Economic analysis of pesticide use and environmental spillovers under a dynamic production environment

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Abstract

Pesticides are used in agriculture to protect crops from pests and diseases, with indiscriminate pesticide use having several adverse effects on the environment. In an era of an increasing public awareness on pesticides' environmental spillovers, the EU is trying to update its pesticide policy by using economic incentives, aiming at reducing pesticide use and environmental spillovers. This dissertation focuses on assessing how pesticide use and its related environmental spillovers are affecting farmers' production environment under a dynamic production setting, thus assisting policy makers in designing optimal pesticide policy tools.

This dissertation met this research aim through an extensive study of relevant literature and the implementation of empirical research. The former included the identification of the contours of an optimal pesticide policy scheme, exploring the information needed for the introduction of such a policy framework in order to identify knowledge gaps to be addressed to support the design of optimal pesticide policies. The empirical research included an empirical evaluation of the impacts of pesticide use and environmental spillovers in agricultural production, empirical assessment of the impacts of pesticide tax and levy schemes on pesticide use and the environment, non-parametric efficiency analysis of arable farms taking into account pesticide dynamics, biodiversity and production uncertainty, and risk-adjusted efficiency analysis of arable farms considering explicitly the risk-increasing or-decreasing nature of pesticides and other inputs.

This research produced a number of key findings: the development of environmental standards, where differentiated tax rates can be based on, needs further attention due to inadequate information on pesticides' environmental spillovers; the indirect impacts of pesticides on biodiversity have a significant impact on farmer's production environment; pesticides are on average overused in Dutch arable farming; pesticide taxes as a single instrument can be characterized as ineffective since they yield small decreases in pesticide use and environmental spillovers; Dutch arable farmers have noticeable output and pesticide environmental inefficiency scores; and when adjusting outputs and inputs to account for the impact of variability in production conditions, estimates of inefficiency decreased dramatically.

The main conclusions drawn from this research were that an optimal pesticide policy should involve incentives to achieve environmental and health standards; future pesticide policies should try to decrease pesticide use and conserve organisms beneficial for the farm; and that our understanding of efficiency levels can be distorted when using models that ignore the dynamics of production and the effects of variability in production conditions.

Keywords: Pesticides, biodiversity, dynamics, environmental spillovers, arable farming, economic incentives, production uncertainty, Netherlands.

Preface

The work described in this dissertation was carried out between April 2008 and March 2012 at the Business Economics Group, Wageningen University. After passing from some interviews early in 2008 my future supervisor Alfons Oude Lansink was announcing me that I was going to be a new Ph.D. student in his department. I do not have words to express how I felt at this moment and it took me some days to realize that a dream was coming true. At a later meeting in March 2008 I met Alfons but this time he was not alone. Spiro E. Stefanou, my other future supervisor, was also there saying the following words: “Theo, the coming years we will roast you on a spit like a lamp during the Greek Easter”. The “roasting” was about to start. When I started to work on my dissertation it was difficult to envisage how it would evolve in the coming years. As my dissertation was funded by the European Union project TEAMPEST, I quickly obtained a broad research proposal. But during these four years the road became clearer resulting in this Ph.D. thesis.

As I complete this thesis, I have drawn on the patience, guidance and collaboration of several people to whom I am indebted. First of all, I would like to take the opportunity to thank my supervisors, Prof. Alfons Oude Lansink, and Prof. Spiro E. Stefanou for the supervision, guidance, and support they gave me during these four years. Alfons, thank you for your guidance, help with different statistical software, enduring support I got from you during times of disappointment, and critical reviews of my work. Your patience, dedication, diplomacy, and quick responses are the most important things I will take away from our cooperation. Spiro, you have not only been an advisor to me, but a mentor, a second father and a dear friend. I owe you a lot and I am truly grateful for the moral and technical support you gave. You transformed my thinking and exposed me to many challenges in the working field. Your teaching and supervising methods will be an important reference in my professional development. The long discussions we had after our meetings on cooking, women, human relationships, and dressing style, enabled me to escape from the working routine and enriched my personality.

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Lastly, and most importantly, I wish to thank my parents, Athanasios Skevas and Chrysoula Karagiozi. They bore me, raised me, supported me, taught me, and loved me. To them I dedicate this thesis. Αγαπητοί Γονείς, αυτή η διδακτορική διατριβή είναι αφιερωμένη σε εσάς. Η αγάπη μου και εκτίμηση για εσάς και για όσα έχετε κάνει για εμένα είναι τόσο μεγάλη όσες οι νυφάδες χιονιού που πέφτουν στις απάτητες βουνοκορφές, όσα τα κύματα του Αιγαίου. Σας ευχαριστώ απο τα βάθη της καρδιάς μου.

Skevas Theodoros (Wageningen, November, 2011)

Contents

Chapter 1

General Introduction

1.1 Background

During the last decades, there is a considerable increase in the global level of food production. This growth that was brought about mainly by technological innovations has its impact on the environment. Agricultural intensification has resulted in environmental degradation but, on the other hand, the development of pollution abatement technologies promises to ease these environmental problems. Sustainable agricultural production is of primary importance in sustaining human needs and protecting the natural habitat.

Plant protection products constitute one of the most important agricultural inputs in developed countries. Being a damage- and risk-reducing input, these products are widely used in agricultural production (EC, 2006). Their stochastic nature (productivity and climatic conditions, pest arrival) is related to uncertainty on the timing and the way of applying them. There is a large range of positive outcomes from the use of pesticides. Pesticides can help in securing and improving crop yields and quality of the obtained products resulting in increased farm and agribusiness revenues. Other benefits of pesticide use is the improved shelf life of the produce, reduced drudgery of weeding that frees labor for other tasks, reduced fuel use for weeding, and invasive species control (Cooper and Dobson, 2007). On the other hand, pesticide application is related to various externalities that call for an immediate rational use of these chemical substances.

Starting with the publication of Rachel Carson’s Silent Spring in 1962 which highlighted the risks of pesticide use, continuous use of chemical inputs such as pesticides produces significant negative externalities that have been broadly documented in the scientific literature (Pimentel et. al., 1992; Pimentel and Greiner, 1997). Pesticides can be dangerous for human health when the degree of exposure exceeds the safety levels. This exposure can be direct, such as the exposure of farm workers applying pesticides to various crops and indirect by consumers consuming agricultural products containing chemical traces or even bystanders near application areas. Additionally, the excessive and uncontrolled use of pesticides can pose serious and irreversible environmental risks and costs. Fauna and flora have been adversely affected while the decline of the number of beneficial pest predators has led to the proliferation of different pests and diseases (Pimentel and Greiner, 1997). Certain pesticides applied to crops eventually end up in ground and surface water. In surface water like streams and lakes, pesticides can contribute to fishery losses in several ways (Pimentel et al., 1992). High chemical concentrations can kill fish directly or indirectly by killing the insects that serve as fish food source. Moreover, the extensive use of pesticides has often resulted in the development of pesticide resistant weeds and pests (Powles et al., 1997; Jutsum et al., 1998). This can trigger increased pesticide applications to reduce pest damage and avoid crop loss. Pimentel et al. (1992) mention many adverse consequences from the overuse of pesticides such as animal poisoning, contaminated products, destruction of beneficial natural predators and parasites, bee poisoning and reduced pollination, crop and biodiversity losses.

Many international and national policies are aiming to reduce pesticide use as consumers are becoming more aware of pesticide externalities and demand pesticide free agricultural products and cleaner and safer natural habitat. Important efforts towards regulating pollution have been made in industrialized countries in the form of increasingly stringent environmental regulations. Regulations on the marketing of plant protection products, maximum residue levels and the thematic strategy on the sustainable use of pesticides compose the puzzle of the European pesticide policy. European Union is aiming at implementing coherent pesticide regulations incorporating economic incentives (i.e. tax and levy schemes) in an effort to reduce pesticide use and indirect effects, thus protecting public health and the environment. Research at the farm-level may provide important information to policy makers seeking to introduce optimal pesticide policies.

1.2 Farm-level empirical research and pesticide policy

Farm-level approaches are important in primary policy analysis as the design and implementation of an efficient pesticide policy (based on economic incentives) requires information on pesticide use, and the indirect effects of pesticides (Hoevenagel et al., 1999; Oskam et al., 1997). Findings coming from farm-level approaches can assist policy makers in introducing optimal economic incentive-based pesticide policies by revealing important information on pesticide demand elasticity, evidence on overuse or underuse of pesticides, impact of pesticides' indirect effects on output realization, and efficiency of the use of pesticides.

Evidence on pesticide demand elasticity provides insights on a potential pesticide tax rate, while information on pesticide overuse or underuse reveal the products to be targeted for reductions. Investigating the production impact of pesticides' environmental spillovers shows the role farmland biodiversity plays on output realization, thus providing useful information for preservation strategies. Information on the extent to which farmers use pesticides efficiently and contribute to spillovers can reveal if there is a potential for reducing pesticide use and their spillovers.

Despite the importance of a farm-level approach on primary policy analysis, little empirical research has been done on jointly investigating the impact of pesticide use and environmental spillovers on production. Lack of detailed farm-level data on pesticides environmental spillovers may be considered a reason for the lack of empirical evidence.

1.3 Study objective and research questions

The objective of this study is to provide a theoretical and empirical evaluation of the impact of pesticide use, and environmental spillovers in output realization under a dynamic production environment, thus assisting policy makers in introducing optimal pesticide policies. To achieve this goal five research questions are addressed:

1. What is the contour of an optimal pesticide policy scheme and what are the knowledge gaps to be addressed to support the design of optimal pesticide policies?
2. Are pesticides' impacts on biodiversity affecting agricultural output?
3. Are pesticide tax and levy schemes effective in reducing pesticide use and environmental spillovers in Dutch arable farming?
4. What is Dutch farmers' technical and pesticides' environmental inefficiency when considering pesticide dynamic effects on biodiversity and production uncertainty?
5. What is Dutch farmers' technical and allocative inefficiency when accounting for undesirable outputs and the risk-increasing or decreasing nature of agricultural inputs?

1.4 Research approach and data

Different approaches were used to answer the five research questions. For question one, a literature review took place to identify the optimal pesticide policy. This was followed by an identification of the elements needed to apply such a policy framework, i.e., the production structure (i.e., production function, pesticide demand elasticities), attitudes toward risk and uncertainty related to pesticides application, the value of pesticides to consumers (e.g., the willingness to pay (WTP) for lower pesticide use), and the effects of pesticide use on biodiversity in relation to existing pesticide policies. Reviewing the literature on the pre-mentioned elements enabled the identification of knowledge gaps that need to be addressed to introduce an optimal pesticide policy scheme.

Question two aims at an empirical assessment of the impact of pesticides' environmental spillovers on farmers' production environment. For this purpose, a dynamic model of optimal pesticide use was employed incorporating two pesticide categories that differ in terms of toxicity, and pesticides' environmental spillovers in different specifications of the production function. Furthermore, shadow prices of the different inputs were computed to assess whether an input is overused or not, thus providing important information to policy makers aiming at designing subsidies and taxes for different inputs.

