VALUING ENVIRONMENTAL DECAY

Quantitative Policy-oriented Studies on Urban and Rural Environments

Printed in Italy
January 2007
Cover photo and design by Claudio Travisi

VRIJE UNIVERSITEIT

Valuing Environmental Decay
Quantitative Policy-oriented Studies on Urban and Rural Environments

ACADEMISCH PROEFSCHRIFT

ter verkrijging van de graad Doctor aan
de Vrije Universiteit Amsterdam,
op gezag van de rector magnificus
prof.dr. L.M. Bouter,
in het openbaar te verdedigen
ten overstaan van de promotiecommissie
van de faculteit der Economische Wetenschappen en Bedrijfskunde
op donderdag 15 maart 2007 om 10.45 uur
in de aula van de universiteit,
De Boelelaan 1105

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Yogāṅgānusthānāt asuddhiksaye jñānadīptih āvivekakhyateh
By dedicated practice of the various aspects of yoga
impurities are destroyed: the crown of wisdom radiates in glory.

Patañjala Yoga Pradīpikā, II.28

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PREFACE

These last years have been years of hard work and intense gratification. I've been constructing a building of intellectual challenges on a topic that I really care about: the interaction between humans and natural ecosystems. Starting to work hard on a number of research questions, I ended up with an avalanche of issues that would deserve both a factual and a theoretical answer. Some of these questions can be easily confined within the research field, some others necessarily invade the personal ground too. So, I have matured the conviction that any process of learning is an enormous task, which generates a myriad of intellectual challenges that, at some point in time, will hopefully collapse into a unique overall question. Of course, I am far behind this point, but I am confident that it exists.

My sincere thanks go to Prof. Peter Nijkamp, for his priceless intellectual stimulus and example, as a man who produces a huge, constant, and devoted academic effort. My appreciation also goes to Prof. Raymond Florax for his example in research rigour and analysis, and to Prof. Roberto Camagni, for his unique creative approach to research.

Finally, I am indebted to my family for initiating me into the art of discernment.

Milan, 21 January 2007

PART I: BACKGROUND

1. MOTIVATION AND ORGANISATION OF THE STUDY

1.1. Valuing environmental decay

Mankind is part of nature and, since the "beginning", it has been utilising the bounty of natural ecosystems in order to survive and flourish. Ever since, any choice we make regarding how to utilise natural resources and ecosystems has repercussions for their maintenance and for the sustainability of the services that they provide to humans and, therefore, for our quality of life standards. We rely on natural ecosystems for an uncountable list of essential aspects of our quality of life, starting from breathing, drinking and eating. We are accustomed to deliberately modifying natural resources and services and reconvert them into man-made ones, but such resources are in large part limited and, in many cases, progressively scarcer or severely damaged. Increasingly, therefore, we are faced with choices of trading some risk and cost to humans or ecosystems in order to at least maintain the status quo in our standard of living or, more often, to gain some extra benefit. A number of significant and still unsolved illustrations of such a perverse mechanism can be found at the crossroads of the conflicting needs for spatial development and spatial well-being in cities and agricultural systems, where the numerous social and economic advantages generated by, respectively, urban agglomeration and proximities, and by rural development and agricultural production, are accompanied by collective diseconomies and multidimensional environmental negative externalities. Air pollution induced by urban mobility, increased traffic noise levels in cities, or water and soil contamination related to the use of chemicals in agriculture are just a few examples of the number of negative externalities which contribute to impoverish the quality of environmental and human health.

In such contexts (but one might generalise the discussion to many others), what we observe is that, in the majority of cases, human pressure on ecosystems and natural resources is the result of a systematic, deliberate and continuous pursuit of wealth by business, individuals and communities, rather than the effect of random events. It is therefore desirable that the problem of how to deal with the drawbacks of economic development should also be approached with systematic, deliberate, and continuous strategies at different administrative and spatial levels. This is the reason why the concept of "sustainable development" defined by the Bruntland report in 1987[1] is steadily gaining recognition, if not having the status of a fully autonomous discipline, and becoming the focus of additional theoretical, normative and empirical reflection. During recent decades, there has been an ongoing discussion on the definitions, and dimensions of sustainability and

[1] The Brundtland Report of the World Commission on Environment and Development defined sustainable development as a "process of change in which the exploitation of resources, the direction of investments, the orientation of technological development and institutional changes are made consistent with future as well as present needs".

sustainable development and an avalanche of literature has been published on this (e.g. Nijkamp and Perrels, 1998; Camagni et al., 1998; Selman, 1996; van den Bergh, 1996). Many discussions have addressed sustainability with a rather global and general perspective, and they have mainly focused on conceptual and epistemology issues. Starting from the 1990s, a more operational approach to the rather vague concept of sustainability has been advocated at international forums, with the intent of overcoming the lack of empirical application of this concept in a concrete policy context (Rio de Janeiro, 1992; Kyoto, 1997). It has gradually become clear, however, that a blueprint for sustainable economic behaviour cannot be given, and that a global operational approach to sustainability is scarcely feasible as it can be fraught with excessive ambiguities. It is evident that the issue to be addressed – i.e. the adverse effects of human-driven environmental degradation on societies and the environment – is significant but difficult to capture and quantify as a whole phenomenon. Interactions between natural or artefact ecosystems, societies and economies – both physical and non-physical – are complex and represent an essential and dominant feature of the modern world.

Complexity in environmental well-being analysis concerns different levels. Firstly, it is manifest in the variety of causes that, in a complex network economy, drive environmental impoverishment and its negative effects on ecosystems and their communities. Secondly, complexity reflects the range of ecological and human populations exposed to risk of damage, as risk targets, and the involved stakeholders can vary substantially depending on the specific environmental issue concerned. Thirdly, it concerns spatial and time scales, since the relevant territorial levels and time horizon differ according to policy objectives. Fourth, it concerns personal, societal and decision makers' preferences. Finally, and as a consequence of what was stated before, it emphasises the need for multiple, complementary and specific policy actions for managing and reducing risks to their minimum.

Therefore, in searching for a framework in which to value environmental decay (or better still its renaissance, to be more optimistic), a number of issues need to be taken into consideration. Identifying the specific phenomenon driving the risks to ecosystems or humans is a precondition, which also allows the proper setting of the spatial and time scale to be considered for designing an environmental valuation approach. As already mentioned, the spatial and time scale of the required sustainability matters, since the smaller the scale the higher the degrees of freedom, i.e. more choices are available at a regional meso-local level than at a global one. So, fragmentation and adopting an analytical local perspective in the assessment of the relevant specific components that contribute to (or affect) environmental quality is crucial to isolate the role of any particular factor at stake and provide solutions suited to each given context. Additionally, personal and societal preferences matter, since what is considered sustainable and safe is to a large extent subject to individual preferences and behavioural choices with respect to environmental quality, the assessment of future technological possibilities, and the attitude towards risks and uncertainty. Risks and impacts to ecosystems actually affect people's well-being and their quality of life, thus conditioning individual behaviour and preferences. Similarly, policy aims and decision-makers' preferences matter, because they set in place reference systems for valuing environmental decay and appraising the options to manage it.

Though broad-spectrum, the impressions described above introduce some issues in the valuation of the environment that will be taken as reference points throughout the remainder of this study. First, the efforts of researchers, both theoretical and methodological, should primarily be directed to the meso-local level of analysis, where environmental phenomena can be clearly identified and the behaviour of complex socio-economic systems can be investigated in greater detail and with lower uncertainty. Next, valuation approaches need to take into consideration complexity in all its facets concerning: ecological and human targets; time and spatial scale; the public's and policy makers' preferences. Additionally, and related to the previous point, any environmental issue implies changes in individual and collective well-being that should be monitored and, whenever possible, measured.

We now therefore proceed to clarify the point that any environmental issue implies one or more decision problems, and that motivations for the economic valuation of environmental well-being strictly depend on the need to provide decision makers with sound scientific knowledge on the pros and cons of development.

1.2. Environmental decision making and the need for quantitative policy-oriented research

The previous discussion helps us to clarify the perspective that is adopted in this book, while dealing with the problem of setting an economic value for the ongoing environmental impoverishment. Decision makers need to manage the effects of economic development – and may need to act – to minimise negative impacts and maximise any beneficial opportunity. At present, however, there is still a lack of reliable information on the societal costs and benefits of changes in environmental quality, which makes it hard for decision makers to judge the amount of resources that they should allocate to manage risks and impacts in any given case. Moreover, for any given policy strategy there is likely to be a number of options that could be pursued to meet the overall policy aim (i.e. the decision criteria). Decision makers need to be informed in order to optimise policies or projectual measures and to adopt a scientific perspective for developing a solution that can lead to higher collective utility.

The successful implementation of these new orientations can be facilitated by support through strategic policy-oriented quantitative research. The role of economics when applied to the environment is multiple (see Figure 1-1). It can act:

- as a tool to analyse the nature and the origins of the risk to ecosystems and humans due to economic development;
- as a tool to provide a measurement of the costs and benefits involved, providing a quantitative estimation of the risks to ecosystems and humans (i.e. the physical dimension);
- as a means of designing options to contain or manage risk, thus offering a range of possible solutions;
- as a systematic and structured approach to appraising the risks and the options designed to manage them on the basis of the costing methodology.

Depending on the issue at stake, research efforts might be oriented in different directions as they might draw from different methodological approaches available to value the environment: for instance, conventional market-based costing-techniques, or non-market valuation methods (revealed and stated preference methods). Similarly, appraisal techniques for alternative options are likely to be employed (e.g. cost-benefit analysis, cost-effectiveness analysis, multicriteria analysis). In principle, it is necessary to look for an optimal context-specific framework of environmental valuation supported with substantive and causal scientific investigation. Optimal analysis needs to be strategic in nature and should be supported by a proper – preferably quantitative – methodology for the systematic evaluation of the analytical knowledge available. In this perspective what is evident, however, is that there is still a need for more context-specific research, which entails considerable effort. More empirical primary studies on the valuation of environmental decay are needed to: i) allow methodological innovation based on experimentation; and ii) produce a sound body of knowledge to be used for research synthesis.

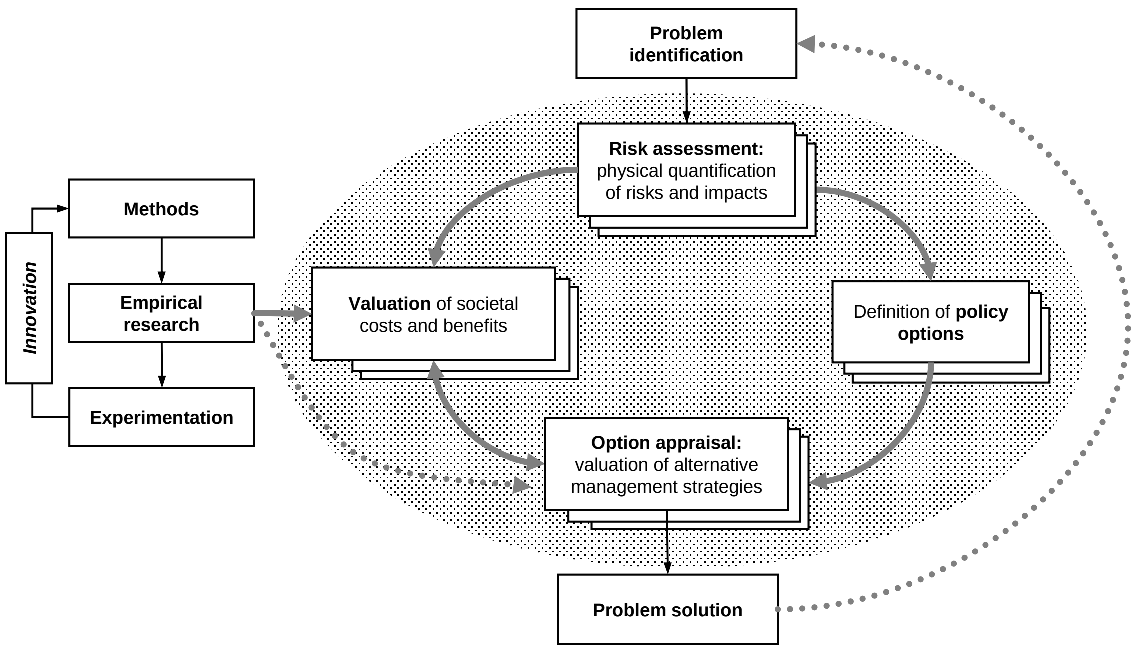

Figure 1-1: The role of economic valuation in sustainability decision making

Given the previous discussion, this study analyses the external welfare-economic effects of economic development, in terms of environmental disruption, in an analytical, policy-oriented perspective. The general research purpose is to contribute to the empirical analysis of context-specific environmental issues with the use of frontier methods of environmental valuation, selected case by case to pursue a rigorous and innovative support and to answer specific research and policy questions.

In doing this, this study has one major merit. It elevates valuation from the status of a mere set of techniques to the status of process of analysis. Hence, by proving a thorough assessment framework, in which several quantitative techniques of environmental valuation can be alternatively applied, this study gives a comprehensive picture of the major virtues, obstacles, and challenges of the economic valuation of environmental decay.

By filling the gaps in research, the studies described in this publication address a number of open policy issues in the context of urban and rural environments. Here, the remaining uncertainty needs to be tackled by a further fragmentation and analysis of the relevant components of each phenomenon, with due attention to contextual factors and improvements in methodologies for an economic quantitative analysis of environmental effects. The research challenges addressed in this dissertation emerge from the policy arena and are intended to provide support to relevant policy issues such as: Which urban and agriculture-induced risks should be prioritised? Which management or mitigation options should be chosen to respond to these risks? And how far it is necessary to go in, for instance, reducing private car or pesticide use? Is action preferable to the do-nothing scenario?

The methodological contribution of this book is based on the paradigm of cost-benefit analysis, which states that, in a world of scarce resources, rational action requires a consideration of relative benefits and societal costs; and that governments need to take some account of public preferences in their decision making. However, researchers face big challenges in evaluating possible responses to the impacts of urban and agricultural development. These include the long timescale of impacts, serious uncertainty over environmental change and human reaction to this change, uncertainty over the effectiveness of management strategies, and the very wide range of impacts that the continuous process of economic development may have.

Taken together, these challenges mean that often it will only be possible to produce order-of-magnitude estimates of cost-benefit ratios. Nevertheless, even this rough advice is likely to be better than no advice. Moreover, it is also desirable that a methodology be adopted which allows for the adjustment of the scale of the analysis to suit the particular problem being addressed. This means that it is often necessary to turn to research synthesis and value transfer type exercises in order to provide efficient and timely advice for policy makers. Similarly, in some cases it is advantageous to look outside the bunch of conventional methods for economic analysis of environmental costs – say, stated and revealed preferences methods – and draw inspiration from the principles and tools of other disciplines, such as ecology and risk assessment, respectively.

The studies developed in this dissertation show how cost-benefit thinking can be used to enhance decision making with respect to risks due to mobility and agriculture, as well as pointing to the contribution that other methods, such as multicriteria analysis and the use of ecological and environmental indicators can make. A characteristic feature of the thesis is the presentation of worked examples of applying a number of well-framed methodologies (stated preference methods, meta-analysis, multicriteria analysis, risk assessment) to issues as diverse as noise pollution, transport disruptions, pesticide ecological risks, and food safety. Aimed at contributing to the previously discussed research challenges, it provides a wide range of empirical research and experiments with some methodological innovations that will hopefully be seen as a valuable contribution to research and policy development. Empirical case studies refer to Italy where, given a relative delay in the use of environmental valuation approaches for costing ecosystems and human health risks, the interest in economic valuation is rapidly increasing.

The overall structure of the book is described in Figure 1-4, while the contents are summarised in Table 1-1. Part I provides the necessary information and theoretical and methodological underpinnings for a thorough understanding of the policy problems addressed in the empirical parts, Parts II and III. Mobility and agriculture are here seen as two specific and representative phenomena of environmental degradation by which to move foreword the discussion on the economic valuation of environment and human health. Regarding mobility, Part II of this dissertation focuses first on the recently-debated issue of noise pollution from railway transport infrastructures, and, secondly, on the diseconomies of cities related to the collective impacts of mobility as a result of the phenomenon of urban sprawl. Regarding agriculture, Part III of this book tackles the problem of the detrimental effects of pesticide use and accumulation in rural environments for non-target ecosystems, as well as its implications in terms of human health risk.

Sections 1.3 and 1.4 now give a brief introduction to the challenging topics addressed in this study. Section 1.5 details the basic research questions and provides an outline of the thesis.

1.3. Challenges in valuing mobility-related environmental externalities

1.3.1. Setting the scene

Among the major determinants of the quality of life in urban areas, the transport sector and its environmental implications play a very important role in modern cities. Transport allows personal mobility for both work and leisure activities, it assures important connections and networks across cities, and provides a vital lubricant for trade and geographical specialisation in production. Such strong advantages in terms of development and collective benefits are, however, also accompanied with pressing environmental problems, which are rapidly becoming a priority for national and European institutions. In this sense the title of the 2001 White Paper on European Transport Policy, “European transport policy for 2010: time to decide”, speaks for itself. It envisages the need for a broader strategy to attenuate the number of undesired negative side effects of transport and reach a higher level of sustainability in this sector (CEC, 2001). Additionally, according to the Thematic Strategy on Urban Environment (CEC, 2004, p. 19), under development at the EU Commission, which embraces the precautionary principle in the vision for sustainable urban management (Annex 2, CEC, 2004), Member States will be encouraged to set out a clear framework policy on urban transport, and to evaluate the impacts and related costs of new urban transport infrastructure projects on the sustainability of a city’s transport system.

Overall, the environmental impacts of transport comprise a complex system of effects, either direct or indirect, which can assume different connotations according to the transport mode under analysis. Major components of transport can be identified in road, aircraft and railway transport, either private or freight. Figure 1-2 classifies the external costs of mobility, first, according to the travel mode: namely, road, railway and aircraft; and, secondly, according to the environmental dimension affected. Impacts can be further classified with regard to whether effects arise from actual transport or from the existence of infrastructures and vehicle production and disposal (modified from Verhoef, 1996). Overall, the environmental external impacts can be divided into direct and indirect effects on human health and the local communities, and direct effects on the various environmental dimensions: air, soil, surface and groundwater. Noise nuisance can be considered a form of pollution which affects both the ecological and the social environment. The impacts and risks for human health can be directly associated with transport, as for the case of injuries incurred during accidents[2], or are the results of environmental pollution due to transport: air, water, soil and noise pollution. In this case, one can distinguish between short-term and long-term risks for human health. Health risks mainly originate from road transport and, to a lesser extent, from air transport. Rail transport is generally considered the most environmentally-friendly transport mode, though, nowadays, the problem of rail noise is receiving ever-increasing attention. Additional impacts on local communities are traffic and parking congestion in urban areas, and the problem of the loss and deterioration of landscapes due to transport infrastructures.

In the light of these externalities, the analysis of the negative side effects of transport and mobility in urban systems will be one of the major challenges for research in this field in the coming years, and this study aims to offer a sound contribution to improving knowledge of the collective costs of transport in urbanised areas. In the following, a brief overview of the major drawbacks of urban mobility is given, paying attention to the issues that will be addressed – with special topics and empirical analyses – in Part II of this dissertation.

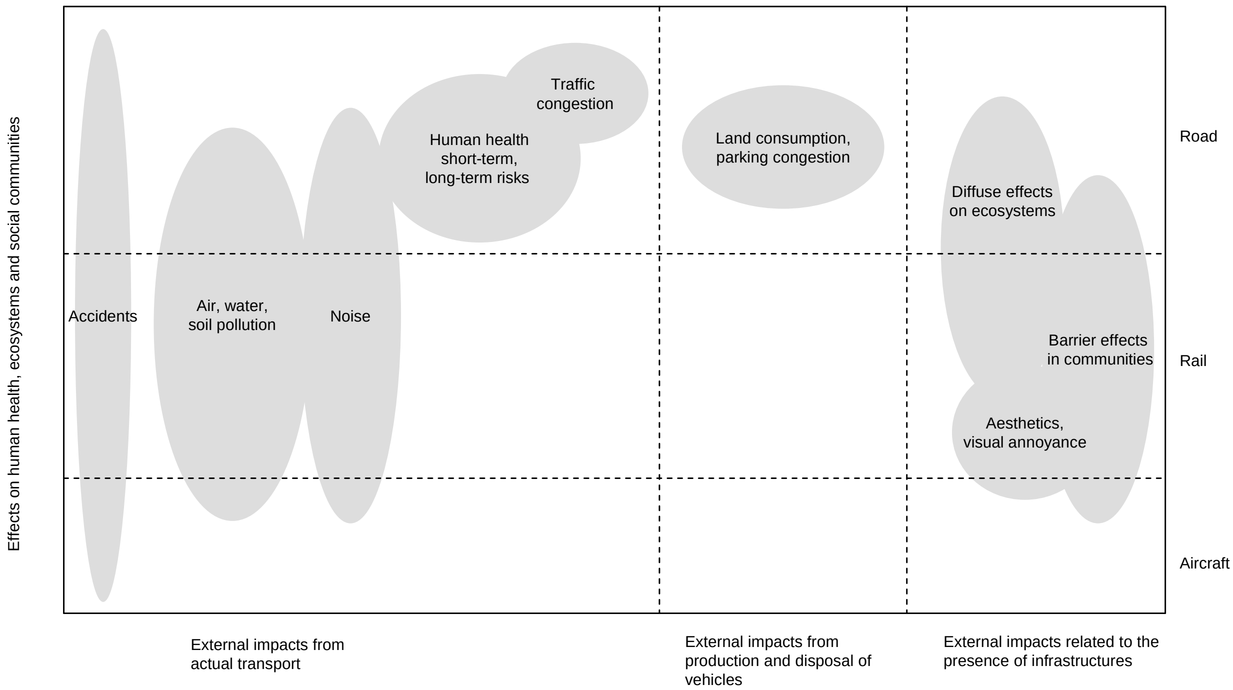

Figure 1-2: A schematic overview of the system of traffic impacts on the environment and health in urban contexts

Note: Modified from Verhoef, 1996.

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[2] Road safety also represents a relevant problem, pertaining to both cities and peripheral areas, as the number of injuries and deaths due to traffic accidents remain unacceptably high. Statistics for the year 2000 indicate that about two-thirds of the 1.3 million traffic accidents in the EU which resulted in injuries took place within urban areas, and one fatal accident in two (INFRAS IWW, 2000).

1.3.2. Valuing noise pollution

Among the range of environmental negative side effects of urban mobility, this book focuses on noise pollution, one of the impacts of urban mobility that has recently gained the attention of both researchers and policy makers. Statistics indicate that 80 percent of noise in cities comes from road traffic (e.g. CEC, 2003). At least 100 million people in European towns and cities, or in the proximity of transport infrastructures are exposed to road traffic noise which exceeds the World Health Organization recommended level of 55 Ldn dB(A)[3]. Such conditions cause serious annoyance and are responsible for negative effects on sleep and the quality of life. Together with noise caused by road traffic, however, excessive noise due to rail transport is also now receiving more attention, especially in those circumstances in which rail infrastructures pass in the vicinity of, or even cross, cities.

The European Commission Green Paper (CEC, 1996) states that: "More attention needs to be paid to rail noise [..] where there is considerable opposition to the expansion of rail capacity due to excessive noise." According to the European Environmental Agency estimates for 2001, 10 percent of the European population are affected by train noise levels over the standard WHO safety level. With special focus on railway noise, a recent document by the European Commission "Position Paper on the European Strategies and Priorities for Railways Noise Abatement" (CEC, 2003) underlines that, in order to protect the current population exposed to rail noise pollution, it will be necessary, on average, to reach a noise reduction by 10-15 dB(A).

Therefore, railway noise abatement has become an important priority in the European environmental policy agenda, and there is a high potential for the reduction of railway noise because the technical instruments for the abatement are available (CEC, 2003). Nevertheless, since noise reduction measures are very costly, in the current situation a major issue is the economically-viable implementation of such expensive noise abatement measures and, hence, the choice of the most cost-effective type of possible interventions. It is therefore of great importance that both the EU and the national policy makers should be informed about the social benefits of reduced rail noise exposure, and about the advantages and disadvantages of the various technical measures available for mitigating noise.

Environmental valuation methods can be employed to estimate the economic value of changes in noise levels and, therefore, to provide decision support for managers and national authorities charged with planning and implementing noise abatement measures (Chapter 2). Revealed Preference (RP) methods have been used extensively to estimate the economic value of reductions and increases in noise emissions (see Navrud, 2002). Nevertheless, so far, the literature on noise has been dominated by the use of RP methods (in particular, Hedonic Price (HP) techniques) and it has mainly focused on the valuation of traffic noise, disregarding the valuation of rail noise.

By filling these gaps in research, after first reviewing the principles of noise valuation in Chapter 3, the present study examines the use of a Choice

[3] Ldn = day/night level over the whole day with a 10 dB(A) penalty for night-time noise, from 22:00 hrs to 7:00 hrs.

Experiment (CE) methodology to assess the economic value of alternative rail noise reduction policy interventions and the respective instruments. In fact, Chapter 4 presents an original piece of work in which, to our knowledge, CE techniques are used for the first time to assess the value of alternative rail noise mitigation plans in Italy. A further strong innovative point of the study is that it employs and tests the econometric robustness of two alternative payment vehicles: namely, a conventional tax scheme, and a new tax-reallocation scheme (proposed by Bergstrom et al., 2004).

1.3.3. Valuing the environmental costs of urban mobility due to sprawl

Urban mobility has significant impacts on the environment and on the health of citizens, as well as on the overall quality of life in towns[4]. At the same time, rising congestion levels are hampering mobility, with increasing costs for the economy and for the communities. These are estimated at 0.5 percent of the Community GDP for road traffic congestion, and are expected to rise to 1 percent by 2010 (CEC, 2001).

In the last decade, the modal composition of traffic in cities has changed dramatically toward less environmentally-friendly, i.e. polluting and energy-consuming, travel modes. The current high level of motorised urban transport, besides worsening air quality, contributes to the increase of sedentary life, with a variety of negative effects on health and life expectancy, noticeably in relation to increased risks of cardiovascular disease[5]. Additionally, the increased used of private over public transport modes, contributes to traffic as well as parking congestion in cities, with drawbacks in intra-sectoral exchanges and increasing loss of public space. Related to this, increased private motorised mobility in urban areas is driving their development by facilitating the expansion of cities into the surrounding rural areas, the phenomenon known as "urban sprawl". Recently, the European Commission has recognized urban sprawl as the most urgent urban design issue as it so often leads to loss of rural and green space, high infrastructure and energy costs, and increased social segregation and functional division in the city (CEC, 2004). In this respect, some commentators have underlined the fact that just as inadequate land-use decision can generate increased traffic, increased traffic can encourage poor land use decisions in response to demands to reduce congestion (CEC, 2004; Camagni et al., 2002a).

There have been many recent publications which quantify collective costs due to diffuse and sprawling patterns of urban expansion over time and space, and in order to achieve a comprehensive assessment of the determinants of urban

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[4] The emission of air pollutants is one of the main problems directly associated with traffic in cities. Nearly all of the European citizens (97 percent) are exposed to air pollution levels that exceed EU quality objectives for particulates, 44 percent to ground level ozone and 14 percent to NO_2. A number of studies suggest that the consequences for the health of urban citizens are considerable, and they range from the increased occurrence of bronchitis and asthma attacks, to increased hospitalisation, morbidity, and mortality (WHO, 1999). The related economic costs of traffic-related air pollution are considerable and they amount to 1.7 percent of GDP (WHO, 1999). In Italy, a recent project called MISA (a meta-analysis of the Italian epidemiological surveys on the short-term health effects of air pollution based on 1990-1999 data) concludes that a statistical correlation between air pollution and increased daily mortality and morbidity does exist.

[5] Cycling for 30 minutes per day can reduce the risk of cardiovascular disease by as much as a half. Yet, more than half of the trips below 5 km are made by car (CEC, 2001).

sprawl, thus being able to devise proper management actions and provide feedback for future urban planning policies. Research, exchange of experience, and the promotion of best practice in urban land issues is therefore of particular importance and highly recommended to attain insights for policy actions. In particular, a starting point should be the identification of the actual state of sprawl-driven negative externalities in towns and cities, and their costs. Next, the identification of the cause-effects relationships that have over time favoured the phenomenon of urban sprawl should follow. We are in fact facing a phenomenon that is driven by a number of heterogeneous components – historical, cultural, social, economic, and structural – which interact in the space playing different roles depending on local conditions.

The possibility to monitor the mobility impacts generated by urban sprawl over time and space, as well as to make clearer what are its major determinants is, therefore, an important prerequisite to prepare a solid background for the definition of effective national, regional and/or local urban environment strategies. This is particularly true for Europe because it presents a very much scattered puzzle of territorial conditions, which vary from country to country, region to region, and even city to city.

Major research challenges can be summarised as follows:

- to qualify and quantify the collective costs imputable to diffuse and scattered patterns of urban development over time and space, with the intent of drawing attention to contingent trends and tendencies, as well as to the likenesses among different cities from which to share experiences;
- to achieve a comprehensive assessment of the determinants of urban sprawl with the aim of determining priority requirements and, thus, a ranking of priority management actions;
- to analyse the effects of past urban planning policies to enhance feedback processes and the definition of good-practice for sustainable urban planning.

The focus of the present thesis is on the first two points, on the connections between sprawl and the impacts of urban mobility. With respect to this, sound empirical and quantitative results on the collective costs of sprawl are still only partial. It is indeed not straightforward to measure the environmental externalities related to the phenomenon of sprawl, especially due to the difficulties of finding sound and reliable performance indicators. Even more challenging is the analysis of the determinants of urban sprawl. By filling the gaps in research, in Chapter 5 an original quantitative analysis of intensity and dynamics of the impact of mobility across seven Italian urban areas is provided. The study employs a mobility impact index based on commuting data and uses multivariate regression analyses to capture heterogeneities across different cities according to sprawl, structural, and transport factors. Causal Path Analysis is then applied to test the causal relationships between the impact of urban mobility and the aforementioned explanatory factors.

1.4. Challenges in valuing agricultural environmental pressure

1.4.1. Setting the scene

Rural areas will face particular challenges as regards growth, jobs and sustainability in the coming years (CEC, 2005). They offer opportunities in terms of their potential for growth in new sectors, the provision of rural amenities and tourism, their attractiveness as a place to live and work, and their role as groundwater reservoirs and highly valued landscapes. On the other hand, with nearly 90 percent of land use in the European Union determined by agriculture and forestry, the relevance of rural areas for sustainability can hardly be underestimated.

Over the years, farming has contributed to creating and maintaining a variety of valuable semi-natural habitats and today still shapes the majority of the EU's landscapes. Nonetheless, the links between the natural environment and agricultural and farming practices and production are complex. Although, many valuable semi-natural habitats are preserved by extensive practice, and a number of wild species can survive and flourish in these areas, it is well known that agricultural practices can have detrimental effects on natural resources and human health. In addition, many of the farming systems with high ecological value, which are beneficial to the environment, are economically marginal and located in less favoured areas which are striving to overcome external factors that limit productivity. Throughout agricultural production, processes occur that can have an impact on the natural and semi-natural ecosystems and human health (see Figure 1-3). Above all, the heavy use of agrochemicals, fertilisers and pesticides, incorrect drainage or irrigation practices, a high level of mechanisation or unsuitable land use can produce environmental degradation. Adverse effects on the quality of terrestrial and aquatic ecosystems are observed, together with fragmentation of habitats and loss of biodiversity (e.g. Finizio, 1999a, b). Similarly, human health can be impacted by unsuitable agricultural practices. The effects of agriculture on human health comprise potential risks for producers, mainly due to direct exposure to agrochemicals or GMOs (Genetically Modified Organisms) in fields, as well as indirect effects on consumers via the provision of foodstuffs, with implications in terms of food safety (e.g. Wilson, 2002; Sivayoganathan et al., 2000).

As for the case of mobility, the complexity of agriculture in terms of the interdependencies among the economy, local communities and the agro-ecosystems suggests the need to adopt of a well-framed approach to sustainable rural development, preferably implemented at a local level and, also, well-targeted to the aim. The reformed CAP (Community Agricultural Policy) and rural development regulation confirms and stresses what was stated in Göteborg 2001 and in the Lisbon Strategy Conclusions in June 2003, that: "strong economic performance must go hand in hand with the sustainable use of natural resources and level of waste, maintaining biodiversity, preserving ecosystems and avoiding desertification". To meet these challenges, CAP and its future development indicates, among its objectives, the need to contribute to achieving sustainable development by increasing its emphasis on encouraging healthy, high quality products, environmentally-sustainable production methods, including organic production, renewable raw materials, and the protection of biodiversity. The future challenges are an increased focus on investments in people, know-how, and capital in the farm sector, new ways of delivering win-win environmental services, and creating more and better jobs through diversification.

Preliminary to implementing these objectives, the new rural development regulation foresees the strategic monitoring of the Community and national strategies to allow the assessment of the starting situation, and to form the basis for the development of the programme strategy (see CEC, 2005). Evaluation activities will take place on an ongoing basis, comprising *ex ante*, mid-term and *ex post* evaluation as well as other evaluation activities useful for improving the programme management. In addition, thematic studies, exchange of good practice, and sharing of evaluation results are strongly recommended to contribute significantly to the effectiveness of rural policies. Scientific support for rural policies, with regard to the increased scope, diversification and value added of agricultural products and services, requires analyses that are sufficiently interdisciplinary and quantitative to provide useful input into the new programme period. Many challenges remain open and, among those recently identified in several European countries as calling for policy-oriented research, some are specifically directed to advancing the quantification of the social costs and benefits of the agricultural sector, as well as providing elements for supporting more environmentally-friendly agricultural practices (see CEC, 2005).

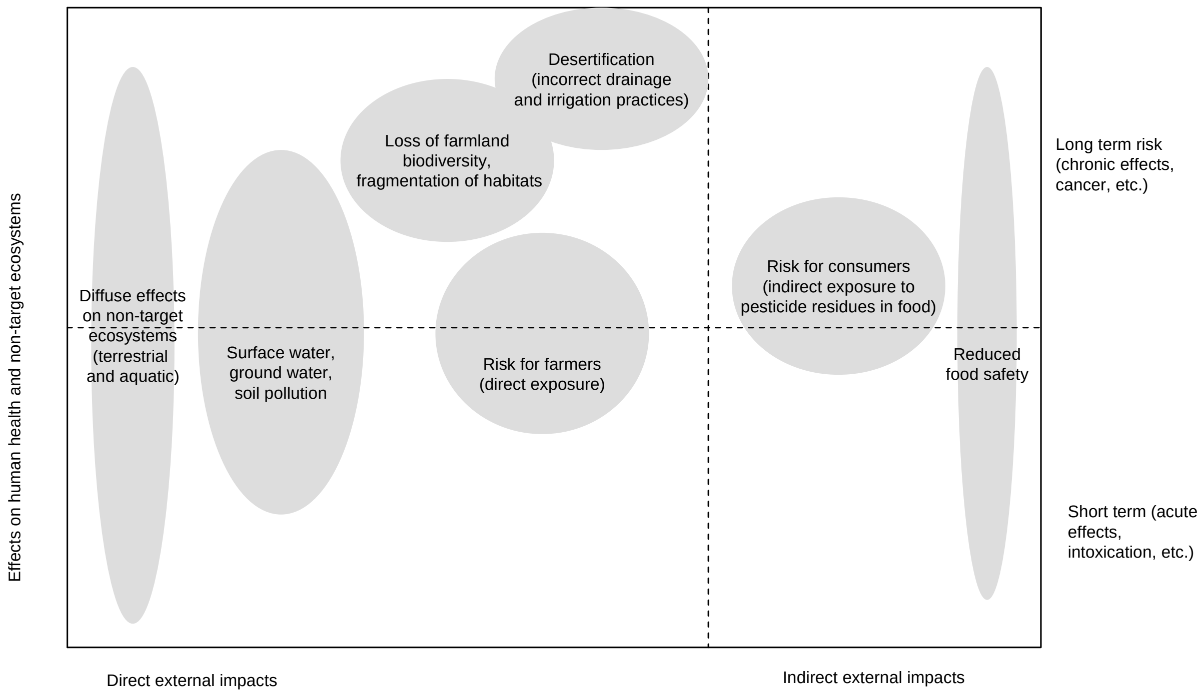

Figure 1-3: A schematic overview of the system of agricultural impacts on ecosystems and human health

It follows that the analysis and quantification of the costs of ecological and human health risks due to agricultural production will be one of the major challenges for research in this field in the coming years. Among the wide range of relevant issues, this book focuses in particular on the environmental drawbacks of agricultural production related to the use of pesticides. With respect to this, the following subsection provides a brief discussion on the motivation for the economic analysis of pesticides risks, paying attention to the issues that will be addressed – with special focus and empirical analyses – in Part III of this dissertation, dedicated to valuing pesticide damage and risk to ecosystems and human health.

1.4.2. Valuing pesticide risks

The negative side-effects of pesticide use are multidimensional, and managing such risks implies trade-offs between stocks at risk to be protected (i.e. risk targets), as well as between different types of potential impacts. The available empirical evidence from medical, toxicological and ecotoxicological studies documents the prevalence of non-negligible hazards to human health and to the quality of aquatic and terrestrial ecosystems. Pesticides can, for instance, contaminate drinking water and food crops, and high-dosage pesticide usage in the production of fruits and vegetables can potentially induce serious health hazards in consumers (Pimentel et al., 1992). The poisoning of farmers due to field exposure to pesticides occurs frequently, especially in developing countries (Sivayoganathan et al., 2000). Pesticides are among the most frequently-detected chemicals in water, particularly in groundwater (Funari et al., 1995), and pesticide usage affects the quality and quantity of the flora (Pimentel and Greiner, 1997), mammalian species (Mason et al., 1986), insects (Murray, 1985), and birds (Luhdholm, 1987).

Furthermore, insights into the intricate cause-effect relationships are necessary to model the phenomenon and to predict its temporal and spatial dynamics. In such situations where risks are multidimensional and trade-offs between them are particularly subtle, and where information on causes and mechanisms is incomplete or uncertain, the trade-offs between risks and benefits should be made explicit and expressed in a way that allows direct comparisons. In addition, the consumers' awareness for food safety and the social preference to improve the environmental sustainability of agriculture are culminating in the design and application of new policy instruments. One such policy instrument is the eco-labelling of fresh produce (Govindasamy et al., 1998a; Blend and van Ravenswaay, 1999), but rules and regulations for the proper use of pesticides and (optimal) pesticide taxes have been designed as well (Swanson, 1998; Mourato et al., 2000). The availability of detailed and disaggregated monetary estimates of the individual's willingness to pay (WTP) for pesticide risk reductions is, however, pivotal for the successful implementation of such policies. In the case of eco-labelling, WTP information provides a basis for price differentiation according to the type and severity of pesticide risks involved in agricultural production. In the case of an ecological tax, for instance, economic theory shows that a Pigouvian tax requires the eco-tax to be set equal to the marginal value of the negative externalities associated with pesticide usage. Moreover, the multidimensionality of pesticide risks implies that potential trade-offs exist in correcting for different types of impacts. The relative importance of each pesticide risk, as measured by the individuals' WTP for reduced risk exposure, is therefore crucial in the price setting and tax-determining behaviour of producers and the government.

Nonetheless, so far, scientific knowledge on the social costs and benefits of pesticide application in agriculture is still partially lacking and no research has systematically analysed the overall range of effects associated with different farming operations on possible targets (for the operators, consumers, environment and ecosystems). Similarly, the availability of detailed monetary estimations of the corresponding environmental and human health impacts is fragmentary, and, when available, estimates mainly refer to context-specific study settings or are based on toxicological and ecotoxicological risk assessment procedures not validated through a comparison with actual damage to natural ecosystems and humans. It is therefore difficult to extrapolate the true external costs and benefits of pesticides and develop general criteria for a system of pesticides taxes or premiums (either in rural development measures or in eco-labelling), or to internalise ecosystems and human health risks in cost-benefit analysis.

In contributing to the advance in the valuation of pesticide costs, the present study provides original pieces of work centred on the valuation of pesticide risks for both ecosystems and human health. After reviewing, with a formal comparative approach, the principles and methods of pesticide risk valuation in Chapter 6, the first stated preference approach on pesticide risk valuation in Italy is presented in Chapter 7. Next, Chapter 8 discusses the results from a formal meta-analysis of the WTP for reduced pesticide risk exposure. To our knowledge, this is the first meta-regression analysis of WTPs for reducing pesticide risk that is available in the literature. Finally, an additional novel approach is presented in Chapter 9, which applies a set of eco-toxicological risk indicators for analysing possible short-term and long-term pesticide risk scenarios to terrestrial and aquatic ecosystems.

1.5. Objectives, research questions, and set-up of the thesis

1.5.1. Objectives

This thesis aims to contribute to the quantitative economic evaluation of environmental disruption, with analytical policy-oriented studies focused on the estimation of the external costs of urban mobility and agricultural production. As argued before in this connection, to enhance policy actions towards environmental risk management, two major active roles of economic valuation can be envisaged: i) as a tool to evaluate the current situation and future prospects of impacts and risks to the environment and human health, and to translate them into monetary terms; ii) as a structured approach for providing *ex ante*, strategic, and comparative appraisal of risks and the options for managing them. The first approach relies on the combined use of risk assessment and market or non-market valuation techniques (see Section 2.3). Alternatively, sustainability or impact/risk indicators can be employed if a monetary estimation of risks is not required, or if it is necessary to monitor and assess environmental conditions and their trends over time and space. Once risks have been assessed – and eventually monetised – risk values can be used as an input in order to provide a ranking of impacts, and an *ex ante* appraisal of alternative management options. In this latter case, Cost-Benefit Analysis (CBA), Cost-Effectiveness Analysis (CEA), and Multicriteria Analysis (MCA) are usually employed.

The potential of valuation for policy advice is explored here on the basis of original empirical research. Research challenges (which have been extensively discussed in the previous sections) emerge directly from the policy arena and are intended to provide support to relevant policy issues such as: How can improving urban planning reduce the collective cost of mobility? How can noise pollution from rail infrastructure be reduced and the most cost-effective noise abatement strategy be chosen? How can a system of taxes on pesticides responsible for severe threats to the environment be set up? Table 1-1 describes the environmental issues, policy issues, research challenges, and methods that are addressed, while Figure 1-4 provides an overall synthesis of the structure of this dissertation

Searching for quantitative policy-oriented answers to these important questions, the methodological contribution of this book is based on the paradigm of cost-benefit analysis, where researchers still face important challenges in assessing possible options to contain and manage the impacts for human health and the environment due to mobility and agriculture. The very wide range of possible threats, the uncertainty about risk assessment procedures, the variability of impact in terms of time and spatial-scale, and the uncertainty over the effectiveness of management strategies all contribute to the need for additional strategic policy-oriented research in this field. On the one hand, the studies presented in this thesis exemplify the role of cost-benefit thinking in enhancing decision making concerning responses to risks in urban and rural systems. Moving within these limits, the thesis focuses on stated choice approaches and, in particular, on the use of choice experiment techniques for the monetary valuation of multiple risks to ecosystems and humans, as well as for the valuation of alternative risk management policies. Nevertheless, on the other hand, the thesis recognizes the role of value transfer exercises to provide efficient and timely advice for policy makers and explores the use of formal meta-analysis for research synthesis. Additionally, the thesis points to the contribution that other approaches, for instance multicriteria analysis, and the principles and tools of other disciplines, such as risk assessment, can play. The dissertation, therefore, embraces the broader valuation paradigm of ecological economics and opens out to the insights and analytical techniques of some environmental science disciplines for particular environmental decision-making requirements. The focus is on the design and use of environmental-economic and environmental risk indicators to be applied within multi-attribute valuation frameworks.

Interestingly, the empirical studies presented in this thesis refer to two diverse but similarly relevant phenomena responsible for environmental impacts, so that the potential and flexibility of the aforementioned valuation approaches can be better observed. Here mobility and agriculture are the 'battlefields' for our empirical research and they raise a number of relevant policy issues.

As ecological quality and human health safety are mutually dependent, in this dissertation, environmental quality is to be interpreted in its widest meaning. It refers to the search for effective policy solutions both to enhance human health safety and to protect integrity of ecosystems[6]. Nonetheless, the studies presented address, explicitly, a number of different environmental issues and deal with their own peculiarities.

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[6] The perspective of this dissertation on these systems of effects is close to the EU perspective that human health but, more generally, the quality of life in urban or rural contexts can be improved via the environment, rather than with a focus on the protection of ecosystems per se (CEC, SEC(2004) 729).

The focus of the analytical studies presented is on Italy, where the interest in up-to-date valuation research is rapidly increasing. Italy went through a substantial change in its environmental management system in the early 1990s when the government set up a new structure of environmental institutions articulated in national, regional and sub-regional agencies. Nowadays, these Regional Environmental Protection Agencies[7] operate with a strong link to the territory concerned, which often requires some decision-making aids to optimise on-field actions toward sustainability. These include the ability to handle a wide range of problems; to capture many important aspects of such problems; and to judge how much a policy/project moves society towards some socially-defined and acceptable goal. In this sense, the present thesis provides some original pieces of work which combine both insights on valuation tools for decision support and empirical quantitative outcomes for mobility and the sustainability of agriculture in Italy.

[7] Agenzia Regionale per la Protezione dell'Ambiente (ARPA).

Table 1-1: Structure, contents, research challenges, and methods of the study

Structure	Environmental issues	Policy issues	Research challenges	Research methods
Part I	**Background**			
Chapters 1-2	Risks to ecosystems and humans	▪ Improve the sustainability of mobility and agriculture ▪ Design effective risk management strategies	▪ Cost risks to ecosystems and humans ▪ Appraise alternative risk management options	Stated choice methods (SC) Meta-analysis (MA) Risk and impact indicators Multicriteria analysis (MCA) Cost-benefit analysis (CBA)
Part II	**Urban environment**			
Chapter 3	Noise pollution	▪ Reduce noise pollution from rail transport infrastructures ▪ Choose the most cost-effective noise abatement strategy	▪ Review the principles of the economic valuation of traffic noise ▪	
Chapter 4	Rail noise pollution	[see above]	▪ Analyse individual preferences for alternative rail noise reduction policies using a choice experiment approach ▪ Estimate the marginal WTP for different features of a noise policy including: noise reduction, aesthetics, environmental and technical attributes, type of project financing ▪ Test the use of an innovative payment vehicle based on Bergstrom et al. 2004 ▪ Test the econometric robustness of CE estimates under three different payment vehicles	Choice experiment (CE)
Chapter 5	Traffic congestion Collective impacts due to sprawl	▪ Reduce traffic congestion ▪ Improve the sustainability of urban mobility ▪ Improve urban planning to reduce the collective costs of mobility	▪ Analyse the dynamics of the impacts of mobility in urban areas ▪ Analyse the environmental costs of mobility due to different types of urban development ▪ Provide deeper insights on the determinants of the environmental costs of mobility	Environmental-economic indicators Cross-section analysis Causal-path analysis (CPA)

Structure	Environmental issues	Policy issues	Research challenges	Research methods
Part III	**Rural environment**			
Chapter 6	Pesticide risks for ecosystems and humans	▪ Manage pesticide risks for human health and the environment ▪ Define effective pesticide policies (e.g. taxes, incentives) ▪ Set premium on organic or integrated-pest management products.	▪ Review the literature on the economic valuation of pesticide risks ▪ Identify which factors explain differences in WTP estimates ▪ Identify gaps in the empirical literature on the valuation of pesticide risks	Comparative analyses (CA)
Chapter 7	Pesticide risks for ecosystems and humans	[see above]	▪ Analyse individual preferences for alternative agricultural production scenarios using a choice experiment approach ▪ Estimate WTPs for more benign agricultural production practices leading to reduced pesticide risks for biodiversity, soil and groundwater, and human health ▪ Estimate WTPs for eliminating all risks from pesticides using contingent valuation	Choice experiment (CE) Contingent valuation (CV)
Chapter 8	Pesticide risks for ecosystems and humans	[see above]	▪ Formally review the pesticide risk valuation literature ▪ Identify factors explaining differences across WTP estimates ▪ Explore the possibility of using value transfer techniques	Meta-analysis (MA)
Chapter 9	Pesticide risks for non-target ecosystems	▪ Improve the sustainability of agricultural practices, reducing risks for non-target ecosystems (both aquatic and terrestrial) ▪ Define new methods for calculating incentives to adopt sustainable agricultural practices based on objective and quantifiable criteria	▪ Apply ecotoxicological risk indicators to define worst-case hazard scenarios at different time and spatial scales ▪ Test the usefulness of ecotoxicological risk indicators within a decision support system	Risk assessment Ecotoxicological risk indicators Multicriteria analysis (MCA)
Part IV	**Retrospect and Prospect**			
Chapter 10	Risks to ecosystems and humans	▪ How economic research can contribute to improving the sustainability of urban and rural systems ▪ How economic research can help in designing effective risk management strategies and optimising sustainability decision making	▪ Make concluding remarks on the research contribution of this book	

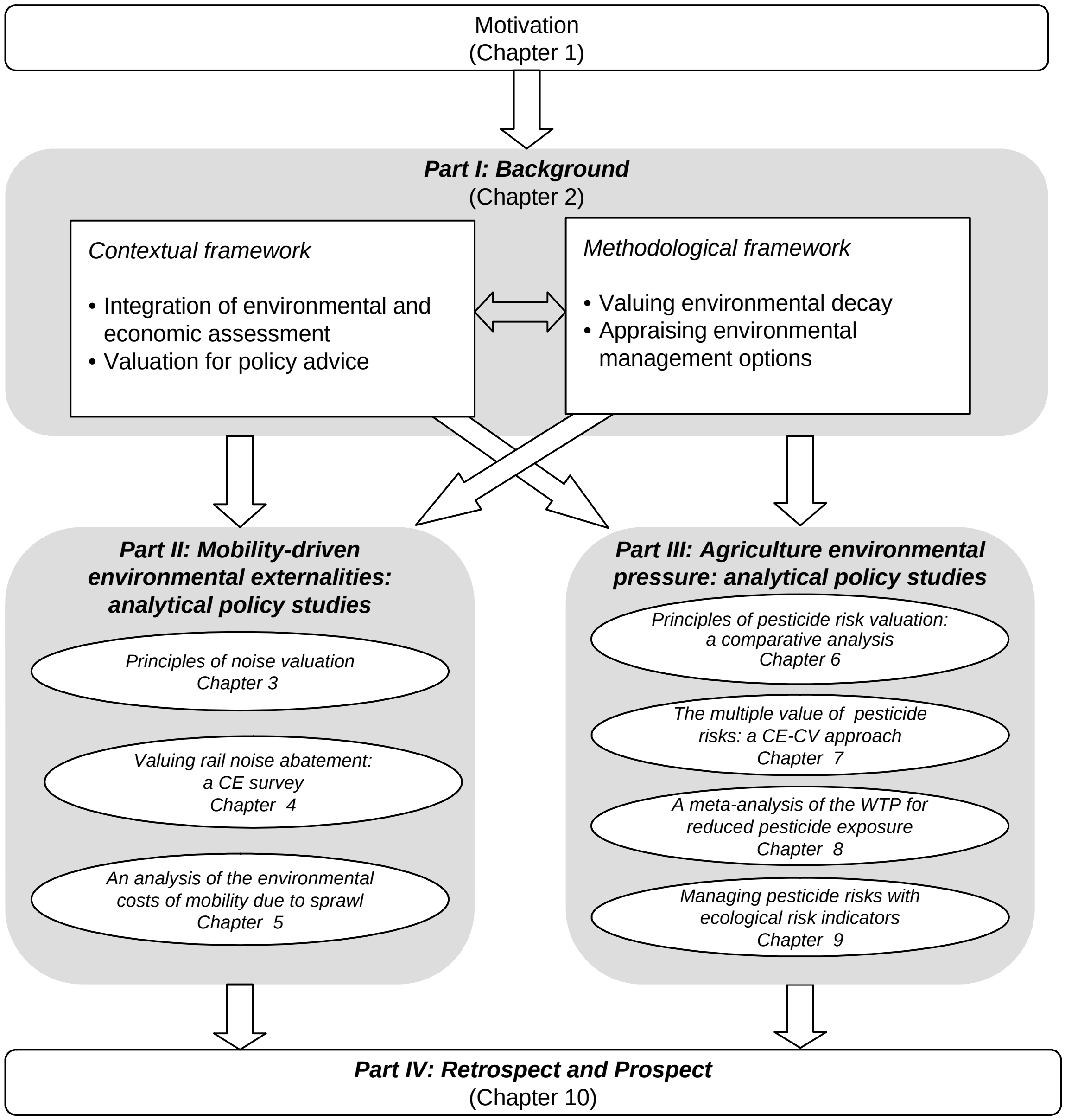

Figure 1-4: Structure of the thesis

1.5.2. Research questions and outline of the thesis

Before going into the empirics of the valuation of environmental risk and decay, which is the core of the present thesis, the notion of environmental externality, environmental value, and the related valuation paradigms assumed and the methods used need to be clarified and arranged. We are in fact looking for a comprehensive, flexible, methodological framework able to tackle the complexity that pertains to the analysis and valuation of environmental systems. This, as argued, concerns different levels. Firstly, it is manifest in the variety of causes that, in a complex network economy, drive environmental impoverishment and the negative effects on ecosystems and their communities. Secondly, complexity reflects the range of ecological and human populations exposed to risk of damage, which can vary considerably depending on the particular environmental issue addressed. Thirdly it concerns spatial and time scales, since the relevant territorial levels and time horizon differ according to policy objectives. Fourth, it concerns personal, societal and decision makers' preferences. Finally, it stresses in the need for multiple, complementary and specific policy actions for managing and reducing risks to their minimum.

It follows that a number of relevant issues need a thorough analysis when looking for a framework to value environmental risk. What are the main contextual, theoretical and methodological issues in the economic valuation of environmental decay? Which valuation paradigm do we need, and how can we choose among available valuation methods? Some questions have already been answered in the literature, or they just require factual responses, while others really need a systematic research approach using a concrete method. Moving from consolidated valuation methods (discussed in Chapter 2), we test their applicability for policy advice with a set of analytical studies on mobility and agriculture. Empirical analytical research enables us to experiment with innovations in methods and provide original insights concerning specific environmental fields of current policy and research relevance. To keep the body of the thesis tractable and harmonious, the empirical analysis is divided into two main areas of study: urban environment (Part II) and rural environment (Part III).

- *Urban Environment (Part II)*

This part of the thesis focuses on the problem of valuing the environmentally-negative side effects of mobility in urban areas. It opens by addressing the issues of mobility-driven noise pollution. Among the wide range of impacts due to urban mobility, noise has recently attracted the attention of European policy makers (CEC, 2003). National and EU policy makers are asking to be informed on the costs and benefits of reducing noise emission levels, and on the most cost-effective type of possible abatement strategy. The first question of this thesis therefore asks:

Question 1: Can we rely on stated choice methods for valuing alternative noise mitigation plans?

Looking for a sound answer to Question 2, Chapter 3 discusses the theoretical underpinnings of the valuation of transport noise, and provides a summary of the state of the art of the empirical economic literature, based on direct valuation methods. Some controversial issues in stated choice approaches for noise valuation are debated, such as whether people understand what a certain reduction in noise would mean to them and the econometric robustness of WTP estimates with respect to the use of alternative payment vehicle schemes. Then, Chapter 4[8] explores the use of a choice experiment survey in order to assess the economic value of alternative policy interventions for noise mitigation, and the respective measures. It presents a choice experiment survey held in Trento, Italy, to assess the marginal WTP for noise reduction, aesthetics and environmental attributes with respect to alternative railway noise reduction plans. In addition,

[8] Based on Nunes and Travisi, 2006a and 2006b.

the study provides an original contribution in the valuation literature since it explores the use of alternative payment vehicles, i.e. an additional local tax and a tax reallocation.

One more policy-relevant field of research addressed in the thesis is the phenomenon of sprawl and its implications in terms of mobility impacts. The European Commission has in fact recently recognized urban sprawl as the most urgent urban design issue as it leads to loss of rural space, high infrastructure and energy costs, and increases functional division and social segregation in the city (CEC, 2004). On the other hand, it is difficult to capture the intensity of the impacts of mobility, which can be ascribed to sprawl, and to understand to what extent congestion depends on sprawl, and vice versa. This motivates the second research question:

Question 2: How can we capture the intensity of the impact of urban mobility? Which factors explain its intensity, and what is the causal chain that drives it?

According to this question, Chapter 5[9] addresses the problem of traffic congestion in urban areas and provides a quantitative analysis of the determinants of the impact of urban mobility, by using environmental-economic indicators. The valuation paradigm assumed is based on an original conceptual interpretation of the causal chain that drives urban traffic, and the related environmental effects. This chapter combines a static and a dynamic perspective on urban mobility and makes use of cross-section regression analysis and Causal Path Analysis. The aim of this chapter is to understand whether there are structural, social and economic elements that contribute, systematically, to increase the demand for urban mobility and, thereby, the related environmental impacts, in order to gain policy relevant insights, focused on the Italian scenario.

- ***Rural Environment (Part III)***

This part of the thesis deals with the valuation of the environmental-drawback effects of agricultural production on ecosystems and human health. In particular, the problem of the valuation of pesticide risks is addressed. This aims to provide new research insights on the monetary valuation of pesticide risks, and respond to the increasing consumer awareness for food safety and the social preference to improve the environmental sustainability of agriculture, as well as the policy need to know the value of pesticide risks in order to devise sound policy instruments. The third research question therefore asks:

Question 3: Which factors influence variations in the willingness-to-pay estimation of risk reductions? How can one estimate the value of pesticide risk reduction with stated choice methods? Is it possible to rely on meta-analysis and value transfer for costing pesticide risks to ecosystems and humans?

Chapter 6[10] discusses the theoretical basis of the valuation of pesticide risks and presents a critical overview of the empirical literature on pesticide risk

[9] Based on Travisi et al., 2006a.

[10] Based on Travisi et al., 2006c.

valuation that provides disaggregated willingness-to-pay estimates (WTPs) of pesticide risk reduction. Recent multidimensional classification methods, such as decision-tree analysis, are used in a comparative approach as tools for explaining the differences in empirical research findings. The analysis shows that the order of magnitude of WTPs is related to both the valuation technique and to the data available from biomedical and eco-toxicological literature, and it shows that WTP estimates of pesticide risks cannot be simply averaged over several empirical studies. Then, Chapter 7[11] presents the results of an empirical study recently conducted in the North of Italy aimed at estimating the economic value of reducing the wide-ranging impacts of pesticide use. A Choice Experiment is used here in combination with Contingent Valuation methods. The experimental design of such a choice modelling approach provides a meaningful tool to assign monetary values to the negative environmental effects associated with agrochemical use. In particular, this survey addresses the reduction of farmland biodiversity, groundwater contamination, and harm to human health. Finally, Chapter 8[12] provides a formal review of the empirical valuation literature dealing with pesticide risk exposure, and develops a taxonomy of environmental and human health risks associated with pesticide usage. Subsequently, it investigates the variation in WTP estimates for reduced pesticide risk exposure with meta-analysis techniques. The income elasticity of pesticide risk exposure is generally positive, although not overly robust. Most results indicate that the demand for human health and environmental safety is highly elastic. The results also show that geographical differences, characteristics of the survey, and the type safety device (eco-labelling, integrated management, or bans) are important drivers of the valuation results.

On the other hand, a proper management of pesticide risks might also require the use of more comprehensive, multicriteria, analytical valuation approaches (OECD, 1999). This is particularly true when the analyses concern future – and therefore uncertain – risk and decision-making scenarios. In such circumstances, where available risk information is uncertain and relevant decision criteria are manifold, effective tools to manage pesticide risks should be capable of reaching a compromise between the demand for a sound scientific approach and the need for a transparent public policy tool. This motivates the fourth and last research question:

Question 4: Is it possible to use eco-toxicological risk indicators to provide sound scientific and user-friendly support for effective ecological risk management?

Chapter 9[13] uses some recently-developed pesticide risk indexes and tests their potential for management purposes. In the search for effective pesticide risk management tools, a pilot approach is proposed, which explores worst-case ecological hazard scenarios at different space-time scales by means of a set of 5 eco-toxicological risk indices. The results are then interpreted from the perspective of a decision support method using the Critical Threshold Value approach. The risk

[11] Based on Travisi and Nijkamp, 2004.

[12] Based on Florax et al., 2005.

[13] Based on Travisi et al., 2006b.

analysis is then enriched within a multicriteria framework which integrates environmental, agronomic, and economic objectives.

2. VALUING EXTERNAL COSTS RELATED TO MOBILITY AND AGRICULTURAL PRACTICES

While in our modern age spatial mobility and advanced agriculture have led to considerable direct benefits for travellers, farmers, and consumers, other individuals or households are likely to suffer from increased environmental damage associated with private transport and the intensive use of agricultural land. Examples are numerous and they range from reduced air quality and noise due to heavy road traffic, to loss of biodiversity, and surface and groundwater pollution in rural areas. In these circumstances, the effect of transport or agricultural market transactions on the welfare of households and communities, via the unpriced impact on the services rendered by the environment, is called an external effect or *externality* in economic analysis. Because the effect is generally negative, the term 'external cost' or 'negative environmental externality' is common. Since proper markets for environmental services and externalities do not exist, in the presence of environmental externalities, market transactions become inefficient, and the market's allocation of scarce resources is distorted and therefore not able to maximise collective utility.

In environmental decision making, under conditions of scarcity, the inability to price environmental external costs generates inefficient decisions. For rational choices on the allocation of public or private resources to the improvement of sustainability in the context of spatial mobility and agricultural land use, a trade-off between the costs and benefits of available alternative policy strategies needs to be made, even though the marginal costs of improving the marginal benefits are not directly observable. For instance, these would consist of the value of reduced traffic noise and air pollutants, or reduced pesticide exposure to humans and natural ecosystems, and so forth. Economists have tried to quantify such benefits by establishing, *inter alia,* measurements based on the well-known concept of *willingness-to-pay*, and consequently a number of quantification techniques are available (e.g. Freeman, 2003). Nevertheless, the process of quantification raises several objections and presents challenging research issues. For some mobility or agricultural land use impacts, for instance, quantitative data on the physical impacts will not be available, so that it will be not possible (or not straightforward) to put a monetary value on the impact. For other impacts, suitable economic valuation techniques will not exist, or will not be applicable because of time and budget constraints. Sometimes, environmental value quantification might raise ethical objections, and technocratic or 'deep ecology' (Singer, 1979) solutions may sometimes be advocated. Yet, for a complete assessment, all the significant impacts must be incorporated into the decision-making process. Therefore, a methodological approach flexible enough to be applied across a range of impacts and scales – from broad aggregated impacts on a region down to very refined disaggregated impacts on a particular target receptor – need to be adopted. Aiming to build the theoretical and methodological framework of this study, the present section tries to shed light on some relevant issues in the valuation of environmental decay, describing the methodological approach employed in this book for valuing external costs related to mobility and agriculture. Section 2.1 highlights the contribution of economic valuation for environmental-management decision making. Section 2.2 links the decision-making problem and the notion of environmental value to available valuation methods. Next, Section 2.3 provides a description of the methodological approach used in this book, and a brief presentation of the single valuation methodologies applied in empirical studies. In particular, stated choice methods, comparative analysis and meta-analysis, and the use of environmental indicators are discussed. Finally, Section 2.4 ends this section by providing some concluding remarks on the pros and cons of the valuation methodologies presented.

