ENVIEVAL

Development and application of new methodological frameworks for the evaluation of environmental impacts of rural development programmes in the EU

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Review of micro level methods and scales

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List of Acronyms

ABMs	Agent Based Model
AES	Agri-Environment Scheme
AFI	Agri-Environmental Footprint Index
CF	Carbon Footprint
CMEF	Common Monitoring and Evaluation Framework
CORINE	Coordinate Information of the Environment
EF	Ecological Footprint
FADN	Farm Accountancy Data Network
FBI	Farmland Bird Index
GIS	Geographic Information System
GLEAMS	Groundwater Loading Effects of Agricultural Management Systems
GOF	Goal Oriented Framework
HNV	High Nature value
IACS	Integrated Administration and Control System
ICSU	International Council for Science
IRENA	Indicator Reporting on the Integration of Environmental Concerns into Agricultural Policy
IPCC	Intergovernmental Panel for Climate Change
LCA	Life Cycle Assessment
MODAM	Multi-Objective Decision support tool for Agri-ecosystem Management
MS	Member State
RDP	Rural Development Programme
RUSLE	Revised Universal Soil Loss Equation
SEAMLESS	System for Environmental and Agricultural Modelling; Linking European Science and Society

USDA	United States Department of Agriculture
WF	Water Footprint

Executive Summary

The environmental impact assessment of rural development policies is an issue to be constantly monitored by both practitioners and policy makers within EU and its Member States. The empirical evaluation of the policy effects reveals, in fact, strengths and weaknesses of the applied strategies, enables better design, and outlines more responsive and effective policies for agri-environmental practices. Thus, impact assessment methodologies are well established in the literature on agri-environmental policies, both at micro and macro scale, considering different public goods (water, biodiversity, etc.) as specific *foci* of these policies. In these terms, a vast range of micro-scale methodologies is available in literature about impact assessment and evaluation. However, there are several challenges and gaps related to their use, concerning both the fitness of indicators, models and methodologies for the expected outcomes, and the adoption of the most suitable scale for the analysis.

Literature is debating on the effectiveness of indicators, models and methodologies in a comparative perspective, also attempting to discuss the need to find common methodologies able to standardise (for example, at EU level) the procedural sets and tools. The main questions are the multi-scale integration and combination of results and the possibility to efficiently generalise micro-scale results in a macro-scale perspective. Finally, for both challenges, fit-for-purpose data, datasets and data sources are required toward more appropriate and holistic analysis and evaluation. These challenges certainly represent the main open questions for ENVIEVAL project and for its advancement in the next two years.

Based on these statements, the objective of the paper is to review the methodologies for the assessment of environmental impacts of policy measures, with specific reference to Rural Development Programmes (RDPs). This assessment needs to be carried out at an appropriate scale able to represent the natural processes, both to support the analysis of multiple benefits and to include the potential for cumulative environmental impacts. Specifically, the review focuses on the methodologies dealing with environmental impacts at micro level that should be able to explicitly link environmental impacts and policy measures through the beneficiaries of the policy. In this vein, the theoretical baseline is that a comprehensive assessment of the extent to which the multiple environmental goals have been achieved requires the application of more than a single methodology.

The present review aims: (i) to explore the vast range of methodologies which may address some of the aforementioned challenges, and (ii) to contribute to develop a flexible and integrated methodological framework for the assessment of environmental impacts in RDPs. A general overview of environmental impact methodologies is presented, coupled with the challenges derived from the use of different analytical scales and levels. Main results are further summarised on the basis of a review of scientific papers with a micro-level perspective. Based on these results, combined with the early findings of the ENVIEVAL project related to the analysis of European evaluation reports (WP2) and to the interviews with stakeholders and evaluators (WP9), the paper explores the current application of micro-level methods in RDP evaluation. Then, it provides an analysis of scientific literature on the evaluation of environmental impacts of agri-environment measures, schemes and programmes across Europe and some international countries, describing the most relevant applied methodologies and stressing their strengths and weaknesses, the latter particularly representing a challenging issue for ENVIEVAL to deal with. Finally, the conclusion proposes the main recommendations for the future activities to advance the ENVIEVAL project.

According to the research questions we can draft four preliminary conclusions. *First*: the lack of appropriate and specific data can undermine the results of the evaluation exercises. Furthermore, taking into account the current and past experience of RDP evaluations, the difficulties encountered by evaluators to use complex methodologies could weaken a good outcome from the evaluation process. *Second:* the most commonly adopted approaches are based on sampling methods and integrated models. Both have some advantages and disadvantages in terms of generalisation of the main related micro-scale findings to a different scale perspective. *Third:* the adoption of the aforementioned models will need specific datasets, for a vast range of socio-economic, environmental and institutional variables and long-term covering for comparative analysis. *Fourth:* one of the future challenges for the advancement of ENVIEVAL is represented by clearly understanding the relationship between micro and macro approaches within complex systems such as agro-ecosystems.

1 Introduction

The objective of this report is to review the methodologies for the assessment of environmental impacts of policy measures, with specific reference to Rural Development Programmes (RDPs). This assessment needs to be carried out at an appropriate scale able to describe and cover the broad range of involved natural processes, to support the analysis of multiple benefits/criticisms and to include the potential for cumulative environmental impacts. The specific focus of the review is on methodologies that deal with environmental impacts at micro level, that should be able to explicitly link environmental impacts and policy measures through the beneficiaries of the policy.

A comprehensive assessment of the extent to which the multiple environmental goals have been achieved requires more than one methodology. The review explores the wide range of methodologies which may address some of the challenges and contribute to the development of a flexible and integrated methodological framework for the assessment of environmental impacts in RDPs. The report is divided in four sections. The first section deals with the general overview of environmental impact methodologies and the challenges arising from the use of different scales and level in the analysis. Furthermore, it tries to summarise the results of a literature review based on a wide collection of scientific papers from a micro-level point-of-view. The second section sheds light on the current application of micro-level methods in RDP evaluation, through the analysis of European evaluation reports, as collected through the inventory of indicators compiled in WP2, and the interviews to national evaluators made in WP9. The third section provides for the analysis of scientific literature on the evaluation of environmental impacts of agri-environment measures, schemes and programmes across the Europe and in some international experiences, providing evidence of the most relevant applied methodologies, particularly shedding light on their strengths and weaknesses, knowledge of which constitutes a crucial benchmark for the advancement of ENVIEVAL. Finally, the fourth section summarises the conclusions both on the topic of environmental evaluation generally, and specifically for the advancement of the ENVIEVAL project, recommending main tasks to pursue in its future activities.

1.1 Overview

Over the years there has been a significant development of agricultural and environmental sciences, and the general public is increasingly concerned about the relationship between agriculture and the environment. In this way, methods and models have been explored to evaluate this complex relationship at different scales and levels. Many studies include in-depth analysis of agricultural systems based on simulation models for various purposes: understanding the involved mechanisms, comparing alternative scenarios and supporting and evaluating policy measures. Indicators are also frequently used to provide concise quantitative figures on aspects such as economic or productive performances and environmental pressure, and to evaluate these aspects in terms of impacts.

Many studies have proposed lists of methodologies, models and indicators for evaluation, most of them emphasising the need to provide a conceptual analytical framework. For example, a document by the International Council for Science (ICSU, 2002) attempted to draft a general scheme where methodologies, models and indicators are framed and integrated in four types of knowledge of a generic evaluation process of human-environmental relationships. *Indicators* represent the first functional component, able to monitor social and environmental developments at various temporal and spatial scales. As previously stated, they should be preferably organised within *conceptual frameworks* that represent the second type of knowledge, i.e. ordering mechanisms that help to logically organise the indicators. A third type of useable knowledge derives from specific forms of *analysis* (e.g. models), to gain information and insight for a specific *assessment* purpose (the fourth type of knowledge). Indicators play a fundamental role as a communication interface between science and policy decision making, whilst models provide methods and tools to support the analysis of specific systems (in this case the agro-ecosystems), or more generally the territorial systems in which agri-environmental issues are considered. Indicators allow better communication and accessibility to information, bridging the gap between producers and information users, i.e. between the information available through scientific resources and the need for information for decision making. A system can be defined as the limited part of reality that contains interrelated elements of specific interest. A model is a simplified representation of a system, whilst simulation is the art of building mathematical models and studying their properties, in relation to those of the systems. Although a model always simplifies reality, it should contain all the essential features of the real system, in order to describe and solve problems. The balance among simplification, comprehensiveness and effectiveness depends upon the scope of the model, which may be very diverse, thus determining a wide range of possible model typologies.

Key methodological challenges for the environmental impact evaluation in relation to public goods are: to provide evidence of true causality; to disentangle the effects of single measures and the programme from other factors; to quantify net-impacts at micro level; to make available a viable body of evidence; to fill the gap between indicator measurement and policy decision making (Lukesch & Schuh, 2010). The Common Monitoring and Evaluation Framework (CMEF) uses a systematic approach based on intervention logic, which links the hierarchy of policy objectives to that of indicators, aiming to measure the extent to which the objectives have been reached.

A better understanding of the relationships between scale and evaluation methods in the agro-ecosystems is crucial to identify the effective policy measures, in a way to improve the impacts of socio-economic drivers from the RDPs. While natural sciences have long understood the importance of scale issues, social sciences has been less explicit, less precise, and more variable. The need for interdisciplinary work to assess the methodological evaluation is essential; however it requires some common understandings and clarifications about scaling and up-scaling issues, as presented in the next sub-section 1.2.

1.2 The Concept of 'Scale' and 'Level

Literature presents different perspectives on distinguishing between scale and level. In the European Evaluation Network for Rural Development working paper on the approaches for assessing the impacts of the Rural Development Programmes (Lukesch & Schuh, 2010), the references to micro scale and level are often related to different meanings of the same concept. In some contexts, 'micro' is used as synonymous with 'local'. Further distinction is provided to show how micro level is linked in RDP to the intervention logic, which drafts a hypothetical trajectory from beneficiary over measure to objective and programme. In other words, the chain of potential effects links the individual measures at micro level. Often the micro level is represented by the farm, which is considered as the simplest management unit of the agricultural system. The micro/macro definition is also used in the field of environmental impacts, where a modelling approach (in terms used by economists) has a quite limited explicatory power. In the case of RD measures, the observation of changes related to the territorial focus prevails, and in this way a common practice is to assess the impact on the micro level and then scale up to the macro level, for example using GIS-based tools.

