THE USE OF PESTICIDE RANKING INDICES IN THE MODELLING OF ENVIRONMENTAL IMPACTS FROM PESTICIDE USE: A CASE STUDY OF THE EUROPEAN APPLE INDUSTRY

FRASER SINCLAIR QUIN

Thesis submitted in candidacy for the degree of Doctor of Philosophy

Institute of Ecology and Resource Management
Edinburgh University
January, 1999.

Declaration

I hereby declare that this thesis represents all my own work unless otherwise stated in the body of the thesis.

August, 1998.

Dedication

This thesis is dedicated to Sean and Heather. I apologise for the many absent nights during recent months. Most of all, however, I owe this thesis to my wife and best friend, Lynn. Without your love, understanding, patience, emotional (and financial) support, I would never have accomplished my dream, and changed my life for the better.

Abstract

Agricultural pollution from pesticides is an example of technological externality. In the presence of externalities decision making will probably never be optimal as externalities typically exist outside the decision making process. One of the main problems of incorporating externalities into the decision making process has been a lack of environmental impact data.

This thesis examines one methodological approach for identifying the environmental impacts associated with pesticide pollution: pesticide ranking indices. It will discuss the general rationale for the use of pesticide ranking indices, discuss the strengths and weaknesses of the various approaches, and recommend the adoption of one particular model for assessing the impacts associated with pesticide use at the farm and regional level. The model was tested against pesticide use data collected from European apple growing regions to ascertain whether results could be obtained that would be useable and understandable to decision makers at all levels.

Accepting that each methodology for identifying the impacts associated with pesticide use has both strengths and weaknesses, improvements in both model structure and data presentation are proposed that render the inclusion of environmental impact information in the decision making process more useable at the farm level. Thus, a modified model is presented that, it is argued, can adequately describe some of the external effects associated with pesticide use. This methodology can then be used by regulators wishing to minimise environmental pollution from agriculture and forestry, by identifying an appropriate threshold of acceptable, or unacceptable, environmental impact.

Acknowledgements

First of all I acknowledge the debt I owe to Professor Murray McGgregor who offered me both employment and the chance to study for a PhD. His professional support and friendship during my first two years of study is something that I will forever cherish.

In addition I also thank Dr Gareth Edwards-Jones for his excellent professional guidance, and for continuing to support me in my developing career. This also has been invaluable, and something I greatly appreciate. I also thank him for offering me a permanent position at the Scottish Agricultural College, and the chance to finally settle in Edinburgh ! Thanks go to Professor Barry Dent who's advice on numerous occasions has helped focus my attention on the job at hand. I have greatly appreciated his calm approach to often difficult problems. Many thanks to Alfredo Albin who worked with me on the DEAC project and is currently in the process of submitting his own PhD thesis. Alfredo's work ethic and commitment to the job at hand were often astonishing, and something which I have taken great encouragement from. I wish him all the best for the future in both his professional and personal life.

Thanks go also to colleagues at SAC, and fellow post-graduate students at IERM who have been too numerous during the last three years to name in their entirety, but include; Dave Oglethorpe, John Finlayson, Kate Corcoran, Wendy Kenyon, Hans Renia, Linda Goodall, Sagendra Tiwari, Alan Hern, Subash Babu, Jane Grandison, Hermann Meyer, Gustavo Ferrera, Margaret Stewart and finally Helen Snodin.

Table of Contents

Abstract iv

Acknowledgements v

Table of Contents vi

List of Abbreviations ix

List of Tables x

List of Figures xii

CHAPTER 1. INTRODUCTION. 1

1.1 Decision making and the environment 3
1.2 Pesticides and externalities 3
1.3 Objectives 3

CHAPTER 2. CROP PROTECTION PRODUCTS AND EXTERNALITY THEORY 7

2.1 Introduction 7
2.2 A definition of externalities 7
    2.2.1 Property rights. 9
    2.2.2 Market failure 13
    2.2.3 Public goods and externalities 14
    2.2.4 Private goods and externalities 15
    2.2.5 A note on pecuniary externalities 15
2.3 Environmental Economics 16
2.4. Crop Protection Products 21
2.5 Pesticides and externalities 23
    2.5.1 The problems of identifying and valuing pesticide related externalities 26
2.6 Summary. 28

CHAPTER 3. AN INTRODUCTION TO THE EUROPEAN APPLE INDUSTRY 29

3.1 The European apple industry in a global context. 29
3.2 Apple production in the EU. 31
    3.2.1 Apple varieties in the EU. 34
3.3 Fixed and variable costs in apple production 35
3.4 The reasons for regional variation in production costs 37
3.5 Breeding disease resistant apple varieties 38

CHAPTER 4. A CRITICAL REVIEW OF PESTICIDE RATING INDICES 40

4.1 Introduction to pesticide impact modelling 40
4.1.1 Pesticide use at the farm level 41
4.2 The need for additional pesticide ranking models 42
4.3 Pesticides and external effects 45
4.4 Methods to assess the environmental impacts of pesticides and pesticide use strategies. 47
4.4.1 The EIQ approach of Kovach, Petzoldt and Tette (1992) 53
4.4.1.2 An example of the results obtained from using the Kovach EIQ model. 57
4.4.2 The environmental yardstick of Reus and Pak (1993). 59
4.4.3 The Pesticide Index approach of Penrose, Thwaite and Bower (1994). 61
4.5 The potential use of pesticide impact ranking models as policy instruments 65
4.6 Reasons for the choice f the EIQ approach 67

CHAPTER 5. PESTICIDES IN THE APPLE GROWING SECTOR: LEGISLATIVE ISSUES, USE PATTERNS, AND INTEGRATED MANAGEMENT TECHNIQUES. 69

5.1 Introduction 69
5.2 The reasons fore current pesticide use patterns 69
5.3 Current pesticide legislation in the EU 70
5.3.1 Agri-environmental policy 73
5.3.2 The Fifth Environmental Action Programme (FEAP) 74
5.4 An overview of pesticide use in apple production 76
5.4.1 A regional analysis of pesticide use 78
5.5 Discussion 82

CHAPTER 6. A PESTICIDE RATING INDEX FOR ASSESSING ENVIRONMENTAL IMPACTS: METHODS AND RESULTS. 83

6.1 Including environmental parameters in the decision making process 83
6.2 Methods 83
6.2.1 The EIQ model 83
6.2.2 Data collection 86
6.3 Results 88
6.4 Discussion 95

CHAPTER 7. MODIFYING THE EIQ MODEL; ADDRESSING THE CRITICISMS. 99

7.1 Introduction 99
7.2. The EIQ damage function 100
7.2.1. Pesticide toxicity and LD_{50} 102

7.2.1.2 Methods 105
7.2.1.3 Results: Mammals 105
7.2.1.4 Results: Birds 106
7.2.1.5 Insects 107
7.2.1.6 Non target flora 110
7.2.1.7 Conclusions on the EIQ damage function. 111
7.2.2 The weighting criteria 112
7.2.3 The averaging of impacts 118
7.2.4 Site specific data 120
7.2.4.1 Soil 121
7.2.4.2 Proximity to water source 121
7.2.4.3 Application method 123
7.2.5 Data presentation method 124
7.2.5.1 Modified EIQ values for fungicides and herbicides: a comparison of existing and modified EIQ values. 124
7.2.5.2 An alternative data presentation format for the Kovach et al., (1992) model. 127
7.2.5.3 Data presentation 129
7.3 Table 7.16 interpretation 133
7.4 Discussion 135

CHAPTER 8. CONCLUSIONS. 137

8.1 Model modifications 137
8.2 Suggestions for future research 139
8.3 Summary 141

Reference List 143

Appendix 1. EIQ Field Use Rating by Country 158
Appendix 2 Production characteristics of existing and new apple varieties, 167
Appendix 3. EIQ tables from Kovach et al., 1992. 184
Appendix 4. Modified EIQ tables. 185
Appendix 5. Statistical analysis 198

List of Abbreviations

A-M	Atomistic-Mechanistic
A.I. (a.i.)	Active Ingredient
CAP	Common Agricultural Policy
CBA	Cost Benefit Analysis
CEC	Commission of the European Communities
CPP	Crop Protection Products
CVM	Contingent Valuation Method
DEAC	Development of the European Apple Crop
ECU	European Currency Unit
ED_{50}	Effective Dose 50
EEC	European Economic Community
EEP	Environmental Exposure Points
EIP	Environmental Impact Points
EIQ	Environmental Impact Quotient
EPA	Environment Protection Agency
ESA's	Environmentally Sensitive Areas
EU	European Union
EXTONET	Extension Toxicology Network
FEAP	Fifth Environmental Action Programme
FUR	Field Use Rating
GP	Goal Programming
ICM	Integrated Crop Management
IFP	Integrated Fruit Production
IPM	Integrated Pest Management
Kg/Ha	Kilograms per Hectare
LD_{50}	Lethal Dose 50
LEAF	Linking Environment and Farming
LGP	Lexicographic Goal Programming
LP	Linear Programming
MCDM	Multi Criteria Decision Making
MYCPP	Multi-Year Crop Protection Plan
NUTS	Nomenclature Units of Territorial Statistics
PI	Pesticide Index
PRI	Pesticide Residues Index
SAC	Scottish Agricultural College
SAV	Submerged Aquatic Vegetation
UK	United Kingdom
USA	United States of America
VI	Value Index
WGP	Weighted Goal Programming
WHO	World Health Organisation
WTP	Willingness To Pay

List of Tables

Table 2.1 Pesticide reduction targets for The Netherlands. Denmark and Sweden. 26
Table 3.1 The expansion of world apple production 1948-1994. 30
Table 3.2 World apple production by region and country. 32
Table 3.3 Apple production in the EU, 1994. 33
Table 3.4 Areas under apple cultivation, 1994. 34
Table 3.5 Quantity produced and market share of the main apple varieties, 1994. 36
Table 4.1 The social and environmnetal cost of pesticide use in the USA 46
Table 4.2 Pesticide risk indices included in the literature review 49
Table 4.3a Evaluation matrix for 20 risk indices 52
Table 4.3b Evaluation matrix for 20 risk indices 53
Table 4.4 EIQ field use rating for three insecticides 57
Table 4.5 A comparison based on environmnetal impact of 6 commonly used pesticides in potato production in the Netherlands. 61
Table 4.6 An example of the pesticide index 64
Table 5.1 The global pesticide market in 1994. 77
Table 5.2 EU pesticide market by country in million ECU (1991). 79
Table 5.3 Annual pesticide use in the EU. 79
Table 5.4 Summary of pesticide use in selected European apple growing regions for the year 1994. 80
Table 5.5 Pesticide use for selected apple growing regions of Europe, 1994. 81
Table 6.1 EIQ for the fungicide captan 84
Table 6.2 Pesticide use and EIQ for France. 89
Table 6.3 Pesticide use and EIQ for the UK. 89
Table 6.4 Theoretical comparison between pest management strategies. 90
Table 6.5 EIQ values at current use rates in the European apple industry. 92
Table 6.6 EIQ Field Use Rating by country. 93
Table 6.7 EIQ values for existing and new varieties. 93
Table 7.1 The EIQ Field Use Rating for the pesticide triadmefon. 100
Table 7.2 The variability of the LD_{50} for the dioxin 2,3,7,8-TCDD. 103
Table 7.3 Different calculations of a suitable dose of LSD for elephants. 103
Table 7.4 Rating criteria of the EIQ equation. 114
Table 7.5 Weighting factors of the EIQ equation. 115
Table 7.6 Alternate ranking system for insecticides, fungicides and herbicides. 117
Table 7.7 The impact categories of the chemical scoring system for 119 hazard identification and exposure.
Table 7.8 Impact categories and scores of the environmental yardstick. 119
Table 7.9 Impact categories for the environmental exposure to pesticides system. 119
Table 7.10 Current data presentation format for the Kovach et al., (1992) EIQ model. 120
Table 7.11 The preferred method of data presentation. 120

Table 7.12 Application method and emission percentage. 123
Table 7.13 A comparison of old and new EIQ values for herbicides. 125
Table 7.14 A comparison of old and new EIQ values for fungicides. 126
Table 7.15 Impact categories and EIQ for farmworkers and consumers. 128
Table 7.16 How the EIQ model might be used in practice: data from Portugal. 134
Table 7.17 Site specific impact ratings. 135
Table 7.18 Impact categories 135
Table 7.19 Site specific variables relating to Table 8.16, Portugal. 135

List of Figures

Figure 1.1 A summary of the key issues and problems presented in the thesis. 2
Figure 1.2 An overview of thesis structure 6
Figure 2.1 Factors affecting the valuation and identification of externalities. 10
Figure 2.2 Property rights as a triadic social relationship. 12
Figure 4.1 Comparing the environmnetal impact of three pesticides 58
Figure 4.2 The pesticide index system 63
Figure 5.1 Factors involved in the choice of pest management strategy. 71
Figure 5.2 An overview of the current legislative situation with regard to pesticide use in the EU: the main legislative instruments, 72
Figure 6.1 EIQ Field Use Rating of capstan and lorsban. 85
Figure 6.2 Summary of the EU EIQ's for existing and new varieties. 92
Figure 6.3 Comparison of EIQ values between regions 95
Figure 6.4 Total EIQ values for France and the UK showing theoretical 97
Figure 7.1 The damage function for the pesticide triadmefon. 101
Figure 7.2 The typical relationship between mortality of organisms and exposure to pesticides under laboratory conditions. 104
Figure 7.3. Dose response curves for essential compounds 105
Figure 7.4 The effect of increased doses of crimidine on voles. 106
Figure 7.5 The dose response curve for the pesticide flocoumafen for hens. 107
Figure 7.6a. 108
and b. Lethal and sub-lethal effects of dose of Margosan-O on pea aphid. 109
Figure 7.7 The effects of increased doses of pyrazophos on carabid beetles. 109
Figure 7.8 Dose response curve for sterility in the Bollworm and Medfly. 110
Figure 7.9 Total damage function from atrazine use around Chesapeake Bay. 111

1. Introduction

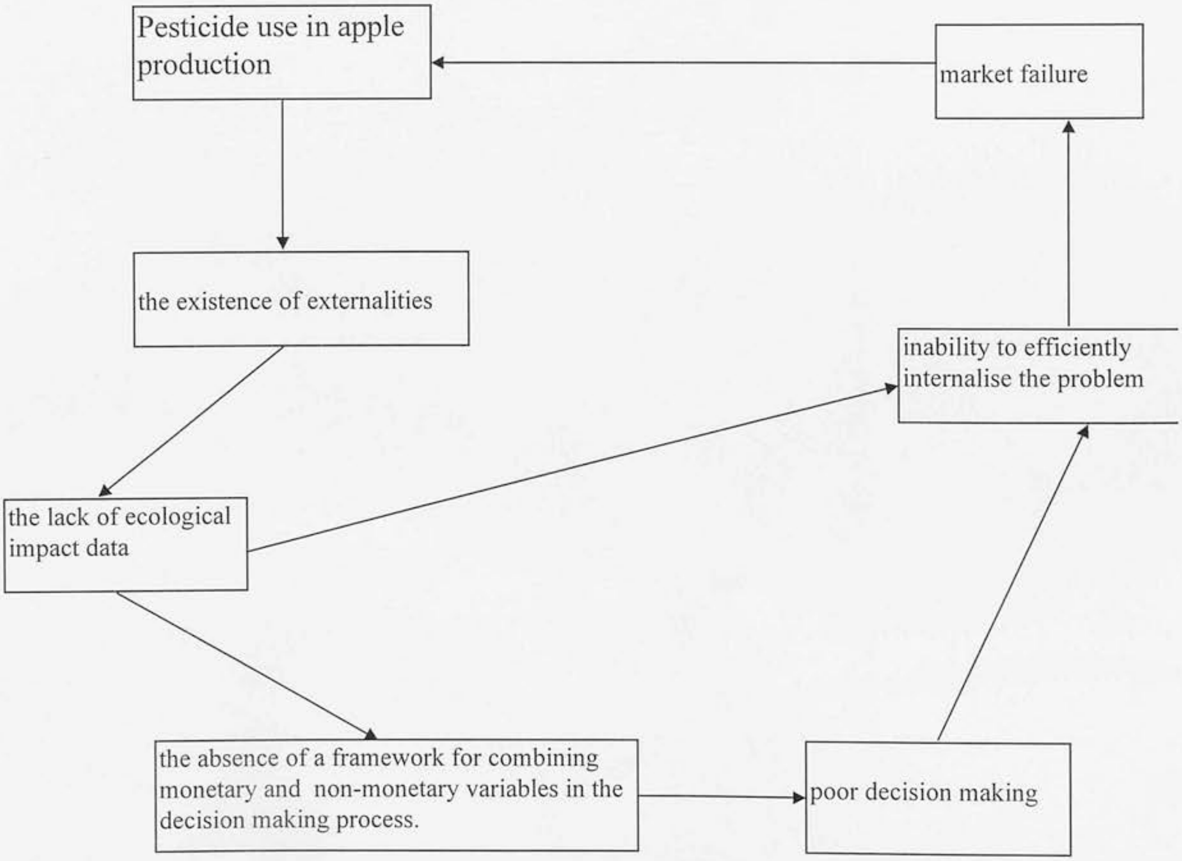
This thesis was undertaken alongside a European Union (EU) funded project entitled *The Development of The European Apple Crop* (DEAC). The EU project was a plant breeding project, the main objective of which was to reduce the amount of pesticides used in the production of apples through the introduction of new disease resistant apple varieties (King et al., 1991, Quin and McGregor, 1995, Quin et al., 1996). This project was in line with many other EU projects and policies, which in general aim to reduce the amount of agro-chemicals used in modern European agriculture. One interesting question related to this aim is to what extent can such reductions in pesticide use be quantified in terms of their environmental impact, and to what extent can this information be incorporated into the decision making arena? This thesis explores a framework for answering such questions.

It is not currently possible to know enough about pesticide contamination and the environment, to be absolutely certain of its safety (Wauchope, 1978, Levitan, 1997). The use of pesticides in agriculture is currently high on the political agenda (Beaumont, 1993, Penrose et al., 1994). A 1991 survey in the UK showed that 74% of respondents thought that some chemical residues on food were dangerous to health (Penrose et al., 1994). Both the EU, and some individual Member States, have acted to attempt to cut, often drastically, the quantity of pesticides to be used by the year 2000 (Oskam et al., 1992, Beaumont, 1993, Green and Mumford, 1995). Quantity reductions in pesticide use, however, are not necessarily the answer to the real or perceived problems (Reus and Pak, 1993, Quin et al., 1997), as old pesticides may be replaced by newer, more efficient, more concentrated formulations requiring much reduced quantities to achieve the same *kill* rate (Pearce and Tinch, 1997). Therefore, a methodology is required that can quantify the implications of pesticide reduction policies by examining individual compounds in terms of the environmental impact per unit of percentage active ingredient (strength) as well as overall quantities used.

1.1 Decision making and the environment

In the absence of environmental impact data it is very difficult to incorporate non-market goods into the decision making process (Cumberland and Kahn, 1982, Kahn and Kemp, 1985, Kahn, 1987, Bahr, 1992, Bergman and Pugh, 1994, Quin et al., 1996, Quin et al., 1997, Pearce and Tinch, 1997). The purpose of this thesis is to examine the feasibility of including environmnetal considerations in the decision making process with regards to pesticide use. When addressing such a purpose it is necessary to consider the interrelation between pesticide usage, market failure, ecological data and decision making structures (Figure 1.1). When considering these interrelations, the key question becomes: to what extent can pesticide impact models be developed in order that they could be used to identify the environmental impact from pesticide use in the apple industry?

Figure 1.1 A summary of the interrelationships between key issues and problems presented in the thesis.



[illegible] walls.

[illegible] will be [illegible] in the [illegible] with [illegible] (1997). This [illegible] by [illegible]

[illegible]

[illegible]

[illegible]

[illegible]

The [illegible] of this paper are,

[illegible] can be [illegible] and [illegible] considered.

[illegible] the environmental [illegible]

1.2 Pesticides and externalities

The potential environmental impacts of pesticide use are well documented, and it is widely accepted that they pose (to some degree) a direct toxicological risks to humans, animals and plants (Meyer, 1993). The potential toxic risks incurred by the use of pesticides could be summarised under the following headings:

1. Health risks to farm workers, particularly those involved in pesticide application.
2. Health risks to the public through groundwater contamination, spray drift and to the consumer through food residues.
3. Damage to non-target flora.
4. Damage to non-target fauna.

Although classifying potential impacts of pesticides into these four categories is relatively uncontroversial, there is lively discussion as to the extent of the potential harm caused by pesticide use (Green and Mumford, 1995, Ramirez and Mumford, 1995). This thesis will not address this argument, rather it adopts the stance that the excessive use of pesticides is a form of public bad (Carlson et al., 1993) and should therefore be reduced wherever possible.

1.3 Objectives

The objectives of this thesis are:

A. To examine pesticide ranking indices in order to select one model which may adequately describe the potential environmnetal impacts associated with pesticide use.

B. To investigate patterns of pesticide use at a regional level for selected apple producing regions throughout the EU, in order that environmental impact data can be applied to actual pesticide use rates, to allow for environmental impact comparisons to be made between regions.

C. To develop an alternative method of presenting environmental impact data as a means of improving the quality of information available to farmers and policy makers and others involved in pesticide use and regulation.

In order to examine and explain the main issues of the thesis it was necessary to examine the theory surrounding externalities, crop protection, and environmental impact models. It was also necessary to build up a picture of pesticide use throughout the EU, at a regional level, through the gathering of survey data from Universities, Agricultural Ministries, Marketing organisations, and Advisory organisations and through the collection of primary data in the form of orchard surveys.

Chapter 2 introduces the key concepts of crop protection and externality theory. It highlights the main issues in the use of crop protection products, and discusses crop protection products as an externality problem. It presents the theoretical background against which the thesis is presented. Chapter 3 introduces the European apple industry, and examines the apple market, its output, value of product, and puts it in the context of overall EU agricultural output. Chapter 4 consists of a critical review of existing pesticide rating indices, often used in conjunction with Integrated Pest Management[1] (IPM) practices, and attempts to highlight the strengths and weaknesses of each methodology in a policy / decision making context. Chapter 5 looks at trends in pesticide use in the production of apple, together with the current legislative situation with regard to pesticide use. It also comments on the dichotomy that exists between the perceived costs and benefits of pesticide use in general. Chapter 6 examines one particular pesticide ranking model and applies it to pesticide use data from the apple industry in order to ascertain whether or not realistic results can be achieved. Chapter 7 re-assesses the strengths and weaknesses of the pesticide ranking indices approach to the examination of agricultural pollution from

---

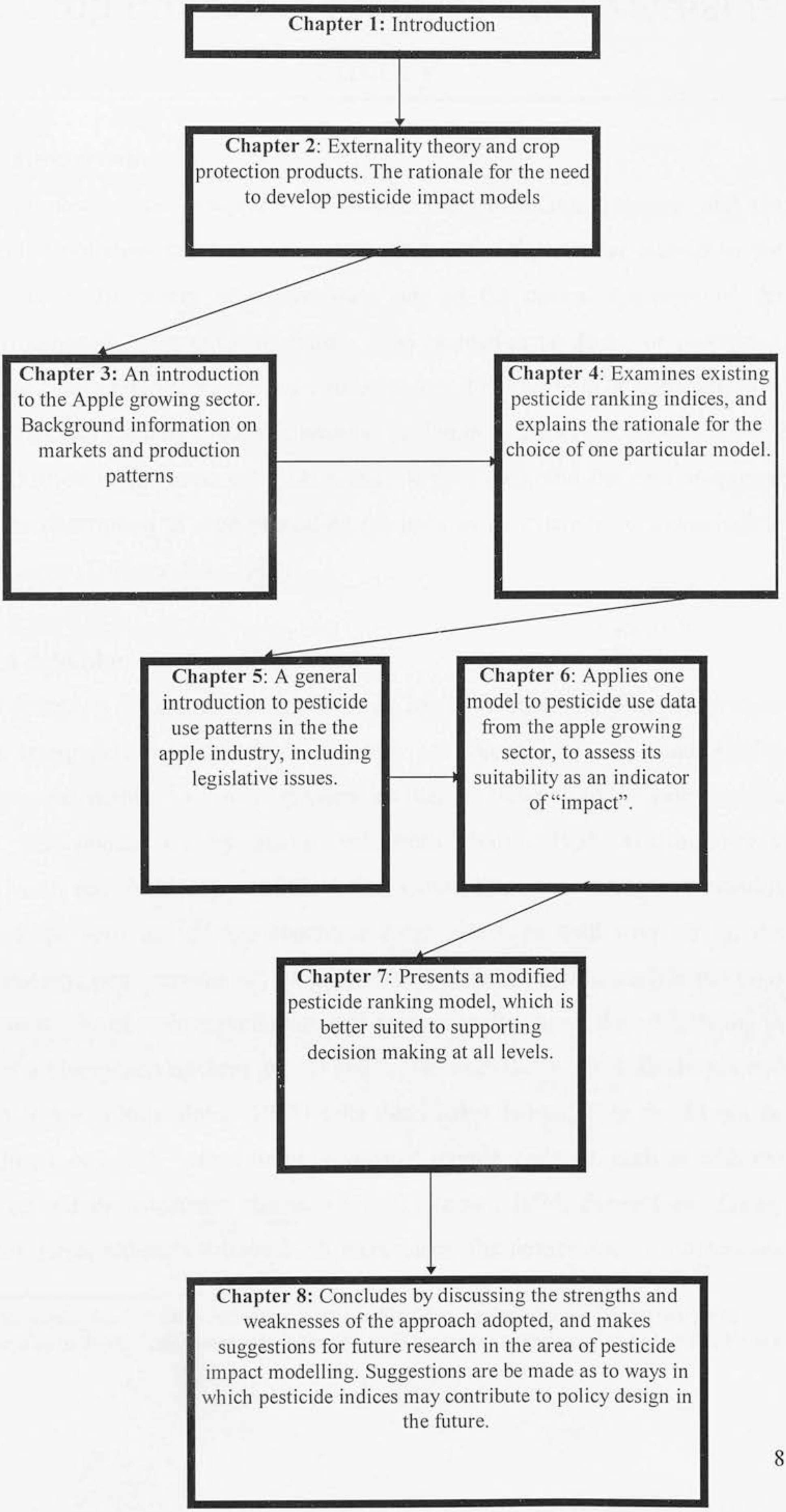
[1] Integrated Pest Management (IPM) is a decision making process that utilises regular monitoring to determine if and when treatments are necessary, and employs physical, mechanical, cultural, biological, educational and chemical tactics to keep pest numbers low

pesticide use, and proposes one modified methodology for use in decision making models, and / or use at the farm decision making level. Chapter 8 concludes by discussing areas where future research might benefit the understanding of the development of environmental-economic models, and in general the incorporation of environmental considerations into the decision making process. For ease of further reference, Figure 1.2 summarises the structure of the thesis and main contents of each chapter.

---

enough to prevent intolerable damage or annoyance. Least-toxic chemical control should be used only as a last resort (Olkowski et al., 1991).

Figure 1.2. An overview of the thesis structure.



2. Crop Protection Products and Externality Theory

2.1 Introduction

The purpose of this chapter is to discuss crop protection products and the potential pollution that may occur from their use. Of particular interest to the analysis is the theory of externalities, one of the central theorems of the Environmental Economics discipline. Crop protection products, or pesticides, will be discussed in relation to externality theory, together with an explanation of how and why socially excessive levels of polluting activity might occur. A brief introduction to Environmental Economics is presented, and the case supported for the description of crop protection products as an example of technological externality (Carlson et al., 1993).

2.2 A definition of externalities.

The definition of externalities is on the one hand straightforward, and yet fraught with interpretational difficulties (Hartwick and Olewiler, 1986). Bator (1958) defines externalities[2] as interdependencies that are external to the price system and unaccounted for by market valuations (Bator, 1958, Griffin, 1991). Buchanan and Stubblebine (1962) define externalities as existing in situations where the activities of one economic agent affect, or spill over on to, the technology, consumption set, or preferences of another. This implies the non-independence of various preference and production functions, the effect being to cause a divergence between private and social cost (Bator, 1958, Buchanan and Stubblebine, 1962). Bator (1958) calls this market failure. This should not be confused, however, with a simple economic interdependence, such as with the farmer and the consumer (Bator, 1958, Weitzman, 1974, Baumol and Oates, 1988). Here, although a transaction takes place, the farmer does not determine

---

[2] Bator sees technological externalities as real externalities, seeing pecuniary externalities as superficial to the analysis. Indeed since Bator (1958) pecuniary externalities have become known

the consumption patterns of the consumer, nor does the consumption of the consumer directly effect the farmers utility function. Of course payment for the farmers' goods affects his utility function, but this brings in the distinction between pecuniary and technological externalities (Baumol and Oates, 1988).

Some have argued, however, that the definition of externalities proposed by Bator (1958) is too broad (Cropper and Oates, 1992). Bator (1958) included the case of increasing returns to scale with natural monopoly under his definition of externality (Cropper and Oates, 1992). This, it is argued, is not what most people are thinking about when externalities are discussed (Baumol and Oates, 1988). The Pareto relevant externality, proposed by Buchanan and Stubblebine (1962) and Baumol and Oates (1988), is to what most of the literature refers when discussing externalities. The Baumol and Oates (1988) approach is to define externalities not by what externalities are, but by what externalities do. That is that a (Pareto relevant) externality is present when, in competitive equilibrium, the marginal conditions for optimal resource allocation are violated (market failure) (Baumol and Oates, 1988, Cropper and Oates, 1992). This, however, still does not inform us how the resource allocation equilibrium is violated.

Mishan (1969), Fisher and Peterson (1976) and Baumol and Oates (1988), contest that two conditions must hold true if the resource misallocation of Buchanan and Stubblebine (1962) is to occur[3]. Many definitions of externalities cover the same ground, the most appropriate, however, is that proposed by Baumol and Oates (1988), from Mishan (1969), which has been widely used in the subsequent environmental economics literature. For an externality to exist the two conditions that must be satisfied are:

Condition 1: An externality is present whenever some individual's (A's) utility or production relationships include real (non-monetary) variables, whose values

---

as pseudo externalities as no resource misallocation need occur to signify their presence (Baumel and Oates, 1988).

are chosen by others (individuals, organisations, Governments) without particular attention to the effect on A's welfare. As Mishan (1971) points out this definition excludes cases where somebody deliberately does something to effect somebody else's welfare[4]. As will be shown later the most important external effects are those which affect a large number of individuals (Johansson, 1987). For a relationship to qualify as an externality, it is argued that a second condition must hold true.

Condition 2: The decision maker, whose activities affects the utility levels or enter into the production functions of others, does not receive (or pay) compensation for this activity, an amount being equal in value to the benefits (or costs) to others. Baumol and Oates (1988) contend that it is this condition that must occur if all of the unpleasant consequences that are associated with the concept of externality, such as resource misallocation, are to occur.

A real paradox emerges when examining externalities in that the objective is to seek to identify the social costs associated with an activity, and reduce them or internalise them. Yet those individuals generating an external cost are also members of society, thus, as Neher (1994) points out, it is as if people require salvation from themselves. It is the identification of the social costs of pesticide use that are of most concern in this study, and the relationship between pesticide use in the apple growing sector and externalities is illustrated in Figure 2.1.

2.2.1 Property Rights

The source of an externality is typically found in the absence of well defined property rights (Coase, 1960, Fisher and Peterson, 1976, Hartwick and Olewiler, 1986, Baumol and Oates, 1988, Cropper and Oates, 1992, Dales, 1992). A resource which is owned by nobody, but open to all, is usually open to misuse

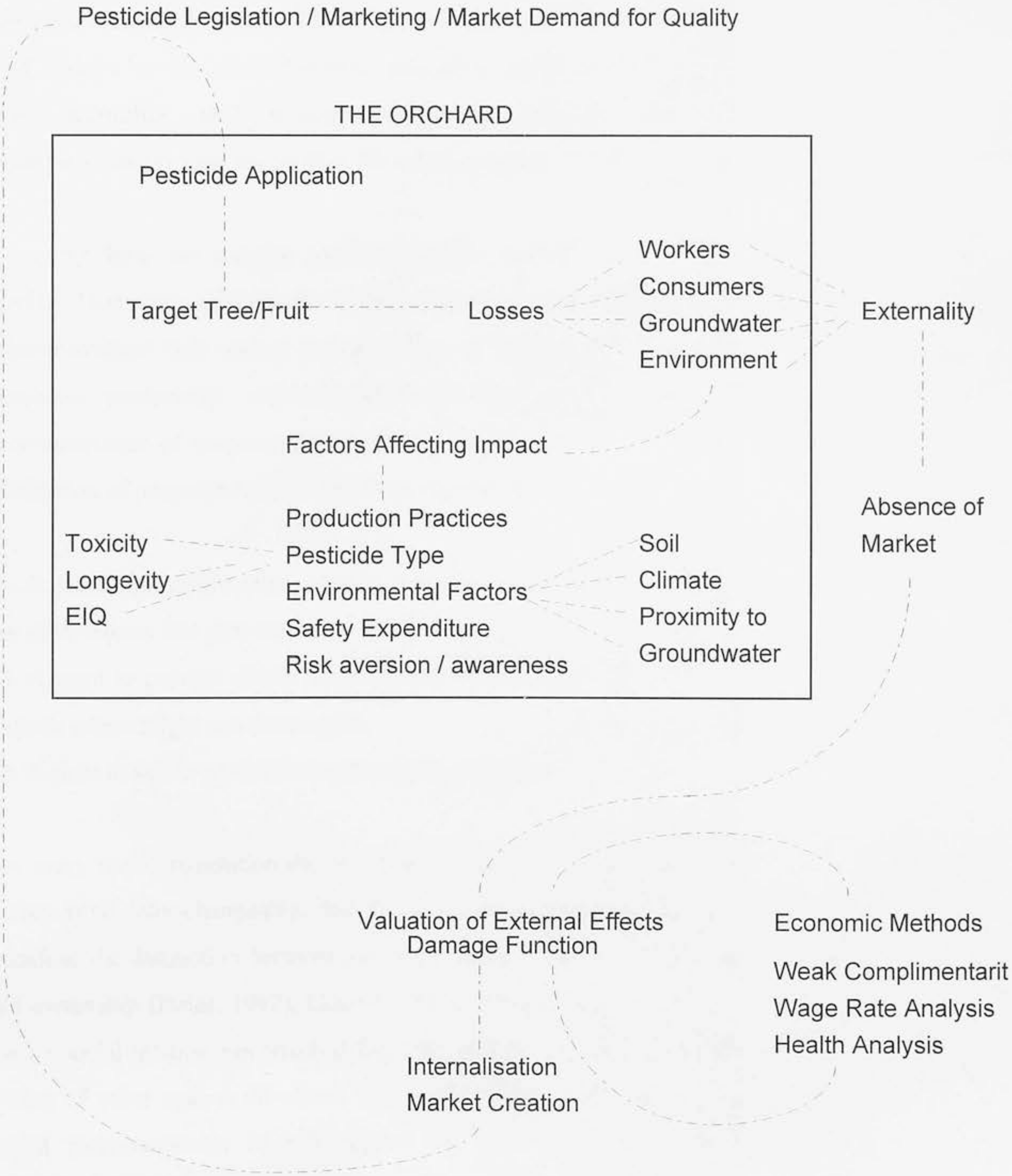
---

[3] Although Mishan (1971), contests that a third condition must also be fulfilled if an externality is to exist, and that is that the impact, or pollution, must be unintentional and unforeseen by either the producer or consumer of the externality.

[4] Mishan (1971) argues that if somebody deliberately pollutes water, or deliberately makes a loud noise to annoy a neighbour, then no externality exists. He contends that for an externality to exist the impact is always an unintentional product of some legitimate activity.

(Hardin, 1969, Tiwari and Quin, 1997). Property rights can either be totally absent, ill-defined or unenforceable, and all of these scenarios may well lead to a situation where the resources contained therein are allocated inefficiently.

Figure 2.1. Factors affecting the identification and valuation of externalities: a simplified model of pesticide use in apples production.



The assumption here is that if property rights are introduced or established, then the externality will be resolved and cease to exist (Coase, 1960). Coase (1960) adds that irrespective of who owns the property rights in a resource conflict situation, the outcome (the socially optimal solution) will always be the same. It will always be the same, however, only when bargaining takes place between those in conflict, and this in turn can only take place when there are only a small number of individuals involved in the conflict (Coase, 1960).

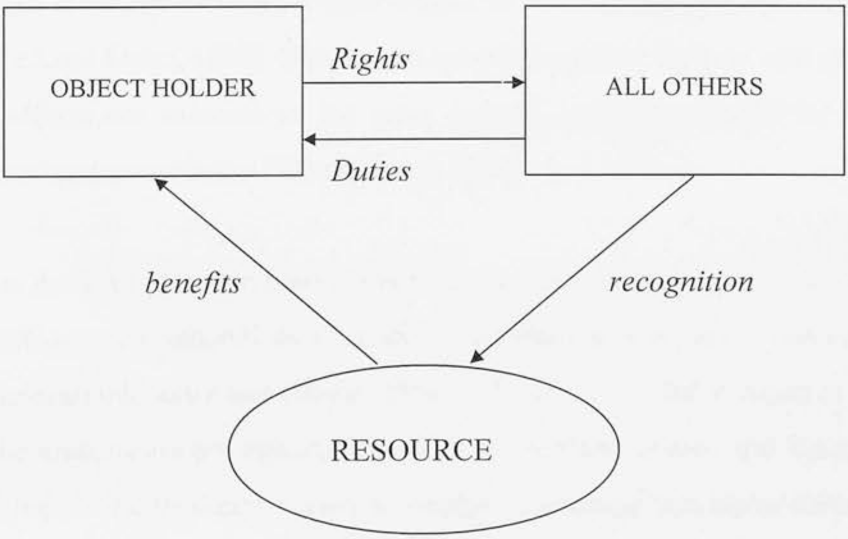
Property rights are a major source of market failure (Hartwick and Olewiler, 1986). Hartwick and Olewiler (1986) define property rights as a "bundle of characteristics" that convey certain powers to the owners of those rights. They mention exclusivity, enforceability and transferability to be the main characteristics of property rights. This is in concordance with Dales' (1992) definition of property rights in which ownership always consists of:

1. A set of rights to use the property in certain ways (and a set of negative rights or prohibitions that prevent its use in other ways).
2. A right to prevent others from exercising those rights or to set the terms on which others might use those rights.
3. A right to sell or re-distribute those property rights.

In every day conversation the concepts of "property" and "property rights" are often used inter-changeably, but the contraction is misleading if it tends to confuse the distinction between viewing property as things rather than as rights of ownership (Dales, 1992). Dales (1992) asserts that property is not an object, but a social relationship which defines the property holder with respect to some thing of value against all others. The terms "rights" and "rules" are frequently used interchangeably in referring to the uses made of natural resources. However, rights are the product of rules. Rights refer to particular actions that are authorised, and rules refer to the prescriptions that create authorisation (Schlager and Ostrom, 1992, Tiwari and Quin, 1997). To possess the right to exploit a resource implies that someone else has a commensurate duty to observe those rights, therefore, rules specify not only rights but also duties.

Figure 2.2. summarises the relationship between property holders and others. So far as exclusivity is concerned in the case of natural resources and environmental goods, exclusive possession is one extreme of a continuum of property rights, and no possession (in the case of pure public goods) is the other. Property rights issues are closely linked with social problems, and two important issues are proposed by Dales (1992). Firstly that there are no perfect legal solutions to social problems, any more than there are perfect economic solutions, and secondly, that a given legal definition of property rights has not only economic consequences, but also social, environmental and political consequences.

Figure 2.2. Property rights as a triadic social relation.



For the sake of simplicity, property rights are usually catagorised as either private property or common property, but four clear categories of property rights are identifiable. They are:

a) private property,

b) state property,

c) common property, and

d) open access property.

(Feeney et al., 1990).

2.2.2 Market Failure

If property rights can be enforced when they are in place, or created where they are absent, then externalities can in theory be eliminated (Coase, 1960). Most externalities related to the use of natural resources involve no exchange through the market leading to sub-optimal resource allocations. This situation arises due to the divergence between social and private costs and is termed "market failure" (Bator, 1958, Coase, 1960, Weitzman, 1974, Fisher and Peterson, 1976, Hartwick and Olewiler, 1986, Johansson, 1987, Pezzey, 1988, Pearce and Turner, 1990).

The concept of market failure is closely linked to that of externalities, indeed if there were no failure of the market there would be no externalities (Johansson, 1987, Baumol and Oates, 1988). This is because public goods, or bads, and their associated effects are external to the price system, and unaccounted for by market valuation, hence market failure (Bator, 1958).

What then is market failure? In theory it is the failure of a more or less idealised system of price-market institutions to sustain desirable activities, and to prevent or reduce undesirable activities (Bator, 1958). Markets have failed when it is clear that the markets are not maximising collective welfare (Pearce and Turner, 1990). It is the central theorem of modern welfare economics[5] that under certain strong assumptions about technology, tastes and producers motivations, the

---

[5] The so called Duality theorem. That is the correspondence of Pareto optimality to a perfectly operating market (Bator, 1958).

equilibrium conditions which characterise a system of competitive markets will necessarily correspond to the requirements of Paretian efficiency[6].

Many factors in the real world violate the correspondence of Pareto efficiency and a perfectly functioning market, such as imperfect information, uncertainty, resistance to change, and poorly defined or non-existent property rights necessitating some form of Government intervention (Baumol and Oates, 1988). Market failure encompasses not only economic concerns, but also social and institutional considerations (Coase, 1960). In the presence of externalities the outcome of this "violation" is to cause the divergence between private and social cost (Bator, 1958, Coase, 1960, Weitzman, 1974, Fisher and Peterson, 1976, Hartwick and Olewiler, 1986, Johansson, 1987, Pezzey, 1988, Pearce and Turner, 1990).

2.2.3. Public goods and externalities.

There is a close correspondence between public goods and externalities (Johansson, 1987). Indeed it is reasonable to view a public good as an externality in consumption (Johansson, 1987, Baumol and Oates, 1988). There are two main characteristics that distinguish pure public goods from private goods (private goods will be examined in more detail later). Firstly, the same unit of a public good can be consumed by many, and secondly, once a public good is provided for some individuals, it is impossible, or at least very costly, to exclude others from benefiting from it (Hartwick and Olewiler, 1986, Pearce and Turner, 1990, Kula, 1994, Neher, 1994). Public goods equate to the "collective goods" of Samuelson, (1958). According to Samuelson (1958) (cited in Johannson, 1987) a collective good is a good which "all can enjoy in common in the sense that each individuals consumption of such a good leads to no subtraction from any other individuals enjoyment of that same good", and also such goods simultaneously enter into the indifference curves of many.

---

[6] A community is on its Paretian frontier if it is impossible to make anybody better off without making somebody else worse off. The concept of Pareto optimality stems from the work of economist Wilfredo Pareto (Just, 1984).

If a lake or ocean is polluted, it is polluted for all individuals in the area and not just one individual (Hartwick and Olewiler, 1986). Therefore water pollution, air pollution and noise pollution are examples of externalities. It is important to remember that externalities can confer positive benefits, in the form of a public good, on consumers and not just negative effects. A public good could be a landscaped garden, as it yields an externality which confers benefits on all viewers of the garden (Cropper and Oates, 1992).

It is accepted that when a public good is identified then the ordinary price system is unable to provide an efficient outcome to the problem (Weitzman, 1974, Johansson, 1987). The basic source of this problem is in the "undepletable" nature of public goods (Samuelson, 1958, Hartwick and Olewiler, 1986, Johansson, 1987, Pezzey, 1988, Cropper and Oates, 1992). Basically, the concept of undepletability applies to public goods because an increase in the consumption of the good by an individual, say A, does not reduce its availability to individuals B, C, D, and so on. The inhalation of polluted air by one individual, does not reduce the quantity available to be inhaled by others (Samuelson, 1958, Hartwick and Olewiler, 1986). Also, the viewing of the landscaped garden by one does not detract from the pleasure gained from another, as long as there is no congestion (Baumol and Oates, 1988)[7].

2.2.4. Private goods and externalities.

It is easy to think of many examples of externalities displaying the characteristics of public goods, such as polluted water, polluted air, noise, a littered city, or indeed the beautifully landscaped garden. More problematic, however, is identifying good examples of "private" or "depletable" externalities (Baumol and Oates, 1988, Cropper and Oates, 1992). Indeed Baumol and Oates (1988) maintain that "it is not easy to provide a convincing example of a depletable (private) externality" (page 20). The example of a depletable externality put

---

[7] It is well accepted that it is inefficient to charge for public goods or bads because the consumption of one individual does not affect the utility or disutility of others. A positive price on a public good for example may actually inhibit use, thereby reducing an individual's utility without increasing that of others.

forward by Baumol and Oates (1988) goes back to conditions in post World War Two Europe. Following the War there were severe shortages of fuel, and it was reported that in many areas of Europe quite large numbers of people spent a great deal of time walking along railroad tracks looking for coal that had been dropped by passing coal trains. It is clear that this is the case of a depletable externality, because for every piece of coal picked up by one gatherer less is available to the next gatherer to come along.

The reason that the coal was left on the tracks was presumably that it was too costly and time consuming for the operators of the trains to stop and clean up after each passage. In principle if there were enough money to be made in doing this then the train operators would have done so. Presumably then the externality was insignificant, and/or the cost of collecting the appropriate fee for the externality must have been excessively high (Baumol and Oates, 1988).

2.2.5. A note on pecuniary externalities

Because there need be no resource misallocation with pecuniary[8] externalities they have become known by some as "pseudo" externalities, and therefore irrelevant to the externality debate (Bator, 1958, Baumol and Oates, 1988, Cropper and Oates, 1992). With pecuniary externalities, one individual's activities affects the financial circumstances of another, or others, but again, there need be no resource misallocation. Pecuniary externalities will not be examined further in this thesis, and the term externality will refer only to technological externality unless otherwise stated.

