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Pesticide Risk Valuation in Empirical Economics

Peter Nijkamp[1,2]
Chiara Maria Travisi[1]
Gabriella Vindigni[3]

[1] *Faculty of Economics and Business Administration, Vrije Universiteit Amsterdam,* [2] *Tinbergen Institute,* [3] *University of Catania.*

Tinbergen Institute
The Tinbergen Institute is the institute for economic research of the Erasmus Universiteit Rotterdam, Universiteit van Amsterdam and Vrije Universiteit Amsterdam.

Tinbergen Institute Amsterdam
Keizersgracht 482
1017 EG Amsterdam
The Netherlands
Tel.: +31.(0)20.5513500
Fax: +31.(0)20.5513555

Tinbergen Institute Rotterdam
Burg. Oudlaan 50
3062 PA Rotterdam
The Netherlands
Tel.: +31.(0)10.4088900
Fax: +31.(0)10.4089031

Pesticide risk valuation in empirical economics: a comparative approach

Peter Nijkamp [☼], Chiara Maria Travisi [☼] and Gabriella Vindigni [Θ]

(☼) Free University
Department of Spatial Economics, De Boelelaan 1105
1081 HV Amsterdam, The Netherlands
Phone: +31-20-44-46090
Fax: +31-02-44-46004
E-mail: ctravisi@feweb.vu.nl, chiara.travisi@unimi.it

(Θ) University of Catania
Via Santa Sofia, 98, 95123 - Catania, Italy
Phone: +39-095-7580333
Fax: +39-095-7580345
E-mail: vindigni@unict.it, gvindigni@yahoo.com

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ABSTRACT

Pesticide use in agriculture poses several risks to both human health and non-target agro-ecosystems. Due to lack of information on the monetary value of reducing pesticide risks, it is difficult to perform an economic analysis that addresses social efficiency of policy and draws conclusions about the appropriate degree of regulation.

The aim of the current paper is to present a critical overview of the empirical literature on pesticide risk valuation that provides disaggregate willingness-to-pay estimates (WTPs) of pesticide risks reduction. Recent multidimensional classification methods, such as coined decision tree analysis, are used in a comparative approach as tools for explaining the differences in empirical research findings. The analysis shows that the magnitude of WTPs is related to both the valuation technique and to the data available from biomedical and eco-toxicological literature. It also shows that WTP 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.

Keywords : pesticide environmental and health risks, willingness-to-pay, comparative analysis
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1. INTRODUCTION

Widespread use of chemicals [1] in modern agriculture has led to increasing awareness of the risks that such products might pose, both to human health and to the environment. Since the 1950s, chemical-based strategies have been the preferred form of pest control in agricultural production, and it has been estimated that, in their absence, up to a third of crop production would be lost (Pimentel, 1978, 1991, 1997). On the other hand, starting at the end of the 1970s, the on-farm benefits of pesticide use have been weighed against concerns over the off-farm costs of pesticide risks to the environment and to human health. This wider perspective has prompted many countries and regulatory agencies, both at national and at international levels, to implement a variety of policies. In their search for an effective management formula, they have employed control strategies 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 (Smith *at al*., 1998).

Setting aside the issue of whether modern agriculture might do without chemical inputs altogether, two major facets of the issue cause much difficulty. On one side, due insight into intricate *cause-effect relationships* are necessary to model the phenomenon and to predict its temporal and spatial dynamics; on the other, the *multidimensionality* attribute of pesticide risk determines the complexity of the consequent trade-offs among conflicting priorities. First, the potential risks posed by agricultural chemicals are context-specific and vary fundamentally depending upon the substance being considered, its physical/chemical properties, the way it is being used and the level of exposure. Chemical and source characteristics can vary tremendously and they directly determine the type of exposure as well as the target potentially at risk ('stock at risk') and the toxicological and eco-toxicological effects. Besides, risks may occur as a result of exposure to a single compound or to a 'cocktail' of agrochemicals (cumulative risk), or may differ in their temporal and spatial dimensions (short-term vs. long-term risk; acute vs. chronic risk). Moreover, the presence of ongoing disturbing factors, such as habitat loss, increased nutrient loads and contamination by other xenobiotics, makes it even harder to relate observed environmental effects to exposure to pesticides (see STOA, 1998). This, in synthesis, explains the first source of complexity and represents an underlying premise of the second one. It follows that since agrochemicals act -and affect the environment- in different ways, selecting among alternative pesticides implies trade-offs between different types of potential risks. This means trade-offs between the 'stocks at risk' to be protected, and trade-offs between differentials in the temporal and spatial dimensions of the hazard to be managed. For example, some products may be environmentally friendly in terms of their impact on the aquatic ecosystem, but may simultaneously damage

[1] Within the European Union, roughly 2,600 chemicals have been categorised as "high production volume" chemicals (over 1,000 tonnes produced per year), and between 15,000 and 20,000 are being produced in volumes between 10 and 1,000 tonnes per year. These chemicals include products that are essential in sectors such as agriculture, health care, home and personal care, manufacturing, education, recreation and many other aspects of everyday life (OECD, 2000).

terrestrial biodiversity. A class of products may have a tolerable level of toxicity for non-target organisms but be extremely long lasting and mobile in the environment, spreading from the emission zone to other previously uncontaminated areas. Some pesticides may be safe for farmers' health but dangerous for consumers' health since they accumulate as residues in fresh agricultural products. In an attempt to provide suitable answers to these difficult questions, toxicology and eco-toxicology have developed predictive approaches since the 1970s (Suter, 1993; Vighi and Bacci, 1998; Edulijee, 2000; Solomon, 1996; Power and Adams, 1997; McCarty and Power, 1997, 2000). In this field, therefore, any regulatory decision should take widely varying valuations into account simultaneously.

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 to be made. Both arguments, within the context of determining chemical control strategies, require, at least implicitly, that regulation be based on a precautionary stance, and that a balance be struck between the costs of reducing the risks and the benefits stemming from risk reduction. Actually, recent trends in chemical risk management are evolving in tune with these ideas. Preference is given to *a priori* precautionary approaches, which encourage action despite a lack of complete understanding of the risk phenomena. Moreover, many countries require that new legislation and/or administrative legislation on chemical risks be integrated with socio-economic analysis (SEA).

It is clear that a radical interpretation of the first stance might lead to the conclusion that hazardous chemicals and activities are considered *tout court* unacceptable because of the uncertain nature of the consequences of their use. Nevertheless, in practice, less radical interpretations are used, either stressing the cost of adopting precautionary actions or utilising a 'safe minimum standard' approach[2]. A more subtle question is whether it makes sense to adopt an *ex-ante* approach to risk management, and what this would involve in circumstances where, as for pesticides, risks are thought to be relatively low. Particularly, pesticide risks are expected to be low when compared to their management costs, which, in contrast, are typically high. Therefore, decisions regarding agrochemicals tend to involve major trade-offs (Lichtenberg, 1991). This, along with the previous argument, explains why regulatory actions increasingly call for some type of quantitative risk-benefit procedure.

To return to our main line of reasoning, once the need for an *a priori* attitude has been defended, we must still determine how to make the pesticide risks/benefits trade-off explicit and directly comparable. Or, from a broader perspective, we must provide information on the level of environmental protection that is socially desirable, the level of human health risk that is socially acceptable and the expected level of potentially excessive cost in terms of both private and public expenditure. In many countries, the call for a formal

[2] A prominent example of radical interpretation of the precautionary principle is the case of the GMOs regulation in the EU (Directive 90/219/EEC; Dir. 90/220/EEC; Dir. 91/414/EEC).

appraisal of chemical policy costs and effectiveness, using one of the established procedures of cost-benefit or cost-effectiveness analysis, risk assessment or multi-criteria assessment, is one of the ongoing responses to this issue. The premise underlying such action stems from the argument that, when making a decision concerning the management of hazardous chemicals, managers need to know if some actions and measures are required or desirable and what the best management strategy would be. To effectively answer these questions, an economic valuation of the changes in chemical risks stemming from the policy's implementation is desirable. If these quantities can be measured, changes in environmental and human risks can be compared on the same basis as the financial costs and benefits of any other project or policy (Pearce and Secconbe-Hett, 2000).

In this sense, the EU is far behind the USA, because in the EU it has only been since the early 1990s that formal appraisal procedures have been improved and more widely applied. Emblematic of the ten-year lag is the irregular and generally weak use of economic valuation within European decision-making processes, where ideological and practical concerns still inhibit its use. Nevertheless, it is fair to note that economic valuation has recently enjoyed a revival in the EU as a consequence of some new legal developments within the European Commission (Pearce, 1998; Pearce and Secconbe-Hett., 2000; Matheus and Lave, 2000).

In terms of chemical and, more specifically, pesticide risk valuation, the European scenario is contradictory. Chemical risk assessment has had a formal status since 20 July 1993, with the creation of the Regulation 93/67/CEE, which clearly contains some of the basic elements of overall risk assessment, but there is still no explicit mention of the cost or of the 'stock at risk'. The consequence of this formulation is that cost-effectiveness cannot be applied (Pearce and Secconbe-Hett., 2000).