In the scientific literature, several definitions have been proposed for the terms 'scale', 'level' and 'unit', sometimes creating confusion due to the different uses of these words for the same purpose. Most of the misunderstandings occur when the analyses move between the ecological and human dimension. Discourse on scale has been extended from those in natural science context (Schneider, 2001; Higgins, 2012) to social science communities (Adger et al., 2006; Keshkamat et al., 2012). This has been driven by the growing demand for integration in systems and complex sciences (Gibson et al., 2000; Vermaat et al., 2005; Cash et al., 2006; Veldkamp et al., 2011; Vervoort et al., 2011). The concepts of scale and reporting by areal units are commonly used in natural and social sciences, but they have emerged from very different theoretical bases. As a result, the collection, analysis and reporting of data, and the analysis undertaken, can be significantly different (Gibson et al., 2000).

The lack of clear definitions frequently leads to the use of 'scale' and 'level' interchangeably. According to Gibson et al. (2000), the term 'scale' refers to "*spatial, temporal, quantitative, or analytical dimensions used to measure and study any phenomenon*", and 'level' refers to "*locations along a scale as the units of analysis that are located at different positions*". Vervoort et al. (2011) introduce the concept of 'dimension' (Figure 1) as the basic structure of analysis, because it allows for recognition of the multiplicity of possible scales. They identify levels (e.g. micro, meso, macro) as positions on a scale.

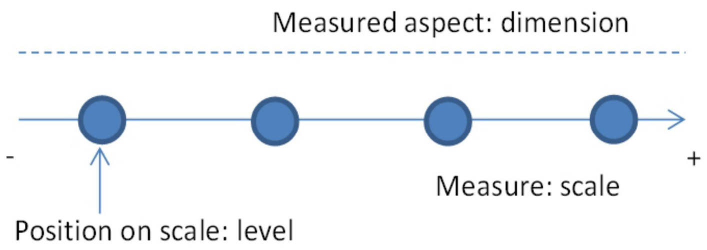

Figure 1 Schematic illustrations of scales, levels and dimensions

Source: Vervoort et al. (2012).

Cash et al. (2006) discriminate different types of scales and levels that can be referred at the geographical space or the spatial context, temporal scale, jurisdictional scales, institutional scale, management scales, network scales and knowledge scales (Figure 2). Each of these scales represents specific aspects, such as environmental, ecological and bio-geophysical phenomena that can be represented in terms of space and time, while social phenomena are better represented through jurisdictional and institutional scales. However, this distinction is rarely clear and in the case of RDP evaluations the analysis can be described both in terms of spatial scale and of jurisdictional and institutional ones. In this sense the jurisdictional scale can also be understood as a subset of spatial scale. Both terms refer to common geographical entities but the interchange of different levels (for example, landscapes instead of provincial) may lead to misunderstandings and to inappropriate solutions proposed for ecological problems due to mismatches among scales in socio-ecological systems (Cumming et al., 2006).

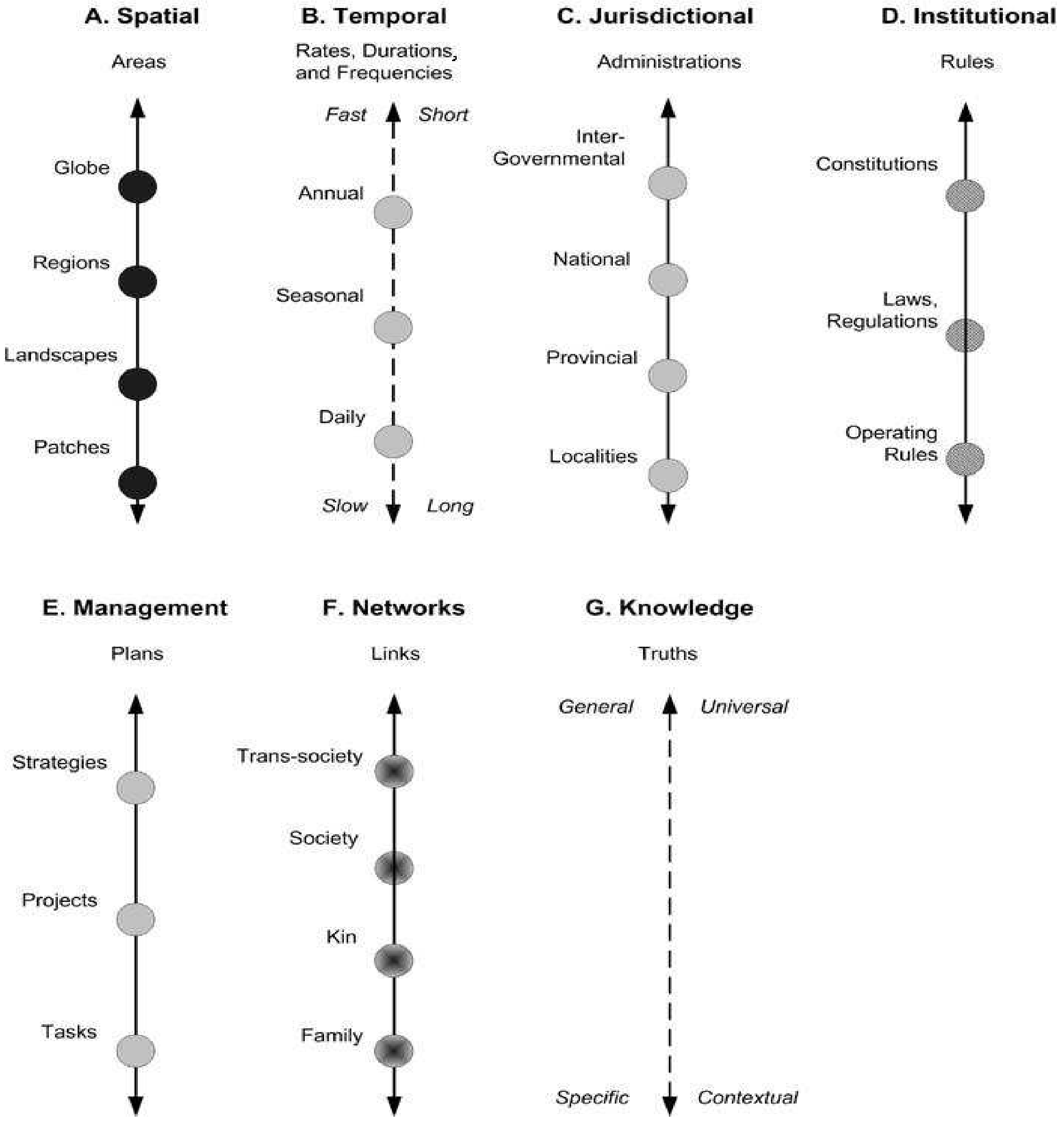

Figure 2 Illustrations of different scales and levels that are critical in understanding and responding to human-environment interactions

Source: Cash et al. (2006).

There are different interactions that may occur within or across scales, leading to substantial complexity in dynamics (Cash et al., 2006; Veldkamp et al., 2011). Cross-level interactions refer to interactions among levels within a scale, whereas cross-scale means interactions across different scales, for example, between spatial domains and jurisdictions. The term ‘multi-level’ indicates the presence of more than one level, while ‘multi-scale’ denotes the presence of more than one scale, but without implying the presence of significant cross-level or cross-scale interactions. Cross-scale and cross-level interactions may change in strength and direction over time in terms of dynamics of their linkages. Changes may arise from the consequences of those interactions or be caused by other variables (Cash et al., 2006).

A better understanding of the relationships between social and ecological systems is needed to improve the impacts of socio-economic drivers on ecosystems and biodiversity with the aim of identifying effective policy measures (Henle et al., 2010; Paloniemi et al., 2012). The lack of interactions among social and ecological systems may lead to scale mismatches, for which the scale of environmental variation and that of social management are not aligned, with consequences in terms of inefficiencies and loss of crucial components (Cummings et al., 2006). A graphical example of mismatched spatial scale is presented in Figure 3, where examples regard the carbon emission regulation, better managed at global level, and fishery harvesting regulation, that conversely needs to be determined at a relatively fine scale. The analysis and discussion with respect to ‘scale’, ‘level’ and ‘reporting unit’ also required consideration of temporal interactions of policy and its impacts on environmental goods. Choices of these elements are still unclear and under development.

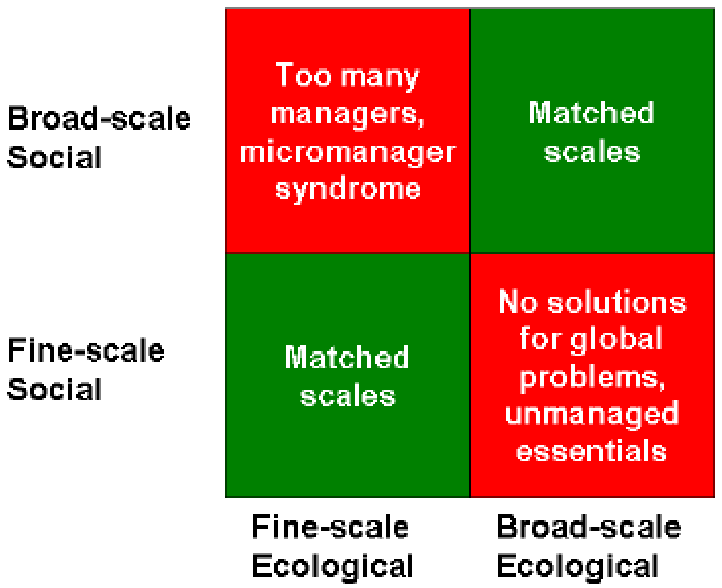

Figure 3 Examples of mismatches in spatial scales

Source: Cumming et al. (2006).