2.3 Environmental Economics.

As Cropper and Oates (1992, pp 675) succinctly stated:

"When the environmental revolution arrived in the late 1960's, the economics profession was ready and waiting".

---

[8] Pecuniary meaning monetary, financial or fiscal.

Economists had what they saw as a coherent set of solutions to the problems of pollution and its associated policy implications (Cropper and Oates, 1992, Bahr, 1992). The concept of externalities and the associated market failure had long been a part of microeconomics theory, indeed the study of external diseconomies, and economic solutions to environmental resource problems, dates back to the work of Arthur Pigou (1920) in the first half of the Twentieth Century. Even before Pigou (1920), however, the importance of environment and resources was well documented. Marshall speaks of external economies in the latter part of the Nineteenth Century. Malthus was concerned with growing populations and the inability of the resource base to cope as far back as 1798. Ricardo introduced the more useful notion of resource scarcity in 1817, and Jevons noted the impact of increasing resource costs and scarcity on the British economy of the nineteenth Century (Kula, 1994).

"Modern" environmental economics, however, could be argued to have begun with Pigou (1920) in the early part of the Twentieth Century. His theories and assertions were analysed and progressed by authors such as Bator (1958) and Samuelson (1958), but the vast and rapid expansion of the subject area began with the systematic economic analysis of alternative uses for natural resources by John V. Krutilla in his 1967 paper "*Conservation Reconsidered*" (Krutilla, 1967, Fisher and Peterson, 1976, Markandya and Richardson, 1992). The problem Krutilla addressed was that of providing for the present and future, the amenities associated with unspoiled natural environments, for which the market fails to make adequate provisions (Krutilla, 1967). He was specifically concerned with natural environments whose use for extractive purposes 1) precluded use for non-extractive purposes that also gave rise to value, and 2) was irreversible (Krutilla, 1967). Krutilla (1967) was also concerned with why there should be a divergence between social and private costs in the presence of externalities. The divergence between social and private costs in the presence of external diseconomies was first proposed by Pigou (1920), although he did not refer to it in those now familiar terms (Cropper and Oates, 1992, Bahr, 1992).

On the face of it the problem facing economists and policy makers is a simple one (Pezzey, 1988). Pollution is seen as the consequence of an absence of prices for certain scarce resources (such as clean air and water) (Hartwick and Olewiler, 1986), thus the firms discharging the pollutants to the environment should somehow be made to pay a price for such discharges related to the amount of environmental damage caused (Cropper and Oates, 1992, Baumol and Oates, 1988, Pezzey, 1988). The problem, however, is not that simple, and additional questions have to be addressed in understanding and regulating Man-environment interactions. As Western society became richer in the 1960's and 1970's the supply of produced goods, such as cars and televisions, was increasing compared to that of environmental goods, such as clean air, clean water and "unspoiled" wilderness areas (Krutilla, 1967, Fisher and Peterson, 1976). Environmental goods were becoming relatively scarce, but the demand for them was rising in relation to real increases in disposable income and leisure time. Since many environmental goods are inherently ones which cannot be produced, or reproduced by the private sector, there was a clear need for public sector regulation (Krutilla, 1967, Fisher and Peterson, 1976, Pearce and Turner, 1990). Governmental concern for the conservation of, and the "putting to better use" of, natural resources dates back a long way. One of the goals of the American Conservation Movement (1890-1920) was the preservation of natural environments in wilderness areas and culminated in the setting up of National Parks under John Muir's guidance (Markandya and Richardson, 1992). Government, however, is faced with answering three complex questions:

i) How much of the environment are we to protect ?,
ii) How much should we pay for protecting it ?
iii) What methods do we use to protect it ?
(Markandya and Richardson, 1992).

The foundations for answering those questions lay in the theory of welfare economics (and now Environmental Economics) and particularly the theory of external effects or externalities[9]. The economists view of the problem, however, had little impact on the initial surge of legislation (Fisher and Peterson, 1976, Pezzey, 1988, Hahn, 1989, Hanley et al., 1990, Cropper and Oates, 1992), that came particularly from the USA, for the control of pollution. As Fisher and Peterson (1976) light-heartedly put it, "it is enough to turn Pigou in his grave"! The reasons for the initial lack of impact of economics in the protection of the environment are manifold. Perhaps most importantly, the concept of environmental damage was both straight forward and yet elusive. For environmental protection policies to be economically efficient, the total economic value of environmental damage had to be known (Kahn and Kemp, 1985, Kahn, 1987, Hanley et al., 1990, Dubgaard, 1991, Pearce and Turner, 1990, Bergman and Pugh, 1994). Only then could the incremental benefits of pollution abatement, or the incremental costs of increased pollution, be known (Krupnick and Alicbusan, 1991). Only then could efficient mechanisms such as pollution charges (Pigovian taxes), or tradable pollution permits be introduced. The inflexibility of the mechanisms proposed by economists has also been cited as a reason for the slowness of their adoption, as has the refusal of economists to compromise on the actions to be taken (Hanley et al., 1990, Cropper and Oates, 1992). For instance, taxes are considered the most economically efficient method of internalising externalities and are the environmental economists most "popular" choice of instrument (Barde and Opschoor, 1994), and yet the more workable tax-standard amalgam of Baumol (1988) would have almost the identical effect (Cropper and Oates, 1992). This is the so called "second best" solution (Baumol and Oates, 1988).

Hanley et al., (1990) summarise the reasons why more is not made of economic instruments in environmental management as being:

---

[9] Environmental economics has been called by some *" simply externality theory dressed up in their Sunday best"* (Markandya and Richardson, 1992).

1. The practical problems of design, of permit markets and taxes, when pollutants are non-uniformly mixed, often non-source point, and impacts are uncertain.
2. Institutional problems as a result of high transaction costs and a shift away from current legislative instruments.
3. Distributional issues, such as the effect of pollution on the poor.
4. Political issues, such as the moral or ethical acceptability of "selling the right to pollute".
5. The need for a vast amount of information, market efficiency, and a strong legal and policing capacity, before economic instruments can be used effectively.

The dichotomy that exists between the nature of environmental systems and the nature of economic systems also poses considerable problems for some. The basic assumptions of neoclassical economics (the atomistic-mechanistic model) do not fit in well with the complex, unpredictable, irreversible view of the natural world proposed by, amongst others, Norgaard (1981 and 1985), Daly (1992), and Birch and Gafni (1993). Indeed as far back as 1966, concerns were being voiced over the dichotomy that existed between the strict predictable laws of economics and the complex unpredictability of "Spaceship Earth" (Boulding, 1992).

There has been a general unwillingness in the past on the part of decision makers to place a monetary value on what are essentially non-market goods, such as the amenity value of a landscape (Bockstael and Kling, 1988), or the value of human life (Mann, 1982). This stems partly from the often perceived unreliability of the results given by the valuation methodologies developed by economists, but also in the inherent tastelessness of the task (Hanley et al., 1990).

Valuation methodologies have become more sophisticated over the last thirty years (Pearce and Turner, 1990, Hanley and Spash, 1993). In 1976 Fisher and Peterson reported that the results from the wide variety of techniques available to economists and policy makers should be "taken with a pinch of salt". One school of thought today, however, would now claim that the methodologies developed, such as the Contingent Valuation Method (CVM)[10], have been able to elicit apparently reliable answers to questions involving the valuation of improvements to the environment, and its associated impact on welfare (Cropper and Oates, 1992, Baumol and Oates, 1988, Pearce and Turner, 1990, Hanley and Spash, 1993). Some would argue, however, that the theoretical structure underlying environmental economics often emphasises elegance at the expense of realism (Hahn, 1989, Norgaard, 1985). It is this elegance at the expense of realism that has lead some to question the very foundation of environmental economics and the relevance of neo-classical models to complex environmental problems[11] (Norgaard, 1991, 1985, Daly, 1992, Birch and Gafni, 1993).

2.4 Crop Protection Products

Crop protection products (CPP) are used in modern European agriculture to control the various pests, diseases and weeds that affect crop production and crop quality (Hull, 1994). Although they may pose important risks for man and the environment, they are indispensable for an efficient agricultural sector, and as such contribute directly to the health and well-being of World populations (Scharpe, 1994, Scharpe and Smeets, 1994, Scheele, 1994, Vlahodimos, 1994, Johnen, 1995).

On a global basis it has been argued that the benefits of CPP can be summarised as: a) allowing the production of food calories to double since 1960, b) allowing the total area under cultivation to remain constant at 1.4 billion hectares since the 1950's, despite a doubling of the population, c) allowing per capita food supplies in Developing Countries to increase by 25%, and d) allowing for spectacular increases in food quality (Vlahodimos, 1994). It is further argued that intensive agriculture based on the efficient use of CPP has resulted in the conservation of

---

[10] A methodology for eliciting monetary values for non-market goods, based on questionnaires, the creation of hypothetical markets, and the identification of individuals Willingness To Pay (WTP), or Willingness To Accept (WTA), for environmental improvements (WTP), or environmental degradation (WTA).

[11] Birch and Gafni (1993) go so far as to call the use of economic instruments to solve environmental problems "the (mis)application of economics to real world situations".

26 million square kilometres of wildlife habitat since the 1950's (Vlahodimos, 1994, Johnen, 1995).

The benefits of the use of pesticides are readily measurable. Lever (1990) and Pimentel (1991) have suggested that the average benefit to the user of pesticides is worth at least three times the cost of treatment. The benefits associated with pesticide use are measured by the value of crop that would otherwise have been lost if pesticides had not been used (Oskam et al., 1992, Beaumont, 1993, Pearce and Tinch, 1997).

There are a number of reasons for the current patterns of pesticide use in the apple industry. They include yield and quality increases that have further increased the financial rewards relating to pesticide use (Lever, 1990), pesticide prices are low in relation to other fixed and variable costs, such as labour and loan repayments (O'Rourke, 1994), the market demands a high quality product (Pimentel, Kirkby and Shoff, 1993), intensive monoculture based agriculture favours pest attack and disease epidemics (Metcalf and Luckman, 1975) and pesticide use has been perceived by some farmers as a form of insurance (van den Bosch, 1978).

There are, however, costs associated with pesticide use that fall outside the scope of farm level economics, and therefore, do not enter into the cost benefit ratio of 1:3 or 1:4 as suggested by Lever (1990). Rachel Carson's 1962 book, *Silent Spring*, popularised for the first time the notion of the additional hidden costs associated with pesticide use. It essentially brought the myriad scientific data on pesticide toxicity within a framework which could be understood by the American public, and as such paved the way for many subsequent studies on the external impacts of pesticide use (Carlson et al., 1993).

The use of pesticides can engender additional internal costs to the agricultural sector directly, as well as external costs to the environment in general. Agricultural costs include; phytotoxicity to the crop (Attwood, 1985, Pilling and

Jepson, 1993), pesticide resistance (Roush and McKenzie, 1987), the development of secondary pests, pest resurgence (van den Bosch and Messenger, 1981), and impact on non-target beneficial organisms, such as natural predators (Flint and van den Bosch, 1981, Pimentel et al, 1991, Schenk and Wertheim, 1992).

Potential additional costs to the environment, and thus society, include health effects on agricultural workers from pesticide exposure (Pearce and Reif, 1990, Rola and Pingali, 1993, Garry et al., 1994, Stephens et al., 1995), health impacts on consumers from residues on food (Moses et al., 1986, Roberts, 1989, Russel, 1989), impacts on non-target flora and fauna (Conway and Pretty, 1991, Faasen, 1994, Beaumont, 1993), the entering of pesticides into the food chain (Kula, 1994, Nebel and Wright, 1996, Pearce and Tinch, 1997), and the presence of pesticides in ground, surface and sea water (Bergman and Pugh, 1994, Copin et al., 1994, Faasen, 1994, FoE, 1989).

The additional environmental costs associated with pesticides borne by society can be classed as technological externalities (Hartwick and Olewiler, 1986, Baumol and Oates, 1988, Cropper and Oates, 1992, Carson et al., 1994, Pearce and Tinch, 1997). Again, these costs are termed external because they are not accounted for when calculating the costs and benefits of pesticide use at the farm-level. The purchase of pesticides, and the cost of application, represent private costs to the user. The impacts of pesticide use on health, non-target flora and fauna, and groundwater contamination, represent costs borne by society as a whole (Carlson, 1994). The presence of externalities signals a socially excessive level of polluting activity (Griffin, 1991), and is typified by a divergence between private and social costs (Fisher and Peterson, 1976, Pearce and Turner, 1990, Kula, 1994, Neher, 1994).

2.5. Pesticides and externalities

As has already been seen, externalities can take two main forms: a public (undepletable) externality, and a private (depletable) externality. How then do pesticides fit in to this picture of public and private externalities? At first glance it is tempting to categorise the excessive use of pesticides in both public and private externality form. An example of a public bad resulting from pesticide use is the contamination of groundwater supplies. All of the conditions of undepletability would hold true (within reason) in this case. The problem of the private externality becomes slightly more complicated however. On the face of it the case can be made for pesticides to fit into the private externality category, but ultimately pesticides do not satisfy the requirements of private externalities.

Types of externality from pesticide use include: impact on non-target organisms, water contamination, human health issues such as neurological disorders and residues that have been found to be alarmingly high on some foods, arguably increasing the risk of cancer to the consumer[12] (Moses et al., 1986). One only has to refer back to the Alar[13] scare on apples in the USA (Russel, 1989, Roberts, 1989a, 1989b), to realise that pesticide residues on food is a real concern for many consumers, particularly in the developed World. It is safe to conclude that those externalities that are meaningful to policy makers and legislators are public in nature, and are specifically public bads (Fisher and Peterson, 1976, Cropper and Oates, 1992). The pollution arising from pesticide use falls clearly into the category of public externalities (Carlson et al., 1993).

The whole area of crop protection is currently high on the political agenda. Agriculture is now seen as a potential area of industrial pollution (CEC, 1992a, 1992b, 1992c), and the case for a move towards a more sustainable[14] agriculture

---

[12] At this stage it should be noted that no firm links are claimed between pesticide residues on food and increased cancer risk in consumers, but merely using this case as a suitable and convenient example of externality.

[13] Alar (daminozide) caused almost hysteria in the USA when residues were found on the apples available in supermarkets and stores. The concern centered particularly on the susceptibility of children to increased cancer risk from the exposure. Although the *increase* in risk to children was estimated at only 0.0025% the scare cost apple producers an estimated $115 million, through boycotts, and generalised reduced consumption through adverse publicity of apples (Roberts, 1989a, 1989b, Russel, 1989).

[14] In recent years sustainabilty has become an environmental byword, and although definitions abound a working definition of sustainable development might be: that it involves maximising the net benefits of economic development, subject to maintaining the services and quality of natural resources over time (Pearce and Turner, 1994).

is based on the argument that, in the long run, current agricultural systems are leading to an undesirable rural environment in social, economic and ecological terms and therefore a decline in the utility of a representative member of society (Webster, 1995).

Agriculture is identified in the Fifth Environmental Action Programme (FEAP) as one of five target sectors, the aim of which is to transform patterns of growth in the Community in such a manner that the path to a sustainable future can be followed (CEC, 1992c, Reus et al., 1994). The guiding principles behind FEAP are the precautionary principle, that of shared responsibility and the polluter pays principle. Under FEAP the following objectives are identified for agriculture:

1. The conservation of water, soil and genetic resources as the basis for the development of a sustainable agricultural sector.
2. Decrease in the chemical input in agriculture to the point where objective 1. can be met.
3. An equilibrium point between nutrient inputs and the assimilative capacity of the environment.
4. An integrated rural management permitting the maintenance of biodiversity and natural habitats, and reducing natural risks such as soil erosion.

(CEC, 1992c).

As a result of FEAP, and other agri-environmental initiatives, there has been an increase in the awareness of alternative agricultural management strategies such as IPM, Integrated Crop Management (ICM), and Integrated Fruit Production (IFP). Each of these management strategies seek, to some extent, to move agriculture away from intensive production practices to one which is more sustainable in the long run. The overall aim is to develop and adopt diverse and improved plant production systems in which the optimal utilisation of inputs maximises economic returns to the farms and protects the environment (Drummond, 1995). ICM, IPM and IFP are, therefore, a management philosophy encompassing all aspect of farm management.

In addition to this some European countries are moving towards often drastic reductions in the amounts of chemicals, and particularly pesticides, used in agriculture. This is particularly true of The Netherlands, Denmark and Sweden (Pimentel, Kirkby and Schoff, 1993, Reus et al., 1994, Beaumont, 1993, Green and Mumford, 1995, Pearce and Tinch, 1997) (see Table 2.1). Why, however, should this now be the case, and is modern European agriculture unsustainable?

Table 2.1. Pesticide reduction targets for The Netherlands, Denmark and Sweden. Source: The Pesticide Trust, 1994, Green and Mumford, 1995.

	Denmark	**Netherlands**	**Sweden**
Baseline Reference Period	**1981-85**	**1984-88**	**1981-85**
1st reduction target year	1990	1995	1990
1st reduction target (kg/a.i/ha)	25%	25 - 28%	50%
2nd reduction target year	1997	2000	1997
2nd reduction target (kg/a.i/ha)	50%	39 - 40%	75%
Pesticide use rate (kg/a.i/ha)	2.2	17.5	1.3

2.5.1 The problems of identifying and valuing pesticide related externalities: pesticides and Cost Benefit Analysis.

As has already been noted a typical list of pesticide related external costs would include: health effects of chronic and acute poisonings, domestic animal poisonings, loss of beneficials, pesticide resistance, honeybee and pollination losses, crop losses, fishery losses, bird losses and groundwater contamination (Pearce and Tinch, 1997). In order to identify socially optimal levels of pesticide use, externalities must be identified and brought within the decision making process, or included in any Cost Benefit Analysis (CBA) (Bowles and Webster, 1995).

As Beaumont (1993), Bowles and Webster (1995) and Pearce and Tinch (1997), point out, there are particular problems in the design and execution of a CBA of a particular pesticide use strategy. Bowles and Webster (1995) identify three reasons why this may be so. They are:

a) Large numbers of active ingredients are generally involved, many of which will have different individual environmental effects (and may engender additional cumulative effects),

b) There may be large numbers of actual and potential effects associated with a particular active ingredient, and,

c) Pesticides may be applied across a range of agricultural (and hence ecological) situations, which may differ in their ability to assimilate the pollution arising from pesticide applications.

In addition to this environmental impact data related to pesticides are incomplete (Beaumont, 1993), and furthermore impact data that do exist are for active ingredient only and do not consider the adjuvant[15] used in the formulation. The adjuvant may alter, or indeed enhance, the nature of environmental impact (Davies, 1997, Pers. Comm). Also, due to the complexity and unpredictability of environmental systems (Birch and Gafni, 1993), CBA has tended to focus on one particular aspect of pesticide related externalities due to technical and operational difficulties encountered in the analysis (Bowles and Webster, 1995). Examples of research aimed at identifying pesticide related externalities in a CBA framework include; Kahn and Kemp (1985), Pimentel (1991 and 1992), Higley and Wintersteen (1992), Rola and Pingali (1993), Lichtenberg et al., (1988) and Harper and Zilbermann (1992).

Currently, Environmental Economics and CBA, whilst providing many insights into the externality problems associated with pesticide use, are failing to provide practical management solutions to the problem. Therefore, this thesis argues that an alternative decision making framework is required that tackles pesticide related externalities on a more practical level. The alternative framework proposed in this thesis is that of the group of environmnetal impact models known as pesticide ranking indices. Here, environmnetal impacts are addressed at the farm level, by informing farmers about the relative "impacts" of individual

---

[15] An adjuvant is the compound used to deliver and fix the active ingredient of the pesticide, thus the active ingredient is mixed with the adjuvant. It may be in the form of a liquid, powder, granules and so on.

pesticide compounds, thus allowing farmers to choose between more or less harmful pest management strategies. This theme will be expanded on in Chapter 4.

2.6 Summary

Pesticides have been called a classic example of externality (Carlson et al., 1993). The technical difficulties in identifying and valuing externalities, in order to bring them within the decision making process, have prevented the adoption of economic instruments and the use of CBA with regard to pesticide use (Bowles and Webster, 1995). And yet there remains little doubt that in order to secure a socially optimal level of pesticide use, the externalities associated with agriculture must be explicitly accounted for in CBA (Pimentel et al., 1992, Bowles and Webster, 1995). Whilst the theory of externalities highlights the need to incorporate non-market effects of pesticide use into CBA and the use of economic instruments to reach socially optimal outcomes, this thesis proposes a different approach to the internalisation of the external effects of pesticide use; the pesticide ranking indice. This will be further discussed in Chapters 4, 5, 6 and 7.

3. An Introduction to The European Apple Industry.

3.1 The European apple industry in a global context.

This Chapter introduces the apple growing sector in the EU, the chosen case study for this thesis. It merely provides an introduction to sales and output levels in the industry, but also begins to discuss reasons why pesticide use might be classed as intensive in the apple growing sector.

Most of the world's apple supplies come from the temperate zone of the northern and southern hemispheres, between latitudes 40 and 50 degrees north, in Europe and North America, and between 30 and 40 degrees south in the southern hemisphere (Hinton, 1987). Production outside these latitudes is made possible where the climate is modified by oceanic influences, or by altitude, such as in New Zealand. Apples require cold winters to induce dormancy and the setting of fruit in the subsequent season, and long warm summers to obtain ripening in most varieties. They are generally not suited to the harshness of continental winters, however, technology has facilitated a gradual expansion in the geographical production possibilities of the apple sector through the adoption of cold hardiness and frost tolerant root stock for cooler climates, and the use of irrigation and heat tolerant varieties allowing expansion into hotter drier areas. Over the last 40 years there has been a gradual expansion in the areas under apple cultivation both in northern Europe and towards the equator (O'Rourke, 1994).

Various factors combine to make the EU the largest world market for fruit (and vegetables), despite its large domestic production (Hinton, 1987). International trade in fruit products is a function of climate and season, and the climatic range is not so great in Europe that fruit production is possible throughout the year. This necessitates out of season imports from sources outwith the EU. Consumption of fresh fruit within the EU is very high. This partly stems from a large domestic production, but is also a result of both high relative incomes, and the awareness of the nutritional benefits of fruit. Traditional trading links between European countries and former colonies, and political allies, such as Australia, New Zealand, South Africa and the USA, Brazil and Chile, also contribute in making the EU the largest world market for fruit (Hinton, 1987, Winter, 1989, O'Rourke, 1994).

World apple production has grown steadily in the decades since World War Two (Table 3.1). This situation is not unique to apple, similar growth has been experienced in other agricultural sectors, partly as a result of the increase in income levels in the industrialised West during that period. Apple is the most (economically and geographically) important of the deciduous fruits, which include pears and peaches, but has experienced competition from increasing supplies of exotic tropical and sub-tropical fruits, such as kiwi fruit, avocados, mango and papaya (Prognosfruit, 1995, O'Rourke, 1994). The value of the European apple crop in 1990 was worth in the region of ECU2500[16] million (King, et al. 1991).

Table 3.1 The expansion in world apple production 1948-1994, average Apple output, '000 metric tonnes. Source: Prognosfruit, 1995, O'Rourke, 1994, Hinton, 1987.

Year	Output
1948-60	13,512
1961-65	18,175
1969-71	28,309
1979-81	34,551
1986-88	40,114
1988-94	40,000

Expansion of apple markets has not been uniform around the world. Particularly rapid growth has occurred in the southern hemisphere, and in formerly centrally planned economies (although statistics for these countries are unreliable). Europe

---

[16] One ECU is worth approximately £0.72. (Financial Times, June 1996).

has remained the dominant supplier of apples since the 1940's, but areas under production have remained constant over the last decade, partly as a result of the grubbing up[17] programme which was introduced by the community to reduce the problems of over supply (Prognosfruit, 1995).

This situation may change over the short term as it is likely that competition from New Zealand, South Africa, Chile, Brazil, the USA, and potentially China, will increase over the next decade (O'Rourke, 1994). Table 3.2 shows apple production across the world, by continent and major apple producing country. It is interesting to note that although Europe's total production is by far the biggest of any region, the growth rate has been considerably faster in many other regions of the world such as in South America, Africa, and Asia. Should these trends continue then the EU is likely to face increased competition from these "New World" sources.

3.2 Apple production in the EU.

The EU is approximately 88% self sufficient in apple, with an average consumption across all Member States of 19 kg per individual per annum (Hinton, 1987, Behr, 1990, Prognosfruit, 1995). In 1994 apple production in the EU was just over 8 million tonnes, constituting approximately 20% of total world production. In 1995 production was in the region of 7 million tonnes (Prognosfruit, 1995). The 12.5% reduction was in part because the 1995 German harvest had been devastated by both frost in the early part of the season and by disease. This loss accounted for 500,000 tonnes of apple in the 1995 growing season, constituting more than 50% of Germany's entire crop (Prognosfruit, 1995). Production and consumption varies from country to country with generally the warmer Mediterranean countries providing more apples than the wetter northern European countries. Table 3.3 illustrates the wide variation in the output of apples between the Member States. The countries that dominate European apple production also have the highest domestic consumption rates.

---

[17] *Grubbing up* is simply the term used to describe the up-rooting of apple orchards to take them out of production.

Table 3.2 Annual World apple production by region and country (average production '000 tonnes, 1948-1991). Source: O'Rourke, 1994.

	Year				
Region or Country	**1948-60**	**1961-65**	**1969-71**	**1979-81**	**1989-91**
World	**13,512**	**18,175**	**28,309**	**34,503**	**40,181**
Europe	**9,508**	**11,822**	**13,966**	**13,359**	**14,148**
EEC	7,093	8,383	7,531	6,877	7,404
E. Europe	1,117	1,806	4,677	4,744	5,013
Scandinavia	213	221	211	205	129
Other	1,085	1,410	1,547	2,133	1,602
North America	**2,758**	**3,208**	**3,550**	**4,457**	**5,486**
Canada	300	416	413	466	513
Mexico	49	108	175	280	548
USA	2,409	2,684	2,962	3,711	4,425
South America	**273**	**536**	**718**	**1,406**	**2,189**
Argentina	102	440	435	945	1,006
Chile	45	54	142	252	683
Other	33	42	140	209	500
Africa	**61**	**173**	**270**	**460**	**645**
S. Africa	39	127	224	375	545
Other	22	46	46	85	100
Oceania	**247**	**426**	**556**	**533**	**718**
Australia	199	340	430	328	325
New Zealand	48	86	126	205	393
USSR (former)	**no data**	**no data**	**4,533**	**6,445**	**6,300**
Asia	**676**	**2,002**	**4,710**	**7,963**	**10,695**
China	118	305	1,953	2,993	3,967
Japan	382	1,066	1,037	887	1,048
Turkey	102	326	716	1,419	1,900
Others	74	305	104	2,664	3780

Table 3.3 Apple production in each member state of the EU in 1994 (tonnes). Source: Prognosfruit, 1995.

Country	Yield
Italy	2,153,492
France	2,128,757
Germany	879,600
Spain	753,300
Netherlands	600,000
Belgium	506,625
UK	330,000
Greece	319,866
Portugal	167,000
Denmark	37,530
Eire	9,000
Luxembourg	5,000
Total	**8,036,968**

The countries with the greatest land areas under apple production are Italy and France. These two countries alone account for 50% of total European output (Prognosfruit, 1995). Table 3.4 indicates the areas under apple cultivation in each of the Member States, and also indicates those countries which are expanding their capacity through the planting of new orchards. Those countries with a large proportion of their orchards aged five years or less are experiencing quite rapid growth in apple cultivation, with the most notable in this category being Belgium and The Netherlands (Peter Jaeker, 1995, pers. comm). It is likely that these countries will have a competitive advantage in the coming years as they will have the largest proportion of new, young, highly productive orchards planted with newer high yielding varieties.

Table 3.4 Areas under apple cultivation in each member state of the EU, 1994. Source: Prognosfruit, 1995.

Country	Area (ha)	Orchards < 5 years old	% less than 1 year
Belgium	11,985	5,504	45.9
NL	16,448	6,563	39.9
Denmark	1,803	581	32.2
Germany	39,223	12,422	31.7
Portugal	19,442	5,990	30.8
Eire	594	183	30.8
UK	19,705	4,397	22.3
Italy	83,201	18,116	21.8
Spain	53,126	11,473	21.6
France	65,999	13,361	20.2
Greece	13,983	1,806	12.9
Lux'g	703	56	7.9
Total	**326,212**	**80,452**	**24.7**

3.2.1 Apple varieties

Apple orchards can remain in production for several decades, but there is, and has been, a continual abandonment of old varieties in favour of newer, higher yielding varieties (O'Rourke, 1994). The reasons once identified are both obvious and elusive. First of all market tastes change. One year consumers may prefer red apples, another it may be green or yellow. Currently in the EU a larger percentage of new orchards are in the form of several varieties of red apples, whereas previously Golden Delicious (green to yellow) was the variety experiencing the most rapid growth (Graham King, 1995, pers. comm). Consumer preferences in apple shape, and skin pattern can also change. There is, however, no accepted model for predicting market preferences, and the likelihood of changes in market preference (Winter, 1989). The launch of a new apple variety is still a risky undertaking even with concomitant market research.

New varieties have several production advantages over older varieties. This has been due to the vast amounts of money that are spent annually on the breeding of new improved varieties in research stations across Europe (King et al., 1991). New varieties are bred for their ability to cope with differing climatic conditions. They are higher yielding, more disease resistant, and are more uniform in appearance and taste. This last point is essential in the marketing of fruit, as the consumer demands uniformity of both appearance and quality. In the mid to late 1970's the dominant variety in the world was Golden Delicious with a market share of 22.8% (O'Rourke, 1994). This situation is still borne out in the EU with Golden Delicious still being the most widely grown variety, although the market share of red apples is increasing (Table 3.5).

Although Golden Delicious still holds nearly 39% of the market share in the EU, evidence tends to suggest that its market share is set to drop considerably over the coming years. There was a 5% drop in 1995 from the 1994 crop, and a 7% drop in 1994 from the 1990 crop. The variety which has experienced the most dramatic growth in recent times is Gala, a red / green apple. Sales of Gala increased between 1994 and 1995 by 21%, while between 1990 and 1994 there was an astonishing rate of growth of 657% (Prognosfruit, 1995). The strength of Gala apples lie in its uniformity of appearance and taste, high yielding characteristics, and its ability to withstand adverse weather conditions, such as frost.

3.3 Fixed and variable costs in the production of apple.

The economics of apple production, as indeed of any perennial cash crop, can be extremely complicated. Commercial growers in the European Union range from the full time commercial farmer cultivating 25 -35 hectares in the UK and Germany, to the small scale, often part time farmer cultivating less than half of one hectare in Greece. The economic conditions facing these two types of producer are clearly quite different, and yet certain underlying principles hold true for both types of production.

Table 3.5 Quantity produced (tonnes) and market share of the main apple varieties in the EU, 1994. Source: Prognosfruit, 1995.

Variety	Quantity (tonnes)	% market share
Golden Delicious	2,753,367	38.6
Red Delicious	825,705	11.6
Jonagold	705,885	10.0
Granny Smith	378,313	5.3
Elstar	313,455	4.4
Gala	214,492	3.0
Morgenduft	197,291	2.8
Cox's Orange Pippin	194,800	2.7
Bramley	137,500	1.9
Renette	117,993	1.6
Boskoop	114,320	1.6
Idared	113,870	1.6
Gloster	96,809	1.4
Spartan	13,800	0.2
James Grieve	2,200	0.1
Others	953,648	13.4
Total	7,133,448	100

Firstly, a considerable time lag exists between planting and harvesting. This time lag can be anywhere from six to ten years depending on tree characteristics. Thus, the costs of establishing an orchard must be covered by output from the orchard during its productive years, which may be up to twenty five years. Secondly, production, and quality of product, may vary greatly from year to year. This phenomenon can be due to any number of reasons from winter frost, to diseases and pests, to hail storms (Winter, 1986, 1989). The smaller unit will suffer a greater percentage variability in production, and therefore not be able to cope as well as bigger production units. To compound this problem apples, like many perennials, suffer from a situation where an above average crop is usually followed by a below average crop (O'Rourke, 1994). Therefore, the average cost per unit output will vary greatly from year to year.

Apples are not a homogenous product, they vary in quality, size, taste and colour. The market values the product based on quality, with fruit sold for juices and processing attracting a much lower price than fruit for fresh consumption (Fenmore and Norton, 1985). In addition to this, in order to ensure the quality of the fruit, the farmer may incur additional costs in terms of increased pesticide use and pruning and thinning, to help guarantee quality. Therefore, in order to increase revenue, the farmer will necessarily incur additional production costs.

Establishing an orchard requires an initial investment in land, trees, irrigation, buildings and machinery. The multiplicity of orchard establishment decisions has a major bearing on orchard profitability (O'Rourke, 1994) as initial capital outlay must be recovered from the future earnings of the orchard. Orchard establishment costs tend to be financed by borrowing, adding to the riskiness of the venture with fluctuating interest rates. Other decisions concerning tree density, choice of rootstock, and size of tree are also all of extreme importance (O'Rourke, 1994).

Fixed costs are those costs associated with fixed factors. In other words they are independent of the level of output, and in particular, must be paid whether or not the orchard produces apples. Fixed costs in apple production naturally include interest paid on investment for orchard establishment, building costs, machinery costs, and land rent or taxes (Gittinger, 1982). Variable costs on the other hand are those costs which change as the output of the orchard changes. These costs include labour, pesticide use, machinery hire or purchase, fuel and irrigation.

3.4. The reasons for regional variations in production costs

Production costs vary between Member States in the EU, and indeed between different regions within individual Member States. For instance, the production costs associated with large scale, intensive apple cultivation in the Alto Adige region of Northern Italy are likely to be significantly different to the small scale, extensive apple production characteristics of the Mezzogiorno region of

Southern Italy. Many factors will combine to create regional differences in production costs, but the most important are: yield, cultivar, farm structure level of knowledge and experience of grower, land availability, level of wages, organisation of market, and political influences (Winter, 1986).

Within a total fruit growing area, regional characteristics will be the result of the overall decisions taken by all of the individual growers (Winter, 1989). Many of these decisions, and ultimately the production characteristics of a region, will be determined by additional factors such as the structure of marketing, technical services availability, distance to consumers, and so on. Physical parameters will also impact on these decisions such as geographical conditions and the associated variations in soil type, topography and climate.

3.5 Breeding disease resistant apple varieties

A plant breeding project, The Development of the European Apple Project (DEAC), began in 1991. The overall objective of this project was to reduce the amount of agrochemical used in apple production, through the use of advanced breeding methods. The goal was to develop disease resistant apple varieties suitable for introduction into the apple industry. The suitability of the new varieties would depend on a whole range of factors including, yield, colour, size, taste, and weather hardiness, as well as disease resistance (King et al., 1991).

The EU, funders of the DEAC project, were interested in identifying the potential social, economic and environmental impacts that may occur as a result of the introduction of new disease resistant apple varieties. The Scottish Agricultural College were selected to design and develop a methodology to achieve this goal. The remainder of this thesis discusses one methodological approach to the identification of the environmnetal impacts from pesticide use: pesticide rating indices.

This Chapter has sought to introduce the apple growing industry in the EU, and discuss the economic considerations of apple production. In doing so, a picture is presented of an industry where the economic necessities of return on investment dictate intensive monoculture practices, with a high quality, high value crop necessary to fulfil that obligation. One of the ways of achieving this high quality output is intensive pesticide use practices, employed in order to control the wide variety of diseases, pathogens and insect pests that affect apple production in the EU. This thesis now examines pesticide ranking indices, but returns to pesticide use patterns in the EU apple industry, and further examines why pesticide use might be classed as intensive.

4. A Critical Review of Pesticide Rating Indices

4.1 Introduction to pesticide impact modelling

Increasingly, governments are considering the use of models that rate pesticides by their environmental impact (Quin and Edwards-Jones, 1997). The purpose of such models are twofold; firstly they seek to influence agricultural practices, either at an individual farm or regional level, and secondly they seek to monitor and/or influence policy decisions at a national level (Penrose et al., 1994). It is widely accepted that such models fall far short of formal quantitative risk assessment, and yet it has been argued that quantitative risk assessment falls far short of the requirements of policy makers. As trade barriers come down throughout Europe and the trend becomes one of policy harmonisation, there is a need for a single policy tool that can strengthen, but above all standardise, agri-environmental regulation. It is essential for those wishing to practice IPM, organic farming, or simply to comply with existing legislation, that there is a Union wide criterion against which such strategies can be confidently adopted.

This chapter reviews selected models of pesticide impact available to decision makers and attempts to highlight the main strengths and weaknesses of the various approaches. In addition to this the results of an extensive pesticide impact modelling review are presented (Tables 4.2, 4.3a and 4.3b.). Currently the use of pesticide ranking models is limited at the strategic policy level, and yet such models are being considered in some states of the USA, Australia, The Netherlands, and now the UK (Newman, 1995). Much of the work on indices and ranking models has been driven by a wish to compare different chemicals, but the current policy need is for broader yardsticks to assess the outcome of policy generally, and reduce risk whilst not adversely affecting agricultural production. The Chapter also discusses the extent to which such models can contribute to the harmonisation of agri-environmental policy. Firstly, however, such models are placed in a policy and externality context and the rationale for the need for additional models of environmental impact from pesticide use discussed.

4.1.1. Pesticide use at the farm level.

Farmers do not generally have access to information regarding the environmental impact of the pesticides that they use. Certainly the information exists and is available at Agriculture and Environment Ministries, through Farmers Unions and through the pesticide manufacturing industries, but generally farmers neither have the time nor the inclination to carry out research into the implications of the pesticide use strategies that they adopt. Instead, there has been a tendency, if not necessity, to accept the view that *"if it has been approved by Government, it must be safe"* (Kovach et al., 1992, pp 1).

In theory this is a sound enough assumption to make considering the advances and developments that have taken place with regard to the protection of the environment in the Member States of the European Union. Of particular relevance to the issues of pesticide use is the development of Council Directive 91/414/EEC. This Directive, formally adopted by the EU during July 1991 (coming into force on the 25th July, 1993), essentially represents a Community wide regime for the authorisation of plant protection products (Scharpe and Smeets, 1994). The basic principles of the directive are:

1. To develop a Community wide list of accepted active substances.
2. To review substances already on the market.
3. To authorise, or deny, Member States individual plant protection products.
4. To harmonise rules on classification, packing and labelling.
5. To allow for the mutual recognition of authorisations.

(CEC, 1991, 1992a, 1992b, 1994).

A plant protection product may not be authorised unless it is established that when properly used:

a) it is sufficiently effective,

b) it is not phytotoxic,

c) it does not cause unnecessary suffering and pain to the vertebrates to be controlled,

d) it has no harmful effect on human or animal health, or on groundwater,

e) it has no unacceptable influence on the environment (fate and distribution, water contamination, non-target species).

(CEC, 1991, 1992a, 1992b, 1994).

Directive 91/414/EEC has been designed to provide the most extensive pesticide regulatory control so far developed on a Europe wide basis[18] (Scharpe, 1994). When coupled with additional environmental protection legislation, such as Directives 79/117/EEC (prohibition of pesticides), 80/68/EEC (groundwater protection), 80/778/EEC (drinking water standards), 76/895/EEC[19] (permissible residue levels on fruit and vegetables), then clearly a comprehensive package of environmental protection measures relating to pesticide use can be seen to be in place (see also Figure 4.2) (CEC, 1992b). If a pesticide has, by the definitions given in Directive 91/414/EEC, "no unacceptable influence on the environment" (Annex III), then the question arises: is there in fact a need for additional measures to reduce the risks associated with pesticide use, and if so why?

4.2. The need for additional pesticide impact ranking models

Environmental risk, and the extent to which it can be accurately measured, has been under intense discussion for three decades (Beaumont, 1994). Alterations in environmental systems due to the activity of humans, it is believed, cannot be accurately evaluated without a framework of environmental risk assessment (The Pesticide Trust, 1992, Rodricks, 1994, Johnen, 1995). It is also widely accepted that diminishing environmental impact, or increasing environmental

---

[18] The Directive covers all aspects of pesticide use from development of individual compounds, to the authorisation, distribution and use of that compound.

[19] This brief list represents only a few of many Directives and Regulations that relate to the use of pesticides and is not intended to represent an exhaustive list.

sustainability, requires a commensurate reduction in environmental risks[20] (Rodricks, 1994, Berg and Scheringer, 1994). Risk assessment can be defined as a highly systematic means for organising available information and knowledge and for specifying the degree of scientific certainty associated with each of the sets of data, models and assumptions, that are needed to reach conclusions regarding health risks of any type (Rodricks, 1994).

The reduction of environmental risks, however, requires that two conditions be fulfilled. They are:

1. The environmental risks related to a certain activity must be known.
2. The social, economic and political conditions must allow the actors to decide in favour of the alternatives with a less severe environmental impact potential and act accordingly.

The actions of human beings, be it the choice of pesticide use strategy or the choice of domestic fuel, are naturally shaped by the knowledge, rules, regulations, laws and social norms of their particular society. Quantitative risk assessment, in general, fails to recognise that quantifying the risks associated with a particular activity have important social aspects as well as purely technical considerations[21] (Wynne, 1992).

It is assumptions, in points one and two above, that necessitate the need for additional (not replacement) pesticide impact information. The first point refers to the fact that the environmental risks of a certain event must be known. To authorise a pesticide under Directive 91/414/EEC would imply that the data concerning the environmental impact of the particular pesticide are complete and accurate, but this is rarely the case. Because of the inherent complexity of

---

[20] It would be useful at this point to define toxicity, hazard and risk. Toxicity is an intrinsic property of the compound, i.e. the compound is toxic at a given concentration. Hazard is a function of toxicity and exposure, i.e. hazard equals toxicity x exposure. Risk is the likelihood or probability for the hazard to actually occur, and can be defined in either qualitative or quantitative terms (Johnen, 1994).

environmental systems there exists a severe lag between scientific knowledge and regulatory needs (Rodricks, 1994). The problems of complexity in environmental systems poses serious problems for effective environmental risk assessment in that ecosystems can neither be described sufficiently, nor can they be defined clearly, and the evolution of ecosystems over time is unpredictable and irregular.

Secondly, the social, economic and political conditions must allow for the actors to decide in favour of one strategy over another. Within pesticide Directive 91/414/EEC there is no provision for the comparison of the impact of the various types of pesticides, that would allow both farmers and decision makers to make trade-offs between economy and environment. Those making decisions on pesticide use at the farm-level do not have information regarding alternatives, and are often presented with one choice and one choice only (Kovach et al., 1992). This lack of information does not allow decision makers to proceed with pesticide use strategies in an environment of adequate knowledge. The central tenet of the Fifth Environmental Action Programme (1992) is, after all, the precautionary principle (Hull, 1994).

Risk assessment is now integral to the classification of new chemical substances and to the re-evaluation of substances already available to the agricultural community. EU Directive 91/414/EEC is based on the risk assessment process, as opposed to the non-scientific approach adopted in the setting of a quasi-zero value of 1μgl per litre used to regulate the quantities of all crop protection products present in drinking water (under Directive 80/778/EEC) (Johnen, 1995). The need for additional models to address environmental impact lies in the fact that pesticide use patterns vary greatly between European regions, therefore, risks will also vary greatly between regions. Consequently, risk should be addressed at the local or regional level, and whilst Directive 91/414/EEC provides a European wide framework of risk assessment, it is the duty of

---

[21] The risk assessment process for pesticides essentially contains four distinct stages: hazard identification, dose-response assessment, human exposure assessment and risk characterisation.

individual Member States to implement the necessary control and risk reduction measures within their particular regions.

4.3. Pesticides and external effects

What evidence is there for the need for additional methodologies that contribute to finding an "optimal" level of pesticide use? The rationale for using pesticides is the extent to which they protect agricultural production. Studies suggest that to stop using pesticides would result in crop losses ranging from zero to 100%[22] (Pimentel et al., 1992). Pesticides, however, are not free resources and a price must be paid for their use, but the financial price paid by farmers does not represent the economic price of pesticide use (Pimentel, 1991, Pearce and Tinch, 1997). In the past, cost benefit analysis (CBA) would have been used to measure the costs and benefits of pesticide use without considering external costs. Whilst the benefits associated with pesticide use are obvious (pesticides generate a benefit in the form of crop damage avoided), the costs associated with pesticide use are less tangible. Even with advances in CBA methodologies, and the explicit inclusion of environmental considerations, the extent to which true environmental costs can be identified is questionable. Proof, it is argued, that additional methods are required to regulate the use of pesticides, lies in the existence of externalities.

Several studies have attempted to quantify the external costs, or externalities, associated with pesticides (Hoag and Hornsby, 1992), perhaps the most notable being those by Pimentel et al., (1991, 1992).

Pimentel et al., (1991), identified ten main categories of external cost in the United States (although it is debatable if some of the costs identified can be regarded as externalities). They were; health impacts, domestic animal deaths, loss of beneficials, cost of pesticide resistance, honeybee and pollination losses, crop losses, fishery losses, bird losses, groundwater contamination and the associated costs of Governmental regulation to control all of the above. Pimentel et al., (1992), went on to estimate that the annual costs associated with these

---

[22] Other studies have suggested, however, that 35-50% reductions in pesticide use could take place in the USA without reducing crop yield (Pimentel et al., 1991).

eleven categories of external cost be in the region of $8 billion per annum (Table 4.1.)