On the other hand, a trend towards the increasing use of economic valuation for the design of pesticide-taxes and eco-labelled products is visible; this is becoming one of the major topics in the agricultural policy agendas of some European member countries (ECOTEC, 1999; Dubgaard, 1987). This is also reflected in an emerging interest within the empirical literature in the valuation of changes in pesticide risks to both the environment and human health (Mourato and Foster, 2000; Schmitz and Ko, 2001-a, Schmitz and Brockmeier 2001-b; Schou *et al*., 2002; Söderqvist, 1998; Press and Söderqvist, 1998; Falconer and Hodge, 2001; Archer and Shogren, 2001).

In light of the increasing concern of the EU Commission about pesticides risk valuation and the narrowness of empirical economic experience in a European context, this paper sets out to provide a critical overview of the empirical literature dealing with this emerging issue throughout the world. In this framework, comparative analysis may play an important role as a tool for explaining the differences in empirical research findings (previous applications on the economics use can be found *inter alia* Nijkamp and Pepping, 1996; and van den Bergh *et al*., 1997).

2. THE VALUE OF CHANGES IN ENVIRONMENTAL RISKS FROM AGROCHEMICALS

2.1 Chemical risk assessment and valuation

An overall economic valuation of the changes in risks related to agrochemical use requires, at least in principle, assessment of both potential human health and environmental hazards. Although the former has become a relatively conventional issue for economists, the latter still presents some elements of complexity, which can be traced back to the human-driven rather than environmentally-driven historical background of chemical risk management.

This unbalanced state has been - and to some extent still is – reflected in both the scientific and the valuation literature. For instance, there appears to have been greater emphasis on developing criteria or benchmarks for examining human safety issues. In contrast, few benchmarks (other than the use of ratios comparing predicted concentration against predicted no-effect concentration) have been established for environmental risks. This is in part due to the wide range of endpoints[3] that require consideration, and also because site-specific considerations and the professional judgement of assessors are usually relied upon more heavily (NRC, 1996). Moreover, if the monetary measurement of changes in human health has a strong theoretical basis, the valuation of changes in ecological risk has not been similarly explored theoretically. In the following discussion, considering human and environmental risks in parallel, we will try to point out some controversial issues stemming from this premise.

The first question is which human and environmental risks should be considered in the valuation, and whether economists can systematically approach such problems by relying on some standardised classification of agrochemical hazards.

In terms of human health, it is straightforward to say that analysts should consider both changes in fatality risks and changes in illness risks; the latter can further be sub-divided into acute effects and the incidence of chronic diseases. Main risk groups will then be considered, typically sub-divided into occupational and non-occupational ones (Table 1 refers to the EU context).

In contrast, ecological risks are more difficult to define as a consequence of the great diversity of species, the multiple levels of biological organisation (individual, population, community, ecosystem), the huge range of interrelationships among organisms, and the number of criteria and endpoints that might be relevant. The purpose of classification is to simplify these complexities, and chemicals’ impacts are usually studied on the basis of the environmental compartments in which xenobiotics might accumulate. Environmental risks are classified according to both the risk groups and the medium of exposure (see Table 2 on the EU approach).

[3] An (eco)toxicological *endpoint*, for a vegetal 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).

It is clear that, in order to avoid the pointless use of resources, analysts should primarily focus on the valuation of those risks that actually have relevant and detectable effects on territory. Dealing with pesticides, a recent document by the STOA (1998) identifies four main risk groups or protection targets in the European region to be considered in a risk-based approach. These are agricultural production, environment and biodiversity, water resources, and occupational and consumer human health. As confirmed by several chemical analyses in EU Member States, agrochemicals are detectable in various environmental compartments (see, among others: SSLRC, 1997; Campbell *et al*., 1997). Whenever agrochemical concentration occurs at a biologically significant level, aquatic and terrestrial ecosystems are damaged and, indirectly, agricultural production and the human population in the contaminated areas are also affected. However, the concept of *risk* should not be confused with the concept of *hazard* (or *potential risk*): the latter represents the potential for a risk scenario, and the former an actual one. This means that there is a potential risk if predictions show that pesticide concentrations in the environment exceed certain environmental quality criteria or threshold values. Yet, the actual risk is assessed by comparing hazard information with information about the environmental system (individual, population, community) actually exposed to the pesticide[4] (see Figure 1).

Inclusive approaches to pesticide risks valuation should not neglect public concerns and perceptions about pesticides' effects on human health and the environment. It is relevant to remark upon how significantly lay people's and experts' perceptions of risks can differ and how crucial balancing popular and expert opinions can be within the risk management process (see also Edulijee, 2000). In the case of agrochemical risks, 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). Here, the discrepancy between public and expert concerns lies in human health impacts. Lay people are most troubled by pesticide residue ingestion via fresh food, whereas experts predict that food risks will be low, and exposure at work more significant. Moreover, people usually fail to consider the on-farm adverse effects of pesticide usage, though in actuality they represent a concrete problem. Their effects include direct crop damage, the development of resistance towards pesticides in the target organisms, and adverse impacts on beneficial insects, etc.

The second, more controversial, question 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,

[4] This comes from the intuitive and well-known argument that either there cannot be hazards without exposure or there cannot be risks without an environmental system being exposed to the hazard (see Figure 1). The difference between the concepts of hazard and risk seems to be subtle but it is rather substantial and it has to be kept in consideration when dealing with risk assessments' results.

the manner in which risk changes can be valued depends upon the information available from the risk assessment.

As noted above, the biomedical and toxicological literature on the effects of human exposure to agrochemicals provides a solid background as a consequence of the human-driven perspective of most major chemical risk policies. As a result, information from human risk assessments on whether or not a risk is considered 'unacceptable' is usually available in several countries [5]. However, comparing one set of criteria with another is often a complex task and some preliminary remarks are in order when approaching a risk valuation exercise.

First, a distinction has to be maintained between *societal* and *individual* risks. The concept of risk to society is often employed when considering the potential for incidents associated with hazardous activities or sites, which might result in large numbers of fatalities. In contrast, when considering the use of chemicals in 'everyday life', attention is focused on the level of individual risk. For agrochemicals the focus is usually on individual risks, although risk assessment results can also be presented in the form of *collective* risks simply by considering the size of the target population.

In addition, one must consider that levels of acceptable/unacceptable risk often vary by type of risk and by country, and they can differ in terms of unit of measure. For example, when concerning risks to individuals, estimated risk values might be expressed either as chances per year or chances per lifetime (the latter being particularly applicable to carcinogenity risk endpoints, where the concern is for lifetime exposure and effects).

In conclusion, it is important to note that the output of the risk assessment can differ, either in the level of uncertainty or in the informational nature of the outcomes (typically quantitative or qualitative). Depending upon these variables, the valuation of fatality-related and illness-related effects might be carried out in different ways.

At the simplest level, an assessment may only provide qualitative outcomes indicating the risk level (say negligible vs. unacceptable) on the basis of acceptable 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 a semi-quantitative valuation of the change in human health risk. Finally, where outcomes from an overall toxicological risk assessment exist and they are expected to have a tolerable level of uncertainty, a quantitative monetary valuation of illness and fatalities effects might be performed. A short discussion about valuation techniques for changes in mortality and morbidity risks is provided in the following paragraph. Here it is sufficient to note that when coping with mortality valuation, a large consensus exists about the adoption of the concept of VOSL (Value of a

[5] In general, the consensus is that there are three levels of risk to human health (OECD, 2000):
i) a level demanding immediate action, usually referred to as *de manifestis,* intolerable or unacceptable; ii) a level regarded as trivial, referred to as *de minimis*, negligible or acceptable; iii) a level in between i) and ii) where a consideration of the costs and benefits of risk reduction should be performed.

Statistical Life) as a monetary measure of individuals' preferences for reducing the risk of fatality. In the context of hazardous chemicals, however, criticism of the use of VOSLs – which are not adjusted for age, health state and latency effects – has led to the development of the alternative concept of VSLY (Value of Statistical Life Years extended)[6]. For a complete literature review on human risk valuation techniques we refer to Cropper and Freeman, 1991; Johannesson and Johansson, 1998.

In terms of environmental risk assessment, the two dominant ecosystem risk-management paradigms today are based, on the one hand, on the traditional *endpoints* stance of ecological risk assessment and, on the other, on the more holistic perspective of ecosystem valuation, the so called *ecological-health* paradigm. The former is the most commonly used approach to ecological risk assessment and is based on the definition of toxicological and eco-toxicological *endpoints*, usually derived in laboratory or field bioassays under highly controlled experimental conditions (see Figure 1 for a complete overview of the information involved in this approach). The latter, in contrast, recognises the importance of simultaneously considering all the components of an ecosystem and all relevant dynamics for ecological well-being, balance and health, especially those that have an analogue in the field of human medicine. Principles of thermodynamic and ecological theory are also included in modelling experiments (Kay, 1991; Costanza *et al*., 1992; Rapport, 1989).

