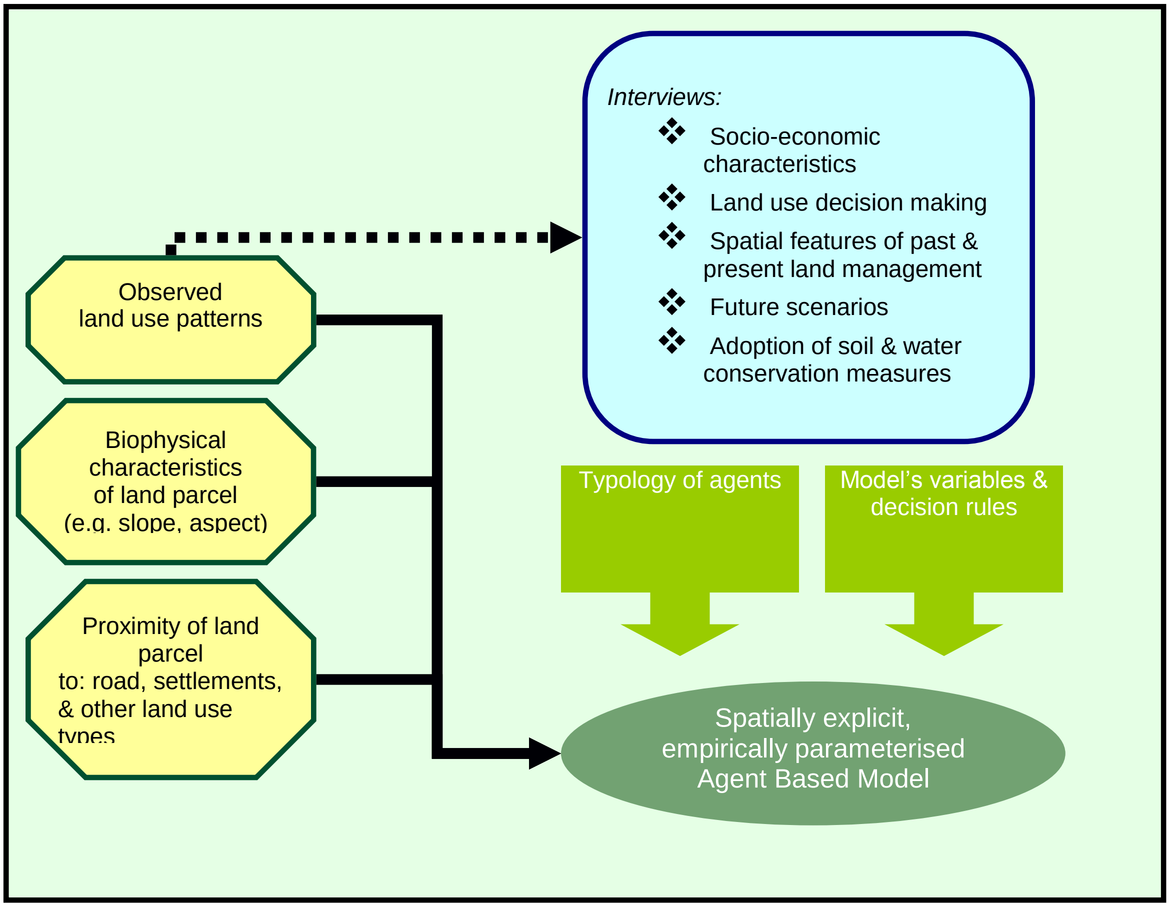

Figure 4.3b: Agent Based Modelling framework of dryland agro-ecosystem management in south-eastern Spain

4.1.4 Phase 4: Modelling adoption of soil and water conservation

This phase of the modelling framework is designed to evaluate how the socioeconomic characteristics of the land managers/farmers, their social interactions, and the economic and biophysical characteristics of the lands that they manage would condition their propensity to adopt particular soil and water conservation (SWC) measures. Both revealed (already adopt) and stated (future propensity to adopt) SWC uptake indication are explored. The output of this specific part of the study is intended to inform policy makers about more effective interventions to encourage the dissemination and uptake of SWC initiatives that could help address land degradation in the dryland agro-ecosystem.