Table 4.1. Total social and environmental costs from pesticides in the USA (Pimentel et al., 1992)

Impact	Cost ($million/ year)
Public health impacts	787
Domestic animal deaths and contamination	30
Loss of beneficials	520
Cost of resistance	1400
Honeybee and pollination losses	320
Crop losses	942
Fishery losses	24
Bird losses	2100
Groundwater contamination	1800
Government regulation to prevent damage	200
Total	8123

The figures presented by Pimentel et al., (1991), must be treated with caution, and are at best very rough estimates (The Pesticide Trust, 1993). Also some of the methodologies used to define economic value are questionable. For instance, for groundwater contamination the study uses clean up costs as the basis of their economic valuation, where individuals willingness to pay (WTP) would reflect a value closer to the total economic value of that particular external costs (Pearce and Tinch, 1997). The costs associated with remedial action would very much represent the lower bounds of value for an environmental good or service. Pimentel et al., (1991), recognise this, however, and state that "if the full........... costs could be measured as a whole, the total cost would be significantly greater than the estimate of $8 billion per year" (Pimentel et al., 1992, pp 758).

The absolute acceptability of the figures presented by Pimentel at al., (1991) (Table 4.1) are perhaps less important than the range of potential external costs identified. If those costs, however, were known accurately and borne by the users of pesticides, then pesticide use may well have to be reduced in order to be economically efficient. To subsidise pesticide use at the farm level encourages the overuse and misuse of pesticides. Furthermore it is biased against alternative pest control strategies such as IPM, and denies money from alternative sectors of the economy (Pearce and Tinch, 1997). To fail to internalise external costs essentially subsidises farmers in the same way. Others (society) burden the costs, and are therefore in some respects subsidising the inefficient use of pesticides.

It is in the existence of external costs (potentially considerable external costs), and the shortcomings of the existing regulation, that the rationale for the use of additional policy instruments are found.

4.4. Methods to asses the environmental impacts of pesticides, and pesticide use strategies

A large amount of research has taken place over the last three to four decades into the impacts of pesticides on the environment, and how to reduce pesticide use to what might appear to be a more rational level (Shahane and Inman, 1987). Methodologies to reduce the dependence on pesticides perhaps began, or were certainly popularised, with Stern et al's., (1959) model of economic injury levels and the integrated control concept. Stern addressed the issues of the use of biological control, monitoring, and the economic injury level[23], in an attempt to move away from a prophylactic pattern of pesticide use to a more conservative management regime. The use of models such as Stern's have been the foundation of subsequent IPM philosophy, and whilst the motivation for the development of such models was initially an economic one, there are potentially considerable environmental benefits to be gained from the use of economic injury levels.

The model of Stern et al., was further developed, in terms of environmental considerations, by Higley and Wintersteen (1992) with the explicit inclusion of an environmental parameter within the economic injury level equation. Here,

---

[23] The economic injury level is defined as the lowest (pest) population that will cause economic damage, and therefore represents the point at which control measures should be implemented (Stern et al., 1959).

Higley and Wintersteen, identified risks associated with pesticide use, ranked those risks, and assigned a monetary value[24] to those risks, the result of which was to give the environment a value which could then be added to the internal costs of pest control. In theory, incorporating the costs of environmental damage raises the economic injury level resulting in a decrease in pesticide use (Levitan et al., 1995). The problem with this and other methodologies using economic injury levels is that a huge amount of data is required at an individual farm level. Economic injury level is likely to vary between farms, regions, crops and management strategies, therefore the results of such an analysis are not readily transferable. This necessitates lengthy and expensive fieldwork to be carried out. Evidence has tended to suggest that the additional information gathering, and monitoring workload required to identify economic injury levels, is not one which farmers readily embrace (McDonald and Glynn, 1994).

Additional methodologies to assess the environmental impact of pesticide use include anecdotal accounts (such as those by Pimentel et al., 1992), the use of a proxy for environmental damage (Berg and Scheringer, 1994), chemical hazard scoring systems (O'Bryan and Ross, 1988), tabular databases and composite environmental impact rating systems (Kovach et al., 1992, Reus and Pak, 1993, Penrose et al., 1994). Pesticide rating may also be as simple as assigning each pesticide the values; hazardous, or non-hazardous (Rola and Pingali, 1993).

Some European countries, such as Denmark, Sweden and The Netherlands, are currently in the midst of programmes aimed at reducing the total amount of pesticides used, often seeking a 50% reduction in current levels (Beaumont, 1993). Aiming for quantity reductions, however, is not necessarily the answer to environmental pollution problems, as pesticides are regularly replaced by compounds often between 10-100 times more powerful (Reus et al., 1994,

---

[24] Higley and Wintersteen (1992) identified the Contingent Valuation Method as the most appropriate tool for assigning economic values to environmental gods, although Levitan et al., (1995) noted that farmers willingness to pay was not a true reflection of the costs associated with the environmental impact of pesticide use. Contingent Valuation seeks to elicit values for environmental goods and services by asking such questions as: how much would you be willing to pay to avoid a high level of risk from a single pesticide use? (Higley and Wintersteen, 1992).

Pearce and Tinch, 1997). It is therefore necessary that pesticides are ranked and judged individually (in terms of their percentage active ingredient as well as their overall use patterns) so that alternative use decisions can be made. Such a ranking was undertaken here prior to selecting on e model for use in analysing pesticide use in the apple growing sector.

Table 4.2 summarises the models that were selected for review for this thesis. Single species impact models were omitted from the analysis, because such information is already available through the pesticide registration process. It was felt that single species models did not constitute an indice, and as such, when taken alone, did not add to the decision making process with regard to pesticide use at the farm level.

Table 4.2. The pesticide risk indices included in the literature review.

Index	Abbreviation	Reference
"Dirty Dozen" Blacklist	DDB	IFPAAW (1995)
Registrations, Sales, Tonnage's, Sectors	RSTS	Various industry sources
Toxicological Register- WHO	TR	WHO (1975, ongoing report)
US EPA Pesticide Classification	EPAPC	US EPA (ongoing) Farm Chemical Handbook
Environmental - Economic Cost Benefit Analysis	ECBA	Pimental *et al.* (1992)
Insecticide Pest Management Rating	PMR	Metcalf (1975)
Environmental Yardstick	EY	Reus & Pak (1993)
Integrated Farming Systems and environmental exposure to pesticides.	EEP	Wijnands & van Dongen (1995)
Stemilt Growers Responsible Choice Point Summary	SGRCPS	Reed (Stemilt Growers) (1993)
Environmental health policy programme ranking system California.	EHP	Pease *et al.* (1991)
Florida Pesticide Use Risk Evaluation	F-PURE	Florida Pesticide Review Council (1984)
The Pesticide Index	PI	Penrose *et al.* (1994)
The Environmental Impact Quotient	EIQ	Kovach *et al.* (1992)
Chemical Scoring system for hazard.	CSSH	O'Bryan. & Ross (1988)
Cost-Environmental Hazard Frontier.	CEHF	Hoag & Hornsby (1992)
SYNOPS 1.0	SYNOPS	Gutsche (1995)
Economic Damage from Herbicide Pollution	EDHP	Kahn (1987)
Economic Injury Levels with economic variables	EILEV	Higley & Wintersteen (1992)
Computer based eco-rating.	ECO	Lewis *et al.* (1996)

Table 4.3a and 4.3b further analyse each of the methodologies selected for review, and go on to examine what type of data is included in the model, who the model was designed for, its usefulness as a policy instrument and so on. Table 4.3a examines the toxicological properties examined by the various modelling approaches and identifies which environmnetal components are examined, such as aquatic, human health, terrestrial, groundwater and so on. Table 4.3b examines each model in a policy context, how the model can be used, at what scale it can be used (farm, regional, national) and whether it was developed for use with a single or multiple crops.

Those models that appeared to fulfil most of the criteria listed in Tables 4.3a and 4.3b, were selected for further review. Some of the models examined considered only human health, such as the WHO Toxicological Register, and so were discarded due to their failure to examine non-human environmnetal impacts. Other approaches examined the economics of environmnetal impacts, such as Hoag and Hornsby (1992), Higley and Wintersteen (1992) and Kahn (1987), but these too were discarded as farm level economic analysis of the environmnetal impacts from pesticide use is time consuming, complex, and would need to be conducted many times in many localities. Rather, those models were selected that appeared to be easy to use and understand, were cost effective and applicable across many different crops[25] and localities, and considered all aspects of the environment, such as human health, groundwater contamination, effects on fish, birds, bees, mammals and so on.

Although the Computer Based Eco-Rating of Lewis et al., (1996) appeared to fulfil most of the criteria indicated in Tables 4.3a and b, it was discarded because its approach was essentially the same as the Pesticide Index (Penrose et al.,

Table 4.3a. An evaluation matrix for twenty risk indices: Toxicological properties. Abbreviations are explained in Table 4.2. 'Y' indicates that the index displays the property as listed in Column 2.

		DDB	RSTS	TR	EPAPC	ECBA	PMR	EY	EEP	SGRCPS	EHP	F-PURE	PI	EIQ	CSSH	CEHF	SYNOPS 1.0	EDHP	EILEV	ECO
Human Health	Worker	Y		Y	Y	Y	Y						Y	Y	Y					Y
	Consumer					Y	Y				Y			Y	Y					
	Water					Y		Y	Y		Y	Y	Y	Y	Y	Y	Y			Y
Environment	Soil							Y	Y		Y	Y	Y		Y		Y			Y
	Air								Y								Y			
	Dermal	Y		Y	Y					Y	Y	Y	Y	Y	Y	Y	Y			Y
Chemical Properties	Oral	Y		Y	Y						Y	Y	Y	Y	Y	Y				Y
	Reproductive													Y	Y					
	Teratogen													Y	Y					
	Mutagen													Y	Y					
	Carcinogen	Y									Y	Y	Y	Y	Y					
	Soil ★-life						Y	Y	Y	Y	Y	Y	Y	Y	Y		Y			Y
	Plant ★-life						Y		Y		Y	Y	Y	Y	Y					Y
	Run off								Y	Y		Y					Y			
	Leaching								Y	Y	Y	Y	Y	Y	Y	Y	Y			Y
	Systemicity													Y			Y			
	Volatilisation																Y			
Test groups	Birds					Y	Y				Y			Y	Y					Y
	Fish					Y	Y	Y			Y			Y	Y					Y
	Mammals					Y	Y				Y		Y	Y	Y					Y
Beneficials	Terrestrial					Y	Y	Y		Y	Y		Y	Y	Y					Y
	Aquatic					Y		Y		Y	Y			Y	Y			Y		
Plants	Aquatic																	Y		
	Terrestrial																			

Table 4.3b. An evaluation matrix for twenty risk indices: Policy considerations. Abbreviations in Table 4.2. Y' indicates that the index displays the property as listed in column 2.

		DDB	RSTS	TR	EPAPC	ECBA	PMR	EY	EEP	SGRCPS	EHP	F-PURE	PI	EIQ	CSSH	CEHF	SYNOPS	EDHP	EILEV	ECO
Human Health	Worker	Y		Y	Y	Y							Y	Y	Y					Y
	Consumer					Y					Y			Y	Y					
Environment	Water					Y		Y	Y		Y	Y	Y	Y	Y	Y	Y			Y
	Soil							Y	Y		Y	Y	Y		Y		Y			Y
	Air								Y								Y			
Spatial	Local											Y						Y	Y	
	Regional																			
	Both	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y		Y	Y	Y	Y	Y			Y
Sector	Sectoral																			
	Universal	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
Output	Numerical		Y			Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
	Descriptor	Y		Y	Y															
	Weighting criteria used						Y			Y	Y	Y	Y	Y	Y					Y
	Is model in use	Y	Y	Y	Y			Y		Y	Y	Y	Y	Y						
	Usefulness as policy tool			Y	Y		Y	Y		Y		Y	Y	Y						Y
	Uses available data												Y	Y						Y
	Data gaps present	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
Operational	Mathematically robust	N/A	N/A	N/A	N/A			Y	Y									Y	Y	
	Easy to calculate	N/A	N/A	N/A	N/A		Y			Y	Y	Y		Y						Y
	Under-standable	Y	Y	Y	Y		Y	Y		Y	Y	Y	Y	Y						Y

1994), and the Environmnetal Impact Quotient (Kovach et al., 1992), but obtained its toxicological data from pesticide labels.

Those models that appeared to address those issues that would most help farmers and policy makers assess the environmnetal impacts from pesticide use were the Pesticide Index (PI) of Penrose et al., (1994), the Environmnetal Impact Quotient (EIQ) of Kovach et al., (1992) and the Chemical Scoring System for Hazard (CSSH) of O'Bryan and Ross (1988). The CSSH of O'Bryan and Ross (1988) was discarded from the analysis because it was extremely difficult to calculate pesticide ratings using this methodology. In addition to this the output was also complex and difficult to understand. Whereas the PI and EIQ models present single figures of impact for each pesticide, the CSSH presented many figures for each pesticide, based on impact in each environmental sub-component. Thus, it was deemed not be user friendly at the farm or policy level.

Thus, the methodologies chosen for further review were the "Environmental Impact Quotient" (EIQ) model of Kovach, Petzoldt, Degni and Tette (1992), the Environmental Yardstick approach of Reus and Pak (1993) and the "Pesticide Index" (PI) of Penrose, Thwaite and Bower (1994). The Environmnetal Yardstick (Reus and Pak, 1993) was chosen for further review because this model is currently being considered for use by the Dutch Government for all Dutch agriculture (Groot, 1996).

4.4.1. The EIQ approach of Kovach, Petzoldt and Tette (1992).

IPM is a pest management strategy that uses a variety of methods to manage pests. Chemical controls are used if no other suitable method is available, but in the past pesticide choice decisions have been based on the efficacy and cost of the compound rather than on environmental hazard (Kovach et al., 1992). It was this situation, and the assumptions by some that if the compound had been approved by Government it must be safe, that prompted the development of the Environmental Impact Quotient (EIQ) model of pesticide impact.

The model was developed to compare different pesticides and different pest management practices in order to determine which pest management programme was likely to have the lowest environmental impact. Extensive data were used to construct the model from a variety of scientific and regulatory sources, such as the Environmental Protection Agencies (EPA) pesticide registration process and the Extension Toxicology Network (EXTONET)[26], a collaborative toxicology and pesticide education project involving Cornell, Michigan State, Oregon and California Universities (Kovach et al., 1992).

The primary module of the EIQ model is an algebraic equation that generates a composite index of environmental impact for each pesticide analysed (Levitan et al., 1995). A second equation, "the field use rating[27]", allows for a site specific analysis based on the active ingredient of the pesticide and total dosage. This then allows for the achievement of the main objective of the model; the comparison between different pest management strategies in terms of environmental hazard or impact.

The EIQ model addresses the issues of hazard to farmworkers, consumers and non-target flora and fauna. To simplify the interpretation of the data the toxicity of the active ingredient of each pesticide was grouped into low, medium or high toxicity categories, and rated on a scale from 1 to 5 (1 being very low impact, 3 being a medium impact and 5 being very high impact). This approach is the same in principle to that of Metcalf (1975), which was perhaps the first attempt to develop a composite index of pesticide impacts (Levitan et al., 1995). According to Kovach et al., (1992), a value is assigned to each category of potential impact: the rating. An additional weight is also assigned to each of the sub-categories farmworker, consumer and ecological impact, again based on a 1 to 5 scale. The effects assigned the 1, 3 or 5 rating are:

Dermal Toxicity - DT

---

[26] Naturally many more sources of pesticide impact data were used in order to consider as many impacts as possible.

[27] The EIQ field use rating is simply EIQ value X % active ingredient X Rate of use.

Chronic Toxicity - C
Systemicity - SY
Fish Toxicity - F
Leaching Potential - L
Surface Loss Potential - R
Bird Toxicity - D
Soil Half Life - S
Bee Toxicity - Z
Beneficial Arthropod Toxicity - B
Plant Surface Half Life - P

The final EIQ equation is equal to the average of the farmworker, consumer and ecological component. The EIQ for each sub-component equals:

$$\text{EIQfarmworker} = C(DT*5) + (DT*P)$$

$$\text{EIQconsumer} = C(\ (S + P\ ) / 2)*SY + L$$

$$\text{EIQecological} = (F \text{ x } R) + (D(\ (S + P) / 2)*3 + (Z*P*3) + (B*P*5)$$

The numbers in each of the equations above refer to the relative weighting (low, medium, high) to each potential impact. A consistent rule throughout the model is that the impact potential of a pesticide is equal to toxicity of the chemical multiplied by exposure (Kovach et al., 1992, The Pesticide Trust, 1993, Dushoff et al., 1994, Levitan et al., 1995). The total EIQ value for each pesticide is the average of the sum of the three sub-components, and is represented by a single figure. The fungicide fosetyl-Al (Aliette) has a low impact value of 13.7, as has the insecticide hexakis (Vendex) at 12.8. A medium impact is the insecticide mevinphos (Phosdrin) at 28.2, whilst methidathion (Supracide) has a high impact at 69.3. An extremely high impact would be in the region of an EIQ of 100, the figure attained by parathion (104.4).

The strengths of the approach are the range of impacts considered in the analysis, the ease of use of the final model for farmers, and the ease of understanding for policy makers. It is also relatively straight forward to identify acceptable thresholds of impact using this analysis, although gaining agreement on what is "acceptable" might be more problematic. However, pesticides are rated as 1, 3, or 5 which limits the range of scores. As Levitan et al., (1995), and Dushoff et al., (1994) point out, distortions caused by rating neutral effects as 1 rather than 0 are compounded because toxicity ratings are multiplied by rates of use. A high dose of a harmless input, therefore, may have a comparable EIQ value to a highly toxic compound used at lower doses, i.e. 1 multiplied by 5, equals 5 multiplied by 1. Also to use the average value of the farmworker, consumer and ecological component of the EIQ equation may distort harmful effects in one category and minimal effects in another. Dushoff et al., (1994) go on to point out that the linearity of the EIQ damage function (see Figure 4.1) is unrealistic. Another failing of the model may relate to data gaps that exist in some areas of toxicological impact (Dushoff et al., 1994), although it is likely that the model can be continually refined and updated as and when new data become available. However, these models were constructed using data from the pesticide registration process, thus, any data gaps that are present in the models are also present in the registration process. A worked example of the model of Kovach et al., is presented in Table 4.2 and Figure 4.1.

Assigning a single environmental impact figure for pesticides may well be misleading unless spatial aspects are considered. For instance, the likelihood of a farmer "polluting" the environment will be far greater if a) he is situated on soils with a high leaching potential, and b) he is situated within a few metres of a river or lake. These two factors could well be added to the EIQ equation without too much difficulty. This would allow farmers to assess the environmental impact of their pest management strategy based not only on the relative toxicity of the pesticides they use, but also on the geographical conditions in which they farm. In addition to this, climatic conditions will also have an impact on the amount of pesticides polluting the environment. Pesticides applied prior to or during heavy rainfall are much more likely to find their way into watercourses for example (Eke et al., 1996). Also temperature, rainfall and humidity will al have an influence on the number of different pests and diseases affecting production, as well as on the incidence of pest attack. A critical factor is also the method used to report this data. Dushoff et al., (1994) have criticised the method of reporting results from the EIQ model as being misleading. This theme will be returned to in Chapter 7.

4.4.1.2. An example of the results obtained using the Kovach EIQ model.

The purpose of the model presented by Kovach et al., (1992) is to provide farmers with greater information on environmental impact (or potential impact) in order that they can make comparisons between different chemical compounds, and between different management strategies (Kovach et al., 1992, The Pesticide Trust, 1993). As shown in Table 4.2. and Figure 4.1, it is possible to map the differences, in terms of potential impact, between different compounds, thus allowing the farmer to choose the least environmentally harmful pesticide option[28]. In some situations it may be desirable to assign a target EIQ, or threshold, above which represents unacceptable pesticide use levels (see Figure 4.1). The EIQ FUR, presented in Table 4.4, represents the environmnetal impact for each pesticide based on the EIQ units (impact) of Kovach *et al.*, 1992. In this case an arbitrary threshold was assigned to illustrate the point. The threshold is based on both EIQ (vertical axis), and dosage or rate of use (horizontal axis).

Table 4.4. The EIQ field use rating (EIQ FUR[29]) of three insecticides.

Compound	EIQ		A.I.	Rate	EIQ FUR
Sevin 50WP (Carbaryl)	22.6	X	0.5	X 6 =	67.8
Thiodan 50WP (Endosulfan)	40.5	X	0.5	X 3 =	60.8
Guthion (azinphos-methyl)	43.1	X	0.35	X 2.2 =	33.2

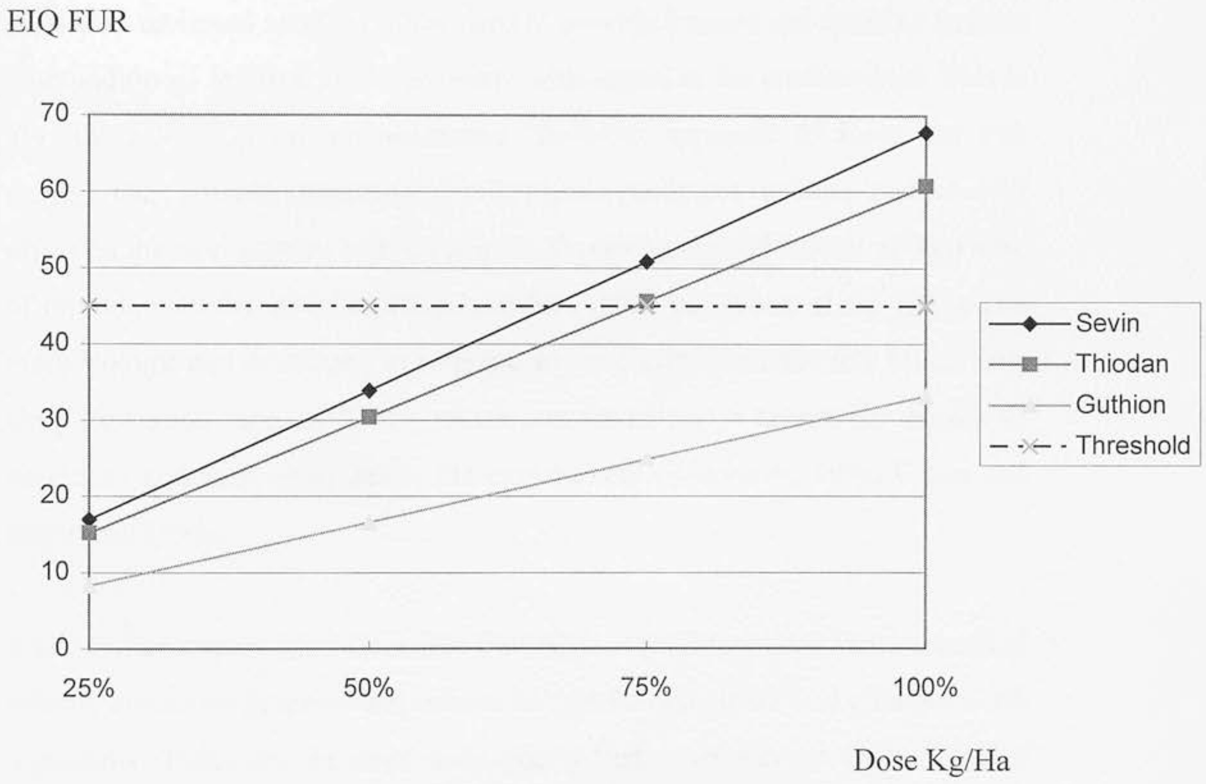
Source: Adapted from Kovach et al., 1992.

---

[28] The model of Kovach et al., (1992), however, makes no judgment as to what level of pesticide use is acceptable or unacceptable.

[29] The EIQ FUR are impact points assigned to each pesticide related to toxicity, solubility, half-life, leaching potential and surface loss potential (Kovach et al., 1992).

Figure 4.1. Comparing the environmental impact of 3 insecticides: Sevin, Guthion and Thiodan, with an arbitrary "impact" threshold indicated. Dose refers to Kg/Ha applied, with 100% being recommended dose and 75%, 50% and 25% representing pesticide use reduction strategies.



The methodology of the EIQ model, together with results of the analysis of the apple industry will be examined in some detail in Chapter 6.

The most interesting proposition in these models may not be in the actual values presented by the models, but in the grouping together of pesticides into low, medium and high risk categories. It gives the user "decision points", which provide information that allows for choices to be made between pesticides that are potentially more harmful than others, and between different pesticide management strategies that yield a different potential for environmental impact. The models present the users with options (Newman, 1995). Despite recent advances in such impact identification models, both Levitan et al., (1995), and

Dushoff et al., (1994), argue that a niche still exists in the development of tools to aid decision makers at both the local and national levels.

4.4.2. The Environmental Yardstick approach of Reus and Pak (1993).

The philosophy surrounding all of the methods for ranking the impact of pesticides reviewed here lies in the need to provide farmers and decision makers with additional information, specifically with regard to the environment. This is also the case with the Environmental Yardstick approach of Reus and Pak (1993). The yardstick enables farmers to choose pesticides with the least harmful effect on the environment and to compare the environmental effects of their way of farming with that of others (Reus and Pak, 1993, Verhoeven et al., 1994). The methodology was developed in response to the Dutch Governments Multi-Year Crop Protection Plan (MYCPP), which sets out reduction targets for the use of pesticides and their emissions to the environment (Beaumont, 1994, Green and Mumford, 1995).

The environmental yardstick for pesticides considers three environmental effects; effects on groundwater, effects on aquatic organisms, and effects on soil organisms. These are the three main effects that have been given the highest priority by the Dutch Government under the MYCPP (Reus and Pak, 1993). The model then assigns "Environmental Impact Points" (EIP) (Verhoeven et al., 1994) for each pesticide analysed, with naturally a higher EIP given to more harmful pesticides. As with the EIQ model of Kovach et al., (1992) the active ingredient for each compound is assessed and analysed, rather than total kg/ha application alone.

The models used to assess the environmental impact of pesticides, and hence used to assign the EIP, are from the ecological evaluation data used by the Dutch Government in their pesticide registration procedures (Reus and Pak, 1993, Verhoeven et al., 1994). A reference point of 100 EIP was established as an acceptable threshold of hazard or impact. The value of EIP (as with the EIQ) depends on such factors as properties of the compound, application factors, and environmental conditions such as soil property. EIP are assigned to a standard application of 1 kg/ha, and if a different dose rate is used then the EIP is multiplied by that dose rate (Reus and Pak, 1993, Verhoeven et al., 1994). Although not specifically designed for the purpose, the EIP approach has been used in farm economic modelling with Linear Programming (Verhoeven et al., 1994).

As Reus and Pak (1993) point out the EIP model can be criticised on three counts:

1. The criteria and standards which have been chosen to assess the environmental impact of pesticides.
2. The methods which have been used to estimate the environmental impact of pesticides.
3. The data which have been used.

Although the model currently considers only three environmental impacts, there are plans to extend the coverage of the model, but only if, firstly, adequate data is available, and secondly, if it does not have an adverse effect on the inherent user friendly nature of the model to farmers and decision makers. Reus and Pak (1993) go on to note that as long as the environmental yardstick is in its development stage a substantial uncertainty margin in the pollution points assigned will have to be taken into account. In view of the intended use of the yardstick, i.e. providing growers with insights into the environmental burden by pesticides, a certain uncertainty margin is acceptable. The yardstick model proposed by Reus and Pak (1993) is currently being used in Dutch agriculture as a means of reducing the environmental impacts of agriculture (Groot, 1996, The Pesticide News, No 33, 1996). Table 4.5 Gives an example of the results from the environmental yardstick.

Table 4.5. A comparison based on environmnetal impact of 6 commonly used pesticides used in potato production in the Netherlands. Reus and Pak (1993).

Pesticide	Dose (kg/a.i/ha)	Pollution points at recommended dose rates Water organisms	 Soil organisms	 Groundwater
maneb	1.5	2500	1500	15
maneb	1.5	2500	1500	15
mancozeb	2.5	18	0	24
mancozeb	2.5	18	0	24
mancozeb	2.5	18	0	24
maneb/fentin	1.1	2800	820	8
maneb/fentin	1.1	2800	820	8
metribuzin	0.5	980	60	260
pirimicarb	0.3	110	230	0
propoxur	0.5	182	275	10000
Total	**14**	**11926**	**5205**	**10378**

4.4.3. The Pesticide Index approach of Penrose, Thwaite and Bower (1994).

The Pesticide Index (PI) approach adapted by Penrose, Thwaite and Bower in 1994, was developed as a result of the Australian Apple and Pear Growers Association making pesticide reduction its number one research priority, but also as a result of the perceived inadequacies in existing methodologies to assess individual compounds in terms of environmental impact and economic efficiency. Whereas the models of Reus and Pak and Kovach readily lend themselves to economic modelling, the PI approach of Penrose et al., explicitly incorporates an economic variable into the pesticide index. The objectives of the model remain the same, however, in that it seeks to identify less desirable pesticide uses (Penrose et al., 1994). The use of the model was to be explicitly in the accreditation of IPM fruit (Levitan et al., 1995).

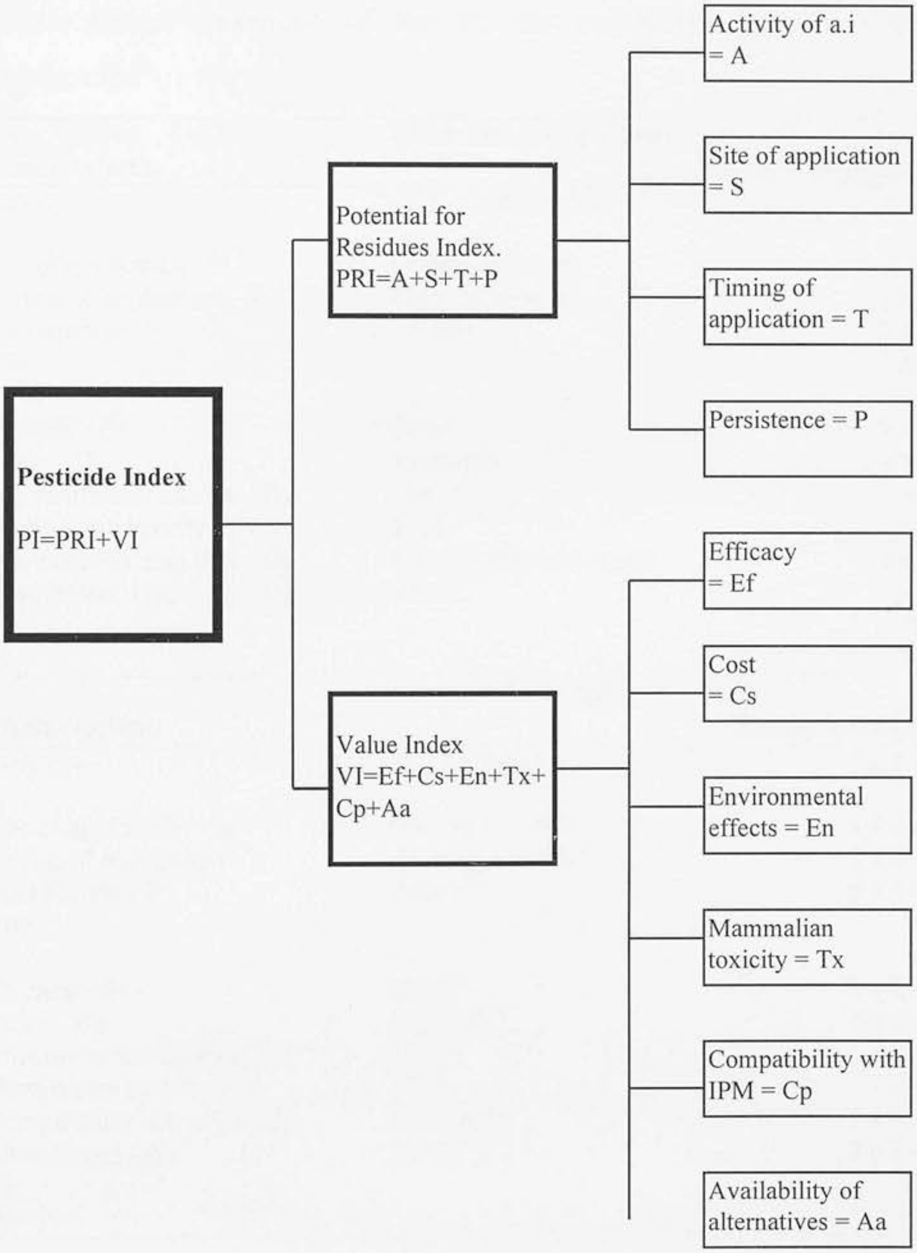
As with the model of Kovach et al., (1992) a broad range of environmental parameters were considered, as was the effect of timing of treatment on the potential for leaving residues on fruit. The attributes of the pesticide are examined, including toxicity, persistence, mode of action, as are the economic variables, efficacy and cost. To be meaningful, Penrose et al., note that the rating index should be calculated for each pesticide, crop, site and management strategy.

The model is initially split into two parts; the Potential for Residues Index (PRI) which estimates the potential for residues to occur and the Value Index (VI) which estimates the importance of the pesticide in a given crop protection system (Penrose et al., 1994). By adding the PRI and the VI, an overall rating index is achieved called the Pesticide Index (PI). The model is structured as shown in Figure 4.2, and an example of the workings of the PI model is presented in Table 4.6.

The PRI is a figure from 4 to 80 calculated by adding the product of each of the ratings, multiplied by a subjective weighting scale of 1 to 4. The higher the index, the greater likelihood of residues at harvest. The VI is a figure from 6 to 120 again calculated by adding the product of each of the ratings, multiplied by the weighting of 1 to 4. A low value indicates that the product is highly valuable to the production of the crop, with a high VI showing that the product is of little value, and should be replaced (Penrose et al., 1994).

The ratings determined for a particular pesticide, crop and management practice, represented in the PRI and VI, are enumerated in a standard format to produce a cryptogram, which represents the PI (Penrose et al., 1994). This information can be entered onto to a computer and run at the farm level for each given set of circumstances. Weightings, of 1 to 4, are assigned by the farmers according to the relative importance of the environment, IPM, efficacy, and so on, dependent on their specific situation. This effectively allows farmers to trade cost, efficacy and availability of alternatives against environmental considerations. One potential problem here, however, is that of allowing farmers to make the trade-off between environment and economy. If economic factors are given precedent, for example, the environmental costs of pesticide use are neither internalised nor recompensed by the economic benefits of that pesticide use (Quin and Edwards-Jones, 1997). One of the main weaknesses of the approach is in the assignation of the weights for each component of the model. It is proposed that this will be carried out by individual farmers, resulting in many different outcomes for similar management strategies, on similar crops using similar pesticides. If the model proposes to form

Figure 4.2. The Pesticide Index system (Penrose et al., 1994).



the basis of an IPM accreditation scheme, then this could pose operational problems at a local, regional and national scale (Quin and Edwards-Jones, 1997).

Table 4.6. An example of the PI: the comparison of Mancozeb with Flusilazole.[30]

	Mancozeb (800g/a.i/ha)	
Characteristic		**Rating x Weighting**
Activity - A	150g x 0.8a.i = 120	3 x 2 = 6
Site of application - S	Leaves and fruit	5 x 2 = 10
Timing of application - T	Through season	3 x 3 = 9
Persistence - P	14 days	3 x 3 = 9
PRI		**34**
Efficacy - Ef	Good	1 x 2 = 2
Costs - Cs	Moderate	3 x 1 = 3
Environmental effects - En	Low	1 x 2 = 2
Mammalian toxicity - Tx	6750	1 x 2 = 2
Compatibility with IPM - Cp	Affects predatory mites	4 x 1 = 4
Alternatives - Aa	Some	3 x 2 = 6
VI		**19**
PI		**53**
	Flusilazole (200g/a.i/ha)	
Characteristic		**Rating x Weighting**
Activity - A	10g x 0.2a.i = 2	1 x 2 = 2
Site of application - S	Leaves and fruit	5 x 2 = 10
Timing of application - T	Through season	3 x 3 = 9
Persistence - P	7 days	3 x 3 = 9
PRI		**30**
Efficacy - Ef	Good	1 x 2 = 2
Costs - Cs	High	4 x 1 = 4
Environmental effects - En	Low	1 x 2 = 2
Mammalian toxicity - Tx	1100	1 x 2 = 2
Compatibility with IPM - Cp	No effects	1 x 1 = 1
Alternatives - Aa	Some	3 x 2 = 6
VI		**17**
PI		**47**

---

[30] The weightings applied to both fungicides are the same, therefore, there would be less need to reduce the use of flusilazole than that of mancozeb. The EIQ equation of Kovach et al., (1992) also rates mancozeb more "harmful" than flusilazole (although does not include any economic criteria).

4.5. The potential uses of pesticide impact ranking models as policy instruments

Pesticide impact ranking models have been designed for, and can achieve, a number of functions and objectives. They seek to either influence individual behaviour at the farm level (Kovach et al., 1992, Reus and Pak, 1993, Verhoeven et al., 1994), at the regional or national decision making level, or both (Penrose et al., 1994, Newman, 1995). It is important to note that such models are aids to the decision making process and adjuncts to existing pesticide regulations and policies, and do not seek to replace or usurp any part of the existing measures to protect the environment (Quin and McGregor, 1995). The models reviewed can achieve the goal of adding to the decision making process in a number of different ways.

Food safety issues and concern over groundwater contamination has lead to an increased interest in IPM over the last few years. Within IPM programmes chemicals were generally chosen on their efficacy or cost effectiveness, rather than on their environmental impact (Kovach et al., 1992). This situation was undoubtedly because additional information that would allow farmers to make choices that would include environmental considerations was not available. Methodologies, such as those presented by Kovach et al., (1992) and Penrose et al., (1994), have sought to redress the imbalance in the informational deficit by supplying IPM practitioners, as well as those farmers wishing to reduce the environmental impact of their operations, with models that explicitly compare the potential for environmental impact of available pesticides, and allow for the calculation of a measure of pesticide impact at an individual farm level.

IPM practitioners have identified additional uses of models such as the EIQ and PI approaches. In response to public concern over pesticide residues on food and groundwater contamination the Australian Apple and Pear Growers Association recently made pesticide reduction its number one policy issue (Penrose et al., 1994). Individual farmers, however, may only adopt such a strategy if there are commensurate benefits associated with a modification of pesticide use behaviour, perhaps in the form of a price premium on IPM produced food.

This in turn can only take place if there is some threshold by which produce can be formally classified as complying with acceptable pesticide use patterns (Penrose et al., 1994). In Europe, Italian apple producers are currently demanding a formal definition of what is, and what is not, IPM produced food. Italian apple producers in the Sud Tyrol region (northern Italy) claim that 25% of their apples are produced under a system of IPM, and are marketed and labelled as such, and suggest that other apple producing regions in Europe are making similar claims whilst not actually practising IPM (Eurofruit, July, 1995). There are potentially clear marketing advantages to the use of IPM as a selling point, but there are at present no Union wide standards against which IPM is classified. One method of achieving such a classification might well be in the use of models such as EIQ, EIP and PI ranking indices. Management strategies can be given an "impact score" as an acceptable threshold of pesticide use. Produce scored as falling on or below this threshold can be accredited as IPM, but produce falling above the threshold cannot be classified as IPM (Penrose et al., 1994, Levitan et al., 1995, Quin and McGgregor, 1995).

Another potential role for models such as the EIQ, EIP and PI approaches is in the setting of pesticide taxes. Once again an acceptable threshold of pesticide use can be identified based on an EIQ, EIP or PI score, and pesticides used in excess of this threshold can be subject to a pesticide tax (Quin and Edwards-Jones, 1997). This scenario is currently being considered in The Netherlands using the "environmental yardstick" approach (Groot, 1996). The difficulty here would be in identifying an impact threshold that would be acceptable to all members of the EU, and policing and administering the scheme once it was in place.

The use of pesticide impact models that assign single impact values to pesticides lends itself to the development of economic-environmental policy and decision making models (Verhoeven et al., 1994, Quin et al., 1996). One of the problems in the past in the development of such models has been how to combine economic data with non-monetary environmental data. The use of models such as the yardstick or EIQ approaches in Linear Programming or Goal Programming models avoids the need to express environmental costs and benefits as monetary variables, as trade-offs can be made within the model structure between environment and economics parameters. The output of such models can be used at a strategic level (as well as at a more localised level) for, amongst others, the purposes discussed above, but above all it affords decision makers the opportunity to explore varying strategies and assess multiple outputs.

The strengths of models such as the EIQ approach of Kovach et al., the EIP of Reus and Pak and the PI approach of Penrose et al., is in their simplicity to the end user. The farmer is presented with a single impact figure for a pesticide which can then be applied at the individual farm level using a field use rating (in the case of Kovach et al.,). These models are user friendly and (relatively) easy to understand by farmers, policy makers, and potentially the general public.

Ultimately, the model of Kovach et al., (1992) was chosen for this thesis to assess the impacts of pesticide use in selected apple producing regions. The model was applied to data regarding pesticide use patterns gathered during the duration of the project. The results are discussed in Chapter 6, and modifications to the approach discussed in Chapter 7.

4.6 Reasons for the choice of the EIQ approach

There were a number of reasons why the EIQ approach of Kovach et al., (1992), was chosen above other available methods to represent environmental impact in the model, but the main criteria of choice can be summarised as follows:

1. The EIQ is easy to use and to understand. Ease of use and more importantly ease of understanding, are a vital characteristic of both the EIQ model and modelling in general. If the end users of the model are

unable to interpret results then the model is to all intents and purposes useless.

2. The EIQ model is applicable across all geographical locations. All that is needed to fulfil the data demands of the EIQ model are pesticide use patterns (quantities used), and the type of pesticide used. Thus comparisons can be made across regions, crops and pest management strategies.
3. The EIQ model reduces environmental impact to a single figure for ease of comprehension, and ease of use and compatibility with other decision making models.
4. Unlike other pesticide impact models, such as that of Penrose et al., (1994), the EIQ model does not seek to trade-off environment against economics. This is potentially a serious flaw in Penrose et al., (1994) model as it allows farmers to choose pesticides on the basis of efficacy and cost above environmental impact.
5. The EIQ model merely describes potential impact without indicating what level of pesticide use is either optimal or acceptable. It is up to the decision maker, the end user of the model, to decide what level of pesticide use is acceptable to society as a whole. This is a great strength of the EIQ approach.

This Chapter has reported on an extensive literature review of existing pesticide ranking indices. From the wide range of models available to decision makers, 4 were chosen for further scrutinisation, and assessed for their suitability for use as aids to decision making with regard to pesticide use. From this detailed examination of 4 models, 1 was chosen for detailed application to pesticide use data from the EU apple growing sector. This model was the EIQ model developed by Kovach et al., (1992). Before applying this model to actual pesticide use data, however, pesticide use in the apple growing sector will be discussed in detail.

5. Pesticides in the Apple Growing Sector: Legislative Issues, Use Patterns, and Integrated Management Techniques.

5.1 Introduction.

The use of pesticides is justified by the extent to which they protect agricultural output (Pimentel et al., 1991, 1992, Beaumont, 1993, The Pesticide Trust, 1992, Carlson et al., 1994, Pearce and Tinch, 1997). The benefit of pesticide use is defined and measured by the value of the output that would otherwise have been lost if pesticides had not been used (Oskam et al., 1992, Carlson et al., 1994, Beaumont, 1993, Pearce and Tinch, 1997). In many areas of the world, especially in the developing countries, pesticide use is heavily subsidised in order to ensure agricultural output, with subsidies potentially leading to the excessive use of pesticides resulting in environmental damage (Pesticide Trust, 1992, Weale, 1992, Carlson et al., 1992, Pimentel, Kirkby and Schoff, 1993, Beaumont, 1993, Pearce and Tinch, 1997). This chapter will examine trends of pesticide use in Europe, highlighting the apple growing sector, and will also comment on the legislative framework within which pesticides are used and whether or not pesticide use is excessive at present levels.

5.2 The reasons for current pesticide use patterns.

The variation in pesticide use within and between Member States is probably due to the interaction of social, biological, economic and political factors presented in Figure 5.1.

Economic conditions in apple production dictate intensive pesticide use patterns in the apple growing sector. Initial investment in orchard establishment is high, with returns on that investment not beginning for between 5 and 10 years depending on apple variety (O'Rourke, 1994). The value of the crop is also high per hectare and the consumer demands high quality, and uniformity of appearance and taste. Pesticide use in the apple industry not only contributes to the security of supply, but also to the cosmetic acceptance of the product (Fenemore and Norton, 1985). The market does not tolerate apples with scars, blemishes and discoloration.