Providing empirical evidence of a real cause-effect link between the observed indicators and the rural development programme is widely considered as the core of the evaluation design. Among the various challenges of the evaluation process, beyond the counterfactual performance (which is the objective of the specific Deliverable 3.1), it has to be mentioned the analysis of the potential direct and indirect effects, both positive and negative. This analysis should consider changes in all relevant impact indicators related to the programme, implying to refer to an appropriate baseline. Moreover it is also crucial to disentangle the effects of a programme support from effects of other exogenous intervening factors that may have also influenced a given impact indicator calculated at the regional/macro level. These influences may stem from other programmes, e.g. Structural Funds, or from programme-independent causes. Measuring the net effects means to subtract the changes occurred without the public intervention from the gross effects, also considering deadweight, leverage, displacement, substitution and multiplier effects. Finally, while some impacts can be observed among direct beneficiaries (e.g. turnover generated for the suppliers of assisted firms), others can only be observed at macroeconomic or macrosocial level (e.g. improving the image of the assisted region). Often, identifying impacts at micro level seems to be easier than identifying overall impacts (DG Agriculture, 2004).

With regard to scale, it is not always possible to clearly distinguish between micro and macro level in the impact assessment of the RDPs. Perhaps, a more intuitive type of distinction is to consider the individual beneficiary of RDP from the sectoral and territorial level as a micro-unit of reference, on which to apply the concept of ‘micro level’. From this starting point, it is possible to effectively identify a correlation between the two levels. As suggested by Lukesch and Schuh (2010) this correlation can be studied along a bi-dimensional line that links the micro-level (impacts on the measure level) with the macro-level (i.e. the program).

1.3 Outcomes of Previous Literature Surveys

A large body of academic surveys attempts to systematise the state-of-art about the environmental effectiveness and impacts of RDPs, agri-environment schemes and measures, and about methods, models and indicators used for the analysis (van der Werf & Petit, 2002). Payraudeau and van der Werf (2005) analyse six main types of methods at different spatial scales (environmental risk mapping, life cycle analysis, environmental impact assessment, multi-agent systems, linear programming and agri-environmental indicators) to assess the environmental impacts of farming activities accounting for 11 case studies, in which one of the six methods was applied. The authors conclude that preferable methods are those which allow the expression of impacts according to several reference units, as they allow the evaluation of different functions of agriculture at farming-region scale (e.g. the production of commodities versus non-market functions). Methods of extrapolation or scaling procedures have to be designed and improved to apply micro-scale indicators to the regional level in terms of classification of farms and of environmental vulnerability. Finally, a system approach to the environmental evaluation of a farming region should integrate into the assessment both inputs and outputs at the regional level as well as the possible effects of interactions among farms.

Regarding the agri-environment schemes (AES), Primdahl et al. (2010) analyse and discuss the actual and potential use of impact models at different stages of the schemes (design, implementation and evaluation). The concept and the role of impact models are discussed surveying 60 European AES studies. Although the authors demonstrate the usefulness of the impact models in agri-environmental policy, both in improving the policy design process and in evaluating policy, they argue more than half of the analysed management packages are based on ‘common sense’ impact models, and thus on general beliefs about the link between agricultural practices and environmental changes, rather than on documented evidence. Impact models offer advanced predictability, more often in underpinned management packages for natural resources than those dealing with biodiversity and landscape. The following criticisms about the effectiveness and evaluation methods for AES have been demonstrated: a need for improved clarity in objectives; great variations in reporting practices concerning the relevance and the use of indicators; lack of appropriate targeting approaches; insufficient or absent baseline data; different approaches to reporting and poor evaluation frameworks (including lack of appropriate impact models). The review suggests increasing the use of evidence-based impact models to improve the effectiveness of AES.

A very recent survey by Uthes and Matzdorf (2013) investigates the AES literature and provides an overview of the most relevant research topics and results, equally considering ecological and economic perspectives. The authors identify three large groups of articles in terms of environmental impact: the first empirically highlights the ecological effects of AES on the basis of field experiments, monitoring data (quasi-experimental) or farm surveys, usually in combination with some statistical analyses; the second identifies the factors influencing farmers’ decisions in participating in AES; the third presents schemes and programmes and their evaluation, in different countries and regions discussing strengths and weaknesses. About 50% of the analysed papers are focused on biodiversity. The most empirical studies concentrate on the assessment of single measures, with few cross-regional comparison and the results are often not comparable or transferable. Ecological effects of AES are heterogeneous and depend on scheme factors and investigated indicators. However, contrasting experiences often result from different study designs and goals. AES seems to have a low effectiveness; however it is difficult to judge whether these analytical results are true or derive from methodological problems. In fact, the evidence of positive impacts is obscured by the fact that the spatial reference, the measures themselves and even the indicators for judging their performances have often changed along the years. Consistent data sets covering more than one or two contract periods (a contract period is usually 5 years) are usually not available. Moreover, the adoption of different measures, combined with the fragmentation of data and data sources for the evaluation, makes the empirical evidence of the environmental impacts inadequate (Uthes & Matzdorf, 2013).

In relation to single public goods, literature surveys have been presented in terms of the effectiveness of AES in promoting biodiversity (Kleijn & Sutherland, 2003; see also Kleijn et al., 2001; 2006). According to the seminal work of Kleijn and Sutherland (2003), the approaches used to evaluate biodiversity effects of the schemes varied enormously. The most common approach compares biodiversity in the AES and control areas at the same time. Assessment based on changes in biodiversity and trends in time are relatively less frequent. The analysed studies demonstrated patchy results of effectiveness with no effects, negative effects, or positive effects on some species and negative effects on others, measured in terms of increased species diversity or abundance. The authors suggest modifications in the prescribed management as outcomes of a regular evaluation of all AES.

Furthermore, the survey by Povellato et al. (2007) deals with climate change in terms of cost effectiveness of greenhouse gas mitigation measures. The authors argue that the complexity and the uncertainty of the phenomena characterising the agricultural and forestry ecosystems make extremely challenging the analysis of causal links between policy measures and the expected positive impacts on climate change. Studies sharing the same methodology and trying to answer the same question usually end up with quite different outcomes, depending on the hypotheses that drive the underlying modelling exercise, the geographical and the sectoral focus. This indicates a substantial dichotomy between the diversity of results of the recent scientific literature and the standardised approaches suggested in the guidance documents at international level (see for instance the IPCC Guidance). Similar dichotomy could arise between the CMEF and the local needs for specific evaluation methodologies.

1.4 Research Questions

The aforementioned literature highlights two main challenges with the evaluation process. First, specifically focusing on micro-level effects, it seems to be still difficult to reach in literature a clear and holistic explanation of cause-effect relationships of policy measures related to the single environmental public good and to specific aspects of each one of them. Second, regarding the adaptability of micro/macro level to this kind of analysis and to a mutual integration of micro and macro levels, efforts are required to explore the micro-level environmental effects in a macro perspective, and to exploit results obtained at the farm level to describe more general performances in AES. For example, discussing the criticisms of European landscape policy, Lefebvre et al. (2013) argue that an optimal management of agricultural landscapes, beyond actions at the farm level, requires the integration of fields and farms in the agricultural landscape at landscape level, and the conservation of the diversity of agricultural landscapes in the European Union as a global public good. Therefore, scientific literature presents multiple and well-tested methodologies for environmental evaluation; however they still require to be systematised, in order to better fit with single measures and overall programmes, and to suitably adapt to the main open questions about evaluation of a single public good.

Starting from these overall considerations, the present review deals with these challenges and tries to answer four main questions. 1. How are the past and current evaluations of RDPs affected by these existing challenges? In the last decade scientific analyses have increased enormously, also thanks to specific requests and financing from public authorities. 2. Which are the most commonly used methodologies to assess micro-level environment evaluations in scientific literature? The question arises from issues related to the specific use for some methodologies (or models, or indicators) for a single public good, and to the suitability of some of them for the evaluation of environmental impacts on a single public good in comparison to others. 3. What are the main gaps and challenges evidenced in literature? Primdahl et al. (2010), among the others, report the following gaps in evaluation methods: need for improved clarity in objectives; great variations in practices about the relevance and the use of indicators; lack of appropriate targeting approaches; insufficient or absent baseline data; different approaches to reporting and deficient evaluation frameworks (including lack of appropriate impact models). 4. What should be the new developments to cope with present gaps and challenges? In this term, future trends of evaluation methods should address new developments able both to integrate diversified approaches (for example qualitative and quantitative), and to consider different perspectives of environmental issues. Finally, aiming to contribute to systematise the current knowledge on evaluation methods, this report sheds new light on the role of methodologies and of their integration in the evaluation process, particularly to fill the gap of knowledge of the agriculture-environment relationships within the complexity of multi-scale and multi-levels approaches.

2 Current Application of Micro-Level Methods in RDP Evaluation

The challenges posed by the environmental impact assessment of past and current RD policies have been accounted in the evaluation process implemented by Member States under the framework proposed by the EU Commission. At the end of the 2000-2006 RD programming period, the first systematic attempt took place to evaluate the environmental impact on an ex post basis. The comparative analysis of the MS evaluation reports (Kantor, 2012) showed that the evidence of environmental impacts is mainly of a qualitative nature, due to the lack of robust baseline and monitoring data and indicators. Besides, the expected long-term effects and the influence of further intervening factors limited the capacity to quantify policy impacts. The application of alternative indicators (different from those proposed by the Commission) was able to generate quantitative data, through the inclusion of secondary data collection (from national/regional sources), surveys and (to a lesser extent) the use of quantitative models and methods. The different sources of data sometimes led to inconsistent and or unreliable results.

The challenge of different levels of evaluation (micro-macro) did not arise from the comparative analysis by Kantor (2012). There was just a reference about the long time required to conduct in-depth analysis of programmes for thousands of beneficiaries. According to this report, four to six months may be an appropriate period for preparing, implementing and analysing the results of case studies. A final recommendation regarded the availability and the continuous measurement of environmental indicators since the start of the programme, which would greatly enhance the capacity to more effectively assess environmental impacts.