2.1. Valuation in environmental policy making: the importance of pricing the unpriced

Externalities arise when the decisions of some economic agents (individuals, firms, governments) – whether in production, in consumption, or in exchange – affect other economic agents, and are not included in the price system of commodities, i.e. are not compensated[14]. Hence, economic agents consider only their private marginal costs, when making decisions, disregarding the total – or social marginal – costs of their actions (e.g. see Hanley and Spash, 1993). As a result, private decision making in the agricultural and transport sectors might lead to goods with socially undesirable characteristics being sold in socially inefficient quantities. Equally, private decision making on the part of consumers might lead to personal behaviour with socially undesirable effects. Imagine, for example, the case of a farmer who applies pesticides on a field to protect harvests from pests. The farmer faces some health risk himself in doing this – he might inhale some of the chemicals, or they might come into contact with his skin, etc. Nevertheless, he does not face the full cost of his actions. He might not care, for instance, about the health of people drinking water that comes from groundwater reservoirs beneath his land. Likewise, the driver faces a personal risk of accident while driving his car, but he might not care about the annoyance that his noisy vehicle causes to the neighbourhood. The well-known source of such problems is market failure: some significant economic factor goes 'unpriced', so that economic activity is undertaken without consideration of its full impact. In environmental decision making, pricing the unpriced becomes crucial for enhancing policy solutions and allowing efficient options appraisal (for a discussion see, e.g., Pearce and Seccombe-Hett, 2000). Valuation acquires a central role that is discussed below in the context of environmental decision making. The general principles presented are valid in the context of decision making for managing both agriculture and mobility impacts.

Environmental policy making can be described as a feedback process that goes from the problem specification through to the *ex post* evaluation (Figure 2-1, modified from Willows and Connell, 2003).

[14] In other words, property rights are not assigned properly, so that the incidence of effect does not coincide with the distribution of legally-recognized controls.

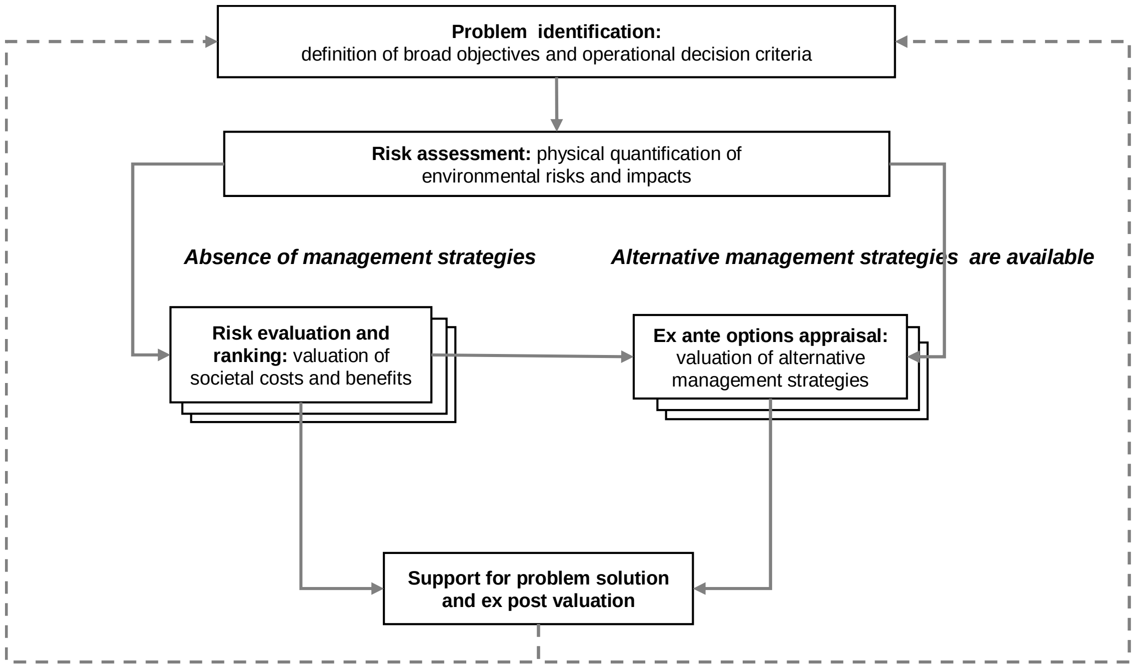

Figure 2-1: Valuation in the context of a framework to support good decision making in the face of negative environmental externalities related to agriculture and transport

Note: Modified from Willows and Connell, 2003.

A problem may arise as a result of, for instance, changes in agriculture (or transport) legislation, reviews of ongoing activities, public concern, or the emergence of new evidence on the environmental and health risks due to the heavy use of chemicals in agriculture (or the increased shift towards private motorised vehicles in the context of urban mobility). The decision maker is the person or the institution that is dissatisfied with the prospect of an ongoing (or future) event, and that possesses the wish and authority to initiate actions designed to adjust this event. In the context of this book, the decision maker may represent a national, regional or local government; a department within one of these levels of government; an environmental regulator; a multinational or small and medium-sized enterprise, whether privately or state-owned; or individual members of society. For instance, a local administration concerned about the excessive level of air pollutants in urban areas, due to private traffic, is potentially a decision maker in this sense. The local administration may be dissatisfied with the low air quality because it compromises a major objective, e.g. the reduction of diseases due to air pollution and the reduction of hospitalisation costs.

To pursue a broad objective, the decision maker must first translate the objective into operational decision-making criteria. In our example, one criterion might involve the reduction of air pollutant emission levels under national regulation limits. These criteria will facilitate the identification of alternative options to alleviate, in this example, the health problem and allow the desired state to be reached. On the other hand, the decision criteria also serve as a basis for the risk assessment and as a basis for assessing the performance of the various policy options under consideration. These options, as well as the state of doubt as to which one is *best*, constitute the heart of the decision problem. For instance, in the case of low air quality, is the health problem best addressed through, say, private mobility demand management, or public transport supply enhancement? Or, even, is it best not to address the health problem at all?

Decision makers can be supported by the results of environmental valuation performed in two stages of making environmental risk (or impact) management decisions:

- Assessment, prioritisation, and ranking of risk and impacts: to generate robust order-of-magnitude estimates of the cost of agriculture or mobility externalities, so that their relative importance can be established.
- Management options appraisal: to generate valid order-of-magnitude estimates of the net benefits of management strategies for tackling specific environmental risks.

In the first stage, the role of environmental valuation is to estimate the economic value (positive or negative) of a given environmental risk, in our case stemming from agriculture or mobility, in the absence of management strategies. The reference scenario would be defined by the ongoing situation, in a given geographical and temporal context, in the absence of policy actions to manage the risk. When analysing future events, more realism might also be introduced by constructing projections of future natural, environmental and socio-economic conditions in the study region, in the absence of mitigation or risk management strategies (Parry and Carter, 1998). The value of this information is that it reveals to decision makers those impacts that are likely to cause the most severe damage and, thus, those risks to which most attention should be given.

In the second stage, we assume that decision makers can undertake some form of action in response to important agricultural or mobility-driven risks to ecosystems and human health. The effect of the policy intervention is to reduce the future exposure of a receptor to the risk concerned. We can think of the reduction in the risk as the effectiveness of the policy response, or the gross benefits of acting against risk. This is given by the estimated impact of a given environmental risk in the absence of policy actions, minus the estimated impact with adaptation, as illustrated in Figure 2-2. In such a management decision context, environmental valuation can be used to estimate the gross monetary benefit of a management strategy. Furthermore, the magnitude of residual impacts on selected receptors across different study areas can also be evaluated. The value of this information to decision makers is that, together with the information on the resource costs of the management strategy, it can be used to answer the following general policy question: Is the gross benefit of the management strategy greater than the cost of the strategy? In addition, this information allows the decision maker to accept or reject a single policy option, or to choose one option out of a number of possibilities.

In this thesis, sound valuation methods are applied in both the aforementioned stages in making environmental risk (or impact) management decisions, as detailed in the following sections. Before describing the overall logic of the valuation methodological approach used here, a general discussion of the notion of value is presented, and the debate on the best valuation paradigm to be applied for policy-making support is summarised.

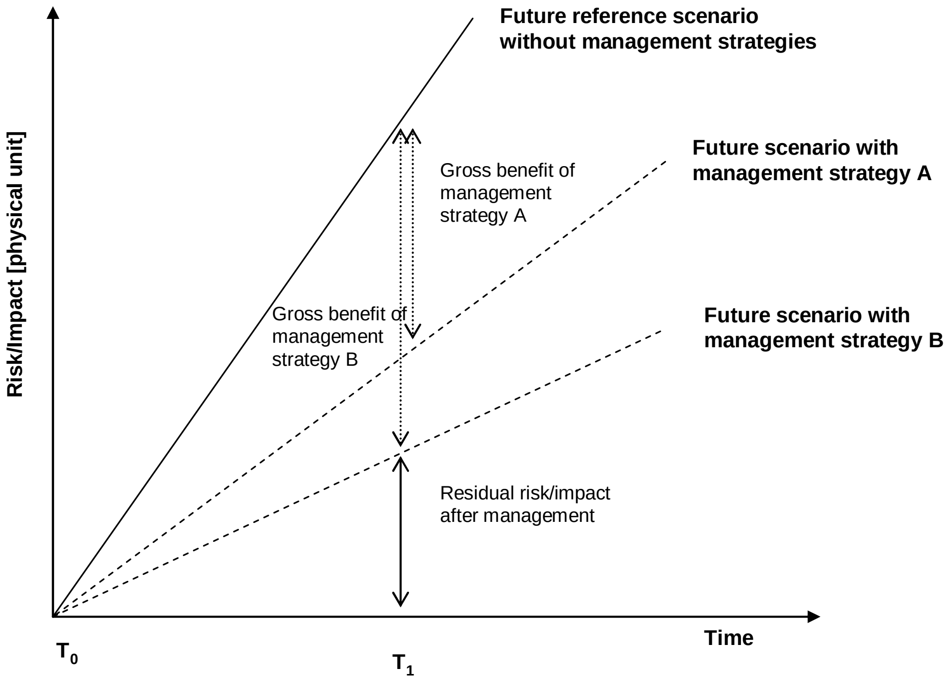

Figure 2-2: The benefit of environmental risk management

Note: Modified from Parry and Carter, 1998.

2.2. Which valuation paradigm do we need?

In common usage, the term *value* means relevance or desirability. It is clear, however, that the transfer between general usage and each discipline's perspective on how relevance or desirability should be gauged has stimulated considerable debate among ecologists, economists, and philosophers (e.g. O'Riordan, 1976; Blamey and Common, 2000). From alternative conceptions of what environmental value is, in fact, different notions of sustainability and environmental quality can be derived, each of which is expected to influence policy aims, valuation tools and instruments for proper management (for a discussion see, e.g., Turner, 1992, 2000).

For environmental economists, the valuation concept relates to human welfare, and human welfare is captured in terms of each individual's own assessment of his or her well-being. The measure Total Economic Value (TEV), therefore, sticks to human, anthropocentric values, including use and non-use dimensions (see Figure 2-3). Use values are values arising from the actual use or consumption of the environmental resources (Pearce and Moran, 1994). Use values are further divided into utilitarian (either direct 'market price' or indirect 'non-market' ones) and option values. Since we focus on agriculture and mobility environmental impacts, *market-price values* refer to the various forms of market production possibilities available to farmers and commuters; the *indirect non-market use value* refers to benefits deriving from ecosystem functions, such as the role of agriculture or mobility in affecting the air, water and soil quality; and the *option value* refers essentially to the individual's WTP for the preservation of the ecological dimension against some (subjective) probability that the individual will make use of it at a future date (Randall, 1987, 1991). Agriculture and mobility have, however, impacts on the well-being of the individuals who are not directly associated with agriculture or mobility consumption. These are referred to as the non-use values, i.e. anthropocentric values that are not associated with current or expected use/consumption of the environmental resource (Carson et al., 1992, 2001). The non-use values are usually divided into the bequest value and the existence value. The *bequest value* refers to the benefit accruing to any individual from the knowledge that others might benefit from environmental goods and services that might be spoilt by agriculture and mobility in the future; the *existence value* refers to the benefit derived simply from the knowledge of the continued protection of the air, water, and soil quality[15]. The non-use values typically have a public good character for which the non-market price is available to disclose accurate monetary valuation. The lack of such market price information may convey the impression that the benefits of agriculture and mobility environmental policy are unimportant, when compared with the market price allocation alternatives (such as, urbanisation and agriculture development). As a consequence, most of the time, policy makers have based their decisions on an undervaluation of the environmental resources, which has thus resulted in a misallocation of scarce environmental resources. The monetary assessment of the use and non-use benefits involved with risk management policy is, therefore, an important step in policy decisions about environmental resource use. The monetary value assessment of such environmental assets requires special tools, which have been proposed by environmental economists (for a review, see Pearce and Markandya, 1989).

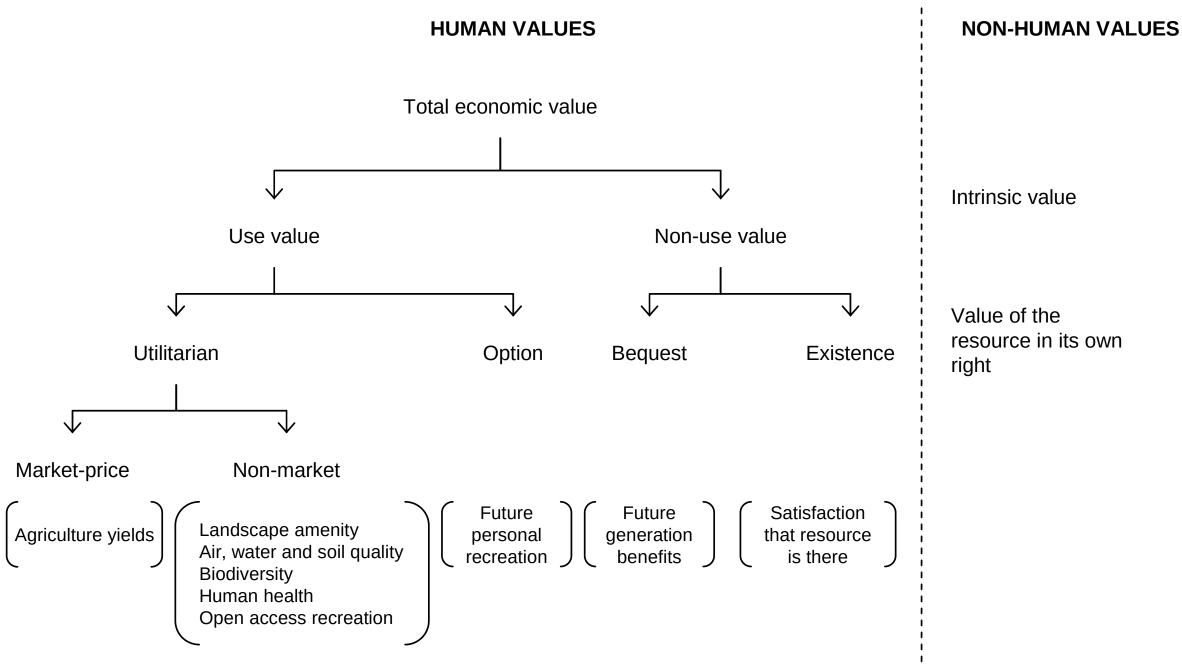

Figure 2-3: The total economic value (TEV) of environmental goods

Note: Modified from Bateman et al. 2003.

[15] The origin of the notion of existence value is attributed to John Krutilla who suggested that people may still value a resource even if they do not use it (Krutilla, 1967).

However, for their part, ecologists and natural scientists doubt the capacity of economists to estimate the value of ecological systems, or even to analyse their dynamics, because of the difficulty of understanding the intricate pattern of physical interrelationships (see Bockstael et al., 1991). To some extent, ecological economists represent a link between such opposite perspectives, and offer the possibility for dialogue and reciprocal cooperation. Ecological economists, in fact, allow extensions to the concept of TEV and embrace the idea of the 'moral reference class' (Turner et al., 1994) for decision making. In this respect, one important question involves the treatment of other human interests (both present elsewhere and future); while another point is whether animals, plants, ecosystems or, even more in general, other entities' interests should be placed on a equal footing with human preferences (see O'Riordan, 1976; Goodpaster, 1978; Watson, 1979; and, more recently, Rollston, 1988). A further divergence from conventional utilitarianism has been proposed by Turner (1992) who argues that all the elements of TEV might be considered as secondary values to primary environmental quality which is a necessary precondition for the generation of all the other values. Even without considering the theoretical case for such philosophical specifications and extensions, a practical problem with these non-TEV dimensions is that they are essentially beyond the scope of conventional, preference-based economic environmental valuation. So, it is necessary to restrict the moral reference classes solely to humans in order to consider the environmental valuation paradigm and the TEV the appropriate standards to define and measure environmental values and, therefore, sustainability.

One suggested solution to the problem of environmental valuation might be to abandon conventional neoclassical economic analysis in favour of modified or alternative appraisals and decision-making strategies (e.g. Bateman et al., 2003). Nevertheless, before coming to conclusions, one ought to wonder whether an environmental and ecological economics evaluation model should, in practice, be considered as competing or complementary. Recently, some commentators have been stressing the methodological and theoretical gap between these two approaches to the analysis of the environment. Bateman et al. (2003) argues that "the choice between ecological and environmental economics could be characterised as one between principle and pragmatisms". The former advocates the full assessment of economic and environmental values, both 'anthropocentric' and 'non-anthropocentric', which moves further towards strict utilitarianism. The latter is somehow closer to the mechanisms by means of which present-day decision making operates. Other commentators highlight the difference between ecological and environmental valuation models as a choice between equity versus efficiency (Blamey and Common, 2000); risk and uncertainty management versus narrow costs-benefits accounting (Hanley, 2000); plural versus hierarchical monetary values (Spash, 2000); experts versus stakeholders judgment (Jassen and Munda, 2000).

The present book takes the stance that the environmental and ecological economics valuation paradigms are complementary rather than competing valuation models, because the problem of sustainability and risk evaluation cannot be separated from the purpose for which it is required, and from the context in which it takes place (for a discussion see, e.g., Norton, 1995; Turner, 2000). In other words, no matter which of the previous dichotomies one wishes to refer to, if one looks into the arena of sustainability and environmental-risk decision making, the role of both paradigms can be clarified to provide indications on the pros and cons of their adoption, depending on the specificities of the evaluation context (policy objectives, environmental impact of concern, groups of interests involved, time and spatial constraints, etc.).

For instance, if one looks at the notion of 'value' adopted by environmental economists, one can agree on the fact that this starts to waver – or at least can be disputed – as one moves towards moral positions (e.g. see Blamey and Common, 2000). Nevertheless, to perform Cost-Benefit Analysis (CBA) in assessing environmental policy options and for purposes of determining liability whenever natural dimensions are threatened, such a concept has considerable precedence as well as legal standing. To some extent, the incorporation of non-market environmental values into policy and liability consideration has been possible because the economists' concept of value has a long history of rigorous thought behind it (for a discussion see Boyd, 2000). It is therefore difficult to set aside these argumentations and stick to ethical standings if so often, in real world circumstances, there are a number of situations in which decision makers would not only benefit from having information on monetary values of environmental dimensions, but explicitly ask for them. First, in Europe, we observe a revival of interest in the formal assessment of environmental policies based on CBA principles (see, e.g. Pearce and Seccombe-Hett, 2000). So, CBA is recommended for *ex ante* and *ex post* environmental policies appraisal, as well as whenever it is necessary to choose among alternative environmental management plans with some budget constraints. In principle, either *ex ante* or *ex post* policy evaluation can be required for any environmental policy. To give just one example, the European Union currently uses a value of € 1 million per human life in safety CBA, generally referred to as the "1-milion-euro rule" (Despotin et al. 1998). Secondly, natural resource damage assessment is qualified by the availability of monetary estimates of the value of the detriment that has occurred. The same goes for legal claims for material (health) or immaterial injuries (moral) against either insurance companies or any person who is responsible for them. Maybe the most famous example is the *Exxon Valdez* oil spill: as both the State of Alaska and Exxon were arguing over the size of the environmental damage, evidence was therefore sought from Contingent Valuation Method studies. Thirdly, proper pricing of renewable and non-renewable resources is suggested to reduce waste and enhance the effectiveness of resources management. This requires a monetary valuation of environmental goods and services. Furthermore, such analysis may also offer a foundation for Pigouvian taxation schemes in spatial-environmental planning, as well as for the design of incentive schemes. These are, respectively, often applied to promote more environmentally-sustainable behaviour or to refrain from highly-impacting ones. Eloquent examples are incentives for the use of low-impact agrochemicals in agricultural production (see Jensen and Stryg, 1996) or, to cite a measure recently adopted in London and debated in Italy, the adoption of road pricing schemes to control urban traffic in metropolitan areas, nowadays choked with smog. Finally, related to this, monetary estimations of some specific environmental dimensions are crucial to set a proper price premium on goods with extra quality features in terms of human health safety, eco-sustainability or, even, ethics. Recent examples of such goods are preservatives, GMOs or pesticide-free foodstuffs, fair-trade products, and so forth.

Likewise, there are a number of situations that entail going beyond the neoclassical cost-benefit perspective without rejecting the usefulness of preference-based values[16]. A revision of CBA principles applies and can be recommended, for instance, to high uncertainty, irreversible problems, as is the care for Genetically Modified Organisms (GMOs), climate change or Persistent Organic Pollutants (POPs). In such circumstances, there is at almost general agreement that the ecological economic valuation principle can help to tackle problems with a broader information basis and is, therefore, expected to provide, if not stronger guarantees of success, at least a wider range of elements for judgement. Moreover, to envisage uncertainty or irreversibility as the *conditio sine qua non* for adopting ecological economics principles is also disputable. Though ecological economics is the 'natural ground' on which to reassert the precautionary principle, this does not preclude the economic analysis of arguably more general situations where the precautionary principle does not apply. In fact, the true novelty of ecological economics is that it links up with other environmental science disciplines, rather than in rejecting welfare economics principles. Nevertheless, the methodology for the integration of socio-economic and ecological variables is usually fraught with many difficulties. In several circumstances, research has to rely on ad hoc information, expert opinions and, so forth, so that one can reach a more precise assessment of the socio-economic and environmental implications of environmental policies. In this context, Nijkamp (2000) has recently noticed that: "The results of valuation and modelling studies are often fed into decision support analysis which may comprise various analysis frameworks". Among others, these may pertain to multiple objective programming models, or to multicriteria analysis for quantitative, qualitative or fuzzy information. These are all methods that have already been extensively applied for environmental policy studies and that have proved their feasibility in past decades. Nonetheless, there are still a number of challenging tasks (for a comprehensive discussion, see Nijkamp, 2000), because research efforts for an economic analysis of the environment need to follow the ongoing evolution in environmental-spatial and socio-economic transformations, and to explore new frameworks of analysis able to offer a consistent and sound scientific picture of environmental, social, and economic linkages.

The next section describes the logic adopted in this book for selecting the most suited valuation method to be applied in each of the empirical studies developed and presented.

2.3. Assessing the external costs of agriculture and mobility: methodological approach

According to the previous discussion, several valuation approaches can be adopted to value environmental externalities due to agriculture and mobility or, more in general, to evaluate environmental risks. Taking a step further from the debate on the moral and theoretical foundations of the notion of total economic value (TEV) and the related valuation paradigms (briefly presented in Section 2.4.),

[16] Recently, Pearce (2003) has noticed: "It is fashionable to criticise the economic approach for all kinds of supposed ethical aberrations, but it has an ethical force of its own. It lies in the democratic expression of individual preferences to rule together with those of 'unelected stakeholders and experts'."

in practice, the selection of the preferred valuation method depends on a number of factors that are illustrated below.

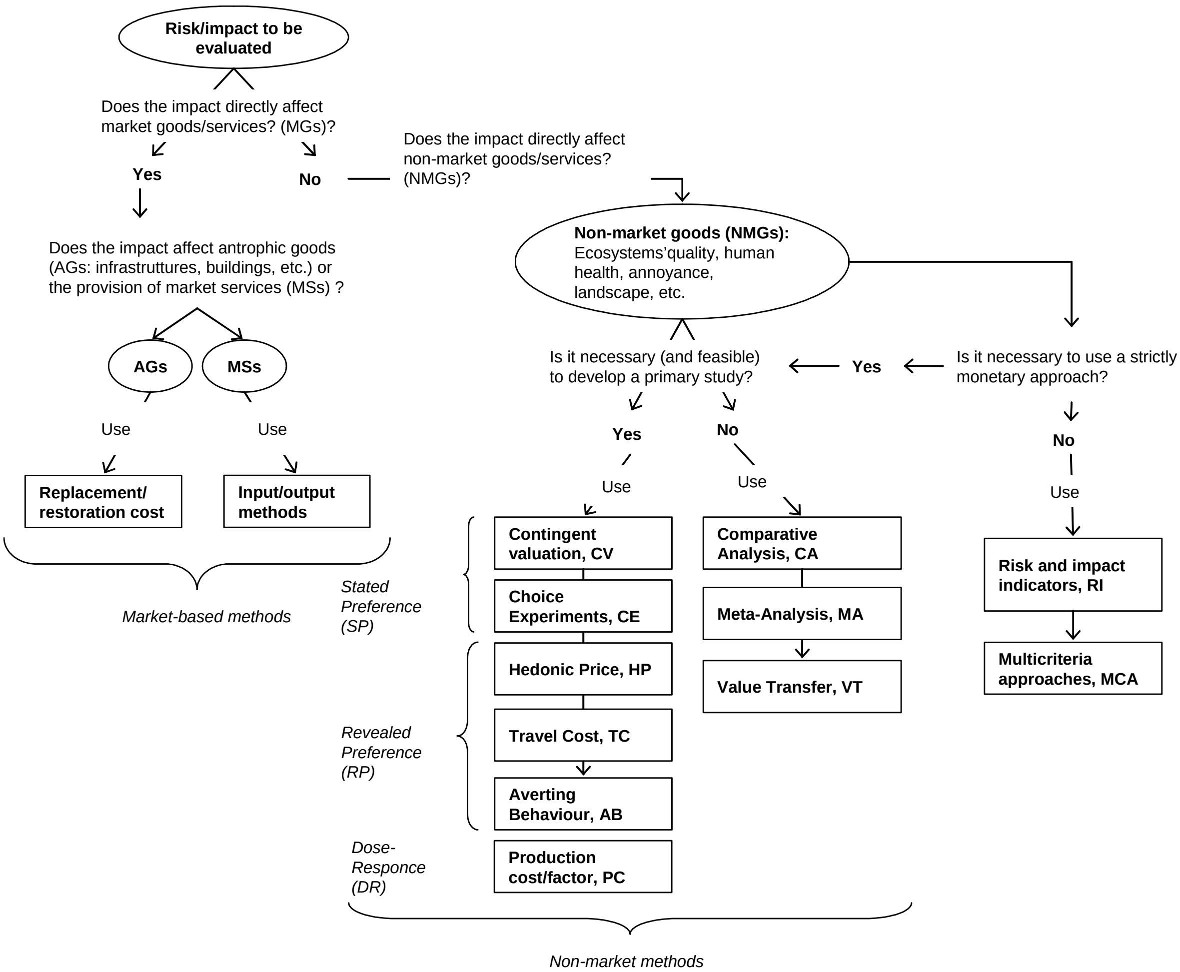

Figure 2-4: Relevant factors for selecting the preferred valuation method

Note: Modified from Metroeconomica, 2004.

A first element to be considered when selecting the preferred valuation method is related to the nature of the good being affected by the risk/impact of concern. In particular, one can make a distinction between goods or services exchanged in regular markets, the 'market goods', and goods/services that are not sold and bought in regular markets, the 'non-market goods'. Clearly, the environmental externalities generated by agriculture and mobility that are considered in this study belong, by definition, to the latter category. For instance, rail noise annoyance, or the reduced water quality caused by pesticide use, are non-market goods. In Figure 2-4, this distinction leads either to market (also called non-preference) or non-market (preference) valuation methods. Following such a choice factor, we enter the right side of the diagram on non-market valuation methods.

A second choice element now concerns the possibility of undertaking an original primary study, or not. As already stated, in fact, sometimes suitable economic valuation techniques will not be applicable due to time and budget constraints. Likewise, before undertaking a costly primary study, an in-depth literature review might be strongly required in order to better orient the empirical work. In such circumstances, the use of research synthesis methods (such as comparative analysis and meta-analysis) and value transfer techniques might be conveniently employed (see Section 2.5.). Next, if a primary valuation study is needed, researchers can rely either on direct (Stated Preference, SP) or indirect (Revealed Preference, RP) valuation methods, or on the Dose-Response Methods (DR). Depending on the issue at stake, the advantages and disadvantages of SP versus RP and DR methods will be compared and the best valuation methodology selected. In the empirical part of this book, preference is given to direct SP valuation methods, as extensively motivated in each chapter.

Finally, a third question is whether it is essential and relevant to employ a strictly monetary valuation approach. For some mobility or agriculture impacts, for instance, quantitative data on the physical impacts will not be available, so it will not be possible (or not straightforward) to put a monetary value on the impact. Sometimes, the environmental value quantification in monetary terms might raise ethical objections, and technocratic solutions be advocated. Similarly, sometimes decision criteria other than economic efficiency, e.g. social equity and environmental quality, might be advocated to enlarge the decision-making perspective. In these cases, risk or impact indicators might be used to quantify the environmental external cost (i.e. impact), though not in monetary terms, and be employed within a multicriteria framework of analysis for options appraisal.

The following section provides a brief description of the valuation methods employed in the empirical part of this thesis. The discussion is not intended to be complete or detailed. For more comprehensive methodological reviews, we refer the reader to the references indicated in the text and to the empirical analytical surveys of this thesis.

2.4. Valuation of environmental risks and impacts

2.4.1. Non-market monetary valuation methods

Various valuation methods are available to put a monetary value on the environmental impact of agriculture and mobility, and a number of classifications of the valuation methods have been proposed (Pearce and Markandya, 1989; Mitchell and Carson, 1989; Nunes, 2002). We distinguish three groups of valuation methods: the Stated Preference (SP), the Revealed Preference (RP) and the Dose-Response (DR) methods (see Figure 2-5). The SP and RP valuation methods have in common that they reveal people's preferences – either directly or indirectly – with a behavioural-linkage approach (Mitchell and Carson, 1989). The DR methods have in common that they put a price on environmental commodities without retrieving people's preference for these commodities (Nunes, 2002). The production cost techniques, for instance, calculate the monetary value of the negative effects of, say, air pollution of buildings, by using a production cost technique and multiplying the increased maintenance and repair prices (Feenstra, 1984). Another example of the dose-response method is when researchers use the production factor approach to estimate, for instance, the economic value of cleaner soil by means of the increased agricultural output by using a demand and supply model (Smith, 1991). Conversely, the SP and RP methods rely on an analysis of the individual preferences for a given environmental good or service.

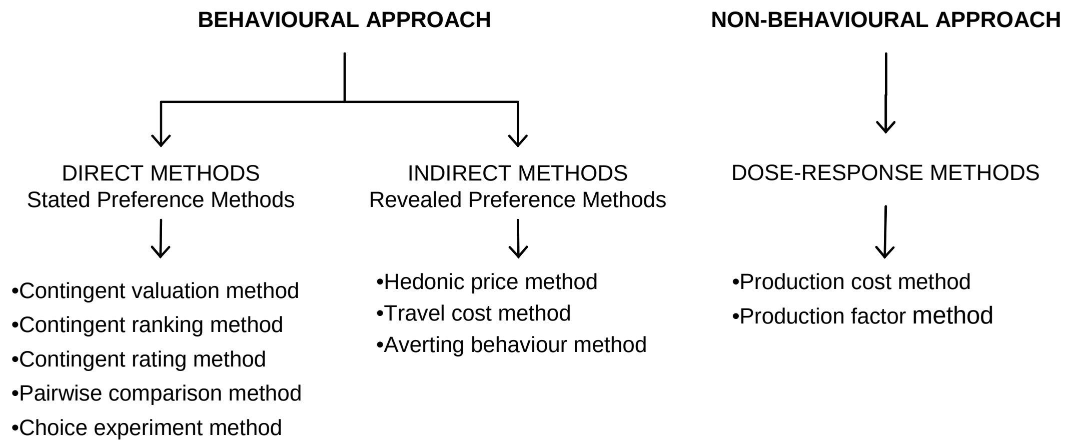

Figure 2-5: A classification of valuation methods

The group of RP valuation methods consists of three techniques: the Hedonic Price (HP) method; the Travel Cost (TC) method; and the Averting Behaviour (AB) method (Branden and Kolstad, 1991). The common underlying feature is a dependency on the relationship between a market good and the environmental commodity to be valued (Nunes, 2002). For instance, when using the travel cost method, researchers estimate the economic value of recreational sites by looking at the costs of the trips made by visitors to the sites (Bockstael et al., 1991). Equally, when using the hedonic price method, researchers estimate the economic value of an environmental good, say noise nuisance, by analysing the relationship between house prices and the surrounding noise levels (Navrud, 2002). Researchers who use the averting behaviour method try to estimate the economic value of environmental quality on the basis of the expenditures made to avert or mitigate the adverse effects of pollution (Cropper and Freeman III, 1991).

The group of SP valuation methods all rely on survey methods to directly infer people's preferences for a given environmental commodity. The underlying feature is the use of ad hoc questionnaires to ask the individuals to directly state their economic values for environmental commodities (Mitchell and Carson, 1989). However, this method has a number of different versions, such as Contingent Valuation (CV), Contingent Ranking (CRk), Contingent Rating (CRt), Choice Experiment (CE) and Pairwise Comparison (PC) (Louviere et al., 2000). These variants of the survey method differ in the way in which the economic values are elicited (Nunes, 2002). For example, whereas the CV method asks respondents to express their preferences directly in monetary terms for some defined environmental good, the CRk (or CRt) method asks the respondent to rate (or rank) a number of described alternatives. The PC is closely related to the CRk method, but the respondents are asked to compare a series of pairs of alternatives. Finally, in a typical CE survey, the researcher presents two or more alternatives to the respondents, and asks the respondents to choose the most preferred one.

The contingent valuation method

The CV method[17] is one of the most used techniques for the valuation of environmental goods together with CE. CV[18] uses survey questions to elicit people's preferences for public goods by finding out what they are willing to pay (Willingness-To-Pay, WTP) for specified improvements in them. The method thus aims at eliciting their WTP in monetary terms. It circumvents the absence of markets for public goods by presenting consumers with hypothetical markets in which they have the opportunity to buy the good in question. The hypothetical market may be modelled after either a private goods market or a political market (Mitchell and Carson, 1989, p. 2-3).

The contingent market defines the good itself, the institutional context in which it would be provided, and the way it would be financed. Typically, a CV survey consists of three sections (Mitchell and Carson, 1989). The first section contains a description of the environmental change as conveyed by the policy formulation and a description of the contingent market. The policy formulation requires describing the quantity (i.e. the availability) of the environmental commodity in both the 'reference scenario' (typically the *status quo*) and 'target scenario' (usually describing the policy action). Since all monetary transactions occur in a social context, it is also crucial to define the contingent market by stating to the respondent both the rules specifying the conditions that would lead to policy implementation, as well as the payment to be exacted from the respondent's household in the event of policy implementation. The second section of the CV survey presents the question in which the respondent is asked about his or her monetary valuation for the described policy action. This is the most crucial part of the CV instrument. The major objective is to obtain a measure of the respondents' maximum WTP for the described environmental policy formulation. Finally, the third section of the CV survey presents a set of questions that collect socio-demographic and attitudinal information about the respondents. This information is crucial to better analyse the respondents' profile and is used to interpret the respondents' stated WTP responses. The third section finishes with follow-up questions, to assess whether the respondents have correclty understood the CV survey in general, and the valuation question in particular (Nunes, 2002, pp. 7-8).

The choice experiment method

The CE is a non-market valuation method that allows people's preferences to be inferred for a set of alternatives, described by a set of relevant attributes. This technique was first developed in market research and has then been widely applied in several other fields with the purpose of analysing choice behaviour, including transportation research, health economics, environmental economics, and the economics of cultural heritage (Louviere et al., 2000; Green, 2002; Bennet and Blamey, 2001). In a typical CE instrument, the researcher presents two or more

[17] Despite the fact that contingent valuation is one of the most widely used techniques to value environmental changes, this method is not free from potential irregularities and inconsistencies. An avalanche of literature is available on this (e.g. Carson et al., 2001).

[18] Since the elicited WTP values are contingent upon the hypothetical market described to the respondents, this approach came to be called the contingent valuation method (Mitchell and Carson, 1989, pp. 2-3).

alternatives to the respondents, and asks the respondents to choose the most preferred one. The alternatives are described as bundles of factors, known as "attributes", which are expected to influence respondents' preferences for the proposed alternatives. The alternatives comprised of bundles of attributes are called "profiles". A combination of two or more profile is called a "choice set" or "card". This scheme allows us to examine the attributes that influence choices and the relative importance of each attribute, through observation of the choice behaviour of the respondents.

In most cases, the main differences between CE and CV can be summarised as follows.

- CE analyses several attributes simultaneously, whilst CV tends to look at attributes in isolation. CE therefore allows the researchers to value several attributes in addition to situational changes. Moreover, the purpose of the study will be less obvious and a weaker tendency to strategic bias can be expected (Wardman and Whelar, 2001).
- CE can examine interaction effects and package effects and is also more useful when the scenario under consideration is multi-dimensional. Another comparative advantage is that CE makes it possible to measure compensation in terms of other goods instead of in money (Adamowicz et al., 1998).
- CE examines different levels of attributes, whereas CV generally does not. Hence, the CE approach supports the detailed and controlled analysis of the functional relationships between the valuation of an attribute and its level, as well as sign and size effects.
- CE tends to ask for order of preference, while CV tends to ask for strength of preference (Wardman and Bristow, 2004). CE responses can be expected to be more reliable for two key reasons. Firstly it is simpler to indicate the order than the strength of preference. Secondly, individuals routinely make choices but are rarely required to establish the strength of preference in real life circumstances.
- CE is a behavioural model from which values are implied, whereas CV is a direct valuation model. Whilst CE is more suited for forecasting applications, CV can avoid the problems involved in the development of the choice model (Wardman and Bristow, 2004).
- CV is relatively straightforward to design. In contrast, a CE survey is usually more complex than a CV one and surrounded by greater uncertainty. In addition, CE requires a bigger cognitive effort for the respondents which might lead to lexicographic responses or other biases.

2.4.2. Research synthesis

Research synthesis or research integration is concerned with summarising empirical research findings by drawing overall conclusions from several separate investigations that address identical or similar research questions. It involves the attempt to discover the consistencies and account for variability in studies that appear similar (Cooper and Hedges, 1994). In addition, research synthesis aims at presenting the state of knowledge about a problem of interest and at pointing relevant issues that previous research has left unresolved (Cooper, 1998). In short, it summarises, compares, integrates, and eventually extrapolates new insights from the results of primary and secondary analysis[19].

There may be various methods for research synthesis and comparison, starting from traditional qualitative narrative reviews. Several commentators have made a strict distinction between comparative analysis and meta-analysis and narrative literature reviews (Glass, 1976; Cooper, 1998; Florax et al., 2002). Similarly to the classification by Button (2002), we distinguish three forms of research synthesis methodologies: literary reviews (LR); comparative analysis (CA); and meta-analysis (MA) (see Table 2-1).

Table 2-1: A summary of research synthesis methods

Note: Modified from Button, 2002.

Technique	Information	Strengths ++	Weacknesses --
REVIEWS	• Publications • Reports • Speeches	• Combine qualitative and quantitative information • Easy to comprehende	• Lack rigour • Subjective
COMPARATIVE ANALYSIS	• Publications • Reports • Speeches • Interviews	• Systematic approach • Up-to-date insights • Combine qualitative and quantitative information •Can look forward	• Lack statistical rigour • Subjective • Fads in expert opinion
META-ANALYSIS	• Publications • Reports	• Transparency • Systematic approach • Statistical basis	• Limited case material • Selectivity • Comparability of studies • Problems of quantification

The longest established and most widely-used method for bringing together information from previous studies is the literary review. The researcher elaborates a written text, supplemented by illustrative data, in which findings of earlier studies are set out and compared, and judgements are made about the strength and quality of the various pieces of work being valued. A major problem of a traditional literary-type approach is that it tends to be qualitative and subjective, and therefore, lacks of rigour. On the other hand, it can combine qualitative and quantitative information and handle a diversity of case studies of various sorts (Button, 2002).

Conversely, subjective quantitative comparative analysis and meta-analysis are quantitative in nature (Glass, 1976; Cooper, 1998; Florax et al., 2002). Generally speaking, the main advantage of quantitative research synthesis is a reduction of the level of subjectivity (Florax et al., 2002; van den Bergh and Button, 1999). Although these methods cannot fully avoid the subjectivity problem, at least they can make certain judgements more transparent (van den Bergh et al., 1997a).

[19] Primary studies are analyses of new data to answer a particular research question. Secondary studies are re-analyses of the data to answer the same research question with new analytical techniques or to answer a new research question with old data (Glass, 1976).

In fact, where traditional reviews tend to be in the form of taxonomies of findings without any specific attempt to relate to the review's purpose (Button, 2002), quantitative forms of research synthesis are actually able to pinpoint and, whenever possible, estimate the impact of theoretical and methodological issues across the full range of studies (Stanley, 2001). Furthermore, another advantage of quantitative research synthesis over qualitative forms of research synthesis is its more systematic approach to analyse the varying results of previous research (Florax et al., 2002). A purely qualitative examination of existing study results on the same research question often fails to include specific study characteristics as potential explanations for consistencies or discrepancies across study results.

Comparative analysis

Comparative analysis (CA) aims at identifying the common and contrasting elements that characterise a certain phenomenon under investigation on the basis of the accumulated expertise of those knowledgeable in a certain area (Nijkamp et al., 1999). The phenomenon analysed may be described in several different ways, such as single descriptive or quantitative case studies; individual data from official sources; scattered and fragmented information in various unpublished research reports; and responses to questionnaires and articles published in scientific journals. In addition, CA may be carried out with the help of various different tools, including classification techniques, artificial intelligence techniques, and other quantitative methods such as rough set analysis (van den Bergh et al., 1997a). The range of tools for subjective quantitative comparative analysis is indeed quite wide. In this sense, it is rather cumbersome to provide a general definition of comparative analysis. However, in the context of this dissertation we use the term 'comparative analysis' in opposition to 'meta-analysis' to indicate research synthesis studies that rely on quantitative, but still not statistical, analytical tools (see Chapter 6). Furthermore, in some ways different from literary reviews and meta-analysis, CA is usually preferred to MA when there is less background material for judging the nature and methods used in earlier studies, or when qualitative as well as quantitative considerations are taken into account (Button, 2002). From a presentational perspective, CA is also powerful when potential users are not familiar with the caveats that surround more statistically rigorous methods (Button, 2002).

Meta-analysis

Meta-analysis (MA) refers to the statistical analysis of a large collection of analysis results from individual studies, for the purpose of integrating research findings (Glass, 1976). Meta-analysis especially focuses on the comparison of the outcomes of previously-performed primary studies by means of statistical techniques (Cooper, 1998; Cooper and Hedges, 1994). It is a statistical approach to reviewing and summarising the literature and a quantitative literature review (Stanley, 2001). More in detail, as defined by Florax et al. (2002): "Meta-analysis is a systematic framework that synthesizes and compares past studies, and extends and re-examines the results of the available data to produce more general results than earlier attempts have been able to do, by focusing on a kernel of previously undertaken research." The MA approach thus offers a series of techniques on measurable phenomena that permits a quantitative, statistical, aggregation of results across different studies (Florax et al., 2002). In doing this, it may help to generate more clearly, for instance, numerical values of the costs and benefits of a given environmental risk from the available data. It can also act as a supplement of more traditional literary-type approaches when reviewing the usefulness of parameters derived from prior studies and help direct new research to related areas. Finally, it may also help to understand the robustness of certain findings by referring to research synthesis as a kind of sensitivity analysis.

It follows that meta-analysis has a wide range of possible relevant applications in economics (van den Bergh and Button, 1999; van den Bergh et al., 1997a) that can be summarised as follows:

- Summarising a collection of similar studies, relationships or indicators;
- Averaging, possibly using weights, by collecting values obtained in similar studies;
- Comparing, evaluating, and ranking studies on the basis of well-defined criteria or goal functions;
- Aggregating studies, by taking complementary results or perspectives;
- Identifying common elements in different studies;
- Comparing outcomes and different methods applied to similar questions;
- Tracing factors that are responsible for differing results across similar studies;
- Performing environmental value and benefit transfer;
- Finding directions for new primary research.

Environmental economic research has contributed a great variety of meta-analytical studies. Examples are: the evaluation of contingent valuation methods for urban pollution (Smith and Huang, 1995; van den Bergh et al., 1997a), congestion (Button and Kerr, 1996), noise nuisance (Button, 1995), and many others. In the context of this dissertation, we employ meta-analysis for tracing factors that are responsible for differing results across similar studies that estimate the WTP for reduced pesticide risk exposure, and for finding directions for new primary research (see Chapter 8).

The different objectives may require different type of techniques. A commonly used method in social science research is *meta-regression* analysis, a statistical technique based on quantitative data that attempts to define the relationship between cause and effect in the problem under investigation. Imagine, for instance, that we have to provide a CBA of a noise reduction programme. We need to estimate what is the value of reducing noise exposure, but the available WTP estimates vary noticeably across available studies. In such a policy problem, meta-regression analysis can be applied to indicate the most reliable estimation of the value of reduced noise and to reach a balanced decision in the present, based upon prior decisions.

Following Florax et al. (2002), the general statistical form of a meta-regression problem can be described as:

$$Y = f(P, X, R, T, L) + \varepsilon \qquad \textbf{Eq- 2-1}$$

where Y is the variable under study also called the *effect size*, say the WTP for reducing rail noise, which has been the focal point of the prior studies under scrutiny; P is what we consider to be the set of determinants of the outcome Y; X represents the characteristics of the set of objects under examination (say, the bunch of primary studies that have provided estimations of the WTPs for reduced noise exposure) affected by P in order to determine the outcome Y; R represents the characteristics of the research methods used in each study (for instance, contingent valuation or choice experiment) and the data (for example, data from in-person or mail surveys); T indicates the time period covered by each study in order to examine the case of time-dependency; L identifies the location in which each study has been carried out; and, finally, ε is the usual error term. Depending on the types of studies considered, all of these variables are supposed to have a relative importance in the analysis. For example, in the field of medical studies where the analyses are experiments in closed and controlled systems, attention is mainly focused on the terms P and X; whereas, in economics, the factors R, T and L may become crucial explanatory variables. Having obtained the regression results, we will have to carry out several tests to verify the accuracy and correctness of our results (for details, see Florax et al., 2002; Bal and Nijkamp, 2001).

Value or benefit transfer

Value or benefit transfer is a technique in which results of studies performed earlier are applied to new policy context (Brouwer, 2000). Three main types of value transfer can be distinguished: a) simple transfer of a mean effect size (for instance a WTP estimate); b) transfer of a demand or bid function (i.e. benefit function transfer); and c) transfer of an estimate based on meta-analysis (Florax et al., 2002). Generally speaking, it is commonly defined as the transposition of monetary environmental values estimated at one site (named the 'study site') through market-based or non-market based economic valuation techniques to another site (named the 'policy site'). In environmental valuation, applying value transfer might therefore be very appealing, as it has the potential to economise on the available knowledge stock (Florax et al., 2002). Imagine, for instance, that we need to provide a monetary estimation of the value of reduced pesticide risk exposure as an input of a broader analysis of the benefits of a given policy on organic farming. It might be the case that the project budget is rather large but nevertheless insufficient to finance a primary stated preference study. On the other hand, imagine that a vast and robust literature on the monetary valuation of the benefits of reducing or eliminating the use of pesticide is available. One might therefore rely on value transfer techniques to infer from previous research findings a reliable estimation of the value of reduced pesticide risk in the new policy site. This is generally very much appreciated by policy makers, since they are usually interested in attaining quantitative policy advice at low cost (Florax et al., 2002). For this reason, within the field of environmental and resource economics, value transfer is now increasingly recognized as a viable technique. Among others, we recall the work by Costanza et al. (1997) on the value of world's ecosystems services and natural capital, which – although severely criticised – has stimulated the discussion about the validity and accuracy of value transfer. The technique is, however, very controversial. A main reason for the scepticism is that the transfer error can be rather substantial (for a discussion, see, e.g., Bergland et al., 2002; Engel, 2002; Brander et al., 2006).

2.4.3. Indicators and indexes

The role of indicators/indexes[20] of environmental risk and impact for evaluative purposes is steadily approaching recognition, if not yet having the status of a fully autonomous discipline (Vismara and Zavatti, 1996). This evolution has been largely driven by increased public awareness of environmental issues, their domestic and international aspects and their linkages with economic and social issues. Over the years, environmental and sectoral policy priorities have evolved, as did the demand for reliable, harmonised and easily understandable information, not only from the environmental community but also from other public authorities, businesses, the general public, environmental NGOs (Non-Governmental Organizations), and other stakeholders. This has stimulated a number of national and international institutions, as well as researchers (e.g. Atkinson et al., 1997; Arrosson, 1997), to produce environmental information that is more responsive to policy needs and public information requirements. The aim is to further strengthen the capacity of countries to monitor and assess environmental conditions and trends so as to increase their accountability and to evaluate how well they are satisfying their domestic objectives and international commitments. In this context, environmental indicators are cost-effective and valuable tools.

Indicators/indexes can be used at local, regional, national and international levels instead of the environmental reporting, measurement of environmental performance, and reporting on progress towards sustainable development. More precisely, reliable, measurable and policy-relevant indicators/indexes can support policy analysis and evaluation and be used in decision making to (Linster, 2006):

- measure environmental progress and performance
- help integrate environmental concerns into sectoral policies (transport, agriculture, tourism, energy, etc.) and monitor policy integration
- allow effective comparisons at different territorial levels (local, regional, national, supra-national)

It is important to recognise, however, that indicators/indexes are not a mechanical measure of environmental performance and state. They need to be complemented with background information, data, analysis and interpretation. One should also note that some issues or topics do not lend themselves to evaluation by quantitative measures or indicators (e.g. ethical positions).

Several classifications of the different types of indicators/indexes are available (e.g. OECD, 1999). We propose a classification that distinguishes four classes of indicators/indexes (see Figure 2-6): i) descriptive indicators/indexes; ii) performance indicators linked to qualitative objectives (aims, goals); iii) performance indicators linked to quantitative objectives (targets, commitments); and iv) risk indicators.

[20] Generally speaking, an indicator can be defined as a parameter, or a value derived from parameters, which points to, provides information about, and describes the state of a phenomenon/environment/area, with a significance extending beyond that directly associated with a parameter value. An index is, instead, a set of aggregated or weighted parameters or indicators.

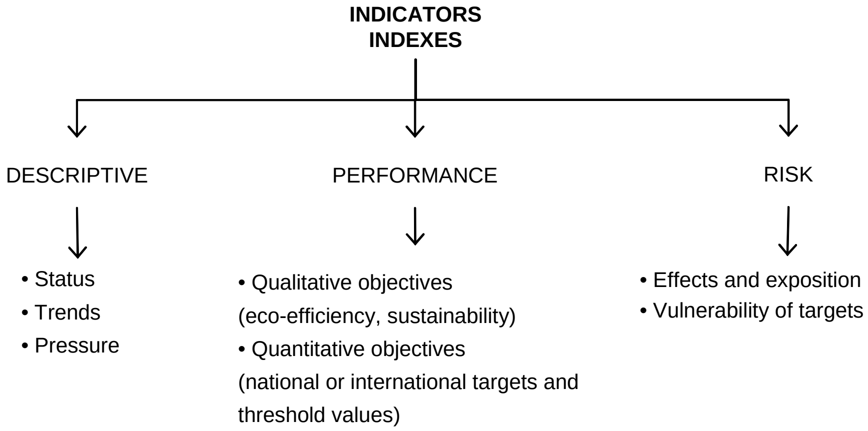

Figure 2-6: A classification of types of environmental indicators

Descriptive indicators can be *status* or *trend* ones, i.e. they provide a picture of the current status (or trend) of a given environmental phenomenon, usually in terms of pressure, e.g. air emission levels/trend, surface and groundwater quality, quantities of agrochemical applied on fields, and so forth. Performance indicators linked to qualitative objectives generally address the concept of performance in two ways: a) with respect to the eco-efficiency of human activities, e.g. emissions per unit of GDP or relative trends of waste generation and GDP growth; and b) with respect to the sustainability of natural resource use, e.g. the intensity of use of forest resources, intensity of the use of water resource, etc. Performance indicators/indexes with reference to quantitative objectives address the concept of performance with respect to national or international targets or threshold values set by regulation, e.g noise emission levels/trends relating to national or international targets, urban air quality relating to national standards, level of pesticide residues in food relating to health threshold levels, etc. Further types are risk indicators/indexes, which are designed on the basis of the information required within environmental risk assessment procedures. Depending on the environmental risk concerned (chemical risk, hydrogeological risk, etc.), the information required to implement the risk indicators will vary. Usually, risk indicators integrate data and information on the potential effects of a given phenomenon (for instance, a chemical), or their likelihood, with some information on the vulnerability of the environmental system or population exposed to it.

For instance, on the first level of assessment, risk indicators may be designed as instruments for predictive risk management approaches, to offer preliminary insights into the *status quo* of environmental risks. They may be developed to obtain baseline information about pesticide use and risks, focusing on one or more realistic hazardous scenarios, and they may assist the identification of potential *trouble spots* and *vulnerable areas* where risk reduction might be necessary.

The previous arguments, though briefly discussed, suggest that challenges for future research in this area are manifold. The first task is striking a compromise between scientific accuracy and decision-making pragmatism. Both experts and managers should be able to give a transparent interpretation of the information provided by indicators. Experts need to accurately interpret, reproduce, and possibly refute results, whereas managers are required to correctly interpret and use outcomes within the decision making process.

2.5. Conclusion

The methods of economic environmental assessment discussed in this chapter all have their established merits concerning the valuation of external costs of mobility and agriculture. However, each approach also has its own theoretical and practical difficulties and, as a result, each can exert a better research and policy contribution, depending on a number of factors. In Table 2-2, each entry shows the potential contribution of different methods for the main possible policy uses identified, and whether they fulfil some standards that are expected to influence their attractiveness for policy application. In particular, we consider: i) research effort required (in terms of intellectual, time, and monetary endeavour); ii) flexibility of the assessment process (i.e. possibility of feedback and 'trial and error' processes, etc.); iii) theoretical robustness versus practical usefulness; and, iv) scientific rigour versus ease of comprehending the results (i.e. whether the approaches are user-friendly or not).

Potential uses of economic assessment in environmental decision making include: cost-benefit analysis (CBA) of policies or projects; pricing policy; design of environmental taxes; national accounting; as a management tool; as a participatory exercise; and, risk/pressure assessment. CBA of projects is the traditional role of environmental valuation, and it remains the context in which stated (and revealed) preference (SP and RP) methods are most used in Europe today (Hanley, 2000). However, recently, changes in EU legislation which mandate some form of environmental appraisal for new policies (e.g. Environmental Impact Assessment, Strategic Impact Assessment, etc.) have increased the usefulness of SP methods within CBA of policies. Furthermore, in the design of pricing policies, for instance, for access to and maintenance of natural resources such as national parks, valuation may be used to elicit the demand curve for the resource and to predict the effect of pricing on behaviour. The connection here arises because SP methods involve seeking the consumer's WTP for the asset. Also, certain techniques: namely, Choice Experiments, allow an estimation of the value of different attributes of the resource in question, enabling resources to be directed most efficiently to maintaining those particular assets (see Chapters 4 and 7). SP techniques might also be applied for designing eco-taxes whereby polluters are charged directly for emitting pollutants. Most of the time these environmental taxes are calculated on the basis of political factors unrelated to their optimal design from an economic point of view. Nonetheless, there is now an increasing trend towards designing taxes so that they reflect the monetary value of the extra damage caused by 1 extra unit of pollution (Pearce and Seccombe-Hett, 2000). This represents an adherence to a general rule for tax design derived from the theory of environmental economics. Moreover, we are witnessing a growing interest in modifying the "national accounts", i.e. the set of accounts that comprise a nation's gross national product (GNP), by also internalising natural stocks and their depreciation due to pollution or other environmental risks[21]. This practice is usually referred to as "green accounting".

All the previously discussed policy purposes might, to some extent, be addressed by research synthesis techniques: namely, meta-analysis and value transfer. These, as said, are techniques that might be used as an alternative to stated preference methods (whenever a suitable body of literature is available) to provide monetary estimations of changes in natural stocks and environmental conditions. Compared with SP methods, however, meta-analysis and value transfer are generally more appealing to policy makers since they might provide quantitative policy advice at low cost. To be fair, however, the considerable research effort required for their application is relevant. The literature retrieval process and the data analysis can be extremely time-consuming, and the robustness of results is not always satisfactory (e.g. Brouwer and Spaninks, 1999). In addition, meta-analysis and value transfer rely on the body of knowledge coming from previously performed studies. Therefore, they do not allow much flexibility in the assessment process. In some cases, for instance, there might be considerable differences in the assets analysed in the literature compared with the new asset of concern, and some compromises may be required. Instead, the major value added of research synthesis comes from its ability to provide statistically robust comparisons of literature results and a systematic framework for establishing 'true' values of environmental goods and services.

Less well understood is the role that monetary valuation (provided with SP techniques) could play in asset management. Valuation indicates the relative strength of WTP for different features of a given asset. Hence, the asset could be managed so as to highlight and expand those features that attract the highest WTP. On the other hand, the contribution of indicators of pressure and risk as management tools appears to be better established. Environmental indicators are currently widely used in environmental reporting, measuring of environmental performance, and monitoring the effects of sectoral policies (transport, agricultural, etc.) and their sustainability. Their capacity to synthesise a vast amount of information in a user-friendly way represents their major comparative advantage for policy applications. Recently, risk indicators based on sound ecotoxicological risk information have also been designed in order to envisage possible future environmental risk scenarios, which draw plausible visions of future environmental conditions (see Chapter 9). In addition, they are appropriate to be used within multicriteria (MCA) approaches.

Finally, SP techniques and (often) multicriteria analysis involve a direct questionnaire approach that allows people to express preferences for or against environmental changes. In addition to the derivation of monetary values for the proposed changes, public participation can help to ensure that the final decision is acceptable to those who are likely to be most affected by it. Valuation also indicates gains and losses to different stakeholders, so that the likelihood for trades between gainers and losers can be identified and managed.

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[21] GNP measures the total flow of goods and services in the economy. Some of this economic activity is taken up with replacing depreciation of assets such as machinery and roads. Hence, only the net national product contributes to average well-being. By the same token, such net measures do not include any depreciation of environmental assets such as coastal zones, rivers, forests, etc. Deducting the monetary value of the damage to the natural assets from the net national product would give a better measure of the "true" level of economic activity.

Table 2-2: Methods of economic environmental analysis and effectiveness for policy making

	Potential for policy use						*Attractiveness for policy use*			
Method	CBA of policies/projects Pricing policy	Eco-taxes	National accounts	Management tool	Participatory exercise	Pressure/ risk assessment	Research effort	Flexibility of assessment process	Theoretical robustness *vs* practical usefulness	Scientific rigour *vs* ease of comprehension
Stated preference	+++	+++	+++	++	++	±	+++	±	+++	++
Research synthesis	++	++	++	±	±	±	+++	±	++	++
Risk/Impact indicators	±	±	++	+++	+++	+++	++	+++	+++	+++

Note: '+++' denotes strong; '++' denotes mild; '±' denotes weak.

PART II: URBAN ENVIRONMENT

3. VALUING TRANSPORT NOISE: AN INTRODUCTORY REVIEW

In many countries, the use of public transport, in particular rail transport systems, is encouraged so as to ameliorate the negative consequences of private mobility. In fact, in the general political intention to shift from more polluting modes of transport to more environmentally-friendly ones, rail transport is assumed to be environmentally friendly. Increasingly, however, railway lines are not acceptable to communities living close to these infrastructures because of concern about high noise levels, which are often over the current cut-off limits set by the international and national legislations. But, in order to stimulate the use of public transport, governments have tended to plan residential areas in the vicinity of rail terminals and infrastructure, while at the same time the accessibility of these areas has increased by expanding the rail network. As a result of this policy, rail noise annoyance has recently become an issue of collective relevance. The European Commission Green Paper "Future Noise Policy" of November 1996 (Com(96)540) states that the public's main criticism of rail transport is the excessive noise level. Excessive levels of noise can potentially lead to both physiological and psychological consequences for people exposed (Miedema and Vos, 1998; Passchier-Vermeer and Passchier, 2000; WHO, 2000). For instance, according to a recent report of the European Commission (CEC, 2003) 10 percent of the European population are affected by rail noise levels higher than 55 L_{dn} dB(A), which is the standard safety level indicated by the World Health Organisation. Moreover, the European Commission "Position Paper on the European Strategies and Priorities for Railways Noise Abatement" (CEC, 2003) underlines that, in order to protect the current population exposed to rail noise pollution, it will be necessary, on average, to reach a noise reduction of 10-15 dB(A).