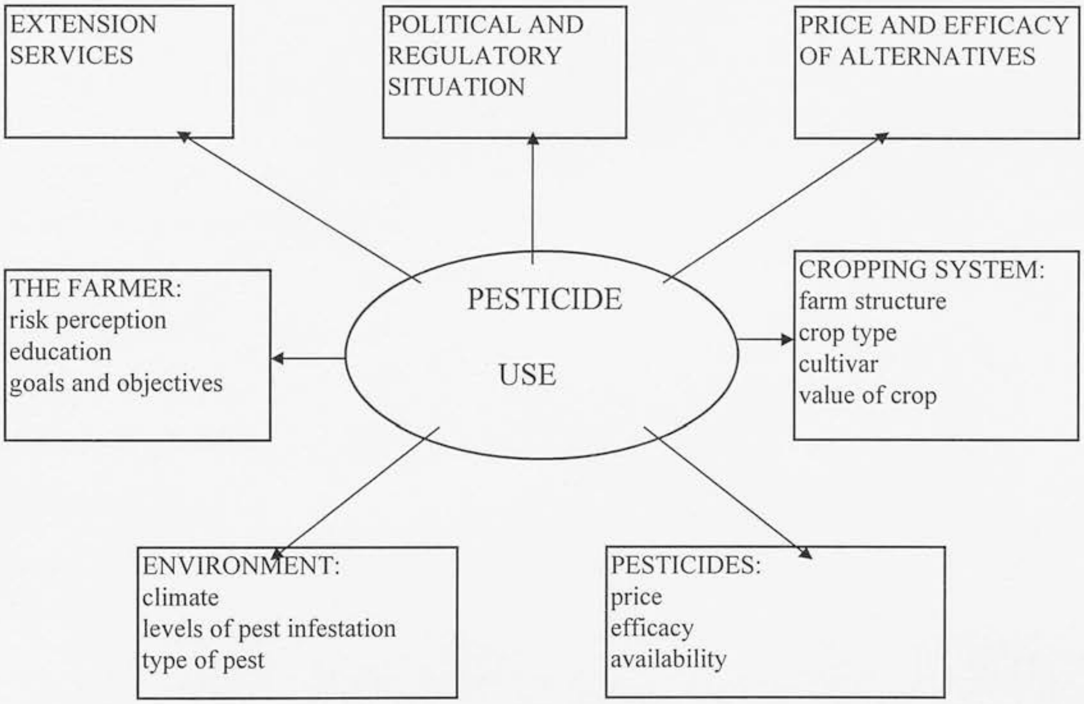
Agricultural activity has always resulted in pollution to a certain extent (Conway and Pretty, 1991, Hanley, 1992). Since World War 2, however, agriculture has undergone considerable changes. Farms have become larger, more highly mechanised and more reliant upon synthetic fertilisers and pesticides (Conway and Pretty, 1991, Oskam, 1992, Hanley, 1992, The Pesticide Trust 1992, Beaumont, 1993). Materials once used as agricultural inputs, such as manure and straw, are now sometimes considered wastes (Conway and Pretty, 1991). Whereas the farmer was once seen as the custodian of the land, he is know perceived as contributing to habitat and wildlife destruction, and agriculture in general seen as a major source of industrial pollution (Hanley, 1992, CEC, 1992c).

5.3 Current pesticide legislation in the EU.

Plant protection products, or pesticides, are used in agriculture to control the wide variety of pests and diseases that ultimately affect crop production and preservation. If not used properly it is likely that they represent important risks to both man and the wider ecosystem (Moses, 1989, CEC, 1992c). Pesticides, however, are indispensable in an efficient agricultural industry, as they contribute directly to the current situation of plentiful food supply, a high quality food supply, and a reasonably priced food supply (Scheele, 1994, Hull, 1994, Johnen, 1995). Under current farming systems the absence of pesticides would cause significant food losses to occur, with food quality and prices also suffering (Scharpe, 1994, Pearce and Tinch, 1997).

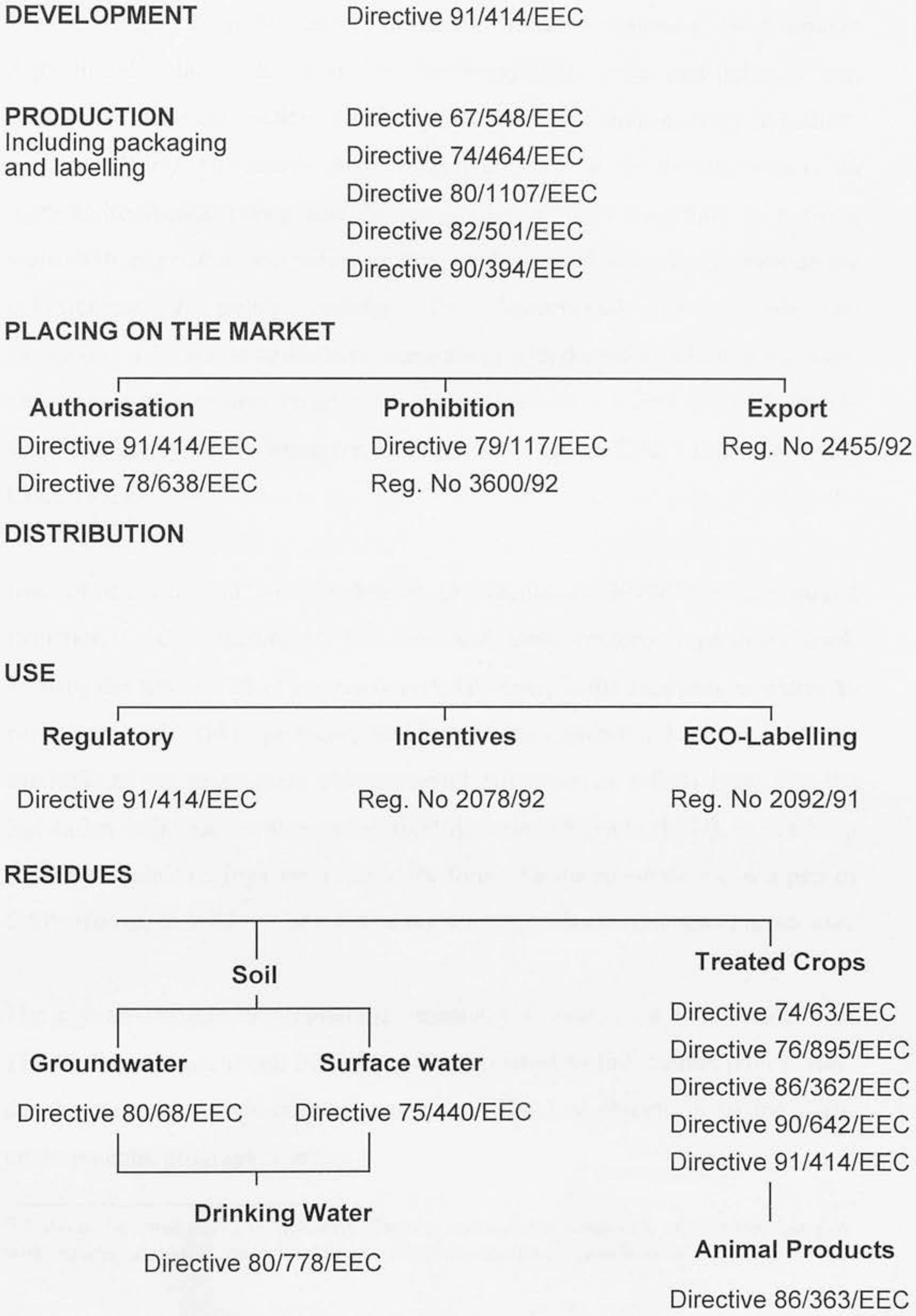
During the last five years the relaxation of trade barriers and the free movement of people and goods within the Community has further necessitated an harmonisation of pesticide regulation across the Community (CEC, 1992, Hull, Scharpe, 1995). European farmers are essentially operating within a market without national borders, allowing the free circulation of pesticides from country to country. In this situation it would be unacceptable to have strict regulation in Germany increasing pesticide prices perhaps, and lax legislation in Italy maintaining low pesticide prices. A rational farmer would undoubtedly take advantage of the freedom of movement of goods and purchase pesticides in Italy.

Figure 5.1. Factors involved in the choice of pest management strategy. Source: Adapted from Pimentel, Kirkby and Schoff (1993).



The new legislation, governing pesticide use in Europe, covers all aspects of pesticides, from the early stages of development to the placing of a product on the market, to use rates, residues on food, and environmental fate. The most important part of the legislation aims to reduce *risks,* and seeks to ensure through obligatory provisions that in all relevant stages of pesticide use that there is no risk to man or the environment (CEC, 1992c, Scheele, 1995, Scharpe, 1995, Johnen, 1995). Figure 5.2 illustrates the large number of documents covering all aspects of pesticide use. This next section will pick out the most important pieces of legislation and explain their contribution to the reduction of pesticide related

Figure 5.2. An overview of the current legislative situation with regard to pesticides in the EU: the main legislative instruments. Source: Adapted from Scheele, 1994.



risks to the environment in general. The legislation listed in Figure 5.2. refers to basic text only.

5.3.1 Agri-environmental policy.

Agri-environmental policy largely developed from the reforms to the Common Agricultural Policy (CAP) under the McSharry Plan, 1990, and links the two previously different policy areas (agriculture and environment) together[31] (Scharpe, 1995). The reason for the perceived need for the development of an Agri-environmental policy was the recognition of the inflexibility in existing legislation, regulation and policy making, and the need to explicitly include the environment in policy making. The fundamental step towards an environmentally sound agriculture, came about with the introduction of the *Agri-environmental Measures Programme,* under Regulation (EEC) 2078/92, Article 43 of the Treaty of the European Union concerning the CAP[32] (Scheele, 1994, CEC, 1992a).

One of the few binding stipulations of Regulation 2078/92 is a required reduction in the quantity of fertilisers and plant protection products used. Perhaps the key aspect of the regulation, however, is the financial incentive to participate with, and to promote, the ideals of the regulation. Limited funds are available as yet to promote environmental protection at a farm level, but the legislation is in itself a step in the right direction (Scheele, 1994). It has been argued that once savings are made in the form of reduced subsidies, as a part of CAP reforms, then this money can be diverted towards environmental protection.

The agri-environmental programme essentially constitutes a new direction of The Common Agricultural Policy (CAP), as it seeks to link market policy, rural development and environmental protection. The key objectives of the Agri-environmental programme are:

---

[31] It should be noted than The McSharry Plan was concerned not only with environment but also with reducing surpluses, trade distortions, and the distribution of farm income.

1. Integration of environmental requirements into the CAP.
2. A reduction in agricultural production resulting in beneficial environmental effects.
3. Improvements in agricultural income by granting the farmer an appropriate reward for the provision of environmental services.

(CEC, 1992C)

5.3.2 The Fifth Environmental Action Programme (FEAP).

Agriculture is one of the five main target areas of the FEAP[33], introduced in 1992, and adopted by Member States in 1993 (CEC, 1992C, Reus et al., 1994). The ultimate aim of the programme is to transform the patterns of growth in the Community in such a way that the path to a sustainable future can be followed (Reus et al., 1994). Indeed there is now a trend in all European Union policy toward environmental protection. Article 2 of the Maastricht Treaty[34] places the environmental imperative at the heart of all Community policy and development areas. Article 2 further calls for harmonious and balanced development of economic activities, sustainable and non-inflationary growth respecting the environment (Hull, 1994).

The central theme of the FEAP is one of European harmonisation and co-operation, with its main principles being:

a) a shared co-operation between Member States, given that environmental problems frequently involve transboundary issues,

b) a harmonisation of environmental policy and standards,

c) to improve the quality of life of European citizens, through the establishment of minimum, but high, environmental standards for drinking water, bathing water, air quality, and nature conservation,

---

[32] Other initiatives and policies have also impacted on Agri-environmental policy such as Agenda 21 of the Earth Summit, and the European Treaty concerning The Common Environmental Policy (Scheele, 1994).

[33] The other four target areas being Industry, Energy, Tourism, and Transport.

d) future Community development must meet the aims and objectives of European sustainability,

e) Member States have a duty to implement European environmental initiatives, and,

f) European environmental policy must be acceptable to its citizens, and has to be seen as being "common sense"

(CEC, 1992c, Reus, 1994, Hull, 1994).

To achieve its aims of environmental sustainability the FEAP advocates a mixture of policy instruments such as command and control; economic and fiscal instruments such as Pigovian Taxes (the polluter pays principle), financial support mechanisms, such as the establishment of Environmentally Sensitive Areas (ESA's) and horizontal instruments such as information and education initiatives.

Through the selection of agriculture as one of its five target areas, FEAP has recognised that agriculture has a significant role to play in future Community agri-environmental policy[35]. It recognises that great leaps have been made in agricultural efficiency, and the supply of high quality, cheap food supplies (and the associated social welfare that this brings). However, changing agricultural practices (especially since World War 2) have lead to the over exploitation and degradation of the natural resources such as soil, water and air, upon which agriculture ultimately depends (CEC, 1992c).

Specifically with regard to the use of agro-chemicals, the FEAP has as its main objective the maintenance of the basic natural processes indispensable for a sustainable agricultural sector notably by the conservation of water, soil and genetic resources. In particular it calls for decreases in the input of chemicals to the point where none of these processes are affected and for the development of

---

[34] Signed at Maastricht, The Netherlands, 7th February, 1992.

[35] 80% of land in the EU is under agricultural use (circa 1994).

an equilibrium between inputs of nutrients and the absorption capacity of soils and plants (CEC, 1992c, Reus et al., 1994, Hull, 1994).

To achieve this the FEAP proposes to set targets for (amongst others) a reduction of nitrates in groundwater, increases in the use of organic materials, "significant" reductions in pesticide use and the introduction of integrated methods of crop management (CEC, 1992c). The FEAP, however, is not a blueprint for a pesticide policy, but rather pesticide use is one compartment of an overall agri-environmental policy, the aim of which is to improve the sustainability of European agriculture.

5.4 An overview of pesticide use in apple production

Data from the sales of pesticides are available from government statistics and national industry associations (Brouwer et al., 1994). Total sales of pesticides in Europe are now in excess of 6 billion ECU per year (Beaumont, 1993). The major markets for pesticides in the EU are France, Germany and Italy, with these countries accounting for approximately two thirds of all sales of formulated product (Brouwer et al., 1994, Beaumont, 1994). The use of pesticides is positively correlated to the output from crop production. Countries with high outputs per hectare also tend to have high pesticide inputs per hectare. The use of pesticides per hectare (excluding glass houses) is highest in areas with intensive horticulture, such as northern Italy, southern France, South-East Spain and the Netherlands (Brouwer et al., 1994). Table 5.1 shows the EU pesticide market by crop, region and pesticide type for the year 1994.

On a global scale the market for pesticides in 1994 was $27,825 million, a 10.1% increase over the previous year (Pesticide News, No 28, 1995). Western Europe maintained its status as the biggest exporter of agro-chemicals, holding 46.7% of global exports in 1994 (38.4% in 1993). Table 5.2 summarises the EU pesticide market.

Table 5.1. The global pesticide market in terms of crop type, region and class of pesticide, 1994. Source: The Pesticide News, No 28, 1995.

By Crop %		**By Region %**		**By Type %**	
Vegetables	24.7	N. America	29.8	Herbicides	42
Cereals	14.2	E. Asia	28.1	Insecticides	28.8
Rice	13	W. Europe	24.2	Fungicides	19.5
Maize	11.2	S. America	9.2	Others	9.7
Cotton	10.2	E. Europe	3.4		**100**
Soya beans	8.4	Rest	5.3		
Sugar beet	2.8		**100**		
Rapeseed	1.7				
Others	13.8				
	100				

Annual sales of pesticides in the EU are estimated to be in the region of 350 million kg of active ingredient, with Italy and France accounting for half this figure. There is a great regional variation in pesticide use patterns throughout Europe with 3kg per hectare used in Denmark, to over 10kg per hectare in Belgium (Brouwer et al., 1994).

Although the values and quantities of pesticides used in fruit production within the EU is less significant than on a global scale, at the local and regional level pesticides used in fruit production may be important (Brouwer et al., 1994). Pesticide use at a sectoral level reveals great variation in expenditure (annually) at a per hectare level, with the price paid for pesticide used on barley being 41 ECU/ha per year, grapes at 60.6 ECU/ha per year, fruit production with 321.1 ECU/ha per year and flowers and ornamentals at 721.6 ECU/ha per year (Table 5.3.).

5.4.1 A regional analysis of pesticide use

For the purposes of this analysis one representative region was selected from each of the European Union countries. The regions chosen for the analysis were those with the most intensive apple production in that country. No data were available from either Denmark or Eire, with financial and time constraints ruling out the execution of field surveys. Denmark was assigned the same data as Germany and Eire the same as the UK.

Data collection was the most serious constraint on the analysis, as data promised by DEAC project participants at the beginning of the project was not forthcoming. The data collection period, therefore, was eighteen months as opposed to the three months originally anticipated. The most difficult task in data collection was in the identification of the most appropriate organisations to contact. In the case of Greece, however, an orchard field survey was conducted with the University of Thessalonika to identify pesticide use patterns in Greek apple orchards. The reason why only one representative region was chosen for each country was simply that detailed pesticide use data does not exist at the local level for Europe.

Table 5.2. EU pesticide market by country in million ECU (1991). Source: Adapted from Brouwer et al., 1994.

Country	Million ECU
France	2204
Germany	929
Italy	728
UK	594
Spain	509
Netherlands	231
Denmark	190
Greece	141
Belgium	136
Portugal	92
Eire	48
Luxembourg	15
Total	**5817**

Table 5.3. Annual expenditure on pesticide use in the EU in ECU/ha for different crops (averages for the years 1989,1990,1991). Source: Adapted from Brouwer et al., 1994.

Crop	ECU / ha
Soft Wheat	73.2
Barley	41
Potatoes	122.4
Sugar Beet	141.5
Rape and Turnip Rape Seed	93
Tomato	219.5
Other Vegetables	192.5
Flowers / Ornamentals	721.6
Grapes (Wine)	60.6
Grapes (Non-Wine)	115.6
Apples / Peaches / Pears	321.1
Other Fruits	104.4
Citrus	156.9

Pesticide use in European apple production is generally intensive (Winter, 1986, 1989), with an average total formulated product use of 74 kg/ha per year, and an active ingredient use of (a.i.) of 40 kg/ha per year. The highest use rates are France (110 kg a.i./ha) and Italy (63 kg a.i./ha), whilst the lowest use rates are for Germany (9 kg a.i./ha) and Belgium (15.24 kg a.i./ha) (Quin and Edwards-Jones, 1997) (Table 5.4).

The figures presented in Table 5.4 are interesting as it is widely perceived that pesticide use in Germany, Belgium, The Netherlands and the UK are intensive (Brouwer et al., 1994, Beaumont, 1993), and yet these countries display use rates at the lower end of the survey range. It may be expected that a high use rate for France, Italy, Spain would be observed, as these are regions with intensive apple production practices (Brouwer et al., 1994). Table 5.5 Indicates which European regions were selected for the analysis and where pesticide use data was accessed from. Again, those regions where pesticide use data were readily available were used.

Table 5.4. Pesticide use on apples in selected European regions for the year 1994. Source: Various (see Table 5.5). Appendix 1 details pesticide use data at compound level.

COUNTRY	Kg / Ha	Kg a.i. / Ha
Italy	94.5	62.66
France	179.35	110.13
Portugal	64.4	37.64
UK	42.1	20.04
Spain	135.2	60.66
NL	36.4	16.25
Greece	64.04	30.3
Belgium / Lux	29.94	15.24
Germany	17.93	9.34
Denmark No Data		
Eire No Data		
EU	**73.76**	**40.25**

Table 5.5. Regions studied and pesticide data source.

Country	Region	Data Source
UK	South-East	Data from ADAS, and marketing organisations English Apples and Pears and ENFRU.
France	Tarn and Garonne	Centre de Economie Rurale and Garonne, CEMEGREF.
Italy	Emilia-Romagna	Confcooperative, Ferrara (marketing organisation). Personal communication on visiting apple co-operatives in Northern Italy.
NL	Whole Country	Landell Mills, Market Research Organisation.
Germany	South	Dr Bernhard Sessler, University of Hochenheim, unpublished report.
Belgium	Whole Country	Dr Peter Jaeker personal communication.
Luxembourg	n/a	Luxembourg is generally taken to be the same as Belgium. Apple production is so small that this is acceptable to this study.
Eire	n/a	Eire taken to be the same as the UK. No data available.
Denmark	n/a	Taken to be the same as Germany. No data available
Greece	Whole Country	SAC - University of Thessalonika orchard survey.
Spain	Cataluna	Apple co-operative, Costa Brava. Personal communication on visits to the region.
Portugal	Central	Ministry of Agriculture. Personal communication on visits to co-operatives in the region.

The type, use rates and number of, individual compounds also varies from country to country. For example, Italy uses 10 compounds, but at a relatively high rate per compound, France uses 22 compounds again at a high rate for certain compounds, whilst The Netherlands uses 19 compounds at relatively low use rates (see Appendix 1).

The data gathered for each of the apple producing regions studied was applied to one particular pesticide impact methodology, the EIQ, in an attempt to quantify the environmental impacts of pesticide use in the apple industry. This will be discussed further in Chapter 6.

5.5 Discussion

Pesticide use in the apple industry has been categorised as intensive (Brouwer et al., 1994, Beaumont, 1994). There are environmental implications associated with pesticide use, but as yet there is no single methodology for identifying environmnetal impacts, and identifying the environmnetal benefits of pesticide reduction policies acceptable to all. Chapter 6 applies pesticide use data from across the EU to the EIQ model of Kovach et al., (1992), to see if comparisons can be made of environmnetal impact between regions.

6. A Pesticide Rating Index for Assessing Environmental Impacts: Methods and Results.

6.1. Introduction

The analysis of the external impacts of pesticide use is highly problematic without accurate data concerning toxicity and (environmental) longevity of pesticides, and the ecological consequences of environmental contamination. Such analysis generally entails a lengthy and in-depth site specific field survey of the impacts of pesticide use on sample organisms, such as birds, voles or invertebrates, and epidemiological laboratory studies of toxicity to humans, again using rats or mice (Conway and Pretty, 1991, Bergman and Pugh, 1994, Copin et al., 1994). In many cases data acquisition at this scale is highly problematic, and the lack of such data has traditionally been one of the major constraints in applying economic instruments to environmental problems. In light of these difficulties there may well be a need for simplified models of environmental impact that have low data requirements but can still aid the decision making process.

The aim of this Chapter is to apply pesticide use data from the EU apple industry discussed in Chapter 5, to the chosen pesticide ranking index from Chapter 4, the EIQ model. The methodological approach adopted and the key results of the analysis are discussed. The purpose of this exercise is to ascertain whether or not the EIQ model can be a useful addition to the decision making process, with regard to pesticide use, at the farm level.

6.2. Methods

6.2.1: The EIQ Model.

The EIQ model seeks to inform farmers about the environmental consequences of their actions in relation, firstly, to individual compounds (Table 6.1. Figure 6.1.), and secondly to pest management strategies (Table 6.2). Initially each pesticide is given an EIQ value based on the relative toxicity of the compound in relation to; dermal and chronic toxicity, systemicity, fish toxicity, leaching and surface loss potential, bird and bee toxicity, soil and plant surface half life and beneficial arthropod toxicity (see Chapter 4). This is expressed in a single (impact) figure as shown in Table 6.1, the total EIQ. An EIQ figure of 12 to 30 represents generally low hazard, 31 to 60 medium hazard, and 61 up to a possible 104.4 (in the case of insecticides) representing potentially high environmental hazard levels.

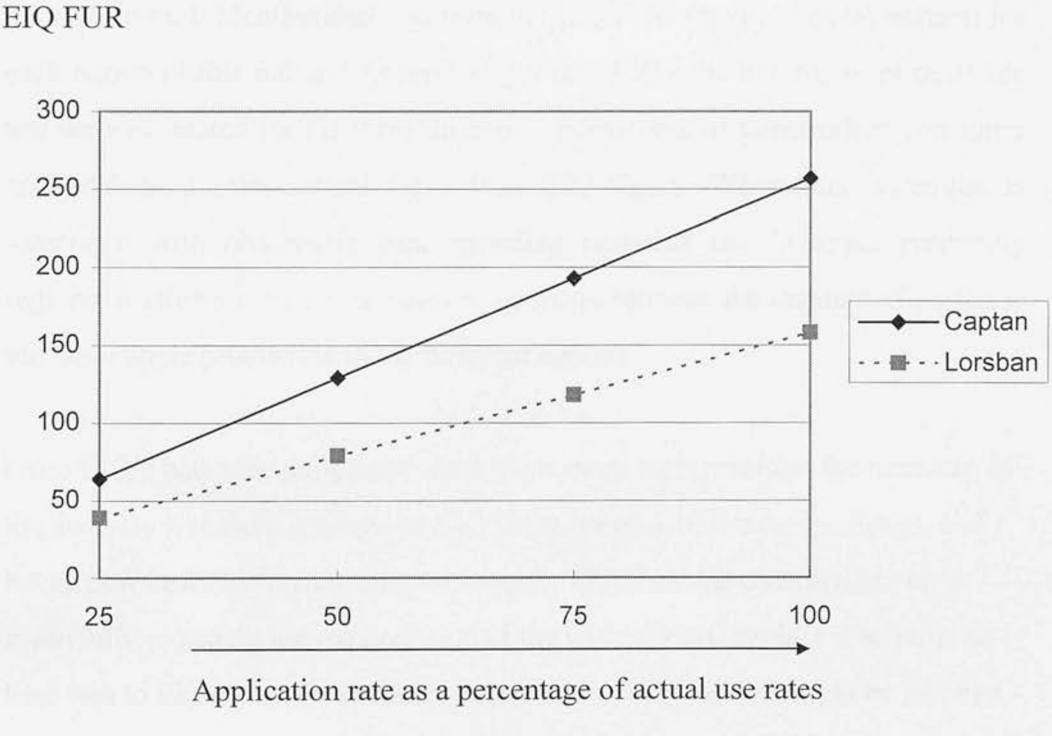
Table 6.1 An example of how the EIQ is derived using data for the fungicide *captan*.

Attributes	EIQ
Applicator effects	10.0
Picker effects	6.2
Total farmworker component	**16.2**
Consumer effects	4.1
Groundwater effects	1.0
Total consumer component	**5.1**
Aquatic effects	5.0
Bird effects	6.2
Bee effects	9.3
Beneficials effects	38.3
Total ecological component	**58.7**
Total EIQ	**26.7**

The final EIQ value consists of three sub-components; a farmworker, consumer and ecological or environmental component. Farm worker risk, for example, is defined by the sum of applicator exposure, plus picker exposure times the long term health effect or chronic toxicity. The overall impact of a pesticide, the EIQ, will be the average of all of the identified potential impacts, that is the average of the farm worker, consumer and ecosystem impacts (Kovach et al., 1992). This thesis, however, also examines the impacts for each of the three components individually. Each aspect of the analysis is given equal weight in the final calculation of EIQ value, but within each component factors are weighted differently to allow for an expression of long term over short term health effects, (see Chapter 7 for a further detailed explanation of weighting).

The subject of weighting factors is a potential source of both controversy and error in the EIQ calculations. Coefficients used in the equation to give additional weight to individual factors are based on a 1-5 scale, factors carrying the most weight are multiplied by 5 and those carrying the least weight multiplied by 1 (Kovach et al., 1992). Clearly if this model, or a hybrid model were to be advanced then the notion of weighting has to be examined very closely. Further the EIQ as it stands at the moment suggests a linear damage function. The validity of such a damage function has not yet been tested, but is discussed further in Chapter 7.

Figure 6.1. Comparing the environmental impact of two different pesticides: the EIQ Field Use Rating (EIQ FUR) of Capstan and Lorsban.



If the EIQ of individual compounds applied to a crop are summed, the total EIQ for the whole crop may be calculated. To account for different active ingredient percentages, different frequencies of application and different application patterns, Kovach et al., (1992), developed a simple equation: the EIQ Field Use Rating (FUR). The rating is achieved by multiplying the EIQ value of the pesticide, by the percentages of active ingredient, and by the quantity applied per hectare (kg/ha) to give:

$$\text{EIQ FUR} = \text{EIQ} \times \% \text{ a. i.} \times \text{Rate}$$

or, for captan once again to give:

$$\text{EIQ FUR} = 26.7 \times 0.83 \times 10 = 221.6$$[36]

Thus, a total figure for hazard can be assigned to each orchard, or management strategy by summing the EIQ values for each compound used as part of that strategy (see Table 6.2 and 6.3).

6.2.2. Data Collection.

The EIQ values of Kovach et al., were combined with observable pesticide use patterns in each Member State, to form the EIQ FUR (environmental impact) for each region (Table 6.6 and Appendix 1). From EIQ's the hazards from pesticide use were estimated for the three different components of farmworker, consumer and ecological effects; and for a total EIQ figure. When this technique is integrated with observable data regarding pesticide use in apple producing regions, it allows for a comparison to be made between the impacts of pesticide use from apple production in the different regions.

Once EIQ's had been assigned to existing management practices the next step in the analysis was then to assign an EIQ value for each of the new varieties, and hence new pesticide management strategies. This exercise was carried out essentially to satisfy the requirements of the wider DEAC project. The purpose here was to illustrate how environmental impact (EIQ values) might be reduced as a result of the introduction of disease resistant apple varieties, thus assessing the effectiveness of plant breeding projects or environmnetal protection policies. However, as no data were available regarding the likely pesticide use characteristics of the new variety, a series of different scenarios were examined (Table 6.7, Appendix 2). Here pesticide quantities were varied in order to

---

[36] Data from France, Table 6.2.

identify changes in EIQ value in response to changes in pesticide use patterns which could occur if different breeding strategies resulted in reduced pesticide applications. This resulted in additional environmental impact coefficients being assigned to each region, again dependant on pesticide type and use rates, so that before and after breeding comparisons could be made of environmental impact. The new coefficients of environmental impact were dependant on the assumptions made about the production characteristics of the new variety. Within each of the Member States, different pesticide use strategies, and different pesticide reduction patterns, were modelled to examine the impact of adoption of the new apple varieties on the overall EIQ FUR. Assumptions regarding the characteristics of the new varieties were based on pesticide reduction patterns of between 25 and 75 % of current use patterns[37]. Experiments in Switzerland have shown that reduction of pesticide us on this scale are achievable, whilst still maintaining the economic viability of the crop (Markus Kellerhals, pers. comm. 1996).

The data used for the new varieties can be used for illustrative purposes only. Pesticide use rates were to have been made available for new varieties for this thesis but were not forthcoming. Thus, pesticide use reductions based on disease resistant apple varieties already in existence were applied (Wauchope, 1992, Winter 1986, 1989). Advice was also taken from members of the DEAC project who had experience in introducing new varieties of apple onto the market.

Variations in pesticide use ranged from minor changes in fungicides only, to reflect the disease resistant characteristics of new apple varieties, to a more general pesticide reduction pattern. It was assumed that if reductions could be made in fungicides then there may be a desire to move towards IPM production resulting in reduced quantities of herbicides and insecticides also. Contrasting pesticide use strategies were modelled within regions to illustrate the changes that might take place in overall EIQ values. Whilst this represents an interesting

---

[37] Some, however, would argue that the contribution of disease resistant apple cultivars to pesticide reduction at the orchard level is minimal (Penrose, 1994).

feature of the analysis, the modelling of new apple varieties was less important than the overall application of the EIQ model to actual pesticide field use data, as data concerning the new apple varieties were entirely hypothetical. A summary of the pesticide use characteristics of the new apple varieties can be seen in Appendix 2.

6.3 Results.

From Tables 6.2 and 6.3. it is clear that (according to the EIQ model) the environmental impact of pesticide use in apple production is greater in France, with a EIQ OF 5453, than in the UK, with an EIQ of 718. It is also clear why this might be the case. Firstly, pesticide use in France is in the region of 4 times that of the UK. This is due to both the number of compounds used and the total quantities of each compound used. The extremely high EIQ FUR attributable to France, however, is explained by the use of just two compounds. Sulphur, with an % a.i. of 0.8, use equivalent to 40 kg/ha, and an individual EIQ value of 45.5, giving a field use rating of 1456. Thiram, with a % a.i. also of 0.8, at the rate of 44 kg/ha, and an individual EIQ of 54.5, results in a EIQ FUR of 1918. These two compounds alone account for 61% of the total potential for environmental impact for France. For the UK, the highest ranking compound in terms of EIQ FUR is captan, with a % a.i. of 0.8, a use rate of 9.8 kg/ha, and an EIQ of 28.6, resulting in a figure of 224.2 for the EIQ FUR.

It could be argued that those member states displaying high levels of environmental impact tend to be heavy users of one or two compounds, resulting in high EIQ FUR scores (see Appendix 1). Sulphur, a pesticide used in organic agriculture, in particular has a negative effect on overall EIQ FUR score, due to the generally large quantities of the compound used, high %a.i. and a medium range individual EIQ score. From Tables 6.2 and 6.3 the guarded assumption can be made that apple production in the UK is currently less environmentally damaging than apple production in French orchards.

Table 6.2. Pesticide use and EIQ (per hectare) for France for one season[38].

Compound	EIQ	%A.I.	Rate	EIQ FUR
sulphur	45.5	0.8	40	1456
thiram	54.5	0.8	7.5	327
captan	28.6	0.83	10	237.4
mancozeb	62.3	0.8	10	498.4
triadimefon	33.3	0.25	2	16.6
thiram	54.5	0.8	44	1918.4
benomyl	69.5	0.5	2	69.5
metiram	55.9	0.75	6	251.5
metalaxyl	29.2	0.45	2	26.3
copper h'ox	33.3	0.08	26	80
gusathion	43.1	0.85	2	73.3
lannate	32.8	0.2	4	26.2
vamidothion	37.7	0.4	1.3	19.6
azinphos	43.1	0.5	2	43.1
hexakis	12.8	0.1	0.5	0.6
methidathion	69.3	0.4	2	166.3
dichlorvos	40.6	0.5	6	121.8
amotrole	37.1	0.23	6.5	55.5
roundup	32.4	0.15	1.3	6.3
simazine	15.7	0.5	3	23.6
2-4-D	56.3	0.5	1.25	35.2
TOTALS			**179.35**	**5453**

Table 6.3. Pesticide use and EIQ (per hectare) for the UK for one season.

Compound	EIQ	%A.I.	Rate	EIQ FUR
Dithianon	35.9	0.75	2.2	59.2
captan	28.6	0.8	9.8	224.2
pyrifenox	34.9	0.2	1.8	12.56
bupirimate	41.2	0.25	4.4	45.3
lorsban	52.8	0.48	3	76.0
pomex	22.6	0.5	8.2	92.7
dicamba	38.7	0.215	5	41.6
amitrole	20.5	0.225	5	23.1
simazine	15.7	0.5	1.7	133.3
diuron	20.5	0.5	1	10.2
TOTALS			**42.1**	**718**

---

[38] Use rate represents one growing season but could be made up of several applications. The quantity reported in the Rate column of Tables 6.2 and 6.3 are total applications per season.

Kovach et al., (1992), went on to classify different EIQ levels as being representative of different management strategies. Table 6.4 shows a theoretical comparison between management strategies (Kovach et al., 1992). Thus, using this approach farmers can identify which compounds are more or less harmful to the environment, allowing them to adopt pesticide use strategies that may correspond to IPM.

Table 6.4. Theoretical comparison between three pest management strategies, conventional, IPM and organic, for one growing season.

Conventional strategy				
Compound	**EIQ**	**a.i.**	**Rate**	**EIQ FUR**
Rubigan	27.3	0.12	2.4	8
Captan	28.6	0.5	18	257
Lorsban	52.6	0.5	6	158
Thiodan	40.5	0.5	3	61
Guthion	43.1	0.35	4.4	66
Cygon	74	0.43	6	191
Omite	42.7	0.68	4	116
Kelthane	29.9	0.35	3	47
Sevin	22.6	0.5	3	34
Total environmental impact				**938**

IPM Strategy				
Compound	EIQ	a.i.	Rate	EIQ FUR
Nova	41.2	0.4	1.2	20
Captan	28.6	0.5	3	43
Dipel	13.5	0.06	4.5	4
Sevin	22.6	0.5	3	34
Guthion	43.1	0.35	4.4	66
Total environmental impact				**167**

Organic Strategy				
Compound	EIQ	a.i.	Rate	EIQ FUR
Sulfur	45.5	0.9	42	1720
Pyrethrin	25.5	0.04	72	73
Ryania	55.3	0.001	116	6
Total environmental impact				**1799**

Source: Adapted from Kovach et al., 1992.

Table 6.5 details the situation with regard to EIQ values for each region analysed, with Appendix 1 displaying this data at compound level, Table 6.7 shows how different pesticide use rates, corresponding to the new varieties, impacted on the final EIQ figure. This exercise was carried out for each of the regions examined. The results displayed in Tables 6.6 and Figure 6.2 suggest that data regarding the environmental impact of pesticide use is achievable using the EIQ model, and Table 6.7 examines changing EIQ FUR in response to new varieties using less pesticides.

Table 6.5 indicates that for the apple producing regions chosen for the study, Germany has the lowest EIQ at 386, and thus lowest potential for environmental impact, whilst France has the greatest EIQ at 5543, and thus the greatest potential for environmental impact. On closer examination of Table 6.6 and Appendix 1, however, some interesting features of the analysis become apparent. For example, it can be shown that Spain uses twice as much total formulated product (135 versus 64 kg/ha), and approximately 38% more active ingredient (61 versus 38 kg a.i./ha) than Portugal, and yet Spain's final EIQ value is less than that of Portugal. The final EIQ figure of Spain is 2359, against a figure of 2697 for Portugal.

The explanation for this difference is apparent from an examination of the use of the individual compounds used at the orchard level. In all respects the EIQ rating for Portugal is less than that of Spain except for one compound: parathion. This alone accounts for 1919 EIQ points, 71% of Portugal's total EIQ rating, and almost 50% of total formulated product. So although more pesticides are used in Spain, in general they tend to be less harmful according to the Kovach model. The only real exception with Spain is the use of 22.5 kg/ha of captan, yielding 322 EIQ points (approximately 14% of Spain's total EIQ rating) (see Appendix 1). If Portugal were to reduce the use of parathion, or replace it with a less "harmful" pesticide, then it could drastically reduce its potential for environmental impact. Figure 6.2 compares EIQ Field Use Rating for existing and new varieties across regions. Again, France, Italy and Portugal stand out as having the highest level of potential impact from pesticide use.

Figure 6.2. Summary of European Union EIQ's for existing and new varieties.

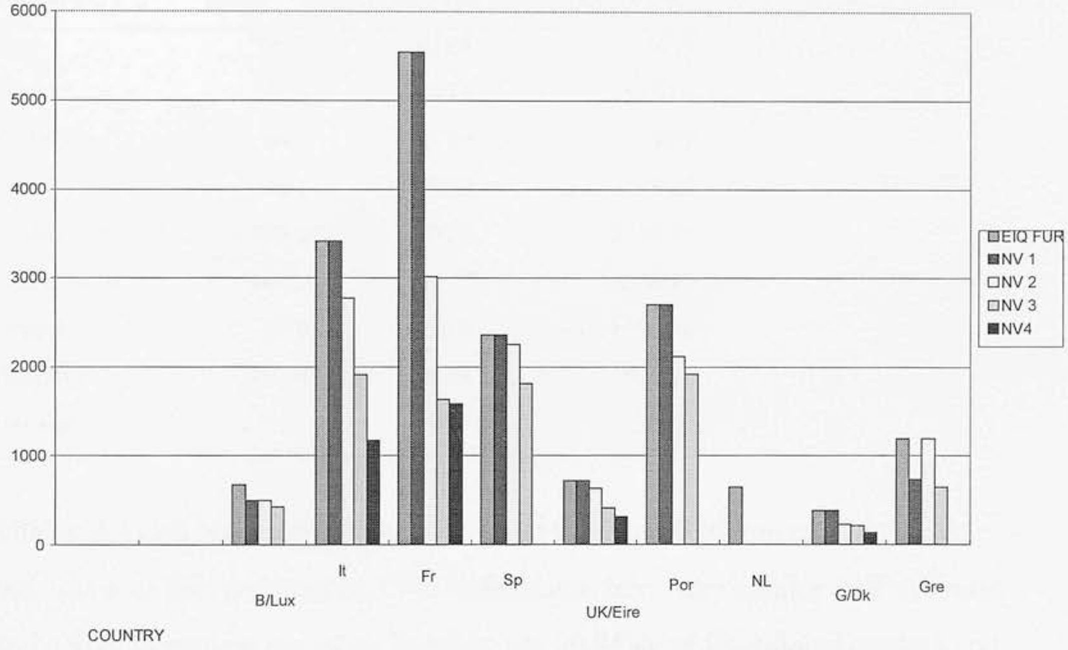


Table 6.5. EIQ values (per hectare) at current pesticide use rates in the European apple industry for 1994.

Country	Total EIQ
Germany/Dk	386.06
Netherlands	651.52
Belgium/Lux	678.75
UK/Eire	718.48
Greece	1190.86
Spain	2358.94
Portugal	2697.01
Italy	3418.77
France	5542.64

Table 6.6. EIQ Field Use Rating by Country. The EIQ figure is at the per ha per annum level.

PRODUCT	kg/ha	%AI	kgAI/ha	EIQ FUR
Italy	94.5		62.66	3435
France	179.3		110.13	5452.6
Portugal	64.4		37.64	2697
UK	42.1		20.04	718.27
Spain	135.2		60.66	2358.94
Netherlands	36.4		16.25	638.85
Greece	64.0		30.3	1190.86
Belgium	29.94		15.24	678.75
Germany	17.9		9.34	386.31

The model also seems to be consistent in its findings. With reference to Table 6.6, it is seen that Belgium and The Netherlands have very similar total use rates and active ingredient use rates. Belgium use 29.94 kg of formulated product and 15.24 kg a.i. / ha, whilst The Netherlands uses 36.4 kg and 16.25 kg respectively. Both countries have a very similar EIQ rating, with Belgium at 678 and The Netherlands at 638. It may be concluded from this is that both countries are using similar quantities of similar compounds, and neither uses one or two "harmful" compounds in large quantities. This assumption is indeed borne out by an analysis of the compounds, and EIQ's, for both Belgium and The Netherlands.

Table 6.7. The effect of reduced pesticide use (NV1, NV2, NV3, NV4) on EIQ FUR.

Country	EIQ FUR	NV 1	NV 2	NV 3	NV4
B/Lux	678.7	496.0	496.0	424.3	0.0
It	3418.8	3418.8	2767.1	1913.1	1178.2
Fr	5542.6	5542.6	3014.2	1633.1	1587.4
Sp	2358.9	2358.9	2252.1	1810.6	0.0
UK/Eire	718.5	718.5	633.2	411.1	314.6
Por	2697.0	2697.0	2115.6	1921	0.0
NL	651.5	0.0	0.0	0	0.0

G/Dk	386.1	386.1	226.6	215.0	135.3
Gre	1190.86	732.8	1190.86	645.37	0

The EIQ model of Kovach et al., (1992) has yielded some interesting and encouraging results. These can be summarised as:

1. The Kovach model allows for comparisons to be made of the environmnetal impact of pesticide use in the apple industry between regions.

2. The model flags up those pesticides that might be targeted for replacement based on their adverse environmnetal affects.

3. Thus, the model provides farmers with information which they currently do not have access to.

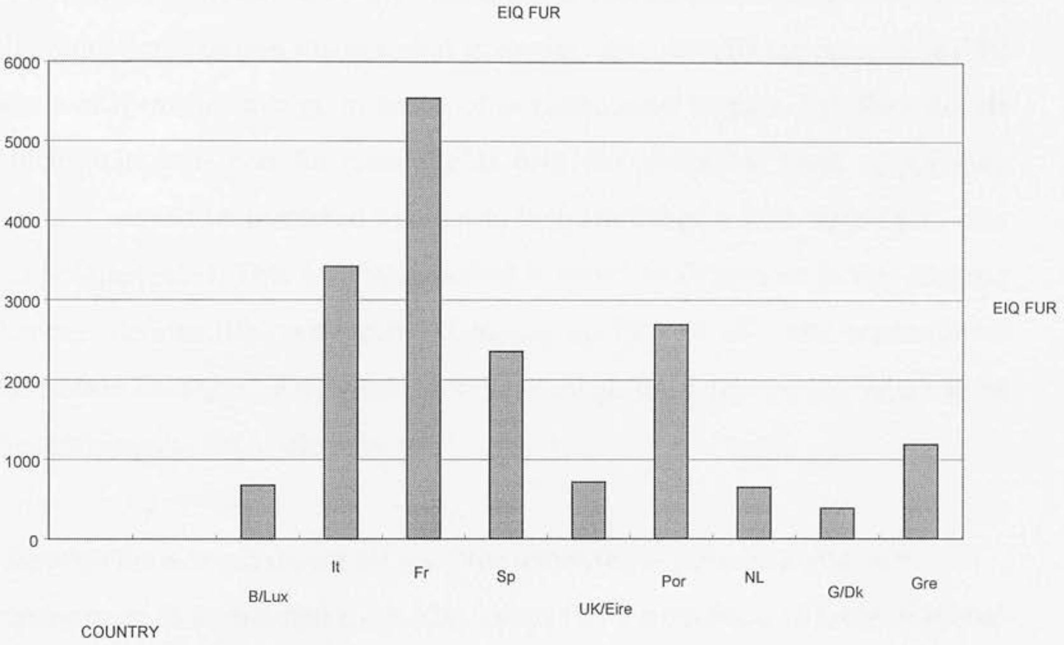
4. If pesticide use data exists throughout the EU, it is possible to compare the environmnetal performance of agricultural regions using the Kovach model.

5. Again, if pesticide use data exists it should be possible to monitor the effectiveness of policies aimed at environmnetal protection in agriculture.

6. Environmnetal monitoring of pesticides should allow for the use of IPM, or Eco, labelling if agreed environmnetal standards can be set, based on the EIQ.

However, many potential weaknesses in the methodology also exist (Dushoff et al., 1994, Levitan et al., 1995). These have been summarised in Chapter 4, and will be re-examined in Chapter 7 in an attempt to propose a model that can potentially contribute to the task of environmental protection more effectively.

Figure 6.3. Comparison of EIQ values between regions.



From the results gained so far (Table 6.6 and Figure 6.3), it is evident that those countries with the greatest potential for environmental impact are France, Italy, Portugal and Spain, with the remainder of the countries displaying pesticide use patterns that appear to be more acceptable. This is interesting as pesticide use patterns, and thus potential for environmental impact, may be expected to be greater than reported in The Netherlands, Germany and southern England (Brouwer, 1994). This, however, may well be explained by the often wide variation in pesticide use patterns between orchards, within regions and between regions. It is also clear from Table 6.6 and Appendix 1 that the main objectives of the Kovach et al., (1992) model were shown to be attainable.