A further comparative analysis was realised on the mid-term evaluation reports prepared by MS at the end of 2010, as requested by the programming schedule of the Commission (Österreichisches Institut für Raumplanung, 2012). The report stressed the relatively low usability of CMEF indicators in the context of environmental impact assessment, and it claimed that a different approach may be required for the following programmes. The mid-term evaluation reports provide some interesting examples of more significant and feasible indicators. The lack of data and the difficulties in identifying the environmental impacts for non-Axis 2 measures are common features of many national reports. The aspects related to the micro- and macro-level evaluation was only cited with reference to the questions posed by the Commission, while in the review of national reports no evidence was found at all.

As showed in the next two sub-sections, additional information may come from the review of evaluation reports and from the interviews with stakeholders. Although these sources of information are not specifically devoted to investigate the issue of micro-level evaluation, they can provide some relevant reflections about the main challenges that affect and constrain current micro-level evaluations.

2.1 Information from the Review of Evaluation Reports

The information of this review has been drawn from the evaluation documents of the following Member States/Regions: Finland, France, Germany, Great Britain, Greece, Hungary, Italy, Lithuania and the Netherlands. More specifically, the focus and priority of the review have been retrieved by the following reports: 2000-2006 Ex-post evaluation results, 2007-2013 Mid-term evaluation and other relevant RDP evaluation reports. The critical point about this concerned the reporting frameworks used by Member States that were not always harmonised. Due to a lack of clarity, a great effort has been made over several evaluation reports and MS on the basis of the public good analysed, to evaluate the methodologies used for the impact indicators of RDP.

Regarding *climate change*, differences are underlined in the methodologies for impact assessment in EU. Generally, the methodological approach for measuring the net C storage lacks scientific knowledge, and thus its estimation is considered to be a poor impact assessment of the impact indicators; similarly the condition in countries such as Austria, France, Greece and Italy. Most have utilised the IPCC methodology for CO_2 absorption and C fixation. Several investigations were conducted for example in Veneto region (IT), in order to quantify the total C storage capacity of forests; however the results should be used cautiously due to uncertainties about the estimation methods proposed for various categories of management and for the forest areas. On the contrary, in England the evaluation of CO_2 absorption and C fixation, derived from forest-environmental measures, is based on indirectly related results and output indicators, since the impact on climate change mitigation is attributed to the wider programme level. The methodological approach used is based on questionnaires filled by the evaluators alongside an extensive literature review. Similar mixed assessments have been also found in Bulgaria, the Netherlands and Scotland. For this public good most of the evaluations were assessed at macro level programme.

Water quality was assessed by the gross nitrogen and phosphorus load (kg/ha), as well as the variation load of pesticides (kg/ha). The main weakness is represented by the limited extent of the area under AESs. Therefore, the methods used in France, England and Hungary only give an indication and estimation of the impact indicator on the public good, showing less effectiveness compared to the water quality objectives. In Latvia, no indicator is defined for assessing water quality, due to scarcity of data. In Greece the indicator 'Pollution by nitrates and pesticides' is not yet available. Hence the impacts on water quality have not been assessed. Similarly, in Poland the impact of measures on water quality cannot be estimated, since the data used are considered too general. In Italy the assessment of AESs could be considered as a counterfactual approach. It has been measured through indicators deriving from evaluation questions of the previous programming period, with micro-level information, and from the FADN database. The GLEAMS2 model is considered as the most efficient for estimating leaching of chemical fertilisers and pesticides. In Lithuania, the impact of farm modernisation is measured by the gross nitrogen balance, the nitrate pollution and pesticide pollution. It is mentioned that none of the indicators are suitable for assessing the impact on water quality. Given that calculations at national level do not reflect either spatial differences or temporal changes, additional measurements are required.

Regarding *biodiversity-wildlife*, most of the methods were assessed at macro level, through the aggregation of micro-level approaches. In Italy, for example, Farmland Bird Index (FBI) has been calculated over 26 species using data from 2000 to 2012. The yearly results are quite volatile; hence the interpretation has to be very cautious. A similar method was used in France, Finland and Scotland. In the evaluation reports for Italian regions, AESs impacts on *biodiversity-high nature value farmland* (HNV) are assessed by measuring the extent of agricultural land and farmland bird species population. The choice of the most apt methodology within a wide range of potential different approaches used for the calculation of HNV farmland is a key point in evaluation activities. Therefore, a combination of land use data (from CORINE Land Cover) with dissemination data of vertebrate species is proposed in France and Greece. In France, the classification of the different types of HNV farmland based on the IRENA methodology is considered questionable. In this case, the areas with high proportion of semi-natural vegetation have been assessed on the base of satellite photos provided by the CORINE Land Cover system, that does not distinguish between extensive and intensive management grasslands. Due to the lack of sufficient information on HNV indicator methodology, the French Mid Term evaluation report provides theoretical and methodological references for the ex-post evaluation in 2013-2014. The investigation of HNV indicators highlights the diversity of combining approaches, given the considered agro-ecological variety. In this regard the evaluations were assessed in most of the countries by aggregating micro-level approaches.

Regarding *soil*, in Italy the AES impact assessment is based on the size of areas under farming systems, aimed either at reducing/preventing leeching, run-off or sedimentation of farm inputs or at preventing/reducing soil loss. Indicators that measure the organic carbon content in the surface layer (0-30 cm), the maintenance/increase of organic matter content, or the risk of soil erosion (these indicators originating from CMEF and IRENA) have been used, mainly retrieving data from IACS, CORINE Land Cover and regional land use or erosion risk maps.

Landscape was mainly assessed at macro level in most of the EU nations. It seems that AESs, followed by the measure 'payments in areas with handicaps other than mountain areas', alongside the measures under Axis 3, 'conservation and upgrading rural heritage' and 'village renewal and development', are the most important measures influencing landscape assessment. Most of AESs aims to maintain or enhance the agricultural landscape. Furthermore, the measure of 'natural handicap payments in areas other than mountain areas' ensures the continuous use of farmland avoiding land abandonment and preserving both the heterogeneity and some specific characteristics of the rural landscape. Regarding this point there is little information and impact indicators established by EU partners, due to a lack of methodology for the estimation of impact indicators on landscape. Moreover, in some evaluation documents, the landscape level is recommended as the necessary bridge-level between micro and macro scale for biodiversity.

Finally there is a worrying lack of information about the evaluation and the impact indicators regarding the *animal welfare*. A concise study about animal welfare was carried out in Italy. The main approach was at micro level (farm level), while a further indicator was used for estimating the impact of measure 133 concerning the ‘share of animals on assisted holdings enjoying improved welfare thanks to assisted investments according to the type of impact (e.g. direct or collateral to animal welfare, related to national or EU welfare standards).

2.2 Information from Stakeholder Interviews

Interviews with stakeholders were carried out with the aim of assessing the practical approach to evaluating RDP impact indicators and the critical points about the assessment methodology. In this regard, some information related to RDP evaluation reports was taken into account. The approach is based on the evaluators’ judgment about the matched analysis of monitoring data and programme documentation. Most models used to assess the impact of the different indicators is summarised in relation to the analysed report from different countries. Generally, in some countries there is a lack of information about the methodologies, monitoring data and models used, that strongly weakens the judgement process and does not help to outline clearly strengths and weaknesses of current evaluations.

There have been some attempts to develop a model for the evaluation of water quality impacts in Scotland, but any models were used for biodiversity, soil and animal welfare impact assessment. Moreover, the models currently used and developed, such as the gross nutrient balance model, are too recent to become widely expanded (Scotland). In England some models have been used for C sequestration through forestry schemes. Examples of models focused on these issues have been suggested as a valuable basis for building CMEF indicators.

For the evaluation of the impact indicators regarding the identification of HNV farmland, no clear models are presented in Germany. In the case of climate change, the Thünen Institute built the Agriculture: National Emission Inventory Report and the GAS-EM model for manure application. Regarding nutrient balance, a series of models could be used for impact assessment, although most of them are not used as they do not meet both the evaluation and evaluators requirements. Evaluators do not develop and/or apply complex bio-physical models, but integrate results from other modelling studies carried out by ministries and government and non-governmental organisations.

In some cases the use of modelling was assessed on the basis of the measures 213, 214, 211 and 212, specifically for biodiversity, landscape, soil quality and water quality. However there is no evidence on the models and their complexity, as in Poland. In Finland models have been mainly used to define impacts on water pollution and climate change. Another model used was MYTVAS, related to reducing nutrient load and fertilisation level. This model takes into account grassland, manure handing, filter strips, buffer zone, winter vegetation cover, diverse crop rotations and organic farming. Conversely, in Hungary no models were used for biodiversity, water and soil evaluation, while landscape assessments were mainly based on landscape visualisation. In Italy most of the collected data were used at territorial level, supported by models for the evaluation of the given public good. Regarding soil quality, the RUSLE model was used with GIS support. In other cases, no specific model was used for biodiversity, water quality, landscape and animal welfare.

The above examples of current evaluation represent a helpful step toward a better understanding of the effectiveness and the impacts of RD programmes and measures. However, they still present a considerable gap when applied to the micro level for two main reasons. First, a lack of data appropriate and specific to the ‘level’ of analysis can seriously undermine the accuracy of the evaluation. Additionally, environmental monitoring data should be directly linked to the assessment data of each public good. Better standardisation of data collection would enable a more fixed, holistic and explicit way of assessing impacts. However, this requires suitable data sources to be identified, causally linked to each other, and frequently monitored.

Second, generally, impact analysis could also benefit from using qualitative approaches in the early programme design, to create a ‘common methodology’. Beyond methods/models that provide quantifiable micro-level impact evaluations, the use of qualitative methods should be corroborated and would be highly desirable. In fact, although a qualitative assessment is unable to give tangible impact analysis, it has the ability to create a logical view of the factors affecting impacts. The cause-effect chain of a programme and of its outcomes is valuable in evaluating impacts.