Thus, railway noise abatement has acquired an important priority on the European environmental policy agenda. There is a high potential for the reduction of railway noise in Europe, because the technical instruments for the abatement of noise are available (CEC, 2003). Nevertheless, in the current EU policy panorama, the main issue is the economically viable implementation of such expensive noise abatement measures, and therefore the choice of the most cost-effective type of possible interventions (see Watkiss, 2001). A crucial question is, then, whether the social benefits of reduced noise can justify the high costs of noise mitigation. Indeed, the implementation of noise abatement measures involves a significant financial cost that is associated either with an investment in the train technology, including wagons and railway tracks, or with an investment in noise barriers (or a combination of both). The effectiveness of the noise abatement will depend on the type of policy intervention adopted, i.e. on the type of noise abatement instrument adopted. Having an economic estimate of social benefits of reduced noise might then allow us to identify the combination of measures providing the highest social benefits per euro of costs, i.e. the highest benefit-cost ratios. In addition, alternative noise mitigation policies will also have different effects in terms of landscape-aesthetics and cost, which should also be considered in order to provide an overall evaluation of the desirability of the possible noise abatement alternatives actually available.

The economic value of changes in noise levels can be elicited by applying environmental valuation methods, thus providing decision support for bodies authorised to plan noise abatement programmes. Both Stated Preference (SP) and Revealed Preference (RP) methods have been used to estimate the economic value of changes in noise exposure by means of the willingness to pay (WTP) concept. The choice between one of these two approaches needs to be motivated depending on a careful consideration of both their pros and cons, and the expected results in terms of theoretical consistency, methodological and estimation robustness, insights for policy. The advantages and disadvantages of RP compared with SP methods are in fact well known and appropriate in their application to noise valuation (for a discussion, see Navrud, 2002 and 2003). These will be reviewed in more detail in the following section.

Before describing the stated preference (SP) survey developed for valuing reduced exposure to rail noise (Chapter 4), the following sections review principles of noise valuation and provide a brief state-of-the-art picture focused on direct environmental valuation approaches.

3.1. Economic background principles

3.1.1. A bottom-up noise valuation approach

Transport noise can be considered a source of environmental pollution, which depends on the site and technology-specific characteristics (receptors, buildings, vehicle technology, traffic situation, etc.). Therefore, as with the framework of reference for the economic valuation of other form of pollution (due to chemicals, GMOs, and so forth) an impact pathway approach need to be used for the quantification of costs due to noise exposure (Droste-Franke et al., 2006; Navrud, 2002). Figure 3-1 presents a typical bottom-up valuation framework which can be employed to provide empirical estimates of the monetary value of declines in the levels of noise exposure.

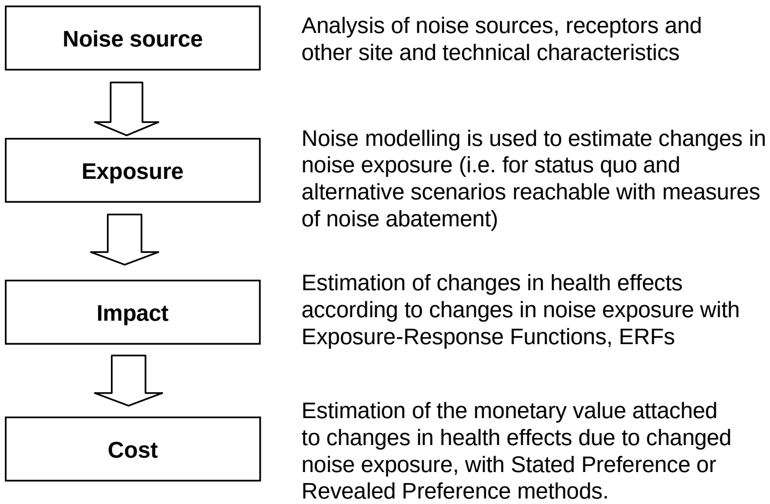

Figure 3-1: Noise valuation framework

Irrespective of the source of environmental noise being considered (road traffic, railways, aircraft, etc.), the noise valuation process starts with the assessment of the potential detrimental effects of exposure to excessive noise levels[22]. Noise modelling and simulations are used to estimate the variation of exposure to noise at different locations and receptors. The spatial distribution of noise levels can be graphically represented by creating maps, within a Geographical Information System (GIS), to illustrate the level of exposure at different receptors. Noise maps[23] enable the economic actors exposed to excessive noise to be identified, and help to select the study area. Maps are created to illustrate the distribution of noise over the study area, and its possible changes from the *status quo* situation. A typical noise valuation exercise in fact can consider the economic welfare effects of changes in noise levels, which can be achieved by implementing either technical measures (car and engine design, low noise tyre, track maintenance, etc.) or non-technical ones (traffic calming, taxation, travel substitution, etc.).

Second, any change in noise exposure needs to be linked to the related change in the level of any given effect caused by noise. With this purpose, exposure response functions (ERFs) have been derived to estimate the correlation between a change in noise exposure and a change in the level of the effect caused by noise (Miedema and Vos, 1998; de Kluizenaar et al., 2001). Experts (e.g. Passchier-Vermeer and Passchier, 2000) report a variety of risks due to noise exposure. These range from physiological disorders, to psychosociological effects (such as annoyance), to pathological and manifest disorders. Pathological and manifest disorders refer to the welfare impacts associated with hearing loss, and stress-

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[22] Or, vice versa, it is subordinated to the assessment of the potential beneficial effects of reduced exposure to noise.

[23] Noise maps represent the noise distribution in space, with different colours corresponding to different noise levels. Noise levels are usually measured in decibels (dB(A)) and with noise indicators such as L_{dn} (see Footnote 3).

related health effects, such as hypertension and the risk of cardiovascular diseases. These effects have been mainly mapped in industrial contexts, where people are exposed to high noise levels over a long and continuous period of time. Recently, however, stress-related health effects have also been monitored in association with exposure to environmental noise (e.g. de Kluizenaar et al. 2001). On the other hand, annoyance refers to a general feeling of displeasure or discomfort and it is mainly mapped in contexts where people are exposed to environmental noise, such as road and rail noise emissions. Annoyance is considered the main type of welfare impact associated to noise caused by traffic and trains (Navrud, 2002).

Third, once noise impacts have been assessed, appropriate monetary values are needed to derive costs of noise pollution. In fact, changes of noise imply changes of welfare that need to be translated into monetary terms. Information on the change in noise impact is therefore used to apply direct or indirect economic valuation techniques and set an economic value for a unit of each end-point taken into consideration (for instance: annoyance, sleep disturbance, hypertension, and so forth). So far, most of the economic valuation literature has focused on annoyance, because of a general lack of epidemiological studies providing evidence on the real health effects of environmental noise. Now, however, there is evidence supporting health impacts in association with environmental noise (e.g. de Kluizenaar et al., 2001), and real health impact will receive more attention.

3.1.2. Capturing noise effect on welfare

According to Metroeconomica (2001) and Hunt (2001), the costs of noise pollution produce three main components of welfare change: a) resource costs, i.e. medical costs paid by the health service in a given country or covered by insurance, and any other personal expenses made by the individual (or family), such as noise insulation costs; b) opportunity costs, i.e. cost in terms of lost productivity (or performing at less that full capacity) and the opportunity costs of leisure, including non-paid work; c) disutility, i.e. other social and economic costs including any reduced or limited enjoyment of leisure activities. According to the recent literature supporting noise health effects, disutility costs should include not only psycho-sociological effects (i.e. annoyance) and sleep disturbance, but also stress-related health effects[24]. In this respect, problems of double counting should be properly addressed, in order to completely separate the effect of simple annoyance and the effect of real health problems on individual utility.

Components a) and b) can be proxied using market prices, and they can be aggregated to provide the so-called 'cost-of-illness' (COI) measure of welfare. However, the latter component needs to be estimated too, since the values of disutility are expected to be much larger than the cost of illness (see Droste-Franke et al., 2006). However, it is clear that the latter component is not captured by the COI approach, and needs to be estimated with a measure of the individual loss of utility. Estimations are therefore required of the willingness-to-pay (WTP) to compensate for the loss of the welfare associated with noise annoyance.

The economic benefits of a reduction of noise levels might be estimated by looking at people's economic behaviour and their preferences, either indirectly or

[24] According to a recent study by Droste-Franke et al. (2006), the cost of real health impacts are about 10 percent lower than the costs of annoyance as calculated by hedonic models.

directly. So far, the literature on noise has been largely dominated by the use of indirect valuation methods (RP) and, in particular, Hedonic Price (HP) techniques. The HP approach to valuing local noise externalities is based on the examination of their impact on property values. Differences in property values due to noise pollution are observed, and it is tested whether property prices decrease as noise levels increase. The main strength of HP techniques is that they rely on actual behaviour in the housing market, where individual preferences for noise and other environmental characteristics can be observed, though indirectly. A major drawback is that the results of HP studies are sensitive to modelling specifications and the condition of the local housing markets, since property values and noise levels often correlate with many contingent factors (for a meta-analysis of airport noise, see Nelson, 2004; Schipper et al., 2001). Additionally, HP techniques are not able to capture non-use values or non-use welfare impacts attached to noise level changes (e.g. reduced enjoyment of desired leisure activities, discomfort or inconvenience, anxiety, concern and inconvenience to family members and others)[25]. In this connection, survey research is important for measuring attitudes towards noxious activities, and to provide quantifications of changes in non-use values related to changed noise exposure. So far, however, stated preference (SP) has been a less frequently-used approach to value noise (see Navrud 2002, 2003). Relatively few SP valuation studies on noise are available and most of these have been conducted over the last 5-10 years, following the trend in the methodological valuation literature regarding the increasing use of SP methods for environmental valuation. The following state-of-the-art description concentrates on SP approaches, putting emphasis on some open research challenges in this field, which have so far induced researchers to refrain from using SP approaches for noise valuation, such as: how to tackle the issue of respondents' subjective noise perception; how to select the proper payment vehicle; how to control for the complexity of valuation scenarios in a survey framework; and, finally, whether the available literature on noise valuation would enable value transfer approaches to be applied.

3.2. An introductory review

3.2.1. Transport noise valuation

In the context of the valuation of transport environmental externalities, several SP studies have been conducted to estimate the costs of noise. These provide a valuation of noise costs in different situations: road traffic noise (e.g. Arsenio et al., 2001; Garrod et al., 2002); air traffic noise (e.g. Baarsma, 2001); a range of impacts including traffic noise (e.g. Daniel and Hensher, 2000); and the intrusion effects of transport (e.g. Eliasson et al., 2002); and both noise and air quality (e.g. Sælensminde, 1999; Hunt, 2001).

Only two original valuation studies focus on rail noise, both of them using an HP approach, while SP approaches to rail noise valuation are even less applied.

[25] Navrud (2002) notices that, among RP methods, the avoidance costs approach has also been applied to noise, but the applicability of this method is severely reduced because only in certain circumstances can the results be interpreted as a proxy of welfare loss.

Strand and Vågnes (2001) use both HP and a Delphi study of real estate brokers to value rail noise in the Oslo region. Holsman and Paparoulas (1982) use an HP approach to estimate for Australia that the occurrence of rail noise, in areas with no benefits from increased accessibility, reduces property prices by 10 percent. A CV survey by Navrud (2000) includes rail noise among noise annoyance and exposure questions but does not provide an estimation of WTP for rail noise abatement. More recently, Droste-Franke et al. (2006) estimate the costs of noise exposure due to road traffic, trains and aviation, using monetary values for noise impacts derived from their own calculations based on Metroeconomica (2001), Navrud, (2003), and Nellthorp et al. (2001). Thus, the study presented in Chapter 4 is, to our knowledge, the first study using a Choice Experiment (CE) approach for valuing rail noise. Among the existing SP studies, the majority of these refer to road traffic noise abatement programmes and, to a lesser extent, aircraft noise as experienced inside the dwellings. Working on the valuation of road noise, major efforts have been oriented to the use of Contingent Valuation (CV) (e.g. Pommerehne, 1998; Soguel,1994; Vainio, 1995, 2001; Barreiro et al., 2000; Navrud, 2000; Lambert et al., 2001), while only Sælensminde (1999) and Thune-Larsen (1995) have applied the CE approach. Similarly, CV dominates in the valuation of aircraft noise, where the sole CE study is the one by Thune-Larsen (1995).

According to Navrud (2002), the scarce use of SC methods so far is, to some extent, explained by complications in designing noise valuation surveys able to tackle the complexity of this environmental phenomenon, without increasing respondents' cognitive effort too much. First, as already stated, in aiming to estimate the value of noise annoyance reduction, complexity in noise valuation comes, primarily, from difficulties in understanding how individuals actually perceive different levels of noise exposure, and how they react to them. To what extent do individuals exposed to excessive noise levels experience a feeling of resentment, displeasure, discomfort, dissatisfaction or offence? How sensitive are individuals to a given noise exposure level? In an SP survey, the somewhat vagueness of the noise annoyance definition complicates the task of researchers in explaining to respondents, as clearly as possible, what exactly is the good being valued. Similarly, uncertainty in noise perception can hamper the interpretation of WTP estimates and welfare analysis. Secondly, complexity in noise valuation stems from the wide-ranging relevant information that should be provided to respondents when they are asked about their preferences for alternative noise mitigation scenarios. In this connection, it is well known that, in CV surveys, researchers have to select only the amount of relevant information that is strictly necessary for the respondents. In some cases, this level of simplification affects the possibility to provide both an exhaustive description of possible alternative policies to respondents and, therefore, a sound analysis of individual preferences. Thirdly, another relevant issue in this literature is the need for realistic and fair payment vehicles. This will contribute, *inter alia*, to minimise a potential low response rate, as often observed in CV studies on noise (e.g. Lambert et al., 2001). Navrud (2002) notes that the most appropriate payment vehicle could differ according to different noise sources and different countries with heterogeneous institutional settings, cultures and preferences. For instance, respondents are more likely to protest against payments to reduce, say, aircraft or industrial noise, for which they are not directly responsible, than to protest against payments for reducing road traffic noise, for which very often they are actually responsible. In the study presented in Chapter 4, an innovative payment vehicle has been used based on the trading tax schemes as originally proposed by Bergstrom et. al (2004) for the contingent valuation setting (for further details, see Nunes and Travisi, 2006a).

Related to the issue of whether respondents can accurately perceive the noise reduction proposed during surveys, interestingly, in the majority of cases noise change is described in terms of a percentage reduction in noise levels (typically a 50% reduction). Some authors, including Sælensminde and Hammer (1994), criticise this approach because, most of the time, there is no additional effort on checking whether respondents actually understand what this reduction in noise would mean to them. More recently, some Contingent Valuation (CV) studies have instead provided respondents with more accurate descriptions of the noise reduction in terms that can be better understood. Some of these have put forward original approaches in the way researchers link the change in noise level to the personal day-to-day noise experience of respondents. Among these, Barreiro et al. (2000) describe the noise reduction by referring to the road noise levels respondents experience at different times on different weekdays ("daytime noise would be reduced from the working day level to that of a Sunday morning"). Vainio (1995, 2001) uses a CV scenario of diverting traffic elsewhere or into a tunnel so that "traffic volume can diminish considerably" on the street the respondents identified as causing them the most noise nuisance. Navrud (2000) and Lambert et al. (2001) both describe the noise reduction explicitly in terms of *annoyance*[26]. Lambert et al. (2001) use a 5-point Likert scale to capture the respondents' level of annoyance, but do not provide details on the way they specify what various levels of annoyance mean to them. Navrud (2000), instead, provides the respondents with a detailed list of avoided impacts in terms of discomfort, including sleep disturbance. This approach has the advantage that, if respondents are asked about their current level of noise annoyance, economic estimates per person annoyed per year for different noise annoyance levels can be estimated. Nevertheless, a drawback is that the meanings of lower/higher level of annoyance still remains subjective, and researchers have to handle a substantial rate of uncertainty, whenever they try to link annoyance reduction to decibel reduction. Ideally, this problem might be overcome by monitoring the actual level of noise exposure at each respondent's home, and then asking the respondents their level of annoyance. In this context, the present study employs the use of a valuation survey that provides respondents with some examples of noisy situations in daily life (a bar, a loud office, traffic or train nuisance, and so forth) and acoustic maps of exposure to rail noise, linking noise annoyance to the distance of the respondent's home from the railway track. Finally, in the valuation section, noise abatement is portrayed both in terms of decibel reduction and its equivalent measure, the distance from the source.

In order to improve survey design with a proper description of the good being valued, Choice Experiment (CE) methodologies can be used either to complement or to substitute CV surveys, especially when the intent is to provide decision makers with insights for the definition of environmental policies and actions. The advantage is that CE treats the public policy programme, such as a noise abatement programme, as a bundle of attributes and derives the marginal valuation of each attribute separately. It therefore allows substantially more information to be provided about the range of possible alternative noise policies, as

[26] In this context, the term 'annoyance' is more specifically intended to be associated with disturbance of activities, sleep, communication, and to cognitive and emotional responses such as anxiety, irritability and nervousness (WHO, 2000).

well as reducing the sample size needed compared with CV, and, in addition, it can reduce the risk of aggregation bias and double counting. Since the purpose of the study is less obvious to respondents, a lower incentive to strategic bias is also expected (for a discussion see Wardman and Whelar, 2001). On the other hand, survey design issues with the CE approach are characterised by a higher complexity due to the multiple numbers of attributes that must be presented to respondents and the relatively strong requirements for the econometric methods used to analyse the survey data. Nevertheless, although CE does not dominate CV from a theoretical perspective, we regard the former to be, on balance, preferable for application to noise valuation (see Chapter 4).

3.2.2. Meta-analysis in noise valuation

Given the high costs of conducting primary studies for the estimations of the benefits from reductions in noise levels and, moreover, given the need to establish interim values for noise from different transport modes (air, road, rail) to be used in cost-benefit analyses performed by the European Commission, the possibility to use meta-analysis and value transfer techniques is receiving increased attention. As argued in Chapter 2, meta-analysis and value transfer have the potential for predicting monetary values from completed valuation studies for use in new study settings, but they can be accepted only after a close review of the existing available studies. To our knowledge, two formal meta-analyses of HP studies on aircraft (Nelson, 2004; Schipper et al., 2001) and one on traffic noise (Bertrand, 1997) are available, whereas formal meta-analysis on direct valuation studies have not been performed.

Table 3-1 provides a complete overview of available estimates of reduced noise annoyance, according to the type of transport mode and the valuation method. Noise estimates are reported as willingness-to-pay (WTP) per decibel of noise reduction per household, per year. The results show that the monetary value for noise annoyance (expressed in euros2006 per decibel per household per year) ranges from about 2 to 110 euros, for road noise, and from about 9 to 1066 euros, for aircraft noise. According to the non-market valuation literature on externalities, potential relevant explanatory factors – that can be anchored on the econometric, methodological and survey design strategy – may explain these variations (see Florax et al., 2005; de Blaeij et al., 2003). In addition, we can observe that Table 3-1 provides a limited amount of information in non-market valuation studies on rail noise. As a matter of fact, we are aware of a single non-market valuation study that focuses on rail noise abatement (Weinberger et al., 1991). More recently, a market valuation study has addressed the monetary valuation of rail noise exploring the use of the hedonic price method (Clark, 2006). Clark studies the dynamics of residential property markets in three counties in Ohio. In particular, the author investigates the empirical magnitude of the impact of rail noise on the prices of houses. According to the results, rail noise is responsible for a negative, and statistically significant, effect on property prices. This magnitude ranges between 6.3 to 31.9 percent. Other hedonic price studies report that rail noise reduces property prices by 2 to 4 percent, according to Simons (2004), and by 10 percent, according Strand and Vågnes (2001). Against this background, Chapter 4 proposes to carry out a monetary assessment of rail noise annoyance using the stated choice method. For this reason this study is a two-fold novelty. First, it constitutes one of the first non-market valuation studies on rail noise. Second, it constitutes the first empirical research on noise annoyance in Italy caused by railroad traffic.

As discussed in the valuation and meta-analytic literature, the observed systematic heterogeneity across WTP estimates for an environmental or health improvement might be explained by differences in several theoretical, methodological or contextual features. Potential relevant explanatory factors, named moderator variables (Sutton et al., 2000), can be derived from three different sources. Theoretical model and previous meta-analyses of health risk provide evidence of WTP-damage and WTP-risk trade offs (e.g. Florax et al., 2005; de Blaeij et al., 2003). Similarly, factors related to the study design process, pertaining either to methodological issues or to the specific study setting may induce systematic variation. The results presented in Chapter 8, for instance, show that WTP values for reduced pesticide risk exposure are sensitive to the level of exposure, geographical location, income elasticity, and to other pivotal features of research design (specifically, the type of survey and type of safety device).

For meta-analysis and value transfer approaches in noise valuation, a number of relevant questions can be raised. Elements such as the initial noise level, the type of noise (either from road, aircraft or trains), the noise valuation unit used and other methodological differences – such as the valuation approach employed – are in fact expected to influence the benefit estimation in predictable ways.

The first relevant question related to transport noise is whether the same WTP values can be applied to noise from different transportation modes. The noise literature in fact reports that some types of noise events are more disturbing than others. For instance, infrequent and sudden events associated with a very high noise level (such as aircraft noise) are considered to be more disturbing than a continuous background noise (such as traffic noise), though the latter can be associated with reduced concentration and fatigue while studying or working (e.g. de Kluizenaar et al., 2001). Moreover, single-tone component noise (such as rail noise) is more disturbing than noise over a wide spectrum (e.g. loud music). Similarly, sudden increases in noise levels (such as a loud horn) are more disturbing than a constant mono-tone noise level (e.g. fans, ventilation system, etc.). Thus, the same noise level for different sources is expected to give perhaps very different levels of annoyance. This suggests, therefore, that the source of transport noise can play a role in explaining differences across monetary values estimates. In this connection, the results from Bateman et al. (2000) show that reductions in aircraft noise are valued more than equal reductions in road traffic noise. Also, in a recent study by Droste-Franke et al. (2006) estimating noise costs for different travel modes, in the absence of no specific Exposure Response Functions (ERFs) for rail transport, a bonus of 5 decibels was applied to railway noise, in order to take into account that rail noise is perceived as less annoying than road noise. This sort of correction is also applied by several noise regulations in a number of European countries (e.g. France, Germany, Italy, see, e.g., CEC 2003).

Moreover, the typical unit of measure for noise, L_{dn}[27] adjusts for different distributions of noise over time, but does not correct for the composition of noise, which, however, is expected to play a role in explaining differences in respondents'

[27] For a definition, see footnote 3.

noise nuisance (for a discussion, see Navrud, 2002). In fact, differences in noise annoyance due to different transport mode are reflected in ERFs. It follows that, those circumstances where individuals are affected by multiple sources of noise should be properly considered, not only to take into account all possible sources of estimation variability in meta-analytical approaches, but mainly to plan adequate and effective noise reduction actions in concrete policy contexts.

Another relevant issue is related to the study setting and the interest group considered during the valuation study. Estimations from SP or HP studies are in fact expected to vary not only according to socio-economic groupings (income groups, gender, age, education level), but also according to the specific noise exposure conditions at the study site. For instance, the level of annoyance from the same level of noise emission outside a dwelling can differ across countries, cities or quarters. This variation depends on several factors, such as different building features (brick, concrete) and climates (insulation or double glazing that protects both against low temperatures and noise); and rate of time spent home during the week and the weekend. This is why some national noise regulations set noise limits, not at the source, but at the receptor (e.g. in Italy, noise levels are monitored at the receptor, both inside and outside the dwelling).

Income level is also expected to influence WTP estimations, as suggested by economic theory, although little empirical evidence is available in this connection. According to Navrud (2002), there seems to be some empirical support for a lower value of noise in accession countries, as well as in EU countries with an average income below the EU15 level. On the other hand, whilst WTP varies theoretically according to income level, a number of cultural and social factors are expected to influence respondents' willingness-to-pay in SP studies, which taken together might influence WTPs even more than income.

Moreover, even though there are an increasing number of SP studies on traffic noise, they present WTP in various units of measures. This entails an important operational problem in meta-analysis because, in order to make the studies comparable, these results must be transformed to a common measurement unit. Noise estimates are usually reported as: i) economic values per annoyed person per year (with separate values for different annoyance levels); or ii) economic values per decibel of noise reduction per person (or household) per year. Navrud (2002) states that the former unit of measure should be the preferred one, since the marginal values required to apply benefit transfer are directly elicited from SP studies, which contain questions about the respondents' current level of annoyance. This approach would eliminate the need for many, and often unrealistic, assumptions to construct marginal values per decibel from estimates for discrete changes measured in noise levels and noise annoyance. In addition, values in euros per annoyed person per year are expected to be more stable across space and time, and easier to adjust in meta-analysis and value transfer exercises, since they are directly based on a measure of individual preferences, rather than on the indirect technical measure of noise (i.e. decibels). Moreover, annoyance-based units of value would allow the same value for different noise sources to be used, while noise exposure-based estimates would have to be diverse for different noise sources in order to correct for their dissimilar characteristics and level of annoyance at the same dB level. Nevertheless, other issues support the use of noise exposure-based values. First, only a few SP studies report WTPs per annoyed person per year, and alternative ii) might be considered to provide interim values. Second, in addition annoyance-based values might require many assumptions to estimate the overall benefit of a given discrete reduction in noise (that usually requires the use of well-established ERFs). During surveys, therefore, an in-depth analysis of the respondents' profiles should be strongly recommended to reduce the sources of error in the estimation of values for different annoyance levels. One possible way of overcoming such drawbacks might be combining dB and annoyance measures in the survey design. This is the approach used in the noise valuation study presented in Chapter 4[28].

On the whole, the aforementioned factors of heterogeneity across noise condition and valuation results complicate the efforts towards the estimation of 'universal' values of noise changes to be used in actual policy contexts for different types of noise sources and end-points. More primary studies are needed, especially for those transport noise sources, such as rail, for which the empirical evidence is close to non-existent.

[28] Respondents are asked about their current level of annoyance (using a 5-point Likert scale) and checked to infer their profile of sensitivity to noise. Visual aids are used to improve the respondents' comprehension of the proposed changes in noise (reported in dB).

Table 3-1: Complete overview of results from Stated Preference (SP) studies

Study	*Data*	*Country*	*Issue*	*ΔdB*	*Method*	*Travel mode*	*WTP (Euro2006)*
Weinberger et al., 1991	1990	Germany	Elimination of noise annoyance	--	CV	Railway	89.16-108.94
Arsenio et al., 2001	1999	Portugal	Avoiding a doubling of the noise level	--	SC	Road	55.69
Barreiro et al., 2000	1999	Spain	Elimination of noise annoyance	--	CV	Road	2.22-3.34
Lambert et al., 2001	2000	France	Elimination of noise annoyance	--	CV	Road	7.78
Navrud, 1997	1996	Norway	Elimination of noise annoyance	--	CV	Road	2.22
Navrud, 2000	1999	Norway	Elimination of noise annoyance	--	CV	Road	25.57-35.57
Pommerehne, 1998	1998	Switzerland	50% reduction in experienced noise	-8	CV	Road	110.06
Saelensminde and Hammer, 1994 Saelensminde, 1999	1993	Norway	50% reduction in experienced noise	-8	CV/SC	Road	52.25-107.83
Soguel, 1994	1993	Switzerland	50% reduction in experienced noise	-8	CV	Road	66.70-78.93
Thune-Larsen, 1995	1993	Norway	50% reduction in experienced noise	-8	CV	Road	21.12
Vainio, 1995, 2001	1993	Finland	Elimination of noise annoyance	--	CV	Road	6.67-8.89
Wibe, 1997	1995	Sweden	Elimination of noise annoyance	--	CV	Road	31.13
Wardman and Bristow, 2004	1996	UK	50% reduction in experienced noise	-10	CV/SC	Road	26.68-40.02
Faburel, 2001	1999	France	Elimination of noise annoyance	--	CV	Aircraft	8.89
Pommerehne, 1998	1988	Switzerland	50% reduction in experienced noise	-8	CV	Aircraft	47.80
Thune-Larsen, 1995	1994	Norway	50% reduction in experienced noise	-8	CV, SC	Aircraft	211.22-1066.10

Note: The results are taken from Navrud (2002) and Brons et al. (2003) and are expressed in euros$_{2006}$ per decibel per household per year, after conversion of the original estimate in the national currency in the year of study.

4. VALUING ALTERNATIVE URBAN RAIL NOISE ABATEMENT PLANS: A CHOICE EXPERIMENT VALUATION STUDY IN ITALY*

Like transport emissions (CO_2, NO_x, PM_{10}, etc.), transport noise is typically an externality with a large number of victims, and sometimes with a large number of polluters, for instance in the case of road transport. Transport noise, which is one of the major sources of noise in the environment (WHO, 2000), is largely produced by mobile sources in well-defined locations, such as high-density streets, highways, and areas around railroad tracks. Therefore, noise pollution is not distributed uniformly over space; instead, it is a localised externality. The localised nature of the noise problem is particularly clear in the case of rail noise, where only noise emitted close to the railway line is considered a source of welfare disruption. The aim of this chapter is to establish marginal values with respect to noise nuisance suffered by the exposed population living in the vicinity of a rail track. The aesthetic and environmental consequences of alternative noise mitigation actions are also addressed. The case study presented is based in the north of Italy, in the province of Trento.

Since 1998 in Italy rail noise pollution has been regulated by a detailed, legal act[29] that sets daytime and night-time limits on receptors, depending on their vulnerability and distance from the railway. Residential areas or vulnerable receptors, such as schools and hospitals, have therefore lower limits than less vulnerable ones. Reception limits refer to a precise spatial area along the railway, which includes receptors within 250 metres from the railway. This area is divided into two parts, named "Zone A" and "Zone B", respectively, 100 and 150 metres away from the rail track, and characterised by different noise reception limits.

However, almost one decade after the introduction of the Italian national noise regulation, the implementation of the required noise abatement measures is still largely incomplete, and only very recently have we witnessed the rise of a national debate on how to proceed in order to abate rail noise below the unacceptable limits. The Brennero railway, which is located in the north-east of Italy in the province of Trento, is one of the first examples in Italy for which rail noise abatement plans are currently under analysis. For this reason, there are now challenging questions and new opportunities to provide policy makers with relevant insights on the best option to be developed against rail noise. Important issues here concern: how to accelerate the implementation of the noise abatement

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* *Based on Nunes and Travisi (2006a, b).*

[29] In Italy, the overall noise regulation was set in place in 1995 (LGQ n. 447/1995). The regulation on rail noise is instead more recent as it was set in 1998 (DPR n. 459/1998 "Regolamento recante norme di esecuzione dell'articolo 11 della legge 26 ottobre 1995, in materia di inquinamento acustico ferroviario").

regulation; how to choose, among the range of possible noise reduction measures, those actions able to provide the highest level of acoustic efficiency at the lowest collective cost (i.e. by looking at the advantages and disadvantages of each possible action for the whole range of stakeholders involved).

In this context, the present chapter examines the use of a Choice Experiment (CE) methodology to assess the economic value of alternative rail noise reduction policy interventions, and the respective instruments. The CE survey was held in Trento in order to assess the marginal WTP for different attributes including noise reduction, aesthetics, environmental and technical attributes with respect to alternative railway plans on the Brennero railway. This allows us to study in detail the potential sensitivity of a set of factors that were identified in meetings with experts as influencing rail noise mitigation plans, including the level of abatement and the respective types of intervention, landscape aesthetics, and the type of financing proposed.

The chapter is organised as follows. Section 4.2 describes the valuation method applied, while in Section 4.3 the survey instrument is developed, and the in-person interviews conducted on a sample of 511 residents exposed to noise pollution in the province of Trento, Italy, are described. Section 4.4 presents the modelling and valuation specifications, while Section 4.5 discusses the range of the economic estimates and evaluates these for different payment scenarios. Section 4.6 provides welfare analysis and policy discussion, and Section 4.7 concludes.

4.1. Valuation method

The economic valuation of rail noise externalities refers mainly to the monetary assessment of welfare losses due to annoyance (see Brons et al., 2003). An accurate, complete and reliable monetary assessment of rail noise externalities requires the application of specific non-market valuation tools, including both contingent valuation and stated choice methods – see Navrud (2002, 2003). Contingent valuation is a survey-based approach that directly estimates the preferences for noise reductions. In short, respondents are asked to express their maximum WTP for one specific change in noise annoyance, as described in the survey. Alternatively, Choice Experiment (CE) confronts respondents with a set of two or more survey described policy alternatives, which differ in terms of their respective attributes and attribute's levels. The respondents are asked to choose their preferred option. For this reason, the CE method brings along with it the advantage that multiple noise reduction policies, which are expressed in terms of different bundles of attributes, can be simultaneously evaluated even if they have not yet been adopted (*ex ante* valuation) or lie outside the current institutional arrangements.

4.1.1. Features of choice experiment

Choice Experiment (CE) is a direct valuation method that can be applied to noise valuation so as to infer people's preferences for a set of alternative noise abatement programmes. Alternatives are presented to the respondents, who are asked to choose the most preferred one. These are described as bundles of factors, known as "attributes", which are expected to impact respondents' preferences for the proposed options. This is called the "choice set". This framework enables us to observe the choice behaviour of the respondents, in order to examine the effect of the attributes that influence preferences and the relative importance of each attribute (for more details, see Chapter 2).

CE, similarly to dichotomous choice contingent valuation methods, which can be considered as a special case of CE, presents some attractive features as a technique for evaluation. First, since choice behaviour is observed in daily life, typically in the form of shopping, the respondents answer the CE questions more easily than other stated preference (SP) techniques, such as rating, ranking, and pairwise techniques, which in contrast do not involve any choice behaviour in decision-making. Second, we can use the hypothetical goods or polices as alternatives so that the respondents' preferences for those goods and policies can be analysed. In the present case, for instance, we can use different rail noise reduction policies as alternatives. This is a valuable improvement over a revealed preference (RP) method such as the hedonic price approach, where the range of noise reduction is usually not clearly measurable and irrelevant to policy. In addition, we can calculate the WTP for noise decrease based on the preferences of a selected sample, whereas it is the householders' preferences that are usually elicited in the hedonic price approach. Moreover, CE has an attractive advantage over CV. A typical noise abatement policy involves various aspects that can have a significant impact on peoples' well-being. What type of noise is targeted by the policy? What level of noise reduction does the policy grant? When and at what cost will be the policy implemented? CE can separately estimate preferences of individuals for these aspects. On the other hand, CV mainly focuses on the valuation of only one aspect or one fixed set of aspects.

Despite these advantages, to our knowledge, CE has not been used so far in the field of the economic valuation of rail noise reduction. We therefore try to explore how CE can be used in eliciting people's WTP for reductions in train noise pollution, and how it can become an important analytical tool in the field of value measurement and welfare analysis of alternative public noise abatement programmes. This exercise will be discussed in detail in Section 4.3. Now, however, in the next subsections we model respondents' behaviour with respect to noise abatement, presenting the cornerstone for any welfare assessment.

4.1.2. Modelling individual respondents' behaviour with respect to noise reduction

This section presents the theoretical framework of the econometric models applied in our noise survey. The statistical analysis of stated preference data is based on the Random Utility Model, which is assumed for the utility of the individual (McFadden, 1974, 1986). When the individual q chooses the noise alternative i, we assume that its utility can be modelled as follows:

$$U_{iq} = V_{iq}(\bar{x}_{iq}) + \varepsilon_{iq} \qquad \textbf{Eq- 4-1}$$

U_{iq} is determined by two components. V_{iq} is the deterministic component, which is a function of the attribute vector $\bar{x}_{iq}$ of the noise management alternative

I, and it can be interpreted as the indirect utility function; and ε_{iq}, is the random part, which is unobservable directly for the researchers.

In our survey, we posit that in each of the choice sets, the respondent selects the noise management alternative with the highest indirect utility. The CE exercise implies a choice between two alternative noise management policies (to be provided by the local administration), each of which can imply different technical measures for noise abatement (e.g. double windows, barriers, etc.), and provide different levels of noise reduction. But, noise policies are costly to the local public administration, and the implementation of one of the possible noise programmes needs to be financed by contributions form citizens. In this connection, the local administration can consider alternative project financing strategies.

Therefore, the noise policies vary with respect to level of noise reduction, type of technical measure, cost to the respondent, and type of financing (see Table 4-2). We assume that the utility function of alternative i for respondent q is:

$$V_{iq} = \bar{x}_{iq}\beta + \bar{z}_{iq}\delta + \varepsilon_{hq} \qquad \text{Eq- 4-2}$$

where q denotes the respondent; i denotes the alternative noise policy; $\bar{x}$ is a vector comprised of the policy attributes; and $\bar{z}$ is a vector of interactions between the attributes and the individual characteristics of the respondent. β and δ are vectors of unknown coefficients. If the error terms ε are independent and identically distributed and follow the type I extreme value distribution, the probability that alternative i is selected out of S alternatives is:

$$P_{iq} = \frac{\exp(w_{iq}\gamma)}{\sum_{j=1}^{J}\exp(w_{jq}\gamma)} \quad \forall i,j \in S \text{ with } i \neq j, \qquad \text{Eq- 4-3}$$

where $\bar{w}$ is a vector containing $\bar{x}$ and $\bar{z}$, and $\gamma = [\beta' \vdots \delta']$. Depending on the assumption about the distribution of the error term, the resulting statistical model is either a conditional logit, a multinomial probit, or a related choice model (Green, 2002). The implicit marginal price of each attribute and the welfare changes associated with changes in the level of the attributes can then easily be derived. Assuming the error terms ε_{iq} and ε_{jq} follow a Gumbel distribution with scale parameter λ (McFadden, 1974), usually standardized to 1, P_{iq} follows a Conditional Logit (MNL) model.

We are now ready to apply this model to predict consumer choice behaviour regarding alternative options to reduce rail noise emission levels. First, however, we need to identify and measure other noise-abatement-related attributes that, together with the price, characterise the utility function of the respondent. This constitutes an important task in our empirical work, and it will be discussed in detail in the following section.

4.2. The CE survey on rail noise abatement

4.2.1. Background and the political debate

As mentioned before, the reduction of railway noise reception levels can be achieved by basic essential types of measure: at the source, including train vehicles and tracks; in the sound propagation path; or at the receptor. In the past, the latter type of measure was most common. As current practice in Europe, measures such as barriers (with high cost) or sound insulation windows (with limited effect) are mostly chosen instead of more cost-effective source-related measures. The reason for this is complicated and involves several issues. First, the sound propagation measures were normally taken because of noise reception limits which have to be observed locally, whereas vehicles are often of broader origin and beyond the influence of the local authorities. Secondly, vehicle emission limits, which could enforce measures on the rolling stock, exist only in a few countries, whereas the application of traditional barriers and sound-insulating windows does not need much innovation. In addition, the instruments to evaluate the best solutions from a cost-benefit point of view and to apportion the contributions of vehicles and tracks and their associated responsibilities have been applied only recently in this field (e.g. CEC, 2003).

In Italy, a decree of the Ministry of the Environment[30], which is consistent with what is stated in the more recent “Position Paper on the European Strategies and Priorities for Railways Noise Abatement” (CEC, 2003), indicates that preference should be given to noise measures at the source (i.e. either on the vehicles or on the tracks) rather than to barriers and building insulation systems. According to Ministry, investments in technical improvements to trains and/or tracks are expected to produce the required noise decrease at the lowest collective cost. Nevertheless, there is no empirical valuation study that can confirm, or deny, such a technical statement for Italy. In addition, local authorities are able to test this technical statement with reference to the local population by assessing the preferences of the affected population for alternative policy solutions, i.e. alternative noise abatement programmes. This flexibility, as introduced in the legislation, gives room for the implementation of an economic valuation study of noise abatement programmes.

The study presented is based in the province of Trento, a fluvial valley surrounded by high mountains in the North-East of Italy. This area is crossed longitudinally by the Brennero railway, which provides transport connections to all the towns located along the Adige River, from West to East. The railway crosses 12 municipalities, and about 2300 households are exposed to excessive noise levels due to the transit of passengers and freight trains. In order to reduce the potential impact on health of exposure to excessive noise levels, the local Administration has programmed investments in noise abatement measures for the coming years. The tentative project implies investments in noise abatement measures on 31 sites on a strip of 74 kilometres, for an overall estimated cost of approximately € 20 million. Noise interventions, which are expected to reduce noise emissions below the national regulation limits, can be implemented with alternative technical measures. In this connection, two radical positions on noise abatement are being

[30] DMA 29/11/2000.

debated. On one side, the local Environmental Protection Agency is recommending the gradual introduction along the rail track of low noise barriers (but perhaps still not sufficient to reduce noise below the limits) to be combined, during a second phase, with some technological innovation in wagons and rail tracks. This would guarantee, in two steps, the required level of noise reduction, minimising the drawbacks of noise barriers for people living or working in the vicinity of the railroad, in terms of aesthetics, landscapes and micro-climate changes (such as reduced light in the case of sound-deadening barriers, or greenhouse effects during summer in the case of transparent barriers). On the other side, the Italian railway company[31] is strongly recommending actions with high barriers and no technological innovation that would be able to guarantee the required level of noise reduction immediately, but with higher collective costs in terms of aesthetic and environmental drawbacks.

To provide advice to the local authorities on the preferred noise abatement option to maximise social utility, a Choice Experiment approach was designed. The survey was distributed to a representative sample of the local affected population. 511 householders were randomly selected from the universe of 1400 households exposed to noise level beyond the acceptable limits and living within 100 metres of rail track[32], in five different municipalities along the Brennero railway[33].

4.2.2. Questionnaire

The questionnaire consisted of three parts. The first part focused on the respondents' noise perception. First, we asked respondents their opinion on the current rail noise situation and asked them to talk about their own experience of noise, by means of a set of eleven questions. First we referred to noise in general terms and used six phrases relating to noise sensitivity to infer the respondents' noise profile. Using a 6-point Likert scale, respondents were asked to say whether they 'not at all' or 'totally' agreed with what was stated in each phrase. Second, we asked respondents how many hours they spent at home during working days and during weekends to infer additional information on their level of noise exposure. We then focused on rail noise and asked respondents to say whether rail noise annoyed them, during the day and night-time, respectively. If they answered yes, the respondents were asked to indicate their own level of annoyance using a 5-point scale (as recommended by ISO, 2001): 'not at all', 'slightly', 'moderately', 'very', extremely' annoyed (see Table 4-1).

Table 4-1: Question on the level of noise annoyance

Do you hear the train?	Does the rail noise disturb you?
□ Yes □ No	□ Not at all □ Slightly □ Moderately □ Very much □ Extremely

[31] Rete Ferroviaria Italiana SpA (RFI).

[32] This area is called "Zone A" in the Italian noise regulation.

[33] Avio, Calliano, Rovereto, Trento, Zambana Nuova.

Those ‘moderately’, ‘very’, or ‘extremely’ annoyed were then asked to indicate which type of disturbance they suffered, during the day and night-time, respectively. A special question was used to understand whether the disturbance arose only from the rail noise emission, or whether it was also related to the track and train vibrations generated during the transit of trains. In fact, according to the results of the focus groups held in two of the seven sites surveyed, vibrations are also perceived as an important source of disturbance generated by closeness to the railway. We also asked whether the level of annoyance declared by the respondents (the householders) was either similar to that suffered by the rest of the household components or higher or lower, and which type of disturbance they suffered. Finally, we asked whether they had ever considered moving because of the rail noise, and whether they thought they lived in a noisy or quite neighbourhood.

The second part of the survey introduced the policy choices and prepared the respondents for answering the CE questions. First, we informed the respondents about the current level of noise pollution to which they are exposed, and the expectations of the level of noise pollution due to the Brennero railway that will be reached by 2010 according to the local Environmental Protection Agency. For instance, the current noise exposure level is approximately 7 to 9 and 9 to 11 decibels over the limits during the day and night, respectively. We also showed the respondent a noise map of the area in which he/she lived showing the relation between noise level and distance from the railway (Figure 4-1). On this map, the respondents could identify the actual site of their own home.

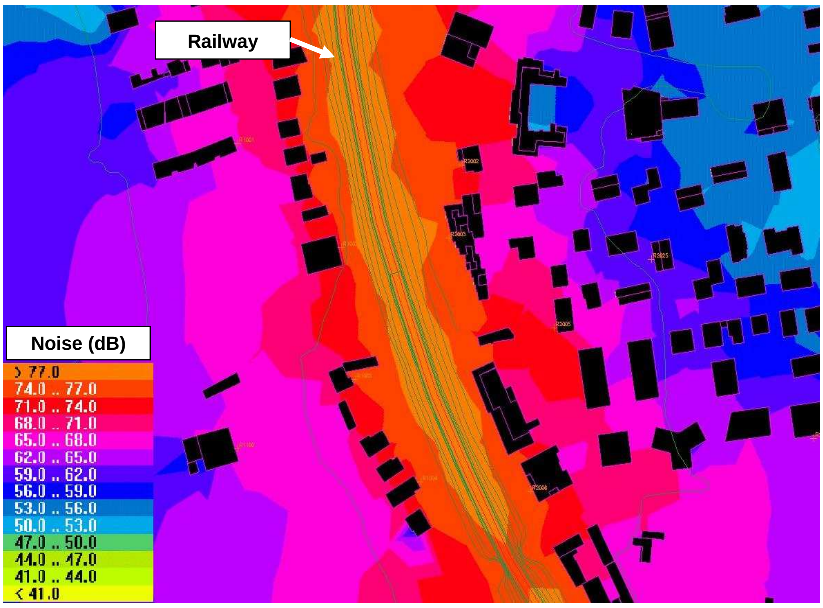

Figure 4-1: Noise map showing the relation between noise levels and distance from the railway

Note: The first and second row of buildings away from the railway experience by different noise levels.

Secondly, we informed the respondent that the local administration was considering the provision of a noise abatement program so as to reduce noise levels, and we described with simple words the main pros and cons of the two alternative types of noise reduction policy instruments that the local administration was considering for implementation. One policy instrument consisted of trackside noise barriers, and the other one involved some technological change either to the train or the railway lines, or to both. We explained that the maximum noise abatement capacity of noise barriers is approximately 15 decibels and that it increases as their height and sound-deadening power increase, although high and sound-absorbent barriers can inconvenience residential areas due to aesthetic and environmental drawbacks, such as reduced light and air circulation. On the other hand, an improvement in technology, which is free from aesthetic impact, can also grant an additional reduction of vibrations, but it has a lower noise reduction capacity (up to 5 decibels). We then showed the respondents several visual simulations of barriers with different heights (four and eight metres)[34]. The graphical simulation consisted of a succession of images showing a given site (corresponding to the respondent's site) either without or with the barrier, and either without or with vegetation. We also show to the respondents a graph, which we called the "noise barometer", with examples of various noise levels that one can experience in daily life, and examples of noise reductions moving from one situation to another one. Here noise reduction levels in decibels are also translated in terms of audible noise and explained in terms of increased distance from the noise source and the receptor (Figure 4-2). The noise barometer was available to respondents during the whole CE exercise.

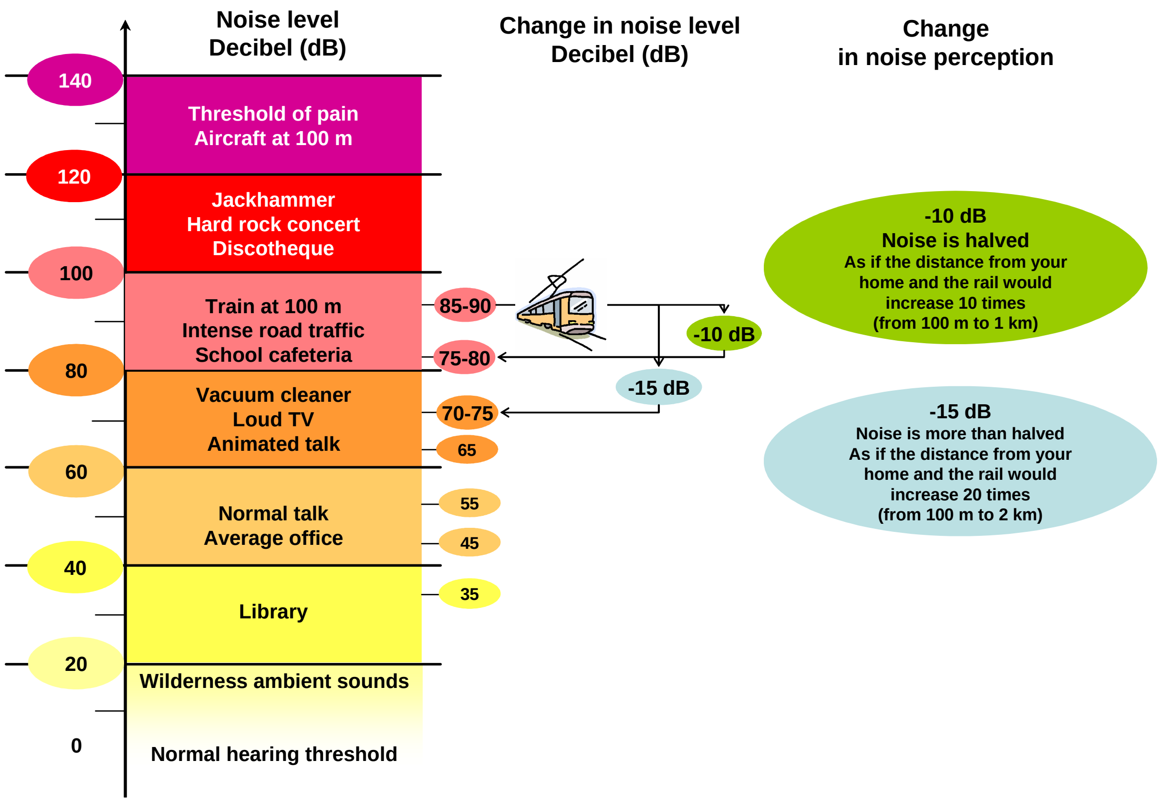

Figure 4-2: Noise barometer with examples of noise levels pertaining to different daily life situations and examples of various noise reduction levels

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[34] During focus groups, we also explored people preferences for aesthetic improvements of noise barriers, provided by covering up the wall with ornamental vegetation. Respondents proved to be insensitive to such aesthetic improvements. Instead, they turned out to be very sensitive to the aesthetic impact due to high trackside barriers as such.

The third part of the survey gathered additional information in order to have a clear image of the respondents' profile, attitudes, socio-economic conditions, exposure to noise, use values provided by the railway, and so forth. Among other things, we asked the respondents: whether their home was provided with thermal and sound insulation systems; whether their home had a garden or a balcony, and, if yes, whether a noise barrier might spoil their recreational value; number of rooms and their exposure to the railway. Questionnaire debriefs closed the survey.

4.2.3. CE questions

The stated choice valuation question is formalized by informing the respondent that the local administration was considering providing a noise abatement programme so as to reduce noise levels. This formalization is characterized by three elements: noise mitigation objective (i.e. reduction of noise level), policy instruments and type of financing (see Table 4-2). Noise mitigation was described to the respondent in terms of a dB(A) reduction, which was set to a range from 9-11 decibels, to 12-14 decibels, and 14-15 decibels. The two policy instruments that are currently available to the local administration so as to mitigate rail noise are the construction of trackside barriers, and/or investments on tracks and trains, including locomotives and wagons. Whilst barriers can only reduce noise, investments on tracks and trains can also reduce vibrations (see Watkiss, 2001) for further details. Furthermore, the survey explains that the full cost regarding the implementation of the proposed noise mitigation programme could not be fully covered by the current financial resources of the local administration. For this reason, it would require an additional contribution from households, either direct of indirect. We therefore explained that the local administration was considering two alternative project-financing strategies. Following Bergstrom et al. (2004), the first one consisted in reallocating the public budget for 2006, and transferring to the noise project the financial resources usually allocated to some other public service, as described in the CE survey, without burdening the households with any additional local tax. The second option was to introduce a new lump-sum local tax for the year 2006, set according to the household income level. The price of this contribution ranges between 35 – 70 euros.

To help the respondents answer the CE questions, the policy instrument attributes were accompanied with an indication of the related level of noise (and vibration) reduction that explains the overall effectiveness associated with each policy profile. In addition, possible reductions in the provincial budget for public transport or administration were described to the respondents in terms of their expected impact: the former one implying a cut that does not allow any improvement in vehicles endowment; the latter implying a cut that reduces some of the benefits for local managers (such as use of cars with drivers and so forth).

The set of attributes, i.e. decibel reduction, height of the trackside barrier, the presence of investment in trains and tracks, together with the type of financing and price of the noise mitigation program constitute the full range of the attributes of the present CE valuation exercise (see Table 4-2).

Table 4-2: List of the attributes used in the CE value application

1. Noise mitigation objective, i.e. reduction of noise level and disturbance [decibels]:
   - Minus 9-11
   [As if the distance between your home and the railway would increase 10 times (e.g. from 100 metres to 1 kilometre)]
   - Minus 12-14
   [As if the distance between your home and the railway would increase 20 times (e.g. from 100 metres to 2 kilometres)]
   - Minus 14-15
   [As if the distance between your home and the railway would increase 30 times (e.g. from 100 metres to 3 kilometres)]
2. Height of noise barrier [metres]:
   - 4 to 6
   - 6 to 8
3. Level of technology:
   - As today
   - Innovation in rail tracks and wagons
4. Price of the programme [€ per household per year_{2006}]:
   - 35
   - 37.5
   - 45
   - 55
   - 60
   - 65 and 70
5. Type of financing:
   - Reduction of 2006 provincial budget for public transport, without any additional tax
   - Reduction of 2006 provincial budget for administration and entertainment expenses, without any additional tax
   - New provincial one-time tax for the year 2006, without any reduction of provincial budget

Following the above explanation, the respondents focused on the CE questions. They were instructed to choose their preference between the two profiles described in the survey. These two profiles correspond to two alternative noise reduction programmes. We prepared all the combinations of the attribute levels, eliminating implausible or inconsistent ones. Choice sets consisted of two alternative profiles. The first one is fixed and corresponds to a benchmark (i.e. a minimum safety standard) policy that guarantees the minimum level of noise reduction able to comply with the limits (i.e. minus 9 to 11 decibels) using noise barriers (4 to 6 metres high) without any improvement in the railway and train technology. The second profile varies from card to card and corresponds to a policy that provides additional noise reduction levels and a reduction in vibrations too, since it combines the use of noise barriers with improvements of train or railway line technology. All combinations were asked in roughly equal frequencies. Each respondent was presented with four questions. Table 4-3 provides an example of a CE question. The econometric analysis of the stated choice responses is anchored in the Random Utility Model (see McFadden, 1974). The random utility model is then estimated with a Multinominal Logit (MNL) model (see Louviere et al. 2000 for more details). Models and valuation results are presented and discussed in Section 4.3

Table 4-3: Example of a CE question

Which of the two noise abatement options would you consider the most attractive for you?		
Attributes	Option A	Option B
Noise abatement	9-11 decibels [As if the distance between your home and the railway would increase 10 times]	14-15 decibels [As if the distance between your home and the railway would increase 30 times]
Noise barrier	4 – 6 meters	6 – 8 meters
Train and tracks technology	No investment	New investments
Cost of the programme	35 euros household/year2006	60 euros household/year2006
Type of financing	New local tax *una tantum* for 2006	Reduction of the provincial budget for public transport for 2006, without any additional tax
	❑ A	❑ B

Table 4-4: Noise reduction options

Noise abatement option	*CE attributes*				
	Noise target [decibels]	Trackside barrier [4-6 metres]	Trackside barrier [6-8 metres]	Train and tracks technology [as today]	Train and tracks technology [new investments]
NOISE1	Minus 9-11	X		X	
NOISE2	Minus 12-14		X	X	
NOISE2	Minus 12-14	X			X
NOISE3	Minus 14-15		X		X

Note: NOISE1 refers to a policy that guarantees the minimum level of noise reduction able to comply with the regulation limits. For computational matters it corresponds to the omitted variable.

4.3. Modelling and valuation results

4.3.1. Indirect utility model specifications

In order to operationalise an empirical formulation of the indirect utility function as described by Equation 4-2, the following six model specifications are examined.[35] Model 1 is the simplest model that we discuss in order to investigate the effect that each of the attributes under consideration have on the respondents' preferences, and therefore on the choice of the noise policy. Formally, we have:

[35] Note that all the indexes for the respondents and alternatives have been omitted.

Model 1

$$V = \beta_1 COST + \beta_2 NOISE2 + \beta_3 NOISE3 + \beta_4 NOISE2 \times HEIGHT \qquad \textbf{Eq- 4-4}$$

In this model formulation, COST refers the cost of the policy to the respondents. NOISE2 and NOISE3 denote the variables for the level of noise reduction (minus 12 to 14 and minus 14 to 15, respectively), which can be provided with different policy instruments (see Table 4-4). NOISE2 can be reached when using a combination of low noise barriers (4-6 metres) and technological innovation, or high barriers (6-8 metres) and no technological innovation. NOISE3, instead, is the highest level of noise reduction that can be provided by the local administration only when using the highest level of barriers and strongest effort in technological innovation. The omitted variable is NOISE1 that corresponds to the minimum level of noise reduction able to comply with regulation limits. It comes from the use of low noise barriers without any technological innovation. The interaction between NOISE2 and HEIGHT controls for the effect of the height of the barrier that we interpret as the most important aesthetic feature of the noise policy. Ceteris paribus, β_4 provides the effect of a unit increment of the barrier's height on the probability to choose a noise policy that reduces noise by 12 to 14 decibels. β_1 can be interpreted as the coefficient of the cost of the noise policy to the respondents regardless of the type of project financing (i.e. payment vehicle).[36]

As we mentioned before, we also want to assess the statistical magnitude of the econometric impact of the different payment vehicles on the consumer's choice and therefore the economic valuation of alternative noise abatement programmes. For this reason, we explore the use of Model 2 and Model 3, which can be interpreted as two formal testings of the payment vehicle.

Model 2 considers only the subsample with the two tax-reallocation payment vehicles, i.e. all the respondents who receive a questionnaire in which the CE question is formulated with either the use of a tax reallocation within the public transport budget or a tax reallocation from the administration.

Model 2

$$V = \beta_1 COST + \beta_2 NOISE2 + \beta_3 NOISE3 + \beta_4 NOISE2 \times HEIGHT +$$
$$+ \beta_5 COST \times PVADM + \beta_6 PVADM \qquad \textbf{Eq- 4-5}$$

where PVADM is a dummy variable for the type of payment vehicle. It takes on the value '0' if the transfer is within the public transport budget, and value '1' if it is to the administration budget. β_1 can be interpreted as the coefficient of the cost of the noise policy to the respondent given a transfer within the public transport budget, whereas β_5 is the coefficient of the cost of the noise policy to the respondent given a transfer to the administration budget.

[36] We also explore other model specifications, but the effect of vegetation on the barriers was not revealed to be statistically significant.

It is therefore interesting to test whether the reported CE responses are influenced by the type of tax reallocation, by controlling that all the respondents face one of the two proposed tax reallocation schemes. This can be formalised with the following hypotheses:

Hypothesis 1: Trading taxes effect on reported CE responses

H1a	**H1b**
$H_0: \beta_5 = 0$	$H_0: \beta_6 = 0$
$H_a: \beta_5 \neq 0$	$H_a: \beta_6 \neq 0$

In addition, we explore the use of Model 3 so as to test the empirical significance of the trading taxes vs. paying taxes effect. For this reason, we now consider all the sample of respondents. On the one hand, we have the subsample with all the respondents who receive a questionnaire in which the CE questions are formulated with either the use of a tax reallocation within the public transport budget or a tax reallocation from the administration. On the other hand, we have the subsample with all the respondents who receive a questionnaire in which the CE question is formulated in terms of a new tax.

Model 3

$$V = \beta_1 COST + \beta_2 NOISE2 + \beta_3 NOISE3 + \beta_4 NOISE2 \times HEIGHT +$$
$$+ \beta_5 COST \times PVTAX + \beta_6 PVTAX \qquad \textbf{Eq- 4-6}$$

where PVTAX is a dummy variable that takes on value '1' if the policy will be financed with a new local tax, and '0' otherwise. β_1 can be interpreted as the cost of the noise policy to the respondent given the tax-reallocation scheme, whereas β_5 is the coefficient of the cost of the noise policy given the introduction of a new local tax.

We therefore test whether the reported CE responses are influenced by the type of payment schemes, and in particular to assess whether CE responses confirm contingent valuation data that suggest the WTP with a tax reallocation is higher than the WTP with a special tax (Bergstrom et al., 2004). This can be formalised with the following hypotheses:

Hypothesis 2: Trading taxes vs. paying taxes effect on reported CE responses

H2a	**H2b**
$H_0: \beta_5 = 0$	$H_0: \beta_6 = 0$
$H_a: \beta_5 \neq 0$	$H_a: \beta_6 \neq 0$

Finally, we investigate the effect of the population characteristics on the implicit price by adding interactions between attributes and socio-demographic and attitudinal variables. All possible attribute combinations were explored in several model specifications in order to test down the model and exclude those interactions that are not statistically significant. The following model includes those statistically significant and relevant for policy analysis and research speculation. Formally, we propose to estimate Model 4:

Model 4

$$V = \beta_1 COST + \beta_2 NOISE2 + \beta_3 NOISE3 + \beta_4 NOISE2 \times HEIGHT + \\ + \beta_5 NOISE2 \times EXPOSURE + \beta_6 COST \times ANNOYANCE + \\ \beta_7 NOISE2 \times INCOME + \beta_8 COST \times EDUCATION \qquad \textbf{Eq- 4-7}$$

This model specification incorporates in the utility function the respondents' level of noise exposure and annoyance, income and education level. It involves the cross-terms of NOISE2 and EXPOSURE, and COST and ANNOYANCE. ANNOYANCE is the level of noise annoyance during the day, based on a 5-point Likert scale, whereas EXPOSURE is a dummy variable that takes on value '1' if the respondent lives in the first row of buildings directly exposed to the railway, and '0' otherwise. Similarly, to control for the level of the respondents' exposure, we also checked the effect of the number of hours per week that the house is not empty. This resulted to be positive but not statistically significant. We can therefore examine the differences in the valuation of 1 unit of rail noise reduction among different respondents' profiles according to exposure and annoyance level. INCOME is a continuous variable and provides the household monthly income. EDUCATION is a categorical variable ranging from '0' to '6' (primary school to PhD). This allows the examination of the effects of the characteristics of individual respondents on the valuation of the single attributes. From the coefficients of interactions we can investigate: whether there is a difference in the marginal utility of price due to different annoyance or education levels; and whether there are differences in the marginal utility of NOISE2, given the respondents' income and noise exposure. We will now present the results of the parameter estimates for each of the models and discuss the welfare implications and the respective repercussions in terms of policy design. Before doing this, however, we will briefly discuss some basic statistics of the questionnaire data.