6.4 Discussion.

The EIQ model allows for two types of decision to be supported. Firstly, for example, if more than one pesticide is available to control the same pest, information can be presented showing which compound should be adopted based on least environmental impact (Figure 6.1). If farmers or decision makers have access to this kind of data, then the most appropriate option can be considered, i.e. they can take the decision to choose one pesticide over another based on environmnetal impact. Secondly, information can be presented examining the difference between pest management strategies, thus, farmers can examine a total pest management strategy in terms of environmnetal impact, and then decide which strategy is best for them (Table 6.4). An acceptable level of pesticide “impact” should be identified in order to increase the ease with which EIQ data can be interpreted. This will be discussed in some detail later on in this Chapter (Kovach defines IPM production as having an EIQ of 167, and conventional production having an EIQ value of 938, although these figures are meant to be illustrative only, see Table 6.4).

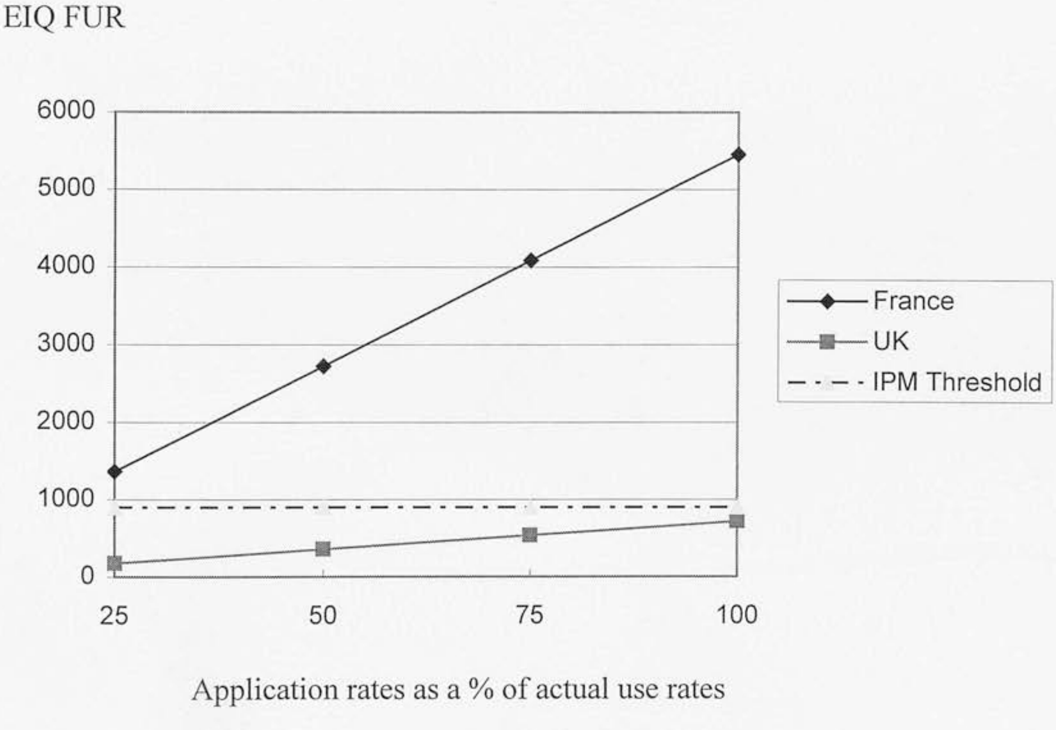
Although the Kovach model makes some tentative judgements about pest management strategies and target EIQ values (IPM production 167, conventional production 938) (Kovach et al., 1992), no optimal EIQ level is suggested. This, however, may well hinder interpretation of the model as without guidelines, or thresholds of damage, the results from using the EIQ model are difficult to visualise. In addition to this it has been suggested that the EIQ model can be used as an accreditation tool for the identification of IPM produced food. Again, however, no thresholds are presented to denote whether or not food can be classed as IPM.

If a threshold of damage could be identified, either for environmental protection, or for labelling IPM produce, then this would greatly enhance the understanding and impact of the model. If agriculturists have agreed on guidelines to work to, and were aware that they must not (or should not) exceed certain boundaries, then the EIQ model could well become a potentially important tool in the control of pesticide use at a farm and regional level, and / or become an important accreditation methodology.

To illustrate this consider the pesticide and EIQ data from France and the UK (Tables 6.2 and 6.3). If an arbitrary IPM threshold of 900 EIQ points were set, the UK could claim to produce apples which could subsequently be labelled as IPM. France on the other hand is exceeding the theoretical threshold by 5 times. Not only are they not producing IPM apples, but it is also likely that there would be a significant threat of environmental damage from pesticide use levels at this rate. Steps could therefore be taken to reduce EIQ value to a level closer to the IPM threshold (Figure 6.4).

It is only when thresholds are agreed that steps can be taken to move towards an agricultural practice that could be described as IPM. The EIQ model is not only useful in indicating when pesticide use rates are too high, but it also assists in the identification of solutions. Table 6.2, for example, shows that by either reducing the quantities used of the two pesticides sulphur and thiram, or by replacing them with less harmful compounds, France could significantly reduce its EIQ score, and perhaps move toward a point where fruit could be sold under an IPM label. The point of threshold setting will be discussed further in Chapter 7.

Figure 6.4. Total EIQ values for France and the UK showing theoretical IPM target EIQ value.



What this suggests is that the model presented by Kovach et al., is capable of making assessments based on the nature of individual compounds rather than on the total quantities used. Indeed this was one of the main stated aims of the EIQ model, that it could differentiate between those pesticides, and management strategies, that had the most harmful and least harmful environmental effects. This is very important when the chemical industry in general replaces older compounds with newer more powerful compounds which are efficient at much lower dose rates (Pearce and Tinch, 1997).

Chapter 7 will re-examine the methodology presented by Kovach et al., and suggest modifications that may make the model more acceptable to the various actors concerned with the agri-environmental system.

7. Modifying the EIQ model: addressing the criticisms.

7.1. Introduction

This chapter further discusses the EIQ model presented by Kovach et al., (1992)[39]. The purpose of the chapter is to suggest possible ways in which the model presented by Kovach et al., (1992), can be improved. The main criticisms of the model, raised in Chapter 4, will be addressed and the model redesigned in order that better use can be made of EIQ's at both the local level and by decision makers at a more strategic level. The Chapter examines whether or not it is possible to increase the acceptability of the model at all levels (farmer, policy maker and consumer). As stated in Chapter 4 the five main criticisms of the EIQ model are:

1. The linearity of the EIQ damage function.
2. The same 1-5 weighting criteria used for all categories of pesticide.
3. The use of the average of the farmworker, consumer and environmental component of the model to give the final EIQ value.
4. A spatial component allowing for site specific impact scores to be identified.
5. The method of data presentation.

Each of these criticisms will be addressed in turn, followed by a re-examination of the Kovach et al., (1992) model, based on the inclusion of changes in model presentation. EIQ figures will be assigned to pesticide use data (from the European apple growing sector) based on modifications to the model. An explanation will be presented of the benefits of the inclusion of additional factors and of altering the vehicle of presentation of the Kovach et al., (1992) model.

---

[39] Permission was sought and granted from Professor Joe Kovach (main author and designer of the EIQ model), to use the EIQ model and to change it, where necessary, for the purposes of this thesis.

7.2. The EIQ damage function

The damage function[40] described by the EIQ equation is linear. In terms of representing actual field conditions, the equation is, it is argued by some, unrealistic (Dushoff et al., 1994, Levitan et al., 1995, Teague et al., 1995). For example, the pesticide triadmefon (data from France) and the subsequent environmental impact (expressed as either EIQ, Field Use Rating or both) is expressed numerically in Table 7.1 and graphically in Figure 7.1 as:

Table 7.1. The EIQ field use rating (FUR)[41] for the pesticide triadmefon.

Product	EIQ	% a.i	Rate	FUR
triadimefon	33.3	0.25	2	16.6

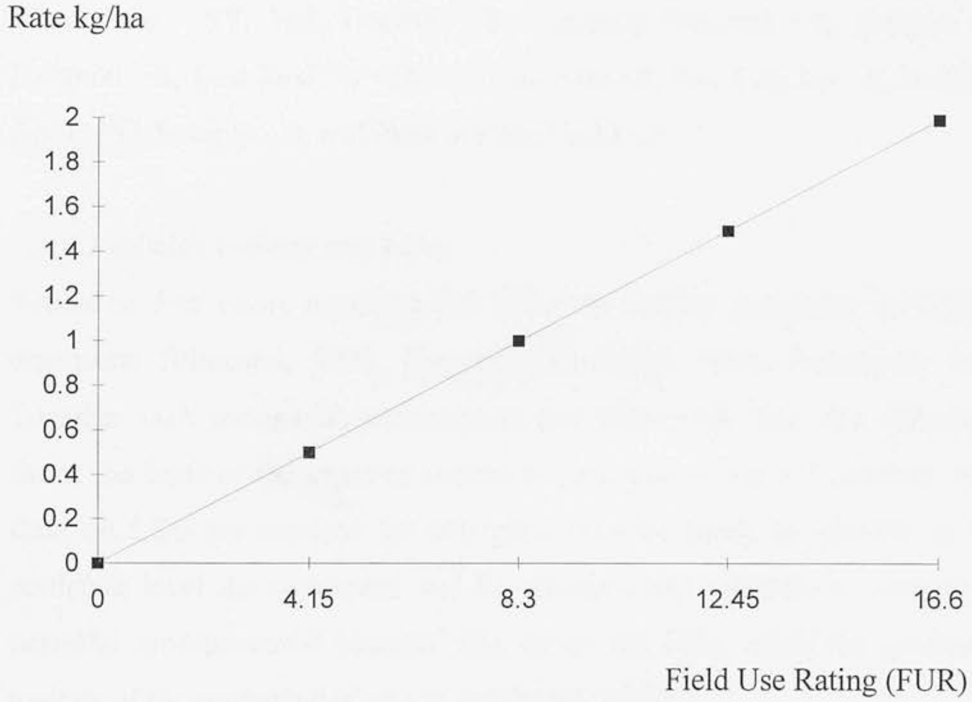
---

[40] A damage function is the expression that relates the presence of pesticides in the environment to the physical damage that occurs.

[41] Again, the EIQ Field Use Rating is simply EIQ value multiplied by the % active ingredient (a.i.) multiplied by the total amount of pesticide used.

Expressed graphically the linearity of the damage function equation becomes clearer (Figure 7.1).

Figure 7.1. The damage function for the pesticide triadmefon.



Such a functional relationship is contrary to some evidence, however, which suggests the relationship between pesticides and environmental impact, be that against target or non-target organisms, is non-linear (Dushoff et al., 1994). In order to examine what a more realistic damage function might look like, and ascertain whether or not the linear damage function described by Kovach et al., (1992) adequately describes the impact of pesticides on the environment, it is necessary to examine the damage functions for each of the categories highlighted above. Damage functions are presented in this chapter to identify the dose response curve for pesticide use for a number of different environmental categories. For the sake of clarity only a limited number of damage functions are examined here, but additional references are noted for studies displaying similar findings.

The EIQ model examines hazard to farmworkers, consumers and non-target flora and fauna. Within these three sub-categories, additional factors are examined which make up each of the farmworker, consumer and ecological component of the equation. Thus, according to Kovach et al., (1992), a value is assigned to each category of potential impact: Dermal Toxicity - DT, Chronic Toxicity - C, Systemicity - SY, Fish Toxicity - F, Leaching Potential - L, Surface Loss Potential - R, Bird Toxicity - D, Soil Half Life - S, Bee Toxicity - Z, Beneficial Arthropod Toxicity - B, and Plant Surface Half Life - P.

7.2.1. Pesticide toxicity and LD_{50}.

Extensive data exists regarding the LD_{50}[42] of various pesticides for different organisms (Pimentel, 1981, Conway and Pretty, 1991, Beaumont, 1993). Together with mutagenic, carcinogenic and teratogenic data this information forms the basis of the approval system of pesticides in the EU (Johnen, 1995). Data on LD_{50} are required for two main reasons, firstly to identify at what particular level the compound will be effective, and secondly to identify any potential environmental hazards. The lower the LD_{50} value the greater the toxicity of the compound (Conway and Pretty, 1991).

As Conway and Pretty (1991) point out, however, there are serious drawbacks in the extrapolation of laboratory test results on rodents, for example, to humans. The most important concern is that species differ greatly in their response to the same compound (Conway and Pretty, 1991) (Tables 7.2 and 7.3). There are serious problems in identifying the most appropriate multiplication factor to be used to extrapolate LD50 from laboratory animals to wild animals or humans. Conway and Pretty (1991) cite the example of the LD_{50} of the hallucinogenic drug LSD for cats and elephants to illustrate this point. Using body weight as the multiplier, an amount of LSD that will "excite" the cat will be rapidly fatal to the elephant (Conway and Pretty, 1991).

---

[42] LD50 is the amount of active ingredient of the compound required to kill (or affect in the case of Effective Dose 50 -ED_{50}) 50% of the exposed test organisms (Conway and Pretty, 1991).

Table 7.2. The variability of the LD_{50} for the dioxin 2,3,7,8-TCDD.

Species	LD_{50} (mg/kg body weight)
Guinea Pig	1
Rat (Male)	22
Rat (Female)	45
Monkey	<70
Rabbit	115
Mouse	114
Dog	>300
Bullfrog	>500
Hamster	5000

Table 7.3 Differing calculations of a suitable dose of LSD for elephants

Based on body weight of elephant and dose effective in cats	97mg
Based on metabolic rate of elephant and cats	80mg
Based on body weight of elephant and dose effective in man	8mg
Based on metabolic rate of elephant and man	3mg
Based on brain size of elephant and man	4mg

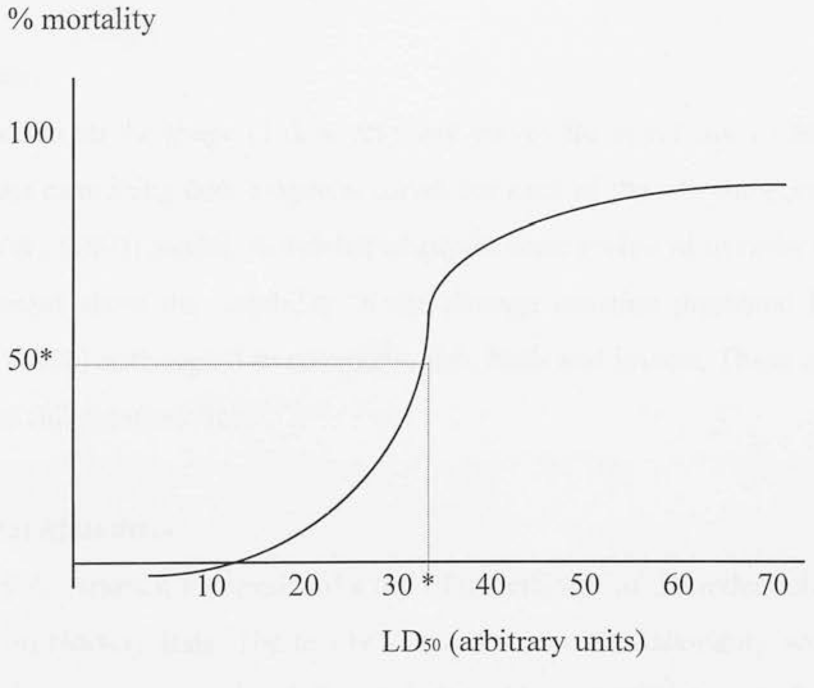
Source: Conway and Pretty (1991).

Although LD_{50} data are in abundance, less is known about the dose response curves of pesticide action on exposed organisms (McKinlay and Evans, pers comm. 1997). Conway and Pretty (1991) suggest that the dose response curve for a group of organisms exposed to a pesticide under laboratory conditions can be summarised and represented by Figure 7.2, whereas Duffus and Worth (1995) argue that a linear dose response curve adequately describes the dose response relationship (Figure 7.3). To be clearer on the form of a dose response curve, or damage functions, however, it is necessary to examine the literature on the impact of pesticides on differing organisms such as insects, birds, mammals and plants.

The group of people who face the most serious threat of exposure to pesticides are farmworkers, particularly those applying pesticides (Moses, 1989, Pearce and

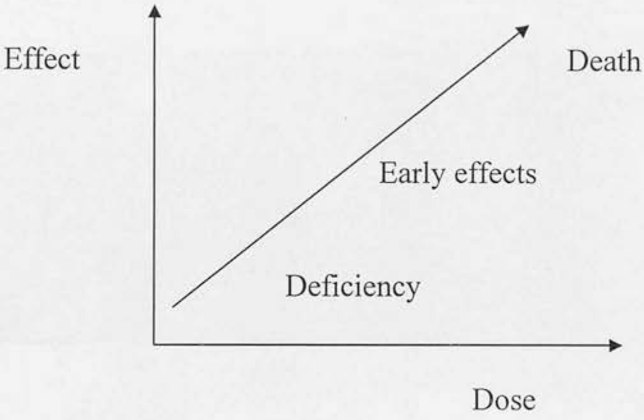
Reif, 1990, Jauhiainen et al., 1991, Lavy et al., 1992, Harper and Zilberman, 1992, Moses et al., 1993, Garry et al., 1994, Chester et al.,1993). It is widely accepted that prolonged exposure to pesticides may lead to a series of illnesses including: skin diseases, a variety of cancers, cardiopulmonary disorders, neurological disorders and haematological symptoms (Chester and Woollen, 1982, Chester et al., 1992, Chester et al., 1993, Brouwer et al., 1994, Rola and Pingali, 1994, Beaumont, 1993). What is not so clear, however, is the relationship between increased contamination, or dose, of pesticides and increased damage (in this case health problems).

Figure 7.2 The theoretical relationship between mortality of organisms and exposure to pesticides under laboratory conditions.



Information regarding the impact of increased dose of pesticides on human health must necessarily come from laboratory studies, generally on rodents, but also on birds, dogs, monkeys and so on (Pimentel, 1981, Conway and Pretty, 1991, Beaumont, 1993). Again, there are problems associated with extrapolating the results of such an analysis to humans (Conway and Pretty, 1991), yet in the absence of better data such experiments are seen to be adequate for the risk assessment of pesticides (Johnen, pers comm. 1995). The question is then asked; what does the dose response curve look like for mammals other than humans?

Figure 7.3. The relationship between dose and effect for essential compounds (Duffus and Worth, 1995).



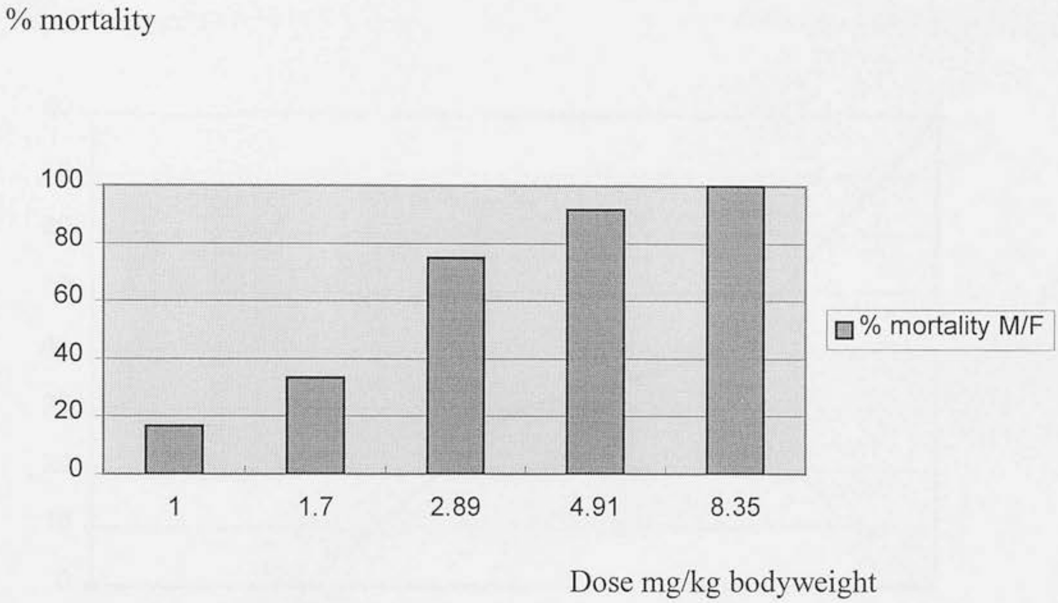
7.2.1.2 Methods.

Data was collected on the shape of dose response curves for mammals, avians, and insects, thus examining dose response curves for each of the sub-categories of the Kovach et., (1992) model. A number of papers were reviewed in order to make a judgement about the suitability of the damage function presented by Kovach et al., (1992) with regard to mammals, fish, birds and insects. These are discussed in the following section.

7.2.1.3. Results: Mammals

In 1995 Quy et al., reported the results of a test of the efficacy of the rodenticide Bromadioline on Norway Rats. The test was carried out in the laboratory with caged rats fed food contaminated with Bromadioline. Moran (1993) also studied the effects of pesticides on rats. In a laboratory study of the rodenticide Bridifacoum on two species of rats, the dose response curve was calculated and is was found to be similar to that of Quy et al., (1995), and Maund et al (1997). Tadlec (1994) found similar results in an analysis of the effect of increased doses of the rodenticide crimidine on voles. Increased doses were fed to the voles and days to death were measured. The results are presented in Figure 7.4.

Figure 7.4. The effect of increased doses of crimidine on male and female voles.



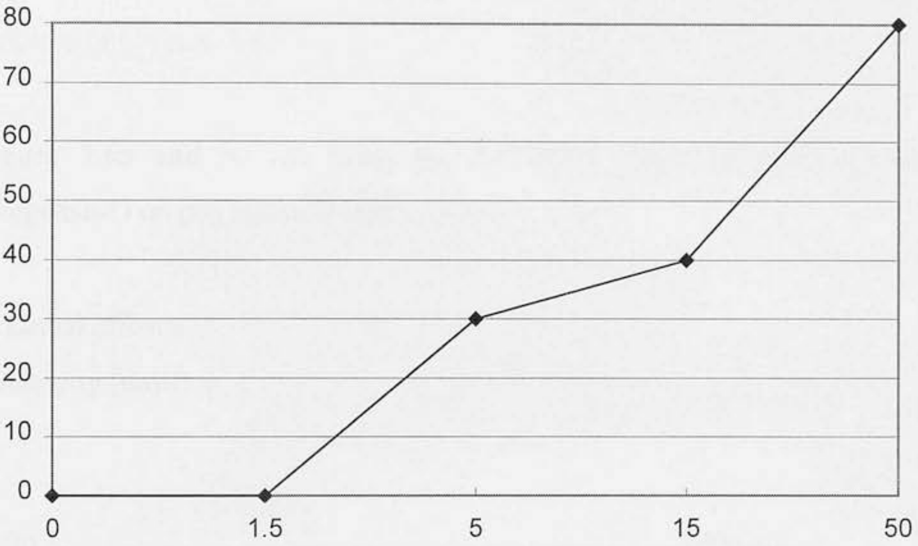
These findings were similar to those identified by Clark et al., (1996) for woodland rodent species numbers after herbicide use over a four year period, Ceron et al., (1995) in rabbits, and Clawson and Clark (1989) in bats.

7.2.1.4. Results: Birds

In 1993 Eadsforth et al., studied the dietary toxicity of the pesticide Flocoumafen to hens. In a laboratory experiment groups of hens were fed varying quantities of the compound over a five day period. The dose response curve for this particular experiment was found to be that presented in Figure 7.5. Similar findings were presented by Coenen and Brouwer (1992) for sub-lethal toxicity of pesticides in Quail, and by Brunet and Cyr (1992) for the impacts of Dimethoate on graniverous bird species.

Figure 7.5 The dose response curve for the pesticide Flocoumafen for Hens. Eadsforth et al., (1993).

% mortality



Dietary concentration of flocoumafen (mg/kg bodyweight).

7.2.1.5. Results: Insects

More data exists regarding the effects of increased doses of pesticides on insects than on any other group of animals. This information is vital in identifying the correct dose that will achieve the desired kill rate of the compound. Stark and Rangus (1994) examined the impact (lethal and sub-lethal) of increased doses of the insecticide Margosan-O on the pea aphid. Their findings are summarised and presented in Figure 7.6.

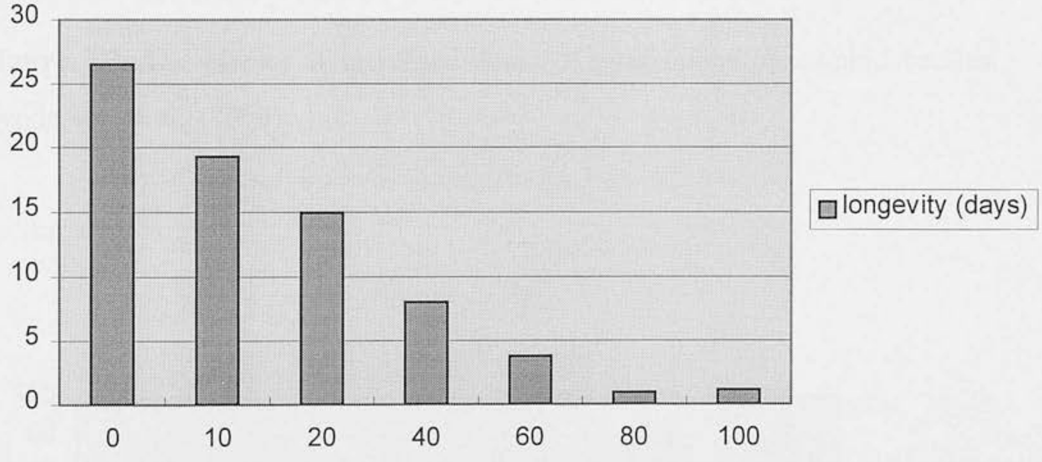
During an acute single species test under laboratory conditions Henderson et al., (1993) found similar results in an analysis of the effects of increased doses of the insecticide pyrazophos on carabid beetles. Their results are summarised and presented in Figure 7. 7.

Pesticides can be applied to either achieve lethal or sub-lethal effects (Metcalf, 1975, Pimentel, 1981). The aim of a particular treatment may be to inhibit either movement of the insect, eating patterns or fertility. Pimentel (1981) identified the dose response curves for an inhibitive pesticide for Bollworm and Medfly, with the aim of the pesticide being to sterilise male insects. The results are presented in Figure 7.8.

Figure 7.6a and b. The lethal and sub-lethal effects of increased dose of Margosan-O on pea aphid. Rangus, (1994).

a) Lethal effects.

Longevity (days)



Dose mg/kg bodyweight

b) Sub-lethal effects.

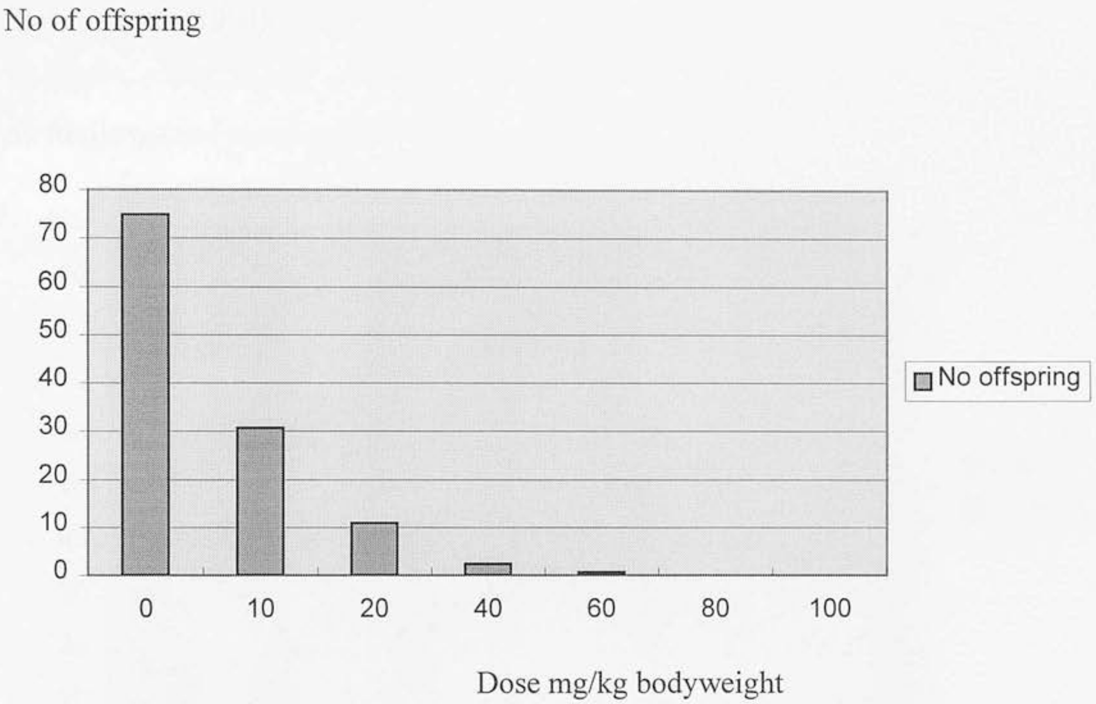


Figure 7.7. The effects of increased doses of pyrazophos on carabid beetles. Henderson et al., (1993).

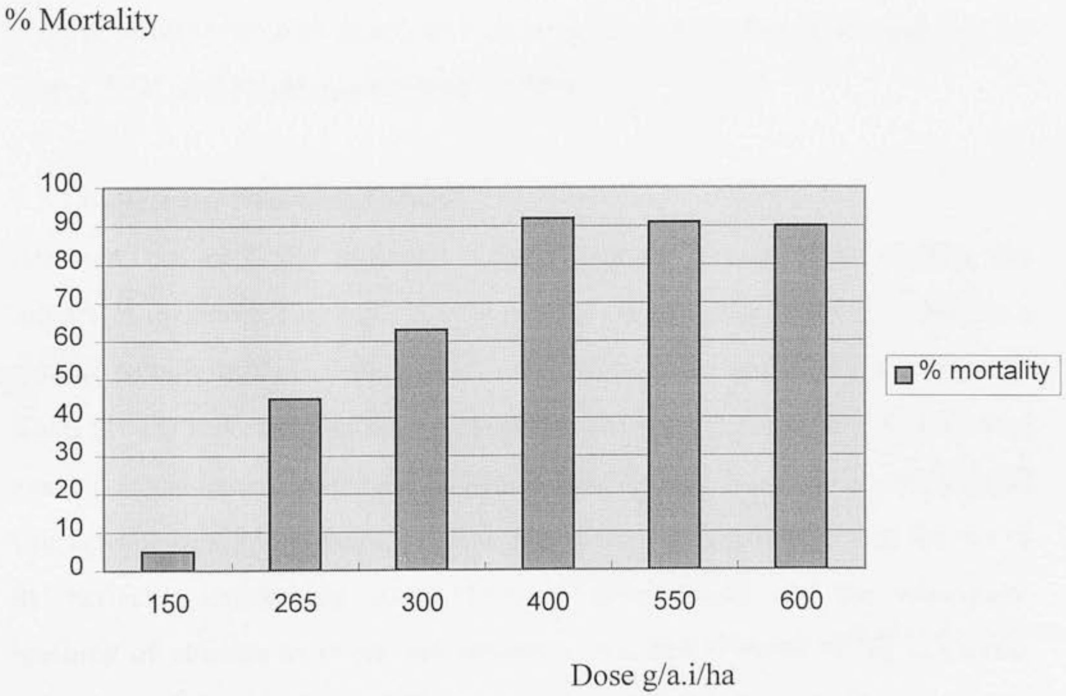
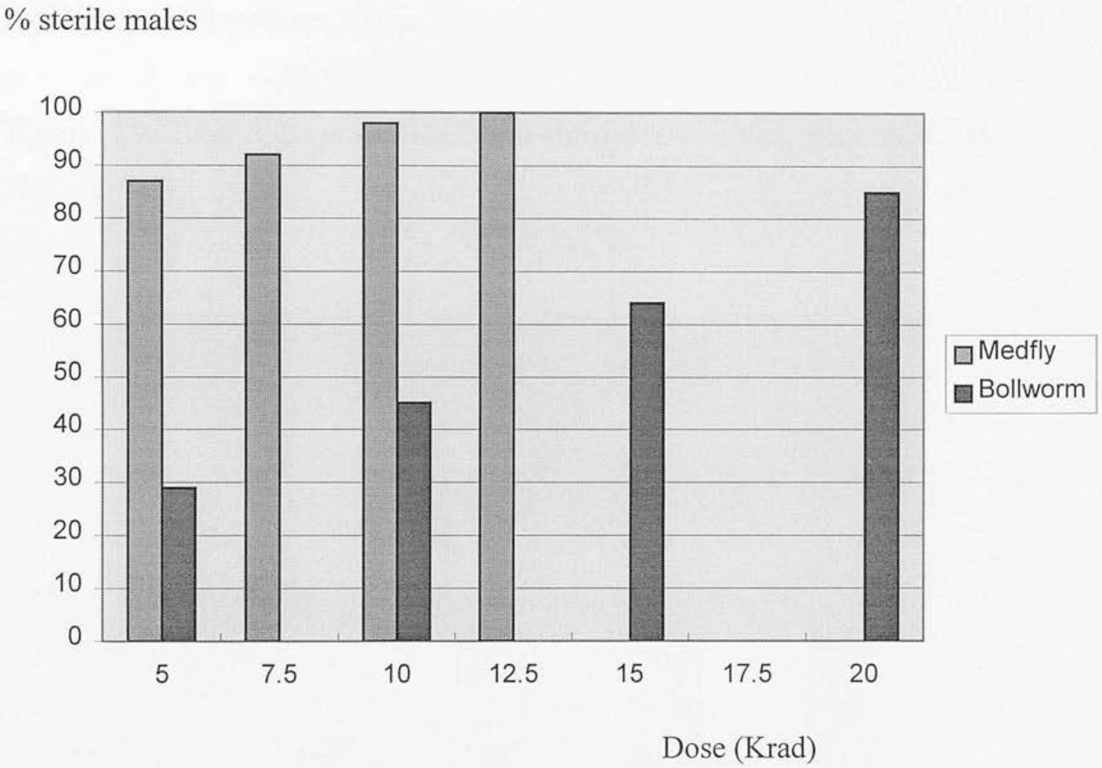


Figure 7.8. Dose response curve for sterility in the Bollworm and Medfly. Helson et al., (1994).

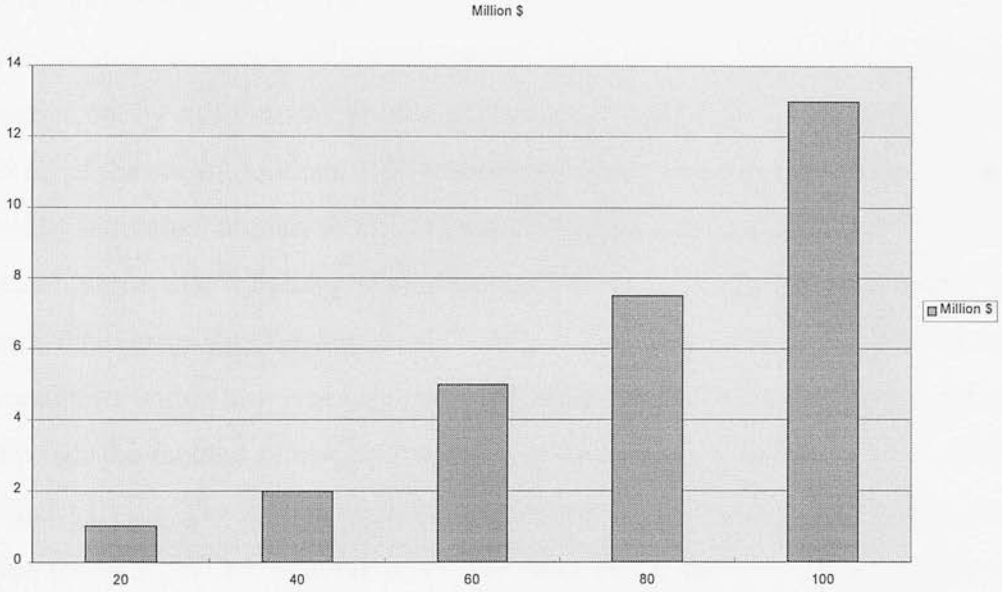


Similar results were also found by Helson et al., (1994), Akkerhuis and van der Voet (1992), and Beyer and Ktitinsky (1989).

7.2.1.6. Results: Non-target Flora

Although not explicitly included in the model of Kovach et al., (1992), the impact of increased doses of some pesticides on non-target flora may follow a similar pattern to that of insects and mammals. Kahn and Kemp (1985), and Kahn (1987) reported that losses of submerged aquatic vegetation (SAV) was causing considerable economic losses to the fishing industry in and around Chesapeake Bay, Maryland, USA. The cause of the losses in SAV was the use of the herbicide atrazine on surrounding agricultural land, and the subsequent leaching of atrazine to rivers and streams in the area. Details of the economic loss associated with SAV depletion can be obtained from Kahn and Kemp (1985) and Kahn (1987), but what is of importance here is the shape of the dose response curve that the research team derived. This is represented in Figure 7.9. It is clear from the analysis of Kahn and Kemp (1985) and Kahn (1989) that the damage function for the destruction of SAV from atrazine use in agriculture is, to all intents and purposes, linear.

Figure 7.9. Total damage function from atrazine use around Chesapeake Bay. Kahn, (1987).



% aquatic vegetation reduction as a result of Atrazine application.

7.2.1.7 Conclusion on the linear damage function of Kovach et al., (1992).

Although an examination of the dose response curves for a number of different organisms reveals that the damage functions are not necessarily linear (Lagadic and Bernard, 1993, Cresswell et al., 1994, Kjaer and Elmegaard, 1996), damage clearly, and intuitively, increases as dose increases. Figure 7.4 to 7.9 above, indicate that the model presented by Kovach et al., (1992) may adequately describe the nature of environmental impact related to increasing dose. If Occam's Razor were applied to the argument, then the representation of dose response curves as linear could be justified. Occam's[43] Razor states that "entities

---

[43] William of Occam was a Franciscan Friar and Philosopher, and together with Roger Bacon and Duns Scotus became known as one of the three great Franciscan Philosophers (William of Occam C1290-1350).

are not to be multiplied without necessity", and that "it is vain to do with more what can be done with fewer", i.e. of two available methods that achieve the same goal, choose the clearest and most simple (Russell, 1991). Accepting the limitations of modelling in general (Norton and Mumford, 1993), it is concluded that the damage function of Kovach et al., (1992) adequately describes the environmental impacts of pesticide use.

7.2.2 The weighting criteria

The relative weighting of environmental impacts from pesticide use can be carried out by either model developers, farmers, a regulatory or accrediting body, or all of the above (Levitan, 1995). Some argue that weighting should be carried out by individual farmers to reflect local conditions (Penrose et al., 1994), whilst others argue that weighting should be carried out by a regulatory authority or accreditation group (Levitan et al., 1995). There is a need to represent local conditions within any model of pesticide impact, but this can be made possible through the method of calculation and / or method of presentation of the final impact figure. The section on data presentation in this chapter further discusses this problem, but an example of the problem lies in the fact that the model of Kovach et al., (1992) does not capture the difference in impact between say an orchardist and a greenhouse horticulturist. The potential for impacts between the two production systems are likely to be very different, and yet the model does not capture this. One possible solution to this problem is in the inclusion of an expression representing pesticide application method in order to capture these differences (Reus and Pak, 1993).

Although weighting requires value judgements, as long as they are not prejudicial or illogical, and represent the opinions of both experts and stakeholders (scientists, farmers, consumers and policy makers) then a number of different weighting systems might satisfy the requirements of environmental impact models (Levitan, 1997). The Kovach et al., (1992) model has both weighted and rated variables. The weighting criteria is based on a subjective assessment of the relative importance of various environmnetal categories, such as effects on applicators, groundwater effects and so on (see Table 7.4), whereas the rating is based on the toxicity of the compound (Table 7.5). Both the weighting and rating criteria used by the EIQ equation are:

a) Low impact = 1.
b) Medium impact = 3.
c) High impact = 5.

All classes of pesticide (herbicide, fungicide, insecticide, and so on) are weighted the same in terms of farmworker, consumer and ecological impact, i.e. each component is given a weight dependant on the relative importance of that component. The rating system proposed by Kovach et al., (1992) is based on different impact criteria for different areas of environmental impact[44]. For instance, dermal toxicity (DT) is based on LD_{50}, with a figure of >2000 being weighted as 1 (little or no impact), 200-2000 weighted 3 (moderate impact), and a LD_{50} of 0-200 weighted 5 (high impact). Plant surface half-life (P) on the other hand is weighted 1-2 weeks equalling 1 (or low impact), 2-4 weeks 3 (or moderate impact, and >4 weeks 5 (high impact). Tables 7.4 and 7.5 summarise the weighting and rating criteria used for each impact category in the EIQ equation. Initially EIQ's were calculated for over 120[45] of the most commonly used pesticides.

Two of the major flaws in the model presented by Kovach et al., (1992), with regard to weighting are, firstly that the impact from individual pesticides cannot receive a weight or rating of 0 (the weighting and rating system is 1, 3 or 5), and secondly all classes of pesticides (herbicides, fungicides and insecticides) receive the same relative importance weights with regard to effect on applicators, pickers, consumers, groundwater, aquatic organisms, birds, bees and beneficials (Table 7.6). This is potentially both unrealistic and misleading.

---

[44] It has to be noted that the toxicity data used by Kovach et al., (1992) are exactly the same data used by the pesticide registration and approval process in the USA and almost identical to that used in the UK and Europe.

Table 7.4 Rating criteria of the EIQ equation (adapted from Levitan, 1997).

Variable	Symbol	Rating Scores & Criteria 1	3	5
Chronic toxicity[46]	C	little or none	possible	definite
Acute dermal toxicity (LD_{50} for rabbits/rats mg kg^{-1})	DT	>2000	200-2000	0-200
Bird toxicity (8 day LC_{50})	D	>1000 ppm	100-1000 ppm	1-100 ppm
Lethality to honey bees (at field doses)	Z	relatively non toxic	moderately toxic	highly toxic
Beneficial arthropod toxicity	B	low impact	moderate impact or post-emergent herbicides	severe impact
Fish toxicity (96 hr LC_{50})	F	>10 ppm	1-10 ppm	< 1 ppm
Soil residue half-life	S	<30 days	30-100 days	>100 days
Plant surface residue half-life	P	1-2 weeks	2-4 weeks	>4 weeks
Mode of Action (Systemicity)	SY	non-systemic and all herbicides	systemic	
Leaching potential (water half-life, solubility, adsorption coefficient, soil properties)	L	small	medium	large
Surface loss potential (water half-life, solubility, adsorption coefficient, soil properties)	R	small	medium	large

Conway and Pretty (1991) note that although on average only 0.5% of pesticides are lost to surface water, the highest losses, of up to 5%, are from herbicides. Furthermore, Conway and Pretty (1991) note that the only agricultural products found consistently in water supplies were the herbicides atrizine and simizine. This is due to the fact that many of the modern herbicides are readily soluble in water (Kahn and Kemp, 1985, Conway and Pretty, 1991, Beaumont, 1993, Eke et al., 1996). Eke et al., (1996) identify the herbicide Isoproturon as being of particular concern.

---

[45] Since the paper was published in 1992, additional pesticides have been assigned an EIQ value, with approximately 200 pesticides now included in the model (Kovach, pers. comm 1997).

[46] Long term health impacts, calculated as the average of ratings from laboratory tests on small mammals designed to assess reproductive, teratogenic (causing deformities in

Table 7.5. Weighting factors of the EIQ equation (adapted from Levitan, 1997).

	Weight	Effects	Max. Score	Weighted Variables	Symbol	Rating
Farmworker Component	5	On Applicators	125	Chronic toxicity	C	1,3, 5
				Acute dermal toxicity (LD$_{50}$ for rabbits/rats)	DT	1,3, 5
	1	On Pickers	25	Acute dermal toxicity (LD$_{50}$ for rabbits/rats)	DT	1,3, 5
				Plant surface half-life	P	1,3, 5
Consumer Component	1	Consumers (food residues)	75	Chronic toxicity	C	1,3, 5
				Soil half-life*	S	1,3, 5
				Plant surface half-life*	P	1,3, 5
				Systemicity (ability to be absorbed by plants)	SY	1,3, 5
	1	On Ground-water	5	Leaching potential (water half-life, solubility, adsorption coefficient, soil properties)	L	1,3, 5
Ecological Component	1	On Aquatic Organisms	25	Fish toxicity (96 hr LC$_{50}$)	F	1,3, 5
				Surface loss potential (water half-life, solubility, adsorption coefficient, soil properties)	R	1,3, 5
	3	On Birds	75	Bird toxicity (8 day LC$_{50}$)	D	1,3, 5
				Soil half-life*	S	1,3, 5
				Plant surface half-life*	P	1,3, 5
	3	On Bees	75	Bee toxicity	Z	1,3, 5
				Plant surface half-life	P	1,3, 5
	5	On Beneficials	125	Beneficial arthropod toxicity	B	1,3, 5
				Plant surface half-life	P	1,3, 5
Total No. Weights Assigned	20		530	**Maximum Total Score (before divided by 3) Max. EIQ = 530 ÷ 3 = 176.7**		

---

offspring), mutagenic (affecting genes and chromosomes), and oncogenic (tumour

This is the most commonly applied pesticide in current use in the UK, with the high tonnage applied, and Autumn application resulting in water contamination in many cereal growing areas (Eke et al., 1996).

On the other hand, insecticides are less soluble, and therefore, less likely to leach to water (Eke et al., 1996). This is due to high adsorption rates of many insecticides, and greater microbial activity, light availability and temperatures in soil resulting in lower half life figures. Herbicides in water have a greater half life than pesticides found in soil due to lower temperatures, and reduced light (Conway and Pretty, 1991). This is not to suggest that insecticides are never found in water sources, however, and do not pose a threat of water supply contamination.