Question three, addresses the impact of tax and levy schemes on pesticide use and environmental spillovers. In other words, the aim of this research question is to investigate whether economic incentives can alter pesticide decisions at the farm level such that environmental spillovers of pesticides are reduced. A dynamic model of optimal pesticide use is developed and estimated econometrically. Following the econometric estimation, a dynamic optimization model is developed that is used to assess the impacts of several scenarios of tax and levy schemes, and quotas on pesticide use and environmental spillovers.

Question four aims at investigating the performance of Dutch arable farms when taking into account pesticide dynamics and production uncertainty (i.e., the variation in production arising from climatic events and other random forces). A dynamic Data Envelopment Analysis (DEA) model is applied to outputs, inputs, and undesirables of Dutch arable farms. The Simar and Wilson (2007) double-bootstrap procedure is employed to explain technical inefficiency using socioeconomic and environmental variables, thus providing empirical representations of the impact of stochastic elements and the state of nature on production. The results of the double-bootstrap procedure are used to adjust firms' outputs and inputs to account for the impact of variability in production conditions.

Question five, aims at investigating the performance of Dutch arable farms taking explicitly into account the risk behaviour of producers. A non-parametric risk-adjusted inefficiency model is used utilizing undesirable outputs and accounting explicitly for the effect of production means on output variability. Output, risk-mitigating inputs and, undesirable outputs inefficiency are computed. Moreover, shadow values of the risk-mitigating inputs are calculated, providing the extent to which these inputs are over- or under-used.

The study uses panel data of Dutch cash crop farms over the period 2002-2007 from the Agricultural Economics Research Institute (LEI). The available data are composed by the Farm Accountancy Data Network (FADN) database and detailed data on pesticide use at the farm level. The dataset includes also biodiversity indicators and climatic variables at the farm level. Biodiversity indicators were obtained from the Dutch Centre for Agriculture and Environment (CLM). For each pesticide that Dutch arable farmers use, there is an environmental indicator which shows the impact on different farmland biodiversity categories (i.e., water and soil organisms, biological controllers). Meteorological data include precipitation and temperature and were obtained from the Royal Netherlands Meteorological Institute (KNMI, 2011).

1.5 Outline

This work is composed of seven chapters including the general introductory chapter. Each chapter concentrates on one of the research questions outlined in section 1.3. This section presents the highlights of each chapter. Chapter 2 presents the contour of an optimal pesticide policy scheme and the knowledge gaps that need to be addressed to introduce such a policy framework. Chapter 3 presents a dynamic model of optimal pesticide use and provides estimates of the impact of pesticides and environmental spillovers on output realization. Additionally, shadow values of pesticides and other inputs are computed, providing important information to policy makers seeking to develop tax and levy schemes.

Chapter 4 uses a simulation model to test the impact of different tax and levy schemes on pesticide use and environmental spillovers. Pesticide taxes, subsidies, and quotas are used to examine their impact on farmers' attitudes, providing valuable information to policy makers aiming at introducing economic incentive-based pesticide policies. Chapter 5 presents a DEA model accounting for pesticide dynamics and production uncertainty to measure the performance of Dutch arable farms. Socioeconomic and environmental variables are used to explain farmers' performance, thus providing empirical evidence for the design of pesticide policy measures. The results highlight the extent to which efficiency measures are distorted when ignoring pesticide dynamics and production uncertainty.

Chapter 6 presents a risk-adjusted DEA model to measure the performance of Dutch arable farms. This chapter shows a way to incorporate undesirable outputs and risk-mitigating inputs in DEA modelling frameworks and take explicitly into account the risk-increasing or –decreasing effect of production inputs on output realization, thus focusing on the risk behaviour on the part of the producer. Finally, Chapter 7 highlights and synthesizes the main findings of the study and discusses the major policy implications, and suggestions for future research.

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Chapter 2

Designing the Emerging EU Pesticide Policy: Lessons Learned

This chapter is in review at *Environmental Policy and Governance* as:
Skevas, T., S.E., Stefanou, and A., Oude Lansink. ″Designing the emerging EU pesticide policy: Lessons learned″.

Abstract

An European Union (EU) wide pesticide tax scheme is among the future plans of EU policy makers. This study presents an optimal pesticide policy framework and examines the information needs for applying such a framework at the EU level. Damage control specification studies, empirical results from pesticide demand elasticity, issues on pesticide risk valuation and uncertainty, and knowledge on pesticides' indirect effects in relation to current pesticide policies are analysed. Knowledge gaps based on reviewing these information are identified and an illustration of the direction future pesticide policies should take is provided.

Keywords: pesticides, pesticide policy, tax schemes, EU.

2.1 Introduction

There has been a considerable increase in the global production of agricultural goods and services in recent decades. Plant protection products have played a major role in driving this growth, as have other technological innovations. However, growth in global agricultural production has a concomitant impact on the environment.

Plant protection products are active substances that enable farmers to control different pests or weeds, and thus constitute one of the most important inputs in agricultural production (Commission of the European Communities, 2006). There is a large range of positive outcomes from the use of different pesticides related to agricultural productivity, but the potential benefits are particularly important in developing countries, where crop losses contribute to hunger and malnutrition (Anon, 2004). Additionally, improving crop yields and the quality of production results in increased farm and agribusiness profits. With weeds being the major yield-reducing factor for many crops, herbicides are the most widely used type of pesticides. Cooper and Dobson (2007) refer to a number of benefits from pesticide use, among which are the improved shelf life of produce, the reduced drudgery of weeding, which frees labor for other tasks, reduced fuel use for weeding, invasive species control, increased livestock yields and quality, and garden plants protection.

Starting with the publication of Rachel Carson's Silent Spring (1962) which highlighted the risks of pesticide use, there has been steady progress in documenting the negative spillovers arising from the continuous use of chemical inputs (Pimentel et. al., 1992; Pimentel and Greiner, 1997). Pesticides are not restricted to use in agriculture: they are used for landscaping, on sporting fields, road and railway side weed control, and public building maintenance. These substances can be dangerous for human health when the degree of exposure exceeds the safety levels. Exposure can be direct, for example when farm workers apply pesticides to various crops, and indirect, such as when consumers ingest agricultural products containing chemical traces, or even when bystanders happen to be nearby application areas. Food and Agriculture Organization (2008) evidence suggests that tens of thousands of farmers are exposed to pesticides each year. The largest number of poisonings and deaths is recorded in developing countries, where farmers often do not use appropriate protective equipment.

Additionally, the excessive and uncontrolled use of pesticides can pose serious and irreversible environmental risks and costs. The decline in the number of beneficial pest predators has led to the proliferation of various pests and diseases with adverse impacts on fauna and flora (Pimentel and Greiner, 1997). Certain pesticides applied to crops eventually end up in the ground and surface water. In surface water environments such as streams and lakes, pesticides can contribute to fishery losses in various ways (Pimentel et al., 1992). Moreover, the extensive use of pesticides has often resulted in the development of pesticide resistant weeds and pests. This can trigger increased pesticide application to reduce the respective damage, resulting in high economic costs that farmers must shoulder. Further, Pimentel et al. (1992) address the adverse environmental consequences from the overuse of pesticides such as animal poisoning, contaminated products, destruction of beneficial natural predators and parasites, bee poisoning and reduced pollination and crop and biodiversity losses.

Currently, the EU aims to upgrade existing pesticide regulations, which includes the introduction of an EU-wide regulatory framework on pesticides grounded upon economic incentives. The foundation of future EU policy schemes aims at the sustainable use of pesticides in European agriculture. This effort involves reducing the risks and impacts of pesticide use on human health and the environment, while still being consistent with crop protection. The design of optimal pesticide policies requires insight into the relationships between production decisions on crop yields and their quality, the environmental and health spillover impacts of pesticide use, and how policies and regulations influence production decision-making. A key policy consideration is balancing the incentives for economic growth against the adverse impact on the environment, which is broadly defined to include the management of land, water and air, as well as the overall stability and biodiversity of the ecological system.

The objective of this paper is to present the contour of an optimal pesticide policy scheme and explore the potential for introducing such a scheme at the EU level. More specifically, this paper reviews the information needed for the introduction of such a policy framework to identify knowledge gaps to be addressed to support the design of optimal pesticide policies. The remainder of the paper is organized as follows. The next section presents an optimal pesticide policy framework. This is followed by a review of the existing literature on pesticides indicating the extent to which the current literature provides information needed for the implementation of optimal pesticide policies. The final section discusses knowledge gaps based on the literature review.

2.2 An optimal pesticide policy framework

Under the Pigouvian tradition the optimal pesticide policy grounded on economic incentives should include taxes (or subsidies) to control pesticide externalities, where the tax (or subsidy) reflects the marginal net damage (benefit) of pesticides' use. The problem in such a policy framework is that obtaining an accurate estimate of the monetary value of pesticide damage (or benefit) is not an easy task, mainly due to prohibitive information requirements. Alternatively, Baumol and Oates (1988) proposed the establishment of a set of standards or targets for environmental quality followed by the design of a regulatory system that could employ unit taxes (or subsidies) to achieve these standards. The authors add that although this will not result in an optimal allocation of resources (such as pesticides) it represents the most cost effective way in attaining the specified standards. A pesticide policy framework that combines market-based instruments with standards for acceptable environmental and health quality will enable policy makers to base the charge rates or prices on the acceptability standards rather than on the unknown value of marginal net damages. In this way taxes can reflect the potential environmental and health damage from each pesticide (Pretty et al., 2001; Hoevenagel et al., 1999; Oskam et al., 1997).

The design and application of a pesticide policy framework grounded on market-based instruments and environmental and/or health standards, requires rigorous information on different dimensions and aspects of pesticide use. The elements needed to apply such a policy framework may be summarized by information on a) the production structure (i.e., production function, pesticide demand elasticities), b) attitudes toward risk and uncertainty related to pesticides application, c) the value of pesticides to consumers (e.g., the willingness to pay (WTP) for lower pesticide use), and d) the indirect effects of pesticide use. Information on the production structure of pesticide use include trends in pesticide use (overuse or underuse), and the direction and extent farmers' behavior will change following the introduction of a pesticide tax. In particular, will a pesticide price increase lead to significantly decreased pesticide use? Information on the riskiness of pesticides in relation to output realization may enhance the effectiveness of pesticide policy tools while evidence on the consumers' WTP for reducing pesticide adverse effects can reveal if there is a demand for more environmental friendly products providing an incentive for farmers to switch to more environmental friendly forms of production (e.g., organic or IPM). Finally, detailed data on pesticides' indirect effects can assist policy makers in setting proper environmental and health standards that can increase the effectiveness of the different economic instruments.