4.3.2. Statistics of the questionnaire

The data were collected through in-person home interviews of 511 randomly sampled householders affected by rail noise pollution, which yielded 482 responses. A trained team of 23 experts from the Statistics Office of the Province of Trento were recruited and carefully instructed how to administer the survey. Prior to the survey extensive focus groups were organised and a pre-test to check the validity of survey instruments was carried out in February 2005 for another 50 households. The responses in the focus groups and the pre-test greatly helped to improve phrases in the questionnaire and develop a more understandable explanation of the good evaluated. In particular, information requirements, comprehension of noise reduction levels, visual aids, payment vehicle, and monetary bids were discussed during the focus groups.

Descriptive statistics on the socio-demographic characteristics of the respondents are summarised in Table 4-5. The sample significantly represents the universe of households affected by rail noise in the province of Trento. We selected samples evenly from households living in buildings directly or indirectly exposed tow the railway (i.e. first or second row of buildings away from the railway). The average respondent is a 56-year old householder who has been living in the vicinity of the railway for more than 20 years. Her/his household consists of about 2 persons, with one member younger than 12 in 16 percent of the cases. The average household income (€ 1,742 per month) is lower than that of Trento's population, which is estimated to be around € 2,400 per month. In 72 percent of the cases respondents own the place where they live, which usually has a garden or a terrace that is exposed to rail noise. Overall the sample is highly sensitive to health and environmental issues and fairly informed on the rail noise issue. In addition, the survey results indicate that respondents hardly use the Brennero railway, as they prefer to travel by car.

Table 4-5: Descriptive statistics for socio-demographic characteristics

	Mean or percentage
Age	56.3
Over 65	31%
Female	49%
Household size	2.39
Has child under 12 years of age	16%
Years of schooling (>13)	51%
No. of years living in the vicinity of the railway	23.00
Owns the place where she/he lives	72%
Has garden or terrace	84%
Cares about health issues	98%
Cares about environmental issues	94%
Fairly, very much, or extremely informed on rail noise before the survey	77%
Uses the Brennero railway for work	7%
Uses the Brennero railway for tourism or activities other than work	38%
Household monthly income (in euros)	1742.5

The survey also contained a set of questions designed to provide a better understanding of how respondents are sensitive to noise in general, and to rail noise in particular. Using a 6-point Likert scale analysis ranging from 'strongly disagree' to 'strongly agree', which has been coded from '1' to '6', the response results show that noise is highly perceived as an element of annoyance, with a sensitivity value up to a score of 5.48 (see Table 4-6).

Table 4-6: Noise sensitivity scores

	Mean
▪ *Noise sensitivity*	
If I were to buy or rent a house, I would avoid proximity to busy streets, nightclubs or restaurants.	5.48
Sometimes noise makes me nervous.	4.12
If it is noisy while studying or working, I shut the door or move in another room	4.70

Finally, the survey contains information regarding noise produced by railway infrastructures (see Table 4-7).

Table 4-7: Descriptive statistics on rail noise perception, annoyance and exposure

	Percentage
▪ *Rail noise perception*	
Rail noise is 'more' or 'equally' important than traffic noise	72%
Rail noise is 'more' or 'equally' important than air pollution	47%
Rail noise is 'more' or 'equally' important than biodiversity depletion	71%
Rail noise is 'more' or 'equally' important that electromagnetic pollution	65%
▪ *Rail noise annoyance*	
Annoyed by noise during the day	85%
'Very much' or 'extremely' annoyed during the day	42%
Annoyed by noise during the night	74%
'Very much' or 'extremely' annoyed during the night	50%
Did consider moving because of rail noise	25%
Disturbed by rail noise when using garden or terrace	65%
Can not rest quietly during the day	15%
Wakes up easily during the night	39%
Gets nervous	14%
Can not talk with relatives or listen to radio and TV	56%
▪ *Noise and vibrations*	
Only noise disturbs me	25%
Only vibrations disturb me	1%
Noise and vibrations disturb me equally	29%
Both noise and vibrations disturb me, but noise more than vibrations	31%
Both noise and vibrations disturb me, but vibrations more than noise	14%
▪ *Noise exposure*	
Building with direct exposition on the railway	53%
Hours spent at home during the week	6.7
Hours spent at home during the weekend	8.2
Thermal or sound insulation systems installed	93%

According to the results, rail noise is perceived by the majority of the sample as an important environmental policy issue, when compared with other issues such as air and electromagnetic pollution, traffic noise, and biodiversity depletion. Second, rail noise is identified by the respondents as a relevant factor of individual nuisance. In fact, this disturbance is identified by 85 and 64 percent of the respondents as causing annoyance during the day and the night, respectively. Third, vibrations in addition to noise are identified as an important element of rail nuisance. About 29 percent of the respondents report that vibrations and noise are equally disturbing. Finally, the design of the sample is characterised by selecting an equally distributed number of respondents interviewed in two distinct spatial acoustic zones. The first area (Zone A) refers to households who live immediately alongside the railway, on average exposed to more than 70 decibels. The second area (Zone B) includes households living in an acoustic area somewhat further away with noise levels ranging from 60 to 70 decibels (see Figure 4-1).

4.4. CE estimation results

The estimation results for Models 1 to 4 are shown in Table 4-8. In Model 1 all variables are highly statistically significant. As expected, the sign of COST is negative and that of the level of noise reduction is positive. Significant coefficients of the level of NOISE2 and NOISE3 show that the valuation of noise reduction varies according to the relative level of provision. Respondents displayed the highest preferences for the measures that provide an additional level of noise reduction equal to NOISE2 rather than NOISE3, with respect to the minimum granted by the 'benchmark policy' (i.e. NOISE1).

As shown in Table 4-8, WTP for NOISE2, corresponding to a policy noise abatement strategy that relies on an investment both in train or tracks together with a noise barrier set at a minimum level (at most 6 metres) is highly valued by the respondents. These show a WTP of about € 156 per household for 2006.

However, if one portrays a maximum decibel abatement, increasing barriers up to 8 metres, then the WTP decreases to € 33 per household. Confronting this estimate with the coefficient of the interaction between NOISE2 and HEIGHT, which is negative and statistically significant, we can understand that respondents have a strong preference for a policy that provides a noise abatement that is achieved by an increase of train or rail technology rather than an additional increase in the height of noise barriers. These results suggest that, as expected, the height of the barrier is perceived as a major drawback of the noise policy. We can interpret this result as signalling a strong disutility from the powerful negative aesthetic impact of such a construction[37].

Model 2 provides a test of the effect of different payment vehicles on individual preferences. According to Model 2 estimates, the cross-term of, COST*PVAMD is positive. This signals that, ceteris paribus, the payment of the noise abatement programme with the reallocation of taxpayers' money from the administrative budget, rather than with the reallocation of taxpayers' money within the transport budget, has a positive effect on the respondent's utility and

[37] We also explore the statistical significance of an aesthetic improvement of the visual impact of the barrier by using ornamental vegetation. Several model specifications have rejected the econometric robustness of this effect. In short, respondents are not in favour of increasing the barrier above the minimum level set by regulation, with or without ornamental vegetation.

therefore in choosing the protection programme. Nevertheless, the respective statistical magnitude does not differ from 0. For this reason, we can not reject the null for H1a. In Model 2 we also consider a direct effect of the payment vehicle on the indirect utility, captured by PVAMD. This is positive. This signals that, ceteris paribus, if one proposes the financing of the programme via reallocation of taxpayers' money from the administrative budget, it has a positive effect in the indirect utility, independently of the amount of payment. Therefore, it captures a sort of psychological objection to the reallocation of taxpayers' money within the transport budget towards the noise abatement (or alternatively a psychological acceptance of the reallocation of taxpayers' money from the administrative budget towards noise abatement per se. As before, the respective statistical magnitude does not differ from 0. For this reason we can not reject the null for H1b. Therefore, we can conclude that CE data does not show that the two tax reallocation schemes under consideration provide different incentives on consumer choice behaviour and thus on choosing a noise protection programme.

On the other hand, estimation results for Model 3 show that the cross-term between COST and PVTAX is negative and highly statistically significant. This suggests the existence of a negative relationship between the introduction of a new local tax and the selection of a noise protection programme. For this reason we can not reject the null for H2a. This in turn is reflected in the monetary valuation of the programme. In fact, respondents are more inclined to pay for a noise policy that is financed by reallocating a quota of the resources normally destined to other public services than for one financed with a new local tax – see Table 4-8. This is an important result that confirms and extends to CE the results presented by Bergstrom et al. (2004), in the case of a CV study to assess groundwater protection policy in Georgia and Maine.

Finally, according to Model 4, whose specification presented the highest goodness of fit when compared with alternative specifications, one can observe that the effect of the cross-terms with EXPOSURE and ANNOYANCE are positive and highly statistically significant. This means that individual utility is sensitive to noise exposure, here objectively measured by the spatial acoustic zone, as well as to individual annoyance, which is subjectively measured by a 5-point Likert attitudinal motivation scale. In order to capture the empirical magnitude of the effect of the annoyance factor, we estimate the price annoyance elasticity. Such elasticity is estimated at 1.045. This figure is computed by multiplying the cross-effect estimate, 0.020, by an individual annoyance level measured at the sample mean, 3.45. This product is then added to the direct price effect, which results in -0.066 (-0.135+0.069). We repeat the same exercise for two other individual annoyance profiles and compute the price annoyance elasticity, defined as the ratio between the change in price, expressed in percentage terms, and the change in the annoyance category, again anchored at the sample mean. This confirms that individuals who show stronger annoyance have an additional price effect. In this context, a respondent with an individual annoyance profile characterised as 'very sensitive to rail road annoyance' is associated with a marginal WTP for noise abatement equal to € 162. On the contrary, the marginal WTP of a respondent with an individual annoyance profile characterised as 'not sensitive to rail road annoyance' amounts to € 143. Similarly, we can estimate that the mean WTP of noise abatement shifts from € 156 to € 170 if we consider that all the respondents live in the spatial acoustic zone with highest exposure to rail noise (i.e. first row of buildings away from the railway).

Ultimately, and consistently with economic theory, NOISE2*INCOME is positive and significant, though the coefficient is narrow in absolute value. This suggests the existence of a positive relationship between the choice of a noise protection programme, and income, as expected. Similarly, the effect of COST*EDUCATION is positive and significant, meaning that the higher the respondent's level of education, the higher his WTP for noise reductions.

Table 4-8: Estimated coefficients

	Model 1	**Model 2**[a]	**Model 3**	**Model 4**
COST	-0.016***	-0.014*	-0.012***	-0.136***
	(0.006)	(0.008)	(0.006)	(0.015)
NOISE2	2.493***	3.076***	2.674***	1.524***
	(0.383)	(0.489)	(0.388)	(0.454)
NOISE3	0.521***	0.415*	0.536***	0.755***
	(0.196)	(0.238)	(0.197)	(0.228)
NOISE2*HEIGHT	-0.314***	-0.402***	-0.337***	-0.278***
	(0.057)	(0.072)	(0.058)	(0.065)
PVTAX			-0.034	
			(0.590)	
COST*PVTAX			-0.007***	
			(0.002)	
PVADM		0.134		
		(0.626)		
COST*PVADM		0.003 [b]		
		(0.002)		
NOISE2*EXPOSURE				0.212*
				(0.123)
COST*ANNOYANCE				0.020***
				(0.003)
NOISE2*INCOME				0.0004***
				(0.738^{-04})
COST*EDUCATION				0.014***
				(0.002)
SAMPLE	1905	1432	1905	1909

Note: Significance is indicated by ***, ** and * for the 1, 5, and 10 per cent level, respectively, with standard errors in parentheses. (a) Model 2 considers the sample of questions with a tax-reallocation payment vehicle, within the public transport or from the administration budget. (b) Other restrictive model specifications have confirmed that this price effect is not statistically significant.

Table 4-9: Willingness-to-pay (WTP) estimates

WTP	Model 1 (pooled)	Model 2a Administration	Model 2b Public transport	Model 3c Tax
NOISE2	156	230	223	139
NOISE3	33	31	30	28
NOISE2*HEIGHT	-20	-30	-29	-18

Note: WTP is expressed in euros per household for the year 2006. (a) WTP estimates from Model 2 given a tax reallocation within the budget for administration and entertainment expenses. (b) WTP estimates from Model 2, given a tax reallocation within the budget for public transport. (c) WTP estimates from Model 3, given the introduction of a new local tax.

Table 4-10: Sensitivity analysis of the valuation results

Individual annoyance profile	*Not sensitive to rail road annoyance*	*Very sensitive to rail road annoyance*
NOISE2	143	162

Note: Willingness-to-pay (WTP) is expressed in euros per household for the year 2006. Upper and lower bounds are calculated using the Delta method.

4.5. Welfare analysis and policy discussions

Standard economic theory suggests that the WTP should be positively associated with the magnitude of noise reduction. Confirming this expectation, present valuation results show that individuals are, on average, willing to pay for noise reduction. In particular, both NOISE2 and NOISE3 are preferred to NOISE1. In other words, respondents welcome additional noise reduction. Furthermore, we see that the WTP associated with NOISE2 is higher than the WTP associated with NOISE3. This apparent counter-intuitive result can be explained by the fact that the noise reduction associated with NOISE3 is only possible with an additional increase in the height of the barrier. The latter is associated with a strong disutility even if the policy maker proposes to provide such a high barrier together with ornamental vegetation. A similar negative impact of the high height of the barrier is embedded in the estimation of the NOISE2*HEIGHT factor. In fact, the respective noise abatement level is comparable to NOISE2. Nevertheless, NOISE2 is associated with a higher and positive WTP, about □ 156, whereas NOISE2*HEIGHT is associated with a negative WTP. This means that the respondent will not accept a further reduction in the regulated noise if this is provided by increasing the current height of the barrier.

The results from Model 3 provide an original contribution for improving the acceptance and realism of the payment vehicle. Model 3 tells us that the acceptance of a payment vehicle based on an indirect payment in the form of a tax-reallocation scheme is higher than that for a conventional tax scheme. The coefficient of COST*PVTAX is, in fact, negative and statistically significant, resulting in a lower evaluation of those noise policies financed via the introduction of a new local tax by 37 percent – see Table 4-9. As in Bergstrom et al. (2004) in the field of groundwater protection policies, the empirical results of our case study indicate that the people in our sample were willing to pay more for noise reduction using a tax reallocation financing mechanism as compared with a special tax financing mechanism. In addition to Bergstrom et al. (2004), whose CV study does not specify the bundle of public services to be traded off for environmental goods, in our survey we were more cautions in describing them to the respondents, and referred just to two specific types of public services: public administration and public transport. The former and the latter being perceived by the residents of the Province of Trento, respectively, as relatively important and very important.

The coefficient estimates for NOISE2*EXPOSURE and COST*ANNOYANCE in Model 2 suggest that the individual noise perception is likely to influence the WTP for noise abatement in a predictable way. In particular, respondents with a stronger individual perception of noise, are more prone to pay to purchase noise abatement. This result signals the importance of knowing as accurately as possible the respondents' profile according to noise perception, and of improving the methods for gathering such information.

Finally, and to conclude, estimation results for the five municipalities under consideration, show that, if no policy action is undertaken so as to make additional investments in the train or rail, and thus be able to reduce aerodynamic noise, traction noise, and vibrations, a significant welfare loss may result. An aggregate estimate of the total welfare loss ranges from □ 358,800 to □ 1,432,900. This value is obtained by: (1) assuming that the respondents who participated in the survey are representative for the entire population that live in acoustic Zones A and B along the Brennero railway in the 5 municipalities; and (2) multiplying the sum of the noise abatement benefits, which is derived from the CE model and ranges from □ 156 to □ 623, by the total number of residents in that same strip line[38] in the Brennero region, estimated at about 2,300. However, the probability that a noise abatement project would be welfare-improving for the community according to CBA or the potential Pareto-improvement criterion will depend positively on the premise that (a) the project was financed by a reallocation within the public budget, (b) the project would involve heavy investment in train and railway technology, and (c) respondents present a high sensitivity to noise exposure and annoyance.

4.6. Conclusions

In this chapter, we have developed a framework for the valuation of several relevant features of rail noise policies using a CE approach. This approach allows us to understand the preferences of people exposed to rail noise for alternative noise abatement measures, which are expected to differ according to their acoustic efficiency, aesthetics, level of technical innovation, and type of project financing. The signs of major estimated coefficients are statistically significant and consistent with the theoretical predictions, including that respondents evaluate price increase negatively, while evaluating noise abatement positively. In addition, estimation results show that respondents strongly prefer a noise policy that relies on technological innovation rather than barriers. In particular, the height of the noise barrier is perceived as a cost priced at about □ 30

[38] A line of dwellings equidistant from the railway.

per household per unit increase. In addition, we explored formal hypothesis testing so as to infer if and how the type of financing mechanism explains a different level of willingness to pay (WTP) for alternative noise reduction policies. In our sample, people were willing to pay more for noise reduction using a tax reallocation financing mechanism rather than a special tax financing mechanism. Moreover, our empirical results suggest that the bundle of public services to be traded off for environmental goods needs to be specified, as WTP varies according to it. We found that respondents were more willing to pay for noise policies financed by drawing resources from the budget normally used for public administration as compared with policies financed using a quota of resources usually allocated to public transport. Furthermore, we proceeded with the econometric analysis of the effect of population characteristics on the reported CE responses and thus on the valuation of the noise abatement programme. The estimated results suggest that it is good practice to control for the valuation transmission mechanism caused by the individual noise exposure and annoyance.

5. ANALYSIS OF ENVIRONMENTAL IMPACTS OF MOBILITY DUE TO URBAN SPRAWL: A MODELLING STUDY BASED ON ITALIAN CITIES*

While the unbridled movement outward of *leapfrog*[39], low-density urban development is usually viewed as an American ill, in the European context *sprawl*[40] is becoming one of the most debated and controversial topics among recent phenomena of urban transformation.

Sprawl is generally defined as 'low-density development outside of city centres, usually on previously undeveloped land'. A central component is the uncontrolled spreading out of the city, and its suburbs, over more and more rural or semi-rural land at the periphery of an urban area. Differently from traditional urban expansion, this pattern of development is not necessarily followed by an increase in the overall population of the city. Migration is no longer directed from rural toward urban areas but, instead, from the core – more densely populated – towards the periphery of urban settlements, and beyond[41]. Moreover, at the same time that cities are expanding outwards, many still contain a large amount of derelict, unused land, and a high number of empty properties (CEC, 2004).

Another extremely significant trait of sprawl is that the process of expansion is typically disordered and unplanned, often leading to inefficient and unsustainable urban expansion patterns. With special regard to Europe, for instance, there is a widely shared consensus that urban dispersion is, at least in part, the result of a long-lasting normative lack of or, more in general, inadequate or not very far-sighted urban planning policies, which have been unable to guide the direction of the '*push* and *pull*' tendencies of European towns and cities over the last 20 years (for a discussion, see Camagni, et al., 1998, 2002a). Nonetheless, sprawl is as much a product of poor land use planning, skewed market mechanisms, uneven tax policies, and fragmented government bodies as it is a product of personal preference[42].

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* *Based on Travisi, Nijkamp and Camagni (2006a).*

[39] 'Leapfrog' is an unlimited and non-contiguous type of urban development outward from the solidly built-up core of an urban area (TRB, 1998, p. 6).

[40] The term 'sprawl' was coined in North America during the second half of the 1960s, when the features, determinants and effects of this peculiar phenomenon of urban development and conversion captured the interest of both researchers and governments and began to be formally analysed (e.g. Real Estate Research Corporation, 1974; Altshuler, 1997; Windsor, 1979).

[41] If one looks at Europe, towns and cities are expanding outwards into rural areas at a faster rate than their population is growing: a 20 percent physical expansion in the last 20 years with only a 6 percent increase in population over the same period (CEC, 2004).

[42] In Europe, the factors that have contributed to the success of the "dispersed city" are numerous (for a complete discussion, see Gibelli, 1999). Some of these pertain to residential preferences, such as: the overall worsening of the quality of life in urban areas (high cost of residential accommodation,

The European Commission recognises urban sprawl as the most urgent of the urban design issues, as it leads to loss of green space, high cost of infrastructure and energy, increased social segregation, and functional land use divisions, which both reinforce the need to travel and increase dependence on the private motorised transport model, leading in turn to increased traffic congestion, energy consumption, and polluting emissions (OECD, 2000; CEC, 2004).

Still, a comprehensive breadth of knowledge about the alleged impacts and costs of sprawl in Europe is lacking, and academics find themselves unprepared to answer the kind of questions that policy makers and city planning communities ask as they prepare to respond to concerns about sprawl. For instance, what are the roots of sprawl and its actual impacts in European cities? The majority of the literature on sprawl focuses on U.S. urban areas which experience geographical, demographic, and socio-economic conditions and government policies that are profoundly different from European ones (Nivola, 2005). Thus, the synthesis of results produced by the U.S. literature on sprawl needs to be rethought and revisited for Europe.

Anticipating this challenge, this chapter is intended to contribute to the empirical body of evidence that analyses one of the most significant impacts of sprawl, i.e. transportation and travel costs (e.g. higher infrastructure and operating costs, longer travel times and more automobile trips, higher social costs of travel, loss of fragile environmental land, higher air pollution and energy consumption, etc.). It presents the results from a quantitative analysis of the environmental impacts of urban mobility in Italy, and explores the major determinants and causal relationships that link sprawl to higher intensities of travel impact. The analysis uses a mobility impact index based on commuting data for 1981 and 1991, and considers seven Italian urban areas with different geographical locations and levels of polycentrism, comprising about 750 cities.

The remainder of the chapter is organised as follows. Section 5.1 provides a concise review of the literature, with special focus on the travel impacts of sprawl. Section 5.2 presents the conceptual underpinnings of the development of the (commuting) mobility impact model and describes hypotheses concerning the reasons for heterogeneities in the intensity of mobility across cities. Section 5.3 presents the results of a dynamic analysis of the intensity of the growth of mobility over the decade 1981-1991. The potential determinants of travel impacts are discussed in Section 5.4, and statistically analysed in Section 5.5. In Section 5.6, a conceptual interpretation of the causal chain that explains the impact of mobility is proposed and empirically tested. Finally, Section 5.7 provides conclusions drawn from the main results.

congestion, air pollution, noise, deterioration of public spaces, etc.); the evolution of individual preferences and taste in favour of single-household dwellings (following the US archetype); the displacement from central locations of residential use in favour of service activities; the higher costs of real-estate redesignation in central areas compared with extra-urban locations; and, often, less stringent city planning and institutional constraints in the periphery. Some other factors relate to economic activities: increased diffusion of back-office activities irrespective of accessibility economies; poor accessibility of central areas by motorised private transport modes; and increasing fiscal and administrative fragmentation are all contingent elements that have contributed to the success of sprawling patterns of urban expansion.

5.1. A concise review of the travel impacts of sprawl

The "sprawl costs all" literature reveals that various commentators have attributed more than two dozen alleged negative impacts to sprawl (for details, see TRB, 1998; Frank et al., 2000). These range from operating costs to transportation and travel costs, to quality of life and social concern (see Table 5-1). Numerous analyses indicate the increased infrastructure cost associated with sprawling development compared with infill or contiguous and compact development (e.g. Altshuler, 1997; Burchell, 1998; Persky and Wiewel, 2000). Similarly, some studies find that compact or managed urban growth, the opposite of sprawl development, may encourage savings in operational costs for public services (e.g. schools, water/sewer utility lines, etc.) (Burchell, 1992). Furthermore, green land and farmland lost to sprawl has almost always been shown to be more than land consumed under compact growth patterns (TRB, 1998, pp.73-74).

The negative impacts of transportation and travel as they relate to sprawl involve mode of travel, pattern of residential development and development access, density of residential development, and location/type of non-residential development. Among simulations related to the effect of density on travel choices and trip duration, that of Downs (1992) develops a hypothetical urban area to test the extent to which changes in the location and density of development would change average commuting distances. The study shows that the density of growth at the urban fringe has a significant impact on reducing commuting distances. Likewise, one study by Metro (1994) shows that more concentrated development, in conjunction with the expansion of public transport, would reduce vehicle-miles of travel and use of the automobile. Among the bunch of empirical studies available, Frank and Pivo (1994) find that density, mix, and jobs/housing balance are all related to travel behaviour, with employment density and jobs/housing balance having the strongest relationships. At higher densities, trips are shorter but take more time. More trips are made using alternatives to the single-occupant vehicle. As land use mix increases, trip distances, times and auto-mode shares decrease. As jobs and housing become more balanced, trip distances and travel times drop. More recently, Cervero and Kockelman (1997) have analysed the connection between travel demand and three dimensions of the built environment: density, land use diversity, and design. The results show that densities have exerted the strongest influence on personal business trips. Diverse land use has also had an impact on travel demand, while several specific design elements of the built-environment have seemed to be particularly relevant to non-work trip-making. The authors conclude that higher densities, diverse land uses, and pedestrian-friendly designs must co-exist to a certain degree if meaningful transportation benefits are to accrue. Other studies have confirmed the correlation between density and vehicle-miles of travel with cross-sectional analysis (TRB, 1998, pp.167).

Table 5-1: A summary of the alleged negative impacts of sprawl

Substantive concern	Negative Impacts
Operating and public-private capital costs	Higher infrastructure costs Higher public operating costs More expensive private residential and non-residential development costs More adverse fiscal impacts Higher aggregated land costs
Transportation and travel costs	Longer travel time More automobile trips Higher household transportation spending Less cost-efficient and effective transit Higher social costs of travel
Land/natural habitat preservation	Loss of agricultural land Reduced farmland productivity Reduced farmland viability Loss of fragile environmental land Reduced regional open space
Quality of life	Aesthetically displeasing Weakened sense of community Greater stress Higher energy consumption More air pollution Lessened historic preservation
Social issues	Fosters suburban exclusion Fosters spatial mismatch Fosters residential segregation Worsens city fiscal stress Worsens inner-city deterioration

Note: Modified from TRB, 1998.

Among the empirical studies on the travel impact of sprawl available for the European context, Camagni et al. (2002b) perform an empirical quantitative analysis on the metropolitan area of Milan, to establish whether different patterns of urban expansion generate heterogeneous levels of land consumption and mobility impacts. The study provides the first insights for Italy on the correlation between variables describing the type of urban expansion and their travel impacts. Travel impacts are taken as an indicator of the pressure on the quality of everyday life in the urban environment. Using an impact analysis based on commuting data (for 1991), they capture the level of environmental impacts of mobility at the commune level, estimated on the basis of trip time and modal choice. The intensity of the mobility impact is then explained by some variables that control for geographical, socio-economic, morphology, and transport-efficiency factors. The results show that a higher impact of mobility is associated with more extensive and sprawling urban development, more recent urbanisation processes, and residential specialisation. The same procedure is used in two subsequent studies on two other Italian urban areas: Brescia (Camagni et al., 2002a, 2002b) and Bologna (Musolino and Guerzoni, 2003), both referring to 1991.

More recently, Salatino (2004) follows the approach by Ewing et al. (2002) and provides, for all the Italian regions, a static analysis of the univariate correlation between an aggregated indicator of spatial dispersion and parameters that capture private costs ascribable to sprawl (e.g. household petrol consumption, household transport expenditures, etc.). Salatino (2006) proposes a similar static analysis at the national level for 11 EU countries, including Italy, and a Causal Path Analysis (CPA) exercise to find causal relationships among the set of variables analysed. Both analyses show a positive and significant correlation among parameters controlling for urban dispersion and transport costs, overall providing further indications that more dense urban forms are accompanied with higher costs of mobility.

In the present chapter, following the approach of Camagni et al. (2002b), we quantify the impact of commuting mobility in seven Italian urban areas that differ in terms of location and level of polycentricism (see Figure 5-1): Bari, Florence, Naples, Padua, Perugia, Potenza, and Turin[43]. The analysis explores the changes that have occurred in the intensity of travel impacts across a ten-year period, from 1981 to 1991. This is a relevant decade to focus on for Italy. It corresponds to an overall deregulation period that is thought to have promoted, indirectly, the unprecedented success of diffuse urban development patterns; and it coincides with an important economic boom, which led to a change of personal travel preferences. Using multivariate cross-section regression analyses, we demonstrate correlations between travel impacts and several dimensions of the built environment, such as density, diverse land use, and level of polycentrism. We also explore whether there are significant differences in the way the model explains variations in the mobility impact across various 'prototypes' of Italian urban areas. Finally, we propose a conceptual interpretation of the causal chain that explains the strength of travel impacts that we test using CPA.

Figure 5-1: Geographical location and a taxonomy of the urban areas concerned

Note: (a) Shown in parentheses are: i) the number of municipalities in each urban area, ii) the type of urban settlement (M: Metropolitan; P: Polycentric); and, iii) the geographical location (N: North; C: Centre; S: South). (b) Polycentrism is measured as the ratio between the population of the chief town of a given urban area (province) and the sum of the population of the ten biggest cities belonging to it. Urban areas are defined as 'Metropolitan' when the index of polycentricism is higher than 0.5, 'Polycentric' otherwise.

[43] Each urban area is a province (from the administrative point of view), and comprises a given number of municipalities. Overall, about 740 communes are analysed (see Figure 5-1).

5.2. Setting a mobility impact index

The first research question addressed in this chapter concerns how to measure the intensity of travel impacts generated by private mobility at the local level, for a group of seven Italian urban areas representative of metropolitan and polycentric agglomerations in the North, Centre, and South of Italy. The lack of reliable mobility data presents a methodological and operational problem. As far as mobility is concerned, we use the only reliable data available at the local (municipal) level. These are commuting data for economically-active residents recordered in 1981 and 1991 by the Italian National Census[44]. Data are disaggregated by mode and trip duration into 6 and 3 classes, respectively (Table 5-2). Following Camagni et al. (2002b), we can therefore apply a weight matrix that associates higher scores with less environmentally-friendly mode/duration categories[45] and estimate the impact generated by commuting trips at the municipal level.

Using the values in Table 5-2, trips recorded in the Census are transformed into Equivalent Impact Commuters (EIC). For each k-th municipality, the intensity of the mobility impact, $IMPACT_k$, can be estimated as the ratio between the EIC and the actual outward trips, as follows:

$$IMPACT_k = \frac{\sum_{ij} m_{ij} w_{ij}}{\sum_{ij} m_{ij}} \qquad \textbf{Eq- 5-1}$$

where m_{ij} is the sum of trips within the k-th city plus outward trips from the k-th city, for the i-th travel mode and the j-th trip-time class; and w_{ij} is the weight assigned to the i-th travel mode and the j-th trip-time class.

Weights are not linked directly to any physical impact dimension, and provide a relative, rather than an absolute, measurement of travel impacts. The relative impact of alternative travel modes is defined according to:

i) external impacts from actual transport: air pollutant emissions, contribution to traffic congestion, risk of accident, noise;

ii) external impacts from disposal of vehicles: land consumption, parking congestion.

In addition, for any given mode, the impact of a trip per unit of time is assumed to decrease with the trip length because: i) pollutant emissions by motorised vehicles are higher at the beginning of the trip; and ii) traffic fluidity increases outside urban areas. The database does not provide trip-length/duration data. Therefore, a drawback of this approach is that we can not control directly for the effects of external factors that might influence trip duration and modal choices, such as congestion or the endowment of transport infrastructure and services. Another limitation is that the data account only for one segment of urban mobility, commuting, disregarding other non-systematic travel purposes, for instance those due to leisure activities. On the other hand, a big advantage of this approach,

[44] ISTAT: Istituto Nazionale di Statistica.

[45] Overall, we define 18 different mode/duration combinations, according to the structure of the available data.

compared with other direct estimations of physical environmental impacts, is that it refers directly to the demand for urban mobility generated in each city. This, therefore, allows an analysis of some specific dimensions of cities, e.g. density, settlement patterns, functional mix, etc., that might explain heterogeneities across travel behaviour and their impacts.

Table 5-2: Weights by travel time and travel mode

	Classes of travel modes (*i*th)	Weights for modes	Time (min)		
Classes of trip time (*j*th)			***0-30 min***	***31-60 min***	***>60 min***
Average trip time			15 min	45 min	75 min
Weight per time unit			1.20	1.00	0.80
Equivalent trip time			18 min	45 min	60 min
	Walking or other soft means	0.00	0.00	0.00	0.00
	Bus	0.33	0.13	0.33	0.44
Travel mode	***Private car (driver)***	1.00	0.40	1.00	1.33
	Motorcycle	0.33	0.13	0.33	0.44
	Private car (passenger)	0.00	0.00	0.00	0.00
	Train, tram, underground	0.20	0.20	0.20	0.27

5.3. Dynamics of mobility impact during 1981–1991

An additional question concerns the dynamic of the demand for urban mobility and its impacts during recent decades. Before exploring the determinants of the intensity of mobility, we therefore analyse the distribution of the mobility impact index (IMPACT) across the urban areas (province) of concern and its variation from 1981 to 1991, computing the IMPACT for 739 Italian cities. The analysis is performed at three different levels. We look at: i) the average impact for each urban area[46]; ii) the average impact of capital cities (i.e. of the chief town of a given province); and iii) the average impact of minor cities. This allows us to take into account that, within a given province, the dynamics of both socio-economic and spatial conditions during the period 1981-1991 can be significantly different for minor towns.

Table 5-3 provides descriptive statistics of the mobility impact index for 1981 and 1991 at the province level. Table 5-4 shows the average values and the percentage rate of increase of impacts for each urban area, their capital city, and minor towns. If we look at absolute values (Table 5-3 and Figure 5-2), a first result is that travel impacts are higher for cities located in the Northern regions (Turin, Padova, Perugia, Florence) and decrease towards the Southern urban areas. We tentatively interpret this as an effect of higher income levels in the northern regions, which normally favour the diffusion of motorised private travel means (Kockelman, 1995). Average impacts turn out to be higher for metropolitan (monocentric) urban areas compared with polycentric agglomerations (Figure 5-2). On the other hand, travel impacts have increased noticeably across the whole peninsula from 1981 to 1991, with higher rates of increase observed for Southern urban areas: namely, Bari and Potenza. These range from a minimum of about 15

[46] This can be considered, in a first approximation, as the urban commuting area.

percent for Turin to a maximum of about 37 percent for Potenza (Table 5-4). Overall, minor towns experienced higher rates of increase than capital cities.

These results suggest that mobility has increased for reasons that go beyond the population growth rate, which was on average about 5.3 percent. Without entering into the details of this discussion[47], among the main drivers of this tendency several commentators report an increasing demand for urban mobility, together with a shift of individual preferences towards private motorised travel modes, and the lower competitiveness of public transport services compared with private ones (e.g. Lattarulo, 2003). The results shown in Table 5-5 confirm that the shift of individual preferences towards private motorized travel modes occurred from 1981 to 1991. The distribution of commuters by travel modes has changed in favour of private transport, with a particularly marked increase in the use of automobiles. The increments range from a minimum of 9 percent (for Naples), to a maximum of 14 percent (for Turin and Padua). At the same time, other private soft modes have been progressively abandoned (walking, bicycle), and the incidence of use of public transport has decreased too.

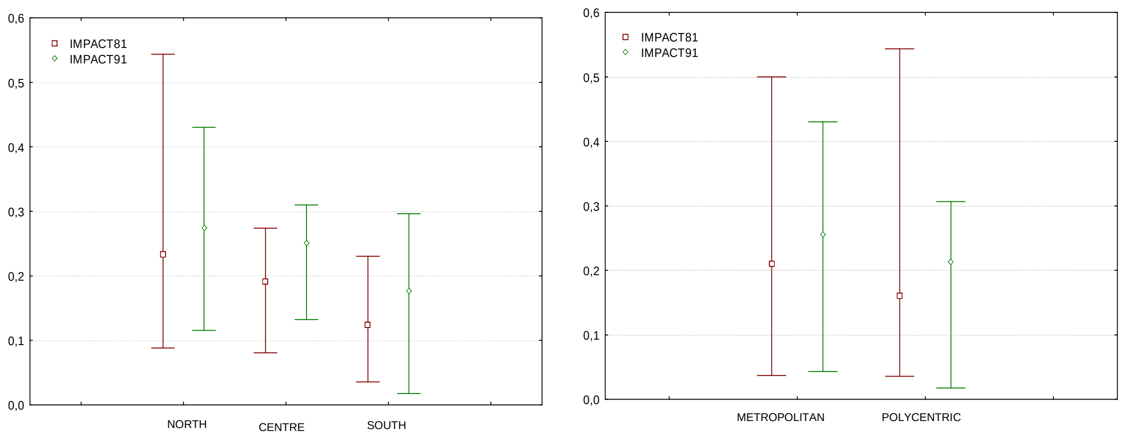

Figure 5-2: Box-plots reporting mean, minimum and maximum values of the IMPACT for 1981 and 1991, for metropolitan and polycentric areas and according to the geographical location

[47] A thorough discussion of the reasons for such an increase lies outside the scope of this chapter, as it would require an analysis of changes in socio-economic features of urban settlements from 1981 to 1991 (e.g. household income, transport networks and infrastructures, changes in production factors and job markets, etc.).

Table 5-3 : Descriptive statistics of IMPACT for 1981 and 1991

Variable	*Nobs*	*Mean*	*Median*	*Minimum*	*Maximum*	*Std.dev*
IMPACT81						
Bari	44	0,105	0,149	0,067	0,146	0,021
Florence	42	0,195	0,235	0,085	0,288	0,037
Naple	91	0,151	0,102	0,039	0,242	0,043
Padua	100	0,198	0,194	0,123	0,572	0,089
Perugia	48	0,063	0,178	0,091	0,240	0,091
Potenza	99	0,109	0,195	0,038	0,202	0,032
Turin	315	0,245	0,108	0,093	0,526	0,059
IMPACT91						
Bari	44	0,159	0,189	0,119	0,201	0,019
Florence	42	0,260	0,287	0,206	0,326	0,028
Naple	91	0,189	0,159	0,045	0,312	0,042
Padua	100	0,237	0,260	0,167	0,289	0,028
Perugia	48	0,244	0,237	0,139	0,323	0,030
Potenza	99	0,174	0,244	0,019	0,246	0,036
Turin	315	0,287	0,174	0,121	0,453	0,049

Table 5-4: Mean value and rate of increase of IMPACT per urban area and time period

	1981			1991			Increase rate 1981-91 (%)		
	(a) Province	(b) Chief town	(c) Other	(a) Province	(b) Chief town	(c) Other	(a)	(b)	(c)
Bari	0,105	0,142	0,115	0,159	0,181	0,155	33,9	21,9	26,0
Florence	0,195	0,184	0,195	0,260	0,206	0,262	25,2	10,8	25,5
Naple	0,151	0,195	0,150	0,189	0,203	0,189	20,4	3,9	20,6
Padua	0,198	0,174	0,198	0,237	0,221	0,237	16,3	21,3	16,3
Perugia	0,189	0,199	0,189	0,244	0,240	0,244	22,7	17,0	22,6
Potenza	0,109	0,148	0,109	0,174	0,199	0,174	37,3	25,7	37,5
Turin	0,245	0,192	0,245	0,287	0,241	0,287	14,8	20,2	14,8

Notes: (a) mean IMPACT value for each urban area (province). (b) IMPACT value for the chief town of a given province. (c) mean IMPACT value referring to the minor towns of a given province.

Table 5-5: Percentage distribution of commuters by travel mode during 1981 and 1991

		Naples	**Turin**	**Bari**	**Florence**	**Padua**	**Perugia**	**Potenza**
Walking or other soft means	*1981*	44%	29%	56%	30%	37%	28%	58%
	1991	41%	22%	45%	23%	26%	21%	41%
Bus	*1981*	24%	25%	13%	23%	17%	24%	17%
	1991	17%	18%	10%	15%	15%	17%	19%
Private car (driver)	*1981*	15%	28%	17%	27%	25%	32%	16%
	1991	24%	42%	28%	37%	39%	45%	29%
Motorcycle	*1981*	1%	2%	2%	8%	12%	6%	1%
	1991	2%	1%	2%	11%	7%	3%	1%
Private car (passenger)	*1981*	4%	7%	6%	6%	7%	7%	6%
	1991	7%	9%	10%	8%	10%	11%	9%
Train,tram,metro	*1981*	11%	10%	6%	6%	2%	3%	3%
	1991	10%	8%	6%	6%	3%	2%	2%

5.4. Potential determinants of the mobility impact

In this section we examine the connection between the travel impact and some specific dimensions of the urban environment. First, we consider factors that control for the level of sprawl (e.g. Frank, 1989; McNally and Kulkarni, 1997; Ewing et al., 2002): density, diversity of land use, and consumption of exurban agricultural land. Density, and more specifically, *low density*, is one of the cardinal defining characteristics of sprawl. Single measures of urban form based on density have been extensively used in the literature (e.g. Spillar and Rutherford, 1990; Dunphy and Fisher, 1996; Bhat and Singh, 2000). We estimate urban density as the gross density of cities (DENSITY), and expect to observe a negative correlation between densities and travel impacts.

In terms of land use types, sprawl includes both residential and non-residential development. Residential development contains primarily family housing; whereas non-residential includes industrial and office parks, shopping centres, as well as other public buildings. In sprawling urban areas, these different types of land use tend to be spatially segregated from one another and unbalanced. A number of studies have therefore considered multiple urban form measures jointly. For instance, Frank and Pivo (1994) consider density and land use mix, whereas Kitamura et al. (2001) employ density and an accessibility measure. In this study we include the measure of land-use mix among the factors that control for sprawl: MIXITE is the ratio between jobs and residents. As it captures the jobs/housing balance, it controls for the diversity of land use and for sprawl. Sprawl is in fact both a cause and an effect of functional land-use divisions, which reinforce the need to commute and increase dependence on private transport modes. We expect to find a negative relationship with the mobility impact indicating that urban mobility becomes more intense as the imbalance between jobs and residents increases (e.g. Boarnet and Sarmiento, 1998; Boarnet and Crane, 2001).

Another significant trait of sprawl is its consumption of ex-urban agricultural and other vulnerable land in abundance, which are the types of land usually found at the periphery of development. We control for this including a variable that measures the rate of agricultural areas in the built environment (RURAL). According to this, we expect to find that the MII is lower for cities with higher quotas of rural areas.

Second, we look at a few additional structural factors that contribute to the characterisation of hierarchies across cities (municipalities) that are part of the same urban area (province). These are expected to influence the spatial distribution of activities and services and, therefore, commuting. We control for: level of polycentrism (METRO, POLYC); geographical distance between capital city and minor cities (DISTANCE); and commuting self-containment capacity (SELFCONT). In this latter case, we focus on whether commuters move within the city borders, or whether they are directed outside their own residential town. This helps us to control for the length of trips, information that is not available in our database. We expect to observe a negative correlation with travel impacts.

The relationship between travel impacts and the quality of public transport services is also relevant since this strongly influences individuals' modal choices. Hence, we consider two explanatory factors that control, respectively, for accessibility (SHAREPUB) and competitiveness (efficiency) of public transport

(COMPUB). In this respect, Camagni et al. (2002b) find empirical evidence[48] that mobility impacts are inversely correlated to the share and competitiveness of public transport.

Other potential explanatory factors include the overall city dimension (POPTOT), the geographical location of cities, and their demographic growth rate (GROWTH). In particular, the demographic growth rate is expected to show a positive relationship with the intensity of impact. High population growth rates are generally associated with areas of recent expansion, typically scattered all around the older urban conurbation. Impacts are expected to increase with the urban dynamism of cities.

Descriptions and descriptive statistics of explanatory variables[49] are provided below in Table 5-6, respectively.

Table 5-6: List and description of explanatory variables.

Type of variable	Abbreviation	Definition
Dependent:	IMPACT91	Average intensity of the impact of urban mobility at commune level. Impact of mobility is calculated as the ratio between the EIC and the number of commuters recorded in the Census
Sprawl	DENSITY	Gross density of the commune calculated as the number of residents over the whole land area [Km^2]
	MIXITE	Ratio between the number of jobs and residents in a city (municipality)
	RURAL	Incidence of rural areas calculated as the rural area [Km^2] over the total land area [Km^2]
Structural	METRO	Takes value 1 if the urban area is metropolitan, 0 otherwise
	POLYC	Takes value 1 if the urban area is polycentric, 0 otherwise
	SELFCONT	Degree of containment of urban mobility within a given urban settlement (at commune level) measured as the ratio between commuters moving out of the commune, and commuters moving within or outward the city
	DISTANCE	Distance [Km] between the centroid of a city and the centroid of the capital city of the province
Mobility	COMPUB	Relative competitiveness of public transport calculated as the ratio between the average time taken for trips made with private transport and the average time taken for trips made with public transport (the ratio is multiplied by 100 for computational reasons)
	SHAREPUB	Market share of public transport calculated as the percentage of all trips made by public transport
Other	GROWTH	Growth rate of the population between 1981 and 1991
	NORTH	Takes value 1 if the city is located in the North of Italy, 0 otherwise
	CENTRE	Takes value 1 if the city is located in the Centre of Italy, 0 otherwise
	SOUTH	Takes value 1 if the city is located in the South of Italy, 0 otherwise
	POPTOT	Total number of residents
	SUPTOT	Total land area [Km^2]

[48] The study refers to the metropolitan area of Milan, Italy.

[49] Results from univariate regression analysis can be found in Travisi et al. (2006).

Table 5-7: Descriptive statistics of independent variables, referring to 1991

Variable	*Mean*	*Minimum*	*Maximum*	*Std.dev.*
IMPACT91	0.24	0.02	0.45	0.06
DENSITY	564.04	1.58	13060.98	1340.67
MIXITE	0.43	0.03	4.07	0.33
RURAL	0.54	0.01	0.99	0.28
SELFCONT	0.36	0.01	1.00	0.16
DISTANCE	40.19	1.00	165.00	26.26
COMPUB	38.58	3.71	446.90	33.37
SHAREPUB	32.90	2.67	75.76	10.79
GROWTH	2.25	-55.30	69.90	10.34

5.5. Multiple regression variants and estimation results

The distribution of travel impacts is examined using an econometric analysis to ascertain whether there are significant correlations with any of the selected independent variables that describe the urban environment. All regression variants refer to 1991, and the IMPACT is used as the dependent variable. It captures, at the commune level, the intensity of the environmental impacts associated with travel demand. A few variables controlling for sprawl, structural, and mobility features of the urban environment are used as explanatory variables. The relationship is established using least squares estimators.

The initial step of our analysis is to assess the average effect of explanatory variables, irrespective of potential heterogeneities across different urban areas. We start, therefore, by running multivariate regressions using pooled data. Preliminary to this, we check for multicollinearity among independent variables and find no significant redundancy among them. The estimation results are reported in Table 5-8. In Model A, all coefficients have the expected sign and are statistically significant. The significant and negative coefficients of DENSITY, MIXITE and RURAL show that sprawl contributes to higher travel impacts. The negative sign of DENSITY suggests that less compact development patterns are associated with higher impacts. We argue that this result is explained by the greater dispersion of activities in sprawl which makes it necessary to spend more time travelling between activities than in a more compact, mixed-use setting. Furthermore, the negative and significant coefficient of MIXITE allow us to suppose that segregation of use requires that the majority of trips be made by automobile, whereas residents of areas with higher mixed-use and higher densities have the option of riding on public buses, biking, or walking. As expected, the sign of the parameter that controls for the consumption of agricultural land (RURAL), typical of sprawl, is also negative and highly statistically significant. Interestingly, SELFCONT too has a negative and highly statistically significant coefficient. This signals that cities with higher self-containment capacities generate lower traffic volumes, usually over shorter distances, and with higher rates of trips using public or non-motorised travel modes[50]. Finally, the geographical location also matters,

[50] The mobility impact indexes estimated for trips within cities resulted in lower values than those estimated on the basis of trips going beyond the city boundary.

confirming that higher impacts are associated with cities located in the Northern regions. This statement is supported by the positive and significant coefficient of NORTH.

In Model B, we use the F-test to assess to what extent the heterogeneity across provinces needs to be taken into account using a weighted least squares (WLS) estimator. The pooled regression model can be affected by heteroscedasticity because the mobility impacts refer to different provinces with differing numbers of observations (i.e. because of different numbers of municipalities in each province: see Figure 5-1). We therefore use the number of observations of the underlying province as a proxy to account for the differing sample sizes available for each of the seven urban areas. The sample size of the different provinces ranges between 42 and 315 observations. The results show that OLS and WLS models provide significant and robust results consistent with our a priori expectations.

Finally, Model C includes a variable controlling for the share of trips made with public transport (SHAREPUB), and a proxy of the efficiency of private *versus* public transport (COMPUB). This model specification has a slightly higher explanatory power than the previous ones. The results show that both the additional regressors are negative and significantly correlated with travel impacts. This suggests that higher accessibilities and higher time-efficiencies of public transport may play an important role in shifting mode choices from private motorised to non-motorised trips. The WLS model is omitted as it does not improve the performance of the analysis.

The aforementioned results largely satisfy our a priori expectations. However, one of our aims is to test whether our model is robust for different Italian cities. Consequently, we proceed, by running multiple regressions in cross-section in order to explore the existence of significant differences among: i) single urban areas; ii) cities located in the North, Centre or South of Italy; and iii) metropolitan and polycentric urban areas. We use the Wald-test on the combined restrictions of model parameters and intercepts across such aggregate samples.

We begin with the analysis of single urban areas. Table 5-9 reports the results of reduced and full specification OLS and WLS models (A, B, C). Similar to the pooled model, the dependent variable IMPACT91 is modelled as a linear additive function of sprawl, structural, and mobility variables, with intercepts specific to each province. Overall, the results presented in Table 5-9 confirm the outcomes of the pooled models (Table 5-8), even though the significance of coefficients is reduced as a result of the limited number of observations available for each of the subsamples based on provinces. The F-test results point to preference for the weighted over the unweighted model. A Wald-test on the combined restrictions of the parameters across different provinces, resulting in seven aggregate samples, shows that restrictions can be rejected and, therefore, that parameters are statistically different for cities belonging to diverse provinces (urban areas). Likewise, province-specific intercepts are also statistically unequal. Overall, the results are mixed. Major variations from the pooled model's results relate to the effects that the self-containment capacity and the proportion of agricultural land have in explaining the IMPACT91 variance in different provinces. For such parameters, whenever significant, coefficients have either a positive or a negative sign. For instance, for SELFCONT, Naples and Turin have negative and significant coefficients, whereas Perugia, Potenza, Florence and Bari have positive and highly significant coefficients not in conformity with our expectation. Another incongruous result is that the proportion of agricultural land favours higher IMPACTs in Naples. As expected, the coefficient of SHAREPUB takes on negative and significant values for the province of Turin, while it is positive for Perugia and Potenza. In this latter case, the results suggest that the overall effect of higher rates of commuters travelling on public transport contributes to an increase of travel impacts, which might be due to longer average trip duration. On the other hand, whenever significant, coefficients of sprawl variables (DENSITY, MIXITE, RURAL) show the expected negative sign. The same goes for DISTANCE and GROWTH that, whenever significant, have a negative and positive sign, respectively.

We move to a broader level of analysis and run a cross-section analysis on the basis of geographical location, and level of polycentrism, using the usual model specifications. Table 5-10 shows that the models perform well in terms of explanatory power and significance of coefficients. Whenever significant, coefficients are consistent with previous results and expectations. The Wald-test on combined restrictions on the parameters across North, Centre, and South aggregated samples shows that the null hypothesis of equality of regressors and intercepts coefficients across subsamples can be rejected. This signals that parameters are statistically different for cities with different geographical locations. In this case, the WLS model is not to be preferred to the OLS model and it is omitted in the table. Finally, Table 5-11, reports the results of cross-section regression models with aggregations based on the cities' level of polycentrism. Once more, the Wald-test on combined restrictions on the parameters across polycentric and metropolitan aggregated city samples shows that the null hypothesis of equality of regressors and intercept coefficients can be rejected. The WLS model is not to be preferred to the OLS model, and is omitted. Just as before, whenever significant, regressors take on the expected sign for each subsample. There are, however, some differences in the elasticity of some explanatory variables. In particular, the effect of diversity in land use, growth rate and density is stronger for cities belonging to polycentric urban agglomerations, whereas the effect of DISTANCE and RURAL is stronger for metropolitan ones.

Ultimately, we can argue from the above-mentioned results that the usual specifications can explain variations of intensity of travel impacts at a broader spatial level than the local one. Nevertheless, Wald-tests on restrictions show that there are significant differences in the elasticities of explanatory parameters, disaggregated according to urban area, location and polycentrism.

Table 5-8: Least squares regression analysis of the mobility impact index 1991 with pooled data

	Model A OLS	**Model B WLS**	**Model C OLS**
Dependent variable:	IMPACT91	IMPACT91	IMPACT91
Independent variables:			
Intercept β	0.31***	0.29***	0.33***
	(0.01)	(0.68^{-02})	(0.89^{-02})
DISTANCE	-0.43^{-03}***	-0.31^{-03}***	-0.44^{-03}***
	(0.66^{-04})	(0.58^{-04})	(0.65^{-04})
DENSITY	-0.39^{-05}**	-0.21^{-05}*	-0.40^{-05}***
	(0.13^{-05})-	(0.11^{-05})-	(0.13^{-05})-
RURAL	-0.51^{-03}***	-0.48^{-03}***	-0.46^{-03}***
	(0.65^{-04})	(0.60^{-04})	(0.64^{-04})
GROWTH	0.34^{-03}**	0.45^{-03}**	0.23^{-03}***
	(0.15^{-03})	(0.15^{-03})	(0.15^{-03})
Log(MIXITE)	-0.011***	-0.011***	-0.010***
	(0.002)	(0.002)	(0.002)
SELFCONT	-0.07***	-0.03***	-0.08***
	(0.01)	(0.01)	(0.01)
METRO	0.01***	0.01***	0.01***
	(0.3^{-02})	(0.3^{-02})	(0.3^{-02})
NORTH	0.01**	0.01**	0.01**
	(0.44^{-024})	(0.33^{-024})	(0.43^{-024})
SOUTH	-0.07***	-0.08***	-0.07***
	(0.52^{-02})	(0.37^{-02})	(0.53^{-02})
SHAREPUB	--	--	-0.65^{-03}***
			(0.15^{-03})
COMPUB	--	--	-0.93^{-04}***
			(0.43^{-30})
No. of obs.	734	734	729
R^2-adj	0.64	0.65	0.66
F-test	147.52***	154.34***	130.11***

Note: The weights are determined as the number of observations related to each of the seven underlying urban areas. Standard errors are given in parentheses. Significance is indicated by ***, ** and * for the 1, 5, and 10 percent level, respectively.

Table 5-9: Least squares regression analyses of the mobility impact index 1991

	Model A **OLS**	**Model B** **WLS**	**Model C** **WLS**
INDMOB91			
β_{Bari}	0.08 (0.06)	0.08*** (0.03)	0.10** (0.04)
$\beta_{Florence}$	0.27*** (0.05)	0.27*** (0.02)	0.30*** (0.03)
β_{Naples}	0.25*** (0.02)	0.25*** (0.01)	0.24*** (0.02)
β_{Padua}	0.21*** (0.04)	0.21*** (0.03)	0.18*** (0.03)
$\beta_{Perugia}$	0.26*** (0.04)	0.26*** (0.02)	0.23*** (0.02)
$\beta_{Potenza}$	0.18*** (0.03)	0.18*** (0.02)	0.15*** (0.02)
β_{Turin}	0.38*** (0.01)	0.38*** (0.01)	0.41*** (0.01)
DISTANCE			
Bari	0.10^{-03} (0.32^{-03})	0.10^{-03} (0.16^{-03})	0.12^{-03} (0.16^{-03})
Florence	-0.65^{-03} (0.53^{-03})	-0.65^{-03} (0.27^{-03})	-0.56^{-03} (0.52^{-03})
Naples	-0.10^{-02}*** (0.41^{-03})	-0.10^{-02}*** (0.30^{-03})	-0.11^{-02}** (0.31^{-03})
Padua	-0.10^{-02}*** (0.41^{-03})	-0.11^{-02}** (0.33^{-03})	-0.11^{-02}*** (0.32^{-03})
Perugia	-0.42^{-03} (0.29^{-03})	-0.42^{-03}*** (0.16^{-03})	-0.58^{-03}*** (0.17^{-03})
Potenza	-0.18^{-03}* (0.96^{-04})	-0.18^{-03}*** (0.75^{-04})	-0.14^{-03}** (0.73^{-04})
Turin	-0.96^{-03}*** (0.12^{-03})	-0.95^{-03}*** (0.17^{-03})	-0.78^{-03}*** (0.17^{-03})
DENSITY			
Bari	0.68^{-05} (0.16^{-04})	0.68^{-05} (0.83^{-05})	0.48^{-05} (0.81^{-05})
Florence	-0.20^{-04} (0.15^{-04})	-0.21^{-04}*** (0.76^{-05})	-0.24^{-04}*** (0.76^{-05})
Naples	0.97^{-06} (0.19^{-05})	0.97^{-06} (0.14^{-05})	0.77^{-06} (0.14^{-05})
Padua	-0.17^{-04} (0.16^{-04})	-0.17^{-04} (0.12^{-04})	-0.16^{-04} (0.14^{-04})
Perugia	-0.75^{-05} (0.66^{-04})	-0.75^{-05} (-0.38^{-04})	-0.47^{-05} (-0.37^{-04})
Potenza	-0.19^{-04} (0.11^{-03})	-0.19^{-04} (0.93^{-04})	-0.21^{-04} (0.90^{-05})
Turin	-0.17^{-04}*** (0.43^{-05})	-0.17^{-04}*** (0.59^{-05})	-0.15^{-04}** (0.58^{-05})
RURAL			
Bari	0.31^{-03} (0.53^{-03})	0.31^{-03} (0.28^{-03})	0.32^{-03} (0.27^{-03})
Florence	-0.23^{-04} (0.41^{-03})	-0.23^{-04} (0.20^{-03})	-0.17^{-03} (0.21^{-03})
Naples	0.37^{-03}* (0.19^{-03})	0.37^{-03}** (0.14^{-03})	0.39^{-03}*** (0.14^{-03})
Padua	-0.15^{-04} (0.28^{-03})	-0.15^{-04} (0.22^{-03})	-0.11^{-04} (0.22^{-03})
Perugia	-0.33^{-03} (0.33^{-03})	-0.33^{-03}* (0.19^{-03})	-0.20^{-03} (0.20^{-03})
Potenza	-0.46^{-03}** (0.21^{-03})	-0.46^{-03}** (0.17^{-03})	-0.41^{-03}* (0.16^{-03})
Turin	-0.60^{-03}*** (0.83^{-04})	-0.60^{-03}*** (0.12^{-03})	-0.50^{-03}*** (0.11^{-03})
GROWTH			
Bari	0.45^{-03} (0.69^{-03})	0.45^{-03} (0.36^{-03})	0.35^{-03} (0.35^{-03})
Florence	0.70^{-03} (0.63_{-03})	0.70^{-03}** (0.31^{-03})	0.69^{-03} (0.30^{-03})
Naples	0.21^{-03} (0.31^{-03})	0.21^{-03} (0.23^{-03})	0.26^{-03} (0.23^{-03})
Padua	-0.33^{-03} (0.50^{-03})	-0.33^{-03} (0.40^{-03})	-0.21^{-03} (0.40^{-03})
Perugia	-0.62^{-03} (0.89^{-03})	-0.62^{-03} (0.51^{-03})	-0.51^{-03} (0.50^{-03})
Potenza	0.61^{-03} (0.60^{-03})	0.61^{-03} (0.47^{-03})	0.60^{-03} (0.45^{-03})
Turin	0.60^{-04} (0.21^{-03})	0.59^{-04} (0.29^{-03})	0.35^{-04} (0.20^{-03})
Log MIXITE			
Bari	0.04 (0.04)	0.03* (0.02)	0.03 (0.02)
Florence	-0.01 (0.01)	-0.01 (0.01)	-0.02* (0.9^{-02})
Naples	-0.31^{-02} (0.02)	-0.32^{-02} (0.02)	-0.30^{-02} (0.01)
Padua	-0.01 (0.8^{-02})	-0.01* (0.7^{-02})	-0.01 (0.7^{-02})
Perugia	-0.02 (0.03)	-0.02 (0.08)	-0.02 (0.17)
Potenza	0.01 (0.02)	0.96^{-02} (0.02)	0.64^{-02} (0.01)

	Model A **OLS**	**Model B** **WLS**	**Model C** **WLS**
Turin	-0.68^{-03} (0.8^{-02})	-0.68^{-03} (0.1^{-02})	-0.26^{-03} (0.11^{-02})
(continues)			
SELFCONT			
Bari	0.07 (0.05)	0.07*** (0.03)	0.06 **(0.03)
Florence	0.04 (0.06)	0.04 (0.03)	0.03 (0.03)
Naples	-0.15*** (0.02)	-0.15*** (0.02)	-0.14*** (0.02)
Padua	0.13*** (0.03)	0.13*** (0.03)	0.14*** (0.03)
Perugia	0.09** (0.04)	0.09*** (0.02)	0.08*** (0.02)
Potenza	0.12*** (0.04)	0.12*** (0.03)	0.11*** (0.03)
Turin	-0.08*** (0.01)	-0.08*** (0.02)	-0.08*** (0.02)
SHAREPUB			
Bari			-0.42^{-0} (0.31^{-03})
Florence			-0.54^{-03} (0.32^{-03})
Naples			0.30^{-03} (0.32^{-03})
Padua			0.67^{-03} (0.61^{-03})
Perugia			0.13^{-02}** (0.46^{-03})
Potenza			0.75^{-03}*** (0.23^{-03})
Turin			-0.15^{-02}*** (0.29^{-03})
N° obs.	734	734	732
R^2 adj	0.72	0.77	0.79
F-test	39.21***	52.04***	49.72***
Wald-test on restrictions	$p<0.00$***	$p<0.00$***	$p<0.00$***

Note: The weights are determined as the number of observations related to each of the seven underlying urban areas. Standard errors are given in parentheses. Significance is indicated by ***, ** and * for the 1, 5, and 10 percent level, respectively.

Table 5-10: Cross-section analysis of the mobility impact index 1991 for subgroups of cities based on geographical location.