The debate on the ethical and methodological strengths and weaknesses of the two approaches is fascinating and the copiously available literature on the subject will satisfy those intrigued by it (see, for example, Shrader-Frechette, 1998; Norton, 1995). However, whether dealing with chemical risk management, the substantial advantages of the *endpoints* approach, as well as its thirty year-old consolidation of scientific experience, justify its sovereignty in the international scientific community[7] (EEC, 1996; CSA, 1996; MSE, 1997; MHPPE, 1989; NRC, 1996; RC, 1998; ANZ, 1995; UK DOE, 1995; USEPA, 1998).

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 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, either in terms of the quality or the informational nature of their output (i.e., providing quantitative or qualitative results). Since knowledge from the process of risk assessment is used in the valuation of risk changes, is it relevant to determine how such differences might affect the degree of freedom of economists.

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

[6] VSLY represents the impact of premature death on an average individual's life span and it allows for distinctions between risk reduction measures on the basis of their effects on longevity.
[7] The advantages of the endpoints paradigm, with respect to the ecosystem-health one, stem from its being simple to use and easy to understand. Because a large database exists for many chemicals and species, it provides an accurate and precise basis for ecosystem management. Besides, as with traditional risk assessment and management, many insights and lessons from the past fit with it. Finally, it works best for assessing and managing chemicals.

assessment and risk characterisation (see Figure 1). The dose-response assessment identifies 'zero damage' thresholds, the so-called PNECs – *Predicted No Effect Concentrations* - based on extrapolations from test data to the environment. Exposure assessment, as is clear, assesses the expected concentration of the chemical concerned in different environmental compartments such as the PEC – *Predicted Environmental Concentration*. Risk 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. 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).

The consequences of this are not trivial. As effectively outlined in a recent report by OECD (OECD, 2000), it follows, at least in principle, that where the output of the risk assessment is expressed as the exposure/toxicity ratio, insufficient information will be available to quantify it and thus to place a proper monetary value on any changes in risk. Since information on the environmental target is lacking, results indicate the potential for harm rather than the actual risk. In situations where the risk assessment does not consider the environmental system that would potentially be damaged, the resulting risk valuation could only be qualitative in nature. Instead, when the assessment provides combined information about the potential for harm with environmental concentration, exposure and 'stock at risk' data, a quantitative prediction of the probability of the specified impact(s) might be developed.

The implications of the above discussion are significant for the valuation issue and will be taken into consideration in our comparative approach.

Figure 1: Scheme of the eco-toxicological risk assessment process as implemented at the EU level.

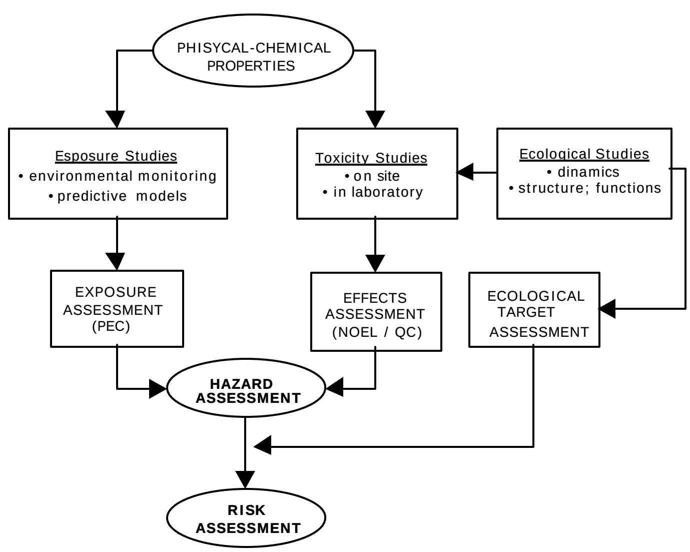


Table 1: Synthesis of human health risk considered in the EU Risk Assessment (Dir. 93/67/EEC; Reg. EC 1488/94).

<table>
<tr><th>Risk group</th><th>Risk end-points</th><th>Associate and indirect impacts</th></tr>
<tr><td>Workers</td><td rowspan="2">• Acute toxicity
• Irritation
• Sensitisation
• Repeated dose toxicity
• Mutagenity
• Carcinogenity
• Reproductive toxicity</td><td rowspan="4">• Fatalities
• Various morbidity effects
• Lost working days and non-working day opportunities
• Health care costs
• Changes in quality of life
• Stress effects related to pain and suffering</td></tr>
<tr><td>General public</td></tr>
<tr><td>Human indirectly exposed via the environment</td><td rowspan="2">• Consumption of drinking water, crops, milk and meat, fish
• Inhalation of air
• Ingestion of soil</td></tr>
<tr><td>Consumers</td></tr>
</table>

Table 2: Synthesis of environmental risk considered in the EU Risk Assessment

<table>
<tr><th>Risk group</th><th>Medium of exposure</th><th>Associated and indirect impacts not explicitly identified by the risk assessments</th></tr>
<tr><td>Aquatic organisms</td><td>Surface water</td><td>• Impact on natural fisheries and associated ecosystems in terms of species diversity, population dimension and support function
• Impacts on commercial fisheries through loss of food sources
• Impacts on recreational fisheries trough loss of certain species, changes in catch rate, size of fish, etc.</td></tr>
<tr><td>Benthic organisms</td><td>Sediment</td><td>• Impact on natural ecosystems in terms of species mix, population dimension and support function
• Through the above, impacts on dependent commercial and/or recreational activities</td></tr>
<tr><td>Terrestrial organisms (flora and fauna)</td><td>Soil</td><td>• Impact on natural ecosystems in terms of species mix, population dimension and support function
• Impacts on agricultural, forestry and other forms of land use
• Through the above, impacts on amenity or aesthetic quality of land</td></tr>
<tr><td>Fish-eating predators</td><td>Fish</td><td>• Impact on natural ecosystems in terms of species mix, populations
• Impacts on commercial and recreational fisheries
• Impacts on recreational values (ex. Birdwatching)</td></tr>
<tr><td>Worm-eating predators</td><td>Earthworms</td><td>• Impact on natural ecosystems in terms of species mix, populations
• Impacts on agricultural, forestry and other forms of land use
• Impacts on recreational values of affected land areas</td></tr>
<tr><td colspan="2">Atmosphere</td><td>• Impact on natural ecosystems in terms of species mix, populations
• Impacts on agriculture (yield and quality)
• Impacts on building materials (corrosion and reduced life)
• Impacts on recreations (loose of visibility)</td></tr>
</table>

2.2 Theoretical background for valuation of environmental risks

As remarked upon in the previous paragraph, valuing changes in risks due to agrochemicals means addressing the issues of human health and alterations in the well being of ecosystems. The following discussion attempts to provide an overview of the available empirical economic techniques useful for this purpose, noting the differences and similarities between approaches when considering the ‘objects’ that are to be valued.

Broadly speaking, the economic literature offers two alternative approaches to environmental 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 is sufficiently flexible for valuing risk to both natural and agro-ecosystems.

The *human capital* approach stems from the idea that the value of an individual is equal to the value of his/her contribution to total production and assumes that a measure of this can be inferred from his/her earnings. This 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. Its most important shortcoming is that it is inconsistent with the individualistic foundation of welfare economics, since it does not take popular preferences about changes in health risks into consideration. Besides, indirect damage to health and injuries, both of directly affected persons and of their relatives, are not considered, nor is the statistical value 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 willingness-to-pay (WTP) measures of environmental health risks have been explored since the 1970s and nowadays have quite a solid background (see, for example, Schelling, 1968; Mishan, 1971; Jones and Lee, 1976; Rosen, 1988; Cropper and Freeman, 1991; Viscusi, 1993; Johansson, 1995). During the past two decades, these have been extended and applied to a wide variety of non-market or public goods and social programs, including public investments in the health services, road safety, the development of water resources and improvements in environmental quality and health. For a complete overview we refer to the Envalue USEPA database [2].

These measures are rooted in utility economics and are based upon a well-known and plain but nontrivial idea. Whenever a good, such as an environmental asset, has either no obvious markets or no market at all, its economic value can be inferred by directly or indirectly analysing individuals’ preferences for it. The concepts used to monetarily quantify such values are WTP (willingness-to-pay) and/or WTA (willingness-to-accept-compensation) for positive and/or negative changes in the environmental asset[8]. From this it follows, at least in principle, that the value of alterations in human and environmental health might be derived from individuals’ preferences, and expressed as a WTP (or WTA) for a reduction (or increase) in hazards in the

[8] Alternatively, when risks are fatality-related, the concept of VOSL or VSLY is usually adopted.

situation[9]. The practical problem, therefore, is one of deriving credible estimates of values in situations where markets are either missing or incomplete.

Preferences can be measured using two basic approaches, one involving *stated* preferences – i.e., preferences conveyed by a response to a question, the other involving *revealed* preferences – i.e., preferences inferred from the behaviour of an individual making choices about some good or option not explicitly connected to the attribute being valued. Figure 2 shows available WTP valuation techniques and how they are related. For an overview of the WTP literature, see, among others: Branden and Kolstad, 1991; Hanley and Spash, 1993; Freeman, 1993.