It is important to notice that optimal pesticide use may be achieved not only through the use of market-based instruments, such as taxes and subsidies, but also of alternative instruments. For instance, command-and-control regulations may be among the means to achieve a policy goal. Unlike market-based instruments encouraging firms' behavior through market signals, command-and-control regulations set uniform standards for firms. An example of a command-and-control measure in relation to pesticide policy is bans on the use of specific pesticides. Stavins (2003) argues that despite the proven success of market-based instruments in reducing environmental pollution at a low cost, they did not come close to replacing command-and-control measures. Given that market-based instruments are difficult to change on short notice, command-and-control measures (e.g., bans on pesticides) can provide flexibility to policy makers. As research on pesticide externalities advances, pesticide bans may always have a place in pesticide policy frameworks.[1] Baumol and Oates (1988) add that a mixed system of regulations, composed of both fiscal and non-fiscal measures, constitutes an optimal regulatory strategy to reduce firms' externalities. Another alternative or complement to public or market intervention instrument can be agricultural production in certified farms (organic or IPM) or self-regulation. Farmers can form groups with common production rules (e.g., IPM), facing the opportunity to gain from their collective capacity to establish a reputation for their products. In this way farmers can experience higher revenues and society can be benefited from reduced pesticide externalities. The formation of producer organizations can be promoted by governments by providing financial facilities (e.g., lower firm taxation).

[1] Many active ingredients have been recently banned based on their adverse effect on human health.

2.3 Production structure

2.3.1 Production function

The concept of damage abatement input, first introduced by Hall and Norgaard (1973) and Talpaz and Borosh (1974), suggests that pesticides have an indirect effect on output in future years arising from pesticide resistance rather than a direct yield-increasing effect. Lichtenberg and Zilberman (1986) were the first to specify production functions that are consistent with the concept of damage abatement input. Apart from pesticides, damage control inputs could include windbreaks, buffer zones and antibiotics. The Lichtenberg and Zilberman (LZ) (1986) damage control framework enables economists to observe that the Cobb-Douglas formulations used in this study resulted in an upward bias in the optimal pesticide use estimations, while recent evidence suggests an overuse (Babcock et al., 1992; Guan et al., 2006). Additionally, the damage control specification accounts for changes in pesticide productivity and enables the prediction of producers' behavior. Pest resistance initially triggers farmers to apply more pesticides until alternative damage control measures become more cost effective. The LZ damage control specification was applied by Babcock et al. (1992), Carrasco-Tauber and Moffit (1992), Chambers and Lichtenberg (1994), Oude Lansink and Carpentier (2001), Oude Lansink and Silva (2004). Guan et al. (2006) and Lin et al. (1993). Table 1 reviews these studies using a set of common criteria: a) setting, b) modeling framework, c) data and application, and d) results and policy implications. The results are mixed with some studies that indicate the over-utilization of pesticides, and others that indicate its under-utilization.

Although the LZ specification has been applied successfully and constitutes a considerable innovation, some authors have expressed concerns. Oude Lansink and Carpentier (2001) have shown that in a quadratic production function, the lack of differentiation between damage abatement inputs and productive inputs does not lead to overestimation of the marginal product, as Lichtenberg and Zilberman (1986) argued. Additionally, Oude Lansink and Carpentier (2001) allow for interaction between damage abatement and other production inputs, where the LZ specification precludes these interactions. Oude Lansink and Silva (2004) challenge the assumption of a non-decreasing damage control function and assumptions imposed on parameters in the damage control model, and propose a nonparametric specification.

Although the LZ specification constitutes a useful and widely acceptable tool in the economics of pesticide use, the various critiques and mixed results developed by some authors perpetuate the debate regarding this specification. The majority of findings show that pesticides are under-utilized, which is contrary to the conventional view. Some interesting insights emerge from studies implementing the LZ specification predicting the over-utilization of pesticides. The choice of specification for the damage abatement function significantly impacts pesticide productivity estimates. Studies permitting specifications allowing a decreasing marginal product of pesticides are more likely to predict pesticide overuse. Furthermore, the examined product/crop can influence the final result regarding the overuse or underuse of pesticides. Babcock et al. (1992) applied the LZ specification on apple production data in North Carolina and found that pesticides are overused. However, apple production requires a considerable amount of preventive pesticide application in order to obtain high quality output. Therefore, this preventive application can justify the over-utilization of pesticides.

Table 1. Studies that have successfully applied the LZ specification

Study	Modeling framework	Data/Application	Trend in pesticide use
Oude Lansink, & Carpentier, 2001.	Generalized Maximum Entropy Estimation.	Secondary; Arable farms, The Netherlands.	Under-utilized
Babcock et al., 1992.	Damage control dynamic model.	Secondary; Apple Production; North Carolina (U.S.).	Over-utilized
Carrasco- Tauber, & Moffit, 1992.	Damage control dynamic model.	Secondary; U.S. agriculture.	Under-utilized
Oude Lansink & Silva, 2004.	Data Envelopment Analysis	Secondary; Arable farms, The Netherlands.	Under-utilized
Chambers & Lichtenberg, 1994.	Dual representation of generalized LZ model.	Secondary; U.S. agriculture.	Under-utilized
Guan et al., 2006.	Damage control model.	Secondary; Potato production; The Netherlands.	Over-utilized
Lin et al., 1993.	Cobb-Douglas Vs exponential, logistic and Weibul damage-control specification.	Secondary; Potato production; Pacific Northwest, US.	Trend depends on model selection.

2.3.2 Pesticide Demand Elasticity

The design of regulatory frameworks for levies on pesticides requires estimates of pesticide demand elasticities. If pesticide demand is inelastic, a new tax or levy will not affect pesticide use significantly, but it will generate revenues that can be distributed to the agricultural sector. Table 2 presents a review of the pesticide demand elasticity estimates of European countries, as well as the United States. A general conclusion based on this table is that the price elasticity of pesticide demand is quite low (in most cases), indicating that pesticide use is indifferent to pesticide price increases. Inelastic demand can indicate a lack of knowledge among farmers regarding alternative production practices, a strong intention toward risk-aversion, or can be due to behavioral factors like professional pride derived from weed-free fields. Inelastic pesticide demand is also reported by Hoevenagel et al. (1999) in their study of an EU wide scheme for levies on pesticides. Therefore, a tax on pesticides can create considerable revenues but it will have a small impact on reducing pesticides' externalities. Another important point is that the more specific the pesticide (e.g., aggregating over all fungicides, insecticides), the higher the elasticity of demand. This suggests that there are few substitutes to these specific products, with the result being that the producers face difficulties in adjusting their agricultural practices. The difficulty of finding lower-risk alternatives or applying alternative crop protection practices is also mentioned by Wilson and Tisdel (2001).

Table 2. Pesticide Demand Elasticity Estimates

Study	Country/Region	Elasticity
Aaltink (1992)	Netherlands	-0.13 to -0.39
Antle (1984)	USA	-0.19
Bauer et al. (1995)	German regions	-0.02
Brown & Christensen (1981)	USA	-0.18
Carpentier (1994)	France	-0.3
DHV & LUW (1991)	The Netherlands	-0.2 to -0.3
Dubgaard (1987)	Denmark	-0.3 (threshold approach)
Dubgaard (1991)	Denmark	-0.7 (herbicides), -0.8 (fung.+insect.)
Ecotec (1997)	UK	-0.5 to -0.7 (herbicides)
Elhorst (1990)	Netherlands	-0.3
Falconer (1997)	UK (East Anglia)	-0.1 to -0.3
Gren (1994)	Sweden	-0.4 (fung.), -0.5 (insect.), -0.9 (fung.)
Johnsson (1991)	Sweden	-0.3 to -0.4 (pesticides)
Komen et al. (1995)	Netherlands	-0.14 to -0.25
Lichtenberg et al. (1988)	USA	-0.33 to -0.66
McIntosh & Williams (1992)	Georgia (USA)	-0.11
Oskam et al. (1992)	Netherlands	-0.1 to -0.5 (pesticides)
Oskam et al. (1997)	EU	-0.2 to -0.5 (pesticides)
Oude Lansink (1994)	Netherlands	-0.12
Oude Lansink & Peerlings (1995)	Netherlands	-0.48 (pesticides)
Papanagiotou (1995)	Greece	-0.28
Petterson et al. (1989)	Sweden	-0.2
Rude (1992)	Sweden	-0.22 to -0.32
Russell et al. (1995)	UK (Northwest)	-1.1 (pesticides in cereals)
SEPA (1997)	Sweden	-0.2 to -0.4
Schulte (1983)	Three German regions	-0.23 to -0.65
Villezca &Shumway (1992)	Texas & Florida (USA)	-0.16 to -0.21

Source: Hoevenagel et al., (1999); Falconer & Hodge (2000); Fernandez-Cornejo et al. (1998)

2.4 Risk and uncertainty in relation to pesticide use

Pest arrival is an uncertain event, and pesticide productivity varies, leading to uncertainty in operator profit. This uncertainty can lead to the overuse of pesticides relative to the private or social optimum. Norgaard (1976) notes that the major motivation for pesticide application is the provision of some “insurance” against damage. Feder (1979) shows that an increase of the degree of uncertainty due to pest damage will cause an increase in the volume of pesticide use. As uncertainty in the pest-pesticide system leads to higher and more frequent use of pesticides, there is also uncertainty regarding the effectiveness of pesticides. Many times, farmers lack full knowledge of the relation between pesticides and pest mortality (Feder, 1979). The effectiveness of pesticides can be influenced by fluctuations in temperature, as well as wind and humidity conditions. Therefore, the pest population can vary with changes in climatic conditions, though these changes can also alter the effect of pesticides, as every chemical product has different durability.

A production function should process enough flexibility that the impact on the deterministic component of production is different from that on the stochastic component. The Just and Pope (1978) approach to modeling production processes in the face of production risk has been a popular addition to the literature, and is widely used in applied analyses related to pesticide use. The variation in production is influenced by the input levels; some inputs may be variation-increasing, while others are variation-decreasing, where risk is defined as the variance of output. Saha et al., (1994) and Griffiths and Anderson (1982) also support the conventional view that pesticides are risk-reducing inputs.

On the other hand, Horowitz and Lichtenberg (1994) show that a limited knowledge of the production process, captured by assuming that pest damage is independent of other factors affecting output, leads to the conventional view that pesticides are risk-reducing inputs. Pesticides may increase risk when pest populations are positively correlated with growth conditions. When pest populations are high and growth conditions are favorable, pesticides will be risk-increasing as they increase the variability of harvests (increase output under good growth conditions). Gotsch and Regev’s (1996) study of Swiss wheat producers shows that fungicides have a risk-increasing effect on farm revenues when rain levels are low. Similar results are reported by Saha et al., (1997) when the production process takes into account the interaction between pesticides and fertilizers, and by Pannell (1995) where herbicides have a risk-increasing effect on wheat farmers in Kansas. Horowitz and Lichtenberg (1993) have shown that pesticides may be risk-increasing inputs and that farmers who purchase federal government crop insurance use more chemicals ceteris paribus. This view on pesticides is contradicted by Smith and Goodwin (1996).

Saha et al. (1997) report the importance of considering the stochastic nature of both the damage control and the production function in order to avoid overestimating the marginal productivity of damage control inputs. With pesticide productivity affected by the level of the developed resistance, the more resistant the pest population, the higher the use of the damage control agents (pesticides) until resistance is sufficiently pervasive and alternative damage control measures are more cost effective.

Uncertainty related to pesticide externalities has become an important health and environmental regulatory issue. The absence of full knowledge on pesticide's side effects can lead to irreversible environmental and health damage if policy-makers postpone pesticide management measures to wait for further scientific knowledge. The precautionary principle, first defined in the 1992 Rio Declaration, addresses this issue by maintaining that uncertainty regarding the environmental or health effects of pesticide use should not act as an obstacle to the timely introduction of pesticide policies.