	Model A **OLS**	**Model B** **OLS**
Dependent variable:	IMPACT91	IMPACT91
Independent variables:		
Intercept β_{North}	0.37*** (0.01)	0.39*** (0.94^{-02})
β_{Centre}	0.25*** (0.02)	0.25*** (0.03)
β_{South}	0.21*** (0.01)	0.20*** (0.02)
DISTANCE		
North	-0.88^{-03}***(0.12^{-03})	-0.74^{-03}***(0.12^{-03})
Centre	-0.43^{-03}* (0.25^{-03})	-0.44^{-03}* (0.25^{-03})
South	-0.24^{-03}*** (0.83^{-30})	-0.23^{-03}*** (0.82^{-30})
DENSITY		
North	-0.18^{-04}*** (0.43^{-05})	-0.15^{-04}*** (0.43^{-05})
Centre	-0.17^{-04}* (0.11^{-04})	-0.16^{-04} (0.11^{-04})
South	0.16^{-05} (0.15^{-05})	0.16^{-05} (0.15^{-05})
RURAL		
North	-0.72^{-03}*** (0.79^{-04})	-0.68^{-03}*** (0.78^{-04})
Centre	-0.30^{-03} (0.23^{-03})	-0.27^{-03} (0.25^{-03})
South	-0.18^{-03} (0.11^{-03})	-0.19^{-03}* (0.11^{-03})
GROWTH		
North	0.47^{-04} (0.20^{-03})	0.67^{-05} (0.20^{-03})
Centre	0.21^{-03} (0.52^{-03})	0.19^{-03} (0.51^{-03})
South	0.46^{-03}* (0.24^{-03})	0.48^{-03}** (0.23^{-03})
Log(MIXITE)		
North	-0.01*** (0.24^{-02})	-0.73^{-02}*** (0.24^{-02})
Centre	-0.01 (0.01)	-0.01 (0.01)
South	-0.92^{-02}* (0.54^{-02})	-0.94^{-02}* (0.53^{-02})
SELFCONT		
North	-0.09*** (0.11^{-03})	-0.10*** (0.01)
Centre	0.07** (0.03)	0.07** (0.03)
South	-0.07*** (0.02)	-0.07*** (0.02)
SHAREPUBB		
North		-0.98^{-03}*** (0.21^{-03})
Centre		0.12^{-03} (0.49^{-03})
South		0.17^{-03} (0.22^{-03})
Nobs	734	732
R^2-adjusted	0.66	0.68
F-test	73.84***	68.16***
Wald-test on restrictions	197.74***	572.49***

Note: The weights are determined as the number of observations related to each of the seven underlying urban areas. Standard errors are given in parentheses. Significance is indicated by ***, ** and * for the 1, 5, and 10 percent level, respectively.

Table 5-11: Cross-section analysis of the Mobility Impact Index 1991 for subgroups of cities based on level of polycentrism

	Model A OLS	Model B OLS
Dependent variable:	IMPACT91	IMPACT91
Independent variables:		
Intercept β_{Polyc}	0.22*** (0.02)	0.22*** (0.21)
β_{Metro}	0.39*** (0.88^{-02})	0.43*** (0.01)
DISTANCE		
Polycentric	-0.33^{-03}*** (0.11^{-03})	-0.33^{-03}*** (0.10^{-03})
Metropolitan	-0.79^{-03}*** (0.13^{-03})	-0.45^{-03}*** (0.12^{-03})
DENSITY		
Polycentric	-0.23^{-04} (0.15^{-04})	-0.23^{-04} (0.13^{-03})
Metropolitan	-0.18^{-04}*** (0.15^{-05})	-0.13^{-04}*** (0.15^{-03})
RURAL		
Polycentric	-0.56^{-03}*** (0.18^{-03})	-0.56^{-03}*** (0.17^{-03})
Metropolitan	-0.97^{-03}*** (0.85^{-04})	-0.70^{-03}*** (0.81^{-04})
GROWTH		
Polycentric	0.77^{-03}* (0.43^{-03})	0.77^{-03}* (0.39^{-03})
Metropolitan	0.51^{-04} (0.21^{-02})	0.14^{-05} (0.19^{-03})
Log(MIXITE)		
Polycentric	0.99^{-03} (0.72^{-02})	0.99^{-03} (0.66^{-02})
Metropolitan	0.2^{-03} (0.29^{-02})	0.53^{-03} (0.26^{-02})
SELFCONT		
Polycentric	-0.13*** (0.23)	0.13*** (0.02)
Metropolitan	-0.14*** (0.13)	-0.13*** (0.01)
SHAREPUB		
Polycentric		-0.22^{-02} ***(0.02)
Metropolitan		0.57^{-05} (0.30)
Nobs	734	732
R^2-adj	0.45	0.55
F-test	47.69***	60.61***
Wald-test on restrictions	151.97***	169.27***

Note: The weights are determined as the number of observations related to each of the seven underlying urban areas. Standard errors are given in parentheses. Significance is indicated by ***, ** and * for the 1, 5, and 10 percent level, respectively.

5.6. Causal chain model of mobility impact

Moving on from the results presented in the previous section, we now try to enrich our analysis by envisaging a conceptual causal chain that explains the relationship between sprawl and travel impacts. In Figure 5-3, mobility impacts are seen as the result of the influence of three main territorial dimensions: structural, economic, and social. We posit that the causal chain that explains travel impacts originates from the structural dimensions of cities. Sprawl, with its low densities and spatial segregation of productive and residential activities contributes to move job opportunities to peripheral areas. This reduces the self-containment capacity of cities, so that congestion virtually follows jobs to the periphery and increases travel demand. This traffic increase uses up all the available road capacity within and across cities, creating higher congestion. If not accompanied by investment in transportation, able to keep pace with the growth in travel demand, the quality of public transport services worsens. Average trip time increases and workers' travel choices favour private motorised modes, with higher social costs. Travel impacts and their social costs increase.

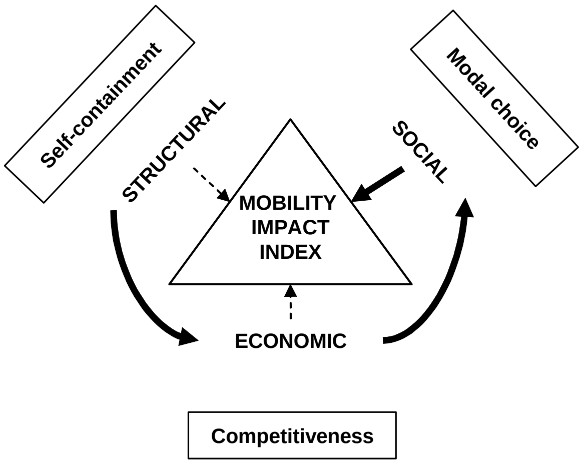

Figure 5-3: Causal chains in the explanation of mobility impact

In short, we argue that cities of relatively compact structure and good functional mix will be characterised by higher self-containment capacity, and will generate more favourable conditions for public transport competitiveness (in terms of journey-to-work time). This will contribute to move people's preferences towards public transport and, consequently, reduce the impacts of urban mobility. From this conceptual interpretation, we proceed to the econometric analysis in order to find some empirical evidence. With this aim, we employ a methodology based on Causal Path Analysis (CPA) (for an in-depth description, see, e.g., Bollen, 1989). This analysis formulates the model as a path diagram, in which arrows connecting variables define the structure of the conceptual framework, and allow the estimation of reaction parameters, i.e. essentially the regression coefficients. The arrow diagram is presented in Figure 5-4. It contains the structure of the causal path that we want to test.

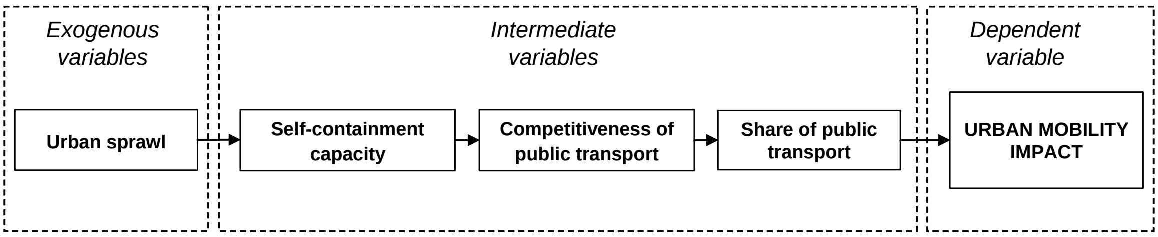

Figure 5-4: A general model for urban mobility impact estimates

On the left-hand side of Figure 5-4, the variables that control for sprawl (densities, functional mix and consumption of agricultural land) are treated as the *exogenous* variables of our model. These influence the *intermediate* variables (self-containment capacity, competitiveness of, and accessibility to, public transport) that, ultimately, explain variations in the endogenous variable. The *endogenous* variable of our model, the mobility impact, is on the right-hand side. This is the dependent variable of our model, which closes our assumed chain of causal relations. We use the Generalised Least-Squares (GLS) method to run the CPA and test the validity of our model. GLS allows us to construct a model of linear equations, in which a given variable can behave both as an independent variable (in one equation) and as a dependent variable in a subsequent equation. We can therefore estimate regression coefficients in simultaneous regression models. Under the assumption that each variable has been standardised to unit variance and mean zero, the value assumed by individual parameters represents the order of magnitude of each independent variable in explaining the following dependent variable. The statistical significance of each parameter is given by the values of the T-student test run in parallel with the coefficient estimation analysis.

We employ the usual three parameters to control for urban sprawl (DENSITY, MIXITE and RURAL). One latent variable is included for each territorial element of our conceptual model. SELFCONT, COMPUB and SHAREPUB are, therefore, employed to control for the structural, economic, and social dimension of cities, respectively. The impact of urban mobility is estimated by travel impacts for 1991 (IMPACT91). The results are presented in Figure 5-5. The causal direction of correlations is given by arrows, with coefficients and T-values in brackets. All estimated parameters are highly statistically significant and show the expected signs. As densities and diversity of land use increase, while consumption of agricultural land decreases, the level of self-containment becomes higher. This signals that, as expected, sprawl contributes to increase the need of commuting, since activities become more dispersed and segregated. On the other hand, having less need to travel within and outward from cities reduces congestion and creates more favourable conditions for a good quality of public transport. This supports the use of public transport for commuting and results in lower travel impacts.

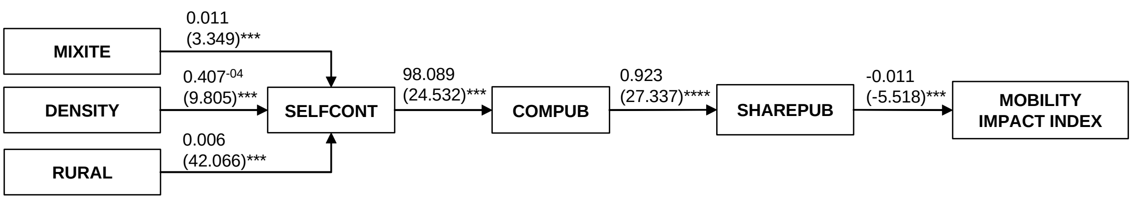

Figure 5-5: Estimated causal path analysis model for Italy

Note: T-statistics are provided in brackets. Significance is indicated by ***, ** and * for the 1, 5, and 10 percent level, respectively.

5.7. Concluding discussion

Sprawl, with its wide dispersion of metropolitan areas and the spread of cities with high consumption of scarce resources, is a relatively recent phenomenon in Europe. This has moved the debate on the most preferable form of urban development towards new issues. One first point that still requires analysis concerns the actual extent of the alleged impacts of sprawl. Is sprawl a problem and, if yes, how serious is it, relative to other problems facing communities and countries? Second, what are the comparative advantages and disadvantages of various proposed solutions to solve the sprawl problem? In this chapter, we have offered a contribution to the former question, focusing on the impacts that sprawl exerts on the transport system of cities. Using a mobility impact index (IMPACT) based on commuting data on 739 Italian cities, we empirically analysed the dynamics and the determinants of travel impacts.

The results show that, during the decade 1981-1991, the impact of mobility has increased in Italy by up to 37 percent. This increment has been generated by a marked shift of modal choices towards private motorised travel modes: namely, the automobile. The increased dependence on the automobile vis-à-vis the reduction of other – more environmentally friendly – means of travel constitutes a relevant policy problem. As the share of motorised trips rises, environmental travel impacts and their costs rise too, leading to a decrease of collective welfare. These include air and water pollution, waste, barrier effects, noise, and the cost of parking and accidents that are not paid by the transportation user and, therefore, affect larger groups of people. However, even if the environmental impact of private transport in urban areas become more and more evident, and shows severe consequences for human health too (e.g. increased occurrence of bronchitis, asthma attacks, increased number of hospitalisation, morbidity and mortality[51]), people still keep on using cars. It seems, therefore, that the benefits provided by private transport, in terms of comfort and "emancipation" from the public transport service, are very high. This phenomenon, in which we see people defending their private benefit, often disregarding the collective one, would require a thorough analysis. How much do people lose if we introduce some restrictions on private vehicle use? But also, how much do people lose if we do not? Even so, as private behaviour is involved, if alternative transport modes, as attractive as cars, are not fed into the mobility market it is unlikely that this issue will be politically addressed in the short run.

In addition, we examined the connection between the IMPACT and some specific dimensions of cities using multivariate cross-section regression analysis. In particular, we considered factors that control for the level of sprawl: density, diversity of land use, and consumption of ex-urban agricultural land. Our analysis provided robust results that confirmed our a priori expectations. In our models, all coefficients have the expected sign and are statistically significant. Significant and negative coefficients of DENSITY, MIXITE and RURAL show that sprawl contributes to higher travel impacts. Less compact and mixed-use cities result in higher impacts, since the greater dispersion of activities in sprawl increases automobile dependency and makes it necessary to spend more time travelling between activities. Yet auto use itself also encourages sprawl. It requires large amounts of land for transportation facilities and makes the development of the urban fringe much easier. Furthermore, our results suggest that, as the segregation of productive and residential activities increases with sprawl, workers need to travel longer and the self-containment capacity of cities is hampered. Ceteris paribus, this shifts congestion from the core toward the periphery of the transport system, resulting in the lower quality of transport services.

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[51] WHO, 1999.

The above-mentioned results are consistent with those of cross-section regression analyses run on subsamples based on geographical location and level of polycentrism. For the usual model specification, the results show that parameters maintain the expected signs. Nonetheless, Wald-tests on restrictions show that there are significant differences in the elasticities of explanatory parameters for such subsamples.

Finally, we proposed a conceptual interpretation of the causal chain that links sprawl to travel impacts and used Causal Path Analysis to test it. The results confirm our expectations. Sprawl, with its low densities and spatial segregation of productive and residential activities contributes to move job opportunities to peripheral areas. This reduces the self-containment capacity of cities, so that congestion virtually follows jobs to the periphery and increases travel demand. The increased congestion uses up all the available road capacity within and across cities, creating higher congestion. If not accompanied by investment in transportation, able to keep pace with the growth in travel demand, the quality of public transport services worsens. Average trip time increases and workers' travel choices favour private motorised modes, with higher social costs. Travel impacts and their social costs increase. These results, therefore, seem to refute for Italy the thesis that sprawl reduces congestion by spreading out trips over more routes, as argued by some commentators (see TRB, 1998, pp.70). Congestion, instead, seems to have followed jobs to the suburbs. Since jobs have moved to areas where there is a poorer public transport service, people have no choice but to drive to these jobs. Overall, sprawl results in an exacerbation of the environmental impacts of urban mobility.

PART III: RURAL ENVIRONMENT

6. VALUING PESTICIDE RISK: A COMPARATIVE APPROACH*

Since the 1950s, chemical-based strategies have been the preferred form of pest control in agriculture, contributing to an unprecedented growth in agricultural production and productivity (Pimentel, 1978; Pimentel and Greiner, 1997). Two decades later, starting at the end of the 1970s, the on-farm benefits of pesticide use started to be weighed against concerns over the off-farm costs of pesticide risks to human health and the environment (Pimentel et al. 1992; Swanson and Vighi, 1998). This wider perspective prompted many regulatory agencies, at both national and international levels, to implement a variety of pesticide risk management policies, ranging from liability rules to market-based instruments, and from command and control approaches to incentives for voluntary action, including moral persuasion. Still, the management of pesticide risks is a difficult task for policy makers (see Smith et al., 1998). The negative sideeffect of pesticide use is multidimensional, and managing such risks implies trade-offs between stocks at risk to be protected (i.e. risk targets), as well as between different types of potential impacts. Moreover, insight into intricate cause-effect relationships is necessary in order to model the phenomenon and to predict its temporal and spatial dynamics. In such situations where risks are multidimensional and trade-offs between them are particularly subtle, and where information on causes and mechanisms is incomplete or uncertain, the trade-offs between risks and benefits should be made explicit and expressed in a way that allows direct comparisons.

Recently, in many countries, the call for a formal appraisal of pesticide policy costs and effectiveness – using one of the established procedures of cost-benefit or cost-effectiveness analysis – has become one of the main responses to this issue. For instance, in the European Union – traditionally less accustomed than the USA to cost-benefit approaches – formal appraisal procedures have been improved and economic valuation has enjoyed a revival (Pearce, 1998; Pearce and Seccombe-Hett, 2000; Matheus and Lave, 2000). An emerging interest within the empirical economics literature is also visible (Söderqvist, 1998; Press and Söderqvist, 1998; Mourato et al., 2000; Schmitz and Ko, 2001, Schmitz and Brockmeier, 2001; Falconer and Hodge, 2001; Arker and Shogren, 2001; Schou et al., 2002).

The availability of detailed monetary estimates of individuals' willingness-to-pay (WTP) for changes in pesticide risks stemming from the implementation of alternative policies is pivotal for such formal assessment of policies. WTP information provides a basis for comparing changes in environmental and human risks on the same basis as the financial costs and benefits of any other project or policy (Pearce and Seccombe-Hett, 2000). In other words, WTP can provide information on the level of environmental protection that is socially desirable, on

* *Based on Travisi, Nijkamp and Vindigni (2006c).*

the level of human health risk that is socially acceptable, and – within a cost-benefit framework – on the expected level of potentially excessive costs in terms of both private and public expenditure. The relative importance of each pesticide risk, as measured by the individuals' WTP for declined risk exposure, is therefore crucial in policy design to handle the multidimensionality of pesticide risks and to properly guide behaviour of decision makers and governments.

In this chapter the problem of pesticide risk valuation in economics is introduced, presenting a detailed discussion of the main theoretical and practical issues encountered in this task. The current state-of-the-art is presented and critically discussed with the use of a comparative analysis based on some artificial intelligence techniques.

6.1. The monetary value of changes in risks from pesticides

The valuation of changes in pesticide risks should reflect the preferences of the economic actors exposed to the risk. The actors include producers applying pesticides in support of agricultural production and productivity, and consumers of products – either fresh or processed – that have been produced using pesticides at some stage of the production process, as well as the broader group of all those affected by the environment. Pesticide use may pose negative side-effects in specific environmental dimensions that should be reflected in the use and non-use values of the environment[52]. An overall economic valuation of the changes in pesticide risks requires, at least in principle, the assessment of both potential human health and ecological hazards. Figure 6-1 presents a valuation framework in which available risk valuation techniques are integrated with scientific information on risks in order to provide empirical estimates (and predictions) of the monetary value attached to declines in risks. Note that the economic valuation process is here considered subordinate to the risk characterisation process, in terms both of the quality and uncertainty of the outcomes, although usually cost-benefit references list them as interdependent but separate processes (Pearce and Seccombe-Hett, 2000). Finally, the value transfer procedure acts as a link between different techniques, since it has the potential for predicting monetary values from completed valuation studies for use in unexplored contexts[53] (see Bal and Nijkamp, 2001). This section will conclude with a discussion of research design issues linked to the valuation of the benefits of pesticide risk decrease.

[52] A recent document by STOA (1998) identifies four main risk groups or protection targets in the European region to be considered in a risk-based approach. These are environmental quality and biodiversity; water resources; occupational and consumer human health; and agricultural production.

[53] Although its application remains controversial and has been debated by economists, it is fair to note that uncertainty in the transfer exercise usually originates from both the monetary valuation and the scientific data. In many cases, risk data and prediction models contain most uncertainty and require stronger theoretical assumptions or simplifications (Matthews and Lave, 2000; Navrud and Bergland, 2001).

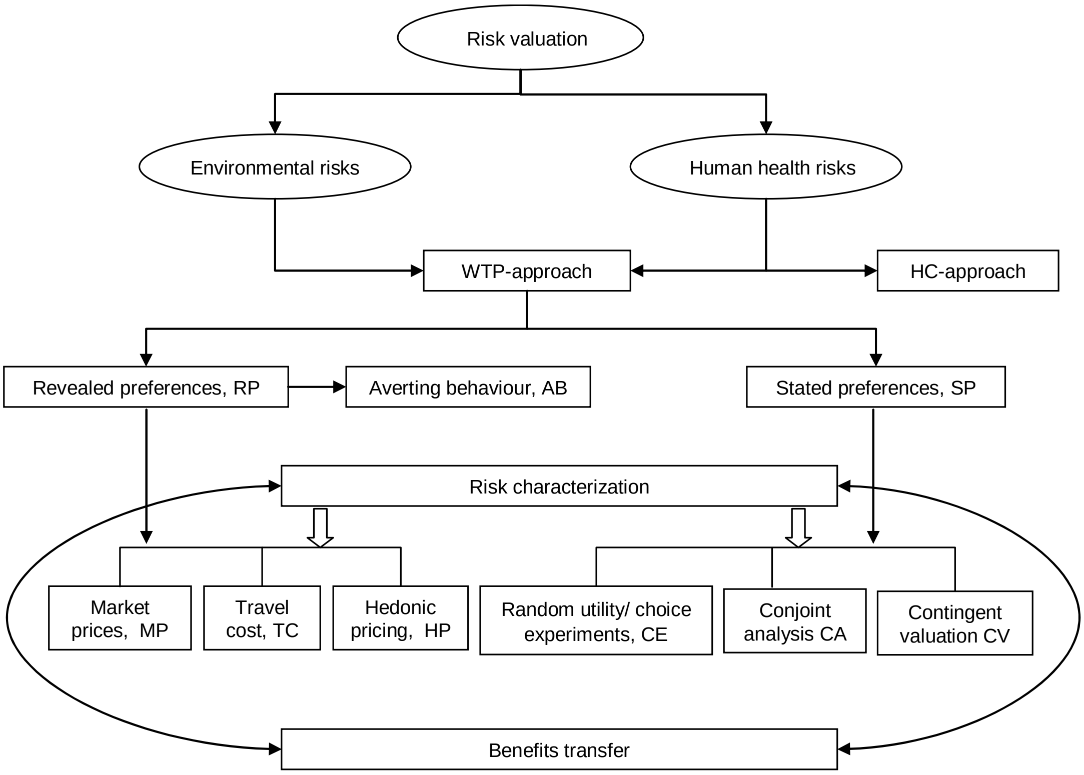

Figure 6-1: Available valuation techniques for environmental and human health risk changes

Note: Modified from Pearce and Seccombe-Hett, 2000.

One of the more controversial questions in this field is whether economists can handle the sound scientific information that is also suitable for evaluative purposes. This is a complicated issue because, to some extent, the manner in which risk changes can be valued depends upon the information available from the scientific risk assessment[54]. At the simplest level, an assessment may only provide qualitative outcomes indicating the risk level (say negligible versus unacceptable)

[54] Procedures exist for risk assessment in most OECD countries, some dictated by national requirements and others by international requirements, such as for the EU (see Solomon, 1996; McCarty and Power, 2000; Power and Adams, 1997; OECD, 2000). However, notwithstanding this well-established context, the output of the risk assessments can be different, in terms either of the quality or the informational nature of their output (i.e. providing quantitative or qualitative results). According to EU procedures (Council Directive: 91/414/EEC; 67/548/EEC; 93/67/EEC; 76/769/EEC), for instance, there are guidelines for *hazard* identification; effects dose-response assessment; exposure assessment and *risk* characterisation. The dose-response assessment identifies 'zero damage' thresholds, the PNECs – *Predicted No Effect Concentrations* – based on extrapolations from test data to the environment. Exposure assessment calculates the expected concentration of the chemical concerned in different environmental compartments, e.g. the PEC – *Predicted Environmental Concentration*.

Hazard characterisation involves the comparison of the PEC with the PNEC and is associated with the reasonable *worst-case scenario*, so as to guarantee the highest level of protection. There is a *hazard* (potential risk) if concentration in the PEC exceeds the PNEC. Actual *risk* occurs only if an environmental system is exposed to that PEC level. The results are then expressed as a risk/hazard quotient, providing semi-quantitative information. The assessment is performed differently in other countries: for example, in Canada and the US, the aim of the risk assessment is to provide the basis for a fully-quantified risk analysis, presented in the form of the probability of occurrence of a particular effect given a certain level of exposure (USEPA, 1998; USEPA, 2000; CSA, 1996).

on the basis of adequate exposure level information. At a more detailed level, the assessment may be able to determine, for a certain target population, the dynamics of the likely number of fatalities or deaths occurring per year. The latter case would allow the semi-quantitative valuation of changes in risks. Finally, where outcomes from an overall risk assessment exist and are expected to have a tolerable level of uncertainty, a quantitative monetary valuation of risks might be performed. Environmental economists are charged with the task of looking for the best available scientific information on pesticide risk for the subsequent economic assessment (for a discussion, see Nijkamp et al., 2002). Estimates of the alteration in well-being can be taken from the biomedical, toxicological and eco-toxicological literature – namely, risk assessments, dose-response and production functions – to predict changes in environmental balance (for pesticides, changes in some risk endpoints[55]: carcinogenity, neurotoxicity, theratogenesis, acute and chronic exposure, etc.). Cost-benefit or risk-benefit references usually mention dose-response or exposure-response functions (for environment and human health, respectively), production functions and expert assessments as possible sources of information about the risk scenario under consideration. However, when dealing with pesticides, whenever possible, economists should rely on the results of pesticide risk assessment procedures (for a discussion, see McCarty and Power, 2000; Power and Adams, 1997; OECD, 2000). Unlike production functions, risk assessment is based on an *ex ante* stance (precautionary principle), which is preferable when handling risk and uncertainty. Like dose-response functions, risk assessment describes a cause-effect relationship between the dynamics of ecological and health effects and chemical exposure levels. In addition, its implementation procedure provides several advantages (USEPA, 1998, 2000). First, uncertainty analysis is usually performed in order to consider the degree of confidence of the assessment explicitly, thus providing a basis for comparing the quality of the results. Moreover, where dose-response functions typically deal with one relation at a time, risk assessment involves a number of dose-response relations, either treated as independent or interdependent. Finally, the iterative nature of the process allows the progressive incorporation of new information whenever it becomes available.

We now come to the economic side of the framework presented in Figure 6-1. Broadly speaking, the economic literature offers two alternative approaches to risk valuation: the *human capital* (HC) approach and the *willingness-to-pay* (WTP) approach. Whereas the first is suited specifically to human health valuation – being based on individual productivity – the second has a foundation in welfare economics and also is sufficiently flexible for valuing risk to natural and agro-ecosystems. The HC approach stems from the idea that the value of an individual is equal to the value of her/his contribution to total production and assumes that a measure can be inferred from her/his earnings. Such a premise, however, has some significant drawbacks and its application is therefore not recommended when one is looking for an inclusive valuation with a strong theoretical basis. First of all, it is inconsistent with the individualistic foundation of welfare economics, since it does not take popular preferences about changes in health risks into consideration.

[55] An (eco)toxicological *endpoint*, for a vegetable or animal species, is usually defined as a certain level of pollution at which a certain (eco)toxicological effect is expected to happen. For a chemical, an (eco)toxicological *endpoint* is usually expressed as the concentration ([μg/l] or [μg/kg]) at which an (eco)toxicological effect is expected to be macroscopically detectable (namely, LD_{50}/EC_{50} and NOAEL/LOAEL for acute and chronic toxicity, respectively).

Moreover, indirect damage to health and injuries – both of directly affected persons and of their relatives – are neglected, as well as the statistical values of retired people. Attempts to overcome such disturbing shortcomings based on simple adjustments of the HC estimates can be useful, but are still insufficient to compensate for the welfare issue (see Johannesson and Johansson, 1998). On the other hand, the theoretical foundations of the WTP measures of risk changes have been explored since the 1970s and nowadays have a solid background (see among the others Schelling, 1968; Mishan, 1971; Jones-Lee, 1976; Rosen, 1988; Cropper and Freeman, 1991; Viscusi, 1993; Johansson, 1995). The monetary value of a decrease in pesticide usage and the associated risks can be expressed as the aggregate individuals' willingness-to-pay for a pesticide risk reduction or, alternatively, the willingness-to-accept (WTA) compensation for exposure to increase risk levels. WTP (and WTA), therefore, reflect preferences, perceptions and attitudes towards risk of the economic actors affected by policy decisions to reduce pesticide usage, implying that the WTP for a risk reduction can vary across different hazardous scenarios (Sjoberg, 1998, 2000). The risk valuation literature typically assumes that preferences can be depicted by continuous and smooth utility functions, where the total WTP is a strictly increasing concave function of the initial risk level and the level of risk reduction[56] (Grossman, 1972; Jones-Lee, 1976). The downward-sloping relationship between the marginal WTP and the risk of experiencing an event with detrimental effects of pesticide usage can conveniently be interpreted as a demand function for improved agricultural safety – i.e. human health and environmental quality – depending on the baseline risk level and the level of pesticide risk at stake. The impacts of pesticide usage will therefore be represented alternatively as health risks and/or the risk of degradation of agro-ecosystems: for instance, acute or chronic human intoxication, surface and groundwater pollution, threats for farmland biodiversity, and loss of natural habitats.

The WTP (or WTA) concept can be empirically captured and measured using two basic approaches, one involving *stated* preferences – i.e. preferences conveyed by a response to a question – the other one involving *revealed* preferences – i.e., preferences inferred from the behaviour of an individual making choices about some good or option implicitly connected to the attribute being valued (for a complete overview of the WTP literature, see, e.g., Branden and Kolstad, 1991; Hanley and Spash, 1993; Freeman, 2003). Both stated and revealed preference techniques have their pros and cons. Only stated preference techniques are capable of capturing the *non-use* values of environmental goods, while revealed techniques simply provide their instrument-related worth. The latter, therefore, are focused on the capital values of environmental goods (either direct or indirect uses), while the former also capture values stemming from existence per se. Revealed preference data are often hampered by lack of data on the choice set considered by the actor, and the actor's perception of risks. In addition, econometric difficulties such as multicollinearity can severely affect the estimation of trade-offs between monetary attributes and safety improvements. These problems can be circumvented using stated preference techniques, although individual's responses can then be influenced rather strongly by the contents and the way in which contextual information is presented. A more general issue, relevant to both techniques, is that respondents may have cognitive difficulties perceiving information on uncertain

[56] Strong empirical support for these assumptions exists, though they have been occasionally refuted too (see, e.g., Smith and Desvouges, 1987).

events, especially when their probability of occurrence is small, as for pesticide hazards. The information provided during stated choice experiments can help to guide the respondent to a proper understanding of the good being valued, and of the size of the safety amelioration offered (Slovic, 1987).

Notwithstanding these conceptual drawbacks, an extensive empirical economic literature on pesticide risk valuation has emerged over the two last decades (see Table 6-1). The WTP estimates available in this literature refer to the effects of different types of pesticide risks, in particular to impacts on human health, and to damages to environmental agro-ecosystems. Because of the historically human-driven rather than environmentally-driven interest of pesticide risk management, economists too have been concentrating their efforts more on human rather than environmental consequences of pesticide usage, and the literature therefore focuses primarily on the valuation of health effects on consumers and farmers (see, e.g., Pingali et al., 1994; Crissman et al., 1994; Antle and Pingali, 1994, Roosen et al., 1998; Thompson and Kidwell, 1998; Blend and van Ravenswaay, 1999; Fu and Hammitt, 1999; Wilson, 2002). Significantly fewer studies address the ecological dimension of pesticide risk (see, e.g. Higley and Wintersteen, 1992; Beach and Carlson, 1993; Mullen et al., 1997; Söderqvist, 1998; Lohr et al., 1999; Foster and Mourato, 2000; Brethour and Weersink, 2001; Cuyno et al., 2001).

The food safety literature centres on the valuation of human health risks associated with the presence of pesticide residues in fresh food, typically using *stated* preference approaches (Figure 6-2). Most studies refer to the US, given the importance of food safety policy there (see, e.g., Misra et al., 1991; van Ravenswaay and Hoehn, 1991a,b; Baker and Crosbie, 1993; Eom, 1994; Bubzy et al., 1995; Roosen et al., 1998). Occasionally, the valuation concerns a cost-benefit analysis of the reduction or ban of a specific pesticide compound (Bubzy et al., 1995; Roosen et al., 1998). Alternatively, the valuation is more marketing-oriented and focuses on consumers' WTP for certified residue-free produce or fresh products certified for integrated pest management (see, e.g., Ott, 1990; Ott et al., 1991; Misra et al., 1991; van Ravenswaay and Hoehn, 1991a; Baker and Crosbie, 1993; Eom, 1994; Blend and van Ravenswaay, 1999). More recently, the study of pesticide risks extends to pesticide health risks for farmers in developing countries (Wilson 2002). Higley and Wintersteen (1992), Mullen et al. (1997), and Brethour and Weersink (2001) extend the focus of the pesticide risk literature by including the valuation of changes in integrated pesticide risk management on the environment in addition to considering acute and chronic human toxicity for farmers. Their environmental targets include groundwater and surface water, aquatic species, avian species, mammals, and arthropods. Cuyno et al. (2001) improve on this approach in order to avoid double counting by distinguishing fewer environmental categories corresponding to non-target organisms at risk. Finally, Foster and Mourato (2000) and Schou et al., 2002 combine the analysis of human health effects and the environment by employing contingent ranking techniques to determine the WTP for the reduction of human health effects, and loss of farmland biodiversity. To our knowledge, the study by Foster and Mourato (2000), in which the authors use a loaf of bread as the payment vehicle, is the only one referring to processed food as the medium for pesticide residue.

Table 6-1: Overview of studies providing empirical WTP estimates for pesticide risk reductions

Study	*Data Country*	*Measurement unit: value per*	*Risk class[(a)]* Human health	Environment	*WTP class[(b)]* Human health	Environment
Anderson et al. (1996)	1994 US	person, produce unit	Consumers	/	Low	/
Baker and Crosbie (1993)	1992 US	person, produce unit	Consumers	/	Low	/
Blend and van Ravenswaay (1999)	1998 US	person, produce unit	Consumers	/	Low	/
Brethour and Weersink (2001)	1993 Canada	household, month	Farmers	Non target systems	High	High
Buzby et al. (1995)	1995 US	person, produce unit	Consumers	/	Low	/
Crissman et al. (1994)	1994 Equador	person, year	Farmers	/	High	/
Cuyno et al. (2001)	1999 Philippines	household, crop season	Farmers	Non target systems	High	High
Eom (1994)	1990 US	person, produce unit	Consumers	/	Medium	Medium
Foster and Mourato (2000)	1996 UK	person, produce unit	Farmers	Biodiversity	High	High
Fu et al. (1999)	1995 Taiwan	person, produce unit	Consumers	/	Medium	/
Govindasamy and Italia (1997)	1997 US	person, produce unit	Consumers	/	Low	/
Govindasamy and Italia (1998)	1997 US	person, produce unit	Consumers	/	Low	/
Hammitt (1993)	1985 US	person, produce unit	Consumers	/	Medium	/
Higley and Wintersteen (1992)	1990 US	person, acre application	Farmers	Non target systems	Medium	Medium
Huang (1993)	1989 US	person, produce unit	Consumers	/	Low	/
Lohr et al. (1999)	1990 US	person, acre application	Farmers	Non target systems	Medium	Medium
Misra et al. (1991)	1989 US	person, produce unit	Consumers	/	Low	Low
Mullen et al. (1997)	1993 US	household, month	Farmers	Non target systems	High	High
Ott (1990)	1990 US	person, produce unit	Consumers	/	Low	/
Ott et al. (1991)	1990 US	person, produce unit	Consumers	/	Low	/
Owens et al. (1997)	1995 US	person, produce unit	Farmers	/	Low	/
Pingali et al. (1994)	1991 Equador	person, year	Farmers	/	Medium	/
van Ravenswaay and Hoehn (1991a)	1990 US	person, year	Consumers	/	Low	/
van Ravenswaay and Hoehn (1991b)	1989 US	person, year	Consumers	/	Low	/
Rosen (1998)	1998 US	person, produce unit	Consumers	/	Low	/
Roosen et al. (1998)	1998 US	person, produce unit	Consumers	/	Low	/
Thompson and Kidwell (1998)	1994 US	person, produce unit	Consumers	/	Medium	/
Weaver et al. (1992)	1990	person, produce unit	Consumers	/	Low	/

Note:

[a] See Figure 6-2 for the mnemonics referring to the different risk classes.

[b] Standardised WTPs are expressed in 2002 USD per person per year. Standardised mean estimates of WTP are categorised in low, medium, or high levels (Low: $0<WTP<5$; Medium: $5\leq WTP<15$; High: $WTP\geq 15$).

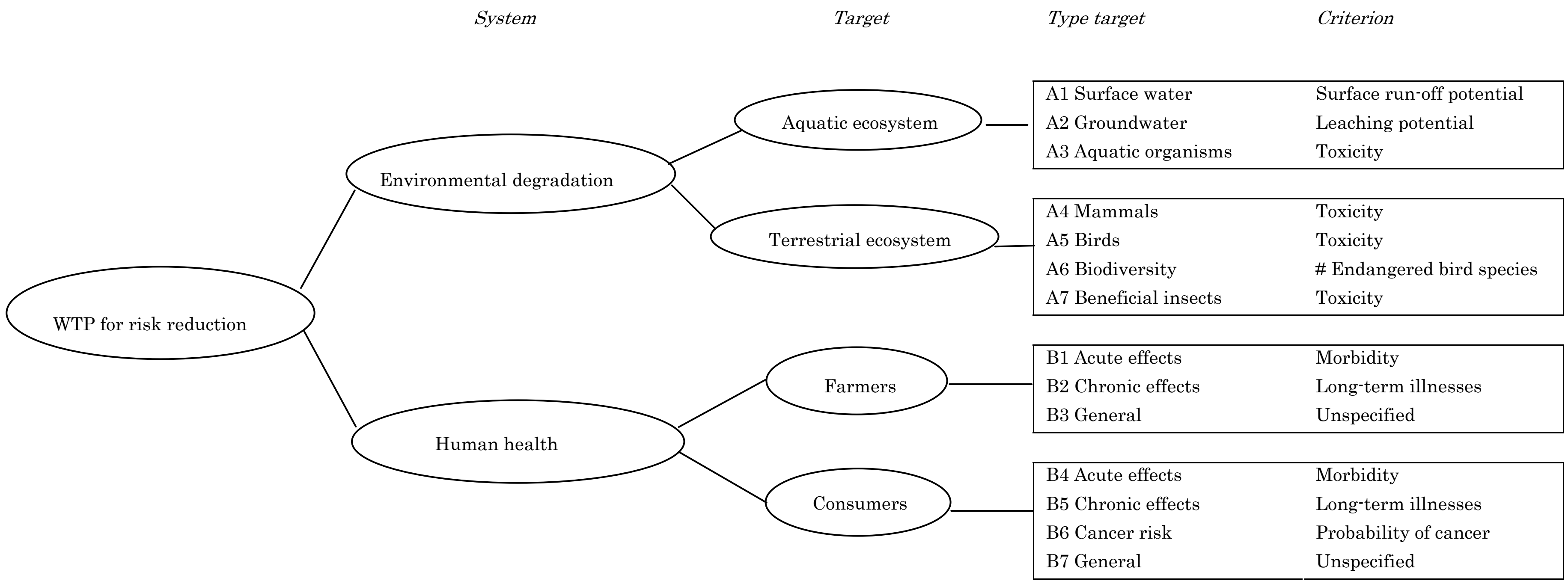

Figure 6-2: Taxonomy of WTP estimates for pesticide risk reduction according to system, target, type, and criterion

Considerably fewer studies rely on *revealed* techniques. Pingali et al. (1994), Crissman et al. (1994), and Antle and Pingali (1994) apply averting/defensive expenditures methods to estimate risks for farmers in the Philippines and Ecuador, while Hammit (1993) relies on the market price method for the estimation of a range of pesticide risks for consumers. Beach and Carlson (1993) and Söderqvist (1998) use the hedonic price method for valuing risk reductions for groundwater.

Human health deterioration and environmental degradation caused by pesticide usage are intrinsically heterogeneous, because targets, exposure mechanisms, and toxicological and eco-toxicological endpoints vary substantially. In order to facilitate the interpretation of the empirical results in the literature, we propose a taxonomy of available WTPs for pesticide risk reduction. Figure 6-2 provides a schematic overview. Starting from the distinction between WTPs for a decline of human health hazards and degradation of agro-ecosystems, we increase the detail of the classification up to the definition of subsets of risk reduction benefits with analogous targets and endpoints.

The class referring to environmental degradation includes WTPs of pesticide risk reduction with respect to various non-target ecosystems. The term 'non-target ecosystems' is used to indicate all living organisms that can be reached and spoiled by pesticides, with the exception of pests specifically intended to be destroyed by the pesticide applications. We distinguish two different targets, aquatic and terrestrial ecosystems, and within those ecosystems, several different types of non-target organisms.

WTP estimates concerning the reduction of pesticide hazards for human health refer either to direct effects on farmers, or effects on consumers due to the ingestion of fresh produce that has been produced using pesticides. Pesticide hazards for farmers are typically related to direct contact with pesticide compounds or, in general, to risks induced by field exposure. Detrimental health effects on consumers are caused by the presumed presence of pesticide residues in produce, specifically fresh fruit and vegetables. In both cases, WTPs can be related to either acute or chronic health effects, caused by pesticide poisoning and long-lasting exposure to low concentrations of pesticides, respectively. The risk of developing cancer is considered explicitly in some studies, although with different specifications. Cancer hazard associated with ingestion of pesticide residues is frequently directly evaluated (that is, it is explicitly mentioned in the valuation question), whereas the hazard related to field exposure is often analysed indirectly by characterising chronic risks using information deduced from carcinogenity and theratogenesis tests.

Table 6-1 provides a synthesis of the major findings of the empirical pesticide risk valuation literature. For the 28 studies we consider, the situation of a rather heterogeneous literature described above – in terms of risk type, valuation techniques, research design, and behavioural assumptions – has an empirical manifestation in the differences in WTPs' magnitude reported in the literature. In view of the open issues discussed above and of the substantial variations within and between studies' outcomes, subsequent sections will give attention to a specific type of comparative analysis in order to obtain preliminary insight into the cause-effect pattern between underlying sources of heterogeneities across pesticide valuation studies and the range of empirical WTP monetary values.

6.2. Comparative analysis of the value of pesticide risk

In recent years comparative analysis methods based on artificial intelligence have been increasingly applied in order to perform a formal analysis of a set of results derived from separate studies in various fields of application, but nevertheless similar in their use of theoretical frameworks pursuing the same objectives. By attempting to cope with the difficulties of pointing out relations among a set of data, classification methods represent a rigorous alternative to the narrative discussion of empirical studies (see, e.g., Vollet and Bousset, 2002).

Apart from Nijkamp and Pepping (1998) and van den Bergh et al. (1997b) focusing on the effectiveness of pesticide price policies, to our knowledge, no other meta-analysis oriented comparative study on pesticide risks exists. In this chapter decision-tree induction − a particular comparative technique belonging to multidimensional classification methods − is applied. Multidimensional classification methods based on artificial intelligence techniques have been developed to order the information contained in a multivariate database so as to discover structural relationships between class characteristics and relevant attributes of the phenomena to be classified. With respect to the valuation of pesticide risk, the phenomenon of concern is the monetary assessment of decline in pesticide risk consequent to reducing pesticide usage in agriculture. The empirical economic literature on pesticide risk valuation provides the information necessary to describe such phenomena. Class characteristics correspond here to different types of WTP estimates (as distinguished in Figure 6-2), while relevant attributes correspond to those elements expected to have a role in explaining the phenomenon dynamics, i.e. variations in WTP estimates. The information provided by the sample of studies is codified into a multidimensional data set according to the set of attributes selected, which can be conveniently viewed as criteria for codification. The multidimensional classification approach therefore focuses on identifying the relationships between the WTP for reductions of various pesticide negative impacts on the various environmental dimensions − called the effect size − and the underlying sources of variation across empirical studies − what we call explanatory or moderator factors. Typical moderator variables include: risk attitude and perception of respondents; the source and nature of the risk data; characteristics of the research design; and behavioural assumptions of the underlying studies. In the remainder of this section, we detail the comparative approach and discuss the sample of studies. Subsequent sections discuss the potential determinants of WTP valuations for reduced pesticide risks, and provide results of the comparative analysis and conclusions.

The method of decision-tree induction, which belongs to the class of multidimensional classification methods − such as neural network analysis, fuzzy set analysis, rough set analysis and decision tree analysis − is widely and increasingly used for classification purposes (Quinlan, 1993). This method aims at analysing and predicting the membership of a class by the recursive partition of a multidimensional data set into more homogeneous subsets (for details, see Breiman et al., 1984). This leads to a hierarchical decision-tree structure where instances are classified by sorting them down the tree from the root node to a leaf node. Each node in the decision-tree specifies a test for an attribute (explanatory factor) of the instance concerned (i.e. the WTP value), and each branch descending from the node corresponds to one of the possible values for this attribute. An instance is classified by starting at the root node of the decision tree, testing the attribute specified by this node and moving to the next node down the tree branch that corresponds to the value of the attribute. This process is then repeated at the node on this branch, and so forth, until a leaf node is reached. A number of systems exist for inducing classification trees from examples, e.g. CART (Breiman et al., 1984), ASSISTANT (Cestnik et al., 1986), and C4.5 (Quinlan, 1993). Of these, C4.5 is one of the most well-known and popular decision-tree systems. The MS Windows implementation of C5, named See5, was used in our experiments.

In a decision-tree algorithm, the most important component is the method used to assess splits at each internal node of the tree. Often the information theory approach, which examines entropy in relation to the information contained in a probability distribution, is employed. The aim is then to select the attribute that is most useful for classifying instances, based on the 'information gain', a measure of the goodness-of-separation for a given attribute used for the training examples according to their classification (for details, see DeFries and Chan, 2000). Entropy is then used as a measure of the reduction of disorder when ordering a set of variables in a data set with respect to different classes. By interpreting information gain as a measure of the expected reduction in entropy, we can – by considering the next node down – define a measure of the effectiveness of an attribute in classifying the training data, caused by positioning the instances according to this attribute. The process of selecting a new attribute and positioning the training examples is then repeated for each non-terminal descendent node, this time using only the training examples associated with the node concerned. Attributes that have been incorporated higher in the tree are excluded, so that any given attribute can appear at most once along any path in the tree.

Formally, the information gain of an attribute is computed by means of the corresponding entropy expression. Given a training data set T, composed of observations belonging to one of k classes {C1, C2 … Ck}, the amount of information required to identify the class for an observation in T is :

$$Info(T) = -\sum_{j=1}^{k} \frac{freq(C_j, T)}{|T|} \log_2 \left(\frac{freq(C_j, T)}{|T|} \right), \qquad \textbf{Eq- 6-1}$$

where freq(Cj,T) is equal to the number of cases in T belonging to class Cj, and |T| is the total number of observations in T, i.e. the average amount of information required to define the class of a sample from the set T. In terms of information theory, the set T is called entropy. The same estimate, after separation of the set T with X, is provided by the following expression:

$$Info_X(T) = \sum_{i=1}^{n} \frac{|T_i|}{|T|} Info(T_i). \qquad \textbf{Eq- 6-2}$$

Then, the criterion of the attribute choice is defined as:

$$Gain(X) = Info(T) - Info_X(T). \qquad \textbf{Eq- 6-3}$$

This criterion is calculated for all attributes and the one that maximises Gain(X) is selected. This latter attribute is the test used in the current tree node, and will be used for further tree derivation.

The empirical literature retrieval process in our comparative experiment started with checking several economic databases (among others EconLit), and was subsequently extended by reference chasing, and by approaching the main authors in the field over e-mail. A series of relevant keywords, such as 'willingness to pay', 'pesticide', 'food-safety', 'environmental risk', and 'human health risk', were used because of the multidimensionality of pesticide risks. This retrieval process resulted in a set of slightly more than 60 studies in both published and unpublished sources. Subsequently, we identified a subset of 28 studies containing monetary estimates. WTPs extrapolated from the sample set up the effect size to handle in the comparative approach. A listing of the studies and their main characteristics is presented in Table 6-1. The studies were published during the 1990s and early 2000s, and they predominantly deal with the situation in the US. Almost 250 observations refer to human health, of which approximately one-fifth is concerned with farmers and the rest with consumers, in particular with the unspecified general health hazard to consumers. Approximately one-third of all observations refer to detrimental effects on ecosystems, with slightly more observations pertaining to aquatic as compared with terrestrial ecosystems. Single mean estimates from each study were considered to avoid a multi-sampling bias. Table 6-1 also shows that comparing effect sizes for different target types, countries, and time-periods comes with an operational problem, because the effect sizes have to be transformed to a common measurement unit, and a common currency in prices of a given year. From here on, all WTP figures are presented as standardised effect sizes in USD 2002 per person per year, conveniently categorised into low, medium, or high level for the comparative approach (Table 6-1).

6.3. Possible sources of systematic WTP variation

A comparative approach can clearly help to present a critical, systematic overview of the pesticide risk valuation literature with a multidisciplinary perspective, and to determine the effect of a number of underlying factors on the empirical results presented in this literature. In the comparative analysis, the standardized mean WTP estimates are used as the effect size, and variables capturing theoretical expected differences, methodological issues, behavioural assumptions, and features related to the study design are used as moderator factors. The relationship is established using the decision-tree algorithm described in Section 6.2 above. In this section, we discuss potentially important explanatory factors and their operationalisations, which are also presented in Table 6-2.

Table 6-2: List of potentially usable moderator variables and their frequencies within the data set

Type of variable	*Variable*	*Description*	*Frequency*
Effect size	WTP	Willingness-to-pay estimate, per person, per year, in US$ 2002	
Target type	HUMAN HEALTH	Pesticide risks for human health	22/27
	FARMER	Risk for farmers due to occupational exposure	9/27
	CONSUMER	Risk for consumers due to the presence of pesticide residues in food	18/27
	ENVIRON	Pesticide risk for the environment	7/27
	NTARGET	Risk for environmental non-target agricultural ecosystems	6/27
	BIODIV	Risk for biodiversity	1/27
Study aim	AIMSCIEN	Scientific aim	23/27
	AIMPOLIC	Policy aim	4/27
Risk scenario	IMPLICIT	Implicit scenario	7/27
	ACTUAL	Actual scenario	8/27
	POTENTIAL	Potential scenario	12/27
Risk assessment	EXANTE	*Ex ante* risk assessment	3/27
	EXPOST	*Ex post* damage assessment	12/27
	GENERIC	Generic assessment	12/27
Risk information	TOXIC	Toxicological and eco-toxicological end-points	7/27
	RISK	Risk estimates	5/27
	DAMAGE	Damage estimates	3/27
	GENERIC	Generic information	12/27
Risk perception	SUBJPERC	Subjective perception	7/27
	OBJPERC	Objective perception	8/27
Risk source	ONEPEST	One pesticide	12/27
	ALLPEST	All pesticides	15/27
Type of product	FRFOOD	Fresh food (fruit or vegetable)	22/27
	PRFRVEG	Processed food	1/27
	CROPS	Crops	4/27
Method	CVM	Contingent valuation method	8/27
	CHOICEXP	Choice experiments (conjoint analysis, contingent ranking, choice modellin	15/27
	REVPREF	Revealed preferences	4/27
Sampling features	RESPFARM	Farmer respondents	6/27
	RESPCONS	Consumer respondents	17/27
	RESPSTRA	Stratified respondents	4/27
Type safety device	ECOLAB	Eco-certification	18/27
	IPM	Integrated pest management or low input agriculture	8/27
	BAN	Ban on specific pesticides	1/27
Payment vehicle	PVPREM	Price premium	21/27
	PVBILL	Separate bill	2/27
	PVYIELD	Yield loss	2/27
	MEDEXP	Medical expense	2/27
Type data	SURVMAIL	Mail survey	12/27
	SURVFACE	Face-to-face survey	4/27
	RETAIL	Retail data	9/27
Geographical locatior	CANADA	Canada	1/27
	PHILLIP	Phillipines	1/27
	UK	United Kingdom	1/27
	EQUAD	Equador	2/27
	TAIWAN	Taiwan	1/27
	USA	United States of America	21/27

An important aspect of our data sample and, more in general, of this literature concerns the differences across WTP estimates in terms of the nature of the risk being valued, represented by different target types in the taxonomy given in Figure 6-2. The main distinction we propose distinguishes between estimates linked to human health threats and degradation of the environment, where the former and the latter can be interpreted, respectively, as the individual (private) and the collective (public) dimension of the benefits of reducing pesticide negative side-effects. What role this distinction plays with respect to WTP is a particularly intricate subject, since it has been largely neglected in empirical studies, with the exception of a few sporadic cases (Jones-Lee, 1985, 1991, 1992; Johannesson et al., 1996). Nevertheless, the microeconomic choice theory underlying WTP estimation suggests that WTP for private goods may be expected to be higher as compared with that for public ones, because of the well-known free-riding behaviour inherent to collective welfare improvements (Johannesson et al., 1996), although the available empirical literature leads to partially misleading results[57]. In our comparative analysis, we introduce a moderator factor controlling for heterogeneities according to target types.

Because of the intrinsically subjective nature of WTP estimates, another debated point is how people's perceptions of risk affect their preferences for environmental risk reduction, if they do at all. In the sociological and psychological risk perception literature, there is a widely shared consensus that individuals have difficulty dealing with uncertain events, especially when their probability of occurrence is low, as with pesticide risks (Slovic, 1987; Magat et al., 1988; Viscusi and O'Connor, 1984). The direct consequence of such a stance is that, once we are modelling choice processes, individuals cannot be assumed either to perfectly know scientific risk estimations or to accurately perceive the risks with respect to expert information or to news coverage. An understanding of the dynamics of individual risk perception is needed. To investigate this, the *attitude-before-behaviour* paradigm is usually accepted as the conceptual framework for depicting the relationship between perceptions, attitudes, and behavioural intentions. The effect of a number of explanatory factors on risk perception is thus studied; typically, these include: socio-economic and demographic characteristics of the involved population; popular attitudes about uncertain events; and concerns about the ongoing risk scenario (Slovic et al., 1990; Alhakam and Slovic, 1994; Sjoberg, 1998, 2000). In the experimental design of the comparative analysis, we can assess the importance of this perception issue, looking into the way in which different studies model people's risk perception. We can also include moderator factors controlling for the type of risk information provided to the respondents in the surveys. Specifically, we can control for: the type of risk scenario presented to the respondents (i.e. an actual, potential or implicit scenario); differences in the source of pesticide risk (one specific pesticide or pesticides in general); and the health risk vehicle (one specific fresh food or fresh food in general). Unfortunately, we could not collect complete socio-demographic profiles for the majority of the studies included in the comparative analysis. Therefore, the analysis can only include a limited amount of information on this issue. Specifically, we can use attributes

[57] Jones-Lee (1985) show that the value of a statistical life (VOSL) increases by about a third if a paternalistic or safety-oriented altruistic attitude of respondents is included. Johannesson et al. (1996) come up with the opposite result, showing that, for some types of altruism, people may be willing to pay more for a private risk reduction than for a uniform risk reduction of the same magnitude.

indicating which stakeholders were considered in the valuation exercise, distinguishing between consumers, farmers or stratified samples, and also we can include attributes identifying the geographical location and the time period of the underlying studies.

Because of previous psychological arguments, for three decades economists have been analysing how individuals' valuation of risk varies with the level of baseline risk, either objectively or as perceived. The conventional hypothesis assumes that the estimated marginal valuation of a risk change increases with an increase in the initial risk level. More specifically, the total WTP is assumed to be a strictly increasing concave function of the level of risk reduction (see Jones-Lee, 1976), a hypothesis also supported by several empirical results[58]. This argument is expected to play an important role in explaining existing differences between estimates and should be taken into account along with other more conventional moderator variables, such as income level (Miller, 2000). In the present survey, however, the high degree of heterogeneity among the approaches adopted for risk characterisation, as well as the variety of risk groups and endpoints within the data set, made it unfeasible to determine an endogenous and comparable initial risk level. A further attempt would require splitting up the data set according to the specific risk group concerned or, alternatively, using exogenous information to determine a comparable initial risk level for each case study in the sample.

An important methodological difference between the studies concerns the valuation technique. Approximately one-quarter of the effect sizes stems from revealed preference (RP) approaches, while the rest originates from stated preference (SP) exercises. The well-known expectation is that SP studies exhibit higher WTP estimates as compared with RP studies. In the comparative analysis we can control for this by considering the specific methods used in the study (as in Figure 6-1) and include, in addition, information on the type of payment vehicle (price premium, separate billing, or yield loss), and the type of data that was used (retail versus survey data).

Finally, we would like to guide the reader's attention to a crucial, although often disregarded, point in the economic valuation of environmental risks. This can be traced back to the apparently trivial observation that the valuation exercise is subordinate to the assessment of the environmental or human health risks, since the information provided by the latter represents the *conditio-sine-qua-non* of the former. Consequently, the choice among different valuation techniques (Figure 6-1), as well as the quality of results, is strictly related to the nature of the information available from the ecosystem risk characterisation. To cope with this, in addition to the previously mentioned economic aspects, the comparative approach explores how different empirical studies have dealt with the risk assessment issue. Specifically, we can control for differences in the approaches adopted to assess risk (*ex ante* versus *ex post*), the type of toxicological and eco-toxicological endpoints and information considered, and the related risk nature of the risk scenario.

The number of potentially relevant control factors and their frequencies within the data set are presented in Table 6-2. As already stated, the majority of

[58] To be fair, we should note that some detailed empirical tests have rejected this theoretical assumption (see, e.g., Smith and Desvouges, 1987). Nevertheless, the hypothesis of the concave nature of the WTP-Risk level function is still dominant.

the surveys focus on human-related negative side-effects of pesticide usage. Particular attention is given to consumers' rather than farmers' preferences. Understandably, consumer risk – related to the involuntary ingestion of pesticide residues in fresh products – prevails over occupational exposure, although today experts are more troubled by pesticide occupational exposure than by food risk via pesticide residue ingestion (STOA, 1998). In this sense, the economic literature seems to suffer from the same type of misperception of pesticide risks that characterises lay people[59]. Only those surveys that consider the farmers' side deal with the valuation of pesticide risks for both human health and the environment, i.e. agro-ecosystems and farmland biodiversity. The others simply consider pesticide impacts on human health. A possible interpretation of this trend is that when researchers are concerned with comparing the trade-offs between on-farm and off-farm pesticide effects for farmers, they are less inclined to also address, simultaneously, the individual (pesticide residue risk) and collective (environmental risks) dimensions of economic actors' preferences. The majority of studies refer to actual rather than potential risk scenarios[60]. Nevertheless, in most cases the approach adopted for risk characterisation is generic, a fact that suggests a lack of suitable scientific information or economists' lack of focus on the linkages between risk characterisation and its economic estimation. Of those studies using sound scientific information, the *ex ante* approach is applied more frequently than the *ex post* one, suggesting that the precautionary principle is more widely recognised. In particular, among other risk endpoints, acute and chronic toxicity indicators occur more frequently than carcinogenity, and theratogeneity. None of the studies consider chemical sensitivity or immune suppression effects. The risk perception variables display a balance between studies that assume that people's perception of risks and technical risk estimates coincide and those that do not. However, despite its theoretical relevance, a considerable part of the sample totally neglects this point. Specifically, SP studies usually model risk perceptions by using a subjective paradigm, while RP surveys make the opposite assumption. In terms of valuation methodology, researchers seem to be prone to choose SP more often than RP techniques. In particular, when referring to an actual risk scenario, contingent valuation and choice experiments are favoured over RP approaches, notwithstanding the hypothetical bias issue concerned with SP approaches[61]. The greater flexibility and the theoretical strength that characterizes stated techniques are therefore attractive to researchers, who seem to be more comfortable with a tool able to cope also with the behavioural basis of individuals' preferences concerning pesticide risk. Moreover, SP surveys are generally supported by a more solid scientific attitude with respect to risk characterisation. For SP studies, the *ex*

[59] An analysis of the background of the national policies on pesticides in several European States shows that public concerns about overall pesticide risks are considerable (STOA, 1998). Contamination of drinking water ranked as the top concern in all countries, followed by concerns about possible adverse effects on ecosystems. Anxieties about risks to human health, both from pesticide residues in food and exposure to residues in water, soil and air, were ranked third, and risks to users came next (Goldenman, 1996).

[60] The actual or potential nature of the risk scenario stems from the difference between *hazard* and *risk* described in Footnote 3. A study refers to an actual scenario of risk if it considers an environmental system that is actually exposed to a hazard. Otherwise, we assume that the study refers to a potential risk scenario.

[61] As observed by Owens (Owens et al., 1997), hypothetical bias might undermine the credibility of some of the surveys of farmers' demand for safer pesticides belonging to our sample (Higley and Wintersteen, 1992; Mullen et al., 1996).

ante approach is most often employed, whereas RP surveys are generally based on an *ex post* perspective. The former consider either information on risk exposure and the related toxicological and (eco)toxicological effects, or information pertaining to risk probabilities; the latter mostly refer to damage measures.

6.4. Results and performance assessment of the comparative approach

The comparative analysis based on decision-tree analysis shows appealing results, which are graphically represented in Figure 6-3. The analysis provides a description of the relationships between effect size and explanatory factors in the form of a classification tree. However, the pattern-class relationship expressed in the tree can also be written as a set of rules in the following way. Each rule consists of a Statistics (n, lift x) or (n/m, lift x) that summarises the performance of the rule (see, Table 6-3). Specifically, n is the number of training cases covered by the rule, while m, if it appears, shows how many of them do not belong to the class predicted by the rule. The rule's accuracy is estimated by the Laplace ratio $(n-m+1)/(n+2)$. The lift x is the result of dividing the rules' estimated accuracy by the relative frequency of the predicted class in the training set.