3 Literature Review

3.1 Research Approach

The references examined in this study have been classified in order to group the papers according to criteria which facilitate the identification of relevant references and the comparison of their contents. The methodological approaches are analysed and compared in order to highlight the most relevant contributions of scientific literature to main challenges of micro-level evaluations. This review offers a survey of the recent literature dealing with the use of methods, models and indicators for the assessment of agricultural systems and their effects on environmental components (water, climate change, air, biodiversity, soil, landscape and animal welfare), from multiple disciplinary perspectives and considering primarily studies with at least a micro-level approach.

The research approach is based on literature review, mainly provided by ISI Web of Science (approximately from 2000 to present) and the bibliographies of the papers found therein. The literature search covered a total of 73 studies, only published in English that explicitly refer to either agri-environment measures or other similar RDP measures. The review is mainly focused on studies from European countries; however it is enriched by some relevant contributions from others countries that presented similar policy frameworks, such as USA, Australia, New Zealand, China and a few other countries. In particular, in case of Ecological Footprint analysis, some non-European scholars have been cited due to their relevance for this topic, and some references relate to books and book chapters.

An inventory of analysed studies has been created. First of all, each paper has been classified on the basis of the type of public goods analysed and of the geographic location. Additionally, another classification is based according to the used scale (micro or micro/macro), assessment approach, use of specific models (biophysical, sampling, spatial, mathematical, cognitive, computational), data requirements, presence of control groups. Short summaries and comments about the general aims and the specific methodology and results of the papers complete the inventory.

Table 1 and Table 2 summarise the result of the classification work, without any attempt to proceed with a statistical analysis of the selected papers. See Annex 1 for the complete list of papers with the classification criteria used.

Table 1 Typologies of models and criteria of classification in the selected papers

	Total	Biophysical	Sampling	Spatial Analysis	Agent Based Model	Math programming	Multi-criteria	Expert	Socioeconomic	Ecological Footprint
Agent-Based Modelling	3	2		1	3	2			1	
Sustainability Indicators	5	4					4	1	1	
Statistical sampling	28		28	2					1	
Statistical Sampling / Meta-analysis	1		1							
Biophysical	1	1								
Spatial analysis	7		1	7			1		2	
Biophysical and Spatial model	4	4		4						
Integrated model	11	10		5		7	1		11	
Integrated model / SEAMLESS	4	4		4		3		2	4	
Network Analysis	1						1	1		
Socioeconomic	5	1		1		2	2		5	
Ecological Footprint	3									3
Total	73	26	30	24	3	14	9	4	25	3

Source: our data base.

Table 2 Typology of models and public goods analysed in the selected papers

	Total	Biodiversity	Climate Change	Landscape	Soil	Water	Indicators
Agent-Based Modelling	3			1		2	
Sustainability Indicators	5						5
Statistical sampling	28	27		5	1	1	
Statistical Sampling / Meta-analysis	1	1		1			
Biophysical	1					1	
Spatial analysis	7	3		1	2		1
Biophysical and Spatial model	4			2	1	3	
Integrated model	11		1	1	3	7	2
Integrated model / SEAMLESS	4					2	2

Network Analysis	1					1	
Socioeconomic	5	1	1	2			2
Ecological Footprint	3		1			2	
Total	73	32	3	13	7	19	12

Source: our data base.

3.2 Outcomes of the Review

3.2.1 Linking changes in farmer behaviour with environmental change

Considering the perspective of environmental impact assessment for RDPs at micro level, it is possible to define different units of analysis (e.g. field, farm or RD measure) at different geographical scale. The farm-level is a significant unit of study for environmental policy analysis, due to the focus on individuals (farmers) and their interactions within a defined physical, socio-economic and policy context. Specifically, considering the environmental impact of RDP as a structural change for the system (land and farm management) relating to the behaviour and interaction of individual farmers, the farm-level decision models can be useful and effective tools for analysis. The environmental impacts of these changes can be estimated introducing linkages with bio-physical models at farm scale. To date, researchers often use farm-level decision models to assess behaviours and changes with Agent-Based Modelling (ABMs) approaches. Scientific literature shows different terms referred to this approaches. Hare and Deadman (2004) provide a taxonomy of agent-based models applied in environmental management, distinguishing among: agent-based modelling; agent-based simulation modelling; multi-agent simulation; multi-agent-based simulation; agent-based social simulation; and individual-based configuration modelling.

Substantially these approaches allow the coupling of environmental models and the social systems embedded in them. In this way the role of social interactions and of adaptive, disaggregated (micro-level) human decision making processes in environmental management can be modelled. In short, the development and use of ABMs for ecosystem management allows us to consider ecological complexity. It is possible to identify the role of individuals and to analyse in more depth and more effectively the different forms of organisation (spatial, networks, hierarchies) and interactions among different organisational levels. Behaviours and interactions are the key issues for understanding and modelling the organisation of ecosystems and the environmental impacts related to ecosystems management (Bousquet & Le Page, 2004). In the specific context of the RDP, several components are to be considered, such as the number of beneficiaries in each region, the rural development at the initial state, interdependencies among farms and the social, economic and technical environment. In this sense, structural changes related to rural development are considered as a complex dynamic process, from which also occur environmental impacts.

Everything features a dynamic stochastic general equilibrium defining a condition of non-linearity given by independent behaviour of subjects, technical restriction, aspects of space and time, that generate a structural change (Balmann, 1997). In short, the ‘agents’ are software entities that respond to stimuli to act upon their environment (Russell & Norvig, 1995). All agent-based systems that represent human behaviour can be called ‘artificial life agents’ (Sengupta et al., 2005). ABMs allow the representation of economic and social systems as the result of individually acting agents. When applied to agriculture, they can simulate, at the micro-level, the behaviour of individual farmers, without the need to aggregate them in ‘representative’ agents, and subsequently generate the macro (aggregate)-evidence. Furthermore, ABMs can catch the iterations of heterogeneous farms when competing over common finite resources (Lobianco & Esposti, 2010). ABMs have received increasing attention in recent years, in particular adapting to land use modelling, mainly because it offers a way to incorporate the influence of human decision making on land use in a mechanistic, formal, and spatially explicit way, taking into account social interaction, adaptation, and decision making at different levels (Matthews et al., 2007; Balbi & Giupponi, 2009).

Recently several studies have been carried out at micro level applying the agent-based approaches to analyse the environmental impacts of land and farm management changes (Le et al., 2008; Valbuena et al., 2010; Le et al., 2010; Luus et al., 2013) and the drivers of land use change (Polhill et al., 2001; Mena et al., 2011). By defining and using different decision making strategies, the conceptual framework allows the inclusion of the diversity of farming systems. This diversity is a critical factor in explaining the interactions between farmers’ decision making, and it can be applied to the landscape structure of a rural region (Valbuena et al., 2010).

The use of ABMs in the agricultural context has been pioneered by Balmann (1997) who developed the Agricultural Policy Simulator (AgriPoliS) model to demonstrate the existence of a path-dependent evolution of land use in specific policy contexts. This opened the way for ad hoc applications to investigate the adaptations to structural changes and agricultural policies under various framework conditions. In these cases, the policy impact analysis is considered as a complex modelling problem at the farm and regional scale, because agricultural policies influence the decision making of individual farms, which is widely heterogeneous in terms of attributes such as factor endowments, ownership structure, location, farm management, and the competitive position on markets for products and production factors. Agent-based simulation offers a conceptual framework to approach this modelling problem because it facilitates the capture of this heterogeneity and the dynamics between agents. Adopting a bottom-up approach, ABMs are able to capture endogenously the process of structural change (Happe et al., 2006). The main focus of the work is on simulations that allow for good ex ante evaluation, while the use of ABMs for ex post evaluation seems less frequent (if not totally absent) in the scientific literature.

Considering case studies on single public goods, many applications of ABMs are focused on water resource management, such as the CatchScape model (Becu et al., 2003), a Multi-Agent System that enables the simulation of the whole catchment features as well as farmers' individual decisions. CatchScape aims at being integrative, spatially distributed and individually-based in order to cope with complex and adaptive issues at the catchment scale and many interactions between socio-economic and biophysical dynamics. The main objective of simulations is to provide a dynamic environment in which local stakeholders can explore the implications of alternative management approaches. The ABMs give a useful framework for predicting the response of multiple firms to general policies for limiting non-point source pollution, even in the case of specific agri-environment measures for improving water quality (Doole et al., 2011). Carpani et al. (2008) present a framework for assessing the effectiveness of agri-environment schemes in promoting social and environmental sustainability in Europe, based on expert knowledge. Expert knowledge - acquired through adequate elicitation strategies and managed with robust and transparent methodologies - can help to build an information system that infers the effectiveness of agri-environment measures, if not in terms of quantitative absolute estimations, at least in comparative terms. The authors develop a conceptual model, in workshops involving local experts, for depicting the physical and socio-economic systems of the case study area.

3.2.2 The use of sustainability indicators

The literature on indicators is quite abundant and partially outside the purpose of the survey. However, some of these studies are used here to show how the micro-level approach has been dealt with, essentially trying to downscale/upscale the analysis at/to farm level. Girardin & Bockstaller (2000) develop an evaluation method applying the multi-criteria methods to generate agro-ecological indicators in order to evaluate the potential impacts of arable farming systems on the environment. Multi-criteria methods represent developing tools that could better investigate the decision making process under environmental limitations (Zander & Kachele, 1999; Uthes et al., 2010). Pacini et al. (2003) develop an environmental accounting method based on site-specific environmental indicators, and environmental externalities generated by farming cycles were measured at farm level. Due to limitations in data availability and a focus on the link between farming practice management and the farmer decision making process, some studies concentrated more on landscape indicators (Piorr, 2003) or pressure indicators (Lütz & Felici, 2008). Moreover, van Passel & Meul (2012) implemented an empirical model to measure farm sustainability using multi-level and multi-user sustainability assessment starting from the FADN dataset.