The following hypothetical example illustrates how the precautionary principle applies to pesticide application in agriculture. Considering the two time period setting, the imposition of a tax or levy scheme to internalize pesticide externalities, thereby leading to socially optimal pesticide use, is not a costless procedure, and its entire regulatory cost creates uncertainty regarding the optimal time of application. In the current period there is uncertainty about the future state of the world. The externalities of pesticides have not been documented fully, nor have the external costs been quantified precisely. Therefore, a policy-maker cannot be sure whether a pesticide tax should be introduced now or later, after further information has been obtained. Imposing a pesticide tax in the current period can be more costly, as there are no precise indicators of the external costs of pesticides. This lack of knowledge may induce a policy-maker to delay intervention and wait to identify the exact external costs of pesticides, where the prices of the different commodities reflect the external costs of pesticide use by imposing a suitable tax in the second period. Therefore, delaying reduces the economic risk of imposing a tax scheme. Nevertheless, acting in the future period can be devastating in terms of biodiversity loss, as there are often difficulties in enhancing biodiversity levels after long periods of intensive agrochemical use (Berendse et al., 1992).

2.5 Pesticide risk valuation

During the last two decades, many attempts have been made to value pesticide risks. The meta-analysis of Florax et al. (2005) and Travisi et al. (2006) provide an overview of the literature on pesticide risk valuation. These analyses find that the literature is diverse, providing WTP estimates not only for various human health risks, but also for environmental risks. However, the majority of studies estimate WTP for the negative externalities on human health., finding great variation in the WTP estimates, as some studies find higher WTP for human safety than environmental quality (Foster and Mourato, 2000), while others show higher WTP for environmental quality than for food safety and human health (Balcombe et al., 2007). This mixed evidence is attributed to the use of different valuation techniques, and to differences among the available biomedical and ecotoxicological data. Foster and Mourato (2000) provide a conjoint analysis of pesticide risks by estimating the marginal value of risk reduction for human health and bird biodiversity. Additionally, Schou et al. (2006) and Travisi and Nijkamp (2008) used a choice experiment approach to estimate the economic value of reduced risks from pesticide use. The latter approach was also used by Chalak et al. (2008), who found high WTP for reduced pesticide use for both environmental quality and consumer health. Moreover, this study indicates the presence of heterogeneous preferences for pesticide reduction in relation to environmental quality and food safety.

2.6 Indirect effects of pesticides

Data on pesticides' indirect effects can enable the development of environmental and health standards, thus favoring the introduction of regulatory schemes that will use economic incentives to attain these standards. Research on pesticide spillovers on human health seems to be more advanced considering the banning of many active ingredients on human health grounds. Sexton et al. (2007) underline the need and difficulty in incorporating and translating pesticide externalities into policy. They also confirm the low level of knowledge on pesticides' environmental effects compared to human health effects. The use of environmental and health standards in EU states' pesticide policies can be depicted in Table 3 which reviews the pesticide policies of different EU countries. Only a few European countries (i.e., Sweden, Norway, France, and Belgium) use environmental and/or health standards to base their pesticide policy tools. Sweden was one of the first countries to introduce a simple tax scheme based on an environmental levy, while Norway uses a tax system where the taxation level is banded by health and environmental properties. In France there are taxes on seven categories of pesticides as non-point sources of pollution, which reflects the differing environmental load of each plant protection product. Belgium has recently introduced a pesticide tax on five active substances which is based on health and environmental risk criteria (OECD, 2008). According to OECD (2008) the use of plant protection products has declined in the abovementioned countries, but it is difficult to separate the impact of taxation on pesticide use from the other factors influencing farmers' use decisions. Moreover, in Norway, the high reductions in pesticide risks should be treated with caution due to the stockpiling of pesticides prior to expected increases of pesticide taxation (OECD, 2008).

Table 3. Pesticide Policies in different European Countries

Country	Description of Pesticide Policy	Values for Pesticide Taxies/Fees/Levies
Sweden	Environmental levy per Kg of active substance	30 SEK/Kg Active Substance (AC) (3.25 €/Kg AC)
Norway	Banded Tax System	Basic Tax: 20 NOK/ha (2.6 €/ha), LT products: 2.6 €/ha, MT: 10.4 €/ha, HT: 20.8 €/ha.[a]
Denmark	-Differentiated pesticide levy. -Overall levy on all pesticides.	Insecticides: 54% of retail price (r.p.), Herbicides/fungicides/growth regulators: 34% of r.p., Wood preservatives: 3% of gross value
Italy	Sales control, Pesticide Tax	0.5% and 1% over the final price of domestic and imported pesticides, respectively.
UK	- Target fee for registration - General fee for industry.	Target fee: 5,000 €, General fee: 5,719 €
Switzerland	- Direct payments, Extra subsidies. - Minimum ecological standards.	-
Finland	Registration charge	840 € + 3.5 % of final price (excluding VAT)
Netherlands	Integrated Crop Protection on certified farms	-
France	Pesticide Tax	Category [a] 1: 0 €/t, 2: 381 €/t, 3: 610 €/t, 4: 838 €/t, 5: 1,067 €/t, 6: 1,372 €/t, 7: 1,677 €/t
Germany	Pesticide Reduction Programme.	-
Belgium	Tax on five active substances [b]	0.395 €/kg

Source: OECD (2008); Hoevenagel et al. (1999); PAN Europe (2005); Parkkinen (2008).
[a] LT, MT and HT denote low medium and high toxicity products, respectively. Pesticides taxes also exist for seed treatment pesticides (1.3 €/ha), concentrated hobby products (130 €/ha), and ready to use hobby products (390 €/ha).
[b] Categories reflect the different environmental load of each plant protection product with 1 and 7 being the lowest and highest toxicity category, respectively.
[c] Based on health and environmental criteria.

2.7 Discussion

The review of the information needed for the introduction of an optimal pesticide policy framework has revealed several knowledge gaps, thus providing useful insights to policy makers. The evidence from studies using the LZ specification is mixed. Overuse or underuse of pesticides may depend on the specification itself but also on the application to different crops. More research needs to be done across EU countries for different crops in order to obtain a clear view of pesticide use trends. Overuse of pesticides implies that policy efforts should focus on decreasing applied quantities while underuse shows that the policy target should not be to reduce pesticide application volume but to stimulate substitution of hazardous products with low toxicity alternatives.

Pesticide productivity varies among others with changes in climatic conditions indicating that taxes on pesticides should be country or region specific. A considerable number of pesticide risk studies opposes the conventional view of pesticides being risk reducing. If pesticides are risk increasing then a pesticide tax (leading to reduced pesticide use), will render agricultural production less risky. Greater dissemination of such scientific findings may increase the effectiveness of pesticide tax and levy schemes. With pesticide demand being in general inelastic, only large pesticide price changes can alter farmers' practices. Considering that high pesticide tax rates may be politically problematic, pesticide taxation might not be considered an effective policy instrument. However, taking into account producers' heterogeneity, economic incentives may still have a role in pesticide policies by encouraging efficiency improvements in pesticide applications or movement to less pesticide intensive forms of cropping (e.g., IPM).

The review of WTP studies has shown that consumers are in general willing to pay to reduce human health and environmental risks from the application of pesticides. This fact favors farmers' switch to IPM or organic agriculture. In this way reductions in pesticide use could be achieved with gains in farm income through conversion to less pesticide-intensive cropping systems. Advice, training and extension in reduced pesticide use practices can encourage farmers' conversion to less pesticide-intensive farming. Subsidizing production on certified farms (e.g., IPM or organic) or promoting self-regulation for pesticide free products may further stimulate farmers to alter their crop protection practices.

As pesticides are not homogeneous goods, pesticide taxation needs classification according to toxic contents (Nam et al., 2007); i.e. higher taxes to be imposed on pesticides that are more harmful to the environment and human health. However, there is no accepted methodology for hazard ranking possibly due to the extensive gaps in knowledge of the environmental impacts of pesticide use. Oskam et al. (1997) strongly encourage the European Commission to adopt a uniform or if possible a differentiated value tax within the EU. Effective pesticide policies should differentiate pesticides according to their health and environmental externalities as current EU pesticide policy (COM, 2006) highlights the importance of reducing risks to both human health and the environment.

More than a decade ago, Hoevenagel et al., (1999) noted the difficulties in discriminating pesticides according to their environmental externalities. Since then, no action has been taken by the EU in stimulating an EU-wide or country specific data collection of pesticide impacts. Zilberman and Millock (1997) argue that the construction of an effective pesticide tax scheme requires rigorous data collection on pesticide use at the farm level. As pesticide application levels and their externalities are very diverse across different regions and under different climatic conditions (Wossink and Feitshans, 2000), country-specific research is of the utmost importance. Pesticide classification through the development of environmental impact-based indicators for each country or region would be important in improving the effectiveness of pesticide policies. The levy systems based on environmental standards used in some countries encompass interesting lessons for an EU wide regulatory framework on ways to charge, collect, differentiate and reimburse the levy. The limited use of the environmental or health standards in national pesticide policies and the small reductions in pesticide use in the countries that use these standards may be attributed to the multidimensionality and lack of data of pesticides' indirect effects. This has led policy makers to be unable to introduce optimal economic incentive-based policies that will not only aim to finance national action plans but will also affect farmers' behavior. Falconer (2002) argues that an effective environmental banding could be based on groups of pesticides with similar hazard scores instead of developing environmental indicators (based on environmental impacts) for each pesticide. A starting point for such a classification could be the development of hazard scores from pesticides' labeling which includes precautions for environmental and human health safety and its mandatory in all

EU member states. Pesticide clusters with higher hazard scores could be prioritized for reductions.

The optimal regulatory strategy does not have to be composed by single policy tools but can involve a mixture of measures and actions such as tax schemes, direct controls, farm certification and self-regulation. In this way the different measures may compensate the deficiencies of each other. Pesticide policies can be coordinated by the EU with taxes or direct controls being country or region specific as pesticides' use and externalities vary regionally due to differences in agronomic characteristics (Schou, 1996). Action taken at EU level implies strong competitive effects between national regulatory systems (Knill and Lenschow, 2005). Integrating economic incentives (defined at the EU level) in existing national regulatory structures may induce strong political pressures on national policy makers to reform these structures. Knill and Lenschow (2005) state that the use of economic instruments at the EU level has been relatively week in comparison to national level, pointing to the required member states' unanimity needed in tax-related decisions as a limiting factor. However, the authors add that such problems may be bypassed by enhanced cooperation among groups of member states.