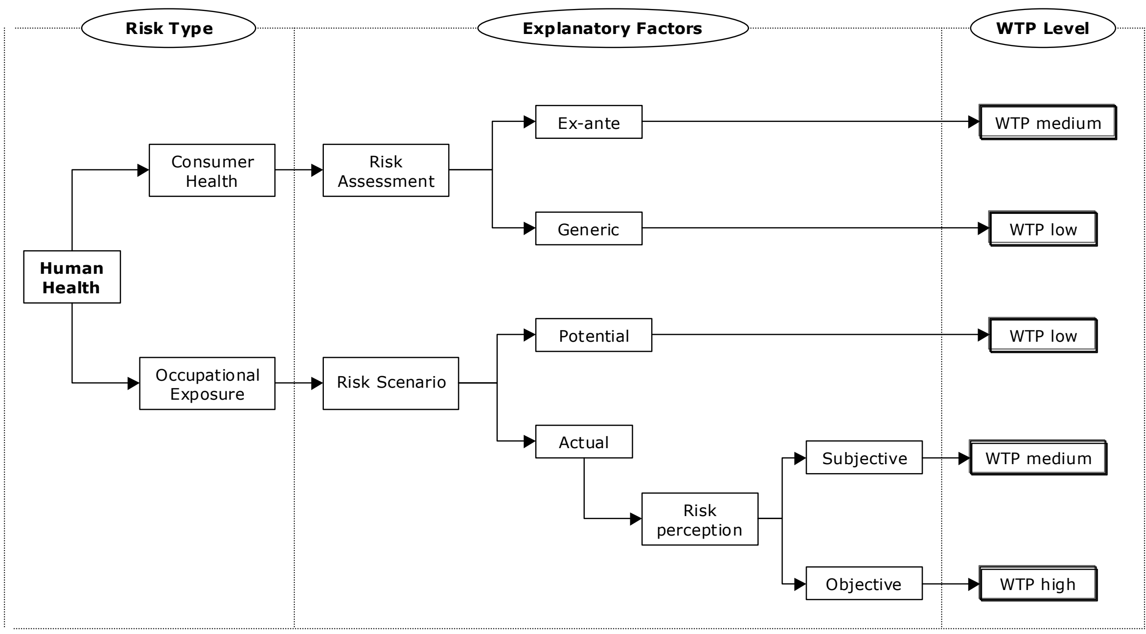

Figure 6-3: Decision tree resulting from application of C5/See5 algorithms

Table 6-3: Extracted rules and performance of the analysis

Extracted rules			
Rule-1	(13/1, lift 1.7) Approach = generic ⇒ WTP low [0.867]	Rule-3	(6/2, lift 2.1) Hum = consumers' health Approach = *ex ante* ⇒ WTP medium [0.625]
Rule-2	(4, lift 2.8) RISKSCEN = actual RISKPER = subjective ⇒ WTP medium [0.833]	Rule-4	(9/4, lift 2.9) Hum = occupational exposure ⇒ WTP high [0.545]
Default class: low			

Evaluation of training data (27 cases)				
Decision tree			*Rules*	
Size	Errors		Nº	Errors
5	4(14.8%)		4	4(14.8%)
(a)	(b)	(c)	⇒ classified as	
12	2		(a): class low	
1	6	1	(b): class medium	
		5	(c): class high	

In Sections 6.2 and 6.3 we formulated the hypothesis that the value of benefits stemming from a decrease in pesticide risks is intimately linked to the nature of the risk being valued, in accordance with the risk taxonomy proposed in Figure 6-2. A similar expectation originated from the plain evidence of substantial differences among various potential pesticide impacts: namely risk targets, exposure mechanisms, and endpoints. We now get a confirmation of this from the comparative analysis of the empirical literature that estimates pesticide-related WTP values, which indicates that the risk-type attribute is able to explain most of the entropy endogenous to the data set, as it appears as the first node of the decision tree. In respect to this, our prime hypothesis indicated a distinction between the individual (human safety) and the collective (ecological) dimensions of benefits stemming from reduced pesticide exposure. Unfortunately, considerably fewer studies in the data set focus on the ecologically-detrimental effects of pesticides, so that the comparative analysis is not able to capture this broader distinction among risks very well. The results instead clearly show the effect of different human health risks on empirical WTP estimates. Specifically, the analysis shows that a pesticide risk related to either consumers' or farmers' health leads to a first split of empirical benefits estimates, corresponding in Figure 6-3 to the upper and lower branch of the tree, respectively. At this stage, it is interesting to note that where the subsequent node in the former branch is the attribute controlling for differentiating in the risk characterisation approach, the determinant in the latter branch is the nature of the risk scenario. How should we interpret this structure? As already noted, biomedical and toxicological research on the effects of human exposure to pesticides (consumers in particular) has a solid background, since most major chemical risk policies were designed first of all to protect people's safety. In Section 6.2, we hypothesized that the more detailed the scientific information on risks available to economists, the greater the effect of this element on the valuation exercise (see Nijkamp et al., 2002). In other words, we expect that when economists deploy scientific information on pesticide risks, then the uncertainty level and nature of risk information will affect the output of the valuation exercise; whereas, by contrast, more generic information will not influence valuation estimates as much. In this sense, it is not surprising that attributes controlling for risk information appear just after risk-type 1. The decision tree shows that an economic valuation that relies on risk information from *ex ante* risk assessment leads to higher WTP levels than one that relies on generic risk information. From this perspective, the hypothesis that lack of adequate scientific information levels out differences in WTP estimates is confirmed. This rule is covered by 13 instances (only 1 instance is misclassified) and has an accuracy estimated to be 86 percent (see Table 6-3).

In the lower branch of the tree, the driving factor is the nature of the risk scenario. As with the risk characterisation attribute, we hypothesised that the potential effect of this feature on WTP estimates would run via its influence of individuals' perception. People are expected to underestimate the severity of negative events described in terms of potential rather than actual risk scenarios. In accordance with this argument, the lower branch of the tree shows that the potential nature of the risk leads to lower WTP estimates than actual risk scenarios. For actual risks, the comparative analysis indicates that the attributes which control for different behavioural assumptions become relevant. The decision rule, with an accuracy estimated at 83 percent, identifies the actual nature of the risk scenario and the subjective paradigm on risk perception as determinant features for a medium WTP. With respect to occupational exposure, subjective risk perception leads to lower estimates of the benefits from decreasing pesticide exposure. To explain this result, one should carefully look into the factors that the sociological and psychological literature considers to affect risk attitudes and perceptions. In a further meta-analytical exercise, therefore, the socio-economic and demographic features of the sample concerned should be included among major explanatory variables.

We have outlined above the mechanism through which the errors' classification affects the rules formation. In order to explore which class distribution will yield the best classifier, we use the 'two-performance measure', i.e. the *classification accuracy* (or error rate) and the *confusion matrix*. Classification accuracy is the most common evaluation metric in machine-learning research. In our system the estimated error at 14% is significantly low. However, using accuracy as a performance measure assumes that the class distribution is known and, more importantly, that the error costs of incorrectly classified instances are equal. Accuracy may be particularly problematic as a performance measure when the data set studied is biased in favour of a majority class (Weiss and Provost, 2001). An alternative method is to analyse the confusion matrix that offers better insight into the classification and misclassification distribution. A confusion matrix contains information about actual and predicted classifications made by a classification system (Kohavi and Provost, 1998). The performance of such systems is commonly evaluated using the data in this matrix. Table 6-4 shows the confusion matrix for a tree classifier, where rows are classes available for use in the classification process and columns are classes chosen during the classification, respectively.

The entry in the confusion matrix represents the number of instances of the row class which have been classified as members of the corresponding column class. Misclassifications occur when the row and column classes of a cell do not match. If the intersection across predicted and actual classes of different levels is empty (or zero), then no misclassification occurs. The results in Table 6-4 show that 12 instances of the known class 'low' were correctly classified using the generated rules as members of the class low; 1 instance of the class 'low' was incorrectly classified using the generated rules as members of the class 'medium'. Considerably fewer instances (6) of the known class 'medium' were correctly classified using the generated rules as members of the class 'medium'; 2 instances of the class 'medium' were incorrectly classified using the generated rules as members of the class 'low'. Finally, 5 instances of the known class 'high' were correctly classified using the generated rules as members of the class 'high'; while one instance of the class 'high' was incorrectly classified using the generated rules as members of the class 'medium'.

Table 6-4: Confusion matrix

Predicted				
class low (a)	*class medium* (b)	*class high* (c)		
12	2		*class low* (a)	**Actual**
1	6	1	*class medium* (b)	
		5	*class high* (c)	

6.5. Conclusions

When we started exploring the economic literature addressing pesticide human-health and ecological-risk valuation, we noticed that – despite several attempts by economists – this research area still suffers from hardly any communication with the environmental sciences, which clearly frustrates research efforts and policy goals. Economic and scientific principles tend to be treated in a separate way and – when interaction is possible – the integration is rather fragmentary, which makes it hard to give an unambiguous interpretation of the available results. Moreover, the environmental dimension of pesticide risk is still partly neglected in the literature, although an overall economic valuation of such risks would require, at least in principle, an assessment of both the human and the environmental impacts. This unbalanced state can be traced back to the human-driven rather than the environmentally-driven historical background of chemical risk management, which still has notable repercussions in the development of valuation surveys.

With this in mind, we have therefore sought to offer a critical overview of the literature on pesticide risk valuation in the light of a multidisciplinary perspective. Different interpretational modes are involved in both the theoretical and the empirical part of our work. On the one hand, we have looked into the scientific background of environmental risk evaluation (i.e. human health and ecological risk assessments); on the other hand, we had to envisage major controversial issues in environmental risk economic valuation, by exploring the frontier which links these two areas. Clearly, the analysis tends to get rather complicated when exploring the context in which different disciplines meet. Nevertheless, this inclusive approach allowed us to perform a comparative analysis, which also addresses relevant heterogeneities. Our contribution should thus be interpreted as an effort to provide a critical research synthesis of the pesticide risks valuation literature, which explores the synergies among complementary theoretical and practical aspects involved in this topic (see Gerrad et al., 2002; Suter, 1995).

In this sense, our comparative application is by no means exhaustive and nor it is representative of the role that a number of theoretical or methodological factors might systematically have in affecting the results of a monetary valuation of environmental and human health risks. Indeed, the aggregation of the utilities proposed to compare different empirical outcomes represents a qualitative interpretation of the state of the art. Nevertheless, the analytical method and procedure presented here is systematic in nature and offers an effective framework for learning from previous case studies. The crucial point lies in the interpretation of results rather than in their indiscriminate use for predictive approaches. This methodology appears, therefore, to be consistent with the broad scope of our analysis and it enables interactions across scientific, social and economic aspects of risk valuation to be evaluated.

From this viewpoint, our experience confirms the expectation that the monetary dimension and the quality of the valuation's results is closely connected to the nature of inputs generated in previous risk assessments, as well as to the psychology of risk perception.

The outcomes of our analysis, though preliminary, suggest that the high degree of variability of WTPs is related to both the valuation technique and the data available from biomedical and eco-toxicological literature. The order of magnitude of a WTP estimate is, in fact, related to the specific type of risk and to the nature of the risk scenario considered, as well to lay people's subjective perception of risks. The analysis also suggests that, in the risk valuation process, more systematic attention should be paid to the formulation of exogenous "framing assumptions" and to their implementation in single case studies.

7. THE MULTIPLE VALUE OF REDUCING PESTICIDE RISK: A STATED CHOICE SURVEY IN ITALY*

Modern intensive agriculture produces significant negative side effects that have been broadly documented in the scientific literature (Pimentel et al., 1992; Pimentel and Greiner, 1997). The order of magnitude of these externalities is dealt with in the scientific, political and economic literature on recent agro-environmental regulations, on pesticide and fertilizer-reduction strategies, and on the assessment of the associated economic costs. Challenging questions and new opportunities to provide policy makers with relevant insights on the best option to be developed against pesticide risks are open to discussion. Relevant issues here concern: how to accelerate the implementation of pesticide risk reduction and management strategies; and how to choose, among the range of possible pesticide reduction measures, those actions that are able to provide the highest level of risk abatement at the lowest collective cost.

In this context, the present chapter examines the use of a Choice Experiment (CE) methodology to assess the economic value of pesticide risk reductions[62]. The CE survey took place in Milan with the aim of providing estimates of the willingness-to-pay (WTP) of consumers to achieve improvements in the environmental and health safety of agriculture. This allows us to study in detail the preferences of consumers for alternative fresh food bundles that differ in production practices, in the sense that they are more, or less, dependent on the use of pesticides. To our knowledge, this is the first study on the valuation of pesticide risk reduction conducted in Italy, and it is one of the few studies that use CE to estimate pesticide risks.

The CE application was designed to estimate the value of some important pesticide-related environmental attributes, using a ‘green shopping’ payment vehicle. Respondents were asked to view the various environmental impacts of pesticide use in agricultural production as foodstuff attributes to be taken into account in the purchase decision. The environmental attributes taken into consideration here were: the reduction in farmland biodiversity; the contamination of soil and groundwater in agricultural land; and the health effects of pesticides on the population in general. The monetary attribute used was the monthly food expenditure bill, by means of which it is possible to estimate the marginal value of the other non-market characteristics. The results confirm that, on average, respondents are willing to pay substantial price mark-ups for safer agricultural production, which thus leads to a reduction of pesticide damage.

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* *Based Travisi and Nijkamp (2004).*

[62] The survey also included a Contingent Valuation question in which respondents were asked to report a maximum WTP for eliminating all negative pesticide impacts (for further details, see Travisi and Nijkamp, 2004).

The remainder of this chapter is organized as follows. Section 7.1 presents the current situation concerning pesticide risk management in the EU political context, and explores the potential to use economic valuation methods in general, and CEs in particular, for assessing the benefits of pesticide risk reductions. It explores the use of the Random Utility Model formulation so as to study respondents' behaviour. Section 7.2 presents the survey instruments and describes the interviews conducted with a sample of 484 consumers approached at three shopping malls in Milan, Italy. Section 7.3 links the selected theoretical model to an empirical exercise, using the CE questionnaire and the respective economic valuation exercise. Section 7.4 discusses the range of the economic estimates. Section 7.5 concludes.

7.1. Background

7.1.1. Pesticide risk reduction in the EU political context

Pesticides are chemicals that require particular attention because most of them have inherent properties that make them dangerous to health and the environment. The European Thematic Strategy on the sustainable use of pesticides (currently being developed) identifies a set of policy objectives that will have to be reached in the coming years to achieve a higher level of sustainability in chemical-based agricultural production. Minimizing the hazards and risks to health and the environment from the use of pesticides is a key point in the strategy that will need to be supported by several policy actions. Among other things, the EU strategy includes: encouraging the use of low input or pesticide-free crop farming, particularly by raising users' awareness; promoting the use of codes of good practice; and consideration of the possible application of financial instruments. In this connection, the EU strategy assumes: i) the imposition of penalties on users by reducing or cancelling benefits under support schemes; ii) the introduction of special levies on pesticides to raise awareness of the detrimental effects of over-intensive pesticide use and further reduce reliance on chemical inputs in modern agriculture; and iii) the harmonization of the value-added tax rates for pesticides (which vary between 3 and 25 percent in the various Member States).

In this context, Italian agricultural policy aims to decrease the risks attached to the use of pesticides by providing economic incentives for organic farming and Integrated Pest Management (IPM)[63]. Moreover, the design of eco-labelling for fresh food that is produced with more benign agricultural practices is a major concern for both agribusiness and policy makers in the Italian agricultural sector. Agribusinesses, such as supermarkets and food producers, appear to be interested in estimating consumer demand for a product with additional environmental attributes, while other agribusinesses, such as seed and chemical companies, and technology and equipment dealers, are interested in farmers' WTP for a new eco-product or service (for a discussion, see Luck and Hudson, 2004).

[63] Italy has the third highest level of pesticide consumption in the OECD countries at 13 percent of total purchases, and a rate of consumption of about 7.7 kg of pesticide per hectare of agricultural land treated.

In relation to pesticide policy purposes, economic theory suggests that an efficient incentive or tax should be set equal to the marginal damage associated with pesticide use. Similarly, estimates of individuals' WTP for pesticide risk reduction would provide key information for policy makers in order to introduce price differentials in products, according to the type and severity of pesticide risks related to their production processes. In this perspective, a proper incentive programme for Italian farmers, or the design of eco-labelling, would require an estimation of individuals' WTP for pesticide risk reduction.

In the current panorama, therefore, the availability of an economic estimate of the social benefits of reduced pesticide risk could be pivotal, allowing us to identify the optimal value-added tax rates for pesticides or incentives to use less risky chemicals.

7.1.2. Economic valuation of the benefits of reduced pesticide risks

Over the last two decades, an extensive empirical economics literature on pesticide risk valuation has emerged (for a detail discussion, see Chapters 6 and 8). Despite the relative abundance of surveys that have provided estimations of WTP for the reduction of several pesticide risks, to our knowledge, there are still only a few Conjoint Analysis (CA) approaches to the valuation of pesticide risks in this literature. Foster and Mourato (2000) and Schou et al. (2002) employed contingent ranking techniques to determine the WTP for the reduction of human health effects, and loss of farmland biodiversity. In their survey, Foster and Mourato (2000) estimated the marginal value of reducing risks for bird biodiversity and human health, whereas Schou et al. (2002) valued the benefits of the reduced use of pesticides in field margins with a focus on the biodiversity of partridges. Nevertheless, in that study the use of pesticides was not mentioned to the respondents, who were simply asked to express their preferences for a generic change in biodiversity. More recently, Wikström (2003) used CE to estimate the WTP for sustainable coffee, and Hasler and Birr Pederson (2004) apply CE for valuing groundwater protection. Whereas Wikström (2003) explicitly addresses the issues of pesticide risk reduction in estimating the WTP for certified organic coffee, Hasler and Birr Pederson (2004), like Schou et al. (2002), do not provide an estimation of pesticide risk reduction since they did not tell the respondents the precise cause of the changes in groundwater protection.

The present chapter provides an empirical application of the CE technique to the valuation of pesticide risks in Italy, by considering the main areas of actual risk for the Italian context: biodiversity, soil and groundwater contamination, and human health risks. Together with the study by Foster and Mourato (2000), to our knowledge, this is the only survey that estimates the multiple impact of pesticide risks with a CA approach. Differently from Foster and Mourato (2000), this study not only considers biodiversity and human health risks but also includes an analysis of soil and groundwater contamination. Moreover, in order to improve the consistency with Random Utility Modelling (RUM) and reduce survey complexity, it applies CE instead of contingent ranking. The main features of CE are detailed in the following section (see also Chapter 2).

7.1.3. Valuing alternative agricultural production scenarios

As presented in Chapter 4, our analysis of the responses to the CE questions uses Random Utility Modelling (RUM) (McFadden, 1974, 1986). We posit that, in each of the choice sets, the respondent selects the alternative with the highest indirect utility. In our questionnaire, the CE exercise implies a choice between three alternative scenarios of agricultural production practices (including the status quo), each of which are more, or less, dependent on the use of pesticides and, therefore, lead to different levels of environmental and human health risks. However, more benign agricultural practices are more costly, and a change in the production process is expected to determine an increase in retail foodstuff prices.

Therefore, the agricultural scenarios differ with respect to food cost, effects on farmland bird biodiversity, contamination of soil and aquifers in farmland areas, and threats to human health. We assume that the utility function of alternative *i* for respondent *q* is:

$$V_{iq} = \bar{x}_{iq}\beta + \bar{z}_{iq}\delta + \varepsilon_{hq} \qquad \textbf{Eq- 7-1}$$

where q denotes the respondent; i denotes the alternative agricultural scenario; and $\bar{x}$ is a 1 x 5 vector comprised of: an alternative *B* and an alternative *C*-specific intercept; the effect of the *i*-th agricultural scenario on bird biodiversity for the *q*-th respondent; the contamination of soil and groundwater related to the *i*-th agricultural scenario for the *q*-th respondent; and the effect of the *i*-th agricultural scenario on the health of the general public for the *q*-th respondent. In Equation (1), $\bar{z}$ is a vector of interactions between the three attributes and the individual characteristics of the respondent. β and δ are vectors of unknown coefficients. If the error terms ε are independent and identically distributed and follow the type I extreme value distribution, the probability that alternative i is selected out of S alternatives is:

$$P_{iq} = \frac{\exp(w_{iq}\gamma)}{\sum_{j=1}^{J}\exp(w_{jq}\gamma)} \quad \forall i,j \in S \text{ with } i \neq j, \qquad \textbf{Eq- 7-2}$$

where $\bar{w}$ is a vector containing $\bar{x}$ and $\bar{z}$, and $\gamma = [\beta' \vdots \delta']$. Depending on the assumption about the distribution of the error term, the resulting statistical model is either a conditional logit, a multinomial probit, or a related choice model (Green, 2002). We run MNL (Conditional Logit) models.

7.2. The CE survey on pesticide risks

7.2.1. Statement of the valuation problem

This study aims to assess people's preferences for alternative scenarios of agricultural production methods which lead to a healthier environment (e.g. integrated pest management, organic agriculture), by focusing on the environmental and economic effects they generate. However, the elicitation of the citizens' preferences for, and economic valuation of, alternative agricultural scenarios is complicated for two reasons. First, the negative environmental side effects of pesticide use – such as pollution of soil, surface and groundwater, higher mortality of sensitive animal and insect species, effects on human health, etc – are not bought and sold on regular markets with proper prices. This implies that we need to apply non-market valuation techniques.

Second, even though low-input agricultural practices have recently been applied in Italy, they have not been monitored with regard to their environmental and economic effects, so that we have to resort to SP non-market valuation techniques, i.e. relying on what people say they would do under hypothetical experimental circumstances, rather than studying their actual behaviour. Here, we deploy an SP method, viz. Choice Experiment (CE), which allows us to estimate WTP values for the negative environmental effects of several pesticides. Though under hypothetical circumstances, we are interested in estimating the value of reducing the main actual risks for the Italian context: biodiversity, soil and groundwater contamination, and human health risks. The details of the survey are provided in the coming subsections (7.2.2 and 7.2.3).

7.2.2. Structure of the questionnaire

The questionnaire used in our experiment consisted of three sections. The first section introduced the subject of the environmental side effects of pesticide use in modern agriculture, by using a cost-benefit perspective, which emphasized existing trade-offs between the positive and negative externalities associated with agricultural production based on the use of synthetic inputs. First, we referred to pesticide risks in general and asked respondents their opinion on the current environmental situation and detrimental effects of modern agriculture. Other questions included: i) How serious are environmental problems compared with other problems in our society, and which of these problems deserve higher public investment?; ii) How serious are pesticide problems compared with other environmental problems, and which of these problems should be the priority for public investment?; iii) Which type of pesticide impact is more severe, and why?; iv) Had the respondents ever personally experienced any of these impacts?; and v) How much were they informed about pesticide impacts? We then focused on some specific dimensions of pesticide risk. The questionnaire described the actual Italian situation concerning pesticides, providing information on, and graphical aids to indicate, both their benefits and risks, and giving the reasons for these positive and negative effects. In particular, the questionnaire focused on three environmental dimensions affected by pesticides: farmland biodiversity; soil contamination; and the health effects on an exposed population.

The second section of the questionnaire contained the CE exercise. In this part, the respondents were asked to view the various side effects of pesticide use due to conventional agricultural practices, as food attributes to be taken into account in daily purchase decisions. Preliminary to the CE questions, we informed the respondents that a reduction of pesticide risk exposure is possible by implementing some pesticide management policies, and that the Italian government was about to do this. Policy options consisted of a change in agricultural production practices that were designed to reduce the rate of pesticide application on field, without any change in the products' quality, but this would increase production costs, leading to an increase in retail costs too. We explained how a reduction in risks would be possible; what range of reduction would be achievable; who would provide this reduction; how it would be provided; and the economic effect of such risk reduction to the respondent.

The third section of the questionnaire gathered additional information in order to obtain a clearer image of the respondents' profile, attitudes, socio-economic conditions, and exposure to pesticides.

7.2.3. CE questions

Before introducing the CE questions, we clearly explained to the respondents that the implementation of the pesticide risk reduction policies, which are designed to reduce environmental and human health risks from agriculture, would be costly to the agricultural sector, and that some of the increased production costs would fall on consumers through an increase in retail prices. Respondents were asked to view the various side effects of pesticide use due to conventional agricultural practices as food attributes to be taken into account in daily purchase decisions.

Following the above explanation, the respondents focused on the CE questions. They were instructed to express their preferences with respect to three profiles described in the survey. These profiles corresponded to three alternative scenarios of agricultural practices, each leading to different agricultural foodstuff shopping conditions. These scenarios differed in the level of risks for biodiversity, soil and groundwater contamination, human health, and the associated price. The attributes and the attributes' levels are described in Table 7-1.

Table 7-1: List of the attributes used in the CE value application

1. Expenditure for fresh foodstuff [€/household per month]:	
-	Current [As indicated by each respondent]
-	Plus 50
-	Plus 100
-	Plus 200
2. Human health [N° cases illness/year]:	
-	250
-	150
-	100
-	50
3. Soil and groundwater [% contaminated agricultural land]:	
-	65
-	45
-	25
-	15
4. Biodiversity [N° endangered farmland bird species]:	
-	15
-	9
-	6
-	3

The alternatives were differentiated in terms of food expenditure and environmental sustainability, which described the range of environmental externalities associated with the underlying production process. In choosing relevant and scientifically-justifiable attributes, we were guided by a group of Italian eco-toxicologists, who helped us to identify the main areas of environmental effects of pesticides in Italy, and to select indicator variables that described each environmental effect. Environmental indicators were selected to describe, as accurately as possible, the main areas of well-documented environmental damage in Italy. Attributes were then tested during focus groups. Specifically, we focused our attention on biodiversity, soil and groundwater (here groundwater contamination is considered to be intimately linked to soil contamination), and human health. A related study, by Foster and Mourato (2000), considered human health and biodiversity. The impact on biodiversity is quantified in terms of the number of endangered farmland bird species, while the impact on soil and groundwater is measured using the percentage of farmland areas contaminated by pesticides. The impact on human health is measured in terms of cases per year of acute illness (i.e. leading to hospitalisation), both as a result of work and domestic exposure.

Special attention was given to the selection of the payment vehicle. A common trend among previous studies estimating WTP for reducing pesticide risks is to use, almost exclusively, the price premium for a single food product with particular additional environmental characteristics (the green shopping payment vehicle), compared with a pre-existing substitute (for a discussion, see Florax et al. 2005). This approach reduces the problem of hypothetical bias because respondents have to focus on a good which is private and deliverable. Nevertheless, the results from our pre-test and focus groups showed that respondents have difficulties in understanding the overall cost to them of the proposed policies and tend to overestimate their WTP[64]. For this reason, our final version of the questionnaire asked respondents about their WTP for alternative low-input agricultural scenarios which lead to a reduction of the negative environmental effects of pesticides and to an overall increase in agricultural foodstuff prices. An additional important advantage of this payment vehicle is that the estimation of the overall benefits from pesticide risk reduction is very much simplified and less biased by approximations[65].

We prepared all the combinations of the attribute levels, eliminating implausible or inconsistent ones[66]. The choice sets consisted of three alternative

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[64] In a pilot version of the questionnaire, the payment vehicle was a price premium on a single packet of spaghetti. Respondents were, however, disturbed by a “single green product” perspective. Respondents were not able to clearly understand the yearly cost to them of the pesticide policy. In fact, they were willing to pay a certain price premium on a single packet of spaghetti, but they were not willing to pay the same price increase when this was expressed as a yearly cost.

[65] For instance, to calculate overall benefits from pesticide risk abatement policies, Mourato et al. (2000) had to convert their WTP estimates expressed in pence per loaf of bread, into pounds per household per year, in order to estimate the total number of loaves consumed by each household per year.

[66] The design of the choice sets is consistent with principles of experimental design (Lazari and Anderson, 1994). In particular, we used a shifted or cyclic design, which generally has a superior efficiency compared with other strategies for generating main effects designs. These shifted designs use an orthogonal fractional factorial to provide the basic alternatives for each choice set. Subsequently, the alternatives within a choice set are cyclically generated. The attribute levels of the new alternatives add 1 to the general level of the previous alternative, until it is at its maximum. At

profiles. The first one was fixed and corresponded to the status quo scenario. The status quo was represented by the conventional scenario of agricultural practices, priced at the household's monthly food expenditure level (reported by each respondent), for which each of the aforementioned environmental attributes was set at its current position (i.e. respectively: 15 endangered bird species; 65 percent of farmland areas contaminated; and 250 cases of acute illness per year). The other two profiles varied from card to card and corresponded to agricultural scenarios that provide lower pesticide risk levels. All combinations were asked in roughly equal frequencies.

7.3. Modelling and valuation results

7.3.1. Indirect utility model specifications

In order to operationalise an empirical formulation of the indirect utility function as described by Equation 1, the following two model specifications are examined.[67] Model 1 is the simplest model that we consider to discuss the effect that each of the attributes under consideration have on the respondents' preferences, and therefore on the choice of the agricultural scenario. Formally, we have:

Model 1

$$V = \beta_1 PRICE + \beta_2 BIODIV2 + \beta_3 BIODIV3 + \beta_4 GRWATER2 + \beta_5 GRWATER3 + \beta_6 HEALTH2 + \beta_7 HEALTH3 \qquad \text{Eq- 7-3}$$

In this model formulation, PRICE refers the cost of the policy to the respondents. BIODIV2 and BIODIV3 denote the variables for the level of biodiversity risk reduction. The omitted variable is BIODIV1 that corresponds to the minimum level of biodiversity risk reduction (i.e. of 6 endangered bird species). GRWATER2 and GRWATER3 denote the variables for the level of reduced contamination of soil and groundwater. The omitted variable is GRWATER1 that corresponds to the minimum level of contamination reduction (of 20 percent of contaminated land). Similarly, HEALTH2 and HEALTH3 denote the variables for the level of reduction of human health risks. The omitted variable is HEALTH1 that corresponds to the minimum level of risk reduction (of 100 cases of acute human poisoning). Ceteris paribus, β_1 can be interpreted as the coefficient of the cost of the pesticide policy to the respondent. β_2 and β_3 provide the effect of a decrement of the biodiversity risk by 9 and 12 endangered bird species, respectively, on the probability to choose an agricultural policy. β_4 and β_5 provide

this point, the assignment returns to the lowest level. We started, therefore, from a set of 81 possible permutations of the hypothetical agricultural scenario ($3\ \text{levels}^{4\ \text{attributes}}$). Then we generated the 'fractional factorial' using a simple routine in the software package SPSS®. Subsequently, we used a cyclic designed to generate 9 choice sets. These choice sets satisfy the principle of orthogonality, level balance, and minimal overlap (see Huber and Zwerina, 1996).

[67] Note that all the indexes for the respondents and alternatives have been omitted.

the effect of a decrement of the groundwater contamination by 40 percent and 55 percent, respectively, on the probability to choose an agricultural policy. Finally, β_6 and β_7 provide the effect of a decrement of human health risk by 150 and 200 cases of acute poisoning, respectively, on the probability to choose an agricultural policy.

We also want to control for differences in the respondent's profile with respect to the consumer's choice and therefore the economic valuation of alternative pesticide programmes. We run all possible attribute combinations in order to test down the model and exclude those interactions that are not statistically significant. For this reason, we present here only the models with the highest explanatory capacity.

Model 2 captures the effect of the population characteristics on the marginal WTP uniform utility function.

Model 2

$$\begin{aligned} V = {} & \beta_1 PRICE + \beta_2 BIODIV2 + \beta_3 BIODIV3 + \beta_4 GRWATER2 + \beta_5 GRWATER3 + \\ & \beta_6 HEALTH2 + \beta_7 HEALTH3 + \beta_8 BIODIV2 \times INCOME + \\ & \beta_9 GRWATER3 \times GENDER + \\ & \beta_{10} HEALTH3 \times AGE \end{aligned} \qquad \textbf{Eq-7-4}$$

In particular, Model 2 incorporates in the utility function the respondents' level of income, gender and age. It involves the cross-terms of BIODIV2 and INCOME, GRWATER3 and GENDER, HEALTH3 and AGE. INCOME is a continuous variable and provides the monthly household income. GENDER is a dummy that takes on value 1 if the respondent is a female, zero otherwise. AGE is a continuous variable providing the respondent's age. From the coefficients of cross-terms we can investigate: whether there are differences in the marginal utility of BIODIV2 given different income levels; whether there is a difference in the marginal utility of GRWATER3 due to different gender, and of HEALTH3 given the respondent's age. We also tried several other interactions resulting in coefficients with the expected sign but these were not statistically significant.

In Model 3 we add the interactions with variables controlling for the respondent's environmental attitude and level of concern about pesticide risks:

Model 3

$$\begin{aligned} V = {} & \beta_1 PRICE + \beta_2 BIODIV2 + \beta_3 BIODIV3 + \beta_4 GRWATER2 + \beta_5 GRWATER3 + \\ & \beta_6 HEALTH2 + \beta_7 HEALTH3 + \beta_8 BIODIV2 \times INCOME + \\ & \beta_9 GRWATER3 \times GENDER + \beta_{10} HEALTH3 \times AGE + \\ & \beta_{11} HEALTH2 \times CONCERN + \beta_{12} GRWATER3 \times ATTITUDE \end{aligned} \qquad \textbf{Eq- 7-5}$$

where CONCERN and ATTITUDE are two categorical variables based on a 5-point Likert-scale, ranging from 0 'not at all' to 5 'extremely' informed on pesticide risk and, respectively, sensitive to environmental and health issues.

7.3.2. Statistics of the questionnaire

The questionnaire was developed by using the results from focus groups and one pre-test[68]. Focus groups and the pre-test were necessary: to test the appropriateness of the attributes (and their levels) included in the questionnaire; to select a proper payment vehicle for the WTP experiment and test bids; and to refine the initial draft questionnaire. On the basis of the results provided by the pilot study, some minor modifications in the draft questionnaire were included. The pre-test was conducted on two university campuses[69] and the final survey was carried out in Milan, Italy, between May and June 2003. The survey questionnaire was self-administered by respondents who were approached at three shopping malls in Milan by a trained team of three interviewers. The enumerators were instructed to stop potential respondents and ask them to take the questionnaire, complete it, and then drop it off after shopping. Overall, 484 questionnaires were distributed, 302 of which were returned in a completed form. The return rate was about 62 percent. Table 7-2 shows the survey statistics and the socio-demographics of the sample.

The socio-demographic features of our sample are to some extent different from those of the population of Milan. The average respondent is 34 years old, has a household income of roughly € 25,000 a year, and has completed high school. The sample is slightly unbalanced toward females, and overrepresents households that are large relative to the Milan average. 15 percent of the sample has at least one person in the household who is younger than 15. The main differences between the socio-demographics of our sample and those of the population of Milan concern age and income level. The average age of our sample is rather low – 34 as opposed to 44 years old – and the household income is 25 percent higher than the Milan average. This suggests that we should control for these individual characteristics in our statistical model of the choice responses. Moreover, 26 percent of the respondents had a strong environmental attitude and 12.2 percent were very concerned about pesticide risks. When compared with other problems in Italy, respondents ranked environment as the third important area for public investment after public health and education. 68.9 percent of the sample population considered public investments for environmental safety very important, compared with the 77.3 and 71.5 percent who found investment for, respectively, public health and education very important. 56.3 percent of the population indicated that they were "informed" or "very well-informed" on pesticide problems, while only 10.2 percent indicated that they were "not at all" or "slightly" informed.

On the basis of the responses to the choice questions and to the control questions, we believe that the respondents had a reasonably good comprehension of the survey material and choice tasks. Only 4.4 percent complained that they had insufficient information, and the relatively low proportion of 8.5 percent reported that they had found some of the questions difficult to understand.

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[68] A pre-test on 40 respondents was undertaken in April 2003 in Milan.

[69] University campuses and shopping centres were considered to be privileged locations to maximize the visibility of our questionnaire and the sampling size, thus curbing the generally high costs of surveys. On the university campuses, interviewers asked people to take the questionnaire, bring it home and ask the member of the family responsible for the daily food shopping to complete it. In shopping centres, people were asked to take the questionnaire before shopping, complete it and then drop it off to the interviewer after shopping.

Table 7-2: Survey statistics and socio-demographics of the sample

Variable	Sample average or percentage	Milan average*
Individual characteristics		
Age	33.9	44
Monthly Household Income in Euros (€/household)	2,098.1	2,791.3
Female	61.6	53.2
Household size	3.5	2.5
Households with one or more persons under 15	15.1	NA
Years of schooling	13.04	NA
Attitudinal characteristics		
Respondents with strong environmental attitude**	26.1	
Respondents very well-informed about pesticide risks**	12.2	
Respondents debriefs		
Found some questions hard to understand	8.5	
Found not enough information provided	4.4	

Note: (*) Authors calculation based on the Milan Municipality Abstract of Statistics, 2002. (**) Based on a 5-point Likert scale.

7.3.3. CE estimation results

The estimation results for Models 1 to 3 are shown in Table 7-3. We first estimated a basic model and, subsequently, we used interactions between the choice attributes and socio-demographic variables to control for individual characteristics.

In Model 1, all variables are highly statistically significant. As expected, the sign of PRICE is negative and that of the level of the various pesticide risks reduction is positive. Significant coefficients of BIODIV2, BIODIV3, GRWATER2, GRWATER3, HEALTH2, and HEALTH3 show that the valuation of reductions in pesticide risks varies according to the relative level of application. Standard economic theory suggests that the WTP should be positively associated with the magnitude of risk reduction. For each type of pesticide risk, the results from Model 1 tell us that an additional level of risk reduction higher than the minimum level proposed is welcomed by respondents, as the coefficients of BIODIV3, BIODIV3, GRWATER2, GRWATER3, HEALTH2, and HEALTH3 are all positive and highly significant. If we calculate the WTP associated with human health, we see that, as expected, the marginal utility of moving from a risk reduction by 100 to 250 cases of acute poisoning is lower than the marginal utility of moving from a risk reduction by 100 to 300 cases of poisoning (see Table 7-3). Similarly, if we calculate the WTP associated with biodiversity risk we see that the marginal utility of moving from a biodiversity risk reduction by 6 to 9 endangered species is lower that the marginal utility of moving from 6 to 12 endangered species. The same applies for the marginal utility associated with different levels of groundwater protection.

To capture variation in the marginal utility of the attributes across individuals, it is necessary to control for the respondents' socio-demographic characteristics. Regarding the preferences for the environmental attributes (biodiversity, soil and groundwater protection, human health), it would be expected that they would vary across respondents' profiles, depending on individual environmental attitude and socio-demographic characteristics. The results of log-likelihood ratio tests show that adding socio-demographic and attitudinal variables adds significantly to Model 1. In Model 2, the effects of interaction of the choice attributes with INCOME, GENDER and AGE are highly statistically significant, meaning that individual utility is sensitive to the individual level of income, gender and age. The interaction with INCOME and BIODIV3 is positive and significant. This suggests the existence of a positive relationship between the choice of a given agricultural scenario and income, as expected. The coefficient of the interaction between GENDER and GRWATER3 is significant and negative, meaning that females are less willing to pay to reduce pesticide risks. On the other hand, the interaction between AGE and HEALTH3 is, as expected, positive.

Finally, in Model 3, the interaction of the choice attributes with CONCERN and ATTITUDE is positive in both cases, as expected, but only the coefficient of ATTITUDE is statistically significant.

Table 7-3: Estimated coefficients of MNL models

	Model 1	Model 2	Model 3
PRICE	-0.005*** (0.595^{-03})	-0.010*** (0.001)	-0.010*** (0.001)
BIODIV2	0.470*** (0.095)	0.658*** (0.231)	0.654*** (0.235)
BIODIV3	0.911*** (0.088)	0.964*** (0.105)	0.965*** (0.107)
GRWATER2	0.201*** (0.089)	1.302*** (0.094)	1.309*** (0.096)
GRWATER3	0.762*** (0.089)	1.503*** (0.163)	0.804* (0.443)
HEALTH2	0.283*** (0.108)	1.148*** (0.102)	0.842** (0.388)
HEALTH3	0.745*** (0.091)	0.927*** (0.253)	0.977*** (0.256)
BIODIV2 × INCOME		0.230^{-03}*** (0.960^{-04})	0.233^{-03}*** (0.972^{-04})
GRWATER3 × GENDER		-0.328* (0.181)	-0.381** (0.185)
HEALTH3 × AGE		0.011* (0.006)	0.010* (0.007)
HEALTH2 × CONCERN			0.092 (0.103)
GRWATER4 × ATTITUDE			0.187* (0.108)
SAMPLE	1358	1345	1322
Log-likelihood	-963.526	-951.569	-927.779
Pseudo-R^2	0.354	0.356	0.361
LR test of significance of all coefficients		21.11 ($p < 0.001$)	25.74 ($p < 0.001$)

Table 7-4: Willingness-to-pay estimates (Model 1)

	WTP	*Upper-bound*	*Lower-bound*
BIODIV2	98	153	56
BIODIV3	190	265	140
GRWATER2	42	84	5
GRWATER3	159	226	112
HEALTH2	59	112	15
HEALTH3	155	222	109

Note: Willingness-to-pay is expressed in euros per household per month. Upper and lower bounds are calculated using the delta method.

Table 7-5: Unit trade-offs across choice attributes

	Human health	*Soil and groundwater*	*Bird biodiversity*
Human health	1	0.4	0.15
Soil and groundwater	2.3	1	0.36
Bird biodiversity	6.4	2.7	1

Note: Trade-offs are calculated on the basis of a basic MNL model including the choice attributes PRICE, BIODIV, GRWATER, HEALTH. Attributes are categorical variables that can assume the values reported in Table 7-1.

7.4. Using contingent valuation to estimate benefits from pesticides elimination

One of the purposes of our study was to estimate the overall benefits of eliminating all negative environmental effects related to the use of pesticides in Italian farmland. With this in mind, we added a contingent valuation (CV) question at the bottom of the sequence of choice sets. In CV surveys, one of the most widely-used approaches to elicit information about the respondents' WTP is the 'dichotomous-choice' format. Following Hanemann et al. (1991), a follow-up valuation question was included so as to improve the statistical efficiency of the WTP estimates. We use this type of elicitation question to estimate the respondents' WTP for eliminating all risks, to both human health and the environment, associated with pesticide applications in agriculture. The dichotomous-choice format mimics behaviour in regular markets, where people usually buy, or decline to buy, a certain good at the proposed retail price. Furthermore, similar to the CCE technique, this CV format is consistent with the incentive-comparability property and is also credited with reducing the cognitive burden placed on the respondents, except that its incentive comparability property might be affected by the previous conjoint questions.

The dichotomous-choice "double-bounded" payment question asked the respondents if they would be willing to pay B_1 percent extra on household monthly food expense to gain the proposed improvement in agricultural safety. In a follow-up question, respondents who answered "yes" to the first bid value were asked if they would pay B_{2+} percent extra on household monthly food expenditure, with $B_{2+} > B_1$, while respondents who answered "no" were faced with a B_{2-} amount, with $B_{2-} < B_1$. The bid value B1 varied randomly across respondents and the amount of the second bid B_2 depended on the amount of the first one, as given in Table 7-5.

According to the double-bounded response model, four response sequences are possible for respondent j: both answers are positive "yes-yes"; both answers are negative "no-no"; j refuses the first bid but accepts the second "no-yes"; or j accepts the first but refuses the second "yes-no". We code these as P^{yy}, P^{nn}, P^{ny}, P^{yn}, respectively.

Table 7-6: Bid cards and distribution of WTP responses

	Amount(*)			Distribution of the WTP responses [%]			
Bid card	Initial	High	Low	yes-yes	yes-no	no-yes	no-no
1	10	20	5	46.9	48.0	2.0	3.1
2	15	30	10	34.3	59.0	6.7	0.0
3	20	40	10	29.3	55.6	13.1	2.0

Note: (*) Expressed as the percentage increase in the household expenditure on food.

Consistent with what we might expect from economic theory, the frequency of "yes-yes" responses falls with the amount the respondent is asked to pay (see Table 7-6). 29.3 percent of respondents say they would be willing to support a more than 40 percent increase in their expenditure on food. This is a pretty high percentage and deserves to be better investigated. In addition to this, the bid amount is not clearly correlated to the proportion of "no-no" responses. Nevertheless, only five respondents state "no" to both the dichotomous WTP questions, corresponding to 1.7 percent of the sample. The remaining response answering patterns, "yes-no" and "no-yes" responses, indicate that the respondents' maximum WTP lies between the initial bid amount and the higher (lower) bid amounts. Considering the four possible response patterns, the sum of contributions to the likelihood function $L(\theta)$ over the sample is maximized:

$$\log L = \sum_{i=1}^{n} \left[I_{yy} \log P_i^{yy} + I_{yn} \log P_i^{yn} + I_{ny} \log P_i^{ny} + I_{nn} \log P_i^{nn} \right], \qquad \textbf{Eq- 7-6}$$

Since the follow-up bid amount is greater (lower) than the first for those who answer "yes" ("no") to the initial payment question, the four pairs above identify the intervals in which the respondents' WTP amount is assumed to fall. Specifically, the respondent's WTP is greater than B_2 for "yes-yes" sequences; the WTP falls between B_2 and B_1 for "no-yes" pairs; it falls between B_1 and B_2 for "yes-no"; and it is lower than B_2 for "no-no". This yields the following log-likelihood function:

$$\log L = \sum_{i=1}^{n} \log\left[F(WTP^H; \theta) - F(WTP^L; \theta) \right], \qquad \textbf{Eq- 7-7}$$

where WTP^H and WTP^L are, respectively, the higher and the lower bound of the interval around WTP. For a univariate model with a Weibull distribution, the mean and median WTP estimate is equal to a percentage increase in household food expenditure of 19.8 percent and 15 percent, respectively (see Table 7-7). The density functions of the WTP with a Weibull distribution are plotted in Figure 7-1.

What we estimate is an "overall" WTP value for reducing all negative side effects of pesticides, compared with a "target specific" WTP inferred using a CCE approach.

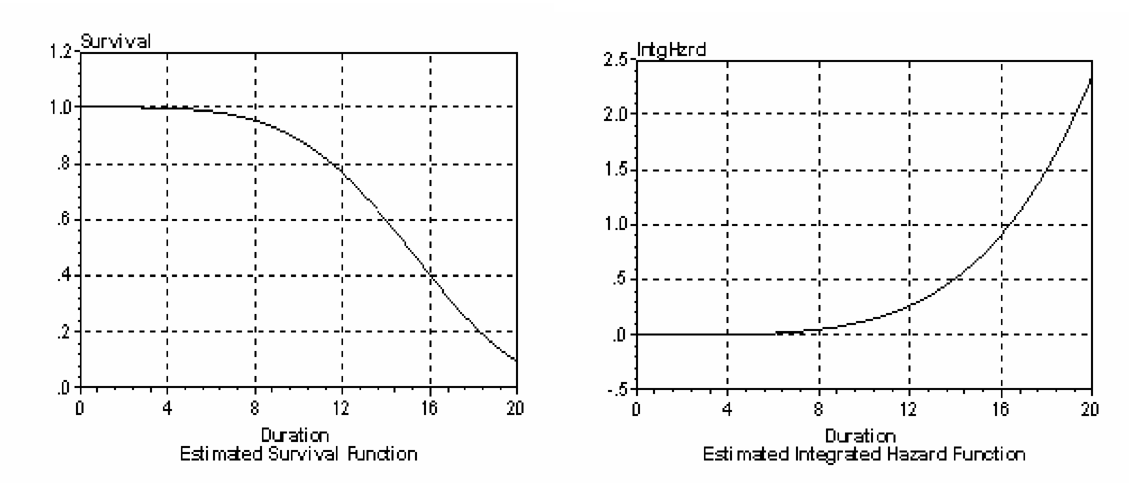

Figure 7-1. Density and hazard function of WTP inferred from the CV question

Table 7-7: Contingent Valuation WTP estimate

WTP	
Mean	19.797
Median	15.009

Note: The WTP is measured as a percentage of the increase in the household food expenditure.

7.5. Welfare analysis and policy discussion

An important prediction of economic theory is that WTP is an increasing function in the individuals' income level. To capture the variation in preferences related to the respondents' income level, we try an interaction of the BIODIV3 variable with the respondents' income level variable. The results from a formal meta-analysis on pesticide risk valuation by Florax et al. (2005) suggest that the income elasticity of the WTP for reduced risk exposure varies across specifications, but seems to indicate that the income elasticity is positive and the relationship is elastic, though low in absolute value. Our results are consistent with those of Florax et al. (2005) and show a positive and highly statistically significant relationship between respondents' income level and WTP estimates.

As we stated before, standard economic theory suggests that the WTP should be positively associated with the level of risk reduction provided. The results from Model 1, which is intended to take into account this effect, are mixed. The results tell us that, for any pesticide risk type, an additional level of risk reduction higher than the lowest one (BIODIV1, GRWATER1, HEALTH1) is welcomed by respondents, as all coefficients (BIODIV2, BIODIV3, GRWATER2, GRWATER3, HEALTH2, HEALTH3) are positive and statistically significant. Moreover, as expected, the marginal utility of moving from a human health risk reduction by 100 to 250 cases of acute poisoning (about € 59 per household per month) is lower than the marginal utility of moving from a risk reduction by 100 to 300 cases of poisoning (approximately €155 per household per month). Similarly, the marginal utility of moving from a biodiversity risk reduction by 6 to 9 endangered species (about €98) is lower that the marginal utility of moving from 6 to 12 endangered species (about €190). The same applies for the marginal utility associated with different level of groundwater protection (respectively €42 and €159 per household per month for a groundwater contamination reduction of 40 percent and 55 percent, respectively).

The survey of Milan's respondents shows that they are, on average, willing to pay €6 per household per month to avoid the loss of one species of farmland bird biodiversity, €2.3 per household per month to avoid the contamination of 1 percent of farmland soil and groundwater, and €0.8 per household per month to prevent one case per year of human ill-health[70]. Though one might be surprised to observe that biodiversity and groundwater received a higher value compared with human health, a comparison of unit trade-offs[71] reveals that Milan's respondents do strongly perceive possible risks for human health related to pesticide use, while there is much less concern about the concept of biodiversity (see Table 7-5). What we see is that, on average, respondents are willing to tolerate only six additional cases of human illness to save one entire species of farmland birds, and two cases of human illness to reduce soil and ground water contamination by 1 percent. Similarly, Foster and Mourato (2000) find that respondents were only willing to tolerate six to eight additional cases of human illness to save an entire farmland bird species. However for bird biodiversity and human illness, our estimates are higher than those by Foster and Mourato (2000). They calculate a WTP of about €20 per household per year to save one farmland bird species, and a WTP of about €3 per household per year to avoid one case of human illness, whereas our average estimates are approximately €76 and €9 per household per year, respectively. We posit that differences might derive both from differences in modelling and elicitation features. In fact, Foster and Mourato employ contingent ranking and use a price premium on a single food product, a loaf of bread, whereas we use a choice experiment and employ a "green shopping" payment vehicle.

The coefficients for GENDER, AGE, CONCERN and ATTITUDE in Model 2 and Model 3 suggest that the individual pattern of noise perception is likely to influence the WTP for pesticide risk abatement in a predictable way. This is a relevant result that confirms the importance of knowing as accurately as possible the respondents' profile according to socio-demographic and environmental attitude and concern about pesticide risk, and of improving the methods for gathering such information.

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[70] WTPs are estimated from a basic MNL model including the choice attributes (PRICE, BIODIV, GRWATER, HEALTH) as categorical variables which assume the values reported in Table 7-1.

[71] It is not correct to make direct comparisons among different pesticide risks and the related WTPs, since the unit of measurement used to quantify different risks in the experiment varies. A more rigorous way of making direct comparisons is to observe unit trade-offs across choice attributes.

7.6. Conclusions

This chapter has provided an economic assessment of the non-market benefits of safety improvements in the environmental and health safety of agricultural production, which can be achieved by non-conventional agricultural practices free from pesticide use. The valuation is based on a questionnaire survey undertaken at Milan, one of the biggest metropolitan areas in the North of Italy. The valuation exercise employs the Choice Experiment (CE) technique, which is an innovation in the pesticide risk valuation literature (for a discussion, see Travisi et al., 2006c). The biggest advantage of CE with respect to Contingent Valuation (CV) is that respondents are forced to make trade-offs – not only between environmental issues and money – but also among different aspects of environmental safety. These are important and typical features of environmental decision making and are central to the debate on the most preferred type of pesticide policy in Italy and Europe.

We use a "green" food expenditure payment package to elicit the respondents' preferences for alternative agri-environmental scenarios, by proposing to them a series of four choice sets made up of three possible agricultural practice options, including the status quo. To our knowledge, this is the first time that such a payment vehicle has been used for the valuation of pesticide damage. The pros of such a payment vehicle are twofold: first, respondents have a clearer view of the cost to them of the proposed policies; second, the estimation of the total benefits from pesticide risk reduction is very much simplified. One possible drawback of this approach is that it may result in an overestimation of respondents' WTP due to initial bid bias; we hope future research will further explore this possibility, which is not addressed here.

From a statistical point of view, the results of the choice modelling experiment perform well in terms of theoretical validity. The signs of major estimated coefficients are statistically significant and consistent with the theoretical predictions, including that respondents evaluate price increase negatively, but evaluate risk reduction positively. Marginal utilities of risk reduction, for any type of pesticide risk, increase as the level of provision increases. Our a priori expectation of the effect of differences in the respondents' socio-economic profiles on attribute coefficients is confirmed by the statistical analysis, with the exception of the effect of gender, which is negative, though results in the valuation literature are also mixed. WTP estimates appear to be positively correlated to income level and concern about pesticides.

Our MNL models of the choice responses indicate that the choice between agricultural scenarios depends in predictable ways on the attributes. For example, respondents consider food purchased in shops to be less attractive if the groundwater pollution generated from the food production process is increased. In addition, respondents are against buying cheaper food that, on the other hand, has more adverse effects on biodiversity and human health. A first result is, therefore, that respondents are capable of assessing agricultural scenarios defined by multiple attributes. Second, respondents assess the agricultural scenarios described in terms of environmental and monetary attributes in the way we expected, showing a positive willingness-to-pay for a gain in agricultural environmental safety.

We also examine the effects of the respondents' attitudinal and socio-demographic characteristics on their preferences, via interactions between choice attributes and explanatory variables, with a special focus on: income level, gender, age, pesticide risk concern, and attitude. Our a priori series of expectations is satisfied, with the exception of the interactions between GRWATER3 and GENDER, which show a negative coefficient (see also Hammitt, 1990). While previous studies on individual preferences for pesticide-related issues (Govindasamy et al., 1998a, 1998b; Foster and Mourato, 2000) show that women usually exhibit a more altruistic attitude than men, our results seem to indicate that women are less willing than men to pay for a reduction of pesticide risks. We tentatively suggest that this might be because Italian women are more able than men to appreciate the impact of the increased expenditure on the household budget (and the welfare effects). Using a 5-point Likert scale, respondents were also asked to declare their level of concern about the use of pesticides and their level of environmental attitude. The interaction with CONCERN shows a positive coefficient. This means that the higher the respondents' concern about pesticide risks, the higher their WTP for risk reductions. Importantly, and consistent with what is predicted by economic theory, the interaction between BIODIV3 and INCOME shows a positive and statistically significant coefficient, even though the elasticity is rather low.

Finally, and to conclude, another result of our study is the estimation of the value of eliminating all risks from pesticide use in agriculture. According to the contingent valuation estimates, the annual mean WTP amounts to an increase of 19.8 percent in household food expenditure. We are aware of a previous study by Higley and Wintersteen (1992) that estimates WTP for eliminating all environmental risks associated with pesticide use, including effects leading to the poisoning of humans, via a CVM survey in which they asked respondents about their WTP to eliminate three different risk levels (low, moderate and high). However, a direct comparison is not straightforward, as they interviewed farmers and estimated WTP as $US per person per acre treated with pesticides. We therefore, compare our results with real market prices for organic food in Italy and see that they appear to be consistent with actual market behaviour. For Italy, in fact, the actual price differential between food grown by conventional or biological agriculture ranges between 10 and 200 percent, with a mean price premium set at about 20 percent. Therefore, the CV estimate of the WTP for reducing all detrimental effects of pesticides on aggregate natural well-being performs very well in terms of criterion validity. In any case, subsequent surveys on this topic should investigate whether combining CE and CV questions might affect the incentive comparability property and, more in general, what the effect of survey design variations might be.

8. A META-ANALYSIS OF THE WILLINGNESS-TO-PAY FOR REDUCING PESTICIDE RISK TO ECOSYSTEMS AND HUMANS*

This chapter takes advantage of the on-field experience gathered during the development of the previous chapters, as well as starting from what is already available in the empirical economic literature, in order to provide the first formal meta-analysis on the monetary value of reducing pesticide risk exposure.

A statistical summary of WTP estimates for reduced pesticide risk exposure taken from the empirical economic literature is presented. This is based on the taxonomy of pesticide risk valuation literature proposed in Chapter 6. Meta-analysis is then employed as a statistical tool to analyse the variation in the estimated WTPs associated with the impacts of pesticide risk on human health and the environment. Meta-analysis is a form of research synthesis in which previously documented empirical results are combined or re-analysed in order to increase the power of statistical hypothesis testing. Some proponents maintain that meta-analysis can be viewed as a quantitative literature review. Others assert that meta-analysis can be used to pinpoint aspects critical to the future development of theory (Stanley, 2001).

This chapter is organised as follows. Section 8.1 presents an exploratory assessment of empirical WTP values for different pesticide risk impacts. Section 8.2 gives an overview of potential determinants for differences in WTP values, where the differences are related to theory, behavioural aspects and/or the research design of the underlying studies. In Section 8.3, we analyse the empirical WTP estimates by means of a meta-regression in order to account for potential differences in a multivariate framework. Section 8.4 provides conclusions.

8.1. Exploratory meta-analysis

Meta-analysis is essentially the 'analysis of analyses' (Hunter and Schmidt, 1990) and has a long tradition in experimental medicine, biomedicine and experimental behavioural sciences, specifically in education and psychology. Its use in the experimental sciences has produced a growing literature on appropriate statistical techniques (for a review, see Cooper and Hedges, 1994), geared towards the combination of effect sizes across studies in order to increase statistical power of hypothesis testing. Effect sizes are statistical summary indicators such as standardised differences in means of experimental and control groups, correlations, and odds-ratios.

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* *Based on Florax, Travisi and Nijkamp (2005).*

These types of effect sizes are rather different from the typical quantitative measures used in economic research. Although substantial parts of economics are quasi-experimental rather than experimental, and meta-analysis was initially developed for experimental disciplines, economists are increasingly starting to use meta-analysis in quasi- or non-experimental contexts (Stanley, 2001). Meta-analysis constitutes a systematic framework for the synthesis and comparison of previous studies, because it systematically exploits existing empirical results to produce more general results by focusing on a joint kernel of previously undertaken research (Florax et al., 2002). The use of meta-analysis in economics originated in environmental economics, and was to a considerable extent driven by the need to attain clarity about WTP estimates for non-marketed environmental goods, and the associated differences in valuation techniques (see Smith and Pattanayak, 2002). By now, there is a considerable meta-analysis literature in environmental economics, and the technique proliferates to other areas, such as labour economics, industrial organisation, and macroeconomics (Florax, 2002a).

Apart from Nijkamp and Pepping (1998), who focus on the effectiveness of pesticide price policies, no meta-analysis on pesticide usage exists.[72] Most meta-analyses in economics employ meta-regression.[73] In our case, the meta-regression analysis centres on identifying the relationship between the WTP for a decline in pesticide threats, and theoretical and behavioural differences towards pesticide risk, as well as differences in the research design of the underlying studies. Typical moderator variables therefore include the baseline risk level, risk attitudes and perceptions of respondents, the source and nature of the risk data, and research design characteristics.

Meta-analysis can, however, also be used to combine effect sizes. We therefore first focus on deriving a combined WTP estimate for the different types of risks distinguished in Chapter 6 (see Figure 6-2), and we assess whether the WTP estimates can be viewed as a homogeneous or heterogeneous sample by means of meta-regression analysis. In the remainder of this section, we discuss the literature retrieval process, and then explore the meta-data set. Subsequent sections discuss the prime determinants of WTP values for reduced pesticide risk exposure, and provide the results of the meta-regression analysis.

The literature retrieval process comprises checking several economic databases (among others EconLit), reference chasing, and approaching key scholars in the field. Several keywords, such as ‘willingness-to-pay’, ‘pesticide’, ‘food-safety’, ‘environmental risk’, and ‘human health risk’ were used in order to cover the multidimensionality of pesticide risks. This resulted in a set of slightly more than 60 studies, of which a subset of 27 contains monetary estimates. Several of these studies do, however, not provide usable WTP estimates. Specifically, in some studies the estimates are expressed as a probability of WTP (see, e.g., Owens et al., 1997; Thompson and Kidwell, 1998; Huang, 1993). Others use the cost of illness approach (see Crissman et al., 1994; Pingali et al., 1994), or they use a hedonic approach to estimate shadow values and only report the mean elasticity for various impacts of herbicides (see Beach and Carlson, 1993; Söderqvist, 1998). As a result, the meta-analysis is concerned with only 15 studies, from which we derive 331 observations.

[72] See also van den Bergh et al. (1997b) for more extensive results.

[73] See Florax (2002a) for an overview of methodological problems in meta-regression analysis.

A listing of the studies and their main characteristics is presented in Table 8-1. The studies have been published during the 1990s and early 2000s, and predominantly deal with the US. Most observations (> 230) refer to human health, of which approximately one-fifth is concerned with farmers and the rest with consumers, in particular with the unspecified general health hazard. Approximately one-third of all observations refer to detrimental effects on ecosystems, with slightly more observations pertaining to aquatic as compared with terrestrial ecosystems.

Table 8-1 shows that comparing effect sizes for different target types, countries and time-periods comes with operational problems, because the effect sizes have to be transformed to a common measurement unit, and a common currency in prices of a given year. The latter two transformations are straightforward, but the transformation to a common measurement unit necessitates the use of approximations. The standardised effect size T is derived from the original effect size reported in the primary study as $T = c \cdot t \cdot m_i \cdot \tilde{T}_i$, where $\tilde{T}_i$ is the original effect size in a specific measurement unit and a given currency of a specific year, and T is the marginal WTP per person, per year, for a given reduction in pesticide risk exposure, in US dollars of 2000. The transformation factors m_i depend on the measurement unit of the underlying studies. In order to standardise the data, information about average household size, annual per capita consumption of produce, annual number of pesticide treatments, and rural density are taken from the original studies or from official national statistics. The transformation factors t and c are operationalised as a GDP deflator, and a Purchasing Power Parity (see the Appendix for details). From here on, all WTP figures are presented as standardised effect sizes using the above definition.