The main inputs for the modelling during this phase come from the land use maps produced in phase 1 and from the land managers/users interviews. A specific section for exploring the nature of SWC adoption to meet this particular objective has been incorporated into the semi-structured questionnaire. This section is designed to capture: a) 'what would make one particular SWC measure attractive compared to others?'; b) 'what conditions may inhibit the farmers (interviews) from adopting certain SWC options'; and c) 'to what extent provision of external incentive can encourage SWC adoption?'. The land managers/farmers may choose from a selection of SWC measures that had been already identified through a participatory process, following the WOCAT approach, led by Partner 6 (Lopez *et al*., 2008a and b). The list includes:

1. *Acolchado paja* (organic mulching);
2. *Labranza reducida en contra de la pendiente en ambientes semi-aridos* (reduced contour tillage in semi-arid environments);
3. *Terrazas de tierra vegetadas* (vegetated earthen-terraces);
4. *Boqueras* (water harvesting from concentrated runoff for irrigation purpose);
5. *Agricultura ecologica de almendros y olivo*s (ecological production of almonds and olives); and
6. *Purines* (the application of manure from pig farm on olive/almond plantation).

Photos and a brief description of each of these SWC are being shown to the interviewees as part of this process. Nonetheless, as not all land managers/farmers were involved during the participatory selection process of the six SWC options, there is a possibility that the interviewees may come up with SWC techniques not represented by the list of options. A separate sub-section has therefore been included to record measures other than the six mentioned earlier known or adopted by the interviewees. In addition, as the list of measures covered by the WOCAT approach are mainly suitable to dryland farming, additional two measures appropriate for irrigated farming have been incorporated as well, namely: 1) measure to improve water use efficiency (e.g. conversion from traditional irrigation into sprinkling system), and 2) crop rotation. Adding these two extra measures is deemed necessary to accommodate those interviewees who run irrigated farming in the study area.

4.1.5 Phase 5: Modelling agents' future response

This phase of the analysis explores how land managers/farmers in the dryland agro-ecosystem being studied will respond to future changes and how their future responses would feed back into the system. The implications of: 1) a future climate change scenario (much drier and hotter climate with very little rainfall); 2) a water resource pricing/regulation scenario; and 3) a policy change scenario (reduction in subsidy and/or uncertainty regarding the long term security of agro-environmental schemes) are examined (see section 4.1.3 for details of how these were selected). Specifically, this part of the study investigates the extent to which particular land managers/farmers, being conditioned by their respective socioeconomic characteristics and the characteristics of the lands they manage, will: a) maintain status quo (doing what they are doing now); b) modify what they are doing now (i.e. reducing or otherwise increasing land areas allocated for particular land uses and/or substituting one land use type with another); c) seize newly introduced land use opportunities (e.g. solar panel field and ecological agriculture); d) do something completely new to the system; or e) abandon the land. A specific section to explore the land managers/farmers' future responses is included in the semi structured questionnaire described in section 4.1.3.

4.1.6 Phase 6: Evaluating the regional soil erosion impacts of agents' agro-ecosystem management decision making

In this phase, the consequences of agro-ecosystem management decisions on soil erosion are evaluated using PESERA (Kirkby *et al.*, 2008). Records of land use change identified through a set of multi-temporal maps produced in phase 1 are fed into the model to better understand how individual decision making at farm level over time across the landscape would impact the agro-ecosystem at a broader scale. In addition, the incorporation of future scenarios into the modelling in phase 6 is expected to produce a proximate of future land uses within the agro-ecosystem under study. Feeding this as input to the PESERA, future forecasting of soil erosion as a consequence of future land managers/users' responses in relation to the management of their lands will be made.

4.2 Input-Output Model

4.2.1 Introduction

Input-output analysis, initially developed by Wassily Leontief in the late 1930s and still widely used today, is a method to analyse interrelations between sectors of an economy. Inputs and outputs can be of any type, but the most common analyses look at monetary flows between economic sectors and final demand. The basis for input-output analysis is an input-output matrix (see example in Figure 4.4).

Input-output matrix (units: million €)

Products	Intermediate consumption									Final demand			
	Agriculture	Electricity supply	Gas supply	Water supply	Oil refining	Food processing	Trans port	Advertising	…	House holds	Government purchases	Investment	Export
Agriculture	2.297	1			32	11.362	1	1		6.856		464	7.609
Electricity supply	455	5.410	29	98	704	439	79	115		6.095		10	417
Gas supply	1	3.145	1	3	346	176	110	1		1.281		1	111
Water supply	291	71		3	64	53	279	3		2.312		7	
Oil refining	1.351	14	3	441	10.948	428	16	4		13.292		23	18.445
Food processing	5.959				99	10.244				20.260		208	7.567
Transport	197	178	5		1.263	1.961	469	33		7.279	754	171	6.453
Advertising	20	66	1	4	33	35	30	2.525		3.959			743
…													