In addition to this herbicides and fungicides may have less of an impact on agricultural workers due to their lower toxicity levels to mammals. This would again bring in to question the justification for ranking all pesticides as a 5 (high impact) for the farmworker component of the EIQ equation. Conway and Pretty (1991) note that "most modern fungicides and herbicides are either non-toxic, or only slightly toxic, to humans". Whereas some caution will always be required when handling pesticides, to class all pesticides as high impact is misleading (Taite, 1997, pers. comm). Thus it is argued that the weighting methodology presented in the Kovach et al., (1992) model be changed to reflect the differing impacts of differing compounds (Table 7.6). Rating hazard to farmworkers as 5 for insecticides is justifiable given the toxicity data relating to insecticides (Moses, 1989, Pearce and Reif, 1990, Conway and Pretty, 1991, Lavy et al., 1992, Chester, 1993, Forastier, 1993, Brouwer et al., 1993).

The WHO Toxicological Register (1992-1993), encompassing human health impact data, confirms the above conclusions. Of 61 pesticides classed 1a (Extremely Hazardous) and of 63 pesticides classed as 1b (Highly hazardous), only 6 are herbicides, and 8 fungicides. The majority of pesticides in either Class

---

growth) effects.

1a or 1b (60%) are insecticides, with the remainder being rodenticides, miticides, acaricides, fumigants, fungicides and herbicides (WHO, 1992-1993). It is therefore recommended that the weighting system be changed to that illustrated in Table 7.6.

Table 7.6. Alternate weighting system for insecticides, fungicides and herbicides[47].

Insecticides			**Herbicides and Fungicides**		
Component	**Weight**	**Effects**	**Component**	**Weight**	**Effects**
Farmworker	5	Applicators	**Farmworker**	1	Applicators
	1	Pickers		1	Pickers
Consumer	1	Consumers (food residues)	**Consumer**	1	Consumers (food residues)
	1	Groundwater		5	Groundwater
Ecological	1	Aquatic Organisms	**Ecological**	5	Aquatic Organisms
	3	Birds		1	Birds
	3	Bees		1	Bees
	5	Beneficials		3	Beneficials
Total No. Weights Assigned	**20**		**Total No. Weights Assigned**	**18**	

In addition to the changes shown in Table 7.6, it is necessary to have the ability to rank some pesticides as 0, rather than 1, 3 or 5 (Dushoff et al., 1994, Levitan et al., 1995). This, it is argued, is justifiable where a particular component, say aquatic organisms, is weighted 1, and the impact, say LD50, is also rated 1. In the existing EIQ model what this describes is an impact category, aquatic organisms, that is not deemed to be under particular threat, and an impact level, LD50, which is also deemed to pose little or no risk (Kovach et al., 1992). Thus in order to allow for a zero value to be entered in the modified EIQ model and entered into the decision support spreadsheet (Table 7.16), 1 multiplied by 1 will equal 0. Where a weight is 1 and a rating 3 or 5, then this may remain unchanged. Also where a weight is 5 or 3 and the rating 1 then this will also

---

[47] Note that the rating for Bees is 3, whilst others in the ecological component are 1 (with the exception of aquatic). This is because in almost all cases the existing rating for bees is 3. It would be unwise to lower the weight for this criteria based on this evidence.

remain unchanged. Clearly any changes made to the weighting criteria and the inclusion of a 0 value will be analysed. Such a comparison made between existing EIQ values and modified EIQ values will be discussed later in this Chapter (Tables 7.14 and 7.15).

7.2.3. Averaging impacts

The final EIQ figure presented by Kovach et al., is calculated by dividing the sum of the farmworker, consumer and environmental categories of the equation by three to give the average EIQ of all categories. This may be misleading to decision makers, at the farm or policy level. It is essential not only to know which pesticides are more or less harmful, but also to know in what areas they are harmful. For instance, herbicides are more readily soluble in water than are insecticides (Kahn and Kemp, 1985) and so may pose a greater threat to groundwater contamination (Bergman and Pugh, 1994). It is vital then that farmers have access to this information, and yet the operation of averaging effects in fact masks impacts in any one of the three categories; farmworkers, consumers and environment. For example, if impact on groundwater contamination (the consumer category in the EIQ equation) is 10, but the impact on human health is 2 and impact on the environment is 1, then the average EIQ value will be 4.33. This grossly understates the potential impact of that particular pesticide on groundwater contamination, and may prove misleading to decision makers.

Other published models, that purport to rank pesticides in order of impact, do not average effects, leaving each category of impact explicitly observable. This it is argued provides decision makers with more useable information (O'Bryan and Ross, 1988, Dushoff et al., 1994). Examples of this can be seen in the models presented by O'Bryan and Ross (1988) (Table 7.7), Reus and Pak (1993) (Table 7.8) and Wijnands and van Dongen, (1995) (Table 7.9).

Table 7.7. The impact categories of the chemical scoring system for hazard identification and exposure (O'Bryan and Ross, 1988).

Chemical	Onco	Geno	Dev tox	Mam tox	Aquatic	Biocon	Env Fate	Vol	Occ Exp	Cons Exp	Env Ex
Acetaldehyde	9	8	9	4	4	1	8	8	9	10	
Benzene	9	4	9	6	4	6	16	10	9	8	
Benzidine	9	8	6	6	6		25	1	1	1	
Bisphenol A	0	0	6	6	4	6	19	10	5	7	
Chloromethylpropene	8	6	3	3	5	0	14	3	7	4	7
Dimethoxyethyl	5	8	7	2	4	3	14	3	5	6	4

Table 7.8. The impact categories and scores of the environmental yardstick (Reus and Pak, 1993).

		Pollution points at recommended dose rates		
Pesticide	**Dose (kg/a.i/ha)**	**Water organisms**	**Soil organisms**	**Groundwater**
maneb	1.5	2500	1500	15
maneb	1.5	2500	1500	15
mancozeb	2.5	18	0	24
mancozeb	2.5	18	0	24
mancozeb	2.5	18	0	24
maneb/fentin	1.1	2800	820	8
maneb/fentin	1.1	2800	820	8
metribuzin	0.5	980	60	260
pirimicarb	0.3	110	230	0
propoxur	0.5	182	275	10000
Total	**14**	**11926**	**5205**	**10378**

Table 7.9. Impact categories for the Environmental Exposure to Pesticides (EEP) model (Wijnands and van Dongen 1995).

	EEPair		EEPair	EEP water		EEPwater	EEPsoil		EEPsoil	A.I.		(kg/ha)
	1988		1992	1988		1992	1988		1992	1988		1992
	C	I	I	C	I	I	C	I	I	C	I	I
ware potato	11.7	3.4	0.9	156	0.9	0.5	3493	376	82	196.6	9.7	2
seed potato	11.7	3.7	3	152	0.9	0.3	3281	185	66	187.5	6.4	1.4
winter wheat	2.9	2.5	2.4	141	1.5	0.4	162	25	10	5	2.6	0.4
sugar beet	1.7	2.3	2.9	1.1	0.1	0.4	103	15	55	3	0.4	1.6
sown onion	4.3	3.8	2.6	7.4	3.8	0.7	722	401	47	23.4	9.8	2.1
averages	5.3	2.9	2.4	86	1.3	0.4	1170	129	46	60.5	4.6	1.4

The method of data presentation in Tables 7.7 to 7.9 displays more information than the model presented by Kovach et al., (1992). Thus it is proposed that the data presentation format be changed from Table 7.10 (the current EIQ data presentation format) to 7.11.

Table 7.10. Current data presentation format of the Kovach et al., (1992) EIQ model[48].

Compound	kg/ha	% a.i	kg/a.i/ha	EIQ	EIQ FUR
Dodine.	10	0.5	5	34.9	174.5
Rubigan	48	0.8	38.4	50.7	1946.88
Dithane	1.4	0.05	0.07	44	3.1

Table 7.11. The suggested method of data presentation[49].

Compound	kg/ha	% a.i	kg/a.i/ha	EIQ	EIQ FUR
Dodine					
Farmworker	0.5	0.45	0.225	20.3	4.6
Consumer	0.5	0.45	0.225	16.4	3.7
Ecological	0.5	0.45	0.225	67.9	15.3
Rubigan					
Farmworker	0.5	0.12	0.06	12	0.7
Consumer	0.5	0.12	0.06	23	1.4
Ecological	0.5	0.12	0.06	47	2.8
Dithane					
Farmworker	12.5	0.8	10	40	400
Consumer	12.5	0.8	10	23	230
Ecological	12.5	0.8	10	68.9	689

7.2.4 Site specific data requirement

One of the main criticisms of the Kovach et al., (1992) model has been that it fails to capture specific local conditions of soil type, proximity to water courses, and application method (Dushoff et al., 1994, Quin and Edwards-Jones, 1997). One of the strengths of alternative models has been that explicit consideration has been given to local conditions, such as natural resource conditions (O'Bryan and Ross, 1988), and application method (Penrose et al., 1994). This section discusses ways in which such factors can be added to the EIQ equation, and displayed in a manner that increases the applicability of the EIQ model to site

---

[48] Note that the final EIQ value is the average of the farmworker, consumer and ecological EIQ values.

[49] Note that impacts are no longer averaged, but expressed separately in order to flag potential areas of concern more clearly.

specific conditions, whilst not compromising the user friendly nature of the model.

7.2.4.1 Soil

Soil type is likely to have a major influence on the potential hazard to environment caused by pesticide use (Arnold and Briggs, 1990). Here there are likely to be several critical factors influencing the fate of pesticides. They are; organic content, water availability, texture, pH, microbial community, and water flow (Arnold and Briggs, 1990, Gaillardon, 1994, Brouard et al., 1994). Soil organic content is particularly important for the adsorption of pesticides (Gaillardon, 1994). Relief, that is the angle of the slope upon which a given farm is situated, may also have a bearing on the environmental fate of pesticide use. Ultimately, the fate of pesticides is determined by a combination of pesticide properties, soil characteristics and climate (Arnold and Briggs, 1990). The Kovach et al., (1992) model must somehow reflect local conditions, one of which will be soil, in order to capture more realistically the impacts associated with pesticides.

Changes in the EIQ equation to incorporate soil characteristics may not be the most appropriate method for incorporating site specific variables into the model. One of the main features of the EIQ model is its simplicity (Dushoff et al., 1994, Levitan et al., 1995, Quin and Edwards-Jones, 1997), and to add additional variables would also add to the complexity of the model. The most appropriate mechanism for including site specific characteristics could then fall outside the EIQ equation, and could be expressed as part of a decision support spreadsheet (Table 7.16).

7.2.4.2 Proximity to Water source

Proximity to water sources will also have a significant effect on the environmental impact of pesticide use (Conway and Pretty, 1991). Orchards close to water courses that do not have buffer[50] strips are more likely to be at greater risk from run-off, leaching, spray drift and spillage than those that have buffer strips in place (Jones, 1990, Cohen, 1990). Additionally the weather may play a critical role in determining the environmental fate of pesticides. Pesticides applied just before heavy rain are much more likely to leach or run-off to water sources (Kahn and Kemp, 1995, Mathiessen et al., 1994, Eke et al., 1996). Although providing buffer strips is potentially expensive for farmers, EU policy is currently one of grubbing up of apple orchards (Prognosfruit, 1995). Grubbing up refers to the removal of orchards, or parts of orchards from production by pulling up the trees to solve the problem of over-supply. Compensation is paid by the EU to farmers for this operation (Prognosfruit, 1995). This policy could prove to be environmentally sound if orchardists directed grubbing up activities to those areas close to water sources, perhaps even creating buffer strips by rivers and streams. Irrigation in apple orchards is common place in Italy, Greece, Spain, Portugal and southern France (O'Rourke, 1994). This is likely to add to the threat of water source contamination.

The use of buffer strips to control runoff from pesticides, nitrates and soluble phosphorous is well documented (Wauchope, 1978, Dillaha, 1985, Beyers et al., 1994, Patty, et al., 1997). Patty et al., (1997) report that studies in France show that herbicide runoff can be reduced by between 75 and 89% using an 11 metre wide vegetative strip. Patty et al., (1997) go on to report that Lindane losses were reduced by 76, 99.8 and 100% on buffer strips measuring 6, 12 and 18 metres wide respectively. As with soil and application method a simple ranking score could be attached to each buffer strip. For example the absence of a buffer strip could be rated 5, a buffer strip of up to 5 metres rated 3, 5 to 10 metres rated 1, and > 10 metres rated as 0. Again the most appropriate mechanism for accounting for water source contamination, thus being site specific, may be to include a water source proximity variable outside the EIQ equation (Table 7.16, 7.17).

---

[50] Buffer strips are simply vegetative strips, or rough land, that separate cultivated areas from water courses.

There may be a justification for changing the scoring system where irrigated spray methods are employed. The majority of orchards in Northern Italy, employ irrigation to counter the long dry summers. It is common practice to mix in pesticides with the water so that they are applied to the whole orchard in a spray mist (Pers. Obs. site visit to Italian apple orchards, 1996). Where orchards are close to water sources, then spray drift may become a problem. If this is the case then the rating for the water criteria could be 5 except where a buffer strip of more than 10 metres is in place. In this case, potentially all pesticides would be given a high EIQ value, indicating that the farmer would need to examine not only pesticide type used but also production techniques, i.e. implementing buffer strips.

7.2.4.3 Application Method

Reus and Pak (1993), Pease et al., (1996), Penrose et al., (1994) and Wijnands and van Dongen (1995) have all stressed the importance of including application method in any analysis of the environmental impacts of pesticide use. Reus and Pak (1994) have categorised emission potential for the most commonly practised application method in Dutch agriculture (Table 7.12).

Table 7.12 Application method and emission percentage (Reus and Pak, 1993).

Application method	Emission percentage
Seed treatment	0
Crop row spraying	0.5
Full field spraying (arable)	1
Full field spraying (fruit)	10
Aerial spraying	100

Ratings, similar to those of the EIQ equation, could be assigned to each application method to indicate potential hazard to the environment. For instance, according to Reus and Pak (1993), for impact on groundwater crop row spraying could receive a rating of 1, whereas aerial spraying could receive a rating of 5 (Table 7.12). Moses (1993) notes that application method will be a critical factor in the occupational exposure of farmworkers to pesticides. Once again, this data would be handled outside the EIQ equation in a decision support spreadsheet (Table 7.16).

7.2.5. Data presentation.

Currently the data presentation format for the EIQ model are as indicated in Table 7.10. This format for data presentation may not represent the most useful vehicle for providing decision makers, at the farm and policy level, with the environmental information that they require. This is particularly the case when one gives additional considerations to site specific variables such as soil type, proximity to water source and application method. When this is combined with changes in the EIQ values, discussed below, then the EIQ model becomes a much more powerful decision support tool. Table 7.11 represents a new method of data presentation, which is then adapted to include modified EIQ values and site specific parameters (Table 7.16). This method of data presentation has been applied to all pesticides for all regions studied.

7.2.5.1 Modified EIQ values for fungicides and herbicides: a comparison of existing and modified EIQ values.

This section reports on the results of the modification of the environmnetal impact figures for fungicides and herbicides. Herbicides and fungicides have been re-calculated based on changes to the weighting and rating criteria discussed in the previous section. These modified EIQ figures have then been compared to those of Kovach et al., (1992) to illustrate how the EIQ values have changed (Table 7.13 and Table 7.14). Finally, the modified EIQ figures are presented in a decision support spreadsheet, the rationale being that this approach describes more accurately the nature of environmental impacts in terms of farmworker, consumer and ecological impact, and captures site specific conditions (Table 7.13).

Table 7.13 A comparison of old and modified EIQ values for herbicides, for farmworkers (FW), Consumers (Con) and the Ecological (Eco) components..

Common Name	Trade Name	OLD EIQ FW	NEW EIQ FW	OLD EIQ Con	NEW EIQ Con	OLD EIQ Eco	NEW EIQ Eco	OLD EIQ	NEW EIQ
2,4-D (acid)	Weedone	72	36	9	21	88	74	56.3	43.6
acifluorfen	Blazer	72	36	12	24	72	72	52	44
alachlor	Lasso	18	6	6	16.5	40	70.5	21.3	31
ammonium sulfamate	Ammate	24	12	8	28	83	71	38.3	37
atrazine	Atrazine	12	6	9.5	29.5	78	123	33.2	52.8
bentazon	Basagran 4S	24	12	11	31	81	85	38.7	42.7
bromacil	Hyvar	12	6	11	31	54	52	25.7	29.7
chloramben	Amiben	15	5	5.5	20	26.6	35	15.7	20
cyanazine	Bladex	26	8.7	7.3	19.3	26	32	19.8	20
cycloate	Ro-Neet	6	0	5	17	35	64	15.3	27
dalapon	Dalapon	36	18	8	28	68.5	62.5	37.5	36.3
DCPA	Dacthal	16	8	9	13	77	83	34	34.7
dichlobenil	Casoron	18	6	7	27	29	34	18	22.3
diethatyl-ethyl	Antor	6	0	3	7	35	64	14.7	23.7
diuron	Karmex	15	5	10.5	22.5	36	87	20.5	38.2
EPTC	Eptam	6	0	5	17	29	34	13.3	17
ethalfluralin	Sonolan	30	10	11	15	51	144	30.7	56.3
fluazifop-butyl	Fusilade	40	20	11	15	81	131	44	55.3
glyphosate	Roundup	16	8	7	11	74.3	122.3	32.4	47.1
linuron	Lorox	16	8	9	21	96	108	40.3	45.7
MCPA	Bronate	32	16	9	13	69	71	36.7	33.3
metolachlor	Dual	12	4	7	19	35	64	18	29
metribuzin	Sencor	8	3	8	28	90	78	35.3	36.3
napropamide	Devrinol	12.8	4.26	9.3	29.26	32	35	18	22.84
nicosulfuron	Accent	12	6	5	28	69.6	74	29.9	36
norflurazon	Solicam	9	3	9.5	29.5	38	65	18.8	32.5
oryzalin	Surflan	12	4	3	5	38	63.5	17.7	24.2
oxyfluorfen	Goal	20	10	8.5	12.5	112	194	46.8	72.2
paraquat	Gramaxone	72	36	13	17	125	155	70	69.3
pendimethalin	Prowl	15	5	8.5	12.5	54	145	25.8	54.2
phenmediphan	Spin-aid	12	6	5.5	9.5	74.6	100.1	30.2	38.5
pronamide	Kerb	24	18	10	14	74	82	36	38
propazine	Milogard	24	12	17	37	75	73	38.7	40.1
pryazon	Pyramin	6	0	7	27	35	38	16	21.7
sethoxydim	Poast	8	3	4.9	16.5	69.6	73.9	27.5	31.1
simizine	Princep	12	4	9	29	26.2	31.2	15.7	21.4
terbacil	Sinbar	12	4	11	31	27.5	30.5	16.8	21.8
trifluralin	Treflan	15	6	8.5	29.5	57	33.5	31.2	23

Table 7.14 A comparison of old and modified EIQ values for fungicides and nematicides.

Common Name	Trade Name	OLD EIQ FW	NEW EIQ FW	OLD EIQ Con	NEW EIQ Con	OLD EIQ Eco	NEW EIQ Eco	OLD EIQ	NEW EIQ
anilizaine	Dyrene	16.2	8.2	5.1	9.1	58.7	68.4	26.68	28.6
benomyl	Benlate	30	18	50	70	128.5	208.5	69.5	98.8
captan	Orthocide	28	14	8	12	49.9	59.9	28.62	28.6
carboxin	Vitavax	9	3	5.5	7.25	45.4	89.9	19.95	33.4
chlorothalonil	Bravo	25	15	11	15	102	184	46	71.3
copper hydroxide	Kocide	12.2	6.15	5.1	9.05	82.7	109.5	33.3	41.6
copper sulfate	copper	81	27	14.5	18.5	47.9	138.9	47.78	61.5
copper sulfate+lime	Bordeaux	108	54	19	23	76	162	67.67	79.7
dichloran	Botran	24.3	12.3	7.2	11.15	76.4	108.4	35.95	43.95
dinocap	Karathane	22	7.3	12	10.5	36.9	97.1	23.63	38.3
dodine	Syllit	20.3	10.25	16.4	20.4	67.9	115.6	34.85	48.75
fenarimol	Rubigan	12	4	23	43	47	138	27.33	61.7
fentin hydroxide	Du-Ter	24	12	5	9	69	137	32.67	52.7
ferbam	Carbamate	8	3	5	9	73.5	75.9	28.83	29.3
flusilazol	Nustar	8	3	9	12.95	81.8	131.2	32.9	49.05
folpet	Phaltan	8.1	3.1	5.7	12.05	52.9	87.7	22.21	34.3
fosetyl-Al	Aliette	12	4	7	8	22	19.5	13.67	10.5
iprodione	Rovral	8.1	3.1	3.1	7.05	68.7	118.4	26.63	42.85
mancozeb	Manzate	40	24	17	21	130	233	62.33	92.7
maneb	maneb	40	24	17	21	135.3	217.3	64.08	87.4
maneb +dinocap	Dikar	32.4	16.4	13.2	17.2	93.9	161.6	46.51	65.1
metalaxyl	Ridomil	8	3	11	31	68.5	62.5	29.17	32.2
metiram	Polyram	50	30	16	20	101.8	93.8	55.92	47.9
myclobutanil	Nova	36.5	18.45	13.8	20.15	73.4	121.4	41.21	53.3
PCNB	Terraclor	15	5	8.5	12.5	42	93	21.84	36.8
streptomycin	Agristrep	18	6	4.6	9.5	33.5	89.15	18.71	34.9
sulfur	Sulfur	10	5	6	10	120	117	45.53	44
thiophanate methyl	Topsin-M	30	18	28	32	96.5	116.5	51.5	55.5
thiram	Thiram	72.9	36.9	7.2	11.15	83.5	125	54.52	57.7
triadmefon	Bayleton	28	14	10	22	62	86	33.32	40.7
triforine	Funginex	24.3	12.3	25.9	32.3	73.4	121.4	41.21	55.3
vinclozolin	Ronilan	24.3	12.3	7.2	13.1	56.7	114.4	29.38	46.6
zineb	Dithane Z	40	20	23	35	68.9	105.1	43.95	53.4

The new figures of environmental impact were then inputted into a decision support spreadsheet, although complete tables will also be available to the decision maker when making the initial choice of pesticide. The tables presented by Kovach et al., (1992) are alphabetically ordered (Appendix 3), rather than ordered in terms of impact. Whilst this is a minor point, the tables have been altered to be ordered in terms of impact, and separated into classes of impact.

This no longer necessitates the decision maker needing to "order or rank" the pesticides themselves (Becker et al., 1989) (Tables 7.15, 7.16).

Pesticides can then be split into "impact" categories. They are: Category 1, Danger, Category 2, Warning and Category 3, Caution (descriptors from the US EPA pesticide hazard classification for humans). Note also that the average EIQ value of total impact is now no longer presented in Table 7.16, and Appendix 3. The critical values of impact are now those for each of the farmworker, consumer and ecological component of the equation.

From Tables 7.15 it is clear that benomyl and fenarimol should be avoided due to their potentially harmful effect on groundwater (EIQ consumer component), and in terms of impacts on farmworkers copper sulphate, thiram, metiram, mancozeb, maneb and zineb should be avoided. Both of these groups of pesticides have been placed in Category 1 and carry the descriptor: DANGER (Table 7.16).

7.2.5.2 An alternative data presentation format for the Kovach et al., (1992) model.

As has been mentioned above the main strengths of the Kovach et al., (1992) model are its simplicity, ease of use and ease of comprehension. It is therefore argued that any additional variable to be included in an analysis of environmental impacts from pesticide use must fall outside the EIQ equation. Thus, a decision support spreadsheet is proposed as an aid to decision making at either a farm, regional or national policy level. The spreadsheet will include the modified EIQ data, as well a site specific variable for water proximity, soil type and application method. It will be used in association with the tables providing information on pesticides, in order of hazard, by farmworker, consumer and ecological components of the EIQ model (Appendix 3).

Table 7.15 An example of arbitrary impact categories and EIQ assigned to the farmworker (FW) and consumer (Cons) categories for fungicides.

Common Name	Trade Name	EIQ FW
CATEGORY 1:	**DANGER**	
copper sulfate+lime	Bordeaux	54
CATEGORY 2:	**WARNING**	
thiram	Thiram	36.9
metiram	Polyram	30
copper sulfate	copper	27
mancozeb	Manzate	24
maneb	maneb	24
zineb	Dithane Z	20
myclobutanil	Nova	18.45
benomyl	Benlate	18
thiophanate methyl	Topsin-M	18
maneb +dinocap	Dikar	16.4
chlorothalonil	Bravo	15
captan	Orthocide	14
triadmefon	Bayleton	14
dichloran	Botran	12.3
triforine	Funginex	12.3
vinclozolin	Ronilan	12.3
fentin hydroxide	Du-Ter	12
dodine	Syllit	10.25
CATEGORY 3:	**CAUTION**	
anilizaine	Dyrene	8.2
dinocap	Karathane	7.3
copper hydroxide	Kocide	6.15
streptomycin	Agristrep	6
PCNB	Terraclor	5
sulfur	Sulfur	5
fenarimol	Rubigan	4
fosetyl-Al	Aliette	4
folpet	Phaltan	3.1
iprodione	Rovral	3.1
carboxin	Vitavax	3
ferbam	Carbamate	3
flusilazol	Nustar	3
metalaxyl	Ridomil	3

Common Name	Trade Name	EIQ Cons
CATEGORY 1:	**DANGER**	
benomyl	Benlate	70
fenarimol	Rugigan	43
zineb	Ditane Z	35
triforine	Funginex	32.3
thiophanate methyl	Topsin M	32
metalaxyl	Ridomil	31
CATEGORY 2:	**WARNING**	
cooper sulphate+lime	Bordeaux	23
triadmefon	Bayleton	22
mancozeb	Manzate	21
maneb	Maneb	21
dodine	Syllit	20.4
myclobutanil	Nova	20.15
metiram	Polyram	20
copper sulphate	Copper	18.5
maneb+dinocap	Dikar	17.2
clorothalonil	Bravo	15
vinclozolin	Ronilan	13.1
flusilazol	Nustar	12.95
PCNB	Terraclor	12.5
folpet	Phaltan	12.05
captan	Orthocide	12
dichloran	Botran	11.15
thiram	Thiram	11.15
dinocap	Karathane	10.5
sulfur	Sulfur	10
CATEGORY 3:	**CAUTION**	
streptomycin	Agristrep	9.5
anilizaine	Dyrene	9.1
copper hydroxide	Kocide	9.05
fentin hydroxide	Du Ter	9
ferbam	Carbamate	9
fosetyl-Al	Aliette	8
carboxin	Vitavax	7.25
iprodione	Rovral	7.05

The decision support spreadsheet will be particularly easy to use as the only information that the decision maker will need to input is pesticide type, total quantity applied, % active ingredient and price. All other variables, such as soil type and proximity to water source, will be pre-programmed and orchard specific. If the farmer routinely uses more than one application method, then this data too will need to be inputted, otherwise this can also be pre-programmed. This operation will require no more additional work than the original EIQ model. The spreadsheet can be pre-programmed by extension workers, Government officials, organisations such as the Environment Protection Agency (EPA), IPM accreditation groups, or by the farmer if preferred. The spreadsheet has been designed in Microsoft Excel 5.0, and thus can be used on a standard personal computer.

7.2.5.3. Data presentation

The new data presentation format is illustrated in Table 7.16. The decision maker inputs pesticide type, quantity, % active ingredient and pesticide purchase price. The model will then identify:

a) whether the compound is a fungicide, herbicide or insecticide.
b) what the EIQ value is for each impact category; farmworker, consumer and ecological.
c) what impact category the pesticide falls into; Category 1, 2 or 3.
d) what impact descriptor should be assigned to each pesticide; DANGER, WARNING or CAUTION, and displays this in the model.
e) what the site specific indicators are; HAZARD or OK.

In addition to this, a use rate criteria is also be included in the model. A pesticide may have the lower impact category: CAUTION, but if use rates exceed a given amount, say 10 kg/ha, then the model flags this quantity as: REDUCE. This will be enough to prompt the farmer to examine the use rates of this pesticide to ensure that the current use rate falls within recommended dose rates. The Field Use Rating (FUR) is the measure used to determine if a pest management strategy can be catagorised as being IPM produced or not. Thus the total FUR is flagged as either "IPM" or "NOT IPM", based on an acceptable threshold of pesticide use. At present an arbitrary IPM threshold figure of an EIQ of 7000 has been allocated, although there is no evidence to suggest that this is entirely justifiable. Further research is required to identify what threshold level might constitute IPM production.

Soil, applicator and water impact are assigned separate impact "flags", in order that the user can once again see where the problems may be arising from the adopted pest management strategy. For instance, there may be no environmental impact from soil due to the nature of the soil and the pesticide. Additionally there may be no environmental impact to water due to a 15 metre buffer strip, but there may be a potential impact to applicators where hand held sprayers are used. Particularly where the pesticide falls into Category 1: DANGER. The user will immediately see that the major cause for concern is with application method, and can therefore act accordingly. The options open to the farmer will be to reduce pesticide use, change the method of application and/or provide better protective clothing, or use an alternative compound with a lower EIQ value in the farmworker category. The flags for water, soil or applicator hazard are:

1. Water "HAZARD"
2. Soil "HAZARD"
3. Applicator "HAZARD"

The term "HAZARD" or "OK" appears in the soil, water or application columns dependent on certain criteria being met. This is dependent on both EIQ and site specific criteria. One of the terms "DANGER, WARNING, CAUTION[51]" appears representing the EIQ descriptor for each pesticide (Tables 7.15 and 7.16). A series of IF > THEN formulae are used to assign impact flags to each pesticide. These take the form:

A. EIQ Criteria =IF(C8 <10, "CAUTION", IF(C8>10<40,"WARNING", IF(C8>40, "DANGER","WARNING")))

---

[51] This follows the US EPA classification of pesticides where even the most benign of chemicals are assigned the Caution rating.

Which reads if EIQ value is less than 10 (the lowest impact group), read CAUTION. If EIQ is between 10 and 40 (the mid-range impact group), read WARNING, and if EIQ is greater than 40 read DANGER.

B. Applicator=IF(C8>40,"HAZARD",IF(AND(C8>20,C5>1),"HAZARD", "OK"))

Which reads if EIQ value is greater than 40 read HAZARD, and if EIQ value exceeds 20 (an arbitrary secondary threshold), but application method is greater than 1 (1 represents safe application methods, whereas 3 or 5 represent less safe methods, Table 7.17), also read HAZARD. This brings out the site specific criteria of application method. An EIQ value, DANGER, will always read HAZARD, because these pesticides should be targeted for replacement with less harmful compounds.

C. Water =IF(C9>30, "HAZARD",IF(AND(C9>10,C4>1),"HAZARD", "OK"))

Which reads if EIQ value (C9) is greater than 30 (the highest impact group for Consumers), read HAZARD, and if EIQ value exceeds 10 (the secondary threshold), but the site specific water variable is rated greater than 1 (signifying a small, or the absence of a, buffer strip) also read HAZARD. Again this allows for the site specific water variable to be accounted for.

D. Soil =IF(C9>30, "HAZARD",IF(AND(C9>10,C3>1),"HAZARD", "OK"))

Which reads if EIQ value (C9) is greater than 30 (the highest impact group for Consumers), read HAZARD, and if EIQ value exceeds 10 (the secondary threshold), but the site specific soil variable is rated greater than 1 (signifying a highly porous, low organic content soil) also read HAZARD. Again this allows for the site specific soil variable to be accounted for.

E. FUR =IF(G14>C14*D14*10, "REDUCE", "OK")

This formula has been included purely to act as a reminder to the farmer. All pesticides are flagged REDUCE if use rates exceed 10 kg / ha of active ingredient. The farmer can then check to see if pesticide use rates correspond to recommended rates, and are thus acceptable. This should prevent farmers from using pesticides excessively. The formula basically states if Field Use Rating (G14) is greater than EIQ value (C14), multiplied by %a.i (D14), multiplied by 10 (kg/ha/a.i) read REDUCE.

It is argued that this system of data presentation is far more effective in advising farmers of the potential environmnetal impacts from pesticide use that the original Kovach et al., (1992) model. The Kovach model presents a single figure of environmnetal impact through the Field Use Rating, for each pesticide, allowing the farmer to choose a less harmful pesticide option. However, adopting the above data presentation format allows the farmer to see exactly where the problem potentially lies. It could be in quantity of pesticide used rather than the toxicity of the pesticide, could be a proximity to water source problem, or could be a more toxic pesticide advising caution with applicators and application methods. The changes suggested in section 7.2.5.2 above advise the farmer exactly what the problem might be rather than simply stating that pesticide A is more harmful than pesticide B.

For example, it might not be justifiable to give a herbicide a very high EIQ value for all circumstances. One of the major problems with herbicides is that of potential impact on water sources due to high solubility levels of herbicides (Conway and Pretty, 1991). Indeed herbicides pose minimal threats to non-target organisms due to their low toxicity levels (Conway and Pretty, 1991). Thus, a farmer may be dissuaded from using certain herbicides if using the EIQ criteria alone. If the farmer is operating well away from ground or surface water supplies, does not use irrigation and is not farming on steep sided hills, there may be no justification in avoiding certain products given that the risk to local water supplies would be extremely low in these circumstances[52] (Jones, 1990,

---

[52] Kovach et al., (1992), assign some herbicides high EIQ values due to their potential for contaminating water sources and impacting upon non-target aquatic organisms alone.

Cohen, 1990). Including site specific variables in the model would address such issues.

7.3. Table 7.16. interpretation.

Reading from the right hand column to the left the first indicator is for pesticide impact alone. Thus for the first pesticide, Benlate, the indicator, "WARNING" is displayed. This suggests that this compound is of mid range toxicity, and may warrant replacement when site specific variables are considered. Site specific variables have been programmed into the decision support spreadsheet, designed in Excel 5.0, and are as illustrated in Table 7.19. These values were arbitrarily assigned, and imply that the hazard to soil is low, hazard to water is low (due to production away from water sources) and application method is satisfactory, again affording low hazard.

The first row of Table 7.16 is explicitly concerned with farmworker impacts, thus the Applicator column reads "OK". This indicates that the application method adopted is safe enough to warrant continued use of the pesticide Benlate (Table 7.17, and 7.19). The next two columns, Water and Soil, read "N/A" because these categories are irrelevant to farmworker safety in this context. The FUR column reads "OK", as the kg/ha/a.i do not exceed 10. The user can then go on to examine both the consumer and ecological rating for Benlate. For both of these categories the indicator "DANGER" has been assigned due to the high EIQ values, thus, the water and soil components of the model automatically read "HAZARD". This is purely because where a pesticide is rated "DANGER" the goal is one of replacement with a less harmful compound.

In the case of Dithane, the FUR column indicates "REDUCE". This is because the kg/ha/a.i are equal to 10kg. Thus the model is prompting the farmer to check the use rate of this compound, especially as this is a mid range compound with an EIQ rating "WARNING". It may be that the farmer is accidentally using an excessive amount of the compound, and the model serves as a reminder to check use rates.

Table 7.16. Decision support spreadsheet for pesticide use for determining acceptable pesticide use levels.

Compound	Impact	EIQ	% a.i.	kg / ha	ECU / Kg	FUR	a.i. / ha	FUR	SOIL	WATER	APPLICATOR	EIQ
Benlate	Farmworker	18	0.5	0.6	37.81	5.4	0.3	OK	N/A	N/A	OK	WARNING
	Consumer	70	0.5	0.6	37.81	21	0.3	OK	HAZARD	HAZARD	N/A	DANGER
	Ecological	209	0.5	0.6	37.81	62.7	0.3	OK	HAZARD	HAZARD	N/A	DANGER
Guthion	Farmworker	36	0.85	6	13.89	183.6	5.1	OK	N/A	N/A	OK	WARNING
	Consumer	5	0.85	6	13.89	25.5	5.1	OK	OK	OK	N/A	CAUTION
	Ecological	88.3	0.85	6	13.89	450.33	5.1	OK	OK	OK	N/A	WARNING
Dithane	Farmworker	20	0.8	12.5	5.38	200	10	REDUCE	N/A	N/A	OK	WARNING
	Consumer	35	0.8	12.5	5.38	350	10	REDUCE	HAZARD	HAZARD	N/A	DANGER
	Ecological	105	0.8	12.5	5.38	1050	10	REDUCE	HAZARD	HAZARD	N/A	DANGER
Dodine	Farmworker	10.3	0.45	0.5	24.22	2.3175	0.225	OK	N/A	N/A	OK	WARNING
	Consumer	20.4	0.45	0.5	24.22	4.59	0.225	OK	OK	OK	N/A	WARNING
	Ecological	116	0.45	0.5	24.22	26.1	0.225	OK	HAZARD	HAZARD	N/A	DANGER
Rubigan	Farmworker	4	0.12	0.5	24.22	0.24	0.06	OK	N/A	N/A	OK	CAUTION
	Consumer	43	0.12	0.5	24.22	2.58	0.06	OK	HAZARD	HAZARD	N/A	DANGER
	Ecological	138	0.12	0.5	24.22	8.28	0.06	OK	HAZARD	HAZARD	N/A	DANGER
Copper ox	Farmworker	6.15	0.5	7.5	3.6	23.063	3.75	OK	N/A	N/A	OK	CAUTION
	Consumer	9.05	0.5	7.5	3.6	33.938	3.75	OK	OK	OK	N/A	CAUTION
	Ecological	137	0.5	7.5	3.6	513.75	3.75	OK	HAZARD	HAZARD	N/A	DANGER
TCHE	Farmworker	30	0.5	4.8	6.73	72	2.4	OK	N/A	N/A	OK	WARNING
	Consumer	4	0.5	4.8	6.73	9.6	2.4	OK	OK	OK	N/A	CAUTION
	Ecological	62.8	0.5	4.8	6.73	150.72	2.4	OK	OK	OK	N/A	WARNING
Ultracide	Farmworker	60	0.4	2	25.08	48	0.8	OK	N/A	N/A	HAZARD	DANGER
	Consumer	8	0.4	2	25.08	6.4	0.8	OK	OK	OK	N/A	CAUTION
	Ecological	140	0.4	2	25.08	112	0.8	OK	HAZARD	HAZARD	N/A	DANGER
Parathion	Farmworker	140	0.5	30	9.41	2100	15	REDUCE	N/A	N/A	HAZARD	DANGER
	Consumer	8	0.5	30	9.41	120	15	REDUCE	OK	OK	N/A	CAUTION
	Ecological	165	0.5	30	9.41	2475	15	REDUCE	HAZARD	HAZARD	N/A	DANGER
						8057.1	**NOT IPM**					

Table 7.17 Site specific impact ratings.

Soil	Rating	Water	Rating	Applicator	Rating	Rating	Impact
A	5	No Buffer	5	Irrigated Spray	5	5	High
B	3	Up to 5m	3	Hand held spray	3	3	Medium
C	1	5 - 10m	1	Tunnel spray	1	1	Low
D	0	> 10m	0	Tractor spray	1	0	None

Table 7.18 Impact Categories.

Min-Maximum permissible thresholds				
Chemical	**Farmworker**	**Consumer**	**Ecological**	**Class**
Benlate	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	F
Copper oxy.	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	F
Dithane	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	F
Dodine	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	F
Guthion	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	I
Parathion	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	I
Rubigan	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	F
TCHE	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	I
Ultracide	<10, 10 to 40, >40	<10, 10 to 30, >30	<50, 50 to 100, >100	I

Low Impact - CAUTION. Medium Impact - WARNING. High Impact - DANGER

Table 7.19. Site specific variables relating to Table 8.16, Portugal.

Criteria	Rating
Soil	1
Water	1
Application	1

7.4 Discussion.

The data presentation format of Table 7.16 increases the usability of the EIQ model at both the farm and policy level. When taken together with the explicit display of values for each component part of the EIQ equation, and re-worked values for fungicides and herbicides based on new weights, it is argued that the modified EIQ model displays more effectively the impacts associated with individual pesticides, as well as overall pest management strategies. However, the use of the EIQ model in this way raises a whole series of questions that will be further addressed in Chapter 8.

As it stands, the Kovach et al., (1992) model allows farmers to choose between pesticides, or whole pest management strategies, based on least environmnetal impact. This is useful information, but taken in isolation can be misleading. If such data are then added to site specific data, such as proximity to water source, application method, soil type and slope angle, and the EIQ is broken down into farmworker, consumer and ecological impacts rather than the average figure presented, then it becomes much more useful. For example, as has already been noted, herbicides used many metres (over 18 metres is generally sufficient, Eke et al., 1996) from a water source, and applied using targeted sprays avoiding spray drift, pose little risk to water organisms and the farmworkers applying the pesticide. Thus, the EIQ figure of Kovach et al., (1992) should be adjusted to show this.

Table 7.16 illustrates how this might be implemented, and displays where the potential hazard actually lies; to farmworkers, consumers or ecology. It also shows what the site specific problems are, such as proximity to water sources. Table 7.16 then assigns an explicit descriptor; CAUTION, WARNING, DANGER, which is much more understandable that the need to interpret a single EIQ figure. Thus, it is argued that Table 7.16 represents a much more effective mechanism for providing farmers with the environmnetal impact data they may require if they wish to pursue an IPM strategy.

8. Conclusions and recommendations for future research

The aim of this thesis was to identify whether or not the environmental effects of pesticides could be accounted for in a manner that would facilitate the inclusion of external effects in decision making. To this end pesticide use patterns were examined for the major apple producing regions of Europe. A number of pesticide impact models were reviewed, and applied to available data, in order to select the model that was the most:

a) user friendly,
b) understandable to decision makers at all levels,
c) understandable to the public, and
d) one that gave the most useable results when applied to data from the apple growing sector.
(See Tables 4.2, 4.3a and 4, Chapter 4).

The model chosen was the Environmental Impact Quotient of Kovach et al., (1992). The purpose of this model was to assist those farmers wishing to practice IPM, to make the most appropriate choice of pest management strategy based on the environmental impact properties of varying compounds. This model had not been tested and reported in academic journals, thus one of the main aims of this thesis was to ascertain whether useable impact data could be achieved. This thesis argues that, when certain changes are made to the EIQ equation and format of data presentation, the EIQ model describes the environmental impacts of pesticide use in a way that can prove useful to farmers and policy makers.

8.1. Model Modifications

Through an examination of the current literature regarding pesticide ranking models, and through the use of the EIQ model with pesticide data from the apple growing sector, certain criticisms of the modelling approach adopted by Kovach et al., (1992) became apparent. They were:

1. The linearity of the damage function.
2. The averaging of effects to achieve a single impact figure.
3. The weighting criteria used.
4. The lack of site specific variables.
5. The method of data presentation.

Each of these criticisms were addressed in Chapter 7 in order that the model would represent local conditions more realistically when identifying environmental impact from pesticide use. It was decided that the linear damage function did in fact adequately describe the dose response relationship between compound and target or non-target organisms. Thus the EIQ FUR was not altered. The averaging of impacts across farmworker, consumer and ecological impacts was seen to be a major flaw in the model, and as such was changed to explicitly display an impact figure for each of these categories.

The weighting criteria used also came under question on two grounds. Firstly, because there was no possibility of a zero score (Dushoff et al., 1994, Levitan et al., 1995), and secondly because all compounds were weighted the same, even though toxicological impacts are likely to be quite different between different groups of compounds (Conway and Pretty, 1991). Once again, the weighting criteria was changed to allow for a zero score, and to capture the differences in the nature of impact between insecticides, fungicides and herbicides.

Site specific variables, which are absent in the EIQ model, were included in a decision support spreadsheet to represent site specific conditions. It was thought that any changes to the actual EIQ equation, in terms of added variables and weights, would only make the approach more complex. Since one of the great strengths of the EIQ model is its simplicity and ease of use, it was felt that site specific variable could be better catered for outside the main equation. Thus a pre-programmed decision support aid was presented in Chapter 7.

The data presentation method of the EIQ model has also been criticised (Dushoff et al., 1994). Thus, once again, a decision support spreadsheet was presented that, it is argued, better displays the nature of the environmental impacts of pesticide use, site specific variables, pesticide price, and informs the decision maker whether or not a particular pest management strategy conforms to acceptable impact thresholds. The spreadsheet can be pre-programmed by an IPM accreditation body, for example (Penrose et al., 1994, Levitan et al., 1995), thus the data input task of the farmer is minimised.

The changes in the EIQ methodology mentioned above, strengthen this approach of identifying the environmental impacts of pesticide use. The model is presented in a user friendly way that is understandable to decision makers at the farm and/or regional level, and should prove understandable to the general public. Pesticide use can subsequently be monitored (potentially on line) by an accreditation body wishing to promote low impact agriculture.

8.2 Suggestions for further research

This thesis argues that the current framework of quantitative risk assessment of pesticides falls short of the informational needs of farmers, and the regulatory needs of policy makers. A lack of information, and a breakdown of consumer sovereignty, is a cause of market failure (Johansson, 1985), which in some circumstances will create externalities. The modelling approach adopted by, amongst others Kovach et al., (1992), Reus and Pak, (1993) and Penrose et al., (1994), seeks to provide additional information to farmers and decision makers in an attempt to explicitly account for environmental considerations in decision making.