2.8 Concluding Comments

In an era where existing EU pesticide policies are streamlined and new policies are planned, this study tries to shed light on the optimal pesticide policy framework and examine the elements needed for applying such a framework. An optimal pesticide policy should involve economic incentives based on standards for environmental and health quality. As the introduction of market-based policy instruments is among the future plans of EU policy makers, this study offers some important insights. Inelastic pesticide demand suggests that tax rates should be high while the development of health and environmental standards, where differentiated tax rates can be based on, needs further attention due to inadequate information on pesticide externalities. Evidence from pesticide use trends (overuse or underuse) among different crops and countries and its relation to risk is mixed, implying that further investigation is needed possibly at state level. The great variety of pesticide risks suggests that more primary research is needed. Pesticides affect human beings and other organisms differently and have various environmental effects across countries due to differences in climatic conditions and species richness. Country-specific research on the effects of existing active ingredients on the environment and a comparison with their effects on human health may enable researchers to introduce differentiated fiscal measures, and trigger the chemical industry to develop effective alternatives. As agrochemical innovation is in general complex, costly and time consuming, the development of economic incentive-based policies grounded in the reality of agriculture can foster crop pest agents innovation.

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Chapter 3

Do Farmers Internalize Environmental Spillovers of Pesticides in Production?

This chapter is in review at *Ecological Economics* as:
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Abstract

Pesticides are used in agriculture to protect crops from pests and diseases, with indiscriminate pesticide use having several adverse effects on the environment and human health. An important question is whether the environmental spillovers of pesticides are also affecting the farmers' production environment. A model that explicitly incorporates the symmetric and asymmetric effect of pesticides' environmental spillovers on crop production is used. The application focuses on panel data set from Dutch cash crop producers and the pesticide contribution to biodiversity is found to impact farm output significantly.

Keywords: pesticides, dynamics, biodiversity, The Netherlands.

3.1 Introduction

Pesticides constitute one of the most important inputs in developed countries' arable farming as they are the most common way of controlling pests. There is a large range of positive outcomes from the use of different pesticides related to agricultural productivity. Pesticides can secure farm income by preventing crop losses to insects and other pests, improve shelf life of the produce, reduce drudgery of weeding that frees labor for other tasks, and reduce fuel use for weeding.

But their use raises a number of environmental and health concerns. Indiscriminate pesticide use can lead to off-target contamination due to spray drift with negative effects for biodiversity, bystanders, soil and water courses. Organic compounds of pesticides that are resistant to environmental degradation can contribute to soil contamination and accumulate in human and animal tissue (Ritter et al., 1995). Pesticides can be dangerous to workers, consumers and bystanders. Farm workers lacking the appropriate protective equipment can present irritations, poisonings and even death. Pesticides have been shown to have devastating effects on water organisms (Fairchild & Eidt, 1993), birds (Boatman et al. 2004), non-target beetles (Lee et al., 2001) and bees (Brittain et al., 2009). Agricultural output can be negatively impacted from the above mentioned pesticidespillover effects. Farm operator's health problems can decrease the efficiency of labor while a decreasing biodiversity deprives the farm from beneficial organisms' productive and damage-abating functions. Pollinators like wild bees can increase plant seed set and output quality (Roldan Serrano and Guerra-Sanz, 2006; Morandin and Winston, 2006) while beetles and birds can control pest populations.

An important question is whether farmers are rational using pesticides taking into account the potential future spillover effects. Myopic decision makers ignore the future effects of their decisions (Alix and Zilberman, 2003). Pesticides' environmental spillovers concern common property resources, including biodiversity populations, for which decision makers perceive that their production decisions might not affect the stock of these resources under myopic behavior (Regev et al., 1976; Pemsl et al., 2008). Lack of information on pest and predator populations' growth and the absence of a market price for the environmental effects of pesticides may drive farmers to ignore them in their production process (Feder and Regev, 1975). Harper and

Zilberman (1989) developed a model to compare output and pesticide use under myopic and optimal behaviour reporting that ignorance on pesticides' dynamic effects on pests, may lead to the pesticide treadmill followed by increased pesticide applications and profit loss. In the same line of reasoning Feder and Regev (1975) argue that myopic pesticide use decisions increase rather than decrease pest damage due to the increased impairment of pest predators.

Hall and Norgaard (1973) and Talpaz and Borosh (1974) introduce the concept of damage-abatement input, which suggests that pesticides have an indirect effect on output in future years arising from pesticide resistance in addition to a direct yield-increasing effect in the current period. Apart from pesticides, damage-abatement inputs include natural predators, and cultural practices such as rotation and planting diversionary crops. Regev et al. (1983) developed a more comprehensive bio-economic model to determine optimal pesticide use using detailed entomological information. Their results show that pesticide use does not only decrease pest populations but also increases pesticide resistance, pointing to the importance of such state variables in policy decision making. Drawing inspiration from these bio-economic models as well as on early analysis of self-insurance through expenditure in loss-reducing agents initiated by Ehrlich and Becker (1972), Lichtenberg and Zilberman (1986) developed an output damage-abatement specification for estimating pesticide productivity, with an extensive range of empirical applications undertaken (Babcock et al., 1992; Carrasco-Tauber and Moffit, 1992; Lin et al., 1993; Chambers and Lichtenberg, 1994; Oude Lansink and Carpentier, 2001; Oude Lansink and Silva, 2004, and Guan et al., 2005). However, none of these studies explicitly account for the impact of pesticides' environmental spillovers on output realization. The objective of this paper is to model whether pesticides' environmental spillovers are also impacting agricultural output. For this purpose, a model is employed that accounts for both the symmetric and asymmetric effect of the environmental spillovers of pesticides on output, since pesticides' environmental spillovers can reduce crop pollination and soil nutritional characteristics (through impact on pollinators and soil organisms respectively), increase crop damage by reducing the number of natural predators, and impact negatively the efficiency of labour (health impact on farm operator). As public awareness in Europe is growing regarding the indirect effects of pesticides on human health and the environment, the European Union (EU) is planning to revise its pesticide policy by introducing tax and levy schemes that internalize pesticides' indirect effects and lead to socially optimal pesticide use. The integration of indirect effects of pesticides in farmer's production technology can assist policy makers in designing appropriate pesticide tax policies.

The remainder of the paper is structured as follows. Section 2 presents the theoretical model of optimal pesticide use. Section 3 introduces the model specification followed by the estimation method and data description. Results are analyzed in Section 4, policy implications are discussed in Section 5 and conclusions presented in Section 6.

3.2 Model of optimal pesticide use

We assume that agricultural production is modeled by production function f and damage abatement function m with the following separable structure:

$$y = f(x_{p_t}(PI_{j_t}), q_{k_t}) * m(Z_t, PI_{j_t}) \qquad (1)$$

where a single output is produced, y, using multiple variable inputs (x_p), fixed inputs (q_k) and damage-abatement inputs (Z, pesticides). Pesticides are separated into two categories, $Z=g(Z_l,Z_h)$, where subscripts "l" and "h" indicate low toxicity (LT) and high toxicity (HT) pesticides respectively. The asymmetric specification in (1) implies that the actual output is scaled by the damage abatement; i.e., abatement is actual output/potential output [or y/f(•)]. This is an assumption maintained in the literature frequently following Lichtenberg and Zilberman (1986). Separability is characterized by the independence of the marginal rates of substitution between pairs of inputs from changes in another input (Saha et al., 1997). More specifically the specification in (1) implies that the marginal rate of substitution between all pairs of inputs in x_p and q_k is independent of Z implying the output elasticities of x_p and q_k are independent of Z in the damage abatement function, m. The Pesticide Impacts (PI) reflect impacts on biodiversity and are a function of pesticide use as they are yearly observations of the impacts of the used pesticide products:

$$PI_{j_t} = g_j(Z_{h_{t-1}}, Z_{l_{t-1}}) \tag{2}$$

where the beginning of the year PI_j is a function of pesticides used in the preceding year. With last year's pesticide use impacting production of the current year, the importance of *PI* on the farm decision environment follows from the potential for biodiversity to control pest populations, increase production through crop pollination, and improve soil nutritional characteristics. As the *PI* variables denote pesticide impacts on farmland biodiversity, we expect the impact of pest resistance to be reflected in the evolution of these variables. Increased volume of pesticide applications increase *PI* values which can be associated with increased resistance levels among biodiversity populations. The specification in (2) implies that the state variable PI_j evolves according to $PI_{j_t} - PI_{j_{t-1}} = g_j(Z_{h_{t-1}}, Z_{h_{t-1}}) - g_j(Z_{h_{t-2}}, Z_{h_{t-2}}) - PI_{j_{t-1}}$ which indicates a 100% depreciation rate.[1] As a result, the current period choices of pesticides (Z_l, Z_h) can be fully characterized as a two period optimization problem. The producer's problem is to maximize profits over two time periods subject to the production technology reflecting the damage abatement and the equation of motion linking last year's pesticide use to this year's *PI*.

3.2.1 Model specification

The empirical application of model (1) requires the specification of functional forms for the production function *f(·)* and the damage-abatement function *m(·)*. The Cobb-Douglas specification is used here and has a long history in the literature for ease of estimation in production studies, in general, and for pesticide impact assessment, in particular (Saha et al., 1997; Carpentier and Weaver, 1997; Carrasco-Tauber and Moffit, 1992).

Following Guan et al. (2005) we use the following damage-abatement specification:

[1] This assumption might be somewhat strong, but is imposed just by the construction of *PI* variables where a state of nature impact is absent.

$$m = exp(-A) = exp[-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \sum_j \xi_j PI_j)^2] \quad (3)$$

3

This specification restricts the value of abatement within a sensible region and allows for both positive and negative marginal product of pesticides. It addresses the damage abatement from the use of pesticides, and their environmental spillovers, and allows for interactions among these inputs.

We can conceptualize the decision problem as follows: Producers are trying to maximize their profit by choosing the optimal quantity of variable inputs (x_p) and pesticides (Z_l, Z_h),

$$\underset{y_t, x_{p_t}, Z_{l_t}, Z_{h_t}}{Max} \; p_t y_t - w_{p_t} x_{p_t} - w_{Z_t}(Z_{l_t} + Z_{h_t}) + \rho[p_{t+1} y_{t+1} - w_{p_{t+1}} x_{p_{t+1}} - w_{Z_{t+1}}(Z_{l_{t+1}} + Z_{h_{t+1}})] \quad (4)$$

s.t.

$$y_t = e^{(c_0 + \sum_i^{N-1} c_i * id_i)} x_{1_t}^{(\alpha_1 + \sum_j \alpha_{1j} PI_{j_t})} x_{2_t}^{\alpha_2} q_{1_t}^{\beta_1} q_{2_t}^{\beta_2} q_{3_t}^{\beta_3} \cdot e^{-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \sum_j \xi_j PI_{j_t})^2} \quad (5)$$

and (2).

The subscript *i* indexes each farm , *N* is the number of farms and *ρ* reflects the discount factor.