Table 8-1 Overview of studies providing empirical WTP estimates for pesticide risk reductions[a]

					Environmental degradation							Human health						
Study	Data	Country	Measurement unit:	# Meta-	Aquatic			Terrestrial				Farmers			Consumers			
			value per	obs.	A1	A2	A3	A4	A5	A6	A7	B1	B2	B3	B4	B5	B6	B7
Baker and Crosbie (1993)	1992	US	person, produce unit	12	—	—	—	—	—	—	—	—	—	—	—	—	—	12
Buzby et al. (1995)	1995	US	person, produce unit	3	—	—	—	—	—	—	—	—	—	—	—	—	3	—
Cuyno et al. (2001)	1999	Philippines	household, crop season	10	2	—	—	2	2	—	2	—	—	2	—	—	—	—
Eom (1994)	1990	US	person, produce unit	12	—	—	—	—	—	—	—	—	—	—	—	—	12	—
Foster and Mourato (2000)	1996	UK	person, produce unit	26	—	—	—	—	—	13	—	—	—	13	—	—	—	—
Fu et al. (1999)	1995	Taiwan	person, produce unit	3	—	—	—	—	—	—	—	—	—	—	—	—	3	—
Hammitt (1993)	1985	US	person, produce unit	115	—	—	—	—	—	—	—	—	—	—	23	23	—	69
Higley and Wintersteen (1992)	1990	US	person, acre application	48[b]	6	6	6	6	6	—	6	6	6	—	—	—	—	—
Lohr et al. (1999)	1990	US	person, acre application	32[b]	4	4	4	4	4	—	4	4	4	—	—	—	—	—
Misra et al. (1991)	1989	US	person, produce unit	1	—	—	—	—	—	—	—	—	—	—	—	—	—	1
Mullen et al. (1997)	1993	US	household, month	24	3	3	3	3	3	—	3	3	3	—	—	—	—	—
Ravenswaay and Hoehn (1991a)	1990	US	person, year	6	—	—	—	—	—	—	—	—	—	—	—	—	—	6
Ravenswaay and Hoehn (1991b)	1989	US	person, year	18	—	—	—	—	—	—	—	—	—	—	—	—	18	—
Roosen et al. (1998)	1998	US	person, produce unit	16	—	—	—	—	—	—	—	—	—	—	—	—	—	16
Wilson (2002)	1996	Sri Lanka	person, year	5	—	—	—	—	—	—	—	—	—	5	—	—	—	—
					15	13	13	15	15	13	15	13	13	20	23	23	36	104
			Total	331	41			58				46			186			

Note:

[a] See Figure 8-1 for an explanation of column headings (A1, A2, etc.) referring to the different target types.

[b] Six observations in Higley and Wintersteen (1992), and four in Lohr et al. (1999) are excluded from the meta-sample because they refer to more than one target type simultaneously. The 32 observations from Lohr et al. (1999) are computed using additional information provided in Higley and Wintersteen (1992, 1997), starting from the four observations referring to environmental and human health risks simultaneously.

The top graph in Figure 8-1 shows that the number of WTP estimates drawn from the studies varies between 1 and 115. Within studies, the distribution of estimates is as a rule rather even, except for the study by Hammitt (1993), which has a very skewed distribution (the median is substantially smaller than the mean). This also carries over to the overall distribution of estimated WTP values for all studies. The mean WTP for reduced pesticide risk exposure is US$ 122 per person per year (in prices of the year 2000), and the median is US$ 16, but the overall standard deviation is rather high at US$ 208. The mean WTP value may not necessarily be a meaningful indicator because it assumes that no significant differences in means exist across different target types. In addition, it ignores the conceptual difference in targets and endpoints as described in the taxonomy of pesticide risks (see Figure 6-2).

We therefore present the range of estimates for human health and environmental risks, categorised according to the taxonomy in target types, in Figure 8-1. Clearly, the distributions for the different target types are sometimes rather skewed. However, the most striking result is that the mean WTP for impacts on aquatic and terrestrial ecosystems, and for health effects of farmers, seem to be very similar, except for the valuation of increased biodiversity through reduced pesticide risk exposure. However, the mean WTPs for the impact of reduced pesticide risk exposure on consumer health are substantially smaller, but at the same time, these distributions are very skewed.

In sum, the exploratory analysis indicates that the WTPs for pesticide risk reduction are rather homogeneous. The mean WTP for a reduction in pesticide risk exposure is very similar for health effects for farmers (US$ 262), and the impact on aquatic (US$ 289) and terrestrial ecosystems (US$ 246) excluding biodiversity (US$ 14). The latter seems to constitute a separate category. Likewise, the mean WTP for a reduction in negative health effects for consumers (US$ 42) is very different. One should note, however, that it is not necessarily meaningful to compare mean WTPs per target type, because such a comparison ignores differences in, for instance, research design, the initial risk level, the change in the risk level, and income differentials. Moreover, the WTP values vary greatly about the mean, and they have been measured with varying precision.

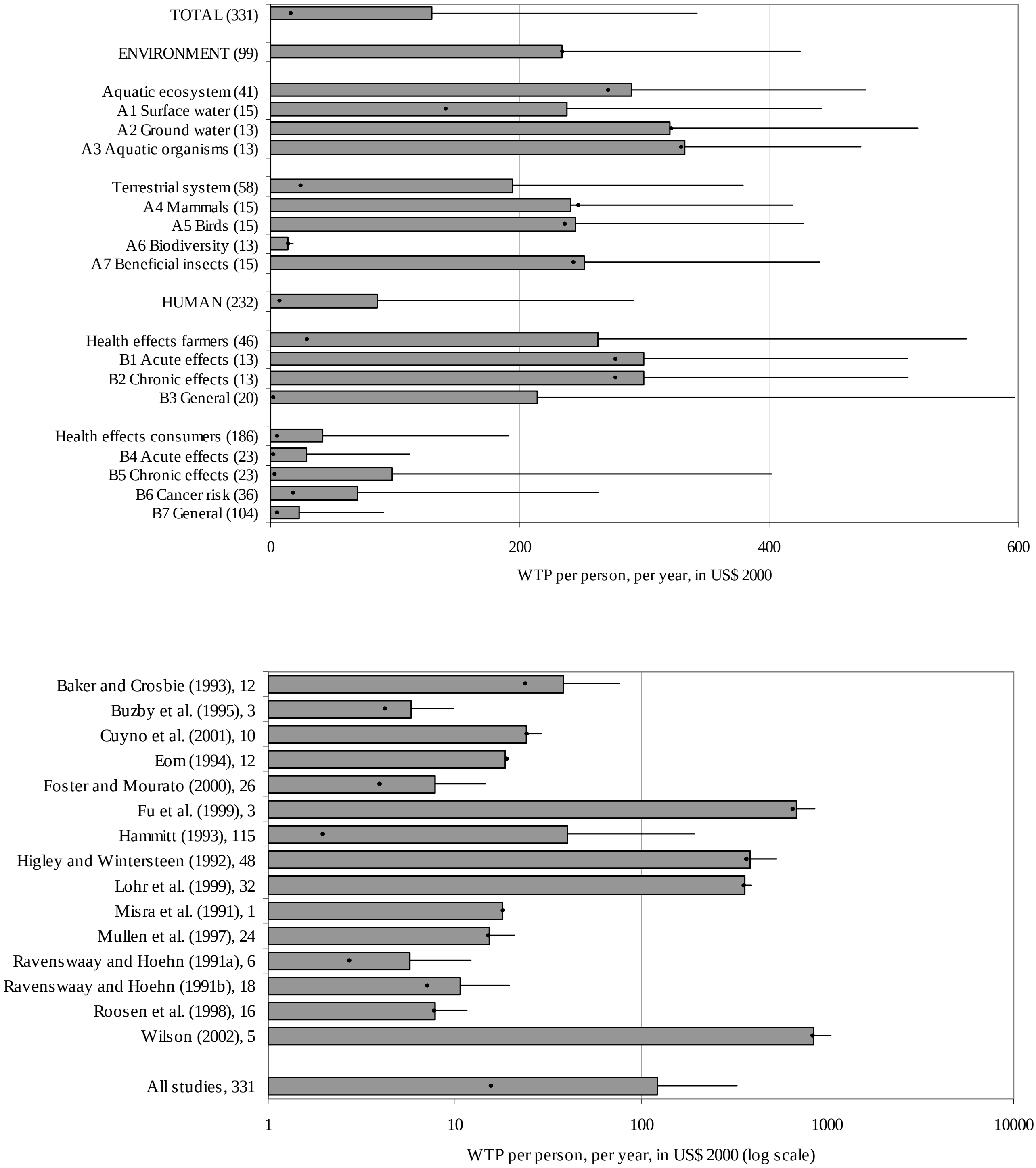

Figure 8-1: Willingness-to-pay per person, per year, in US$ referring to 2000, organised by study or by target

Note: Bars represent the average value, the median value is indicated by black dots, and the error bars represent the standard deviation of the WTP values within each study or target type

8.2. Potential determinants of WTP variation

The meta-analysis therefore focuses on explaining differences in WTP estimates by means of a multivariate meta-regression whose dependent variable is the standardised WTP measure, and whose explanatory variables are related to theoretically expected differences, methodological issues, and differences in the study setting.

The standardised WTP estimate for the reduction and prevention of pesticide risk exposure ranges from –26 to 1,375 US$ per person, per year.[74] In total, there are 331 observations, of which 15 (taken from Hammitt, 1993) are negative. Because the negative values are theoretically implausible and the heteroscedasticity inherent in a meta-analysis is generally mitigated by a semilog specification for which the dependent variable has to be strictly positive, we exclude the negative values. The meta-analysis is therefore based on 316 positive observations, with a mean and median of US$ 136 and 17, respectively.

Potentially relevant explanatory factors, usually called moderator variables (Sutton et al., 2000), can be derived from three different sources. Theoretical models of individual rationality suggest WTP-risk trade-offs, and factors related to the study design process pertaining either to methodological issues or to the specific study setting (time period considered, geographical location, etc.) may induce systematic variation. We briefly discuss the relevant variables and operationalisations.

The main distinction among target types in our taxonomy separates human health deterioration and degradation of the environment. This distinction can also be interpreted as distinguishing between private and public effects of reduced pesticide risk exposure. Microeconomic choice theory underlying WTP estimation predicts the WTP for private goods to be relatively higher, because of free-riding behaviour inherent in collective welfare improvements (Johannesson et al., 1996). In the empirical analysis, we use dummy variables to assess and control for heterogeneity according to target types.

A simple expected utility framework can be used to describe how individuals are willing to trade wealth for increases or decreases of health risks, under the conventional assumption that the estimated marginal valuation of a risk decline increases with an increase in the baseline risk level, with the absolute size of the risk reduction, and with the baseline income (Grossman, 1972; Jones-Lee, 1976; Hammitt, 2000). Previous meta-analyses of health hazard valuations have found significant and positive correlations between the risk level and WTP, and a negative correlation with risk decline (Miller, 2000; Mrozek and Taylor, 2002; de Blaeij et al., 2003). In our meta-analysis, we use the following operational definitions. First, in order to make the studies comparable, the information on the baseline risk has to be expressed in a discrete three-step variable (ultimately transformed into three different dummy variables) identifying a low, medium and high baseline risk. Second, in virtually all studies the risk reduction equals the change from the baseline risk level to zero, and it can hence not be identified

[74] A fairly small number of primary studies reports trimmed rather than ordinary mean WTP-values (i.e. the mean of a middle group of a series of individual estimates), because trimmed means are less sensitive to outliers, and trimming reduces the distance between the mean and the median of the distribution of individual WTP values (see also de Blaeij et al., 2003).

separately[75]. Finally, because a complete data series on the baseline income level for all the original studies is lacking, we include this determinant in the analysis using exogenous information on GDP per capita levels for countries (World Bank, 2002).

An important methodological difference between the studies concerns the valuation technique. Contingent valuation (CV) and revealed preference (RP) methods each account for about 40 per cent of the observations, and approximately 20 per cent use some kind of choice experiments (CEs) (either conjoint analysis, contingent ranking, or choice modelling). Stated Preference (SP) studies are generally expected to exhibit higher WTP estimates than RP studies (see, e.g., List and Gallet, 2001; Little and Berrens, 2004; Murphy et al., 2005).

Another potential source of variation is the subjective nature of the WTP estimates and the related issue of the individual's perception of risk. The sociological and psychological risk perception literature shows that individuals have difficulty dealing with uncertain events with a low probability of occurrence. Individuals also find it hard to perceive actual risks accurately on the basis of expert information or news coverage (Viscusi and O'Connor, 1984; Slovic, 1987). The individual's perception of risk is therefore influenced by the nature and quality of the available risk information, and the degree to which subjective perception problems occur. In the meta-analysis we can assess the importance of some of these perception difficulties, although only for stated preference studies. We experiment by including dummy variables controlling for the type of risk information provided to respondents in the valuation surveys. Specifically, as suggested in Chapter 6, we can control for differences in the type of risk scenario presented to the respondents (i.e. an actual or potential scenario), and the health risk vehicle in consumer health studies (one specific type of fresh food, or fresh food in general). In addition, we can include information regarding the type of payment vehicle (price premium, separate billing, or yield loss), which type of interview was performed (mail versus face-to-face), and whether pre-tests and controls for biases were adopted. Finally, with respect to all types of studies we can distinguish *ex ante* from *ex post* risk and general risk, differentiate according to the source of pesticide risk (one specific pesticide or pesticides in general), and the type of safety-enhancing measure proposed (adoption of Integrated Pest Management versus eco-certification of food commodities or a ban on particular pesticide compounds).

It is well known that respondents' socio-demographic characteristics also affect their risk perception and WTP (Huang, 1993; Govindasamy et al., 1998b; Sjoberg, 2000). Complete socio-demographic profiles cannot, however, be derived from the information available in the primary studies. We therefore experimented by including dummy variables indicating which stakeholders were interviewed in the valuation survey (farmers, consumers, or both), and a dummy variable referring to the geographical location of the study (US versus non-US).

[75] The only studies for which precise continuous information on the baseline risk and the risk decline is available are the studies on the relationship between pesticide exposure and cancer (Buzby et al., 1995; Eom, 1994; Fu et al., 1999). A detailed explanation of the operationalisation of the baseline risk level is given in the Appendix to this Chapter.

8.3. Meta-regression variants and estimation results

Our meta-regression analysis began by assessing the heterogeneity of effect sizes, controlling for differences in the risk level and the hypothesised risk change, as well as in per capita income levels across studies. Two modelling aspects are pivotal. First, a meta-analysis is intrinsically heteroscedastic because the effect sizes come from studies with differing numbers of observations. As a result the estimated standard errors of the effect sizes are different. Unfortunately, estimated standard errors were available only for a small part of the data set (89 observations). We therefore used the number of observations in the underlying studies as a proxy to account for the precision with which the effect sizes have been estimated (see also Dalhuisen et al., 2003). The sample sizes of the primary studies range from 21 to 1157 observations[76].

Second, a choice has to be made with respect to modelling the potential differences in the underlying population effect sizes. These differences can either be considered fixed, random or a combination of both (i.e. typically referred to as 'mixed'). In the meta-analysis literature there has been an extensive discussion of whether fixed or random effects models are more appropriate (see Sutton et al., 2000). The meaning of the terms 'random' and 'fixed' in the methodological meta-analysis literature is, however, slightly different from the standard econometric terminology used in the context of panel data (see also Florax, 2002b). Experimental meta-analyses in medicine and psychology usually do not have a panel-like data set-up resulting from sampling multiple estimates from primary studies. Typically, experimental meta-analyses are based on single sampling: each effect size in a meta-analysis sample comes from a different study. In a meta-regression framework the adjectives 'fixed' and 'random' then simply refer to whether the underlying population effect sizes are considered fixed or randomly drawn from a pre-specified distribution. In addition, it is common in the standard experimental meta-analysis literature to weight the estimated effect sizes with their associated standard errors[77].

We started with a simple meta-regression specification in order to determine whether fixed or random effects are more appropriate. Since the meta-sample contains multiple estimates from the same study, we used standard panel data estimators rather than the above-mentioned meta-estimators. Specifically, we performed a meta-regression in which we assumed that the heterogeneity in population effect sizes can be modelled using random effects. From a multitude of specifications with random effects for different characteristics (see Rosenberger and Loomis, 2000), we chose three obvious candidates. In one specification, we assumed unobserved heterogeneity between studies, and, in the others, between target types and between different estimation methods used in the underlying

[76] It is common in meta-analysis to use the reciprocal of the sampling variance as weights so that the estimated effect sizes that have been measured with the greatest precision are given most weight (see, e.g., Sutton et al., 2000). As the variance tends to be inversely related to the number of observations, we use the number of observations of the original studies as weights.

[77] In standard econometric terms, the fixed effects meta-estimator is therefore equivalent to the weighted least squares (WLS) estimator using the estimated variances (obtained in the primary studies) as weights and re-scaling the standard errors of the meta-regression by means of the square root of the residual variance (see Hedges, 1994). The random effects estimator is akin to a random coefficient model in which the within- and between-study variances are used as weights.

studies (CV, CEs, and RP). The random effects model is an attractive specification because it assumes that the population effect sizes for different studies (or target types, or methods) are randomly drawn from a normal distribution. The results are therefore easier to generalise to the larger population, and the specification is such that substantially more degrees of freedom are left. Moreover, as a result of incorporating random study effects (or, alternatively, target type and method effects), the error variance-covariance matrix has a block-diagonal structure with non-zero covariances, which is very similar to a specification that allows for dependence between measurements sampled from the same primary study – or, alternatively, from the same target type, or using the same method (see Florax, 2002b). The results of the random effects model, using weights for the precision with which the WTP has been measured in the underlying studies, are presented in Table 8-2.

Table 8-2: Random effects specifications, with random effects for studies, target types, and method types[a]

Variable / Random effects	Studies[b]	Targets	Methods
Constant	5.97	1.49	–5.48**
	(1.26)	(0.65)	(–2.08)
Risk assessment and income			
Medium risk	0.12*	0.79***	0.21*
	(1.72)	(3.62)	(1.64)
High risk	0.82***	0.90***	0.76***
	(12.35)	(4.36)	(5.89)
Log(GDP)	–0.31	0.26	0.77***
	(–0.65)	(1.13)	(3.02)
N	315	316	316
LM(FE/RE vs no effects)	1599.68***	1185.89***	785.18***
LM(Hausman)	3.42	53.39***	0.63

Note:
[a] The variables are weighted using the number of observations in the underlying studies as weights. Significance is indicated by ***, ** and * for the 1, 5, and 10 per cent level, respectively, with *z*-ratios in parentheses. The omitted category is low risk.
[b] For reasons of identification, the single result of Misra et al. (1991) was omitted in this specification.

Table 8-2 shows that, for all specifications, the corresponding Lagrange Multiplier (LM) tests prefer a fixed or random effects specification over a specification without such effects. The Hausman test indicates preference for the random over the fixed effects specification when the random effects refer to studies or methods, but the fixed effects model is preferable for the specification with random target types.

Although the random effects model is based on an attractive estimator because of its less restrictive assumptions, the downside is that the estimator leads to bias in the coefficient estimates if the random effects are correlated with the other regressors. This is actually very likely in this case because studies, target types, and methods are correlated with the risk levels and/or the level of GDP per capita. Moreover, the differences in terms of studies, targets as well as methods are observable. If part of that information were included in the specification by means of dummy variables, the assumption of random effects (according to either studies, targets or methods, whichever is not operationalized using dummy variables) being uncorrelated with the explanatory variables would become even more questionable. Hence, for the remainder of the analysis, we used the fixed effects estimator in a simple linear, additive specification.

Ideally, a general-to-specific stepwise regression approach would be used to arrive at a robust specification following a clear methodology (see, e.g., Dalhuisen et al., 2003, for an example in the context of meta-analysis). However, for meta-regressions, in which typically a rather limited number of observations coincides with the need to use a relatively large number of dummy variables to capture heterogeneity across studies, such an approach is not without problems. Specifically, linear combinations of dummy variables capturing differences in target types, risk level, and other study characteristics may create undue multicollinearity problems because of near exact linear dependencies between two or more variables. We investigated this prior to performing the meta-regression by inspecting bivariate correlations, the condition number of various design matrices, and variance inflation factors (for details, see Belsley et al., 1980). As a result, we identified two potentially serious collinearity problems. First, combining the series of study characteristics identified in Section 8-2 with the target or target type variables (excluding a reference category) leads to excessive collinearity. Second, even if we omit the target or target type variables from the design matrix, the variables "stratification of respondents" and "*ex post* risk" create collinearity problems. The latter can be remedied by re-classifying these variables. We therefore distinguished between respondents by creating a dummy variable "consumer respondents" and a group "farmers" or a stratified combination of consumers and farmers in one category, and we distinguished "general risk" from a combined category *ex ante* or *ex post* risk. The collinearity problem, however, had implications for the specification strategy because it precluded starting from the most general specification.

In the first step, therefore, we used a model distinguishing between different baseline risk levels, and income, as before. In addition, we included dummy variables for geographical location (non-US countries versus the US), the valuation method (choice experiments and revealed preferences vs. stated preferences), the survey type (sampling of consumers vs. sampling of farmers or stratified sampling, face-to-face vs. a mail-in survey, and quality checks on the survey in terms of pre-testing or bias control), risk perception (a potential scenario vs. an actual scenario, and general vs. *ex ante* or *ex post* risk), the payment vehicle (a price premium or separate billing vs. yield loss), the type of safety device (integrated pest management and a ban on specific pesticides, with eco-labelling as the omitted category), the health-risk vehicle in consumer health studies (fresh food in general rather than one specific fresh food), and the source of pesticide risk (one specific pesticide vs. pesticides in general). The regression results are WLS estimates, and they are given in column (1) of Table 8-3.

In the second step, we developed a specific model using stepwise deletion of variables that were not significant ($p > 0.10$)[78]. Given our specific interest in the demand for environmental quality and food safety, we always retained the risk variables, as well as the income variable. See column (2) of Table 8-3 for the results. An LR test on the restrictions is not rejected.

[78] An alternative approach would be to first test groups of variables using an F or χ^2 test (see Table 8-3 for the group headings), and subsequently perform tests on individual variables within groups.

In the third step, we added either the overall target variables to the specific model, or the series of target type variables (as distinguished in Figure 6-2). See columns (3) and (4) of Table 8-3 for the results.

Table 8-3: Extended specifications with fixed effects for differences between studies, using the weighted least squares (WLS) estimator[a]

Variable / Specification	(1) General	(2) Specific	(3) Incl. targets	(4) Incl. target types
Constant	2.20	7.87^{***}	4.31	2.85
	(0.22)	(2.62)	(0.38)	(0.23)
Targets and target types[b]				
Farmers			0.96	
			(0.29)	
Acute effects				0.42
				(0.12)
Chronic effects				0.42
				(0.12)
General				0.73
				(0.21)
Consumers			omitted	
Acute effects				–0.06
				(–0.04)
Chronic effects				0.18
				(0.12)
Cancer risk				–0.15
				(–0.40)
General				omitted
Aquatic ecosystem			1.21	
			(0.37)	
Surface water				0.63
				(0.18)
Groundwater				0.68
				(0.20)
Acquatic organisms				0.56
				(0.16)
Terrestrial ecosystem			1.17	
			(0.36)	
Mammals				0.54
				(0.16)
Birds				0.55
				(0.16)
Biodiversity				2.39
				(0.69)
Beneficial insects				0.56
				(0.16)
Risk assessment and income				
Medium risk	0.11^{*}	0.13^{**}	0.14^{**}	0.17^{***}
	(1.66)	(2.06)	(2.19)	(2.76)
High risk	0.82^{***}	0.82^{***}	0.81^{***}	0.78^{***}

Variable / Specification	(1) General	(2) Specific	(3) Incl. targets	(4) Incl. target types
	(12.49)	(12.62)	(12.77)	(12.58)
(continues)				
Log(GDP)	0.58	0.11	0.38	0.51
	(0.71)	(0.44)	(0.43)	(0.54)
Geographical location				
Non-US	1.95			
	(0.75)			
Method				
Choice experiments	–3.70**	–4.50***	–4.77***	–5.05***
	(–2.26)	(–6.40)	(–4.48)	(–4.25)
Revealed preferences	–7.32***	–8.13***	–7.52***	–7.40***
	(–3.46)	(–11.49)	(–3.69)	(–3.46)
Type survey and sampling				
Consumer respondents	–0.05			
	(–0.05)			
Face-to-face survey	5.54	5.88***	6.06***	6.23***
	(1.61)	(14.68)	(9.03)	(8.03)
Pre-test	–0.16			
	(–1.17)			
Bias control	–0.17***	–0.19***	–0.19***	–0.18***
	(–3.04)	(–3.35)	(–3.42)	(–3.50)
Risk perception				
Potential scenario	1.44			
	(0.29)			
General risk	–0.23			
	(–0.16)			
Payment vehicle				
Price premium	–8.54***	–8.27***	–7.57***	–7.40***
	(–2.81)	(–12.17)	(–3.32)	(–3.07)
Separate billing	–4.76**	–3.19***	–3.16***	–3.15***
	(–2.26)	(–23.99)	(–19.02)	(–18.75)
Type safety device				
Integrated pest management	–2.75*	–3.31***	–3.70***	–2.94*
	(–1.70)	(–4.80)	(–3.10)	(–1.92)
Pesticide ban	0.66	1.17***	1.24***	1.42***
	(0.33)	(4.50)	(3.64)	(2.79)
Health risk vehicle				
All fruits and vegetables	5.46***	6.52***	6.84***	7.29***
	(2.82)	(9.28)	(5.73)	(4.91)
Risk source				
One pesticide	0.47			
	(0.32)			
N	316	316	316	316
R^2-adjusted	0.92	0.92	0.93	0.93
Log-likelihood	–555.48	–556.64	–548.63	–531.66
LR-test[c]	2.32		16.02***	49.96***
F-test	213.33***	323.88***	270.93***	176.26***

[a] The weights are determined as the number of observations in the underlying studies used to determine the risk value. Significance is indicated by ***, ** and * for the 1, 5, and 10 per cent level,

respectively, with *t*-ratios in parentheses. [b] The omitted categories in columns (3) and (4) are health risks to consumers, and general health risks to consumers, respectively. [c] Likelihood Ratio test of the restricted model in column (2) against the unrestricted models in columns (1), (3) and (4).

Table 8-3 shows that the results are very robust across specifications. In particular the marginal effects of increasing the baseline risk level are largely unaffected by the different specifications. Going from low to medium or low to high risk levels increases the WTP by approximately 15 or 80 per cent, respectively. The estimated income elasticity is more volatile, but it is not statistically different from zero. Neither is the WTP for reduced pesticide exposure statistically different for countries outside the US as compared to the US either. The valuation technique is crucial: choice experiments result in lower WTP estimates than contingent valuation, and revealed preference studies lead to the overall lowest WTP values. The table also shows that characteristics of the survey design in stated preference studies have a non-negligible impact on the WTP estimate. Specifically, face-to-face interviews are associated with substantially higher WTP estimates, and bias control slightly lowers the estimated values. Although it is frequently maintained that risk perception is an important phenomenon, we cannot discern systematic differences according to the scenario type or the risk type involved in soliciting WTP values. The payment vehicle is important though: price premiums and separate billing, as compared with yield loss, lead to a significantly lower WTP. Another important result for policy makers is that the type of safety device influences people's WTP. The WTP for risk reduction associated with integrated pest management and eco-labelling is significantly higher than for a ban on specific pesticides. Finally, the WTP for pesticide risk reduction is systematically higher if it relates to all rather than just one fruit or vegetable, although the risk source concerning just one or a multitude of pesticides seems to be irrelevant.

It is reassuring to see that, when we added dummy variables for the different targets or target types, the results for the other variables do not change substantially. Interestingly enough, the results for the targets or target types are not significantly different from zero. Hence, it does not seem to be relevant to consumers whether a pesticide risk reduction is brought about by making the environment safer or by increasing food safety. This is different from what has been found in the literature on the valuation of statistical life, where it generally makes a difference whether people are asked to value risk in the work place or transport safety risk (see, e.g., Miller, 2000). The latter, taken together with the need to slightly deviate from a strict general-to-specific strategy in specifying our model and the inflated standard errors for the target and target type variables, shows that we can still benefit from more primary studies in order to create more variation and alleviate multicollinearity problems. The results for the LR tests on the restrictions also indicate that there is potentially a relevant difference across targets or target types, but multicollinearity precludes a robust identification of such differences.

8.4. Conclusions

Productivity growth in agriculture has been closely related to the increased use of chemical inputs such as fertiliser and pesticides. As an important side-effect, chemical inputs in agricultural production create non-negligible hazards for human health and the quality of aquatic and terrestrial ecosystems. Food safety and environmental sustainability of agriculture have been promoted using policy instruments such as eco-labelling, pesticide bans, integrated pest management and pesticide taxes. Preferably, such policy measures should be related to individuals' WTP for reduced pesticide risk exposure.

We reviewed the pesticide risk valuation literature, and showed that substantial information on individuals' WTP for reduced pesticide risk exposure is available. The literature is, however, very diverse. It provides WTP estimates not only for various human health risks, but also for the risk of environmental degradation. Our taxonomy of the different effects of pesticide risk exposure distinguishes effects on farmers, consumers, the aquatic and the terrestrial ecosystem, including more detailed target types per category.

Our data retrieval process eventually yielded 316 usable individual WTP assessments sampled from 15 studies containing monetary estimates, allowing the calculation of mean and median effects of the different pesticide risks, both by target type and by study.

A meta-regression framework to account for inherent differences in the WTP values for reduced risk exposure provided strong evidence that the WTP for reduced risk exposure increases by approximately 15 and 80 per cent in going from low to medium and low to high risk-exposure levels, respectively. The income elasticity of the WTP for reduced risk exposure is not significantly different from zero, and there do not seem to be geographical differences in valuation. The results also show, however, that differences across studies, in terms of characteristics of the research design (specifically, the valuation technique, the type of survey, the payment vehicle, and the type safety device), are important drivers of the valuation results.

The results of our meta-analysis reveal that it may still be too early for a meta-analysis to be able to provide a consistent and robust picture of the large range of WTP assessments across different target types. Given the intrinsic heterogeneity in effects of pesticide usage across different target types (food safety, health effects on farmers, and aquatic and terrestrial ecosystems), as well as across geographical space, and given the non-negligible impact of research designs on the estimated WTP values, more primary research on pesticide risk valuation is called for. Some important implications for future primary research can, however, be drawn from this meta-analysis. Apart from the above-mentioned implications of research design characteristics, it is important that future valuation work carefully specifies both the baseline level of risk and the change in the risk level. More attention is also needed for the income and potentially location-specific nature of the valuation of reductions in pesticide risk exposure.

Appendix

Standardisation of effect sizes

The WTP estimates given in the underlying studies, $\tilde{T}_i$, are transformed to standardised WTP estimates, *T*, defined as the WTP value per person, per year, in US dollars of the year 2000, using the transformation function $T = c \cdot t \cdot m_i \cdot \tilde{T}_i$. The subscript *i* refers to three different measurement units: (1) per household, per time period; (2) per unit of produce weight; and (3) per pesticide application, per acre of cropland treated. Corresponding transformation factors are defined as:

(1) $m_1 = d/h$, where h is the average household size in a specific country and year, and d a conversion factor to change a given time period to the per-year basis;
(2) $m_2 = c/w$, where c is the average annual per capita consumption of the produce concerned, and w a conversion factor from the weight unit concerned to the weight unit of c; and
(3) $m_3 = s/r$, where s is the average annual number of pesticide treatments for the crops concerned, and r the rural density of the country concerned, defined as the ratio of the rural population over the total acreage of land area.

The transformation factor t refers to the conversion of current prices to 2000 proces, and is in fact a GDP deflator. The conversion of local currencies to US dollars of 2000 is implemented using the 2000 Purchasing Power Parity (PPP). Both the GDP deflators and the PPPs are taken from *World Development Indicators* (World Bank, 2002). The same procedure is applied to standardise GDPs used as a proxy of the baseline income level. Further details are available upon request.

Baseline risk level

The baseline risk levels reported in the original studies can be classified into a three-level risk scale, distinguishing between low, medium, and high-risk. Some studies already use this classification. Studies concerning environmental and farmers risk by Higley and Wintersteen (1992), Lohr et al. (1999), Mullen et al. (1997), Brethour and Weersink (2001), and Cuyno et al. (2001) estimate the initial risk level (for each of the environmental targets analysed) by considering analogous toxicological end-points and classify these end-points according to the aforementioned 3-level risk scale. For some other studies, the baseline risk levels have to be transformed into the 3-level risk scale. We used the following adjustments, based on expert advice of (eco)toxicologists. Further details are again available upon request.

Foster and Mourato (2000) measure negative pesticide impacts on consumers and farmland bird biodiversity using damage estimates. They set the baseline level of human health risk to 100 cases of pesticide intoxication per year, while the number of endangered bird species is set at 9. We classify the risk levels for human health and bird biodiversity as medium and high, respectively.

Wilson (2002) does not report the baseline risk level; nevertheless, useful information on the pesticide risk for human health in Sri Lanka is taken from Sivayoganathan et al. (2000). We classify the human health risks reported in Sivayoganathan et al. (2000) as high.

Bubzy et al. (1995), Eom (1994), Fu et al. (1999), and van Ravenswaay and Hoehn (1991b) estimate WTPs for reducing cancer risk and measure the initial risk level as the number of cases per 10,000 or per 100,000 people. We classify these cancer risks as low, medium, or high if the actual risk is, respectively, lower than 5 cases, between 5 and 12 cases, and higher than 12 cases per 10,000 persons.

Finally, van Ravenswaay and Hoehn (1991a), Misra et al. (1991), Roosen et al. (1998), Hammitt (1993), and Baker and Crosbie (1993) estimate consumers' preferences for a decrease in the health effects due to pesticide residues in fresh food. None of these studies provides the baseline risk level. As a proxy we use the percentage of products in violation of national pesticide residue regulation, as found during the national annual monitoring campaigns, and characterise residues risk as low, medium or high if the percentage of products found to be in violation of national limits is, respectively, lower or equal to 0.5, between 0.5 and 2, and higher than 2.

9. MANAGING PESTICIDE RISKS FOR NON-TARGET ECOSYSTEMS WITH RISK INDEXES: A MULTICRITERIA APPROACH*

Since the late 1970s a wealth of scientific research from different disciplines has shed light on the off-farm human health and environmental risks[79] of pesticide use; it has provided evidence that an indiscriminate use of chemical inputs in the agricultural production system would have not been environmentally or socially sustainable in the long run. This new awareness has prompted OECD countries to design and implement a variety of programmes and policies to reduce risks associated with pesticide use. For a detailed synthesis we refer to OECD, 1996.

After two decades of research activity in the OECD countries, the findings from various pesticide risk scenarios are still contradictory. Due to the historically human-driven rather than environmentally-driven background of pesticide risk management in the past decades, which have led to a situation where potential pesticide risks to human health (general population and agricultural workers) can be considered to be reasonably under control, at present the dimension and nature of pesticide ecological effects is still largely unknown. Recent scientific insights highlight the problem of potential environmental side-effects of plant protection products (to control pests or weeds) on sensitive species and habitats, which are called 'non-target organisms and ecosystems'[80] .

In the search for effective tools to manage pesticide risk for non-target ecosystems, a broad agreement is arising in the scientific community on the usefulness of pesticide *risk indicators* as instruments capable of achieving a meaningful compromise between the demand for a sound scientific approach and the need for transparent public policy tools (OECD, 1997, 1999).

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* *Based on Travisi, Nijkamp, Vighi and Giacomelli (2006b).*

[79] In risk assessment, the terms 'hazard' and 'risk' are used to describe human health, and environmental effects. Hazard is a function of toxicity and exposure; it demonstrates a potential. Risk is the assessment of the actual risk, which is the potential (or the probability) of the hazard to actually occur, given a biotic system exposed to it. In this paper, both the terms 'effect' and 'risk' refer to hazard.

[80] The term 'non-target organism' indicates all the living organisms, with the exception of the pests which are specifically intended to be killed by pesticide applications. Non-target ecosystems are those that, even if not directly treated, can be reached and spoilt by pesticides. Natural processes of environmental diffusion (run-off, drift, volatilization, bioaccumulation, etc.) can mobilise chemicals from the area of application to other non-target compartments, so that pesticide effects can spread – at different space-time scales – beyond local boundaries. As a consequence, surface water, groundwater, epigean and hypogean terrestrial systems can be exposed to potential hazard compounds without being directly touched by pesticide treatments. Some studies mention that as much as 90% of the pesticides applied may or will reach non-target environments (Faasen, 1994).

In the present chapter, some recently developed pesticide risk indices are employed and their potential for management purposes is tested. A pilot approach is proposed, which explores pesticide worst-case hazard scenarios at different space-time scales by means of a set of five ecotoxicological risk indices. The results are then interpreted from the perspective of a decision support method using the Critical Threshold Value approach. Our risk analysis is then enriched within a multicriteria framework which integrates environmental, agronomic, and economic objectives.

In the remainder of the chapter, Section 9.1 discusses the basic principles and challenges in risk indices design. In Section 9.2 the empirical case study is introduced, and the outcomes from our risk assessment analysis are put in the context of a decision support method. Finally, Section 9.3 presents and discusses the main findings.

9.1. Towards risk management tools: pesticide risk indicators

The lively debate on the design of pesticide risk indicators is based on the stance of their assumed complementarities to more consolidated and standardised procedures, such as risk assessment and registration (OECD, 1996). From this perspective, it is an intriguing challenge to strengthen the definition and development of new policy-relevant tools, providing government officials and stakeholders with additional sound scientific and user-friendly support. Indicators should not be used to substitute for existing procedures nor to quantify pesticide risks in a strict sense; rather, they are expected to help national regulatory institutions to estimate general trends in pesticide risk reduction and to judge the effectiveness of their programmes.

The final report of the first OECD "*Workshop on Pesticide Risk Indicators*" (OECD, 1999) emphasises that indicators may be designed for different purposes which will determine how much a sophisticated methodology is required, and how much and what types of data are needed. This argument is essential to guarantee that any interpretation of the meaning of indicators is consistent with the knowledge required, and that the indicators' results are not misinterpreted or employed beyond their proper contests.

On a first level of assessment, risk indicators may be designed as instruments for predictive risk management approaches, to offer preliminary insights into the *status quo* of pesticide risks. They may be developed to obtain baseline information about pesticide use and risks, focusing on one or more realistic hazardous scenarios, and they may guide the identification of potential *trouble spots* and *vulnerable areas* where risk reduction might be requisite. A proper design of risk indicators may also provide insights to compare, and eventually classify, several pesticide risks with respect to both the substance of concern – or a mixture of substances – and the environmental target at stake. Finally, cumulative risks associated with the use of multiple pesticides may also be explored.

On a second level of action, indicators may be specifically designed for monitoring the impacts of pesticides policies and programmes during their several stages of implementation. In this case, indicators should be generated to track risk *trends* over time and space, which also reflect the dynamics in boundary conditions affecting pesticide risks[81].

The previous arguments, though briefly discussed, suggest that challenges for future research are manifold. The first task is striking a compromise between scientific accuracy and decision-making pragmatism. Both experts and managers should be able to give a transparent interpretation of the information provided by indicators. Experts need to accurately interpret, reproduce and eventually refute results, whereas managers are asked to correctly interpret and use outcomes within the decision-making process.

To satisfy such conditions, indicators should be consistent with modern principles of pesticide risk assessment and registration, thus combining information on pesticide hazard and exposure for each of the environmental compartments at risk[82] (see EPPO, 1993, 1994a and 1994b). In this sense, a set of indicators, separately dealing with risks for different environmental targets, would be highly preferable to a single overall one. The use of sets of indicators would allow the results of scientific predictions concerning the severity of different pesticide risks to be compared with some a priori patterns of preferences/priorities, as expressed either by the risk-managers or by the stakeholders involved in the decision process. To some extent, the availability of sets of indicators – providing predictions for each of the endangered environmental compartments – would also enable the evaluation of risk/risk and risk/benefit trade-offs, which are particularly relevant considering the multidimensionality of pesticide impacts.

To return to our main line of reasoning, whenever the indicators' design fulfils risk assessment principles, these instruments might help to develop *plausible visions* on the negative side-effects of pesticide use at different time and spatial scales. Depending on the data and on the specifications set during the design process, the results may provide useful information on both risks for varying endangered compartments (surface water, groundwater, epigean soil, etc.), and risk scenarios at different space-time scales (short-term *vs* long-term horizons; local, regional or global scale)[83].

As already noted, the main challenge is the development of *dynamic* indicators including data on risk (hazard and exposure) and data on the conditions and quantity of pesticide use (amount applied per unit area, total area treated, total quantity of pesticide used, frequency of application, crop timing, formulation type etc.). Such a dynamic perspective would make it possible to measure the impacts of pesticide risk reduction programmes and policies, and to follow risk trends over time and space. Outcomes from a non-static analysis would provide intriguing visions on past and present experiences, giving important feedback for future risk reduction actions.

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[81] To give an example, the time variable might be included as endogenous data in indicators by means of updating information on the level of pesticide use.

[82] This chapter focuses on pesticide risks for non-target ecosystems; the discussion does not address risks to human health.

[83] One should not confuse the ability of indicators to reflect trends over time, i.e. a dynamic perspective, with that of predicting risk scenarios at different time and spatial scales. In the former case, updating data on pesticide use is included in the indicators, thus allowing risk trends to be tracked over time. In the latter case, instead, the time dimension is not endogenous in the scenario, which gives a static picture of potential risks. Different scenarios can refer to varying time horizons, but they remain static visions referring to some fixed spots on the time axis.

To conclude this section, some elements deserve a further comment. First, indicators are crude measures of risks and should not be the sole basis for decision making. Risk trends and other information shown by indicators need to be confirmed and enriched by a closer investigation before regulatory or management action is taken. In addition, indicators are relative measures, not exact measures of real risk, and whether or not it is important for an indicator to correspond closely to real risk depends on the purpose of the indicator and how it will be employed. In this sense, different indicators provide different results, and it is not yet possible to say which results are most accurate or which indicator provides the true information. Given the complexity of real-world conditions and the intrinsic uncertainty of risk, the use of one – or of a combination of – various *alarm systems* should always be preferred as this would lower the chance of missing the management of some non-negligible risks.

9.2. Testing risk indexes: a study of herbicide strategies in Italy

In recent years, a number of *pesticide risk indicators* have been developed to provide information on the level of environmental hazard associated with pesticides by pre-specified criteria. The criteria defined to evaluate the acceptability of environmental risks are generally based on the concept of the toxicity-exposure ratio (TER)[84]. This ratio should be calculated for each of the environmental non-target compartments at risk (surface water, groundwater, soil) to establish critical *threshold values* as a trigger for the need for further investigations. On the other side, TERs can be used in comparative analysis and preliminary analysis of pesticide risks, with the introduction of adequate *safety factors* representing the limits of acceptable risk for each component of the non-target compartments.

In general, the proposed index-systems are founded on the development of a score for a set of physicol-chemical, toxicological, and ecotoxicological properties of the substances considered (see Finizio et al., 2001; Kovach et al., 1992). The scores are then combined by means of an appropriate algorithm in order to obtain a numerical expression of the level of potential risk related to the compound. Rating systems are usually based on standard environmental *worst-case scenarios* to guarantee the highest level of protection in accordance with the stance of the precautionary principle. Consequently, they assess the potential risk (or hazard) inborn in the use of pesticides[85].

[84] A TER is defined as the ratio between a toxicological end-point (i.e. *LD50*, *NOEL*) and a Predicted Environmental Concentration (PEC): *TER* = (*LD50* or *NOEL*)/*PEC*. A *PEC* for a certain pollutant is the pollutant's concentration in a certain environmental compartment. Ideally, *PECs* should be measured in the environmental compartments of concern during monitoring campaigns, but – more often – *PECs* are predicted by means of environmental diffusion modelling.

[85] Risk should not be confused with hazard (or potential risk). The 'hazard' of a pesticide is its potential to impose some negative effects on biological systems. There is a potential risk if predictions show that pesticide concentration in the environment exceeds the environmental quality criteria or risk-based residue limits. Yet, for an assessment of the actual risk, the actual (real) exposure of a biological system has to be compared with concentrations known to actually exert negative effects.

The relatively low complexity of this kind of approach, with respect to complete *on-site* risk assessment procedures, as well as its predictive and comparative potential, suggests that it might be successfully employed to support and orient risk reduction strategies beyond their use within registration procedures. Seen from this perspective, risk indexes would find their natural setting in the preliminary stage of collection of information that precedes the definition and implementation of regulations. In this framework, their adoption might be particularly useful for the development of *plausible visions* of the negative side-effects of pesticide use – at different space-time scales – and for the identification of patterns of priority for action. In a more advanced stage, the adoption of regional or local risk scenarios instead of standard worst-case scenarios would also allow a *site-specific* analysis with a direct connection to the relevant territory. On such conditions, the elaboration of decision-support tools for pesticide-oriented regional sustainability would represent an additional promising dimension of research development.

The focus of our chapter is on the Italian context where rating systems for environmental pesticide risk have been recently developed – with analogous theoretical foundations – within a project sponsored by the Italian Environmental Protection Agency (ANPA). For a comprehensive description of principles and results of the ANPA project, we refer to: Finizio, 1999a and 1999b; Finizio et al., 2001). The usefulness of such tools for management purposes is tested here with an application to conventional agriculture in Italy. The analysis addresses herbicide use and explores herbicide worst-case hazard scenarios, at different space-time scales, for two of the most diffuse field crops: maize and rice[86]. The procedure developed is illustrated in Figure 9-1. Step 1 implies the analysis of the Italian herbicide market – for maize and rice – and the identification of the main risk sources. Step 2 performs hazard assessments of risky agricultural practices and provides plausible visions of different types of ecological impacts. Finally, in Step 3, the outcomes from Step 2 are enriched with additional information on agronomic and economic objectives, and risk scenarios are analysed within the context of decision support methods.

[86] An analogous analysis was also performed for two other important field crops: soy beans and field beet. For synthesis reasons, the chapter only illustrates the case of maize and rice. In Italy, the herbicide market represents more than 44% of the overall pesticide sector (insecticides 29%; fungicides 21%) (Sbriscia Fioretti et al., 1998). The economic value of the herbicide sector reaches 40% of the total pesticide market. More than 70% of the total amount of herbicides consumed is incorporated in the production of maize, wheat, rice and field beet, while fruit and vegetable farming absorbs the remaining part. Among others, rice and field beet are the most herbicide-dependent farming practices (Sbriscia Fioretti et al., 1998).

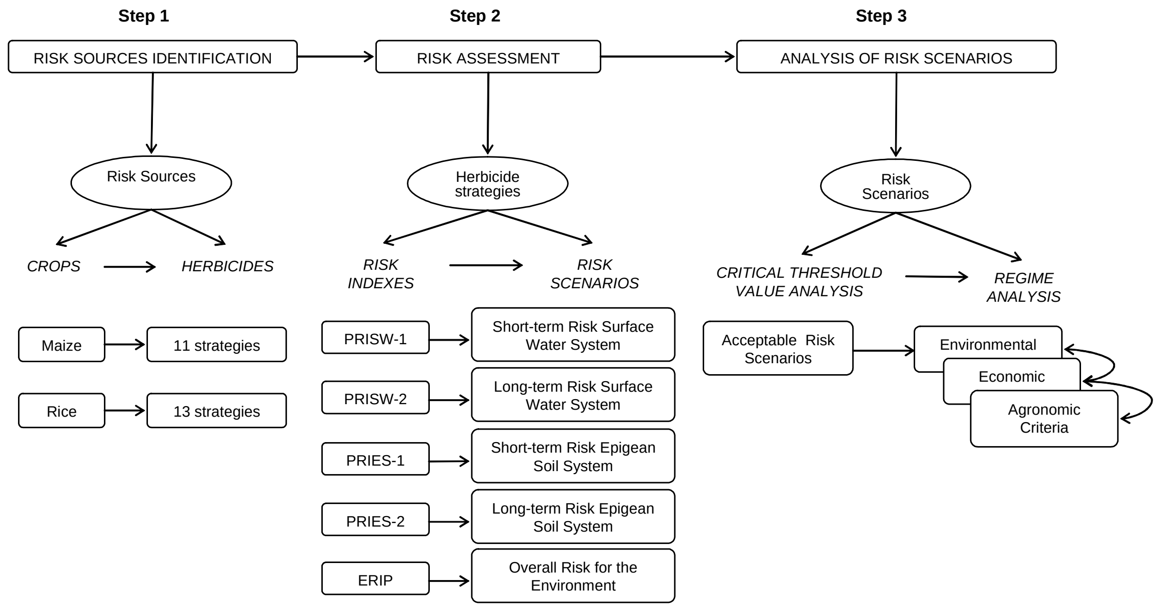

Figure 9-1: Procedure for the analysis of hazard scenarios for non-target environments

Note: Step 1 identifies major risk sources; Step 2 performs risk assessments of hazardous activities; Step 3 analyses risk scenarios in a decision support frame.

9.2.1. Hazard assessment

Thanks to close cooperation with a group of Italian agronomists, we could depict a realistic image of the potential ecological impacts attached to the herbicide practices employed in maize and rice production. For each crop of concern, we identified the herbicide strategies mostly used at a national level with similar agronomic action and purpose[87]. Overall, 24 strategies were selected, which typically implies the use of mixtures of two or more herbicide active ingredients[88]. Table 9-1 offers a summarised overview of the strategies.

To seek a correlation between the use of herbicides and the related risk of occurrence of negative side effects on environmental compartments, each strategy was characterised by means of five *eco-toxicological* risk indices assessing hazards for both aquatic and terrestrial non-target ecosystems[89]. The indexes, developed in a project sponsored by the Italian Environmental Protection Agency ANPA, are entirely based on information required by Annex VI of the EU Directive

[87] For each crop type, the selected strategies are those that are applied to most of the national crop acreage; they have similar agronomic usage and are virtually mutually alternative to one another.

[88] Since crops can be treated at different stages of the vegetative cycle, the selected strategies include either pre-emergence and post-emergence herbicides (or a combination of the two). Pre-emergence and post emergence herbicides are applied, respectively, before and after the outgrowth of the plant.

[89] In the original ANPA project, three different environments were considered to provide a set of 7 indexes: surface water; terrestrial hypogean; and terrestrial epigean systems. Because of the partial lack of basic toxicological data on the hypogean system, the present study limits the analysis to surface water and epigean soil.

414/91/EEC for the registration and re-registration of plant protection products (Uniform Principles)[90].

Table 9-1: Sets of alternative herbicide strategies representative of maize and rice production, in Italy

	Maize		Rice
A.	Alachlor + Terbuthylazine	A.	Tiocarbazil
B.	Metolachlor + Terbuthylazine	B.	Dimepiperate
C.	Metolachlor + Pendimetalin	C.	Dimepiperate + Molinate
D.	Isoxaflutol + Aclonifen	D.	Tiobencarb
E.	Rimsulfuron + Prosulfuron + Primisulfuron	E.	Tiobencarb + Molinate
		F.	Tiobencarb + Propanile
F.	Rimsulfuron + Dicamba	G.	Molinate
G.	Rimsulfuron + Terbutilazine	H.	Quinclorac
H.	Rimsulfuron + Fluroxipir	I.	Propanile
I.	Nicosulfuron + Dicamba	L.	Pyrazoxyfen
L.	Nicosulfuron +Sulcotrione	M.	Oxadiazon
M.	Nicosulfuron + Prosulfuron + Primisulfuron	N.	Pretilaclor
		O.	Azimsulfuron

Note: In maize production, the current trend is to adopt one single herbicide treatment per crop season, either in pre- or post- agronomic emergence. Alternatives A to D are pre-emergence treatments, while alternatives E to M refer to post emergence action. The peculiar environmental conditions of rice paddyfield are particularly suitable to the development of a complex vegetable biocenosis. Among such weed varieties, those species and ecotypes of monocotyledonous grass belonging to the genus *Echinochloa* represent the most diffuse and aggressive rice weeds. Selected alternatives for rice are those acting against *Echinocloa* distribution in rice paddies.

For surface water and the hypogean soil system, two different time-spatial scales are taken into consideration. On the one side, the short-term (local-scale) indices refer to a risk imposed by a pesticide immediately after the on-field application (PRISW-1: Short-Term Pesticide Risk Index for Surface Water System; PRIES-1: Short-Term Pesticide Risk Index for Epigean Soil System). On the other side, two indices aim at evaluating pesticide risk in a medium-time horizon and on an area beyond the local boundaries (PRISW-2: Long-Term Pesticide Risk Index for Surface Water System; PRIES-2: Long-Term Pesticide Risk Index for Epigean Soil System). Finally, a comprehensive index evaluates the overall risks imposed by pesticides on the environment (ERIP: Environmental Risk Index for Pesticides).

Consistent with modern risk assessment approaches, the indices integrate exposure parameters (rate of application, environmental distribution, and bioaccumulation, and soil persistence) and information on the effects (EC_{50}, NOEL) that pesticides can exert on a set of non-target organisms considered as representative of each environmental system[91].

[90] Uniform Principles aim at standardising pesticides admission procedures in Europe, by providing a general outline and guidance of the effects evaluation. According to the Uniform Principles, all aspects of human health and the environment – including biota – have to be considered in the evaluation of fate, distribution, and probable effects of pesticides.

[91] Representative bioindicators are selected according to Directive 414/91/EEC. For surface water and epigean soil, representative non-target organisms are, respectively: Algae, *Daphnia magna*, Fish; Plants, Bees, Beneficial arthropods, Birds, Mammals. The general index ERIP considers all previous

The general procedure for the development of the indices consists of two sequential steps. First, a PEC is calculated using simple dilution models or more complex models, based on the fugacity approach and specific for surface water inputs (Fugacity Level I: by Mackay, 1991; SoilFug: by Di Guardo et al., 1994)[92]. Once a PEC is available, TERs are calculated by mean of toxicity data concerning relevant bioindicators. The algorithms assign sub-scores to each TER value, which are then weighted by considering the ecological role of each bioindicator and its meaning in the overall risk evaluation. Finally, sub-scores are combined in a suitable algorithm to provide a final synthetic numerical outcome (see Finizio et al., 2001]). For long-term and general indexes, because of difficulties in quantitatively calculating a PEC, the procedure may eventually be different. In this case, a *scoring approach* substitutes the *PEC approach*, which is used to convert fate variables (i.e. persistence, bioaccumulation, etc.) into scores, subsequently combined into an overall score for exposure. When dealing with mixtures, the original algorithms have been modified under the hypothesis that all considered chemicals act with the same mode of action (in Greco et al., 1992: the *Concentration Additional Model* by Loewe)[93].

Indexes can assume values ranging from 0 (virtually no risk) to 100 (maximum risk).

Table 9-3 illustrates risk scores as computed for the 24 herbicide treatments assessed. It goes without saying that managers and stakeholders might run into cognitive difficulties when dealing with scientific outcomes synthetically expressed as numerical scores. The ecotoxicological information synthesized by risk indexes, need to be converted into unequivocal terms. To overcome this problem, for each ecotoxicological index we developed a *risk classification* in which increasing scores correspond to more severe risk levels. Risks range from *negligible* to *very high*[94] . Each risk level was then illustrated in qualitative terms providing a brief description of the possible impacts that experts ascribe to it. Different risk

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bioindicators plus Earthworms. In the case of lack of toxicity data, some default assumptions were used (for details, see Finizio et al., 2001).

[92] An exhaustive explanation of the environmental scenarios in which *PECs* are calculated is reported in Finizio, 1999b.

[93] The unspecific narcotic effect of herbicides on non-target organisms justifies the adoption of such a stance. However, we are conscious that this approximation might increase the level of uncertainty of our risk predictions. Particularly, we expect that risk measures related to mixtures might be slightly overestimated here. Nevertheless, considering that the actual knowledge on mixture toxicity is still poor and that available data on pesticide use often underestimate the magnitude of this environmental problem, we believe that this bias might be tolerated here. The actual knowledge on mixtures' toxicity is still extremely slight, and many efforts are still required for a complete understanding of the interaction mechanisms among substances. For mixtures, algorithms were modified as follows: for each *jth* representative bioindicator, an overall toxicity-exposure ratio, denoted by TER_{mixj}, substitutes *TERs* due to single substances. TER_{mixj} is calculated by adding the unit of toxicity (TU_i) relative to each ith component of the mixture, as follows:

$$TER_{mixj} = 1 \Big/ \sum_i \left(1/TU_I\right),$$

where: $TU_i = (1/TER_i)$; $i = (1,..,n)$, with n = number of components of the mixture; $j = (1,..,m)$, with m=number of representative bioindicators. The complete database and an electronic page for the calculation of the 5 indexes are available upon request to the author.

[94] Risk classes are defined in the light of results obtained from the indexes' validation procedure, performed within the ANPA project.

levels are therefore associated with plausible visions of potential environmental effects, as illustrated in Table 9-2.

In a management context, the previous step is essential to establish a dialogue with managers and stakeholders, since each of the parties involved needs to accurately know what the actual state of risk is before formulating any hypothesis on management strategies. In our case, the identification of risk scenarios through risk indices allowed us to set – for each environmental compartment at stake – an a priori set of reference values for the analysis of possible hazard scenarios.

Table 9-2: Classes of risk and related potential negative impacts for each ecotoxicological risk indices

Risk Level	PRISW-1	Potential Short-term Effects on Aquatic Ecosystem
Negligible	≤ 5	Negligible impacts
Low	>5- ≤ 15	Moderate alteration of the aquatic biotic communities.
Medium	>15- ≤ 40	Alteration of the aquatic biotic communities. Local dying out of most sensitive species of fishes and invertebrates.
High	>40- ≤ 80	Alteration of the aquatic biotic communities with reduction in sensitive species of fishes and invertebrates. Potential reduction in the community productivity.
Very high	> 80	Serious effects on aquatic communities with a reduction in growth and productivity. Damping-off of fishes and invertebrates.

Risk Level	PRISW-2	Potential Long-term Effects on Aquatic Ecosystem
Negligible	≤ 5	Negligible impacts
Low	>5- ≤ 10	Moderate alteration of the aquatic biotic communities.
Medium	>10 ≤ 30	Alteration of the aquatic biotic communities. Local dying out of most sensitive species of fishes and invertebrates; bioaccumulation in the trophic chain.
High	>30≤ 60	Serious alteration of the aquatic biotic communities with changes in the existing species of fishes and invertebrates. Potential reduction in the community productivity. Bioaccumulation in the trophic chain.
Very high	> 60	Very serious effect on aquatic communities with a reduction in growth and productivity. Damping-off of fishes and invertebrates.

Risk Level	PRIES-1	Potential Short-term Effects on Terrestrial Ecosystem
Negligible	≤ 5	Negligible impacts
Low	>5- ≤ 15	Moderate alteration of terrestrial non-target communities
Medium	>15 ≤ 50	Alteration of terrestrial non-target communities: beneficial arthropods (pollinator insects and natural pests' antagonists), birds and small mammals. Local migration of most sensitive species.
High	>50≤ 70	Serious alteration of population dynamics of terrestrial non-target species. Damping-off of most sensitive arthropods, birds and small mammals.
Very high	> 70	Very serious effects on terrestrial organisms with a reduction in growth and productivity. Significant increase of the death rate of populations.

Risk Level	PRIES-2	Potential Long-term Effects on Terrestrial Ecosystem
Negligible	≤ 5	Negligible impacts
Low	>5- ≤ 15	Moderate alterations of terrestrial non-target communities: beneficial arthropods (pollinator insects, natural pests' antagonists), birds and small mammals.
Medium	>15 ≤ 40	Alteration of terrestrial non-target communities. Local migration of most sensitive species. Bioaccumulation in the trophic chain.
High	>40 ≤ 70	Serious alteration of population dynamics of terrestrial non-target species. Damping-off of most sensitive arthropods, birds and small mammals. Bioaccumulation in the trophic chain.
Very high	> 70	Very serious effects on terrestrial organisms with a reduction in growth and productivity. Significative increase of the death rate of populations. Bioaccumulation in the trophic chain. Effects on bird populations might extend beyond local boundaries.

Risk Level	ERIP	Potential Effects on General Environment
Negligible	≤ 10	Negligible impacts
Low	>10- ≤ 25	Low probability of alterations of aquatic and terrestrial ecosystems in a local scale.
Medium	>20 ≤ 40	Moderate probability of alterations of aquatic and terrestrial ecosystems in a medium scale horizon
High	>40 ≤ 60	Significant probability of alterations of aquatic and terrestrial ecosystems in a medium scale horizon
Very high	> 60	High probability of serious alterations of aquatic and terrestrial ecosystems in a medium scale horizon

9.2.2. Analysis of hazard scenarios

The design methodology for the analysis of risk scenarios is based on a joint use of various multicriteria evaluation methods. The core of the methodology is represented by the Regime Analysis, extended with a complementary methodology, viz. the Flag Model.

Flag Model

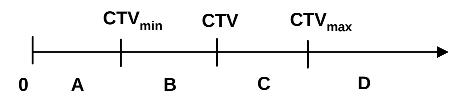
The main purpose of the Flag Model is to analyse whether one or more policy alternatives can be classified as acceptable or not in light of an a priori set of constraints. The model does so by comparing impact values with a set of reference values (called Critical Threshold Values). The Flag Model assesses the degree to which competing alternatives fulfil pre-defined standards or normative statements in an evaluation process (for applications of the Flag Model, see: Nijkamp and Ouwersloot, 1998; Nijkamp and Vreeker, 2000). The Flag Model can operate both as a classification procedure and as a visualisation method. In the former case – for example, in combination with Regime Analysis – the Flag Model can determine acceptable alternatives; accordingly, examined alternatives can then be ranked by means of Regime Analysis. In the latter case, the Flag Model can be used to visualise in an appealing way the results obtained, for example, from Regime Analysis or from other sets of classification or evaluation methods.

In this chapter, the Flag Model operates as a classification procedure to identify acceptable herbicide practices that do not pose critical risk scenarios. Acceptable alternatives are then ranked by means of Regime Analysis. At the same time, the visualisation of the risk assessment outcomes by means of the Flag model facilitates the analysis of risk scenarios. For each risk indicator, a bandwidth of critical threshold values (CTVs) is defined, which is used to set our reference system for judging the environmental impacts of alternatives[95]. In the present analysis, the bandwidth ranges from a maximum value (CTVmax) to a minimum value (CTVmin). For each risk index, CTVmin, CTV and CTVmax are set equal to the corresponding *low*, *moderate* and *high* upper bound of risk level, respectively. The resulting reference scheme is represented in Figure 9-2. Table 9-3 illustrates results of the Flag Analysis.