It has been noted that the development of models, such as the EIQ, fill data gaps in the risk assessment process, and as such are useful aids to farm decision making (Dushoff et al., 1994, Levitan et al., 1995). Before they are fully acceptable to decision makers, however, further research is required that refine the models, assess the wider acceptability of such models in terms of consumer confidence and farm level economics, and identify ways in which the success of initiatives such as EIQ based IPM schemes can be monitored. Thus, the key areas for future research can be summarised as follows:

1. It is necessary to analyse what the effects of, for example, using pesticides with a lower EIQ (hence lower environmental impact), might have on farm level economics. It is likely that farmers will only embrace such environmental initiatives if they represent no loss of income. This poses a whole series of additional questions that have not been addressed by this thesis, such as:

    a) Are pesticides with a lower EIQ as efficacious as those with a higher EIQ?

    b). How does the risk aversion of farmers affect the uptake of the use of models such as the EIQ?

    c) Does using a lower EIQ pesticide necessarily mean using higher dosages?

    d) Would a shift toward IPM result in reduced cosmetic appearance of the product, and would the consumer tolerate this?.

    e) Is the consumer prepared to pay a price premium based on an IPM label, and would they have any confidence in an IPM label?

2. In the majority of models presented in Chapter 4, data gaps exist that reduce the acceptability of the modelling approach. Further research is required to fill these data gaps, and it has been suggested by some that these models can actually assist in the highlighting of areas that require a more intensive research effort (Newman, 1995).

3. The EIQ model lends itself to incorporation into either decision support spreadsheets (see Chapter 7), or inclusion in Linear Programming and MCDM models (Verhoeven et al., 1994, Quin et al., 1996, 1997). This is an area where

further research and development could result in potentially very useful decision making and policy tools. Decision support spreadsheets may also be practical as an environmental and policy monitoring tool. All that would be required would be for a farmer to download pesticide use data onto a centrally held database (an IPM accreditation body for example) via e-mail or the Internet. Thus initiatives such as sustainable agriculture schemes could be monitored by any one of a number of groups (co-operatives, extension services, Government departments, commercial research stations and academic institutions).

4. Attitudes of the public to a pesticide related "eco", green or IPM label based on the EIQ or PI models need to be assessed. The explicit purpose of the Penrose et al., (1994) model is as an IPM accreditation scheme. The EIQ model of Kovach et al., (1992) has also been developed as an aid to IPM practitioners, the logical extension being an IPM label.

5. In order that an IPM label can be assigned to food products acceptable, or unacceptable, thresholds must be identified. Thus an analysis is required of a) what constitutes IPM production and b) what pesticide use rates represent IPM. Only then can an EIQ value be assigned to products, and hence a label.

8.3. Summary

The attitudes of some in the agro-chemicals industry towards models such as the EIQ, are that they duplicate data presented in quantitative risk assessment, they do not adequately describe risk, and that the current legislation (91/414/EC) is sufficient to afford the environment the protection that the EU demands (Bernard Johnen, Director, Environmental Research and Development, Zeneca Ltd. pers. comm, 1995). This thesis contends, however, that current EU legislation, risk assessment and the agro-chemicals industry has failed to provide the various actors in agricultural systems (from producer to consumer) with information that will allow them to choose between pest management strategies (and therefore produce for consumers) that are more or less harmful to the environment.

Pesticide ranking indices go some of the way to redress the informational imbalance that currently exists.

The weaknesses of the modelling approach to the environmental impacts of pesticide use are discussed in some detail in Chapter 7, but can be summarised as being:

1. Problems with the mathematical approach of models such as the EIQ, including the summing of different impact parameters.
2. The potentially over simplified relationship between dose and damage in a linear damage function.
3. Toxicological data gaps within these models.
4. The subjective weighting and rating of impacts and impact categories.
5. The choice of the most appropriate means of presenting and communicating results.

This thesis argues that the modelling approach adopted by Kovach et al., (1992), and modified in Chapter 7 (Table 7.17), does adequately describe the environmental impacts associated with pesticides and can contribute greatly to the decision making process with regards to pesticide use.

REFERENCES

Akkerhuis, G and van der Voet, H (1992). Dose effect relationship for the effect of detamethrin on a linyphiid spider population in winter wheat. Arch. Of Environ. Contam and Toxicol, 22, pp 114-121.

Arnold, D. J and Briggs, G. G (1990). Fate of pesticides in soil: predictive and practical aspects. In Hutson, D. H and Roberts, T. R. (Eds). Environmental fate of pesticides. John Wiley and Sons.

Attwood, P. (Ed). Crop Protection Handbook: Cereals. BCPC.

Bahr, N (1992). Economic theory and the welfare state: a survey and interpretation. Journal of Economic Literature, Vol XXX, pp 741-803.

Barde, J. P and Opschoor, J. B (1994). From stick to carrot in the environment. The OECD Observer, No 186, pp 23-27.

Bator, F. M (1958). The anatomy of market failure. Quarterly Journal of Economics, LXXII, pp 351-379.

Baumel, W. J. and Oates, W. E. 1988. The Theory of Environmental Policy (2nd Edtn). Cambridge University Press.

Beaumont, P (1993). Pesticides, Policies and People: a guide to the issues. The Pesticide Trust, London.

Becker, R. L. Herzfeldt, K. R. Ostlie, E. and Stamm-Katovich, E. J. (1989). Pesticides: surface runoff, leaching and exposure concerns. Minnesota Extension Service Publications. AG-BU-3911.

Behr, H. C (1990). An evaluation of alternative EC market policies for fruit and vegetables. European Review of Agricultural Economics, 17, pp 1-17.

Berg, M and Scheringer, M (1994). Problems in environmental risk assessment and the need for proxy measures. Fresnius Envir. Bull, 3, pp 487-492.

Bergman, L and Pugh, M. (Eds) (1994). Environmental Toxicology, Economics and Institutions: The Atrazine Case Study. Kluwer.

Beyer, W. M and Kritinsky, A. J (1989). Long-term persistence of dieldrin, DDT and heptachlor in earthworms. Ambio, 18, pp 271-273.

Beyers, D. W et al., (1994). Toxicity of carbaryl and malathion to two federally endangered fish species, an estimated regression and anova. Environmental Toxicology and Chemistry, 13, pp 101-107.

Birch, S and Gafni, A (1993). Changing the problem to fit the solution: Johannesson and Weinstein's (mis) application of economics to real world problems. Journal of Health Economics, 12, pp 469-476.

Blommers, L. H. M (1994). Integrated Pest Management in European apple orchards. Annual review of entomology, 39, pp 213-241.

Bockstael, N. E and Kling, C. L (1988). Valuing environmental quality: weak complimentarity with sets of goods. American Journal of Agricultural Economics, 67 (4), pp 654-662.

Boulding, K. (1992). The economics of the coming spaceship earth. In Markandya, A and Richardson, J . (Eds) The Earthscan reader in Environmental Economics. Earthscan, London.

Bowles, R. G and Webster, J. P. G (1995). Some problems associated with the analysis of the costs and benefits of pesticides. Crop Protection, 14 (7), pp 593-600.

Brouard, C et al., (1994). Mobility of phenylurea herbicides in soil columns. In Copin et al., (Eds), Environmental behaviour of pesticides and regulatory aspects. European Study Service, Brussels.

Brouwer, F. M. Terluin, I. J. and Godeschalk, F. E. 1994. Pesticides in the EC. LEI-DLO.

Brunet, R and Cyr, A (1992). The impact of dimethoate on rhythms of three graniverous birds. Agriculture, Ecosystems and Environment, 41, pp 327-336.

Carlson, G. A. Zilbermann, D. and Miranowski, J. A. 1993. Agricultural and Environmental Resource Economics. Oxford University Press, New York.

Carson, R (1962). Silent Spring. Houghton Mifflin, Boston.

Centro Operativo Ortofrutticolo, 1994. Apple and pear crop forecast. Prognosfruit.

Ceron, J. J et al., (1995). Toxicological effects in rabbits induced by endosulfan, lindane, and methylparathion representing agricultural by-products contamination. Bull. Environ. Contam and Toxicol, 54, pp 258-265.

Chester, G and Woollen, B. H (1982). Studies of the occupational exposure of Malaysian plantation workers to paraquat. British Journal of Industrial Medicine, 39, pp 23-33.

Clark, D. R et al., (1996). Small mammal populations in Maryland meadows during four years of herbicide applications. Environmental Toxicology and Chemistry, 15 (9), pp 1544-1550.

Clark. L et al., (1993). Acute toxicity of the bird repellent Methyl Anthranilate to Fry of Salmo salar, Oncorhynus mykiss, Ictalurus punctatus and Lepomis macrochirus. Pesticide Science, 39, pp 313-317.

Clawson, R and Clark, D (1989). Pesticide contamination of endangered grey bats and their food base in Boone County, Missouri in 1982. Bull. Env. Contam. and Toxicol, 42, pp 431-437.

Coase, R (1960). The problem of social cost. Journal of law and economics, 3, pp 1-44.

Coenen, T. M. M and Brouwer, A (1992). Sub-chronic toxicity and reproduction effects of tri-n-butylin oxide in Japanese Quail. Arch. Of Environ. Contam and Toxicol, 23, pp 457-463.

Roberts, T. R. (Eds) Environmental fate of pesticides. John Wiley and Sons.

Commission of the European Communities (1991). Council Directive of 15th July, 1991, concerning the placing of plant protection products on the market (91/414/EEC). Official journal of the European Communities, No L230, pp 1-31.

Commission of the European Communities (1992a). Towards Sustainability. Vol. II, CEC, Brussels.

Commission of the European Communities (1992b). Council Directive 92/32/EC. Official Journal of the European Communities. L154, Vol 35.

Commission of the European Communities (1992c). Commission Regulation No 3600/92. Official Journal of the European Communities. L366, Vol 10.

Commission of the European Communities (1994). Council Directive 91/414/EEC. Official Journal of the European Communities. L227, Vol 31.

Conway, G. R and Pretty, J. N. (1991). Unwelcome harvest: agriculture and pollution. Earthscan Publications Ltd, London.

Cresswell, C. J et al., (1994). Control of greenhouse Whitely on Gerberas by systemic application of Acephate. Pesticide Science, 42, pp 13-16.

Cropper, M. L. and Oates, W. E. Environmental Economics: A Survey, 1992. Journal of Economic Literature. Vol XXX 675-740.

Cumberland, J. and Kahn, J. 1982. The evaluation of marginal damage functions: An assessment of the information requirements. Northeast Regional Science Review, 12, 1-12.

Dales, J. H. (1992). The property interface: In Markandya, A. And Richardson, J. Earthscan Reader in Environmental Economics. Earthscan Publications Ltd, London.

Daly, H. (1992) The economic growth debate: what some economists have learnt but many have not. In Markandya, A and Richardson, J . (Eds) The Earthscan reader in Environmental Economics. Earthscan, London.

Davis, J. U et al., (1992). American Journal of Agricultural Economics, 74 (1), pp 907-917.

Dillaha et al., 1985. Vegetative filter strips for agricultural non-point source pollution control. Trans. ASAE, 32, pp 513-519.

Drummond, C. J (1995). Landscape management: central to the whole farm policy. Symposium Proceedings No 63, Integrated Crop Protection: Towards Sustainability.

Dubgaard, A (1991). Pesticide regulation in Denmark. In Hanley, N (Ed). Farming and the countryside: an economic analysis of external costs and benefits. CAB International.

Duffus, J and Worth, R (1995). Fundamental Toxicology. Royal Society of Chemistry, London.

Dushoff, J. Caldwell, B. and Mohler, C. L. (1994) evaluating the environmental effect of pesticides: a critique of the Environmental Impact Quotient. American Entomologist, Fall 1994, pp 180-183.

Edwards-Jones, G (1993). Environmental auditing in agriculture: potential and practicality. BCPC - Weeds, 8B-3, pp 1125-1134.

Edwards-Jones, G et al., (1992). Environmental auditing and its relevance to agriculture. Farm Management, 8 (2), pp 73-81.

Eke, K.R et al., (1996). Impact of agricultural pesticides on water quality, in Hester, R.E and Harrison, R. M (Eds). Agricultural Chemicals and Environment. The Royal Society of Chemistry, London.

Eurofruit Editorial. Italian growers call on EU to rule on IPM. Eurofruit, July 1995.

Faasen, R. 1994. Workshop on a framework for the sustainable use of plant protection products in the EU. Proceedings. Brussels

Feeney, D. Berkes, F. McKay, B. G. and Acheson, J. M. (1990). The tragedy of the commons: twenty two years later. Human Ecology, 18 (1), pp 1-9.

Fenmore, P. G. and Norton, G. A. 1985. Problems of implementing improvements in pest control: A case study of apples in the UK. Crop Protection, 4 (1), 51-70.

Fisher, A. J and Peterson, F. P (1976). The environment in Economics: A Survey. Journal of Economic Literature, 14, pp 1-33.

Flint, M. L and van den Bosch, R (1981). Introduction to IPM. Plenum Press, New York.

FoE (1989). An investigation of pesticide pollution in drinking water in England and Wales. FoE, London.

Forastiere, F. (1993). Cancer among farmers in central Italy. Scandinavian Journal of Work and Environmental Health, 19, 382-389.

Freeman III, A. M (1974). On estimating air pollution control benefits from land value studies. Journal of Environmental Economics and Management, 1, PP 74-83.

Gaillardon, P (1994). Study of the availability of Isoproturon and its degradation in soil products. In Copin et al., (Eds), Environmental behaviour of pesticides and regulatory aspects. European Study Service, Brussels.

Garry, V. F et al., (1994). Survey of health and use characterisation of pesticide appliers in Minnesota. Archives of Environmental Health, 49 (5), pp 337-343.

Gittinger, J. P (1982). Economic analysis of agricultural projects (Second Edition). EDI Series in World Development, The World Bank.

Green, E and Mumford, J (1995). Policy needs and options for agricultural pesticide controls in the European Union. Proceedings from the Workshop on Pesticides.

Grifin, R. C. 1991. The welfare analytics of transaction costs, externalities and institutional choice. American Journal of Agricultural Economics, 73 (3), 601-614.

Groot, M. J (1996). A decision support system for fruit crops for economic and ecological considerations. Acta Horticulturae Economics Review.

Gutsche. V (1995). The influence of pesticides and pest management strategy on wildlife. Sustainable Farming Systems Symposium Proceedings No 63. Integrated Crop Protection: Towards Sustainability. SFS - BCPC.

Hahn, R. W (1989). Economic prescriptions for environmental problems: how the patient followed the Doctors orders. Journal of economic perspectives, 3 (2), pp 95-114.

Hanley, N and Spash C, L (1993). Cost Benefit Analysis and the Environment. Edward Elgar, London.

Hanley, N. Hallet S and Moffatt, I (1990). Research policy and review, 33. Why is more notice not taken of economists prescriptions for the control of pollution? Environment and Planning A, pp 1421-1439.

Harper, C. R. and Zilbermann, D. 1992. American Journal of Agricultural Economics, 74 (1), 68-78.

Hartwick, J. M. and Olewiler, N. D. (1986). The economics of natural resource use. Harper and Row, New York.

Helson, B. V et al., (1994). Laboratory toxicity of six pesticides for four species of bees. Arch. of Environ. Contam and Toxicol, 27, pp 107-114.

Henderson, P. T et al., (1993). Risk assessment for worker exposure to agricultural pesticides: review of a workshop. Ann. Occup. Hygiene, 37 (5), pp 499-507.

Higley, L. and Wintersteen, W. (1992). A novel approach to environmental risk assessment of pesticides as a basis for incorporating environmental costs into economic injury levels. American Entomologist, 39, pp 34-39.

Hinton, L. 1987. The European market for fruit and vegetables. Elsevier Applied Science.

Hoag, D. L and Hornsby, A. G (1992). Coupling groundwater contamination with economic returns when applying pesticides. Journal of Environmental Quality, 21, pp 579-586.

Hull, R. (1994). General introduction to the Fifth Environmental Action Programme. Proceedings of the Workshop for the framework for the sustainable use of plant protection products in the EU. Brussels, pp 119-126.

International Journal of Land Management. In Press.

Johansson, P (1987). The economic theory and measurement of environmental benefits. Cambridge University Press.

Johnen, B, (1994). Development of a sound basis for regulatory and market related instruments. Proceedings of the Workshop for the framework for the sustainable use of plant protection products in the EU. Brussels, pp 119-126.

Johnen, B. (1995). Risk assessment and crop protection products use reduction. Proceedings from the Workshop on Pesticides. Wageningen Agricultural University, August, 1995.

Jones, R. L (1990) Pesticides in groundwater: conduct of field research studies. In Hutson, D. H and Roberts, T. R. (Eds) Environmental fate of pesticides. John Wiley and Sons.

Juahiainen, A et al., (1991). Occupational exposure of forest workers to Glyphosate during brush saw spraying work. Am. Ind. Hyg. Assoc. Journal. 52 (2), pp 61-64.

K. A. Lewis, C. E. Bloom, M. J. Newbold, A. M. Hall (1996). Environmnetal Management for Arable Agriculture: An eco-rating system for pesticide use. Hertfordshire University Working Paper.

Kahn, J. 1987. Economic damage from herbicide pollution in Chesapeake bay. Project Appraisal 2 (3), 142-152.

Kahn, J. and Kemp, W. M. 1985. Economic losses associated with the degradation of an ecosystem: The case of submerged aquatic vegetation in Chesapeake bay. Journal of Environmental Economics and Management, 12, 246-263.

King, J. G. Alston, F. H. Battle, I. et al., 1991. The European apple genome mapping project: Developing a strategy for mapping genes coding for agronomic characters in tree species. Euphytica 56: 89-94.

Kjaer, C and Elmegaard, N (1996). Effect of herbicide treatment on host plant quality for a leaf eating beetle. Pesticide Science, 47, pp 319-325.

Klein, A. W. and Goedicke, J. (1993). Environmental assessment of pesticides under Directive 91/414/EEC. Chemosphere, 26, pp 979-1001.

Kovach, J. Petzoldt, C. Degni, J. and Tette, J. 1992. A method to measure the environmental impact of pesticides. New York's Food and Life Sciences Bulletin, 139, 1-8.

Krupnick, A and Alicbusan, A (1990). Estimation of health benefits of reductions in ambient air pollutants. Divisional Working Paper No 1990-11, The World Bank.

Krutilla, J. A (1967). Conservation reconsidered. American economic review, 57, pp 777-786.

Kula, E (1994). Economics of natural resources, the environment and policies. Chapman and Hall, London.

Lagadic, L and Bernard, L (1993). Topical and oral activities of Imidacloprid and Cyfluthrin against susceptible laboratory strains of Heliothis virescens and Spodoptera littoralis. Pesticide Science, 38, 323-328.

Lavy, T. L et al., (1992). Conifer seedling nursery workers exposure to Glyphosate. Arch. Environ. Contam. Toxicol, 22, pp 6-13.

Lever, B. G (1990). Crop Protection Chemicals. Ellis Howood, Chichester.

Levitan, L (1997). An overview of pesticide impact and risk assessment systems. OECD, 1997.

Levitan, L. Merwin, I. and Kovach, J. (1995). Assessing the relative environmental impacts of agricultural pesticides: the quest for an holistic method. Agriculture, Ecosystems and Environment, 55, pp 153-168.

Lichtenberg, E, Parker, D. D and Zilbermann, D (1988). Marginal analysis of welfare costs of environmental policies: the case of pesticide regulation. American Journal of Agricultural Economics, 67 (4), pp 867-874.

Mann, S. H (1982). The Economic utility of health affected by environmental change. Journal of Environmental Management, 15, pp 301-310.

Markandya, A and Richardson, J (1992). The Earthscan Reader in Environmental Economics. Earthscan, London.

Marsh, J (1995). The policy approach to sustainable farming systems in the EU. Symposium Proceedings No 63, Integrated Crop Protection: Towards Sustainability. SFS and BCPC.

Mathiessen, P et al., (1994). Monitoring pesticide runoff and leaching in surface dominated catchment: implications for aquatic risk assessment. In Copin et al., (Eds), Environmental behaviour of pesticides and regulatory aspects. European Study Service, Brussels.

Maund, S. J et al., (1997). Ecological considerations in pesticide risk assessment for aquatic ecosystems. Pesticide Science, 49, pp 185-190.

McDonald, G. and Glynn, C. J. 1994. Difficulties of measuring adoption of apple IPM: A case study. Agricultural Ecosystems and the Environment, 48, 219-230.

Metcalf, R. L and Luckman, M (1975). Insecticides and Pest Management. In Metcalf, R. L (Ed). Introduction to Insect Pest Management, Wiley, New York.

Meyer, H (1993). Competition between technologies: the case of organic versus conventional farming. Unpublished PhD thesis.

Mishan, E. J (1969). The relationship between joint products, collective goods and external effects. Journal of Political Economy, LXXV11, PP 329-348.

Moran, S (1993). The toxicity of Brodifacoum wheat bait to the field rodents Microtus guentheri and Meriones tristrami. Pesticide Science, 38, pp 303-307.

Moses, M (1989). Pesticide related health problems in farmworkers. AAOHN Journal, 37 (3), pp 115-127.

Moses, M et al., (1993). Environmental equity and pesticide exposure. Toxicology and Industrial Health, 9 (5), pp 913-959.

Nebel, B. J and Wright, R. T (1996). The way the world works. Environmental science, Fifth Edition. Prentice Hall, London.

Neher, P. A. (1994) Natural Resource Economics: Conservation and Exploitation. Cambridge University Press.

Newman, A. (1995). Ranking pesticides by environmental impact. Environmental Science and Technology, 29 (7), pp 324-326.

Norgaard, R. B (1981). Sociosystem and ecosystem coevolution in the Amazon. Journal of Environmental Economics and Management, 8, pp 238-254.

Norgaard, R. B (1985). Environmental Economics: an evolutionary critique and a plea for pluralism. Journal of Environmental Economics and Management, 12, pp 382-394.

Norton, G. A and Mumford, J (1994). Decision tools for pest management. CAB International.

O'Bryan, T. R. and Ross, R. H. (1988). Chemical scoring system for hazard and exposure identification. J. Toxicol. Environ. Health, 25, pp 119-134.

O'Rourke, D. A (1990). Anatomy of a disaster. Agribusiness, 6 (5), pp 417-424.

O'Rourke, A. D. 1994. The World Apple Market. Food Products Press.

Oka, I. N (1991). Success and challenges of the Indonesian national Integrated Pest Management programme in the rice based cropping system. Crop Protection, 10 (3), pp 163-165.

Olkowski, W. Olkowski, S. and Daar, S. 1991. What is IPM? The IPM Practitioner XlIl (11/12), 1-8.

Oskam, A et al., (1992). Pesticide use and pesticide policy in the Netherlands. Wageningen Economic Studies, 26, The Netherlands.

Patty, L et al., (1997). The use of grassed buffer strips to remove pesticides, nitrate and soluble phosphorous compounds from runoff water. Pesticide Science, 49, pp 243-251.

Pearce, D. and Tinch, R. (1997). The true price of pesticides. In Vorley, W. (Ed). Bugs in the system: the pesticide industry and sustainable agriculture. Eearthscan, London.

Pearce, D. W and Turner, R. K (1990). Economics of natural resources and the environment. Harvester Wheatsheaf, London.

Pearce, N and Reif, J, S (1990). Epidemiological studies of cancer in agricultural workers. American Journal of Industrial Medicine, 18, pp 133-148.

Pedigo, L. P and Higley, L. G. (1992). The economic injury level concept and environmental quality. American entomologist, Spring 1992, pp 12-21.

Pedigo, L. P and Higley, L. P (1992). The economic injury level concept and environmental quality: a new perspective. American Entomologist, Spring, pp 12-21.

Penrose, L. J. Thwaite, W. G. and Bower, C.C. (1994). Rating index as a basis for decision making on pesticide use reduction and for accreditation of fruit produced under IPM. Crop Protection, 13 (2), pp 146-152.

Pezzey, J (1988). Market mechanisms for pollution control: Polluter Pays, Economic and Practical aspects. In Turner, R. K (Ed). Sustainable environmental management: principles and practice. Bellhaven Press, London.

Pigou, A. C (1920). The Economics of Welfare. Macmillan, London.

Pilling, E. D and Jepson, P. C (1993). Synergism between EBI fungicides and a Pyrethroid Insecticide in the Honeybee. Pesticide Science, 39, pp 293-297.

Pimentel, D, Kirkby, C and Shroff, A (1993). The relationship between cosmetic standards for food and pesticide use. In Pimentel, D and Lehman, H (Eds). The pesticide question, environment, economics and ethics. Chapman and Hall, London.

Pimentel, D. et al., (1991). Environmental and economic effects of reducing pesticide use. Bioscience, 41 (6), pp 402-409.

Pimentel, D. et al., (1992). Environmental and economic costs of pesticide use. Bioscience, 42 (10), pp 750-759.

Proceedings of the Brighton Crop Protection Conference; Pests and Diseases, November, 1996.

Prognosfruit (1995). European Apple and Pear Forecasts, 1995. COO, Italy.

Quin, F. S and Edwards-Jones, G (1997). The use of ranking indices in the regulation of pesticide use: the case of taxes. BCPC - Weeds, 6A-4, pp 469-476.

Quin, F. S. Albin, A. and McGregor, M. J. (1996) Developing an environmental - economic model of the European apple industry: assessing the impacts of change. Acta Horticulturae Economics Review, pp 56-66.

Quin, F. S. and McGregor, M. J. (1995). Damage Functions: Potential and Problems for Efficient Pesticide Regulation in the European apple industry. Proceedings from the Workshop on Pesticides. Wageningen Agricultural University, August, 1995.

Quin, F.S Albin, A and McGregor, M. J (1997). A framework for identifying the social, environmental and economic impacts of a biotechnology project: the European APPLE project. Project Appraisal 12 (1), pp 2-10.

Quy, R. J et al., (1995). Control of a population of Norway rats resistant to anticoagulant rodenticdes. Pesticide Science, 45, pp 247-256.

Reus, J. A. W. A et al., (1994). Towards a future pesticide policy, CLM, The Netherlands.

Reus, J. A. W. A. 1991. An environmental yardstick for pesticides: Development and testing. Centre for Agriculture and Environment, Utrecht.

Reus, J. A. W. A. and Pak, G. A. 1993. An environmental yardstick for pesticides. Med. Fac. Landbouwuniv. Gent, 58 (2a), 249-255.

Roberts, L (1989). Alar: the numbers game. Science, 243, pp 1430.

Roberts, L. (1989). Pesticides and Kids. Science, 243, pp 1280-1281.

Rodricks, J.V. (1994). Risk Assessment, the environment and public health. Public Health Perspectives, 102, pp 258-264.

Rola, A. G and Pingali, P (1993). Pesticides, rice productivity and farmers health: an economic assessment. IRRI - WRI.

Roush, R.T and McKenzie, J. A (1987). Ecological genetics of insecticide and acaricide resistance. Annual review of entomology, 32, pp 361-380.

Russell, T. 1989. An Alar dose a day: A cause for concern? BIBRA Bull, 28, 281-283.

Sagendra Tiwari and Fraser Quin (1997). Property Rights Revisited: Institutional Change and Land Reform. A Case Study of Community Forestry in Nepal.

Scharpe, A and Smeets, L (1994). The gaps in the regulatory aspects of environmental guidelines. In Copin et al., (Eds), Environmental behaviour of pesticides and regulatory aspects. European Study Service, Brussels.

Scharpe, A. (1994). General introduction to current EU legislation on plant protection products. Proceedings of the Workshop for the framework for the sustainable use of plant protection products in the EU. Brussels, pp 63-76.

Scheele, M (1994).General introduction to Agri-environmental policy. Proceedings of the Workshop for the framework for the sustainable use of plant protection products in the EU. Brussels, pp 51-62.

Schenk, A. M. E and Wertheim, S. J (1992). Components and systems research for integrated fruit production. Netherlands Journal of Agricultural Science, 40, pp 257-268.

Schlager, E. And Ostrom, E. (1992). Property rights regimes and natural resources: a conceptual analysis. Land Economics, 68 (3), pp 249-262.

SFS and BCPC.

Shahane, A. N and Inman, R. C (1987). Review of available pesticide rating systems. Hazard Assessment of Chemicals, 5, pp 103-132.

Stark, J. D and Rangus, T. M (1994). Lethal and sub-lethal effects of the insecticide formulation Margossan-O on the pea aphid. Pesticide Science, 41, pp 155-160.

Stern, V. M. et al., (1959). The integrated control concept. Hilgardia, 29 (2).

Tadlec, E (1994). Response to voles to the concentration of crimidine rodenticide baits. Crop Protection, 13 (6), pp 474-478.

Teague, M. L, Mapp, H. P and Bernardo, D. J (1995). Journal of Production Agriculture, 8 (3), pp 405-415.

The Pesticide Trust (1992). Pesticides: Missing the Target. A report by the Pesticide Trust for WWF, UK.

The Pesticide Trust (1995). Global Pesticide Sales. Pesticide News, 28.

Towards Sustainability. SFS and BCPC.

Van den Bosch, R. (1978). The pesticide conspiracy. Doubleday, New York.

Verhoeven, J. T. W. Wossink, G. A. A. and Reus, J. A. W. A. 1994. An environmental yardstick in farm economic modelling of future pesticide use: A case study of arable farming. Netherlands Journal of Agricultural Sciences, 42 (4), 331-341.

Vlahodimos, K. P (1994). General introduction to future trends in the use of plant protection products. Proceedings of the Workshop for the framework for the sustainable use of plant protection products in the EU. Brussels, pp 69-74.

Wageningen Agricultural University, August, 1995.

Wauchope, R. D (1992). Environmental risk assessment of pesticides: improving simulation model credibility. Weed Technology, 6, pp 753-759.

Wauchope, R. D. (1978). The pesticide content of surface water draining from agricultural fields-a review. Journal of environmental quality, 7, pp 459-472.

Weale, A (1992). The new politics of pollution. Manchester University Press, Manchester.

Webster, J. P. G (1995). Analysing the economic consequences of sustainability in farming systems. Symposium Proceedings No 63, Integrated Crop Protection:

Weitzman, M. L (1974). Prices Vs Quantities. Review of Economic Studies, 41 (4), pp 477-491.

WHO Toxicological Register for Pesticides, WHO, 1992-93.

Wijnands, F. G and van Dongen, G. J. M. (1995). Integrated Farming Systems and environmental exposure to pesticides. Proceedings of the Workshop on Pesticides, Wageningen, the Netherlands..

Winter, F (1986). Trends of production, cultivars and planting systems in apples and pears in western Europe. Acta Horticulturae, 160, pp 261-265.

Winter, F (1989). Parameters affecting the costs of apple production in different fruit growing areas. Acta Horticulturae, 243, pp 383-387.

Wynne, B. (1992). Uncertainty and environmental learning. Butterworth-Heinemann Ltd.

Appendix 1. EIQ Field Use Rating by Country. The EIQ figure is at the per ha per annum level (Source: Quin and Edwards-Jones, 1997).

COUNTRY:	**ITALY**				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicide					
Copper oxy.	10	0.5	5	33.3	166.5
Dithane	48	0.8	38.4	62.3	2392.32
Anvil	1.4	0.05	0.07	26.7	1.9
Saprol	8	0.19	1.52	32.9	50
total	**67.4**		**44.99**		**2610.72**
Insecticides					
Metasistox	3	0.25	0.75	82.5	61.9
Nomolt	0.6	0.15	0.09	58	5.2
Brution	5	0.42	2.1	69.3	145.5
Gusathion	16	0.85	13.6	43.1	586.16
total	**24.6**		**16.54**		**798.76**
Others					
Carbaryl	2.5	0.45	1.13	22.6	25.42
total	**2.5**		**1.13**		**25.42**
	94.5		**62.66**		**3435**

COUNTRY	**FRANCE**				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicides					
Sulphur	40	0.8	32	45.5	1456
TMTD	7.5	0.8	6	54.5	327
Captan	10	0.83	8.3	28.6	237.4
Ditahne	10	0.8	8	62.3	498.4
Bayleton	2	0.25	0.5	33.3	16.65
Thiovit	44	0.8	35.2	54.5	1918.4
Benlate	2	0.5	1	69.5	69.5
Mikal	6	0.75	4.5	55.9	251.52
Sandomil	2	0.45	0.9	29.2	26.28
Copper oxy.	26	0.082	2.13	33.3	80
total	**149.5**		**98.53**		**4881.15**
Insecticides					
Gusathion MS	2	0.85	1.7	43.1	73.3
Lannate 20L	4	0.2	0.8	32.8	26.2
Kilval	1.3	0.4	0.52	37.7	19.6
AzinugecPM	2	0.5	1	43.1	43.1
Cesar 100	0.5	0.1		12.8	0.64
Ultracide	2	0.4	0.8	69.3	166.3
Dichrolvos	6	0.5	3	40.6	121.8
total	**17.8**		**7.82**		**450.94**
Herbicides					
Aminotriazole	6.5	0.23	1.46	37.1	55.46
Basta	1.3	0.15	0.2	32.4	6.32
Simazine	3	0.5	1.5	15.7	23.55
2-4-D	1.25	0.5	0.63	56.3	35.2
total	**12.05**		**3.78**		**120.53**
	179.35		**110.13**		**5452.6**

COUNTRY:	**PORTUGAL**				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicides					
Dodine	0.5	0.45	0.23	34.9	7.85
Rubigan	0.5	0.12	0.06	27.3	1.64
Dithane	12.5	0.8	10	62.3	623
Benlate	0.6	0.5	0.3	69.5	20.85
Copper oxy.	7.5	0.5	3.75	33.3	124.9
total	**21.6**		**14.34**		**778.24**
Insecticides					
Gusathion	6	0.85	5.1	43.1	219.81
TCHE	4.8	0.5	2.4	32.3	77.52
Ultracide	2	0.4	0.8	69.3	55.44
Parathion	30	0.5	15	104.4	1566
total	**42.8**		**23.3**		**1918.77**
	64.4		**37.64**		**2697**

COUNTRY:	**UK**				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicides					
Dithianon	2.2	0.75	1.65	35.9	59.23
Captan	9.8	0.8	7.84	28.6	224.22
Pyrifenox	1.8	0.2	0.36	34.9	12.56
Bupirimate	4.4	0.25	1.1	41.2	45.32
total	**18.2**		**10.95**		**341.33**
Insecticides					
Lorsban	3	0.48	1.44	52.8	76.03
Pomex	8.2	0.5	4.1	22.6	92.66
total	**11.2**		**5.54**		**168.69**
Herbicides					
Dicamba +	5	0.215	1.08	38.7	41.6
Amitrole	5	0.225	1.13	20.5	23.06
Simazine	1.7	0.5	0.85	15.7	133.34
Diuron	1	0.5	0.5	20.5	10.25
total	**12.7**		**3.55**		**208.25**
	42.1		**20.04**		**718.27**

COUNTRY	SPAIN				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicides					
Captan	22.5	0.5	11.25	28.6	321.75
Folpet	4	0.8	3.2	22.2	71.04
Atemi	0.4	0.05	0.02	62.3	1.25
DNCO	30	0.05	1.5	41.2	61.8
Sulphur	35	0.8	28	45.5	1274
Copper oxy.	20	0.5	10	33.3	333
total	**111.9**		**53.97**		**2062.84**
Insecticides					
Gusathion	11.25	0.2	2.25	43.1	97
Dimetoato	4	0.4	1.6	74	118.4
Mecarban	2	0.5	1	32.8	32.8
total	**17.25**		**4.85**		**248.2**
Herbicides					
Roundup	2.1	0.36	0.76	32.4	24.5
Basta	2.55	0.15	0.38	32.4	12.4
Simazine	1.4	0.5	0.7	15.7	11
total	**6.05**		**1.84**		**47.9**
	135.2		**60.66**		**2358.94**

COUNTRY:	**NETHERLANDS**				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicides					
Copper 500wp	6	0.5	3	47.8	143.4
Benomyl 500wp	1	0.5	0.5	69.5	34.75
Captan 500sc	3	0.5	1.5	28.6	42.9
Bayleton 050wp	1	0.05	0.05	33.3	1.66
Nimrod 250wp	0.5	0.25	0.13	20	2.5
Baycor 250wp	1.5	0.25	0.38	55.9	20.85
Delan 750 sc	1.5	0.75	1.13	44	49.5
total	**14.5**		**6.68**		**295.56**
Insecticides					
Zolone 500sc	1.2	0.5	0.6	23.9	14.34
Dimilin 480sc	0.75	0.48	0.36	39.5	14.22
Ultracid 400wp	1	0.4	0.4	69.3	27.72
Dimethoate 400ec	2	0.4	0.8	74	59.2
Pirimor 500wg	0.5	0.5	0.25	30.5	7.62
Apollo 500sc	1	0.5	0.5	52.8	26.4
Insegar 250wp	0.3	0.25	0.08	66.9	5.02
total	**6.75**		**2.99**		**154.52**
Herbicides					
Glyphosate 360sl	5	0.36	1.8	32.4	58.32
Diuron 800wp	3	0.8	2.4	20.5	49.2
Simazine 500sc	3	0.5	1.5	15.7	23.55
Paraquat 200sl	4	0.2	0.8	70	56
total	**15**		**6.5**		**187.07**
Others					
Carbaryl 500wp	0.15	0.5	0.08	22.6	1.7
total	**0.15**		**0.08**		**1.7**
	36.4		**16.25**		**638.85**

COUNTRY:	**GREECE**				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicides					
Delan	5.04	0.75	3.78	34.9	131.92
Systhane 125	2	0.13	0.25	41.2	10.71
Ri midin	2	0.06	0.12	27.3	3.3
Melprex	3	0.68	2.03	34.9	71.2
Captan	15	0.5	7.5	28.6	214.5
Dithane	4	0.8	3.2	62.3	199.36
Baycor	2	0.25	0.5	23.6	11.8
Cu	10	0.5	5	47.8	239
total	**43.04**		**22.38**		**881.79**
Insecticides					
Zolone	2.8	0.35	0.98	23.9	16.73
Dimilin	0.5	0.25	0.13	39.5	4.94
Guzathion	1	0.4	0.4	43.1	17.24
Imidan	4	0.5	2	23.9	47.8
Apollo	0.5	0.5	0.25	52.8	13.2
Omite	4	0.3	1.2	42.7	51.24
Ultracide	4.2	0.4	1.68	69.3	116.42
total	**17**		**6.64**		**267.57**
Herbicides					
Round up	3	0.36	1.08	32.4	35
Basta	1	0.2	0.2	32.4	6.5
total	**4**		**1.28**		**41.5**
	64.04		**30.3**		**1190.86**

COUNTRY:	**BELG**				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicides					
Mancozeb	4	0.8	3.2	62.3	199.36
Baycor	1	0.25	0.25	23.6	5.9
Euparen	1	0.5	0.5	22.2	2.8
Bayleton	0.5	0.25	0.13	33.3	41.62
Ronilan	0.5	0.5	0.25	29.4	7.35
Captan	2	0.83	1.66	28.6	47.48
Cu	4	0.5	2	47.8	95.6
Polyram	3	0.8	2.4	55.9	134.16
Combi					
total	**16**		**10.39**		**534.27**
Insecticides					
DNOC	2.5	0.56	1.4	29.9	41.86
Decis 025	0.3	0.03	0.01	34.2	0.31
Dimilin	0.6	0.25	0.15	39.5	5.92
Insegar 250	0.3	0.25	0.08	66.9	5.02
Apollo	0.4	0.5	0.2	52.8	10.56
Nissorum 100	0.04	0.1	0	12.8	0.05
total	**4.14**		**1.84**		**63.72**
Herbicides					
Basta	5	0.2	1	32.4	32.4
Roundup 360	2.8	0.36	1.01	32.4	32.66
Simazine 500	2	0.5	1	15.7	15.7
total	**9.8**		**3.01**		**80.76**
	29.94		**15.24**		**678.75**

COUNTRY:	GERMANY				
PRODUCT	kg/ha	%AI	kgAI/ha	EIQ	EIQ FUR
Fungicides					
Delan	5	0.75	3.75	34.9	130.9
Bayfidan	0.5	0.05	0.03	33.3	0.83
Rubigan sc	0.6	0.12	0.07	27.3	1.96
Benocap	0.25	0.2	0.05	32.9	1.64
Omnex	0.5	0.63	0.31	62.3	19.53
Dithane	2	0.8	1.6	62.3	99.7
Euparen	3	0.5	1.5	35.9	53.85
Benomyl	0.3	0.5	0.15	69.5	10.42
total	**12.15**		**7.46**		**318.83**
Insecticides					
Rubitox	1	0.35	0.35	23.9	8.36
Insegar	0.4	0.25	0.1	66.9	6.7
Metasystox	0.25	0.3	0.08	82.5	6.2
Pirimor	0.5	0.5	0.25	30.5	7.62
Apollo	0.08	0.5	0.04	52.8	2.11
Torque	0.4	0.5	0.2	49.6	9.92
Dimilin	0.5	0.25	0.13	39.5	4.94
total	**3.13**		**1.14**		**45.85**
Herbicides					
Round up	1	0.36	0.36	32.4	11.66
Basta	1.5	0.2	0.3	32.4	9.72
total	**2.5**		**0.66**		**21.38**
Others					
Carbaryl 500wp	0.15	0.5	0.08	22.6	0.25
total	**0.15**		**0.08**		**0.25**
	17.93		**9.34**		**386.31**

Appendix 2. Production characteristics of existing (country number 1) and new (countries 2 onwards) varieties.

	YIELD	NITROGEN	PHOSPHOROUS	POTASSIUM	FUNGICIDE	INSECTICIDE	HERBICIDE	OTHER	HARVESTING	PRUNING	OTHERS	PRICE
COUNTRY/NUTS												
GREECE 1	26.1	274	96	151	8.6	6	0.74	0.62	356	193	441	0
COUNTRY/NUTS												
GREECE 2	25.6	471	165	516	29.63	10.6	0	0	880	316	816	0
COUNTRY/NUTS												
GREECE 3	24.5	452	0	0	9.1	7.2	0.25	0.1	438	116	263	0
COUNTRY/NUTS												
BELG/LUX 1	33.1	90	20	170	5.15	1.8	3	2	350	170	200	0
COUNTRY/NUTS												
BELGIUM 2	33.06	90	20	170	5.15	0.92	1	0	350	170	200	10
COUNTRY/NUTS												
BELGIUM 3	33.06	81	20	153	5.15	0.9	1	0	350	170	200	10
COUNTRY/NUTS												
ITALY 1	10.92	70	30	30	44.99	16.54	0	1.1	333	130	168	0
COUNTRY/NUTS												
ITALY 2	25	70	30	30	33.74	16.54	0	1.13	265	153	83	0
COUNTRY/NUTS												
ITALY 3	30	63	27	27	22.5	12.4	0	0.85	333	170	79	5
COUNTRY/NUTS												

ITALY 4	36.7	56	24	24	11.25	10.75	0	0.73	333	170	79	5
COUNTRY/NUTS												
FRANCE 1	31.1	50	25	60	98.53	7.82	3.78	0.15	371.5	200	244	0
COUNTRY/NUTS												
FRANCE 2	31.7	50	25	60	18.46	8.52	2.47	0.15	371.5	200	244	2
COUNTRY/NUTS												
FRANCE 3	35.21	47.5	23.75	57	24.63	7.04	3.4	0.13	371.5	200	232	5
COUNTRY/NUTS												
FRANCE 4	35.9	45	22.5	54	24.63	6.25	3.02	0.12	379	200	232	5
COUNTRY/NUTS												
FRANCE 5	32.6	42.5	21.25	51	49.3	5.9	2.8	0.11	371.5	200	232	0
COUNTRY/NUTS												
SPAIN 1	27.8	75	40	125	53.97	4.85	1.84	0.05	400	120	120	0
COUNTRY/NUTS												
SPAIN 2	32.1	90	55	140	27.86	4.93	3.51	0.02	250	130	80	5
COUNTRY/NUTS												
SPAIN 3	20.74	81	45	126	31.2	4.4	2.4	0.031	325	125	95	5
COUNTRY/NUTS												
UK/EIRE COX	35.6	70	30	30	44.99	16.54	0	1.13	333	170	168	0
COUNTRY/NUTS												
UK 2	32.83	70	30	30	33.74	16.54	0	1.13	333	153	81.3	5

COUNTRY/NUTS												
UK 3	30.26	56	24	24	22.5	14.9	0	1.02	333	153	79	5
COUNTRY/NUTS												
UK 4	35.6	52.5	22.5	22.5	11.25	13.23	0	0.9	333	153	79	5
COUNTRY/NUTS												
PORTUGAL 1	7.9	60	60	60	14.34	5.9	0	0	450	210	583	0
COUNTRY/NUTS												
PORTUGAL 2	8.3	100	120	120	7.17	5	0.6	0	430	250	300	0
COUNTRY/NUTS												
PORTUGAL 3	10.5	114	115	115	3.6	1.75	0.6	0	424	312	268	5
COUNTRY/NUTS												
NL	36	90	20	100	3.34	2.99	6.5	0.09	226	165	179.2	5
COUNTRY/NUTS												
GERMANY	37.1	40	7	33	7.46	1.14	0.66	0.1	238	100	176	0
COUNTRY/NUTS												
GERMANY 2	42.6	40	7	33	3.73	1.14	0.66	0.1	266	120	193	0
COUNTRY/NUTS												
GERMANY 3	42.6	40	7	33	3.73	0.85	0.66	0.1	266	120	174	5
COUNTRY/NUTS												
GERMANY 4	42.6	36	6.3	29.7	1.86	0.85	0.66	0.1	266	120	173.7	5

COUNTRY/NUTS												
GERMANY 5	42.6	34	5.9	28	0	0.57	0.66	0.1	266	120	173.7	5

COUNTRY/NUTS												
DENMARK	37.1	40	7	33	7.46	1.14	0.66	0.1	238	100	176	0

Appendix 2 continued. EIQ values for existing and new varieties for selected apple growing regions of Europe.