The previous stance represents the most commonly used basis upon which to discriminate between the available valuation methods. Broadly speaking, this perspective draws attention to the fact that only stated preference techniques are capable of capturing the *non-use* values of environmental goods; revealed ones simply provide their instrument-related worth. The latter are, therefore, focused on the capital values of environmental goods (either direct or indirect uses), while the former also capture values stemming from existence *per s*e. For a thorough analysis of this interesting issue, the reader can refer to a copious amount of literature. Here, we note that such divergent 'talent/attitude' can make a large difference when one is selecting between alternative approaches.

In Figure 2, Contingent Valuation (CVM) and Conjoint Analysis (CA) are *stated* preference techniques (Higley and Wintersteen, 1992; Mullen *et al*., 1997; Brethour and Weersink., 2001; Cuyno *et al*., 2001; Foster and Mourato, 2000; Buzby *et al*., 1995; Fu *et al*., 1999; Ravenswaay and Hoehn, 1991-a; Misra *et al*., 1991; Baker and Crosbie, 1993). All others are *revealed* preference ones: related to averting expenditures or to defensive expenditures (Pingali *et al*., 1994; Crissman *et al*., 1994; Antle and Pingali, 1994; Antle, 1991), the hedonic price method (SÖderqvist, 1998; Beach and Carlson, 1993), market demand functions (Hammit, 1993; Thompson and Kidwell, 1998), travel cost method and random or discrete utility choice modelling (Lohr *et al*., 1999; Eom, 1994; Blend and Ravenswaay, 1999; Huang, 1993; Govindasamy and Italia, 1997).

Another important consideration is whether or not the technique employed explicitly assesses the risk change from the valuation of its monetary output (Cropper and Freeman, 1991). Some approaches rely on estimates of the alteration in well-being taken from biomedical and eco-toxicological literature – namely, risk assessments, dose-response and production functions - to predict changes in environmental balance (i.e., for chemicals, changes in some risk endpoints: carcinogenity, acute and chronic exposure, neurotoxicity, etc.). They then employ either direct or indirect WTP methods to estimate the value individuals place on changes in risk. By contrast, the averting behaviour approach views the relationship between environmental impoverishment and its monetary value apart from any technical or expert predictions explaining cause-effect

[9] It is useful to remember that WTP/WTA reflects not only individuals' preferences, but also their perception of and attitude towards risks.

links. It yields inferences about value from observations of how people modify their behaviour in response to environmental alterations.

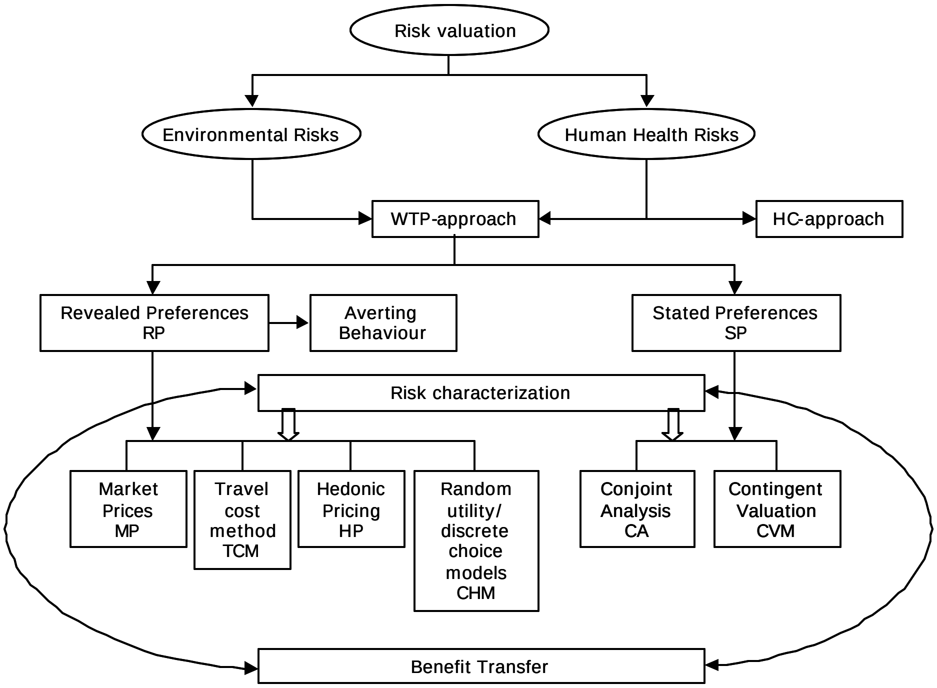
Another question is whether a 'taxonomy' of the valuation techniques should consider the two steps, i.e., the cause-effect assessment and the economic valuation, as separate procedures. Usually cost-benefit references list them as interdependent but separate (Pearce and Secconbe-Hett., 2000). However, from the broader perspective of Lancaster's attribute theory of consumer choice (Lancaster, 1966, 1971), production functions or risk assessment procedures can be viewed as integral parts of several valuation procedures. When individuals are supposed to have utility functions in which they combine various goods and services, including environmental ones, as far as the exposure-response interpretation is concerned, valuation is applied to the outcome of the risk assessment. Hence, it makes sense to consider the two procedures as a whole (see Figure 2).

From this perspective, it is natural to consider the valuation process subordinate to the assessment process, both in terms of the quality and uncertainty of the outcomes. Analysts should look for the best available initial information.

Cost-benefit or risk-benefit references usually mention dose-response or exposure-response functions (for the 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 chemicals, whenever possible, analysts should rely on the results of the risk assessment procedure (as described in subsection 2.2). Unlike production functions, risk assessment is based on an *ex-ante* stance, 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 results' quality. 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 inter-dependent. Finally, the iterative nature of the process allows the progressive incorporation of new useful information when it becomes available.

To settle this discussion, the *benefit transfer* - or *value transfer* - procedure acts as a link among the different techniques, since it 'borrows' the monetary values from completed valuation studies for use in unexplored contexts (Bal and Nijkamp, 2001). 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 the most uncertainty and require stronger theoretical assumptions or simplifications (Matthews and Lave, 2000; Navrud and Bergland, 2001).

Figure 2: Available valuation techniques for environmental and human health risk changes.



3. COMPARATIVE ANALYSIS AND THE USE OF META-ANALYSIS

Far from being an uncontroversial subject, environmental valuation has generated a number of claims about its ethical, theoretical and technical drawbacks. Without entering into detail about this ongoing debate, we will limit our discussion to those aspects that appear to be meaningful for our subsequent comparative analysis.

Setting aside ethical claims that are mostly related to consumer sovereignty (see, for example, Sen, 1977; Penz, 1986; Naess, 1973; Singer, 1979; Spash, 2000), we focus instead on practical objections that are relevant when dealing with empirical results.

- ***The monetary valuation***

From a pragmatic perspective, one of the main sources of controversy in this field is whether *money* is a useful unit of measure for quantifying the value of changes in environmental and human health risks and, operatively, whether economic valuation can provide useful input to decision-making.

Once the ethical foundation of welfare economics has been accepted, some argue that answering this question is not a matter of science but instead a matter of judgement, since there is an independent source of measurement of such values against which to test these results (see Blamey and Common, 2000). Consequently, the use of WTP estimates is the only means for directly comparing the values of non-market goods on a quantitative basis - as for pesticide risk changes. In this sense, debating the legitimacy of this premise is futile, since valuation is a means of measuring public preferences for environmental resources rather than a direct valuation of the resources. Moreover, using quantitative results enables us to employ, whenever budget and technical constraints require it, the *benefit transfer* procedure for inferring monetary values for an original case study from complete valuations applied in similar but different contexts (see Bal and Nijkamp, 2001; Brouwer and Spaninks, 1999; Brouwer, 2000, Navrud and Bergland, 2001).

Otherwise, if the ethical foundations of CBA are rejected when the environment is at stake, the question of how else to inform managers about individual preferences is raised. Researchers consider multi-criteria analysis the main alternative, but this also has *pros* and *cons*. If it might partially satisfy the criteria of ethical arguments in the domain of monetary reasoning, it fails in the respect that it does not allow the direct comparison of risk/benefit trade-offs, which are particularly relevant considering the multidimensionality of pesticide risks. Besides, the weighting procedure of the involved hazards, typically based on decision-makers' preferences, adds an additional arbitrary dimension without solving the problem of how to meaningfully elicit individual preferences. In this sense, some commentators advocate public consultation, but the major problem, direct comparability, would remain.

The previous arguments caused us to focus the comparative exercise on the empirical economic literature providing disagregated monetary estimates of the risks posed by pesticides to both human health and the environment. The results of an accurate literature search were a data set of more than 60 studies[10]. A subsequent selection on the aforementioned monetary basis resulted in a smaller data set of 27 studies. WTP estimates extrapolated from the sample set up the background for our comparative approach [see Table 4].

- ***Risk perception***

Due to 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.

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 the chosen process, 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 dynamic 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

[10] The bibliographical search lasted three months. It was carried out using several economic databases (EconLit and other online databases), a review of the references provided by each individual survey, and direct contacts with the main authors.

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; Slovic *et al*.,1997; Slovic, 1999; Sjoberg, 1998, 2000]

In our comparative analysis, we looked into differences between studies in the way they model popular risk perception, with the aim of investigating their causality direction with respect to WTPs. This information is captured by the RISKPERC variable (Table 5).