Some studies attempt to build up a farm level composite indicator to assess social, economic and environmental issues (Purvis et al., 2009; Gómez-Limón & Sanchez-Fernandez, 2010; Reig-Martínez et al., 2011; Louwagie et al., 2012). This approach combines different methodologies mainly derived from social sciences, such as Data Envelopment Analysis and Multi-Criteria Decision Making methods. The implementation of composite sustainability indicators emphasises farm heterogeneity within a single agricultural system with respect to sustainability, as well as analysing the intervening structural and decision-oriented variables. Some of these studies use stakeholder knowledge to fill the gap between measurability of human environmental action and resulting environmental change to better define policy evaluations, where immediate quantitative data may not be available (Wilson & Buller, 2001; Lütz & Felici, 2008). For example the Agri-Environmental Footprint Index (AFI) is a composite indicator designed to quantify environmental impacts at farm level for statistically representative samples. AFI is based on multiple criteria methods and represents a harmonised approach to evaluateingRDP environmental impacts (Purvis et al., 2009).

A further relevant approach based on sustainability indicators is ‘eco-footprinting’, a quantitative measurement that encloses the potential indicators for measuring sustainability in natural resource management by social systems (Hoekstra, 2008; Cucek et al., 2012). Footprint methods are based on the effects of human carrying capacity on environmental, economic and social sustainability. Regarding the environmental dimension, Cucek et al.

(2012) identify different types of footprint based on the Ecological Footprint (EF) concept as provided by Wackernagel & Rees (1996). This is defined as the area of biologically productive land and water an individual, population or activity requires to produce all the resources it consumes and to absorb the waste it generates, using prevailing technology and resource management practices.

In this way, EF analysis accounts for both population size and resource consumption, and thus provides a measure of human 'load' considering the human carrying capacity as the maximum persistently supportable load; thus the larger the footprint, the greater the load. The main strength of EF is the conceptual and intuitive simplicity. It focuses on natural resource consumption, and consolidates the data on the associated energy and material flows into a single concrete variable, land area (Rees, 2000). However, EF does not produce a complete picture of ecological sustainability. Its main weakness is related to the excessive simplicity of the analysis. EF is not all-inclusive, and the method does not account for the cause or effects of an environmental problem. In fact calculations do not even describe the whole land and water story. Only major categories of consumption can be included, excluding the net effects deriving from the categories most difficult to be evaluated. This runs the risk of an over-simplification of the complexity of environmental effects (Rees, 2000) that may affect the effectiveness of the final evaluation by EF.

In recent years, Carbon Footprint (CF) and Water Footprint (WF) have been implemented on different activities and sectors. CF measures the amount of carbon dioxide emissions that are directly and indirectly caused by an activity or accumulated over the life stages of a product (Wiedmann & Minx, 2008). Carbon has been considered as one of the biggest emitters of global GHGs. Methodologically, CF is assessed through a whole analysis of total emissions from all inputs in a Life Cycle Assessment (LCA) of production or consumption (Cheng et al., 2011; Finkbeiner, 2009). CF can be considered as a direct derivation from LCA approach, that is based on International Standards (ISO 14040, ISO 14044) and on environmental labels and declarations (ISO 14020, ISO 14024, ISO 14025).

WF is based on the concept of virtual water (Chapagain et al., 2006; Zhang et al., 2011) and represents the total volume of direct and indirect fresh water used, consumed, and/or polluted. WF consists of different measurements: a) the blue water footprint, which represents the consumption of surface and ground water; b) the green water footprint, derived from the consumption of rainwater; c) the grey water footprint, the volume of water required to dilute pollutants to water quality standards (Mekonnen & Hoekstra, 2010; Klemes et al., 2009).

3.2.3 Application of statistical sampling and up-scaling questions

An evaluation of environmental impacts based on empirical evidence needs statistical sampling methods to be applied. The studies try to verify the existence of causal links between farming practices and environmental impacts, in many cases analysing the changes induced by policy measures. The bulk of scientific literature is focused on biodiversity, particularly in the analysis of variation in animal and plant species richness, abundance, diversity and frequency in sample sites and plots. The species number in a community is referred to the species richness and the relative abundance of species (Walker et al., 2007). For example, although two communities may contain the same number of species, one community may be dominated by one species while the other one may contain large numbers of all species (Zollinger et al., 2013). Most of the ecological investigations (number of species, nutrient content, microbial analysis in the soil) may be useful to measure the diversity of one taxonomic group. For example, if plant ecologists are interested in studying plant species, they would measure plant diversity and exclude other kinds of organisms.

Furthermore, diversity can also be separated into different components - as an average diversity on a plot scale, and between plots, giving in this case the measure of variation in species composition between plots (Flohre et al., 2011). From the key work of Klejin et al. (2001) and Klejin and Sutherland (2003) about the role of protection and promotion of biodiversity for European AES, a series of biodiversity-focused papers has thus developed sampling designed analysis, giving valuable references for scholars and evaluators (Batáry et al., 2011). For this review, the statement of a prevalence of sampling models in our results is influenced by the fact that about 40% of reviewed papers focus on biodiversity, as confirmed by Uthes and Matzdorf (2013) who show that about half of studies on agri-environment measures, schemes and programmes are dedicated to biodiversity. Notwithstanding other public goods, such as water, have a significant but less important number of scientific studies. For examples, Ekholm et al. (2007) present a study based on monitoring of data on fluxes of nutrients and total suspended solids at catchment level, emphasising the role of nutrient reserves of the soil in the apparent inefficiency of the agri-environment schemes of RDP.

The evaluations tested with sampling designs often imply also a counterfactual approach, through the comparison of trends on treatment fields and control fields, both before and after implementation of the treatment (Klejin et al., 2006). The high relevance of sampling models, at least in evaluation methods on biodiversity, also means a high number of evaluations have been carried out at micro level with surveys in case study areas (Zollinger et al., 2013; Roth et al., 2008; Baldi et al., 2013). In fact, for biodiversity, the macro perspective has less to say about the link with species variables, but there is the sense that the individual and detailed analysis can give more useful results (Knop et al., 2006). In this way, most of the studies focus on the evaluation of specific impacts at local level (micro scale) and the dialogues with the macro level occurred almost completely independently (Fuentes-Montemayor et al., 2011; Gleason et al., 2010; Olsson et al., 2009). Some effort was made to apply sampling methods in a consistent way at field, farm and landscape scales to take into account the different taxa studied and the interactions among variables describing climate, topography, land use, socio-economic and soil conditions (Gabriel et al., 2010; Batáry et al., 2011). The 'bottom-up approach', with a study area at plot or field level, allows results to be obtained through the collection of data from site-specific surveys following experimental protocols, and to extrapolate them from micro level with GIS, satellite images or spatial analysis (Reynolds et al., 2006; Rundlöf et al., 2010; van der Horst et al., 2010).

Sometimes, this approach - and sampling models in general - suffers from a lack of consistency and robust empirical data. In fact, according to Primdahl et al. (2010), even when appropriate sampling designs are applied, it may be difficult to identify the reasons for an absence (or insufficient magnitude) of effects. For example, it may be difficult to distinguish between an incorrect impact model that is implemented as specified, and a correct impact model that is not properly implemented. This approach may inappropriately trivialise the complexity of the system, and compromise the understanding of the set of factors involved in the evaluation, giving a mistaken idea of the system. Thus, this can misrepresent the advantages or weaknesses of sampling models, both the analytical and up-scaling methods.

3.2.4 Integrated models and the complexity of agro-ecosystems

The application of integrated models in agri-environment evaluation acknowledges the complexity of intervening factors impacting on biophysical processes. According to Le Gal et al. (2010), interactions between techniques implemented by farmers, their sequencing in time and space and environmental process are complex and intricate. It is necessary to focus on the farm scale and its sub-sets, such as cropped areas. In fact, this is the level for which farmers allocate available land, labour, equipment, and capital resources to the various tasks in their production systems. Furthermore, farmers have more options at the farm level when technical solutions at the field level are difficult to implement. Given the complexity within the potential range of intervention strategies, the design of innovative farming systems relies on modelling to represent the biophysical, technical and decision processes involved, and to evaluate the impacts of technical or organisational innovations.

Integrated models are therefore able to shed light on the environmental, economic and social components and contributions of a multifunctional agriculture. They are necessary to address more holistically the evaluation questions. Among the more commonly-used integrated models for analysing environmental impacts, the SEAMLESS (System for Environmental and Agricultural Modelling; Linking European Science and Society) project represents an integrated assessment of agricultural systems that allows a consistent multi-scale analysis to assess and compare, ex-ante, alternative agricultural and environmental policy options. In particular, this approach allows the analysis at the full range of scales (from farm to EU and global), so focusing on the most important issues emerging at each scale and shedding light on the environmental, economic and social contributions of a multifunctional agriculture (van Ittersum et al., 2008; Ewert et al., 2009; Olsson et al., 2009). Data require to be collectable, consistent and available for dynamic biophysical models, static bio-economic farm models and partial equilibrium market models. This means integrating several data sources related to European agriculture, including economic, biophysical, climatic data, model simulation input and output data, scientific workflow configurations and calculation of indicators into a single relational database schema (Janssen et al., 2009). In this way, an important task of SEAMLESS has been the development of an indicator framework (goal-oriented framework (GOF)) that could assist the users of the SEAMLESS Integrated Framework (SEAMLESS-IF) to select indicators to be used in the assessment of the impacts of a new policy.

Integrated models seem to be mainly preferred for evaluating environmental impacts on water and soil quality, land use and climate changes. In the case of *water quality*, the integrated models combine, for example, hydro-economic and hydro-geological models, to simulate the evolution of groundwater quality, with agronomic and economic components to assess the expected costs, effectiveness, and benefits of AES implementation (Uthes et al., 2010; Hérivaux et al., 2013). Within SEAMLESS, a simulation tool was developed to assess the influence of changes in the decision system at the farm level in terms of grass irrigation, hay mowing, and upstream water distribution on the production system’s agronomic performance and environmental impact. The analytical tool integrates both the farmers' and upstream water managers' decision systems as well as the functional relationships between water supply, hay production, and water flows at the field scale. Furthermore, it uses a dynamic decision making model at the farm scale, a dynamic crop model at the field scale, and a hydraulic model at the border scale, that are considered as three independent and modular modules (Le Gal et al., 2010).