The solution to this optimization problem leads to the optimal x_1 and x_2:

$$x_{1_t} = \left(\frac{w_1}{p e^{(c_0 + \sum_i^{N-1} c_i * id_i)} (\alpha_1 + \sum_j \alpha_{1j} PI_{j_t}) x_{2_t}^{\alpha_2} q_{1_t}^{\beta_1} q_{2_t}^{\beta_2} q_{3_t}^{\beta_3} e^{-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \sum_j \xi_j PI_{j_t})^2}} \right)^{1 / (\alpha_1 + \sum_j \alpha_{1j} PI_{j_t} - 1)} \quad (6)$$

$$x_{2_t} = \left(\frac{w_2}{pe^{(c_0+\sum_i^{N-1} c_i * id_i)} x_{1_t}^{(\alpha_1+\sum_j \alpha_{1j} PI_{j_t})} \alpha_2 q_{1_t}^{\beta_1} q_{2_t}^{\beta_2} q_{3_t}^{\beta_3} e^{-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \sum_j \xi_j PI_{j_t})^2}} \right)^{1/(\alpha_2-1)} \tag{7}$$

and the optimal pesticide use (e.g for Z_l [2]):

$$\begin{aligned} & -w_{Z_t} - 2(\gamma_1 + \gamma_3 Z_{h_t}) p_t x_{1_t}^{(\alpha_1+\sum_j \alpha_{1j} PI_{jt})} x_{2_t}^{\alpha_2} q_{1_t}^{\beta_1} q_{2_t}^{\beta_2} q_{3_t}^{\beta_3} \\ & \cdot (\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \sum_j \xi_j PI_{j_t}) e^{-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \sum_j \xi_j PI_{j_t})^2} \\ & -2\beta(\gamma_1 + \gamma_3 Z_{h_{t+1}}) p_{t+1} x_{1_{t+1}}^{(\alpha_1+\sum_j \alpha_{1j} PI_{j_{t+1}})} x_{2_{t+1}}^{\alpha_2} q_{1_{t+1}}^{\beta_1} q_{2_t}^{\beta_2} q_{3_{t+1}}^{\beta_3} \\ & \cdot (\gamma_1 Z_{l_{t+1}} + \gamma_2 Z_{h_{t+1}} + \gamma_3 Z_{l_{t+1}} Z_{h_{t+1}} + \sum_j \xi_j PI_{j_{t+1}}) e^{-(\gamma_1 Z_{l_{t+1}} + \gamma_2 Z_{h_{t+1}} + \gamma_3 Z_{l_{t+1}} Z_{h_{t+1}} + \sum_j \xi_j PI_{j_{t+1}})^2} = 0 \end{aligned} \tag{8}$$

Expression (8) implies that the discounted flow of marginal profit arising from current period pesticide use, equals the current cost of applying another unit of pesticide. The rationale of the behavioural model in equation (5) is that each production period starts off with a specific biodiversity status which has been shaped from previous period's pesticide decisions. Then producers decide on the optimal use of pesticides taking into account the impact in the current period and all future periods.

3.2.2 Empirical estimation

The system to be estimated must reflect the pesticide choices with the intertemporal linkages, found in (8), the profit maximizing variable input choices, reflected in (6) and (7), and the technology, in (5). With no closed form solution available for optimal pesticide use, these decisions are approximated by reduced form estimation. As a result, three equations are

[2] The optimal pesticide use for Z_h is derived by replacing (γ_1+$\gamma_3 Z_h$) with (γ_2+$\gamma_3 Z_l$).

estimated simultaneously using 3SLS[3], where y, x_1, x_2, Z_l and Z_h are treated as endogenous variables. The assumption of profit maximization suggests the direct estimation of the production relationship in (5) is compromised by the productive variable inputs, x, being correlated with the disturbance term. The instrumental variables that were used in the estimation to avoid the simultaneous equation bias are the fixed variables (q_1, q_2, q_3), the output and input price indexes, and the quadratic terms of these variables. The instrumental variable specification has been tested using a Hausman test indicating that the 3SLS provides consistent parameter estimates.

The parameters to be estimated are α, β, γ, c and ξ. Variable inputs are denoted as x, with x_1 for fertilizers and x_2 for other inputs. The arguments q_k are fixed inputs, with k=1 for labour, 2 for capital and 3 for land. PI are the impacts of pesticides on various biodiversity categories, with j = "w" for water organisms, "s" for soil organisms, and "b" for biological controllers. The farm-specific dummies are denoted by ci and e is a disturbance term that includes factors that are not accounted for in the model such as stochastic events (e.g. weather) and measurement errors.

The computation of the output elasticities of pesticides' use reflects the impact of pesticide applications on the current period output and next period's output through the PI components of the damage abatement function. Therefore, the overall elasticity for example of LT pesticides is composed of the direct elasticity:

$$\frac{\partial y_t}{\partial Z_{l_t}} \cdot \frac{Z_{l_t}}{y_t} = \frac{\partial \ln m(Z_{h_t}, Z_{l_t}, PI_t)}{\partial \ln Z_{l_t}} \qquad (9)$$

and the future elasticity:

[3] The 3SLS estimation takes place after taking the logarithms of equations (5), (6), and (7) and including random disturbance terms to count for the effects of variables that cannot be taken explicitly into account in the model.

$$\frac{\partial y_{t+1}}{\partial Z_{l_t}} \cdot \frac{Z_{l_t}}{y_{t+1}} = \sum_{j=w,s,b} \frac{\partial m(Z_{h_{t+1}}, Z_{l_{t+1}}, PI_{t+1})}{\partial PI_{j_{t+1}}} \cdot \frac{\partial g_j(Z_{h_{t+1}}, Z_{l_{t+1}})}{\partial Z_{l_{t+1}}} \tag{10}$$

Expression (10) indicates that the effect of present pesticide use is transmitted to future production through next year's *PI*. [4]

3.3 Data

The available data are composed by the Farm Accountancy Data Network (FADN) database and detailed data on pesticide use at the farm level from the Agricultural Economics Research Institute (LEI) for arable farms in the Netherlands. Panel data are available over the period 2002-2007 from 130 farms (514 observations). The panel is unbalanced and on average farms stay in the sample for four to five years.

Variable definitions and summary statistics are provided in Table 1. One output and 8 inputs are specified. The output consists of root crops (potatoes, sugar beets, carrots and onions), cereals (wheat, barley, triticale, corn, oats and rye) and other crops (green beans and peas and grasseed). Output is measured as total revenue from all products, deflated to 2005 values using an index of prices from Eurostat. The inputs were classified as productive inputs and damage-abating inputs. Productive inputs have a direct impact on agricultural output while damage-abating inputs impact output indirectly through the reduction of crop damage. The productive inputs are separated into fixed ones which include land, capital and labour, and variable ones which consist of fertilizers and other specific crop inputs. Land was measured in hectares, capital includes the replacement value of machinery, buildings and installations, deflated to 2005 using a Tornqvist index based on the respective price indices, and labour is measured in annual work units (AWU[5]). Fertilizers were measured as expenditures deflated to 2005 using the fertilizer price index. The "other inputs" variable includes expenditures on energy, seeds and other specific crop costs, deflated to 2005 using a Torngvist index for the disaggregated "other inputs" components. The damage-abating inputs include pesticides. Pesticides were

[4] For the pesticide impact function *g,* a quadratic function was used (i.e., *PI* variables were regressed on the levels, squared terms and cross-product of pesticide inputs).

[5] One AWU is equivalent to one person working full-time on the holding (EC, 2001).

measured as expenditures deflated to 2005 using pesticide price index and divided into LT and HT products based on their environmental impact scores. Finally, the price of pesticides Z_l, Z_h are assumed to be the same. [6]

Table 1. Summary statistics

Variable	Symbol	Number of observations	Dimension	Mean	S.D.
Output	Y	514	1000 Euros	210.77	180.7
Output price	P	514	price index	1.12	0.08
Fertilizers	x_1	514	1000 Euros	11.59	8.85
Other inputs	x_2	514	1000 Euros	58.98	49.06
Labour	q_1	514	Annual work units (AWU)	1.6	0.82
Capital	q_2	514	1000 Euros	335.04	365
Land	q_3	514	Hectares (ha)	88.51	55.75
Low toxicity pesticides	Z_l	514	1000 Euros	18.55	13.5
High toxicity pesticides	Z_h	514	1000 Euros	9.34	7.7
Pesticides' interaction term	Z_{lh}	514	1000 Euros	219.18	430.5
Impact of pesticides on water organisms	PI_w	514	Impact points	5.38	7.53
Impact of pesticides on soil organisms	PI_s	514	Impact points	7.1	10.57
Impact of pesticides on bio-controllers	PI_b	514	Kg*10 of active ingredient	2.96	3.08

3.3.1 Data on Pesticide Impacts (PI)

The available data are obtained from the Dutch Centre for Agriculture and Environment (CLM). For each pesticide that Dutch arable farmers use, there is an environmental indicator which shows the impact on aquatic, surface water organisms (PI_w), terrestrial life (PI_s), and biological controllers (PI_b). The effects of pesticides on water organisms and soil organisms are

[6] Price indexes for LT and HT pesticides are not available and instead we use the price index of all pesticides from EUROSTAT to deflate HT and LT pesticides. The implicit assumption made here is that relative price changes in LT pesticides are the same as relative price changes in HT pesticides.

expressed in environmental impact points. The impact points for water organisms (i.e., aquatic insects) depend on pesticide toxicity and the amount of spray drift to watercourses. The amount reaching a watercourse depends on the application technique. For arable farming the percentage spray drift is 1%.The impact points for soil organisms (i.e., soil insects), are computed based on the organic matter content, pesticide characteristics (degradation rate and mobility in soil) and pesticide toxicity. The organic matter content in conjunction with the pesticide characteristics determine the amount of pesticides that over the course of time remains in the soil. There are five classes of organic matter content with the case study farms belonging to the 3-6% category. Originally, the environmental impact points (for both PI_w and PI_s) are expressed for an application of 1 kg/ha (standard application). To calculate the application specific PI_w and PI_s, the environmental impact points under a standard application are multiplied by the actual applied quantity per hectare (CLM, 2010). The final farm-specific PI_w and PI_s are computed by summing up the impact points of the individual pesticide applications.

The environmental impact points increase when pesticides have a greater impact on the environment. For soil organisms a score of 100 impact points is in line with the acceptable level (AL) set by the Dutch board for the authorization of pesticides (CTB) which reflects the concentration which implicates minor risk for the environment. Since 1995, the AL for aquatic organisms is 10 impact points per application (CLM, 2010).

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The risk for biological controllers (PI_b) (e.g. ladybugs, predatory mites, hymenopteran parasitoids) is indicated in the data ordinally with a symbol.[7] This symbol reflects the suitability for integrated cropping systems and is a combination of all pesticide effects (direct effects, such as mortality or non-hatching of eggs and pupae, have been taken into account as well as indirect effects, such as reduced fertility, repellency, persistence etc.) for individual beneficial organisms. The division of pesticides into LT and HT products is based on their PIs. HT product is characterized by a pesticide where at least one of its PIs exceeds the acceptable

[7] There are four symbols for bio-controllers: symbol 'A' indicates that the pesticide is useful for integrated cropping systems (i.e, no side effects on bio-controllers); symbol 'B' slightly useful (i.e., minor side effects); symbol 'C' not useful (i.e., large side effects); and symbol '?' not well known impact (CLM, 2010). The PI_b variable is a continuous variable that represents the sum of the kilograms of active ingredient of the most hazardous for beneficial organisms applications ("C"). In this way PI_b variable captures the intensity of the most hazardous for bio-controllers applications.

levels [8] set by CTB or belongs to the most hazardous [9] category. On the other hand, LT product is a pesticide that all its PIs are below the acceptable levels or belong to the least harmful categories.