- ***Baseline risk level***

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 (i.e., the initial risk level specified in preference surveys, not to be confused with the actual risk).

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 [11].

In the meta-analytical approach, 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, otherwise, using exogenous information to determine a comparable initial risk level for each case study in the sample.

- ***Public vs. private nature of risk***

Another major issue in the valuation of health risks is how to deal with *altruism*, and what role it plays with respect to WTP. This is a particularly intricate subject, since it has been neglected in empirical studies to a large extent, with the exception of a few sporadic cases (Jones-Lee *et al*., 1985, 1991, 1992; Johannesson *et al*., 1996). Furthermore, the available empirical literature leads to partially misleading results, alternatively suggesting a positive or a negative correlation between individuals' willingness-to-pay attitude and the public nature of the improvement in safety[12]. Consequently, investigating the existence of a causality relation - and its direction - between WTPs and the public/private nature of the risk reduction is a fascinating challenge not

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

[12] Jones-Lee *et al*. (1985) show that the 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.

only in the theoretical but also in the meta-analytical literature. In the present survey we have made a clear distinction between environmental and human health risks, consumers and producers and collective health risks. In Table 4 such information is directly described by the RISKTYPE variable.

- ***Risk assessment and the valuation of environmental risk changes***

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 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. The actual or hypothetical/potential nature of the risk scenario, the approaches adopted to characterise the risks, as well as the type of endpoints and information used are included among the explanatory attributes (see Table 5).

4. AN APPLICATION OF A META-ANALYTICAL APPROACH TO PESTICIDE RISK VALUATION

4.1 Codification and brief description of the data set

The four points introduced in the above discussion have influenced the comparative analysis that we propose here as a preliminary investigation of the pesticide risk valuation literature.

A data set of 27 surveys providing disagregated monetary values for pesticide risks to both the environment and human health was interpreted and properly codified for our meta-analytical exercise. WTPs extrapolated from the sample set up the *effect size* to handle. Single mean estimates from each study were considered to avoid a multi-sampling bias. An appropriate standardisation of the estimates was necessary to make them directly comparable [13] (see Table 3).

Results provided by the various case studies are summarised in Table 3. Table 4 shows an annotated overview of the studies, codified according to several explanatory factors. A more detailed description of the codification is given in Table 5. Figure 3 gives a graphic account of the frequency of different items for the main attributes.

[13] Single mean WTPs extrapolated from each study were standardised and expressed in USD_{2002}/ per person per year. Standardisation from $/acre and $/household was performed considering the average rural density and the average household size in the country at issue, respectively. Standardisation from %price premium to $/ pound and from $/ pound to $/person year was performed considering the average price of the product and the related average per capita consumption in the country at issue.

Table 3: Brief description of the surveys' contents and results.

Study ID		Results (*)						
		Content	[WTP]	WTP hum	WTP env	WTP tot	WTPsnd [$/person year]	
1	Higley and Wintersteen, 1992	valuation of risks for 8 environmental categories associated with a single use of a specific insecticide (32 in total). 4 level of risks considered	[$/ year person]	-	-	12.95	13	medium
2	Lohr *et al* ., 1999	valuation of avoidance of moderate risk to environment eliminating one insecticide application. trade-offs between risk of yield losses and possible environmental quality improvements are analysed	[$/ year person]	-	-	11.37	11.4	medium
3	Mullen *et al* ., 1997	valuation of changes in pesticide risks for 8 environmental categories, described by time-period and for 3 risk levels	[$/ year person]	18.23	17.48	35.71	35.7	high
4	Brethour and Weersink, 2001	valuation of changes in pesticide risks for 8 environmental categories, described by time-period and risk level. WTP estimates are derived from CV survey's results by Mullen et al., 1997. Unit Value Transfer with income adjustment is used	[$/ year person]	24.22	23.97	48.19	48.2	high
5	Cuyno et. al., 2001	benefits valuation of Integrated Pest Management (IPM) program implementation reducing pesticide applications	[$/ year person]	28.48	23.59	52.06	52.1	high
6	Foster and Mourato, 2000	valuation of human health and biodiversity impacts associated with pesticide applications (cases of illness due to pesticide exposure during cultivation and n. of endangered farmland bird species)	[$/ year person]	1.92	14.52	16.44	16.4	high
7	Pingali *et al* ., 1994	analysis of chronic health effects and medical costs related to pesticide occupationalexposure	[$/ year person]	9.01	-	9.01	9.0	medium
8	Crissman *et al* ., 1994	analysis of health consequences of pesticide use. medical expenses and lost income are considered	[$/ year person]	22.75	-	22.75	22.8	high
9	Owens*et.al*., 1997	valuation of farmers demand for 3 new formulations of the herbicide Atrazine at different price premiums, for % of respondents and avarage acres	[$/unit of produce]	1.18	0.89	2.07	1	low
10	Buzby *et al*., 1995	cost-benefit analysis of a potential pesticide ban (postharvest fungicide for grapefruit)	[%premium]	38	-	38	1.6	low
11	Eom, Y. S., 1994	analysis of purchasing intentions for a pesticide residues-free produce and factors affecting those choices. price/risk trade-offs are then used to infer the values of risk reduction embodied in the selection of safer product	[$/unit of produce]	1.08	-	1.08	8.9	medium
12	Fu T-T *et al* ., 1999	valuation of different levels of reduction in health risk due to pesticide-residues in chinese-cabbage	[$/unit of produce]	1.05	-	1.05	10.8	medium
13	Blend and Ravenswaay, 1999	analysis of demand for regual, ecolabeled and unlabeled apples at different price premium. effects of explanatory variables on purchase probability are considered	[$/unit of produce]	1	-	1	3.1	low
14	Ravenswaay and Hoehn, 1991-a	valuation of demand for eco-labelled apples. WTP for risk reduction and WTA cosmetic damage are derived	[$/unit of produce]	1.26	-	1.26	2.8	low
15	Ravenswaay and Hoehn, 1991-b	valuation of the impact of pesticide health risk information on apples demand.	[$/unit of produce]	2.33	-	2.33	9.5	medium
16	Misra *et al* ., 1991	analysis of demand for fresh products certified as free of pesticide residues	[%premium]	10	-	10	0.8	low
17	Huang, 1993	develompent of model for analysis of consumers' food safety concerns relative to pesticide; estimation of interelationships among risk perceptions, attitudes about pesticiedes use and WTPs	[%premium]	5	-	5	0.3	low
18	Roosen *et al* ., 1998	benefits valuation of partial reduction of insecticide use in apple production.	[$/unit of produce]	0.59	-	0.59	4.0	low
19	Hammitt, 1993	analysis of price differences between organic and conventional version of 27 fresh products. Valuation of willingness to pay to reduce mortality risk.	[$/unit of produce]	0.39	-	0.39	7.6	medium
20	Thompson and Kidwell, 1998	analysis of the demand for organic produce using actual retail price premiums. Cosmetic characteristics of produce and demographic factors are considered	[%premium]	107	-	107	9.3	medium
21	Govindasamy and Italia, 1997	analysis of WTP for IPM and organic fresh products as a function of demographic characteristics	[%premium]	10	-	10	0.8	low
22	Govindasamy *et al* , 1998-a	analysis of comsumers' demand and perceptions of Integrated Pest Management produce	[%premium]	6	-	6	0.6	low
23	Ott, 1990	analysis of consumers' concerns on pesticide residue; preference on type of assurance for pesticide residues free produce; WTP for fresh produce certified free of pesticide residues; WTA lower cosmetic quality	[%premium]	5	-	5	0.3	low
24	Ott *et al* ., 1991	anaysis of consumers' concerns on pesticide residue; purchasing habits; WTPs for fresh produce certified and tested as pesticide residue free	[%premium]	5	-	5	0.3	low
25	Weaver *et al* ., 1992	analysis of consumers' concerns; changes in purchasing behaviour; willingness to purchase or to accept several produce characteristics	[%premium]	12	-	12	1.5	low
26	Anderson *et al* ,. 1996	analysis of consumers' response to IPM and certification of food-safety	[%premium]	10	-	10	0.5	low
27	Baker and Crosbie, 1993	analysis of individual preferences for food-safety attributes related to pesticides use	[$/unit of produce]	1.03	-	1.03	7.5	medium

(*) Standardised WTPs are expressed in 2002 USD per person per year. Standardised mean estimates of WTP_{TOT} and their categorisation in *low*, *medium* or *high* levels are reported in the last column. *Low*: $0<WTP<5$; *Medium*: $5\leq WTP<15$; *High*: $WTP\geq 15$.

Table 4: Annotated overview of the dataset codification used to run the comparative analysis.