Value added	Agriculture	Electricity supply	Gas supply	Water supply	Oil refining	Food processing	Trans port	Advertising
Labour	3.904	1.775	269	1.275	5.578	5.220	8.239	1.821
Tax	-1.251	438	41	-32	22	-31	18	31
Capital	18.780	9.098	2.115	730	4.726	2.942	9.430	4.633
Import	6.508	501			27.299	9.250	2.610	733

IO shows how:
- the sectors of an economy are interrelated (in €)
- **an** economic **activity *demands***, in its production process, ***inputs*** from **other** economic **activities**
- an **increase in final demand** of a good or service **produces an *indirect demand* of other goods and services** that serve as intermediate inputs to producing that specific good.

Figure 4.4: An example 8-sector excerpt of an input-output matrix showing inter-industry intermediate consumption and various categories of value added and final demand

To illustrate how input-output analysis works we can take a look at a hypothetical example from one of the economic sectors considered - food processing industries. This would include margarine production plants. To produce a margarine output worth €1.00, the plant would need to source €0.40 of oilseeds from agricultural suppliers (the agricultural sector). It would also need €0.05 worth of electricity, €0.03 worth of gas supply and €0.02 worth of water supply to process the oilseeds into margarine. The factory would have to purchase plastic containers worth €0.10 from the oil refining (incl. plastics) industrial sector. It would also hire the services of a transport sector firm to get oilseeds to the factory and margarine to the distribution channels (€0.05). Finally it would perhaps contract an advertisement firm to set up a publicity campaign to increase output (€0.05). Besides the above intermediate products, the plant would need to pay salaries to its employees (an equivalent €0.07 per €1.00 worth of margarine), pay various taxes

(€0.05) and source some inputs not available from the local economy (i.e. the region considered by the input-output analysis) from imports (e.g. palm oil worth €0.15). The shareholders and capital investors in the plant would finally be paid the remaining €0.03[24].

When one adds all interactions between the various sectors of an economy a matrix results with inter-industry intermediate product (value) fluxes, and value added and final demand categories (Figure 4.4). The example shows how the margarine processing plant would contribute to the column of food processing industries, sourcing various intermediate products and adding value through employment, tax, and interest. In turn, the margarine output of the plant would contribute to the output row of the entire sector, where it is considered as an intermediate product for the economic sectors and different categories of final demand (including household consumption, government purchases, export and investment). Once an entire economy is characterised as a matrix of input-output interrelations, one can use it to perform matrix calculations. If final demand for margarine increases, one can use the matrix to estimate how this will affect the economy. Unitary inputs per output (as in the above example) are called technical coefficients. An increase in margarine production will greatly affect the demand for oilseeds. Producing oilseeds in turn requires increased inputs of machines, fertilisers, etc. After solving the large set of linear equations resulting from a single change (e.g. increased demand for margarine), the impact of that change on the economy can be determined, i.e. the whole supply chain effects are considered potentially including environmental effects.

Originally, natural resources were not taken into consideration in input-output models, but various resources are increasingly accounted for. Guan and Hubacek (2008) review the application of input-output models to water issues, and present a body of research that has developed since the 1980s. Land as a production factor has also been incorporated in input-output models (e.g. Hubacek and Sun, 2001; Hubacek and Giljum, 2003), as have energy use, employment and various types of pollution. While the effect of land degradation on economy has been studied occasionally (e.g. Alfsen *et al.*, 1997 for Ghana; Bandara *et al.*, 2001 for Sri Lanka), inter-sector effects of soil conservation remain, to our knowledge, unstudied to date. The goal of the I/O model described below is to evaluate the wider effects on the regional economy of adopting mitigation strategies for land degradation.

To fill this lacuna, we will develop an input-output model for the Autonomous Region of Murcia, Spain. The model will be coupled to the agent-based model described in section 4.1 and to the modified PESERA model described in section 2 to allow scenario analyses of the regional economic impact of the adoption patterns of remediation technologies, in turn influenced by policy and climate change scenarios.

4.2.2 Model construction

[24] As will become clear when reading on, input-output analysis can also be used to say e.g. how much water (in physical units) would be required to construct one unit of output.

The starting point of the input-output model is the construction of an inter-industry I/O matrix (see Figure 4.4 above). No such table is available for the region, so the first step will be to construct this from a recently published national I/O table for Spain for 2005, which distinguishes 73 sectors (Instituto Nacional de Estadística, April 2009[25]). Regional economic data from the Centro Regional de Estadística de Murcia[26] will be used to adapt the national table to the regional economy. The first step herein is to multiply the national data with regional sector employment coefficients, based on the assumption that the labour productivity in Murcia is comparable to that of Spain as a whole (following Miller and Blair, 1985). As regional employment statistics are available for 26 sectors, the original 73 x 73 I/O table will first be aggregated to a 26 x 26 I/O table.