In the case of *soil erosion/land use*, a typical integrated model is the bio-economic MODAM (Multi-Objective Decision support tool for Agri-ecosystem Management), that assesses the soil-related characteristics of cropping activities and the economic effects of agricultural land use (Schuler & Sattler, 2010). MODAM provides information on all specific pesticide applications, fertiliser usage, or the periods and when each work step is done on a field. For several production-intensity levels, either for crop or animal production, a specific set of input combinations is generated for each product, corresponding to different points on the production function. In the same way, its economic model is based on a linear programming model, that maximises the gross margin of a region under restrictions such as policies, land and labour availability and technical coefficients of production; this simulates the behaviour of farmers with the aim of gaining profit from their farming activities (Zander & Kächele, 1999).

Additionally, an interesting project is LUMOCAP, which consists of a selection of models, all linked into a single integrated model able to simulate the linked bio-physical and socio-economic developments in the European Union (EU-27) up to 30 years forward (van Delden et al., 2010). This model is characterised by a holistic multi-scale approach that matches models working at four different levels: European, national, regional and local. The main limitation of LUMOCAP is where there is a lack of consistent data sets for a long time series of land use and land use change, and to retrieve a 10-years base for the model is challenging.

Another category of integrated models is essentially based on the links between biophysical models and spatial analysis, but without explicit reference to the decision making process developed in the socioeconomic sphere (Ekholm et al., 2007; Kersebaum et al., 2006; Tzanopoulos et al., 2012; Yuan et al., 2011). Many studies use spatial information on soils, climatic zones, land use, and distribution of AES to assess the reduction effect of policy measures on nitrogen pollution of water resources. In these cases, the use of different data sources and maps raises problems with incongruent topologies that exacerbate the uncertainty of adopting different databases. In this sense sophisticated physically-based models suffer from problems related to model complexity, computational efficiency, and parameterisation constraints (Kersebaum et al., 2006).

Levels and scales of environmental impact evaluation studies mainly depend on the environmental components. In the case of water, for example, the major impairments to aquatic ecosystems in many watersheds consist of physical habitat degradation, suggesting that some relevant components of aquatic habitat represent potential variables to analyse for ecological impacts and their interactions (Granlund et al., 2005; Shields et al., 2006). In these cases, often the scale of analysis is the catchment level. Quite frequent is also the use of multi-criteria approaches through sorting, selecting, or classifying cropping systems or farming systems according to their environmental effects (for example, Girardin & Bockstaller, 2000; Hill et al., 2005; Uthes et al., 2010).

Outside the European Union there are other interesting experiences. For example the USDA's model 'Annualized Agricultural Non-Point Source Pollution – AnnAGNPS' that Yuan et al. (2011) apply on a large scale, through the exploration of spatial resolution and accuracy of the model, and applying it to current and future landscape scenarios to look at potential N-loading changes caused by increased corn production. The authors show the applicability of the model as guidelines for future planning to evaluate the impact of future land use changes. Fennessy & Craft (2011) estimate the cumulative ability of the conservation practices implemented under the Farm Bill, to retain sediment, nitrogen, and phosphorus in Upper Mississippi River Basin watersheds, highlighting the need for indicators that can document the ability of conservation practices to deliver ecosystem services and the spatial distribution of conservation practices.

To a certain extent, integrated models have symmetrical features in comparison with statistical sampling methods, as they obtain consistent results at multiple levels based exclusively on simulation results without empirical evidence of the environmental impacts. The lack of experimental measurements is substituted by calibration procedures that have the aim to adapt the models to local conditions. In general the use of large amounts of spatially-explicit data, mainly represented by land use, farming practices, soil and climate conditions, helps to generate more robust estimations.

3.2.5 Comparative score of the models to be used at farm level – Strengths and Weaknesses

As previous paragraphs have shown, five main categories of methodologies/models have been used at micro level:

- Sustainability Indicators
- Statistical Sampling and Monitoring
- Bio-physical and spatial models
- Agent Based Modelling
- Integrated Models (with modules at farm level).

These five categories are not mutually exclusive and present some overlaps. For example, integrated models are based on empirical evidence coming from statistical sampling and monitoring; sustainability indicators could result from the outputs of integrated models, while biophysical models and spatial analysis are frequently considered within integrated models.

Based on the state-of-the-art resulting from the literature review of models at farm level and according to Tables 1 and 2, the most used ones have been considered and discussed in the following Table 3, in order to show their strengths and weaknesses in terms of RDP environmental impacts assessment. Each method has been scored according to their relevance in terms of micro-level applicability, based on four Strengths and three Weaknesses that are common to each selected model. Furthermore, relevant specific Strengths and/or Weaknesses have been added for each model, to comprehensively describe their suitability at micro level.

Four models have been considered within the discussion: Agent Based Modelling (3.2.1), sustainability indicators (3.2.2), statistical sampling (3.2.3), integrated models (3.2.4). Biophysical and spatial analysis models have not been described because they present similar Strengths and Weaknesses to sustainability indicators, and thus their analysis has been included within the sustainability indicators. Conversely, EF, described within the sustainability indicators in the Paragraph 3.2.2, has been considered in the Table 3 as independent, because it represents an innovative and work in progress methodology that can stimulate ENVIEVAL research interests, particularly for the water quality and climate stability public goods.

Table 3 Categories of Models to be used at farm level - Strengths and Weaknesses

Categories of models	Strengths	Relevance	Weaknesses	Relevance
Sustainability Indicators	Micro/macro linkage	++	Limited availability of farm level data	++
	Applicability to counterfactual analysis	++	Lack of consistency, robustness and representativeness of micro level data	++
	Bottom up approach	+++	Not applicable to all public goods	++
	Interaction and integration with other models	+++	More oriented to pressure indicators (farm practices instead of actual environmental impacts)	+++
	Possibility to obtain composite indicators	+++		
Ecological Footprint	Micro/macro linkage	+	Limited availability of farm level data	+
	Applicability to counterfactual analysis	++	Lack of consistency, robustness and representativeness of micro level data	+
	Bottom up approach	+	Not applicable to all public goods	+
	Interaction and integration with other models	+	Complexity models require high-skilled evaluators	++
	Measure of the environmental impacts of the Life Cycle (e.g., energy, water)	+++	Strict requirements related to data processing	+++
			Strictly dependent by the selected productive system	+++
Statistical Sampling	Micro/macro linkage	++	Limited availability of farm level data	++
	Applicability to counterfactual analysis	+++	Lack of consistency, robustness and representativeness of micro level data	+++
	Bottom up approach	++	Not applicable to all public goods	+++

	Interaction and integration with other models	+++		
	Good quality of data processing	+++		
Agent Based Modelling	Micro/macro linkage	+	Limited availability of farm level data	+
	Applicability to counterfactual analysis	+	Lack of consistency, robustness and representativeness of micro level data	++
	Bottom up approach	+++	Not applicable to all public goods	++
	Interaction and integration with other models	+++	Difficulties in case of ex-post evaluation	++
	Possibility to recognize and simulate the role of individuals	+++	Complex models require high-skilled evaluators	++
Integrated Models (with modules at farm level)	Micro/macro linkage	+++	Limited availability of farm level data	++
	Applicability to counterfactual analysis	+++	Lack of consistency, robustness and representativeness of micro level data	++
	Bottom up approach	++	Not applicable to all public goods	+
	Interaction and integration with other models	+++	Need to have large amount of consistent databases	+++
	Integration of data sources	+++	Use of representative farms	++
	Possibility to simulate the environmental system conditions and impacts	+++	Necessity of bio-physical models to estimate environmental impacts	+++
			Complex models require high-skilled evaluators	++
			Strictly dependent by the selected productive system	+++

Source: our data base.

Therefore, three scores have been assigned to indicate the Low (+), Medium (++), High (+++) relevance of the specific Strength: (i) Micro/macro linkage; (ii) Applicability for counterfactual analysis; (iii) Bottom up approach; (iv) Interaction and integration with other models, and Weakness: (i) Limited availability of farm level data; (ii) Lack of consistency, robustness and representativeness of micro level data; (iii) Not applicable to all public goods. Table 3 highlights differences, large in some cases, within Strengths and Weaknesses among the selected models.

Regarding Strengths, in terms of micro/macro linkage a low score has been assigned to the Agent Based Modelling, due to it being strictly related to the behaviour of the single farmer, that cannot be easily generalised and/or converted to macro-level behaviour. Conversely, micro/macro linkage capacity represents a relevant strength for statistical sampling, whose sample is intrinsically flexible and adaptable to different sizes and scales. In this way, for example, the bottom-up approach is more applicable in the case of statistical sampling rather than EF, where the up-scaling process is strictly dependent on the types of targets to be evaluated. Thus, specific farm-level data cannot significantly represent the complexity of the statistical universe.

Regarding the counterfactual approach, this can be easily adapted to statistical sampling and integrated models, given that they allow the targeting of different typologies of farms, and thus it can be easier to find samples of farms presenting similar characteristics to be evaluated in the comparison RDP beneficiaries/not beneficiaries.

Furthermore, integrated models allow us to simulate the environmental systems' conditions, and thus they may potentially estimate and assess RDP environmental impacts. This is mainly due to these models (such as SEAMLESS) using different data sources and sets, and can have access to good quality data at farm level in terms of processing, temporal dimension and representativeness of the samples. Therefore, the outputs of these models are independent modules that can be integrated to easily up-scale the resulting assessment. For example, in the case of water quality, for the nitrogen leaching measure, two independent modules can be used respectively for soil nitrogen contents and nitrogen leaching in the water. This modularity represents a strength also for sustainability indicators and statistical sampling. In fact, this modularity allows us to use data characterised by different status (for example, rough or processed, or with different temporal dimensions or different levels, such as cadastral or landscape) and thus to produce evaluation results independently of the quality and the availability of data, data sources and sets.

With regard to Weaknesses, limited data availability often affects the applicability of the models for the environmental impact assessment. In fact, in the case of EF and ABMs, when data are poor or unavailable there is a lack of consistent, robust or representative results. Finally, not all the models can be applied to all the public goods. For example, statistical sampling is strictly limited to biodiversity, while oppositely integrated models and sustainability indicators can be suitable for different public goods. In this way, the limited application of EF in environmental evaluation (water quality and climate stability) is probably due to the relatively new method and the specific knowledge over the plast ten years.