	STUDY ID	DOC	YEA	COU	AIM	AGR	PEST	RISKTYPE		RISKSCEN	RISKCH			RISKPER	METHOD	GROUP	MOD		WTP level
								Hum	*Env*		*Approach*	*End-point*	*Info*				*Payment vehicle*	*Interview*	
1	Higley and Wintersteen, 1992	JA	1990	USA	BV/PE	field crops	ins	occupational exposure	non-target ecosystems	potential	ex-ante	acute/ chronic	exp, tox	objective	CVM	farm	price premium	mail	medium
2	Lohr *et al*., 1999	JA	1990	USA	CB/PE	field crops	ins	occupational exposure	non-target ecosystems	actual	ex-ante	acute/ chronic	exp, tox	subjective	CHM	farm	acceptable yield loss	mail	medium
3	Mullen *et al.*, 1997	JA	1993	USA	BV/PE	fruit	all	occupational exposure	non-target ecosystems	actual	ex-ante	acute/ chronic	exp, tox	objective	CVM	strat	monthly grocery bill	mail	high
4	Brethour and Weersink, 2001	JA	1993	Canada	BV/PE	field crops	all	occupational exposure	non-target ecosystems	actual	ex-ante	acute/ chronic	exp, tox	objective	CVM	strat	monthly grocery bill	mail	high
5	Cuyno et. al., 2001	JA	1999	developing	BV/PE	veg	all	occupational exposure	non-target ecosystems	actual	ex-ante	acute/ chronic	exp, tox	objective	CVM	farm	price premium	n.a.	high
6	Foster and Mourato, 2000	JA	1996	UK	BV/PE	cereal	all	occupational exposure	biodiversity	actual	ex-post	acute	damage	objective	CA	strat	monthly grocery bill	mail	high
7	Pingali *et al.*, 1994	JA	1991	developing	BV/PE	cereal	ins	occupational exposure	n.c	actual	ex-post	chronic	damage	subjective	HC	farm	med expense	n.a.	medium
8	Crissman *et al*., 1994	JA	1994	developing	BV/PE	veg	all	occupational exposure	n.c	actual	ex-post	acute/ chronic	damage	n.c.	HC	farm	med expense	n.a.	high
9	Owens*et.al*., 1997	REP	1995	USA	CB/PE	cereal	herb	occupational exposure	surface water	potential	generic	cancer	generic	n.c.	DS	farm	price premium	phone/mail	low
10	Buzby *et al*., 1995	JA	1995	USA	BV/PE	fruit	fung	consumer health	n.c	actual	ex-ante	cancer	risk	n.c.	CVM	cons	price premium	phone/mail	low
11	Eom, Y. S., 1994	JA	1990	USA	BV/PE	fruit/veg	all	consumer health	n.c	potential	ex-ante	cancer	risk	subjective	CHM	cons	price premium	store	medium
12	Fu T-T *et al.*, 1999	JA	1995	USA	BV/PE	veg	all	consumer health	n.c	actual	ex-ante	cancer	risk	objective	CVM	cons	price premium	face-to-face	medium
13	Blend and Ravenswaay, 1999	JA	1998	USA	CB/PE	fruit	org/low-input	consumer health	n.c	implicit	generic	chronic	generic	n.c.	CHM	cons	price premium	phone	low
14	Ravenswaay and Hoehn, 1991-a	REP	1990	USA	BV/PE	fruit	all	consumer health	n.c	potential	generic	chronic	generic	subjective	CVM	cons	price premium	mail	low
15	Ravenswaay and Hoehn, 1991-b	BO	1989	USA	CB/PE	fruit	all	consumer health	n.c	actual	ex-ante	cancer	risk	subjective	CHM	strat	price premium	retail data	medium
16	Misra *et al*., 1991	JA	1989	USA	CB/MS	fruit/veg	all	consumer health	n.c	potential	generic	chronic	generic	subjective	CA	cons	price premium	mail	low
17	Huang, 1993	JA	1989	USA	CB/PE	fruit/veg	all	consumer health	n.c	potential	generic	chronic	generic	subjective	CHM	cons	price premium	mail	low
18	Roosen *et al*., 1998	JA	1998	USA	BV/PE	fruit	ins	consumer health	n.c	actual	expert judgment	neurotox	tox	objective	CVM	cons	price premium	face-to-face	low
19	Hammitt, 1993	JA	1985	USA	BV/PE	fruit/veg	org/low-input	consumer health	n.c	actual	ex-ante	acute/ chronic	exp, tox	subjective	MP	cons	price premium	retail data	medium
20	Thompson and Kidwell, 1998	JA	1994	USA	CB/MS	fruit/veg	org/low-input	consumer health	n.c	implicit	generic	chronic	generic	n.c.	MP	cons	price premium	retail data	medium
21	Govindasamy and Italia, 1997	REP	1997	USA	CB/MS	veg	org/low-input	consumer health	n.c	implicit	generic	chronic	generic	objective	CHM	cons	price premium	store	low
22	Govindasamy *et al*, 1998-a	REP	1997	USA	CB/MS	veg	org/low-input	consumer health	n.c	implicit	generic	chronic	generic	n.c.	DS	cons	price premium	store	low
23	Ott, 1990	JA	1990	USA	CB/MS	fruit/veg	all	consumer health	n.c	implicit	generic	chronic	generic	n.c.	DS	cons	price premium	store	low
24	Ott *et al*., 1991	BO	1990	USA	CB/MS	fruit/veg	all	consumer health	n.c	potential	generic	chronic	generic	n.c.	DS	cons	price premium	mail	low
25	Weaver *et al*., 1992	JA	1990	USA	CB/MS	veg	all	consumer health	n.c	implicit	generic	chronic	generic	n.c.	DS	cons	price premium	store	low
26	Anderson *et al*,. 1996	JA	1994	USA	CB/MS	cereal	org/low-input	consumer health	n.c	implicit	generic	chronic	generic	n.c.	DS	cons	price premium	store	low
27	Baker and Crosbie, 1993	JA	1992	USA	CB/PE	fruit	org/low-input	consumer health	n.c	potential	ex-ante	cancer	risk	n.c.	CA	cons	price premium	store	medium

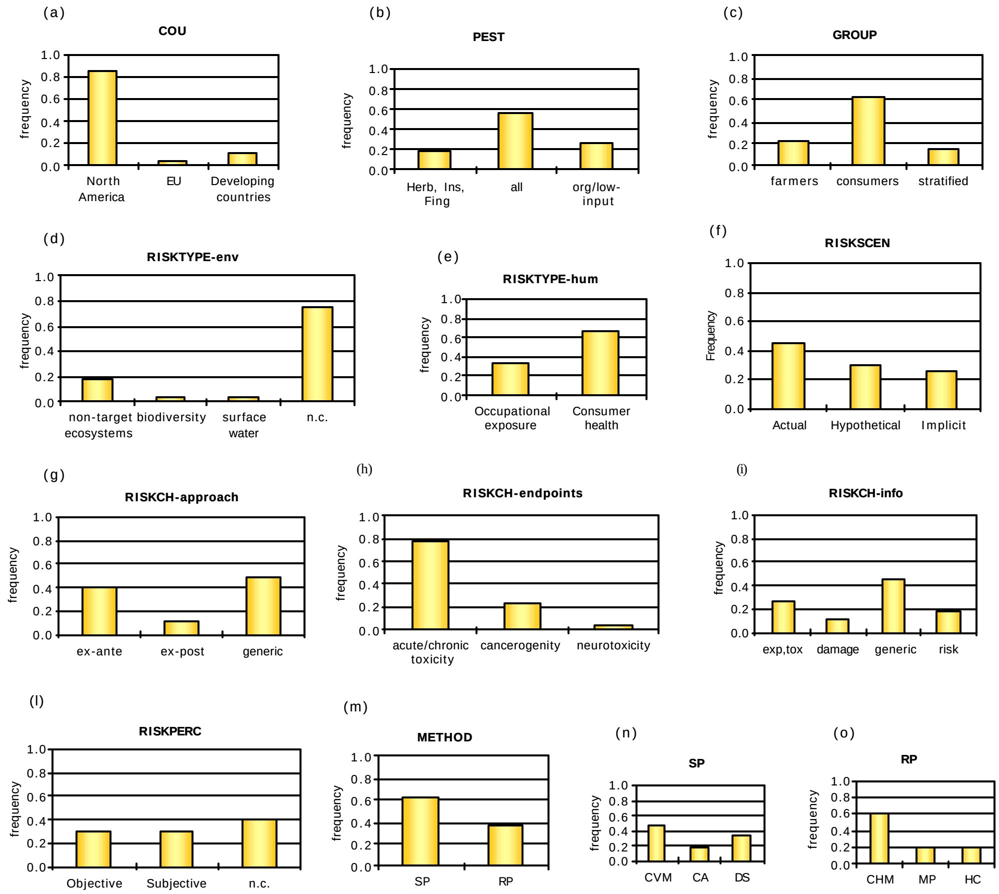
Table 5: Brief description of the dataset codification.