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
GREECE 1								
Fungicide	883.04	335.5	294.7	112	1468	557.8	881.79	335.1
Insecticide	212.1	169.7	34.3	27.4	576.2	461	267.57	372.7
Herbicide	20.5	12.3	9	5.4	95.1	57.1	41.5	25
TOTALS	1115.6	517.5	338	144.8	2139	1075.9	**1190.86**	**732.8**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
GREECE 2								
Fungicide	883.04	883.04	294.7	294.7	1468	1467.9	881.79	881.79
Insecticide	212.1	212.1	34.3	34.3	576.2	576.22	267.57	267.57
Herbicide	20.5	20.5	9	9	95.1	95.1	41.5	41.5
TOTALS	1115.6	1115.64	338	338	2139	2139.22	**1190.86**	**1190.9**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
GREECE 3								

Fungicide	883.04	353.2	294.7	118	1468	587.2	881.79	352.8
Insecticide	212.1	212.1	34.3	34.3	576.2	576.22	267.57	267.57
Herbicide	20.5	4.1	9	1.8	95.1	19	41.5	25
TOTALS	1115.6	569.4	338	154.1	2139	1182.42	**1190.86**	**645.37**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
BELG/LUX 1								
Fungicide	475.6	237.8	143.9	71.95	896.4	448.2	534.27	351.5
Insecticide	14.4	14.4	10.6	10.6	115.5	115.5	63.72	63.72
Herbicide	44	44	23.05	23.05	175.4	175.4	80.76	80.76
TOTALS	534	296.2	177.6	105.6	1187	739.1	**678.75**	**495.98**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
BELGIUM 2								
Fungicide	475.6	237.8	143.9	71.95	896.4	448.2	534.27	351.5
Insecticide	14.4	14.4	10.6	10.6	115.5	115.5	63.72	63.72
Herbicide	44	44	23.05	23.05	175.4	175.4	80.76	80.76
TOTALS	534	296.2	177.6	105.6	1187	739.1	**678.75**	**495.98**

COUNTRY/NUTS		FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
BELGIUM									
3									
	Fungicide	475.6	237.8	143.9	71.95	896.4	448.2	534.27	351.5
	Insecticide	14.4	7.2	10.6	5.3	115.5	57.75	63.72	32.42
	Herbicide	44	22	23.05	11.52	175.4	87.7	80.76	40.41
	TOTALS	534	267	177.6	88.77	1187	593.65	**678.75**	**424.33**

COUNTRY/NUTS		FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
ITALY 1									
	Fungicide	1609.8	1609.8	692.3	692.33	5534	5534	2610.72	2610.7
	Insecticide	703.3	703.3	111.2	111.15	1610	1609.7	808.05	808.05
	Herbicide	0	0	0	0	0	0	0	0
	TOTALS	2313.1	2313.1	803.5	803.48	7144	7143.7	**3418.77**	**3418.8**

COUNTRY/NUTS		FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
ITALY 2									
	Fungicide	1609.8	1207.35	692.3	519.24	5534	4150.5	2610.72	1959

Insecticide	703.3	703.3	111.2	111.15	1610	1609.7	808.05	808.05
Herbicide	0	0	0	0	0	0	0	0
TOTALS	2313.1	1910.65	803.5	630.39	7144	5760.2	**3418.77**	**2767.1**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
ITALY 3								
Fungicide	1609.8	804.9	692.3	346.2	5534	2767	2610.72	1306
Insecticide	703.3	527.47	111.2	86.4	1610	1207.3	808.05	607.1
Herbicide	0	0	0	0	0	0	0	0
TOTALS	2313.1		803.5		7144		**3418.77**	**1913.1**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
ITALY 4								
Fungicide	1609.8	402.45	692.3	173.1	5534	1383.5	2610.72	653.01
Insecticide	703.3	457.1	111.2	72.2	1610	1046.3	808.05	525.2
Herbicide	0	0	0	0	0	0	0	0
TOTALS	2313.1	859.55	803.5	245.3	7144	2429.8	**3418.77**	**1178.2**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ

FRANCE 1

Fungicide	4178	4178	888	888	9590	9590.47	4971.15	4971.2
Insecticide	358.4	358.4	31	31	621.9	621.85	450.97	450.97
Herbicide	133.4	133.4	40.6	40.6	187.1	187.1	120.52	120.52
TOTALS	4669.8	4669.8	959.6	959.6	10399	10399.4	**5452.64**	**5452.6**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
FRANCE 2								
Fungicide	4178	2089	888	444	9590	4795.2	4971.15	2442.7
Insecticide	358.4	358.4	31	31	621.9	621.85	450.97	450.97
Herbicide	133.4	133.4	40.6	40.6	187.1	187.1	120.52	120.52
TOTALS	4669.8	2580.8	959.6	515.6	10399	5604.15	**5542.64**	**3014.2**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
FRANCE 3								
Fungicide	4178	1044.5	888	222	9590	2397.7	4971.15	1221.4
Insecticide	358.4	322.56	31	27.9	621.9	559.7	450.97	303.4
Herbicide	133.4	120.1	40.6	36.5	187.1	168.4	120.52	108.3

TOTALS	4669.8	1487.16	959.6	286.4	10399	3125.8	**5542.64**	**1633.1**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
FRANCE 4								
Fungicide	4178	1044.5	888	222	9590	2397.7	4971.15	1221.4
Insecticide	358.4	286.72	31	24.8	621.9	497.5	450.97	269.7
Herbicide	133.4	106.7	40.6	32.5	187.1	149.7	120.52	96.3
TOTALS	4669.8	1437.92	959.6	279.3	10399	3044.9	**5542.64**	**1587.4**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
FRANCE 5								
Fungicide	4178	2089	888	444	9590	4795.23	4971.15	2442.7
Insecticide	358.4	268.8	31	23.25	621.9	466.4	450.97	758.45
Herbicide	133.4	100	40.6	30.4	187.1	140.3	120.52	90.2
TOTALS	4669.8	2457.8	959.6	497.65	10399	5401.93	**5542.64**	**3291.4**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
SPAIN 1								
Fungicide	780.15	780.15	366.4	366.35	5030	5030.4	2062.84	2062.8

Insecticide	202.2	202.2	36.6	36.6	505.6	505.6	248.2	248.2
Herbicide	26.6	26.6	14.3	14.3	102.9	102.9	47.9	47.9
TOTALS	1009	1008.95	417.3	417.25	5639	5638.9	**2358.94**	**2358.9**

COUNTRY/NUTS		FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
SPAIN 2									
	Fungicide	780.15	741.1	366.4	348	5030	4778.9	2062.84	1956
	Insecticide	202.2	202.2	36.6	36.6	505.6	505.6	248.2	248.2
	Herbicide	26.6	26.6	14.3	14.3	102.9	102.9	47.9	47.9
	TOTALS	1009	969.9	417.3	398.9	5639	5387.4	**2358.94**	**2252.1**

COUNTRY/NUTS		FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
SPAIN 3									
	Fungicide	780.15	585.1	366.4	274.8	5030	3772.8	2062.84	1544.2
	Insecticide	202.2	182	36.6	32.9	505.6	455	248.2	223.3
	Herbicide	26.6	23.9	14.3	12.9	102.9	92.6	47.9	43.1
	TOTALS	1009	791	417.3	320.6	5639	4320.4	**2358.94**	**1810.6**

COUNTRY/NUTS		FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ

UK/EIRE COX

Fungicide	293.6	293.6	108.2	108.2	622.5	622.5	341.33	341.33
Insecticide	114	114	24.54	24.54	367.2	367.2	168.9	168.9
Herbicide	67.9	67.9	41.75	41.75	185.9	185.9	208.25	208.25
TOTALS	475.5	475.5	174.5	174.49	1176	1175.6	**718.48**	**718.48**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
UK 2								
Fungicide	293.6	220.2	108.2	81.1	622.5	466.9	341.33	256.1
Insecticide	114	114	24.54	24.54	367.2	367.2	168.9	168.9
Herbicide	67.9	67.9	41.75	41.75	185.9	185.9	208.25	208.25
TOTALS	475.5	402.1	174.5	147.39	1176	1020	**718.48**	**633.25**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
UK 3								
Fungicide	293.6	146.8	108.2	54.1	622.5	311.25	341.33	170.7
Insecticide	114	102.6	24.54	22.1	367.2	330.5	168.9	151.7
Herbicide	67.9	61.1	41.75	37.6	185.9	167.31	208.25	88.7

TOTALS	475.5	310.5	174.5	113.8	1176	809.06	**718.48**	**411.1**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
UK 4								
Fungicide	293.6	73.4	108.2	27.05	622.5	155.6	341.33	85.35
Insecticide	114	91.2	24.54	19.6	367.2	293.8	168.9	134.9
Herbicide	67.9	54.3	41.75	33.4	185.9	148.7	208.25	94.35
TOTALS	475.5	218.9	174.5	80.05	1176	598.1	**718.48**	**314.6**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
PORTUGAL 1								
Fungicide	460	460	209.2	209.2	1667	1666.77	778.24	778.24
Insecticide	2403.6	2403.6	161.5	161.5	3189	3189.4	1918.77	1918.8
Herbicide	0	0	0	0	0	0	0	0
TOTALS	2863.6	2863.6	370.7	370.7	4856	4856.17	**2697.01**	**2697**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
PORTUGAL 2								
Fungicide	460	230	209.2	104.6	1667	833.4	778.24	389.3

Insecticide	2403.6	2163.2	161.5	145.3	3189	2870.5	1918.77	1726.3
Herbicide	0	0	0	0	0	0	0	0
TOTALS	2863.6	2393.2	370.7	249.9	4856	3703.9	**2697.01**	**2115.6**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
PORTUGAL 3								
Fungicide	460	115	209.2	52.3	1667	416.7	778.24	194.7
Insecticide	2403.6	2163.2	161.5	145.3	3189	2870.5	1918.77	1726.3
Herbicide	0	0	0	0	0	0	0	0
TOTALS	2863.6	2278.2	370.7	197.6	4856	3287.2	**2697.01**	**1921**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
NL								
Fungicide	363.1	181.55	112.6	56.3	400.9	200.45	295.56	146.1
Insecticide	129.1	129.1	60.77	60.77	316.8	316.8	168.89	168.89
Herbicide	90.4	90.4	61.7	61.7	359.4	359.4	187.07	187.07
TOTALS	582.6	401.05	235.1	178.77	1077	876.65	**651.52**	**502.06**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ

GERMANY

Fungicide	195.6	195.6	114.5	114.5	646.4	646.4	318.83	318.83
Insecticide	29.3	29.3	8	8	99.5	99.5	45.85	45.85
Herbicide	10.6	10.6	2.6	2.6	49.1	49.1	21.38	21.38
TOTALS	235.5	235.5	125.1	125.1	795	795	**386.06**	**386.06**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
GERMANY 2								
Fungicide	195.6	97.8	114.5	57.25	646.4	323.2	318.83	159.4
Insecticide	29.3	29.3	8	8	99.5	99.5	45.85	45.85
Herbicide	10.6	10.6	2.6	2.6	49.1	49.1	21.38	21.38
TOTALS	235.5	137.7	125.1	67.85	795	471.8	**386.06**	**226.63**

COUNTRY/NUTS	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
GERMANY 3								
Fungicide	195.6	97.8	114.5	57.25	646.4	323.2	318.83	159.4
Insecticide	29.3	22	8	6	99.5	74.6	45.85	34.2
Herbicide	10.6	10.6	2.6	2.6	49.1	49.1	21.38	21.38

TOTALS	235.5	130.4	125.1	65.85	795	446.9	**386.06**	**214.98**

COUNTRY/NUTS GERMANY 4	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
Fungicide	195.6	48.9	114.5	28.6	646.4	161.6	318.83	79.7
Insecticide	29.3	22	8	6	99.5	74.6	45.85	34.2
Herbicide	10.6	10.6	2.6	2.6	49.1	49.1	21.38	21.38
TOTALS	235.5	81.5	125.1	37.2	795	285.3	**386.06**	**135.28**

COUNTRY/NUTS GERMANY 5	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
Fungicide	195.6	0	114.5	0	646.4	0	318.83	0
Insecticide	29.3	14.65	8	4	99.5	49.75	45.85	22.8
Herbicide	10.6	10.6	2.6	2.6	49.1	49.1	21.38	21.38
TOTALS	235.5	25.25	125.1	6.6	795	98.85	**386.06**	**44.18**

COUNTRY/NUTS DENMARK	FWFUR	NVFWFUR	CFUR	NVCFUR	EnFUR	EnNVFUR	TOTAL EIQ	NVEIQ
Fungicide	195.6	195.6	114.5	114.5	646.4	646.4	318.83	318.83

Insecticide	29.3	29.3	8	8	99.5	99.5	45.85	45.85
Herbicide	10.6	10.6	2.6	2.6	49.1	49.1	21.38	21.38
TOTALS	235.5	235.5	125.1	125.1	795	795	**386.06**	**386.06**

FWFUR = Fieldworker Field Use Rating, CFUR = Consumer Field Use Rating, and EnFUR = Environment Field Use Rating. The prefix NV represents the new variety field use rating for each of the components.

APPENDIX 3. EIQ tables of Kovach et al., 1992.

Fungicides

Common Name	EIQ
anilizaine	26.68
benomyl	69.5
captan	28.62
carboxin	19.95
chlorothalonil	46
copper hydroxide	33.3
copper sulfate	47.78
copper sulfate+lime	67.67
dichloran	35.95
dinocap	23.63
dodine	34.85
fenamiphos	78.86
fenarimol	27.33
fentin hydroxide	32.67
ferbam	28.83
flusilazol	32.9
folpet	22.21
fosetyl-Al	13.67
iprodione	26.63
mancozeb	62.33
maneb	64.08
maneb +dinocap	46.51
metalaxyl	29.17
metiram	55.92
myclobutanil	41.21
PCNB	21.84
streptomycin	18.71
sulfur	45.53
thiophanate methyl	51.5
thiram	54.52
triadmefon	33.32
triforine	41.21
vinclozolin	29.38
zineb	43.95

Herbicides

Common Name	EIQ
2,4-D (acid)	56.3
acifluorfen	52
alachlor	21.3
ammonium sulfamate	38.3
atrazine	33.2
bentazon	38.7
bromacil	25.7
chloramben	15.7
cyanazine	19.8
cycloate	15.3
dalapon	37.5
DCPA	34
dichlobenil	18
diethatyl-ethyl	14.7
diuron	20.5
EPTC	13.3
ethalfluralin	30.7
fluazifop-butyl	44
glyphosate	32.4
imazapyr	18.7
linuron	40.3
MCPA	36.7
metolachlor	18
metribuzin	35.3
napropamide	18
nicosulfuron	29.9
norflurazon	18.8
oryzalin	17.7
oxyfluorfen	46.8
paraquat	70
pendimethalin	25.8
phenmediphan	30.2
picloram	31.8
pronamide	36
propazine	38.7
pryazon	16
sethoxydim	27.5
simizine	15.7
terbacil	16.8
triclopyr	31.2
trifluralin	26.8
	29.5

Insecticides

Common Name	EIQ
acephate	17.9
aldicarb	37.13
azinphos-methyl	43.08
Bacillus thuringiensis	13.48
carbaryl	22.57
carbofuran	56.78
chlorpyrifos	52.78
cryolite	21.44
diazinon	34.17
dichlorvos	40.58
dicofol	29.85
diflubenzuron	39.5
dimethoate	73.97
disulfoton	121.92
endosulfan	40.52
esfenvalerate	49.58
ethion	41.04
ethoprop	44.58
fensulfothion	66.85
fenvalerate	49.58
fonofos	44.58
formetanate	21.45
hexakis	12.8
malathion	23.15
methamidophos	64.08
methidathion	69.27
methomyl	32.83
methoxychlor	58
methyl parathion	35.22
mevinphos	28.17
naled	37.67
oil	27.53
oxamyl	22.9
oxydemeton-methyl	82.53
oxythioquinox	44.35
parathion	104.37
permethrin	56.43
phorate	68.2
phosmet	23.9
phosphamidon	26.3
piperonyl butoxide	20.79
pirimicarb	30.5
propargite	42.72
propoxur	87.27
pyrethrin	17.98
rotenone	33
ryania	55.3
sabadilla	35.63
soap	19.45
terbufos	32.25

APPENDIX 4. Modified EIQ Tables.

HERBICIDES

Common Name	Trade Name	EIQ Eco
CATEGORY 1:	**DANGER**	
oxyfluorfen	Goal	194
paraquat	Gramaxone	155
pendimethalin	Prowl	145
ethalfluralin	Sonolan	144
fluazifop-butyl	Fusilade	131
atrazine	Atrazine	123
glyphosate	Roundup	122.3
linuron	Lorox	108
phenmediphan	Spin-aid	100.1
CATEGORY 2:	**WARNING**	
diuron	Karmex	87
bentazon	Basagran 4S	85
DCPA	Dacthal	83
pronamide	Kerb	82
metribuzin	Sencor	78
2,4-D (acid)	Weedone	**74**
nicosulfuron	Accent	74
sethoxydim	Poast	73.9
propazine	Milogard	73
acifluorfen	Blazer	**72**
ammonium sulfamate	Ammate	71
MCPA	Bronate	71
alachlor	Lasso	70.5
norflurazon	Solicam	65
cycloate	Ro-Neet	64
diethatyl-ethyl	Antor	64
metolachlor	Dual	64
oryzalin	Surflan	63.5
dalapon	Dalapon	62.5
bromacil	Hyvar	52
CATEGORY 3:	**CAUTION**	
pryazon	Pyramin	38
chloramben	Amiben	35
napropamide	Devrinol	35
dichlobenil	Casoron	34
EPTC	Eptam	34
trifluralin	Treflan	33.5
cyanazine	Bladex	32
simizine	Princep	31.2
terbacil	Sinbar	30.5

Common Name	Trade Name	EIQ Con
CATEGORY 1:	**DANGER**	
propazine	Milogard	37
bentazon	Basagran 4S	31
bromacil	Hyvar	31
terbacil	Sinbar	31
atrazine	Atrazine	29.5
norflurazon	Solicam	29.5
trifluralin	Treflan	29.5
napropamide	Devrinol	29.26
simizine	Princep	29
ammonium sulfamate	Ammate	28
dalapon	Dalapon	28
metribuzin	Sencor	28
nicosulfuron	Accent	28
dichlobenil	Casoron	27
pryazon	Pyramin	27
CATEGORY 2:	**WARNING**	
acifluorfen	Blazer	**24**
diuron	Karmex	22.5
2,4-D (acid)	Weedone	**21**
linuron	Lorox	21
chloramben	Amiben	20
cyanazine	Bladex	19.3
metolachlor	Dual	19
cycloate	Ro-Neet	17
EPTC	Eptam	17
paraquat	Gramaxone	17
alachlor	Lasso	16.5
sethoxydim	Poast	16.5
CATEGORY 3:	**CAUTION**	
ethalfluralin	Sonolan	15
fluazifop-butyl	Fusilade	15
pronamide	Kerb	14
DCPA	Dacthal	13
MCPA	Bronate	13
oxyfluorfen	Goal	12.5
pendimethalin	Prowl	12.5
glyphosate	Roundup	11
phenmediphan	Spin-aid	9.5
diethatyl-ethyl	Antor	7
oryzalin	Surflan	5

Common Name	Trade Name	EIQ FW
CATEGORY 1:	**DANGER**	
2,4-D (acid)	Weedone	**36**

acifluorfen	Blazer	**36**
paraquat	Gramaxone	36
fluazifop-butyl	Fusilade	20
CATEGORY 2:	**WARNING**	
dalapon	Dalapon	18
pronamide	Kerb	18
MCPA	Bronate	16
ammonium sulfamate	Ammate	12
bentazon	Basagran 4S	12
propazine	Milogard	12
ethalfluralin	Sonolan	10
oxyfluorfen	Goal	10
CATEGORY 3:	**CAUTION**	
cyanazine	Bladex	8.7
DCPA	Dacthal	8
glyphosate	Roundup	8
linuron	Lorox	8
alachlor	Lasso	6
atrazine	Atrazine	6
bromacil	Hyvar	6
dichlobenil	Casoron	6
nicosulfuron	Accent	6
phenmediphan	Spin-aid	6
trifluralin	Treflan	6
chloramben	Amiben	5
diuron	Karmex	5
pendimethalin	Prowl	5
napropamide	Devrinol	4.26
metolachlor	Dual	4
oryzalin	Surflan	4
simizine	Princep	4
terbacil	Sinbar	4
metribuzin	Sencor	3
norflurazon	Solicam	3
sethoxydim	Poast	3
cycloate	Ro-Neet	0
diethatyl-ethyl	Antor	0
EPTC	Eptam	0
pryazon	Pyramin	0

INSECTICIDES

Common Name	Trade Name	EIQ Farmworker
disulfoton	Di-Syston	150
parathion	Niran, Phoskil	140
oxydemeton-methyl	Metasytox-R	96
carbofuran	Furadan	72
dimethoate	Cygon	72
propoxur	Baygon	72

Common Name	Trade Name	EIQ Ecological
disulfoton	Di-Syston	187.8
propoxur	Baygon	176.8
parathion	Niran, Phoskil	165.1
phorate	Thimet	154.6
fensulfothion	Dasanit	146.6
methamidophos	Monitor	141.3

dichlorvos	Vapona	60
methidathion	Supracide	60
ethoprop	Mocap	57.5
methyl parathion	Penncap-M	54
naled	Dibrom	54
rotenone	Chem Fish	54
ryania	Ryania	45.6
aldicarb	Temik	45
chlorpyrifos	Lorsban	45
fonofos	Dyfonate	45
fensulfothion	Dasanit	40
methamidophos	Monitor	40
phorate	Thimet	40
propargite	Omite	40
sabadilla	Red Devil	39.3
azinphos-methyl	Guthion	36
dicofol	Kelthane	36
endosulfan	Thiodan	36
ethion	Ethion	34.5
pirimicarb	Pirimor	34.2
mevinphos	Phosdrin	30
piperonyl butoxide	Butacide	30
terbufos	Counter	30
methoxychlor	Marlate	25
malathion	Cythion	21
permethrin	Ambush	20
phosphamidon	Swat	18
oxythioquinox	Morestan	16
diazinon	Diazinon	15
diflubenzuron	Dimilin	15
oxamyl	Vydate	15
cryolite	Kryocide	13.1
Bacillus thuringiensis	Dipel	12
carbaryl	Sevin	12
phosmet	Imidan	12
soap	M-Pede	11.4
esfenvalerate	Asana	8
fenvalerate	Pydrin	8
oil	Oil	8
acephate	Orthene	6
formetanate	Carzol	6
hexakis	Vendex	6
methomyl	Lannate	6
pyrethrin	Pyronone	6

Common Name	**Trade Name**	**EIQ Consumer**
carbofuran	Furadan	29
oxydemeton-methyl	Metasytox-R	29
disulfoton	Di-Syston	28
aldicarb	Temik	14

dimethoate	Cygon	140.9
permethrin	Ambush	140.8
methidathion	Supracide	139.8
esfenvalerate	Asana	136.8
fenvalerate	Pydrin	136.8
methoxychlor	Marlate	135.5
oxydemeton-methyl	Metasytox-R	122.6
ryania	Ryania	113.3
oxythioquinox	Morestan	110.1
chlorpyrifos	Lorsban	104.9
diflubenzuron	Dimilin	98
azinphos-methyl	Guthion	88.3
ethion	Ethion	86.2
fonofos	Dyfonate	82.8
propargite	Omite	82.2
methomyl	Lannate	81.5
diazinon	Diazinon	79.5
endosulfan	Thiodan	78.6
oil	Oil	71
carbofuran	Furadan	69.4
ethoprop	Mocap	67.2
terbufos	Counter	62.8
sabadilla	Red Devil	61.6
dichlorvos	Vapona	58.8
phosmet	Imidan	56.7
naled	Dibrom	55
formetanate	Carzol	54.4
phosphamidon	Swat	52.9
carbaryl	Sevin	52.7
aldicarb	Temik	52.4
dicofol	Kelthane	48.6
mevinphos	Phosdrin	48.5
methyl parathion	Penncap-M	47.7
pirimicarb	Pirimor	45.9
cryolite	Kryocide	45.2
oxamyl	Vydate	45.2
pyrethrin	Pyronone	45
malathion	Cythion	44
acephate	Orthene	43.7
soap	M-Pede	41.8
rotenone	Chem Fish	41
hexakis	Vendex	28.8
piperonyl butoxide	Butacide	28.7
Bacillus thuringiensis	Dipel	22.5

fensulfothion	Dasanit	14
methoxychlor	Marlate	13.5
propoxur	Baygon	13
pirimicarb	Pirimor	11.4
methamidophos	Monitor	11
methomyl	Lannate	11
phorate	Thimet	10
ethoprop	Mocap	9.1
dimethoate	Cygon	9
chlorpyrifos	Lorsban	8.5
oxamyl	Vydate	8.5
permethrin	Ambush	8.5
diazinon	Diazinon	8
methidathion	Supracide	8
parathion	Niran, Phoskil	8
phosphamidon	Swat	8
endosulfan	Thiodan	7
oxythioquinox	Morestan	7
ryania	Ryania	7
Bacillus thuringiensis	Dipel	6
cryolite	Kryocide	6
fonofos	Dyfonate	6
mevinphos	Phosdrin	6
propargite	Omite	6
sabadilla	Red Devil	6
diflubenzuron	Dimilin	5.5
soap	M-Pede	5.1
azinphos-methyl	Guthion	5
dicofol	Kelthane	5
malathion	Cythion	4.5
acephate	Orthene	4
esfenvalerate	Asana	4
fenvalerate	Pydrin	4
formetanate	Carzol	4
methyl parathion	Penncap-M	4
naled	Dibrom	4
rotenone	Chem Fish	4
terbufos	Counter	4
hexakis	Vendex	3.7
oil	Oil	3.7
piperonyl butoxide	Butacide	3.7
carbaryl	Sevin	3
dichlorvos	Vapona	3
phosmet	Imidan	3
pyrethrin	Pyronone	3
ethion	Ethion	2.5

FUNGIICDES

Common Name	**Trade Name**	**EIQ Farmworker**	**Common Name**	**Trade Name**	**EIQ Consumer**
CATEGORY 1:	**DANGER**		**CATEGORY 1:**	**DANGER**	

copper sulfate+lime	Bordeaux	54
thiram	Thiram	36.9
metiram	Polyram	30
copper sulfate	copper	27
mancozeb	Manzate	24
maneb	maneb	24
zineb	Dithane Z	20
CATEGORY 2:	**WARNING**	
myclobutanil	Nova	18.45
benomyl	Benlate	18
thiophanate methyl	Topsin-M	18
maneb +dinocap	Dikar	16.4
chlorothalonil	Bravo	15
captan	Orthocide	14
triadmefon	Bayleton	14
dichloran	Botran	12.3
triforine	Funginex	12.3
vinclozolin	Ronilan	12.3
fentin hydroxide	Du-Ter	12
dodine	Syllit	10.25
CATEGORY 3:	**CAUTION**	
anilizaine	Dyrene	8.2
dinocap	Karathane	7.3
copper hydroxide	Kocide	6.15
streptomycin	Agristrep	6
PCNB	Terraclor	5
sulfur	Sulfur	5
fenarimol	Rubigan	4
fosetyl-Al	Aliette	4
folpet	Phaltan	3.1
iprodione	Rovral	3.1
carboxin	Vitavax	3
ferbam	Carbamate	3
flusilazol	Nustar	3
metalaxyl	Ridomil	3

benomyl	Benlate	70
fenarimol	Rugigan	43
CATEGORY 2:	**WARNING**	
zineb	Ditane Z	35
triforine	Funginex	32.3
thiophanate methyl	Topsin M	32
metalaxyl	Ridomil	31
cooper sulphate+lime	Bordeaux	23
triadmefon	Bayleton	22
mancozeb	Manzate	21
maneb	Maneb	21
dodine	Syllit	20.4
myclobutanil	Nova	20.15
metiram	Polyram	20
CATEGORY 3:	**CAUTION**	
copper sulphate	Copper	18.5
maneb+dinocap	Dikar	17.2
clorothalonil	Bravo	15
vinclozolin	Ronilan	13.1
flusilazol	Nustar	12.95
PCNB	Terraclor	12.5
folpet	Phaltan	12.05
captan	Orthocide	12
dichloran	Botran	11.15
thiram	Thiram	11.15
dinocap	Karathane	10.5
sulfur	Sulfur	10
streptomycin	Agristrep	9.5
anilizaine	Dyrene	9.1
copper hydroxide	Kocide	9.05
fentin hydroxide	Du Ter	9
ferbam	Carbamate	9
fosetyl-Al	Aliette	8
carboxin	Vitavax	7.25
iprodione	Rovral	7.05

Common Name	**Trade Name**	**EIQ Ecological**
CATEGORY 1:	**DANGER**	
mancozeb	Manzate	233
maneb	maneb	217.3
benomyl	Benlate	208.5
chlorothalonil	Bravo	184
copper sulfate+lime	Bordeaux	162
maneb +dinocap	Dikar	161.6
copper sulfate	copper	138.9
fenarimol	Rubigan	138
fentin hydroxide	Du-Ter	137
flusilazol	Nustar	131.2
thiram	Thiram	125
triforine	Funginex	121.4
myclobutanil	Nova	121.4

iprodione	Rovral	118.4
sulfur	Sulfur	117
thiophanate methyl	Topsin-M	116.5
dodine	Syllit	115.6
vinclozolin	Ronilan	114.4
copper hydroxide	Kocide	109.5
dichloran	Botran	108.4
zineb	Dithane Z	105.1
CATEGORY 2:	WARNING	
dinocap	Karathane	97.1
metiram	Polyram	93.8
PCNB	Terraclor	93
carboxin	Vitavax	89.9
streptomycin	Agristrep	89.15
folpet	Phaltan	87.7
triadmefon	Bayleton	86
ferbam	Carbamate	75.9
anilizaine	Dyrene	68.4
metalaxyl	Ridomil	62.5
captan	Orthocide	59.9
CATEGORY 3:	CAUTION	
fosetyl-Al	Aliette	19.5

HERBICIDES

Common Name	Trade Name	EIQ Eco
CATEGORY 1:	**DANGER**	
oxyfluorfen	Goal	194
paraquat	Gramaxone	155
pendimethalin	Prowl	145
ethalfluralin	Sonolan	144
fluazifop-butyl	Fusilade	131
atrazine	Atrazine	123
glyphosate	Roundup	122.3
linuron	Lorox	108
phenmediphan	Spin-aid	100.1
CATEGORY 2:	**WARNING**	
diuron	Karmex	87
bentazon	Basagran 4S	85
DCPA	Dacthal	83
pronamide	Kerb	82
metribuzin	Sencor	78
2,4-D (acid)	Weedone	**74**
nicosulfuron	Accent	74
sethoxydim	Poast	73.9
propazine	Milogard	73
acifluorfen	Blazer	**72**
ammonium sulfamate	Ammate	71
MCPA	Bronate	71
alachlor	Lasso	70.5
norflurazon	Solicam	65
cycloate	Ro-Neet	64
diethatyl-ethyl	Antor	64
metolachlor	Dual	64
oryzalin	Surflan	63.5
dalapon	Dalapon	62.5
bromacil	Hyvar	52
CATGORY 3:	**CAUTION**	
pryazon	Pyramin	38
chloramben	Amiben	35
napropamide	Devrinol	35
dichlobenil	Casoron	34
EPTC	Eptam	34
trifluralin	Treflan	33.5
cyanazine	Bladex	32
simizine	Princep	31.2
terbacil	Sinbar	30.5

Common Name	Trade Name	EIQ Con
CATEGORY 1:	**DANGER**	
propazine	Milogard	37
bentazon	Basagran 4S	31
bromacil	Hyvar	31
terbacil	Sinbar	31
atrazine	Atrazine	29.5
norflurazon	Solicam	29.5
trifluralin	Treflan	29.5
napropamide	Devrinol	29.26
simizine	Princep	29
ammonium sulfamate	Ammate	28
dalapon	Dalapon	28
metribuzin	Sencor	28
nicosulfuron	Accent	28
dichlobenil	Casoron	27
pryazon	Pyramin	27
CATEGORY 2:	**WARNING**	
acifluorfen	Blazer	**24**
diuron	Karmex	22.5
2,4-D (acid)	Weedone	**21**
linuron	Lorox	21
chloramben	Amiben	20
cyanazine	Bladex	19.3
metolachlor	Dual	19
cycloate	Ro-Neet	17
EPTC	Eptam	17
paraquat	Gramaxone	17
alachlor	Lasso	16.5
sethoxydim	Poast	16.5
CATEGORY 3:	**CAUTION**	
ethalfluralin	Sonolan	15
fluazifop-butyl	Fusilade	15
pronamide	Kerb	14
DCPA	Dacthal	13
MCPA	Bronate	13
oxyfluorfen	Goal	12.5
pendimethalin	Prowl	12.5
glyphosate	Roundup	11
phenmediphan	Spin-aid	9.5
diethatyl-ethyl	Antor	7
oryzalin	Surflan	5

Common Name	Trade Name	EIQ FW
CATEGORY 1:	**DANGER**	
2,4-D (acid)	Weedone	**36**
acifluorfen	Blazer	**36**
paraquat	Gramaxone	36

fluazifop-butyl	Fusilade	20
CATEGORY 2:	**WARNING**	
dalapon	Dalapon	18
pronamide	Kerb	18
MCPA	Bronate	16
ammonium sulfamate	Ammate	12
bentazon	Basagran 4S	12
propazine	Milogard	12
ethalfluralin	Sonolan	10
oxyfluorfen	Goal	10
CATEGORY 3:	**CAUTION**	
cyanazine	Bladex	8.7
DCPA	Dacthal	8
glyphosate	Roundup	8
linuron	Lorox	8
alachlor	Lasso	6
atrazine	Atrazine	6
bromacil	Hyvar	6
dichlobenil	Casoron	6
nicosulfuron	Accent	6
phenmediphan	Spin-aid	6
trifluralin	Treflan	6
chloramben	Amiben	5
diuron	Karmex	5
pendimethalin	Prowl	5
napropamide	Devrinol	4.26
metolachlor	Dual	4
oryzalin	Surflan	4
simizine	Princep	4
terbacil	Sinbar	4
metribuzin	Sencor	3
norflurazon	Solicam	3
sethoxydim	Poast	3
cycloate	Ro-Neet	0
diethatyl-ethyl	Antor	0
EPTC	Eptam	0
pryazon	Pyramin	0

FUNGIICDES

Common Name	Trade Name	EIQ Farmworker
CATEGORY 1:	**DANGER**	
copper sulfate+lime	Bordeaux	54
thiram	Thiram	36.9
metiram	Polyram	30
copper sulfate	copper	27
mancozeb	Manzate	24
maneb	maneb	24
zineb	Dithane Z	20
CATEGORY 2:	**WARNING**	
myclobutanil	Nova	18.45
benomyl	Benlate	18
thiophanate methyl	Topsin-M	18
maneb +dinocap	Dikar	16.4
chlorothalonil	Bravo	15
captan	Orthocide	14
triadmefon	Bayleton	14
dichloran	Botran	12.3
triforine	Funginex	12.3
vinclozolin	Ronilan	12.3
fentin hydroxide	Du-Ter	12
dodine	Syllit	10.25
CATEGORY 3:	**CAUTION**	
anilizaine	Dyrene	8.2
dinocap	Karathane	7.3
copper hydroxide	Kocide	6.15
streptomycin	Agristrep	6
PCNB	Terraclor	5
sulfur	Sulfur	5
fenarimol	Rubigan	4
fosetyl-Al	Aliette	4
folpet	Phaltan	3.1
iprodione	Rovral	3.1
carboxin	Vitavax	3
ferbam	Carbamate	3
flusilazol	Nustar	3
metalaxyl	Ridomil	3

Common Name	Trade Name	EIQ Consumer
CATEGORY 1:	**DANGER**	
benomyl	Benlate	70
fenarimol	Rugigan	43
CATEGORY 2:	**WARNING**	
zineb	Ditane Z	35
triforine	Funginex	32.3
thiophanate methyl	Topsin M	32
metalaxyl	Ridomil	31
cooper sulphate+lime	Bordeaux	23
triadmefon	Bayleton	22
mancozeb	Manzate	21
maneb	Maneb	21
dodine	Syllit	20.4
myclobutanil	Nova	20.15
metiram	Polyram	20
CATEGORY 3:	**CAUTION**	
copper sulphate	Copper	18.5
maneb+dinocap	Dikar	17.2
clorothalonil	Bravo	15
vinclozolin	Ronilan	13.1
flusilazol	Nustar	12.95
PCNB	Terraclor	12.5
folpet	Phaltan	12.05
captan	Orthocide	12
dichloran	Botran	11.15
thiram	Thiram	11.15
dinocap	Karathane	10.5
sulfur	Sulfur	10
streptomycin	Agristrep	9.5
anilizaine	Dyrene	9.1
copper hydroxide	Kocide	9.05
fentin hydroxide	Du Ter	9
ferbam	Carbamate	9
fosetyl-Al	Aliette	8
carboxin	Vitavax	7.25
iprodione	Rovral	7.05

Common Name	Trade Name	EIQ Ecological
CATEGORY 1:	**DANGER**	
mancozeb	Manzate	233
maneb	maneb	217.3
benomyl	Benlate	208.5
chlorothalonil	Bravo	184
copper sulfate+lime	Bordeaux	162
maneb +dinocap	Dikar	161.6
copper sulfate	copper	138.9
fenarimol	Rubigan	138
fentin hydroxide	Du-Ter	137

flusilazol	Nustar	131.2
thiram	Thiram	125
triforine	Funginex	121.4
myclobutanil	Nova	121.4
iprodione	Rovral	118.4
sulfur	Sulfur	117
thiophanate methyl	Topsin-M	116.5
dodine	Syllit	115.6
vinclozolin	Ronilan	114.4
copper hydroxide	Kocide	109.5
dichloran	Botran	108.4
zineb	Dithane Z	105.1
CATEGORY 2:	**WARNING**	
dinocap	Karathane	97.1
metiram	Polyram	93.8
PCNB	Terraclor	93
carboxin	Vitavax	89.9
streptomycin	Agristrep	89.15
folpet	Phaltan	87.7
triadmefon	Bayleton	86
ferbam	Carbamate	75.9
anilizaine	Dyrene	68.4
metalaxyl	Ridomil	62.5
captan	Orthocide	59.9
CATEGORY 3:	**CAUTION**	
fosetyl-Al	Aliette	19.5

INSECTICIDES

Common Name	Trade Name	EIQ Farmworker
disulfoton	Di-Syston	150
parathion	Niran, Phoskil	140
oxydemeton-methyl	Metasytox-R	96
carbofuran	Furadan	72
dimethoate	Cygon	72
propoxur	Baygon	72
dichlorvos	Vapona	60
methidathion	Supracide	60
ethoprop	Mocap	57.5
methyl parathion	Penncap-M	54
naled	Dibrom	54
rotenone	Chem Fish	54
ryania	Ryania	45.6
aldicarb	Temik	45
chlorpyrifos	Lorsban	45
fonofos	Dyfonate	45
fensulfothion	Dasanit	40
methamidophos	Monitor	40
phorate	Thimet	40
propargite	Omite	40
sabadilla	Red Devil	39.3
azinphos-methyl	Guthion	36
dicofol	Kelthane	36
endosulfan	Thiodan	36
ethion	Ethion	34.5
pirimicarb	Pirimor	34.2
mevinphos	Phosdrin	30
piperonyl butoxide	Butacide	30
terbufos	Counter	30
methoxychlor	Marlate	25
malathion	Cythion	21
permethrin	Ambush	20
phosphamidon	Swat	18
oxythioquinox	Morestan	16
diazinon	Diazinon	15
diflubenzuron	Dimilin	15
oxamyl	Vydate	15
cryolite	Kryocide	13.1
Bacillus thuringiensis	Dipel	12
carbaryl	Sevin	12
phosmet	Imidan	12
soap	M-Pede	11.4
esfenvalerate	Asana	8
fenvalerate	Pydrin	8
oil	Oil	8
acephate	Orthene	6

Common Name	Trade Name	EIQ Ecological
disulfoton	Di-Syston	187.8
propoxur	Baygon	176.8
parathion	Niran, Phoskil	165.1
phorate	Thimet	154.6
fensulfothion	Dasanit	146.6
methamidophos	Monitor	141.3
dimethoate	Cygon	140.9
permethrin	Ambush	140.8
methidathion	Supracide	139.8
esfenvalerate	Asana	136.8
fenvalerate	Pydrin	136.8
methoxychlor	Marlate	135.5
oxydemeton-methyl	Metasytox-R	122.6
ryania	Ryania	113.3
oxythioquinox	Morestan	110.1
chlorpyrifos	Lorsban	104.9
diflubenzuron	Dimilin	98
azinphos-methyl	Guthion	88.3
ethion	Ethion	86.2
fonofos	Dyfonate	82.8
propargite	Omite	82.2
methomyl	Lannate	81.5
diazinon	Diazinon	79.5
endosulfan	Thiodan	78.6
oil	Oil	71
carbofuran	Furadan	69.4
ethoprop	Mocap	67.2
terbufos	Counter	62.8
sabadilla	Red Devil	61.6
dichlorvos	Vapona	58.8
phosmet	Imidan	56.7
naled	Dibrom	55
formetanate	Carzol	54.4
phosphamidon	Swat	52.9
carbaryl	Sevin	52.7
aldicarb	Temik	52.4
dicofol	Kelthane	48.6
mevinphos	Phosdrin	48.5
methyl parathion	Penncap-M	47.7
pirimicarb	Pirimor	45.9
cryolite	Kryocide	45.2
oxamyl	Vydate	45.2
pyrethrin	Pyronone	45
malathion	Cythion	44
acephate	Orthene	43.7
soap	M-Pede	41.8

formetanate	Carzol	6
hexakis	Vendex	6
methomyl	Lannate	6
pyrethrin	Pyronone	6

Common Name	Trade Name	EIQ Consumer
carbofuran	Furadan	29
oxydemeton-methyl	Metasytox-R	29
disulfoton	Di-Syston	28
aldicarb	Temik	14
fensulfothion	Dasanit	14
methoxychlor	Marlate	13.5
propoxur	Baygon	13
pirimicarb	Pirimor	11.4
methamidophos	Monitor	11
methomyl	Lannate	11
phorate	Thimet	10
ethoprop	Mocap	9.1
dimethoate	Cygon	9
chlorpyrifos	Lorsban	8.5
oxamyl	Vydate	8.5
permethrin	Ambush	8.5
diazinon	Diazinon	8
methidathion	Supracide	8
parathion	Niran, Phoskil	8
phosphamidon	Swat	8
endosulfan	Thiodan	7
oxythioquinox	Morestan	7
ryania	Ryania	7
Bacillus thuringiensis	Dipel	6
cryolite	Kryocide	6
fonofos	Dyfonate	6
mevinphos	Phosdrin	6
propargite	Omite	6
sabadilla	Red Devil	6
diflubenzuron	Dimilin	5.5
soap	M-Pede	5.1
azinphos-methyl	Guthion	5
dicofol	Kelthane	5
malathion	Cythion	4.5
acephate	Orthene	4
esfenvalerate	Asana	4
fenvalerate	Pydrin	4
formetanate	Carzol	4
methyl parathion	Penncap-M	4
naled	Dibrom	4
rotenone	Chem Fish	4
terbufos	Counter	4
hexakis	Vendex	3.7
oil	Oil	3.7
piperonyl butoxide	Butacide	3.7
carbaryl	Sevin	3
dichlorvos	Vapona	3

rotenone	Chem Fish	41
hexakis	Vendex	28.8
piperonyl butoxide	Butacide	28.7
Bacillus thuringiensis	Dipel	22.5

phosmet	Imidan	3
pyrethrin	Pyronone	3
ethion	Ethion	2.5

APPENDIX 5. Chapter 7. Testing the relationship between the linear EIQ curve and laboratory tested dose response curves. Chi Square test.

Table 8.4

Laboratory - 0	EIQ - E	Chi-Sq
18	18	0
33	38	0.657895
58	59	0.016949
82	80	0.05
100	100	0
		0.724844

Table 8.5

Laboratory - O	EIQ-E	Chi-Sq
0	0	0
30	26.5	0.462264
40	53.5	3.406542
100	100	0
		3.868806

Table 8.6a

Laboratory - O	EIQ-E	Chi-Sq
2	28	0
8		
19	22	0.409091
15.5	18	0.347222
8	15	3.266667
3.5	10	4.225
0	6	6
		14.24798

Table 8.6b

Laboratory - o	EIQ-E	Chi-Sq
75	75	0
31	62.5	15.876
11	50	30.42
2.5	38	33.16447
1	25	23.04
0	12	12
0	0	0
		114.5005

Table 8.7

Laboratory - O	EIQ-E	Chi-Sq
0	0	0
46	40	0.9
65	50	4.5
100	100	0
100	100	0
100	100	0
		5.4

The purpose of carrying out a Chi-Square on laboratory observed, and EIQ linear dose response curves was to ascertain whether or not it could be concluded that the EIQ model adequately described the impacts associated with pesticides.

Results	Chi-Sq	Probability	Degrees of freedom	Hypothesis accepted
Table 8.4	0.724	0.9	4	Yes
Table 8.5	3.87	0.49	4	Yes
Table 8.6a	14.25	0.09	4	Yes
Table 8.6b	114.5	n/a	4	No
Table 8.7	5.4	0.27	4	Yes

Of the five tables examined all were within the acceptable statistical range indicating that there is no significant difference between the two data sets, except for Table 8.6b. Casual observation of this curve intuitively confirms that it is not linear. Statistically then, it is possible to suggest that, from the examples of laboratory studies presented in Chapter 8, the EIQ model adequately represents the environmental impacts associated with pesticide use. The male/female mortality curve tested in Table 8.4 represents an extremely good fit, with approximately a 0.9 probability with 4 d.f. Therefore, we can conclude that the EIQ linear dose response curve adequately describes environmnetal impact when compared to the laboratory studies reviewed.