3.4 Results

3.4.1 Pesticide use and environmental spillovers

Data analysis has shown that Dutch cash crop farmers used in total 337 different pesticides. The average pesticide applications and products used per year were 27 and 21, respectively (Figure 1). The sudden increase of pesticide applications in 2003 can be attributed to a 10.4 % increase of fungicides, in comparison to the previous year, that was caused by relatively high temperatures and humidity.

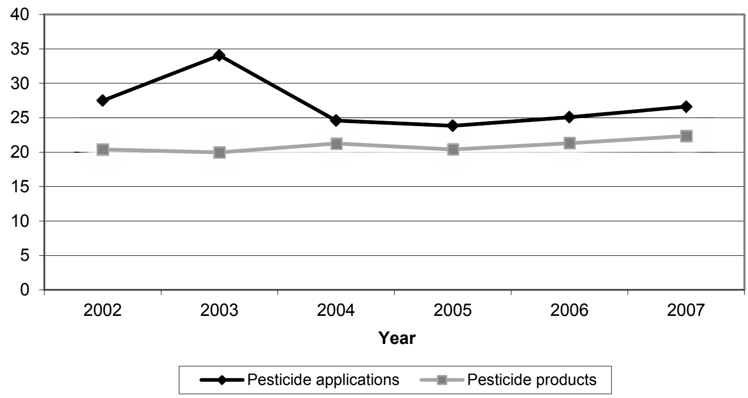

Figure 1. Average pesticide applications and products per year used by the Dutch cash crop farms (2002-2007).

The majority of pesticide applications are in potatoes followed by sugar beet, wheat, onions and barley (Figure 2). Concerning the division of pesticides into LT and HT products, 104

[8] Acceptable levels exist only for PI_w and PI_s.
[9] For PI_b the most hazardous category is considered the “C”.

pesticides (31%) [10] were characterized as HT (Table 2). From the HT ones, the majority are herbicides and fungicides. It is worth noting that the majority of the used insecticides belong to the HT category while in almost all other types of pesticides the LT products have the highest share.

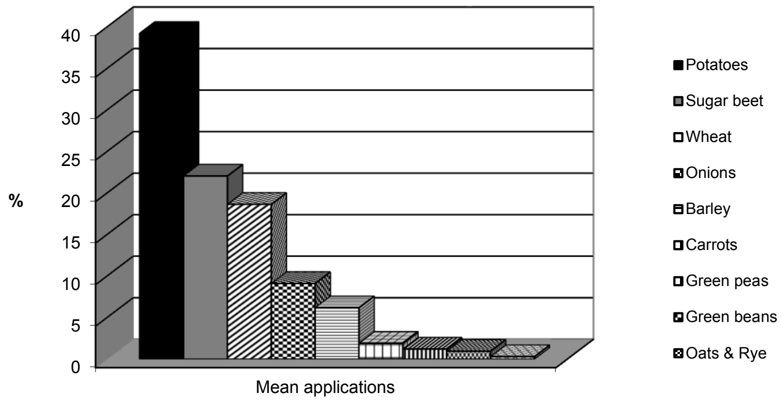

Figure 2. Average pesticide applications (% of number) per year for different cash crops in the Netherlands (2002-2007).

Regarding the *PI* of the pesticide used there are a number of products whose impact on bio-controllers (PI_b) is not well known is denoted as category "?". This category constitutes around 25% of the used plant protection products and indicates that the specific pesticide can be either harmful or harmless for beneficial organisms. The effects of pesticides on beneficial organisms are mainly monitored on indoor crops where Integrated Pest Management (IPM) can be easily applied by the use of natural enemies to reduce harmful insects' populations. It is important to notice here that our data concern arable crops where different pesticide products are applied in comparison to indoor crops. IPM is hardly applied in arable farming, hence the 25% of chemicals used there without information on beneficial organisms' impacts (Moerman, 2009).

[10] Around 87% of the HT pesticides had extreme environmental impact scores (or belonged to the most hazardous category) for more than one *PI*.

Furthermore, research on pesticide impacts on beneficial organisms has mostly focused on insecticides [11] while Dutch arable farmers use mostly herbicides and fungicides.

Table 2. Descriptive statistics of used pesticides.

Category	Total (#)	Percentage of Total	Low toxicity products (#)	High toxicity products (#)
Herbicides	148	43.92	99	49
Fungicides	107	31.76	83	24
Insecticides/Acaricides	32	9.49	9	23
Growth regulators	23	6.83	22	1
Additives (mineral oil)	8	2.37	8	0
Ground Disinfectant	6	1.78	4	2
Unclassified	6	1.78	4	2
Sulfur (Zwavel)	3	0.89	3	0
Rodenticides	2	0.59	1	1
Detergents	2	0.59	0	2
Total	337	100	233	104

3.4.2 Production technology of Dutch cash crop farms

The estimation results of the 3SLS model are presented in Table 3.[12] Farm fixed effects estimates are not presented due to space limitations. The coefficient estimates α_1-β_3 are interpreted directly as elasticities. Most of the productive inputs have a significant impact on production at the 1 or 5% significance level. The significant coefficients of fertilizers (a_1) and other inputs (a_2) indicate that variable inputs do play an important role in crop production.

[11] The focus on insecticides stems from the fact that as this kind of chemicals target harmful for the crop insects, it is probable that they can impact negatively similar organisms like natural enemies and pollinators.
[12] A Cobb-Douglas versus a translog specification was tested and the restrictions of the Cobb Douglas were not rejected (p=0.258).

Table 3. Estimated coefficients of 3SLS system of equations

Parameter	Symbol	Estimate	p-value
Productive inputs			
Fertilizer	α_1	0.053***	0.000
Other inputs	α_2	0.223***	0.000
Fertilizer-PI_w	α_{1w}	0.0001	0.771
Fertilizer-PI_s	α_{1s}	-0.0003***	0.000
Fertilizer-PI_b	α_{1b}	-0.001**	0.011
Labour	β_1	0.188**	0.026
Capital	β_2	-0.093	0.341
Land	β_3	0.517***	0.000
Damage abatement inputs			
Low toxicity pesticides	γ_1	-0.014**	0.016
High toxicity pesticides	γ_2	-0.021	0.190
Pesticides' interaction term	γ_3	0.001	0.180
Pesticide impact on water organisms	ξ_w	0.027**	0.021
Pesticide impact on soil organisms	ξ_s	0.0004	0.968
Pesticide impact on bio-controllers	ξ_b	0.061***	0.000

Note: Fertilizer-PI_w to Fertilizer-PI_b denote pesticide impact on fertilizer use through pressure on water organisms, soil organisms, and biological controllers respectively; (**), and (***), indicate that the estimate is significantly different from zero at the 5 and 1 % significance level, respectively.

The elasticity of other inputs is higher than the one reported by Guan et al. (2005) implying the increasing significance of other inputs in agricultural productivity.[13] The land elasticity is greater than the productive inputs, implying that land is a scarce input that constrains the cash crop sector. Guan et al. (2005) come to a similar conclusion but they report a higher land elasticity.[14] The elasticity of capital is negative but insignificant which shows that the sample farms are overcapitalized in the short run. The elasticities a_{1w} to a_{1b} denote the symmetric effect

[13] e.g. improved seed varieties may increase agricultural productivity in comparison to a decade ago.
[14] The lower estimate of our study is due to an increase of the mean acreage in comparison to the period studied by Guan et al. (2005).

of the *PI* variables on crop production through fertilizers.[15] Increased pesticide pressure on soil organisms and biological controllers seems to have a negative effect on crop yields, possibly through a decrease in soil nutritional characteristics and pollination, respectively.

Concerning the damage abatement inputs, a Wald test of the joint significance of parameters γ_i and ξ_i for i= 1, 2, 3, and j = 1, 2, 3 finds that the null hypothesis is rejected (p=0.000). This indicates the existence of crop losses due to non-optimal production conditions (e.g., pest infestations), thus highlighting the importance of damage abatement inputs in avoiding yield reductions. Farm specific dummies absorb elements that are not modeled directly in this study. These elements can include education, farming experience, and farm soil type. Indeed, around 91% of the farm-specific dummies are significant at the 5% significance level. The mean farm effect is 4.3 (with standard deviation 3.7) and shows how much will the production shift up by the elements that are taken indirectly into account.

The coefficients in the damage abatement function (γ_1-ξ_b) of Table 3[16], individually are not directly interpretable in terms of meaningful relations, elasticity responses are reported in Table 4 which provide further information on the output response to each input and on the economies of scale in the Dutch cash crop sector. Pesticides have a direct impact on production and a future impact. The direct impact of both types of pesticides is positive with HT products having slightly lower impact on production than LT products while their production impact is insignificant. This opposes the conventional view that highly toxic products might be more effective in preventing crop damage. The future impact of both types of pesticides is negative (implying that use of pesticides in the current period impacts negatively next year's output through the *PI*) but insignificant. Concerning the elasticities of *PI*, we can see that they all have a negative impact on output. This finding indicates that water and soil organisms and biological controllers can have a beneficial impact on output by reducing crop damage through the control of pest populations, enhancing soil nutritional characteristics and contributing to increased crop pollination. If farmers increase the pressure on the biodiversity categories (by using pesticides

[15] A model specification where pesticide impacts on biodiversity affect both fertilizers and other inputs has been tested, with the impact of pesticides' indirect effects on other inputs being insignificant.
[16] Prior to estimating the 3SLS model, pesticide and *PI* variables were examined for multicollinearity. The multicollinearity test has shown that no correlation exceeded |0.5|. This was expected as the different *PI* variables are calculated based on the observed pesticide use measured in Kg/ha while the Z variables reflect pesticide expenses measured in Euros.

that increase PI_w, PI_s and PI_b) they will realize some output losses. Another explanation for the negative impact of PI on output realization can be that increased pressure on farmland biodiversity (i.e., higher *PI*) can increase the level of resistance among pest populations and induce some crop damage. It is important to notice that most *PI* elasticities are significant at the 10 % significance level, indicating the important role of pesticides' environmental spillovers on output realization. The input elasticities sum to 0.91 indicating decreasing returns to scale which is consistent with the results reported by Oude Lansink (1997). Guan et al. (2005), in their study on conventional and organic arable farms in the Netherlands, report an elasticity of 0.98 adding that these farms may operate beyond the optimal scale.

The value of the marginal product which is the shadow price of the different inputs can be used to assess whether an input is overused or not. Therefore, the value of the marginal product (VMP) can be used in the design of subsidies or taxes for individual inputs. Table 4 presents the VMP estimates which are computed at the sample means, at average output price index 1.12.[17] The shadow price of labour is 27.06 while a statistical test suggests that it is not significantly different from labour price. Capital investment is realizing a net loss showing that Dutch arable farms are over-capitalized. This finding is consistent with results from Guan et al. (2005) and Guan and Oude Lansink (2003). The VMP of land is 1.47 and is not significantly different from the average rent of land. Therefore, from the productive inputs only capital is used intensively in Dutch arable farms.