Dataset codification	
DOC	Type of document JA: journal article, REP: research paper, BO: book
YEA	Year of datas employed in the survey.
COU	Country of concern.
AIM	Aim of the study BV: benefits/risks valuation; CB: analysis of consumers'/farmers' behaviour; PE: improving the effectiveness of environmental/food safety policies; MS: improving the effectiveness of marketing strategies.
AGR	Type of farming or type of agricultural produce of concern
PEST	Type of pesticides (herbicide, insecticide, fungicide, pesticides as a whole) or type of low-input agriculture (organic, IPM, etc.) of concern.
RISKTYPE	Type of risk analysed either explicitly or implicitly Hum: human health risks related either to pesticides residues in food (consumers' health) or to pesticide exposure in the farmland (typically exposure at work during the preparation and application of pesticides: occupational exposure); Env: pesticide risk for environmental ecosystems (non-target aquatic and terrestrial ecosystems) and biodiversity, in the farmland.
RISKSCEN	Scenario of risk *Implicit*: the survey implicitly refers to pesticide risks (typically, whether dealing with not conventional agricultures, i.e. organic or low-input agricultures). *Actual*: the survey refers to an actual risk for a certain target environment or human population. *Potential*: the study refers to a potential risk scenario for a generic environment or human population.
RISKCH	Risk characterization *Approach*: approach to risk characterisation (Ex-ante: features affecting pesticide potential hazard are considered; Ex-post: damage indicators are considered; Generic). *End-points* : toxicologial endpoints considered (Acute and/or chronic toxicity; cancerogenity; neurotoxicity). *Info*: type of information used to define pesticide risks (Exp: exposure parameters; Tox: toxicity parameters; Damage: damage estimates; Risk: hazard/risk quantitative estimates; Generic: in case of implicit risk scenario)
RISKPER	Risk perception *Objective*: respondents are assumed to accurately perceive the risks in response to information. Risk is supposed to be an objective element (prior perceived risk = technical risk estimate). *Subjective*: risk is supporsed to be a subjective element (i.e. prior perceived risk ≠ techical risk estimate). n.c.: not considered
METHOD	Research method (See Figure 2)
GROUP	Group of stakeholders of concern
MOD	Additional methodological specifications: payment vehicles; type of interview.

As one can easily observe (Figure 3a), the sample is strongly biased towards North American countries: the number of studies performed either in developing countries or in the EU is small. However, this imbalance is not related to sample bias but instead stems from the regulatory and research background of the different areas. As already noted, the US's experience in regulating environmental chemical-related risks has been a major topic of discussion, starting with the *Toxic Substances Control Act* (USEPA, 1978). Moreover, since the early eighties, the application of WTP approaches to risk valuation has been strongly encouraged by the systematic use of CBAs for the formal appraisal of major US environmental policies.

Few studies consider the negative externalities of the use of particular compounds or groups of pesticides (i.e., herbicides, insecticides, fungicides, etc.), see Figure 3b. Most of the case studies view pesticide impacts as a whole or implicitly refer to pesticide impacts when dealing with the environmental effects of organic or low-input agricultural production[14] (see Figure 3f).

Figure 3: Frequency of main explanatory variables.



The majority of the surveys focus on consumers' rather than farmers' preferences (see Figure 3c). Understandably, consumer risk - related to the involuntary ingestion of pesticide residue in fresh products – prevails over occupational exposure (see Figure 3e). In this sense, the economic literature seems to suffer from the same type of distortion that we pointed out when discussing lay vs. expert perceptions of pesticide risks (subsection 2.1). Yet, in the scientific and regulatory community there is widely shared agreement about the fact that the management of occupational exposure is a priority objective compared with the presence of pesticide residues in fresh food (see STOA, 1998). Nevertheless, this bias towards the consumers' side seems mostly due to the location bias previously mentioned, resulting from the importance of food safety policy in the USA.

Only those surveys considering the farmers' side deal with the valuation of pesticide risks for both human health and the environment, i.e., agro-ecosystems and farmland biodiversity (see Figure 3d and 3e). The others simply consider pesticide impacts on human health. A possible interpretation of this trend is that when

[14] Typically, IPM (Integrated Pest Management) production is considered.

researchers are concerned with comparing the trade-offs between on-farm and off-farm pesticide effects for farmers, they are less inclined to address the private (pesticide residue risk) and public (environmental risks) dimensions of consumers' preferences simultaneously. An increase in consumer interest in the valuation of pesticide effects on biodiversity and non-target ecosystems is notable, but some ongoing projects on this issue are still in the embryo stage and could not be included in the present analysis.

Of those that discuss risk characterisation, the majority of studies refer to actual rather than potential risk scenarios[15] (Figure 3f). In contrast, in most cases, the approach adopted for risk characterisation is a generic one (Figure 3g). This results partly from a lack of suitable scientific information but mainly from the fact that a considerable number of studies refer to an implicit scenario of risk. The *ex-ante* approach is applied more frequently than the *ex-post* one, suggesting that the precautionary stance is more widely recognised. Finally, the several risk endpoints, acute and chronic toxicity, occur more frequently than carcinogenity, in accordance with the fact that pesticide cancer risks for humans are expected to be relatively low (Figure 3h).

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. A considerable part of the sample totally neglects this point, however, despite its theoretical relevance (Figure 3l).

In terms of methodology, researchers seem to be prone to choose stated preference more often than revealed preference techniques (see Figure 3l; 3m; 3n)[16]. In particular, CV and CHM surveys are the most common SP and RP techniques, respectively.

A few additional points can be put forward by means of plain cross-tabulations among different types of valuation techniques and a small number of risk specifications (Figure 4). Firstly, when referring to an actual risk scenario, contingent valuation and conjoint analysis, surprisingly, are favoured over revealed preference analysis. The obvious drawback of CV and CA methods is that the simulation of the market is hypothetical. Nevertheless, the SP methods' greater flexibility is attractive to researchers; they may prefer to deal with hypothetical bias instead of a rigid tool (especially when coping with multidimensional issues, as with pesticide risks)[17]. Secondly, CV and CA surveys are supported by a more favourable scientific attitude with respect to risk characterisation. For SP studies, the *ex-ante* approach is most often employed, whereas RP surveys are generally based on an *ex-post* perspective. The former considers either information on exposure and (eco)toxicological effects or information pertaining to risk estimates; the latter refers to damage measures. Finally, CV and CA studies usually use a subjective paradigm for modelling risk perceptions, while RP surveys make the opposite assumption.

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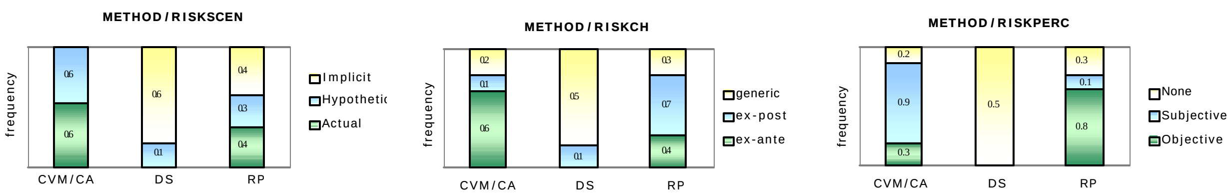
[15] The actual or potential nature of the risk scenario stems from the difference between *hazard* and *risk* described in subsection 2.1 and Figure 1. A study refers to an actual scenario of risk if it considers an environmental system that is actually exposed to a hazard. Otherwise, we assume that the study refers to a potential risk scenario.

[16] The reader should note that we have also included within the SP group the surveys that only provide descriptive statistics of interviews' results preliminary to inclusive CV or CA exercises (i.e., DS: direct surveys). Excluding these, the proportion of SP and RP surveys is fairly balanced.

[17] As observed by Owens (Owens *et al*., 1997), this weakness 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).

Figure 4: Cross-tabulations among methodologies and risk features.



4.2 Application of decision-tree algorithms to pesticide risk valuation studies

- ***Learning decision trees with C5/See5***

Through classification it is possible 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. 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. This method aims at analysing and predicting the membership of a class by the recursive partition of a multi-dimensional data set into more homogeneous subsets (see, for details, Quinlan, 1986). 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, which provides the classification of the instance. Each node in the decision-tree specifies a test for an attribute of the instance concerned, 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.

In a decision tree algorithm, the critical step method is 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 (see Shannon, 1948). The aim is then to select the attribute that is most useful for classifying instances, based on the so-called information gain (a measure for the goodness-of-separation for a given attribute 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 :

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

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

$$\mathrm{Info}_X(\mathrm{T}) = \sum_{i=1}^{n} \frac{|T_i|}{|T|} * Info(T_i)$$

Then, the following formula is the criterion of the attribute choice:

$$\mathrm{Gain}(X) = \mathrm{Info}(\mathrm{T}) - \mathrm{Info}_X(\mathrm{T})$$

This criterion is calculated for all the attributes and the one that maximises the expression is then selected. This attribute is the test used in the current tree node, and will be used for further tree derivation.