4 Concluding Remarks and Recommendations

The present review has addressed several issues concerning the evaluation of environmental impacts of RD measures and programmes from the micro-level point of view. According to the research questions presented in the sub-chapter 1.4 we can draft four preliminary conclusions.

1) The degree of conditioning of past and current RDP evaluations by the existing challenges. As shown from the analysis of evaluation reports and confirmed by evaluators, the lack of appropriate and specific data can undermine the results of the evaluation exercises. Sometimes poor data availability is also due to lack of access to administrative and statistical databases because of privacy regulations or ill-coordinated efforts to collect information for monitoring purposes. The minimal requirements for suitable data sources should be causally linked to each other and frequently monitored. Furthermore, the use of qualitative approaches (common sense) is quite frequent, not only for the lack of data or of financial resources for creating new database but also for the difficulties encountered by evaluators using complex methodologies that could guarantee a good outcome from the evaluation process. This knowledge gap has to be taken into account when the following three conclusions are drawn and discussed.

2) The most commonly adopted methodologies within scientific literature. The review of papers by international academic journals demonstrates that the most adopted approaches are based on sampling methods and integrated models. Sampling methods allow the account of a given plot or field, and try to generalise the main related micro-scale findings to a different

scale perspective (e.g. for the evaluation of specific local AEMs or case study based evaluations of impacts on grassland biodiversity including adequate control group design of non-participating farms or parcels). However, significant problem is the data availability and the reliability and degree of accuracy for findings to be generalised. In fact, the greater risk is to depict a condition that, actually, is more complex than the scientifically reported one. Conversely, integrated methods represent a well-established trajectory of current agri-environmental evaluation in the academic field. Integrated models can help to generalise micro-scale results in a macro-scale perspective. Their interchangeable and flexible characteristics are able to create a set of tools to build an analytical framework that a) is more holistic and can integrate ad hoc evaluations, and b) can become a candidate to propose common strategies across the multi-level context for the evaluation of agri-environmental impacts. Furthermore, a specific focus can be addressed to the EF, whose use in environmental evaluation, although limited, is increasing particularly for water quality and climate stability. The strengths of these models are related to the capacity to analyse natural resources consumption in agricultural sector, consolidating the resulting data on the associated energy and material flows into a single concrete variable, land area. EF allows us to measure the human 'load' considering the human carrying capacity as the maximum persistently supportable load. Conversely, EF does not account for the cause or effects of an environmental problem, including only major categories of consumption and excluding the net effects deriving from the categories most difficult to be evaluated (Rees, 2000).

3) Unavailability of data. Addressing the main methodological gaps, the adoption of the aforementioned models will obviously continue to be problematic regarding the long term unavailability of specific datasets for a vast range of socio-economic, environmental and institutional variables. As also stressed with the stakeholders interviews, the lack of appropriate and specific data for the 'level' of analysis and for the examined public good can seriously undermine the accuracy of the evaluation. Standardising the procedures to collect data would enable a more unchanging, holistic and explicit way of assessing impacts. This requires that suitable data sources are identified, causally linked to each other, and frequently monitored. Moreover, within the literature, the sampling models are widely used, mainly focusing on biodiversity, and are versatile in depicting the systemic structures both of animal and plant communities. On the other hand, the models can be descriptively accurate for the given plots or farms, but less reliable in evaluating the effectiveness of a RD strategy as a whole at a non-local or timely scale. Additionally, sampling models in biodiversity are strictly

related to the scale questions. The fact that the available literature shows a high heterogeneity of ecological effects of AES and other RDP measures demonstrates that this vast ‘range of effectiveness’ depends on adopted scales, scheme factors and investigated indicators. Despite the general impression of low AES effectiveness, it is in fact difficult to judge a) whether the effects of AES really are disappointing, or b) are the result of methodological problems and general criticisms in the scientific discourse that may lead to ‘failure’ cases. Evidence of positive impacts is obscured by the fact that the spatial reference, the measures themselves (codes and content) and even the indicators for judging their performance often changed along the years (Uthes & Matzdorf, 2013).

4) Relationship between micro and macro approaches. One of the future challenges for the advancement of ENVIEVAL is represented by clearly understanding the relationship between micro and macro approaches within complex systems such as the agri-environmental ones. In fact, the ‘macro-system’ is characterised by interacting components (micro level), where each component is part of the system (farming systems). Consequently, its behaviour does not derive from the sum of effects, but from the presence of ‘emergent properties’ that occur only in a certain state of organisation. These properties have a direct impact on ecosystems. Taking into account methodologies based on complexity theories such as ABMs that reflect the interactional effects (e.g. substitution effects), it is possible to draw a study approach. These methodologies can be represented by ABMs and integrated models. Through these models ecological, environmental, socio-economic and institutional complexity can emerge, both identifying the role of individuals (ABMs) and analysing in depth the different forms of organisation (spatial, networks, hierarchies) and interactions among different organisational and intervening levels (ABMs and integrated models).

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Annex 1 - List of selected paper from scientific academic literature

Authors	Year	BIO	CLI	LAN	SOI	WAT	IND	Biophysical	Sampling	Spatial Analysis	Agent based Model	Math programming	Multi-criteria	Expert	Socioeconomic	Ecological Footprint	Type of Model Used
Albrecht et al.	2007	1							1								Statistical sampling
Albrecht et al.	2010	1							1								Statistical sampling
Allouche et al.	2012	1							1	1							Spatial analysis
Baldi et al.	2013	1							1								Statistical sampling
Batary et al.	2011	1		1					1								Statistical Sampling / Meta-analysis
Becu et al.	2008					1		1		1	1	1			1		Agent-Based Modelling
Blanco-Gutiérrez et al.	2013					1		1		1		1			1		Integrated model
Carpani et al.	2008					1							1	1			Network Analysis
Carvell et al.	2007	1							1								Statistical sampling
Chapagain et	2006					1										1	Ecological

al.																	Footprint
Cheng et al.	2011		1													1	Ecological Footprint
Concepcion et al.	2008	1		1					1								Statistical sampling
Critchley et al.	2003	1							1								Statistical sampling
Davros et al.	2006	1							1								Statistical sampling
Deumlich et al.	2006				1					1							Spatial analysis
Doole et al.	2011					1		1			1	1					Agent-Based Modelling
Doole et al.	2011				1	1		1				1			1		Integrated model
Ekholm et al.	2007					1		1		1							Biophysical and Spatial model
Ewert et al.	2009					1		1		1				1	1		Integrated model / SEAMLESS
Feehan et al.	2005	1							1								Statistical sampling
Flohre et al.	2011	1							1								Statistical sampling
Fuentes-Montemayor et al.	2011	1							1								Statistical sampling

Gabriel et al.	2010	1		1					1								Statistical sampling
Girardin et al.	2000						1	1					1				Sustainability Indicators
Granlund et al.	2005					1			1								Statistical sampling
Hanspasch et al.	2011	1							1								Statistical sampling
Henseler et al.	2009		1					1				1			1		Socioeconomic
Hérivaux et al.	2013					1		1		1					1		Integrated model
Hill et al.	2005				1					1			1				Spatial analysis
Janssen et al.	2009						1	1		1		1		1	1		Integrated model / SEAMLESS
Kersebaum et al.	2006					1		1		1							Biophysical and Spatial model
Kleijn et al.	2004	1							1								Statistical sampling
Kleijn et al.	2006	1							1								Statistical sampling
Kleijn et al.	2009	1							1								Statistical sampling
Knop et al.	2006	1							1								Statistical sampling
Kohler et al.	2007	1							1								Statistical

																	sampling
Le Gal et al.	2010					1		1				1			1		Integrated model
Lefevbre et al.	2012			1											1		Socioeconomic
Louwagie et al.	2012						1	1					1				Sustainability Indicators
Lutz and Felici	2009						1							1			Sustainability Indicators
Marini et al.	2011	1		1					1						1		Statistical sampling
Marriott et al.	2006	1			1				1								Statistical sampling
Marshall et al.	2006	1							1								Statistical sampling
Melman et al.	2008	1								1							Spatial analysis
Merckx et al.	2007	1							1								Statistical sampling
Neumann et al.	2008						1			1							Spatial analysis
Norton et al.	2009	1		1					1								Statistical sampling
Olsson et al.	2009						1	1		1		1			1		Integrated model / SEAMLESS
Pacini et al.	2003						1	1							1		Integrated model

Piorr	2003						1	1					1		1		Sustainability Indicators
Purvis et al.	2009						1	1					1				Sustainability Indicators
Reino et al.	2010	1							1								Statistical sampling
Reynolds et al.	2006	1							1	1							Statistical sampling
Roth et al.	2008	1							1								Statistical sampling
Rundlof	2010	1		1					1	1							Statistical sampling
Sal and Garcìa	2007						1						1		1		Socioeconomic
Savard and Bohman	2003				1	1		1		1		1			1		Integrated model
Schuler and Sattler	2010				1			1				1			1		Integrated model
Shields et al	2006					1		1									Biophysical
Tzanopoulos et al.	2012			1				1		1							Biophysical and Spatial model
Uthes et al.	2010					1		1		1		1	1		1		Integrated model
Valbuena et al.	2010			1							1						Agent-Based Modelling
Van Delden et	2010			1				1							1		Integrated model

al.																	
van der Horst	2007	1		1						1					1		Socioeconomic
van Ittersum et al.	2008					1		1		1		1			1		Integrated model / SEAMLESS
van Passel and Meul	2012						1								1		Integrated model
Vlahos and Louloudis	2011			1						1					1		Spatial analysis
Walker et al.	2007	1								1					1		Spatial analysis
Wei et al.,	2009		1			1		1		1		1			1		Integrated model
Yuan et al.	2011			1	1	1		1		1							Biophysical and Spatial model
Zander and Hachele	1999						1					1	1		1		Socioeconomic
Zhang et al.	2011					1										1	Ecological Footprint
Zollinger et al.	2013	1							1								Statistical sampling