When the size of the 26 regional industries has been established, the output from each sector can be distributed across industries as intermediate consumption using the technical coefficients from the national table, and to final consumption and export. In the case that the regional production does not suffice to fulfil intermediate consumption by other regional industries, we will assume that the deficit will be imported. In order to validate these calculations, we will use the regional statistics on import and export. In doing so, we take stock of Boomsma and Oosterhaven's (1992) warning that failure to take into account regional trade data tends to overestimate a regions inter-industry relations (and any subsequent multiplier effects when performing scenario analyses based on erroneously constructed tables)[27].

A next step in model construction is to split the agricultural sector, currently in its entirety considered as one of the 26 sectors in regional statistics, to account for the major land use categories (tree crops, annual crops and (irrigated) horticulture and livestock production). Regional agricultural economic statistics will be used to inform this step.

Once the inter-industry I/O table is complete, the model can be extended with environmental matrices. An important source of information is the Integrated Environmental and Economic Accounting System (IEEA) maintained by INE. Its water use budgets have been used successfully in I/O studies at the national level (Duarte *et al.*, 2002) and regional level (for Andalucía, Dietzenbacher and Velázquez, 2007). Water use is important to consider in our model both because water resources depletion is a serious environmental issue in Murcia, and because the agricultural sector is by far the largest regarding water consumption[28].

In considering the effect of soil erosion (mitigation) on agricultural land on the wider economy, it should be noted that soil is usually not a natural resource of economic importance[29]. Rather, the effect of soil erosion (mitigation) should be accounted for by

[25] Available at http://www.ine.es/daco/daco42/cne00/cneio2000.htm

[26] Available from http://www.carm.es/econet/home.html

[27] However, due to resource constraints we will not be able to use a detailed trade survey as suggested by Boomsma and Oosterhaven (1992), but will instead rely on reasonably detailed regional trade statistics

[28] Apart from water quantity, water quality is an important issue, especially for domestic consumption, tourism and some industries. Polluting activities can affect water resources quality. Agriculture is both important in the absolute volume of water it consumes as well as a sector affecting the quality of remaining water sources (e.g. by runoff of fertilizers and pesticides).

[29] Very severe soil erosion could eventually result in badlands without any productive value, but this is exceptional and would likely especially affect land that is already of marginal economic value. Note that when soil is regarded as a substrate (i.e. having a carrier function) for economic activities, we define it as land. Land may have considerable economic value, especially in urban environments and very intensive agricultural areas. However, we refer here to 'soil' (…erosion/conservation), not to 'land'.

incorporating matrices for carbon and nitrogen cycles, being the most important elements contained in soil for inclusion in economic accounting. Regarding the carbon cycle, soil conservation helps build up organic matter in the soil and e.g. replacement of fertilizers by organic mulches reduces fossil carbon dependency. Regarding the nitrogen cycle; the better organic matter content due to soil conservation increases yields without the need of chemical fertilizers, while ongoing land degradation on the other hand increases dependency on chemical fertilizers. Application of locally produced organic matter on agricultural fields adds value for these residual products otherwise lost (even leading to eutrophication of surface water resources). The (important) effect of soil erosion (and technologies to remediate it) on green water – water stored in the soil and used by plants – will be indirectly considered through changed water budgets and crop productivity in scenarios with as opposed to without mitigation strategies (Section 4.2.3).

The I/O model will finally also include accounting rows for agri-environmental subsidies and capital input.

4.2.3 Scenario analyses

The goal of the I/O model is to evaluate the wider regional economic effects of adopting mitigation strategies for land degradation. Mitigation strategies include the application of various structural, vegetative, management and agronomic soil and water conservation measures on rainfed land cropped to annual crops, ecological production of tree crops, and application of organic mulches on tree crops and possibly irrigated crops (annual crops, tree crops and irrigated crops being distinguished as separate economic sectors).

Based on the Cost-effectiveness modelling (section 3), we can alter input and output structures of the relevant sectors in the I/O model for a future land use situation (i.e. the after adoption situation). By exploring differences between different states (policies and technologies expressed in changes in the IO matrices), we can then derive implications for the wider regional economy. Other scenarios can be based on extrapolation of land use changes assessed by the ABM model (section 4.1) or on results from WB 1, 3 etc.

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