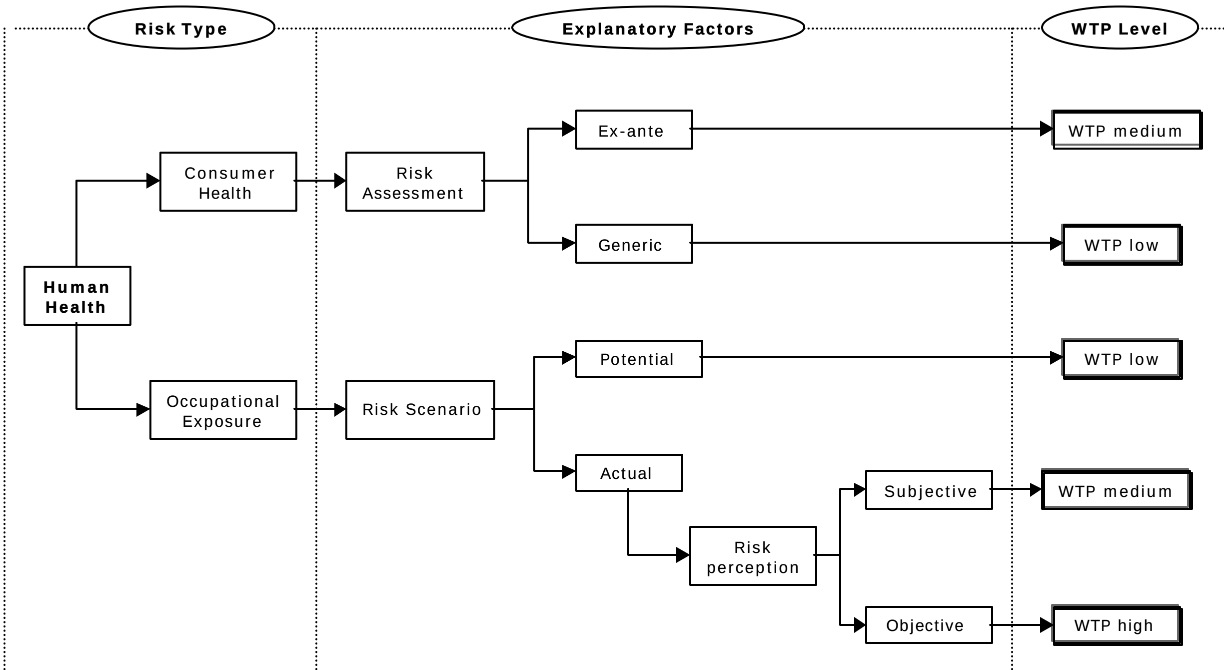
4.3 Discussion of results

- ***Decision tree***

The application of decision-tree algorithms appears to lead to appealing results, graphically represented in Figure 5.

The type of pesticide risk considered in the study explains most of the entropy endogenous to the data set (in Table 4: RISKTYPE-hum). In particular, the type of human health risk hides the environmental one that, as expected, does not appear in the decision tree. Within the data set, the small number (7/27) of studies addressing environmental pesticide risks easily explains such an outcome. The decision tree therefore splits into two branches, one referring to consumer health risk, and the other to occupational exposure. At this stage, it is interesting to note that whenever the following node in the former branch is the approach used for risk characterisation (RISKCH-approach), the determining attribute in the latter branch is the nature of the risk scenario (RISKSCEN). How should we interpret this structure?

Figure 5: Decision tree resulting from application of C5/See5 algorithms.



As already noted, biomedical and toxicological research on the effects of human exposure to pesticides has a solid background, since most major chemical risk policies were developed from a human-driven perspective. This could suggest two contradictory expectations, which would either be coherent or incoherent with our results. On one hand, we might reasonably expect that such a well-established scientific background would generally lead to a low level of heterogeneity within studies with respect to the risk characterisation variable (RISKCH approach) and, indirectly, to a low explanatory power for this attribute. On the other hand, however, the more detailed the scientific information on human risks available to analysts, the greater the importance of that factor in the valuation exercise (see subsection 2.1). In other words, when analysts utilise proper scientific information on pesticide risks, the level of uncertainty and the qualitative or quantitative nature of the information will affect the resulting valuation exercise, whereas, by contrast, more generic information will not influence the valuation outcome as much. In this sense, it is not surprising that this attribute appears at the beginning of the decision tree. In particular, when the survey uses an *ex-ante* or generic approach to risk characterisation, WTPs fall in the *medium* or *low* levels. From this perspective, arguments that the lack of adequate scientific information levels out differences in the WTP estimates for risk reduction, are confirmed.

In the other branch of the tree, the driving factor is the nature of the risk scenario (RISKSCEN). As with the risk characterisation attribute, this feature can affect the outcome of the valuation, since it is among the forms of information used as background for the WTP estimation. However, although the risk characterisation attribute is expected to have a notable influence on the valuation outcomes (WTPs), because it affects the way in which researchers design the survey and chose among the available techniques, the risk characterisation attribute influences the valuation results mostly because it affects individual preferences for risk reduction. The actual or potential nature of the risk scenario is expected to modify lay people's perception of risk and, consequently, their WTPs for risk reduction or safety improvements. People are expected to underestimate the probability of the occurrence of low-probability events referring to a potential risk scenario, as compared with events related to an actual risk scenario.

In accordance with this argument, the lower branch of the tree shows that the potential nature of the risk generally corresponds with *low* WTP estimates. On the other hand, when the case study deals with an actual risk, the risk perception feature (RISKPERC) is a major criterion in hypotheses about the expected WTP level. Specifically, whenever a survey makes a distinction between individual risk perceptions and technical risk estimates, the resulting WTPs are lower than when the subjective nature of risk perception is disregarded. WTPs have *medium* and *high* levels, respectively, in the two different scenarios.

It follows that the assumption that individual risk perceptions are subjective rather than objective leads to lower estimates of the value of changes in pesticide risks. To explain this result, one should carefully look into the factors that the sociological and psychological literature considers affecting risk attitudes and perceptions. In a further meta-analytical exercise, therefore, the socio-economic and demographic features of the involved sample should be included among the explanatory variables. When considering pesticide risks, in most cases the literature indicates that gender and income are among the major determinants of risk perception (Govindasamy *et al*., 1998-a; 1998-b). Willingness-to pay increases as income level increases and females are usually more willing to pay a premium for a certain amount of pesticide risk reduction. Additional socio-demographic characteristics, such as age and education level, have been found to have conflicting influences on pesticide risk concerns and WTPs (see Govindasamy and Italia., 1997; Anderson *et al*., 1996; Hammitt, 1990).

- ***Decision rules***

However, the pattern-class relationship expressed in the tree can also be written as a set of rules in the following way (see Table 6).

Table 6: 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 on training data (27 cases)

Decision tree		Rules	
Size	*Errors*	*N°*	*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

Each rule consists of a so-called Statistics (n, lift x) or (n/m, lift x) that summarises the performance of the rule. 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 $(n\text{-}m+1)/(n+2)$. The lift x is the result of dividing the rules estimated accuracy by the relative frequency of the predicted class in the training set.

In our system of calculation the two of the most reliable rules will be discussed. These two rules demonstrate the way in which the results of the tree analysis are expressed. The first rule states that a generic scientific information on human risk available to the analyst leads to a low value of the WTP. It is covered by 13 instances, while only 1 instance is misclassified. Its accuracy is estimated at 86%. The second rule identifies the actual nature of the risk scenario and the subjective importance of the risk perception as determinant features for a medium WTP. This rule is supported only by 4 instances, altough the accuracy estimated at 83,3 % is rather high.

- ***Performance assessment***

In order to explore which class distribution will yield the best classifier, we have chosen the two-performance measure, i.e. *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 incorrect classified instances are equal. Accuracy is particularly problematic as a performance measure, when the dataset 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 derived from a classification system (Kohavi and Provost, 1998). The performance of such systems is commonly evaluated using the data in the matrix. The following table shows the confusion matrix for a tree classifier. In the matrix both the rows and columns have the same headers, but there is a distinction between them; the rows of the table are the classes available for use in the classification process; the columns of the table are the classes chosen during the classification (see Table 7).

The number in the matrix represents the number of instances of the row class, which have been classified as a member 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 does not contain any number, then no misclassification occurs.

Table 7: Performance of the analysis

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)	

The results in this table can be interpreted as follows:

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*.

6 instances 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*.

5 instances of the known class *high* were correctly classified using the generated rules as members of the class *high*; 1 instance of the class *high* was incorrectly classified using the generated rules as members of the class *medium*.

5. CONCLUSIONS

When we started exploring the economics literature addressing pesticide human health and ecological risk valuation, we noticed that –despite several attempts of economists– this research area still suffers from a scarce 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 a plain interpretation to the available results. Moreover, the environmental dimension of pesticide risk is still partially 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 environmentally-driven historical background of chemical risk management, which still have notable repercussions in the developement of valuation surveys.

With this in mind, we have therefore sought to offer a critical overview of the literature on pesticide risks valuation in the light of a multidisciplinary perspective. Different interpretational modes are involved into both the theoretical and the empirical part of our work. On the one hand, we have looked into the scientific background of environmental risk, human health and ecological risk assessments; on the other hand, we had to envisage major controversial issues in environmental risk 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 addresses also 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 Gerrard *et al*., 2002; Suter, 1995).

In this sense, our comparative application is by no means exhaustive nor representative regarding 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 heath risks. Indeed, the aggregation of the welfare approaches proposed to compare different empirical outcomes represents a qualitative interpretation of the state-of-art. Nevertheless, the analytical method and procedure presented here, and based on artificial intelligence tehcniques, 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 an indiscriminate use of them for predictive approaches. This methodology, therefore, appears to be consistent with the broad scope of our analysis, and it allows us to highlight synergies across scientific, social and economic aspects of risk valuation.

From this viewpoint, our results confirm the expectation that the monetary dimension and the quality of valuation results is closely connected to the nature of outcomes 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 in WTPs is related to both the valuation technique and to 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.

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