For the maize production, the Flag Analysis of risk scenarios provides quite encouraging results, and none of the alternatives gets a black flag. The graphical visualisation suggests that alternative herbicide strategies can be grouped into two major sets which impose comparable risks to non-target agricultural ecosystems. In particular, the alternatives from A to D are related to more severe risk scenarios (yellow or red flags), whereas the remaining alternatives get yellow or green flags. The main differences refer to long-term risk scenarios, both for aquatic and terrestrial ecosystems, which can be traced back to the higher persistency of the herbicide active ingredients used in treatments A, B, C and D. It is interesting to note that such subgroups correspond, exactly, to pre-emergence and post-emergence strategies, respectively. The case of rice is slightly more complex. Two strategies – E and F – present critical risk scenarios associated

[95] Since in many cases experts and decision makers may have conflicting views on the precise level of the acceptable threshold values, a bandwidth of critical threshold values – by way of sensitivity analysis – is often preferred to a single CTV.

with red or black flags both for the aquatic and the terrestrial ecosystems, and are therefore excluded from the set of possible treatments for rice. Alternatives A, C, D, and G are associated with alarming risk scenarios for the surface water environment; whereas, overall, the remaining strategies seem to be more environmentally-benign.



Section **A**	Green	No reason for concern
Section **B**	Yellow	Need for further analysis
Section **C**	Red	Alarming risk scenario
Section **D**	Black	Critical risk scenario

Figure 9-2: Reference scheme for judging alternatives within the Flag Model

Table 9-3: Results of the Flag Model analysis for maize and rice, in Italy

Maize Risk Scenarios	Alternative herbicide strategies										
	A	B	C	D	E	F	G	H	I	L	M
Aquatic Ecosystem - Short Term	Y	Y	Y	Y	Y	G	Y	G	G	Y	Y
Aquatic Ecosystem - Long Term	R	R	R	R	Y	G	Y	G	G	G	Y
Terrestrial Ecosystem - Short Term	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
Terrestrial Ecosystem - Long Term	R	R	R	Y	G	Y	Y	Y	Y	Y	G
General Environment	R	Y	R	Y	Y	Y	Y	Y	Y	G	Y

Rice Risk Scenarios	Alternative herbicide strategies												
	A	B	C	D	E	F	G	H	I	L	M	N	O
Aquatic Ecosystem - Short Term	R	Y	R	R	R	Y	R	Y	Y	Y	Y	Y	Y
Aquatic Ecosystem - Long Term	R	Y	R	R	B	B	R	Y	Y	Y	Y	Y	Y
Terrestrial Ecosystem - Short Term	Y	Y	Y	Y	B	B	Y	Y	Y	Y	R	Y	Y
Terrestrial Ecosystem - Long Term	Y	Y	Y	Y	R	R	Y	Y	Y	Y	Y	Y	G
General Environment	Y	Y	R	Y	R	R	Y	Y	Y	Y	Y	Y	Y

Regime Analysis

The risk analysis is finally enriched within a MCA frame which integrates environmental, agronomic, and economic objectives. Multicriteria analysis comprises various classes of decision-making approaches. The multi-assessment method used in our methodology is Regime Analysis, a discrete multi-assessment method (for details, see Nijkamp et al., 1990). Regime is a generalised form of concordance analysis, based in essence on a generalisation of pairwise comparison methods. The fundamental framework of the method is based upon two kinds of input data: an impact matrix (structured information table), and a set of (politically determined) weights (for details, we recommend Hinloopen et al., 1983; Hermanides and Nijkamp, 1998). The impact matrix is composed of elements that measure the effect of each considered alternative in relation to each policy relevant criterion. The set of weights incorporates information concerning the relative importance of the criteria in the evaluation. If there is no prioritisation of criteria in the evaluation process, all criteria will be assigned the same numerical weight value.

In our analysis, Regime is applied separately to each of two sets of acceptable alternative herbicide strategies for maize and rice production. The relevant criteria include short-term and long-term risks for non-target ecosystems, agronomic efficacy[96], and the cost of herbicide alternatives. The impact matrixes are presented in Table 9-4. We compare results obtained using two different weight vectors[97]: one is aiming for the higher level of environmental protection; the other strongly favours the agronomic performance (see Figure 9-3 and Figure 9-4). In particular, the 'environmental' weight vector favours the long-term environmental sustainability against the cost of the herbicide treatment; nevertheless, the agricultural efficacy is still considered a relevant decision criterion. The 'agricultural performance' vector, by contrast, is unfavourable to expensive and/or inefficient herbicide strategies. A closer look at the results shows that the rankings of the alternatives obtained using the two aforementioned decision perspectives are quite close to each other, especially in the case of maize production. Both for maize and rice, the best herbicide strategy does not change as the weight vector changes; this suggests that the first-ranked alternative satisfies both the environment and agriculture-driven perspectives. For maize, even the following three alternatives remain unchanged, although their relative order is different. Some larger variations can be noticed in the lowest positions of the ranking; however, the whole variation of the decision scores does not exceed 0.3 points. For rice, the decision scores show a slightly higher variation which reaches 0.35 – using the environmental weighting vector – and 0.29 using the second weighting vector. Consistent with the results of the Flag Model for rice, if the two rejected alternatives – E and F – are included in the analysis, they do indeed get the two lowest scores in the 'environmental' ranking. However, in the 'agricultural' ranking, alternative F is strongly favoured thanks to a high agronomic efficacy and a low cost (Figure 9-5).

To examine the usefulness of this type of analysis for management purposes, let us take a closer look at the Italian herbicide market and see whether our best alternatives correspond to what is actually used by the Italian farmers. The case of maize is particularly interesting; Table 9-5 shows the retail data of alternative herbicide treatments for the maize production in Italy (1999). As can be observed, Italian farmers strongly prefer adopting pre-emergence herbicide treatments (options A to D) as opposed to post-emergence ones (alternatives E to M). The reason for such preference is easily imputable to the lower cost of pre-emergence alternatives and, secondly, to a slightly stronger agronomic performance compared with post-emergence actions. The actual situation of the herbicide practices in Italy is therefore in contrast with what – in our analysis – emerges as the herbicide options to be adopted for the satisfaction of basic economic, agronomic, and environmental criteria. Alternative M, for instance, which has the highest decision score using both the weighting procedures, is used to treat only

[96] In particular, we consider the agronomic efficacy of alternative herbicide treatments against grass weeds and dicotyledonous weeds. The data employed in the analysis refer to the potential agronomic efficacy of different chemical weed control strategies, as evaluated in field experiments (Rapparini, 1998, 1999a, 1999b; Rapparini et al., 1998). For rice, in particular, the agronomic efficacy against grass weeds refers to *Echinochloa* spp., one of the most aggressive weeds in rice production.

[97] The 'environmental' vector assigns: 4 and 3 to the long-term and short-term risk indicators, respectively; 2 to the agronomic efficacy criteria; and 1 to the cost criterion. The 'agricultural performance' vector, by contrast, assigns: 4 to the agronomic efficacy criteria, 3 to the cost criterion, and 2 and 1, to the long-term and short-term risk indicators, respectively.

1.4% of the total area under corn cultivation. By contrast, alternative B, employed on 22.7% of the total area under maize, is placed at the bottom of the Regime ranking; even when embracing a more agricultural-driven perspective it is surpassed by a number of other alternatives.

Table 9-4: Impact matrix of alternative herbicide treatments for maize and rice crops, respectively, in Italy

CRITERIA (*)	MAIZE: Alternative Herbicide Strategies										
	A	B	C	D	E	F	G	H	I	L	M
Cost of the treatment	57.8	56.8	75.9	54.2	81.1	94.5	72.3	90.9	90.6	118.5	77.2
Agronomic Efficacy - Grass weeds	10.0	10.0	10.0	8.0	9.2	9.2	9.4	9.2	9.7	9.7	9.7
Agronomic Efficacy - Dicotyledon weeds	9.5	9.2	8.3	8.8	9.2	9.8	9.6	8.5	9.9	9.7	9.4
Risk for Aquatic Ecosystem - Short-term	21.5	17.5	27.0	17.5	6.0	0.0	17.5	0.0	0.0	24.0	6.0
Risk for Aquatic Ecosystem - Long-term	56.3	41.2	58.8	39.0	7.5	2.1	23.7	1.8	3.6	3.5	9.0
Risk for Terrestrial Ecosystem - Short-term	19.0	28.0	31.0	20.5	6.0	33.0	36.0	35.0	39.0	35.5	9.0
Risk for Terrestrial Ecosystem - Long-term	48.3	48.4	51.6	20.9	5.0	9.4	25.2	6.8	9.8	6.0	5.4
Risk for General Environment	45.2	40.2	53.6	36.6	15.9	11.2	21.3	16.2	12.0	7.4	16.7

CRITERIA (*)	RICE: Alternative Herbicide Strategies												
	A	B	C	D	E	F	G	H	I	L	M	N	O
Cost of the treatment	149.3	117.8	67.1	132.2	127.6	51.1	126.5	130.2	85.2	198.3	43.6	39.3	124.0
Agronomic Efficacy - Grass weeds	4.0	4.0	4.0	4.5	5.0	5.3	4.4	4.1	5.9	7.0	6.1	5.8	8.5
Agronomic Efficacy - Dicotyledon weeds	10.0	10.0	10.0	10.0	10.0	10.0	10.0	7.0	10.0	10.0	8.0	9.0	8.5
Risk for Aquatic Ecosystem - Short-term	43.0	17.5	43.0	51.0	51.0	39.0	54.0	24.0	39.0	5.5	29.5	29.5	12.0
Risk for Aquatic Ecosystem - Long-term	30.6	17.0	52.2	35.0	70.2	61.4	35.2	14.7	26.4	14.0	20.0	22.9	7.4
Risk for Terrestrial Ecosystem - Short-term	30.0	36.0	36.0	39.0	73.0	72.0	40.0	19.5	47.0	35.0	53.5	29.0	7.0
Risk for Terrestrial Ecosystem - Long-term	27.0	18.0	39.4	36.0	57.4	52.0	21.4	10.8	16.0	18.8	20.5	14.0	3.2
Risk for General Environment	20.6	17.1	42.1	25.8	50.8	46.1	25.0	14.7	20.3	11.3	27.2	23.1	12.3

Note: For each decision criteria we report values before standardization. (*) Cost of the treatment expressed in euros per hectare. Agronomic efficacy can assume values ranging from 4 to 10. Risk indicators can assume values ranging from 0 to 100.

Table 9-5: Retail data and spread of alternative weed control strategies employed in maize production (1999), in Italy

	Alternative herbicide practices	% Area (°)	
Pre-emergence	A. Alachlor + Terbutilazine	16,4	65,5
	B. Metolachlor + Terbutilazine	22,7	
	C. Metolachlor + Pendimetalin	4,5	
	D. Isoxaflutol + Aclonifen	21,8	
Post-emergence	E. Rimsulfuron + Prosulfuron + Primisulfuron	1,4	21,4
	F. Rimsulfuron + Dicamba	8,2	
	G. Rimsulfuron + Terbutilazine	0,9	
	H. Rimsulfuron + Fluroxipir	0,5	
	I. Nicosulfuron + Dicamba	5,5	
	L. Nicosulfuron +Sulcotrione	3,6	
	M. Nicosulfuron + Prosulfuron + Primisulfuron	1,4	

Note: (°) In Italy, 1,100,000 *ha* of the total cultivated area is under maize crop. 70% is treated with pre-emergence herbicides, whereas the remaining 30% is treated with some post-emergence strategies.

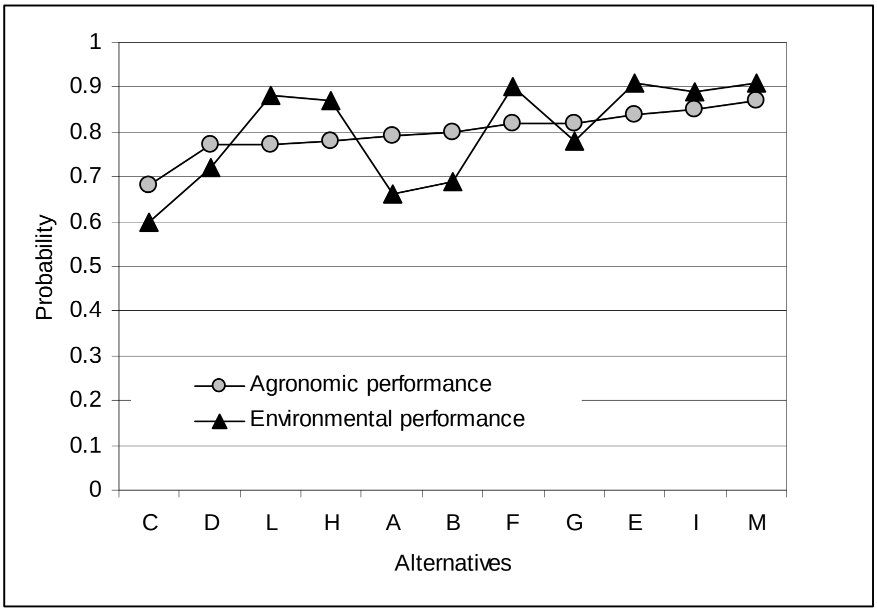


Figure 9-3: Results of Regime Analysis for maize produced in Italy

Note: The rankings of alternatives are obtained applying the 'environmental' and the 'agronomic performance' weight vector.

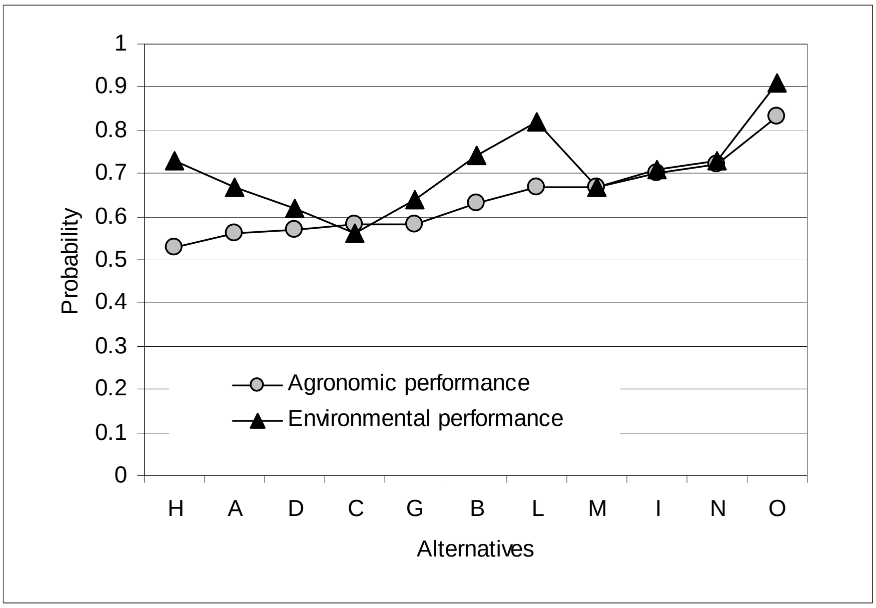


Figure 9-4: Results of Regime Analysis for rice produced in Italy

Note: The rankings of alternatives are obtained applying the 'environmental' and the 'agronomic performance' weight vector.

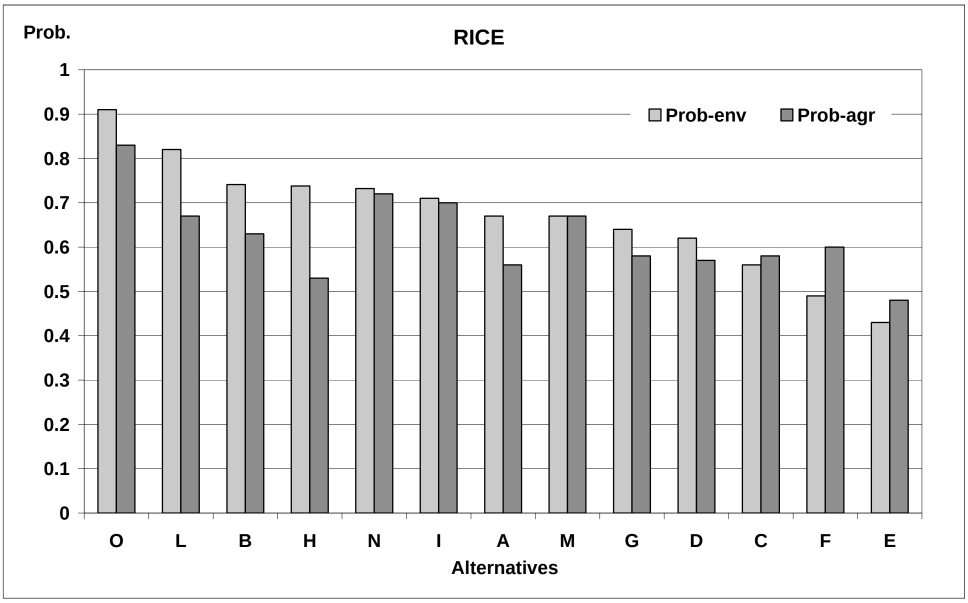


Figure 9-5: Results of Regime Analysis for rice produced in Italy, considering all the initial herbicide alternatives

Note: The rankings of alternatives are obtained applying the 'environmental' and the 'agronomic performance' weight vector.

9.3. Conclusion and policy discussion

The policy issue here is how best to use scientific information on the environmental risk of pesticide use to reduce the impact of conventional agricultural practices. In particular, a more immediate issue is how to combine and analyse the available ecotoxicological risk information for useful management actions.

One aim of this chapter was to draw attention to the design of pesticide risk indicators as innovative risk management instruments. The synthesis of information on pesticide hazard and exposure into risk indices is found to be useful for providing plausible visions of the *status quo* of pesticide risks and to identify potential trouble spots where risk reduction might be a main concern. The inclusion in the analysis of a set of indicators representing pesticide hazards along a number of ecological dimensions is also found to be important for articulating trade-offs in management objectives across different environmental concerns. In addition, our empirical analysis confirms that multicriteria techniques constitute a suitable framework to apply risk indices as decision support tools.

Overall, this study suggests the potential of risk indicators to support the definition of risk management priorities. In this sense, a major contribution of risk indicators is to allow the acquisition of information of primary importance, in an easily interpretable format, and at a relative moderate cost. The level of detail of such tools, either compound or mixture-specific, is indeed very high compared with the cost involved in the design and implementation of this kind of approach. Nevertheless, one should be aware that the cost-effectiveness of an indicator depends on both its purpose and how it will be deployed. Only from this perspective, therefore, is it relevant to express satisfaction or discontent with the quality of results.

Our tentative empirical application suggests that – whenever the analysis aims at giving preliminary insights on the severity and dimension of environmental risks – risk indicators based on standardised worst-case scenarios have relevant management potential. However, as the analysis becomes site-specific and requires direct connection to the territory, the potential of such tools decreases since risk scenarios need to incorporate the environmental variability of local or regional boundary conditions. Finally, whenever the study aims at monitoring risk trends over time and space – keeping data on toxicity and exposure at a lower level of detail – integrated data on conditions and quantity of pesticide use are required.

Yet, many research challenges remain open. In particular, efforts are needed to clarify the basic scientific requirements necessary to design risk indexes, given different management tasks. In this sense, a close cooperation among scientists and managers is advocated to finalise the indexes design procedure, thus increasing their cost-effectiveness and implementation potential.

PART IV: RETROSPECT AND PROSPECT

10. CONCLUSION

10.1. Foreword

At the beginning of this thesis, we considered the current multiple risks and impacts posed by agricultural production and mobility to ecosystems and human health. Seen through the kaleidoscope of economic valuation, the pressure that these phenomena exert on the environment reveals its complexity. Ecosystems are dynamic systems that act and react against the development of our network economy on the basis of a variety of causal relationships, often characterised by uncertainty. Similarly, complexity concerns personal, societal, and decision makers' preferences concerning the trade-off between economic development and environmental protection. Environmental concern has been growing steadily over the years, but it has been growing like a sinusoidal function, where the ups and downs have been driven by the well-known conflicting needs for regional development and growth. Moreover, technological advances in agriculture and mobility and the substantial liberalisation of trade have contributed to ever-increasing stress on the environment over the past 50 years, and will potentially lead to unknown long-term effects. For instance, we do not know exactly what environmental and human health effects GMOs (Genetically Modified Organisms) might have in 100 years time. Likewise, 50 years ago we did not know exactly the environmental and human health risks of DDT and POPs (Persistent Organic Pollutants) used to protect harvests, so that we are now more engaged with remediation than with preventive policy actions. The same applies, for instance, to the heavy use of fossil fuels for transport and their effects in terms of greenhouse gases emissions. Thus, as environmental risks are persistent, dynamic, and complex phenomena, and they are the results of a systematic, deliberate, and continuous pursuit of wealth by business, individuals, and communities, it is crucial that the problem of how to deal with the drawback effects of economic growth should also be addressed with systematic, deliberate, and continuous risk management policy strategies.

As argued in this thesis, environmental valuation can provide a substantial contribution in this direction as it removes one layer of discretion in decision making, by supporting and enhancing the process of environmental-risk (or -impact) policy making at different stages with sound scientific insights. It supports assessment, prioritisation and ranking of risks, and allows the appraisal of alternative management strategies to minimise them. On the other hand, in this thesis we considered environmental valuation as a process in itself. It proceeds from the identification and the analysis of the risk of concern, to the selection of the most suited valuation methods (among those available), up to the provision of estimations of costs and benefits, and the discussion of welfare and policy implications. From this perspective, this dissertation has been looking at environmental valuation not just as a simple collection of techniques (stated or revealed preference (SP or RP) methods, dose-response function, and so forth). Valuation assumes the status of process, whose structure and internal coherence needed to be settled. We were looking for a comprehensive, flexible methodological framework capable of handling the wide-ranging variety of environmental risks, as well as providing reliable responses to the broad variety of relevant policy questions on the environmental impacts caused by agriculture and mobility. This introduced the discussion presented in Chapter 1. A retrospective reflection on such issues contributed to the definition of a general valuation framework where the potential uses of various economic valuation tools are established. Nevertheless, any general framework for policy-oriented analysis needs to be tested on-field. The consistency and the suitability of the proposed valuation framework were therefore experimented and tested with empirical quantitative policy-oriented studies. Interestingly, the empirical studies presented in this thesis referred to two diverse, but equally significant, phenomena responsible for environmental impacts. Mobility and agriculture were the arena for our empirical research, each of which raised a number of significant issues that, in recent years, have reached the top of the agenda of environmental policy makers in Europe. The research questions addressed are repeated here to serve as the background against which to draw our conclusions.

Question 1: Can one rely on stated choice methods for valuing alternative rail noise mitigation plans?

Question 2: How can one capture the intensity of the impact of urban mobility? Which factors explain its intensity, and what is the causal chain that drives it?

Question 3: Which factors influence variations in the willingness-to-pay estimation of risk reductions? How can one estimate the value of pesticide risk reduction with stated choice methods? Is it possible to rely on meta-analysis and value transfer for costing pesticide risks to ecosystems and humans?

Question 4: Is it possible to use eco-toxicological risk indicators to provide sound scientific and user-friendly support for effective ecological risk management?

The remainder of the concluding chapter is organised as follows. Section 10.2 returns to research questions and summarises the main research results and implications of the answers that we found to these questions. Next, in Section 10.3 we provide concluding remarks with respect to the research findings and, then, we discuss the policy implications. Finally, Section 10.4 provides some suggestions for future research by identifying some of the issues that remain to be further investigated.

10.2. Summary of the results

Chapter 1 introduced the role of economic valuation for environmental decision making, and proposed a focus on two phenomena, mobility and agriculture, for which valuation is expected to act as a valuable decision-support tool (Pearce and Seccombe-Hett, 2000). We first referred to the wide-ranging impacts caused by mobility, and focused on two challenging open issues. On one side, we introduced the problem of noise pollution due to rail transport infrastructure; while, on the other side, we looked at the phenomenon of urban sprawl and at its implications in terms of collective impacts. Next, we presented the research challenges that valuation can address when looking at the impacts and risks of agriculture on ecosystems and human health. In this connection, we identified pesticide risk valuation as a relevant research area (Travisi et al., 2006c). Chapter 2 provided a further discussion on the way economic valuation can be integrated into decision making at the local, regional, and national level (Willows and Connell, 2003). The focus was on the use of economic valuation for policy-making purposes. A definition of "environmental externality" and "environmental value" was provided, with a discussion on the various notions of value proposed in the literature (see, e.g., O'Riordan, 1976; Goodpaster, 1978; Rollston, 1988; Bockstael et al., 1991, Turner, 1992) and how they are related to different (competing or complementary?) valuation paradigms. We came to the conclusion that the extreme variability of changes in environmental goods and services, as well as the growing policy demand for environmental economic analysis, requires the adoption of a broad perspective on the problem of the preferred notion of environmental value and, therefore, on the preferred paradigm of environmental valuation. This has strong practical implications. First, it leads to the need to embrace valuation methods and techniques that are not always necessarily rooted in neoclassical welfare economics (Jassen and Munda, 2002). Risk assessment, ecological and ecotoxicological risk indexes, as well as multicriteria analysis, are only few examples of analytical tools that can complement the results provided by valuation methods based on cost-benefit analysis principles, such as RP and SP methods. Second, it leads to a shift in the status commonly attributed to environmental valuation. There are, in fact, several dimensions to this fundamental problem of choosing across various methods that – placed one after the other – elevates valuation from the status of a mere set of techniques to the status of process of analysis. First, heterogeneity across environmental goods and services creates a preliminary step in the valuation process as it imposes the need to know both the object of the analysis and its ultimate purpose. For instance, a strict distinction between effects that have implications on intangible non-use values, on the one hand, and environmental impacts, whose value might be inferred by looking at market prices, on the other, is needed. Similarly, research and policy aims should be made explicit to express a preference concerning either strict quantitative welfare analysis or qualitative and multicriteria studies, eventually involving stakeholder participation. Sometimes, for instance, the quantification of environmental values in monetary terms might raise ethical considerations (Blamey and Common, 2000). At other times, decision criteria other than economic efficiency, e.g. social equity, might also be considered relevant within the decision-making process. Or, technocratic decisions based on a strict prioritisation of risk may be required. Aside from that, uncertainty about risks and complexity of environmental phenomena might form an obstacle to the use of quantitative monetary valuation methods (e.g. SP) or hamper the robustness of willingness-to-pay (WTP) estimations. In these cases, the valuation process might be supplemented and reinforced by research synthesis, such as comparative and meta-analysis (Florax et al., 2002). Moreover, research synthesis and value transfer might be very appealing whenever valuation is required for policy advice at low cost, and quickly. This retrospective look at the process of economic valuation and its integration within environmental decision making brought us from the theory to the empirics of valuation. It is thanks to empirical research that we could actually try to provide methodological innovation based on experimentation.

The second, empirical part of the thesis focused on the problem of valuing the environmental negative-side effects of mobility in urban areas. Among the wide range of impacts due to urban mobility, noise has recently reached the top of the European policy agenda (CEC, 2003). Now national and EU policy makers are asking to be informed on the costs and benefits of reducing noise emission levels, and on the most cost-effective type of possible abatement strategy. Looking for a sound answer to Question 2, Chapters 3 and 4 discussed the theoretical underpinnings of the valuation of noise, and provided a summary of the state of the art of the empirical economic literature on noise valuation, as well as an original empirical study. In the analysis presented in Chapter 3, we motivated the advantages of using SP rather than RP approaches, and our preference for choice experiment (CE) over contingent valuation (CV) methods. In Chapter 4 we then developed a framework for the valuation of several relevant features of rail noise policies using a CE approach. This approach allowed us to understand the preferences of people exposed to rail noise for alternative noise abatement measures, which are expected to differ according to their acoustic efficiency, aesthetics, level of technical innovation, and type of project financing. The study also provided an original contribution to the valuation literature, since it explored the measures and tested the marginal WTP estimates by analysing their econometric robustness with respect to the use of alternative payment vehicles, i.e. an additional local tax and a tax reallocation. The signs of major estimated coefficients are statistically significant and consistent with the theoretical predictions, including that respondents can evaluate price increase. In conformity with the expectation that the WTP should be positively associated with the magnitude of noise reduction, our valuation results showed that individuals are, on average, willing to pay for noise abatement. In particular, respondents welcome additional noise reduction with respect to a minimum decrease able to fulfil national regulation limits. Moreover, the type of noise intervention appeared to influence the respondents' WTP for alternative noise reduction plans. A noise decrease achieved with an additional increase in the height of the barrier is priced at a lower level than an equal noise reduction achieved by technological innovation on train wagons and tracks. The former is associated with a strong disutility even if the policy maker proposes to provide this high barrier together with ornamental vegetation. Therefore, this suggests that the respondent will not accept a further reduction in the regulated noise if this is provided by increasing the current height of the barrier.

The results from Chapter 4 also provided an original contribution for improving the acceptance and realism of the payment vehicle. The outcomes showed that the acceptance of a payment vehicle based on an indirect payment in the form of a tax-reallocation scheme (Bergstrom et al., 2004) is higher than that for a conventional tax scheme. The estimations resulted in a lower evaluation of those noise policies financed via the introduction of a new local tax by 37 percent. As in Bergstrom et al. (2004) in the field of groundwater protection policies, the empirical results of our case study indicate that people in our sample were willing to pay more for noise reduction using a tax reallocation financing mechanism as compared with a special tax financing mechanism. In addition to Bergstrom et al. (2004), whose CV study does not specify the bundle of public services to be traded off for environmental goods, in our survey we described them explicitly to the respondents and referred to two specific types of public services: public administration and public transport, the former and the latter being perceived, respectively, by the residents of the Province of Trento, as 'relatively important' and 'very important'. Moreover, the results show that the individual noise perception is likely to influence the WTP for noise abatement in a predictable way. In particular, respondents with stronger individual perception of noise are more prone to pay to purchase noise abatement. This result signals the importance of knowing as accurately as possible the respondents' profile according to noise perception, and of improving the methods for gathering such information. Finally, estimation results (for the 12 municipalities under consideration) show that, if no policy action is undertaken so as to make additional investments in the train or railroad, and thus be able to reduce aerodynamic noise, traction noise and vibrations, a significant welfare loss may result. An aggregate estimate of the total welfare loss ranges from □ 358,800 to □ 1,432,900.

One more policy relevant field of research addressed in the second part of the thesis was the phenomenon of urban sprawl and its implications in terms of mobility impacts. The European Commission has, in fact, recently recognised urban sprawl as the most urgent urban design issue, because it leads to higher transportation and travel costs, higher energy costs, the loss of rural and green space, and an increase of social segregation and functional division in the city (CEC, 2004). In this connection, Chapter 5 addressed the problem of the impacts of mobility in urban areas, and provided a quantitative analysis of the correlations that associate the defining characteristics of sprawl (density, functional diversity and consumption of rural lands) with the travel impacts. The intent of this chapter was to understand whether there are structural, social and economic elements that contribute, systematically, to increase the demand for urban mobility and, thus, the related environmental impacts, in order to attain policy relevant insights, focused on the Italian context. The valuation paradigm assumed was based on an original conceptual interpretation of the causal chain that drives urban traffic, and the related environmental effects. This chapter combined a static and a dynamic perspective on urban mobility and used cross-section regression analysis and Causal Path Analysis (CPA). The travel impacts were computed for a set of 747 Italian cities, using the mobility impact index (IMPACT) developed by Camagni et al. (2002b). The impacts were estimated using commuting data for 1981 and 1991 at the municipal level. The results showed that, during the decade 1981-1991, the impact of mobility has increased in Italy by up to 37 percent. This increment has been mainly due to a marked shift of modal choices towards private motorised travel modes. Moreover, we examined the connection between the IMPACT and some specific dimensions of cities using multivariate cross-section regression analysis. In particular, we considered factors that can control for the level of sprawl: density, diversity of land use, and the loss of ex-urban agricultural land. The analysis provided robust results that confirm our a priori expectations. In our models, all coefficients had the expected sign and were statistically significant. Less compact and mixed-use cities resulted in higher impacts, since the greater dispersion of activities in sprawl increases automobile dependency, and makes it necessary to spend more time travelling between activities. Moreover, auto use itself also reinforces sprawl. It requires large amounts of land for transportation services and encourages the development of the urban fringe. These results, therefore, suggest that, as the segregation of productive and residential activities increases with sprawl, workers need to travel longer and the self-containment capacity of cities is hampered. Ceteris paribus, this shifts congestion from the core toward the periphery of the transport system, resulting in lower qualities of transport services.

Ultimately, we proposed a conceptual interpretation of the causal chain that links sprawl to travel impacts and used CPA to test it. We argued that cities of relatively compact structure and good functional mix support higher self-containment capacities, and generate more favourable conditions for the competitiveness of public transport in term of average trip time. This will contribute, inter alia, to move people's preferences towards public transport and, consequently, reduce the impacts of urban mobility. The results confirmed our expectations. Sprawl, with its low densities and spatial segregation of activities, moves employment opportunities toward more peripheral areas. This reinforces the need for commuting, so that congestion virtually follows jobs to the city's margins and increases travel demand. The augmented congestion uses up all the available road capacity, creating higher impacts. If not accompanied by investment in transportation, the quality of public transport services worsen. Average trip time rises and workers' travel choices favour private motorised modes, with higher social costs. The travel impacts and their social costs increase.

The third part of the thesis (Chapters 6, 7, 8 and 9) dealt with the valuation of the environmental drawback effects of agricultural production on ecosystems and human health. This aimed at adding research insights on the monetary valuation of pesticide risks, and responding to the consumers' increasing awareness for food safety and the social preference to improve the environmental sustainability of agriculture, as well as to the policy need to know the value of pesticide risks in order to formulate sound policy instruments.

Chapter 6 discussed the theoretical basis of the valuation of pesticide risks and presented a critical overview of the empirical literature on pesticide risk valuation that provides disaggregated willingness-to-pay (WTP) estimates of pesticide risk reduction. Recent multidimensional classification methods were used in a comparative approach as tools for explaining the differences in empirical research findings. The analysis showed that the order of magnitude of WTP is related to both the valuation technique and to the data available from biomedical and eco-toxicological literature. This signals, therefore, that these estimates of pesticide risks cannot be simply averaged over several empirical studies. The order of magnitude of a WTP estimate is, in fact, related to the specific type of risk and to the nature of the risk scenario considered, as well to lay people's subjective perception of risks. The analysis also suggests that, in the risk valuation process, more systematic attention should be paid to the formulation of exogenous "framing assumptions" and to their implementation in single case studies.

Then, Chapter 7 presented the results of an original empirical study recently undertaken in the North of Italy. The study aimed to estimate the economic value of reducing the wide-ranging impacts of pesticide use. The valuation was based on a questionnaire survey undertaken in Milan, one of the biggest metropolitan areas in the North of Italy. A choice experiment (CE) survey was designed to estimate the value of some important pesticide-related environmental attributes, using a "green shopping" payment vehicle. Respondents were asked to view the various environmental impacts of pesticide use in agricultural production as foodstuff attributes to be taken into account in the purchase decision. The environmental attributes taken into consideration were: the reduction in farmland biodiversity; the contamination of soil and groundwater in agricultural land; and the health effects of pesticides on the population in general. The monetary attribute used was the monthly food expenditure bill, by means of which it was possible to estimate the marginal value of the other non-market characteristics. The results confirmed that, on average, respondents are willing to pay substantial price mark-ups for safer agricultural production, which leads to a reduction of pesticide damage. The signs of major estimated coefficients are statistically significant and consistent with the theoretical predictions, including that respondents evaluate price increase negatively, but evaluate risk reduction positively. For any type of pesticide risk, marginal utilities of risk reduction increase as the level of provision increases. Our a priori expectation of the effect of differences in the respondents' socio-economic profiles on attribute coefficients was strengthened by the statistical analysis, with the exception of the effect of gender (negative for women), though results in the valuation literature are also mixed. WTP estimates appear to be positively correlated to income level and concern about pesticides. Our models of the choice responses indicated that the choice between agricultural scenarios depends in predictable ways on the attributes. For example, respondents consider food purchased in shops to be less attractive if the groundwater pollution generated from the food production process is increased. In addition, respondents are against buying cheaper food that has more adverse effects on biodiversity and human health. Finally, and to conclude, another result of our study was the estimation of the value of eliminating all risks from pesticide use in agriculture. According to the contingent valuation estimates, the annual mean WTP amounts to an increase of 19.8 percent in household food expenditure.

Chapter 8 used meta-analysis to provide a formal review of the empirical valuation literature dealing with pesticide risk exposure. We reviewed the pesticide risk valuation literature, and showed that substantial information on individuals' WTP for reduced pesticide risk exposure is available. The literature is, however, very diverse. It provides WTP estimates not only for various human health risks, but also for the risk of environmental degradation. Our taxonomy of the different effects of pesticide risk exposure, based on the results from Chapter 6, distinguishes effects on farmers, consumers, and the aquatic and the terrestrial ecosystem, including more detailed target types per category. Our data retrieval process eventually yielded 316 usable individual WTP assessments sampled from 15 studies containing monetary estimates, thus allowing the calculation of mean and median effects of the different pesticide risks, both by target type and by study. A meta-regression framework to account for inherent differences in the WTP values for reduced risk exposure provided strong evidence that the WTP for reduced risk exposure increases by approximately 15 and 80 per cent in going from low to medium and low to high risk-exposure levels, respectively. The income elasticity of the WTP for reduced risk exposure is not significantly different from zero, and there do not seem to be geographical differences in valuation. However, the results also show that differences across studies, in terms of characteristics of the research design (specifically, the valuation technique, the type of survey, the payment vehicle, and the type of safety device), are important drivers of the valuation results. To conclude, our meta-analysis reveals that it may still be too early for a meta-analysis to be able to provide a consistent and robust picture of the large range of WTP assessments across different target types. Given the intrinsic heterogeneity in effects of pesticide usage across different target types (food safety, health effects on farmers, and aquatic and terrestrial ecosystems), as well as across geographical space, and given the non-negligible impact of research designs on the estimated WTP values, more primary research on pesticide risk valuation is called for.

On the other hand, a proper management of pesticide risks might also require the use of more comprehensive, multicriteria, analytical valuation approaches (OECD, 1999). This is particularly true when the analyses concern future – and therefore uncertain – risk and decision-making scenarios. In such circumstances, where available risk information is uncertain and relevant decision criteria are manifold, effective tools to manage pesticide risks should be capable of reaching a compromise between the demand for a sound scientific approach and the need for a transparent public policy tool. Chapter 9 used some recently developed pesticide risk indexes and tested their potential usefulness for management purposes. In the search for effective pesticide risk management tools, a pilot approach was proposed, which explores worst-case ecological hazard scenarios at different space-time scales by means of a set of five eco-toxicological risk indexes. The results were then interpreted from the perspective of a decision support method using the Critical Threshold Value approach. The risk analysis was then enriched within a multicriteria framework which integrates environmental, agronomic, and economic objectives.

10.3. Policy implications

Recent changes in EU legislation mandating some type of environmental appraisal of new policies[98] have reinforced the need for sound scientific insights on the potential environmental effects of European sectoral policies (transport, agricultural, etc.). Interest has arisen, especially in government, in investigating the extent to which the expected benefits of policy proposals can also be accompanied with negative-side effects on local communities and the environment. The problem becomes one of assessing the preferred policy option to manage environmental risks and impacts in the most cost-effective way. It is also incumbent on the researcher to ensure that existing research insights are not misinterpreted, or used inappropriately, by naïve policy and decision makers with little knowledge of the issue involved in the valuation. The following discussion, therefore, presents lessons and policy implications that can be drawn from each of the empirical studies presented above.

From a policy perspective, the results of our study on rail noise (Chapter 4) underline the relevance of investing in actions for the reduction of noise emission levels in urbanised areas. While in the past, railway noise has been reduced in Europe[99], in fact, the technological improvement was not primarily planned as a

[98] For instance, the Commission's new assessment process, Impact Assessment, "is intended to integrate, reinforce, streamline and replace all the existing separate impact assessment mechanisms for Commission proposals". Communication on Impact Assessment, 5 June 2002, COM(2002)/276.

[99] The equipping of most new coaches with disc brakes instead of iron block brakes has led to a significant reduction of noise generation. The replacement of jointed track with continuously welded

noise reduction measure, but was adopted because of other operational requirements. Disk brakes had to be used on modern coaches to allow speeds above 140 km/h. This was not, however, required for freight wagons, which is why noise generation from this type of stock is now the predominant railway noise issue in Europe, particularly concerning operations at night. The lack of technical progress in railroad tracks and wagons, therefore, does represent an impediment in tackling noise reduction. In addition, current transport plans in Europe foresee high speed trains running at speeds of up to 350 km/h to form a Trans-European high-speed railway network. Noise from high-speed lines, mostly operating during the day time, is the second main noise issue. It often arises at the planning stage of new high speed lines or services, when noise mitigation becomes a key requirement. Noise from high speed trains has different characteristics from that of freight wagons. With increasing speed, aerodynamic noise from the upper part of the trains becomes dominant and most of the existing noise barriers are too low to shield this source. It follows that the height of noise barriers also matters. The third noise issue concerns urban rail transport. Trams and urban light systems mainly operate in densely populated areas, where rail noise annoyance has become a subject of social relevance. Here, noise reduction measures have a high profile. Nevertheless, the benefits from annoyance nuisance reduction can be hampered by aesthetic or micro-climatic disutilities that are often associated with noise barriers. In this connection, the social acceptance of different noise abatement strategies can play a role too. The results from our SP study case suggest, in fact, that, depending on acoustic efficiency, aesthetics, and level of technological innovation, people affected by excessive noise levels associate lower or higher utilities to alternative noise abatement plans. For instance, a noise reduction achieved with an additional increase in the height of the barrier is priced at a lower level than equal noise abatement achieved by technological innovations on trains and tracks. Noise barriers are associated with a strong aesthetic disutility, even if the policy maker offers it disguised with ornamental vegetation. This signals that investments in train or rail technology would be highly appreciated. Hence, even if national authorities charged with planning and implementing noise actions can operate discretionally, our study suggests that research advice on the best strategy that can guarantee higher benefits for local communities is relevant.

The process of railway reform that started with the EU Directive 91/440 is also characterised by changes in responsibilities or functions. Although there are different institutional settings in the various EU Member States[100], it is commonplace to have a variety of different entities: operators, rolling stock owners, maintenance companies for rolling stock or infrastructure, infrastructure managers and manufacturing industry. This splitting results in several formally separated parties responsible for railway noise abatement, which makes it even more important to create links for a common cost-effective strategy. For instance, in Italy, the recent splitting of responsibilities between rolling stock owners and infrastructure managers has created conflicts on the most preferred action to abate rail noise. Doubts have also arisen about which party should pay and finance noise reduction measures, which is why there have been continual postponements of interventions against noise. Irrespective of legal arguments, our empirical analysis

rail across much of the European network has also led to significant local reductions in noise creation (CEC, 2003, p. 19).

[100] These include integrated companies with a split of functions or separated companies.

shows that people affected by noise are sensitive to the type of project financing adopted. The overall welfare gain from, as well as the public acceptance of, rail noise reduction measures, might therefore be affected by the type of project financing adopted.

As far as urban sprawl is concerned, the empirical results of the analyses presented in Chapter 5 support the pessimistic assessment of the phenomenon of sprawl that has been embraced by the European Commission. As a matter of fact, the favoured vision of high density, mixed-use settlements with no brownfields and empty property, and planned expansions of urban areas rather than ad hoc urban sprawl, has been reinforced in each successive EC policy document on urban development starting from the "Green Paper on the Urban Environment" (CEC, 1999). Since then, a number of community initiatives, based on urban land regulation, have been implemented to reduce sprawl, supporting urban infilling and densification[101]. Because land regulation offers more direct control, it is more appealing than other approaches that propose to reduce sprawl by attacking the root cause – market failure – through changes in taxes and fees, especially on land development (e.g. Breuckner, 2000). Through land use regulation local communities can identify areas appropriate for development and areas that are not, rather than trying to shape consumer preferences concerning alternative types of urban settlements. Nevertheless, these measures can be difficult to implement politically, especially in areas with well-organised property rights interests or weak long-term economic growth prospects (Frank et al., 2000). Therefore, one additional approach to combat sprawl may possibly be to focus on the negative consequences of sprawl, without necessarily addressing the underlying causes of sprawl or regulating land use per se. For example, purchase of development rights for open space, tighter fuel economy standards for cars, and increased investments in workforce and business development in the city centre might contribute to solve many of the most critical problems associated with sprawl.

As far as the transport impacts of sprawl are concerned, a number of urban design philosophies – new urbanism, transit-oriented development, traditional town planning – have gained popularity in recent years as ways of shaping the travel demand (e.g. see Camagni and Gibelli, 1997). All share three common mobility aims: i) reduce the number of motorised trips; ii) increase the share of trips that are non-motorised; iii) reduce travel distances of motorised trips, and increase vehicle occupancy levels. An expected result of weaning people from their cars would be a lessening of the negative consequences of an automobile-oriented society: namely, air pollution, fossil fuel consumption, and class and social segregation. Europeans, though, seem to be ever-increasingly attracted by automobiles, because of the benefits that cars provide in terms of personal comfort and freedom of travel.

Our empirical study on Italy shows, for instance, that individual preferences for alternative transport modes have progressively changed towards

[101] Among the others, the European initiatives URBAN II and INTERREG support mixed use and environmentally-friendly brownfield redevelopment, involving reduced pressures on greenfield site development and urban sprawl. In addition, the Community supports different research projects related to the revitalisation of city centres and neighbourhoods, the restoration and reuse of contaminated and brownfield sites, the sustainable rehabilitation of urban areas such as large housing estates, and strategies to reduce urban sprawl based on the integration of land use and transport planning.

private motorised travel means. The role of modal choices in affecting the intensity of the collective impacts of mobility appears, therefore, pivotal. Nevertheless, as private behaviour is involved, it is unlikely that this issue will be politically addressed in the short run, if alternative transport modes, as appealing as automobiles, are not fed into the mobility market.

To conclude, given the evidence that the costs of sprawl probably do exceed its benefits, public policy ought to pursue at least one of the sets of strategy discussed above. It is premature, however, to recommend one strategy over another, and local communities will probably need, at least in the near future, to find their own way.

Related to pesticide risk, the empirical studies presented (Chapters 6 to 9) show the relevance of the economic costs of pesticide use, and support the perspective of some commentators (e.g. Swanson and Vighi, 1998) that optimal strategies against pesticide risk should be capable of taking into consideration the variety of local conditions that contribute to the rate at which chemicals will affect different environmental targets. In fact, the comparative and statistical reviews of the literature presented in Chapters 6 and 8, respectively, as well as the choice experiment (CE) approach in Chapter 7, show that the cost of pesticide risk can vary substantially according to the type of risk concerned. The value of reducing pesticide risk appears to change according to: i) the risk target (so that effects on farmers, consumers, and the aquatic and the terrestrial ecosystems should be distinguished); ii) the type of safety-enhancing device (integrated pest management, eco-labelling, or a ban on specific pesticides); and iii) the geographical location of the respondents. These results signal that market instruments to manage pesticide risk (taxes, incentives) should take into account such differences. Similarly, these suggest that alternative safety devices could be implemented at different cost-effectiveness ratios.

The above-mentioned factors are important and typical features of environmental decision making and are central to the debate on the most-preferred type of pesticide policy in Italy and Europe. The question is, therefore, which instrument can best be used that takes into account target differences and location-specific criteria. This poses some serious questions with respect to the environmental policy within Europe. Taking into account the economic criteria for standard-setting, one could argue that location-specific criteria should be taken into consideration, and that, therefore, environmental standards should not be the same throughout the whole EU[102]. In conformity with this perspective, for instance, Faure and Lefevere (1998) suggest that the most efficient way of regulating the use of pesticide is by setting rules on two different levels. First of all, quality standards on receptors (i.e. aquatic and terrestrial ecosystems) should be set at the European level. These should be enforceable upon Member States by both the Commission and individuals who suffer damage because Member States do not meet EU standards. Secondly, Member States might more efficiently reach these goals by using a system of zoning of regions where pesticides are frequently used. In each type of zone, rules should be set that ensure the correct use of pesticide with regard to the characteristics of the area concerned (e.g. rules relating to the type of

[102] For instance, efficient standards might be relatively high in the industrial areas of the EU, with a high population density and a consequent heavy load of environmental pollution, but might be less stringent in situations where the cleansing capacity of the ecosystem can still absorb a certain amount of pollution.

chemical applied, level of the chemical's accumulation in different ecosystems, potential environmental targets, vulnerability of targets, etc.). In addition, zoning rules should be enforced upon the farmers by the national, regional or local authorities. However, as far as pesticide taxes or incentives are concerned, a location-specific approach – as applied to pesticide producers or farmers – would imply unequal market conditions since operators would be charged at different prices in various European regions. Instead, as already argued, taxes on, and incentives for, producers and farmers might be set, efficiently, to take into account differences in risk targets, risk intensity, and risk persistency.

10.4. Suggestions for future research

Given the importance of increasing our understanding of the causes and the effects of environmental decay, many issues are worth further investigation. One point, which is more of a general consideration than a criticism, is that further efforts to propose a new framework for evaluation seem warranted. In fact, if from a research perspective, the economic valuation of externalities appears to be, overall, theoretically and methodologically well-established, from a policy perspective, valuation exercises can still improve in their applicability and usefulness (Pearce and Seccombe-Hett, 2000). In particular, the data collection issue is crucial, as it often leads to costly and time-consuming valuation exercises. In addition, the difficulty in interpreting the results – by a public of non-experts – can discourage policy makers from using them.

As far as the issue of data collection is concerned, meta-analysis may be a cost-effective and expeditious way of economising on research efforts by focusing on the main determinants of a phenomenon, seen from a comparative approach. The same applies to value and benefit transfer studies. Clearly, there is need for some caution. The moderator variables have to be carefully investigated and proper care regarding study and context-specificity is required (Florax at al., 2002). Further evidence is also needed with respect to the validity and reliability of value transfer. To date, the analysis of the performance of value transfer has mainly focused on the comparison of mean values. As the body of knowledge increases, the comparison may be cast in a statistical setting (Florax, et al., 2002).

In this connection, a number of issues arise from our study. While we have provided the first statistical meta-analysis in the field of pesticide risk valuation (Chapter 8), to date a value transfer has not been performed for our pesticide model. The transfer of values developed in a non-market setting is difficult in itself. Moreover, the great variety of pesticide risks concerned and methods applied in the literature suggests that more primary research is needed. In particular, it is important that future valuation effort carefully specifies both the baseline level of risk and the change in the risk level. More attention is also necessary for the income-specific and potentially location-specific nature of the valuation of reductions in pesticide risk exposure.

Equally, while the added value of meta-analysis for the valuation of noise had previously been illustrated by meta-analytical studies on the valuation of aircraft noise (Schipper et al., 2001; Nelson, 2004), and on the valuation of traffic noise (Bertrand, 1997), so far such a meta-analytical attempt has not been performed for rail noise. The few valuation studies on rail noise, in fact, still refrain from attempts in this area. Thus, a new field in noise valuation that is likely to play an interesting role is rail noise valuation. In this dissertation we have provided the first estimates – based on a stated-choice approach – of the value of reducing rail noise with alternative types of measures (Chapter 4). We have attempted to address this issue by including attributes that control for some relevant features of alternative noise interventions (e.g. technological innovation, aesthetics, type of financing), but further efforts are warranted to improve the body of knowledge in this field. Above all, it is necessary that future research more carefully uses an annoyance-level-based unit of value, and controls for the respondents' profile. In addition, there is now evidence concerning the negative health impact due to noise, and this represents about 10 percent of the cost of annoyance, as estimated by hedonic price models (Droste-Franke et al., 2006). Of course, there might be problems of double counting but the real health impact of noise will deserve more attention in the future. Alternative payment vehicles, such as the tax-reallocation scheme proposed by Bergstrom et al. (2004), and applied in our rail noise survey, should also be further tested.

Finally, another area where meta-analysis can usefully be developed is in a review of the alleged impacts due to sprawl. The literature provides mixed results of the various costs of sprawl (Frank et al., 2000), and it would therefore benefit from a rigorous statistical review. Future research on the costs and benefits of sprawl needs to recognize the potential for variation between metropolitan areas related to differences in transportation costs (congestion and public transport service) and city-suburbs' wage differential. In addition, more research is needed on the most cost-effective measure for preventing the environmental damage associated with sprawl and private mobility, and on the political feasibility of anti-sprawl policies compared with expanded environmental regulation. Research is also underway to attempt to measure whether laissez-faire land development policies produce a more efficient city form than policies intended to promote a more compact city (Persky and Wiewel, 2000). Similarly, more attention should be given to the analysis of the most preferable transport system in cities. In this respect, the microeconomic base of mobility should be further discussed. Why people keep on using cars to move in cities notwithstanding the serious collective costs that this habit produces in terms of risk to human health and quality of living? The private benefits provided by private transport, in terms of comfort and "emancipation" from the public transport service, should be quantified and compared against its collective and private cost.

As far as the issue of providing user-friendly results is concerned, the use of environmental impact/risk indexes is promising. Nevertheless, this approach can not be a substitute for conventional valuation methods: neither when it is necessary to monetise the environmental impacts concerned, nor when ad hoc risk assessment procedures should be implemented to estimate physical risks. By combining ecotoxicological indexes – while examining other contextual factors – within a multicriteria framework of analysis, we have defined and compared alternative future agricultural risk scenarios at various time and spatial scales (Chapter 9). As long as risk indexes are based on sound scientific information, this appears to be a new area where these tools are likely to play an interesting role in order to support valuation. As indexes are usually measured with simple ordinal or cardinal scales, they are easily interpretable by policy makers; moreover, they can integrate a wide wealth of relevant information. In addition, such characteristic features make indexes particularly useful to simulate future risk and socio-economic scenarios, which might be easily corroborated with sensitivity analysis and Montecarlo simulations.

Ultimately, participatory assessment processes are gradually becoming common practice in normal environmental decision making, and this may reinforce the need for preference-based analysis, with the direct involvement of stakeholders. Thus, there may be new perspectives for environmental valuation if it is linked systematically to the ever-increasing processes of environmental decision making that entail stakeholders' participation (e.g. Agenda 21, Strategic Environmental Assessment). As such, in fact, stated preference techniques and (often) multicriteria analysis require a direct questionnaire approach that allows people to state their preferences for alternative environmental scenarios. These tools might be integrated into forums to provide prior sources of information concerning people's preferences for or against environmental changes, or to test valuation questionnaires. This would facilitate the derivation of monetary values for the proposed changes, and ensure the public acceptance of the ultimate decision.

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SUMMARY

Valuing Environmental Decay. Analytical Policy-oriented Studies on Urban and Rural Environments

The environment has become exposed to a range of tresses from a wide variety of economic activities. Increasingly, we are faced with choices of trading some risk and cost to humans or ecosystems in order to reinforce economic development. A number of significant and still unsolved illustrations of such a perverse mechanism can be found in cities and rural systems, where the numerous social and economic advantages generated by, respectively, urban agglomeration and proximities, and by agricultural production and rural development, are accompanied by collective diseconomies and multidimensional environmental negative externalities.

Interest has therefore arisen, especially in government, in investigating the extent to which the expected benefits of policy proposals can also be accompanied with negative-side effects on local communities and the environment. The problem becomes one of assessing the preferred policy option to manage environmental risks and impacts in the most cost-effective way. It is also incumbent on the researcher to ensure that existing research insights are not misinterpreted, or used inappropriately, by naïve policy and decision makers with little knowledge of the issue involved in the valuation.

By filling the gaps in research, the studies described in this thesis focus on a number of open policy issues in the context of urban and rural environments. With a retrospective look at the process of economic valuation and its integration within environmental decision making, this thesis brings us from the theory to the empirics of valuation, in order to provide methodological innovation based on experimentation. Firstly, the thesis refers to the wide-ranging impacts caused by urban mobility, and focuses on two challenging open issues. On one side, it introduces the problem of noise pollution due to rail transport infrastructure; while, on the other side, it looks at the phenomenon of urban "sprawl" and at its implications in terms of collective impacts. Next, it presents the research challenges that valuation can address when looking at the impacts and risks of agriculture on ecosystems and human health. In this connection, it identifies pesticide risk valuation as a relevant research area.

The research challenges addressed in this dissertation emerge from the policy arena and are intended to provide support to relevant policy issues such as: Which urban and agriculture-induced risks should be prioritised? Which management or mitigation options should be chosen to respond to these risks? And how far it is necessary to go in, for instance, reducing private car or pesticide use? Is action preferable to the do-nothing scenario? The methodological contribution of this book is based on the paradigm of cost-benefit analysis, which states that, in a

world of scarce resources, rational action requires a consideration of relative benefits and societal costs; and that governments need to take some account of public preferences in their decision making. However, researchers face big challenges in evaluating possible responses to the impacts of urban and agricultural development. These include the long timescale of impacts, serious uncertainty over environmental change and human reaction to this change, uncertainty over the effectiveness of management strategies, and the very wide range of impacts that the continuous process of economic development may have.

The studies presented show how cost-benefit thinking can be used to enhance decision making with respect to risks due to mobility and agriculture, as well as pointing to the contribution that other methods, such as multicriteria analysis and the use of ecological and environmental indicators can make. A characteristic feature of the thesis is the presentation of worked examples of applying a number of well-framed methodologies (stated preference methods, meta-analysis, multicriteria analysis, risk assessment) to issues as diverse as noise pollution, transport disruptions, pesticide ecological risks, and food safety. Empirical case studies refer to Italy where, given a relative delay in the use of environmental valuation approaches for costing ecosystems and human health risks, the interest in economic valuation is rapidly increasing.

SAMENVATTING

Analyse van de verslechtering van het milieu. Kwantitatief beleidsmatig onderzoek naar de stedelijke en plattelandsomgeving

Het milieu wordt blootgesteld aan een reeks van negatieve invloeden veroorzaakt door velerlei economische activiteiten. Als het gaat om economische vooruitgang worden wij in toenemende mate geconfronteerd met keuzes tussen het nemen van bepaalde risico's en de gevolgen die ze kunnen hebben voor mensen of ecosystemen. Een aantal belangrijke, maar vooralsnog onopgeloste voorbeelden van een dergelijk verstoord mechanisme is te vinden in steden en op het platteland, waar veel maatschappelijke en economische voordelen van respectievelijk stedelijke agglomeraties en nabijheid van voorzieningen, en van de agrarische productie en plattelandsontwikkeling, die worden vergezeld door collectieve economische nadelen en multi-dimensionale negatieve milieu-invloeden.

Juist daarom is er veel belangstelling ontstaan, vooral bij de overheid, om te onderzoeken in hoeverre de verwachte voordelen van de beleidsvoorstellen, negatieve neveneffecten op de lokale gemeenschap en op het milieu zouden kunnen hebben. De uitdaging begint bij het beoordelen van het voorkeursbeleid hoe er op de voordeligste manier moet worden omgegaan met risico's van en invloeden op het milieu. Voor een wetenschapper is het van groot belang zich ervan te verzekeren dat de bestaande wetenschappelijke inzichten niet verkeerd worden begrepen of dat deze niet verkeerd zullen worden gebruikt door de naïeve politici en beleidsmakers met weinig kennis van zaken.

Door de open plekken in het onderzoek op te vullen, focust de studie, zoals deze wordt beschreven in deze scriptie, zich op een aantal open beleidstukken in de context van steden en platteland. Door een retrospectieve kijk naar het proces van economische evaluatie en diens integratie in het kader van het milieubeleid, brengt deze scriptie ons van de theorie naar de empirische evaluatie, met het aantonen van de methodologische innovatie gebaseerd op het experimenteren als doel.

Ten eerste richt deze scriptie zich op wijdverspreide invloeden, veroorzaakt door de stedelijke mobiliteit en het focust zich op twee uitdagende open kwesties. Aan de ene kant wordt het probleem van geluidsoverlast geïntroduceerd, dat door de infrastructuur van het vervoer over het spoor veroorzaakt wordt; aan de andere kant, richt het zich op het verschijnsel van de stedelijke "uitdijing" en op de gevolgen van de collectieve impact. Vervolgens komen de wetenschappelijke uitdagingen aan bod, die de evaluatie aan de orde stelt, als men kijkt naar de gevolgen en risico's van landbouw op de ecosystemen en menselijke gezondheid. In dit verband wordt de evaluatie van het risico van pesticiden geïdentificeerd als een relevant onderzoeksgebied.

De wetenschappelijke uitdagingen, die in deze scriptie zijn opgenomen, komen voort uit de politieke arena en zijn bedoeld om ondersteuning te bieden aan de relevante politieke onderwerpen, zoals: Welke risico's, afkomstig van de steden en het platteland, dienen als prioriteit te worden gesteld? Welke managementstrategie en verzachtende omstandigheden moeten worden gekozen om op deze risico's te reageren? En hoever moet je gaan om bijvoorbeeld het autogebruik of pesticidengebruik te verminderen? Verdient actie de voorkeur boven niets doen? De methodologische bijdrage van dit boek is gebaseerd op het paradigma van de kosten-batenanalyse, die zegt dat in de wereld van schaarse middelen, rationele handelingen een overweging vereisen van relatieve baten en maatschappelijke kosten; en dat overheden rekening moeten houden met de publieke voorkeuren in hun besluitvorming. Desalniettemin lopen onderzoekers tegen grote uitdagingen aan, als het gaat om het evalueren van de mogelijke respons op de gevolgen van stads- en plattelandsontwikkeling. Hier zijn bij inbegrepen: de gevolgen op lange termijn, grote onzekerheid omtrent milieuveranderingen en menselijke reactie op deze veranderingen, onzekerheid over de effectiviteit van de management strategieën en een heel breed scala aan gevolgen dat het voortdurende proces van economische ontwikkeling teweeg kan brengen.

Deze studie toont aan hoe kosten-batendenken kan worden gebruikt ter ondersteuning van beleidsvorming, als het gaat om de risico's die mobiliteit en landbouw met zich meebrengen, alsook wijst het op de bijdrage van andere methoden, zoals multi-criteria analyse en het gebruik van de ecologische en milieu indicaties. Een karakteristiek kenmerk van deze scriptie vormt een presentatie van bewezen voorbeelden die gebruikt kunnen worden bij een aantal goed omschreven methodieken (stated preference methods, meta-analyse, multi-criteria analyse, risicobeoordeling) ten opzichte van kwesties zoals geluidsoverlast, transport onderbrekingen, ecologische risico's ten gevolge van pesticidengebruik en gezonde voeding. Empirische case studies hebben betrekking op Italië waar, gezien een relatieve achterstand in het gebruik van milieu evaluaties omtrent kosten voor ecosystemen en risico's voor de menselijke gezondheid, de belangstelling voor de economische evaluatie in hoog tempo toeneemt.