The VMP of LT and HT pesticides are 0.25 and 0.23, respectively. A comparison of these shadow values with pesticide prices shows that both LT and HT[18] pesticides are overused. Oude Lansink and Carpentier (2001) report a shadow price of 3.2[19] in their study of Dutch arable farms over the period 1989-1992. The large difference may result from the failure in the latter study to take into account the heterogeneity across farms. Even higher estimates are reported by Oude Lansink and Silva (2004) in a non-parametric study of pesticides use in the

[17] By construction, the estimation using (6) and (7) guarantees that the average VMP of fertilizers and other productive inputs match up with the input price (i.e., the first-order conditions of profit maximization are imposed on the estimation, which means that the VMP of fertilizers and other productive inputs equals the input price).
[18] Using the point estimate of the elasticity of HT pesticides, the VMP is much less than the input price. Considering that this elasticity is not significantly different from zero, implies that HT pesticides are surely overused.
[19] Weighted over 3 types of pesticides (herbicides, fungicides and other pesticides) and 4 different model specifications.

Netherlands over the same period, but the authors add that this may be a result of outliers. Both Oude Lansink and Carpentier (2001) and Oude Lansink and Silva (2004) conclude that almost all pesticides are underutilized, on average, which is a result that is not in line with our finding. Carrasco-Tauber and Moffit (1992) and Chambers and Lichtenberg (1994) also find that pesticides are underutilized in U.S. agriculture. In our study, the average VMP of LT and HT pesticides is 0.24, which is lower than the average pesticide price, suggesting that farmers could increase their profitability by decreasing the use of pesticides.

Table 4. Production elasticities and values of marginal products (VMP in EUR 1,000)

	Elasticities			p-value	VMP	Input price (IP)
	Direct	Future	Overall			
Fertilizer	0.052	-	0.052	0.000	1.127	0.98
Other inputs	0.223	-	0.223	0.000	0.856	0.99
Labour	0.188	-	0.188	0.042	27.06	22.19[a]
Capital	-0.093	-	-0.093	0.427	-0.065	0.09[b]
Land	0.516	-	0.516	0.006	1.471	1.60[c]
LT pesticides	0.020	-0.0003	0.020	0.024	0.254	1.02
HT pesticides	0.010	-0.001[d]	0.009	0.197	0.227	1.02
PI_w	-0.014	-	-0.014	0.048	-0.626	-
PI_s	-0.001	-	-0.001	0.984	-0.019	-
PI_b	-0.019	-	-0.019	0.013	-1.548	-

[a] Labour price is calculated as the average hourly wage of entrepreneurs in 2002-2007 (CBS, 2010)
[b] Capital price is calculated as 10 % of average capital price index.
[c] Land price is computed as the average farmland rent per ha for 2002-2007 (CBS, 2010).
[d] P-values for the future elasticities of low (LT) and high toxicity (HT) pesticides were computed using bootstrapping techniques. The elasticities of HT and LT pesticides were found to be insignificant (i.e., p=0.364 and p=0.209, respectively).

Guan et al. (2005) report a VMP of 1.25 and conclude that pesticides were optimally used at the farm level, but they add that this might lead to an overuse if the indirect effects of pesticides are taken into account. This hypothesis is verified by the current study where the inclusion of pesticides' indirect effects showed that pesticides are on average overused. Overutilization of pesticides is reported by Babcock et al. (1992) in their study on apple farms in North Carolina. The considerable amount of preventive pesticide applications that apple production requires, might be one of the reasons for the reported overutilization.

3.5 Policy implications

The important role of biodiversity in output realization shows a need for protecting farmland organisms. Overuse of pesticides can be associated with decreased numbers of beneficial organisms at a farm level and reduced efficiency of labour due to their health effects. The results of this study can assist policy makers in designing pesticide policies that are based on economic incentives.

Different tax or levy schemes can lead to socially optimal pesticide use. A number of different taxes can be applied including a tax per kilogram of applied active substance, a flat tax on all pesticides, a flat tax on categories of pesticides that differ in terms of toxicity (i.e. LT and HT products), and taxes on *PI*. The flat tax on all pesticides and the tax per kilogram of active substance are easily implemented and involve low transaction costs. However, they do not differentiate between HT and LT products and ignore the impacts of pesticides on the environment and human health. [20] The flat tax on different categories of pesticides takes into account pesticides' environmental spillovers as these categories have been constructed based on each product's toxicity impact on biodiversity. A high tax on HT applications can reduce environmental spillovers as these products contribute more to *PI*s. On the other hand, a lower tax rate can be applied to LT pesticides as the results show that LT products cause less damage and have a slightly higher impact on production. Concerning taxes on *PI*, the tax can be a monetary value per *PI*. For this purpose, it is possible to maintain a threshold level and tax only the PIs above the threshold levels. From the PIs exceeding the threshold levels, PI_b and PI_w can be taxed at a higher rate than PI_s as its increased level impacts significantly (i.e. decrease) agricultural output; i.e., the higher tax for the first two categories is due to the preservation of biological controllers and water organisms is associated with increased farm productivity.

Levy systems can also be used where the revenues collected under a tax is redistributed back to farmers in the form of subsidies or to other involved stakeholders. Subsidies can be direct

[20] A tax per kilogram of applied active substance cannot capture the true environmental/health impact of the applied pesticide as applying less active ingredients does not necessarily mean that the environmental/health impact is automatically smaller.

through the application of a farm subsidy for the use of LT pesticides and/or indirect through resources directed to extension services that will promote information on the existence of LT alternatives, more precise pesticide applications and safe handling of empty pesticide packages. Tax revenues can also be directed to R&D on the development of LT alternatives.

Differentiated tax or levy schemes that reflect the potential environmental and health damage of pesticides are strongly encouraged (Hoevenagel et al., 1999; Oskam et al., 1997). However, concerns are raised over its exact rate and the differentiation procedure in the light of inadequate information on pesticides' indirect effects and the lack of an accepted methodology for hazard ranking (Pretty et al., 2001). Moreover, the design and implementation of differentiated taxes or levies, based on some measure of the hazards related to each pesticide, involves high costs due to high information requirements (Sheriff, 2005).

Despite the efficiency advantages that economic instruments may have compared to command and control approaches (Baumol and Oates, 1988), economic instruments' share in current pesticide policies is relatively small. In Europe, where a few countries have embedded economic incentives in their pesticide policy frameworks, the primary objective of the existing tax schemes is to provide resources for research and extension, rather than influencing user behavior (Wossink and Feitshans, 2000). The collection of detailed data on pesticide use at the farm level and environmental impacts of different pesticides may enable policy makers to introduce optimal pesticide tax and levy schemes. These schemes may alter pesticide decisions at the farm level such that environmental spillovers of pesticides are reduced.

3.6 Conclusions

This study presents a model of optimal pesticide use on specialized cash crop farms in the Netherlands. The inclusion of two pesticide categories that differ in terms of toxicity, and pesticides' environmental spillovers in both the production and the damage abatement specification is an improvement compared to earlier specifications in terms of richness of the results. Shadow prices of pesticides and other inputs are estimated and compared with market prices to assess the degree of over- or under-utilization.

The empirical results indicate that the indirect impacts of pesticides on biodiversity are affecting the farmer's production environment. These results suggest that future pesticide policies should conserve organisms beneficial for the farm, as they protect farm yields from loses through the control of pest populations. The results also show that pesticides are overused on average. Organisms beneficial for the farm can be impacted negatively from the overuse of pesticides. The use of economic incentives like taxes may lead to optimal use of pesticides. When LT alternatives exist for some of the HT products, then taxes can also help switching to the LT category that has a lower impact on biodiversity.

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Chapter 4

Can Economic Incentives Encourage Actual Reductions in Pesticide Use and Environmental Spillovers?

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Abstract

Chemical pesticides constitute an important input in crop production. But their indiscriminate use can impact negatively agricultural productivity, human health and the environment. Recently, attention is focused on the use of economic incentives to reduce pesticide use and its related indirect effects. The aim of this work is to assess the effectiveness of different economic instruments such as taxes and levies in encouraging farmers to decrease pesticide use and their environmental spillovers. A policy simulation model is employed using data from Dutch cash crop producers including two pesticide categories that differ in terms of toxicity and pesticides' environmental spillovers. Four different instruments were selected for evaluation: pesticide taxes, price penalties on pesticides' environmental spillovers, subsidies, and quotas. The results of the study indicate that even high taxes and penalties would result in a small decrease in pesticide use and environmental spillovers. Taxes that differentiate according to toxicity do not lead to substitution of high with low toxicity pesticides. Subsidies on low toxicity products are not able to affect the use of high toxicity products. Pesticide quotas are more effective in reducing pesticide use and environmental spillovers.

Keywords: pesticides, economic instruments, The Netherlands

4.1 Introduction

Pesticides are integral components of modern crop production systems. However, excessive pesticide use has a negative impact on a large number of dimensions such as contamination of surface and ground water, soil, food, biodiversity and human health (Pimentel et. al., 1992; Pimentel and Greiner, 1997; Wilson & Tisdell 2001). Reducing the applied pesticide quantities or using less toxic products are among the most challenging environmental policy objectives. The challenge in achieving these objectives is to maintain a balance between the continued contribution of agriculture to production and greater human health and environmental protection.

Recently, particular attention is given to the role of market mechanisms in achieving environmental policy aims, especially through the introduction of economic incentives. The European Union's (EU) pesticide policy envisages the use of pesticide tax and levy schemes (EC, 2007). Economic instruments such as taxes and subsidies may guide farmers toward pest management strategies which are more in line with society's concerns for sustainable agriculture. Although the environmental economics literature suggests that economic instruments may have efficiency advantages compared to command and control approaches (Baumol and Oates, 1988), the economic instruments' share in current pesticide policies is relatively small. In Europe, only a few countries have embedded economic incentives into their pesticide policy frameworks. However, in many cases the primary objective of the existing tax schemes is to provide resources for research and extension, rather than influencing user behavior (Wossink and Feitshans, 2000). The design and implementation of an efficient system of pesticide taxes and levies requires information on pesticide use, demand, and the risk and toxicity characteristics of the used products to account pesticides' indirect effects (Hoevenagel et al, 1999; Oskam et al, 1997).

Knowledge of the relationship between input applications and environmental damage is a key element in designing environmental taxes (Falconer, 1998). However, the design and implementation of differentiated taxes or levies based on some measure of the hazards related to each pesticide involves significant costs due to demanding information requirements (Sheriff, 2005). Another difficulty facing policy makers in designing pesticide tax and levy schemes is whether to tax pesticide use or price. For example, a tax per kilogram of applied active substance cannot capture the true environmental or health impact of the pesticide applied, since applying less active ingredients does not necessarily mean that the environmental or health impact is smaller automatically. A low application of a highly toxic product may still cause more environmental or health damage than a high dose of a less toxic product. On the other hand, a pesticide price tax may not yield the desired effects when pesticides' demand elasticity is low or when the older and more hazardous pesticides are a cheaper alternative.

Concerning the environmental effectiveness of pesticide tax schemes, an important issue is how to tailor the burden of the tax to the potential damage of a pesticide. Falconer (1998) points that market mechanisms can have a share in pesticide policy but their environmental effectiveness depends on their careful design. Differentiated taxes that can somehow reflect the potential environmental and health damage of pesticides are strongly encouraged (Pretty et al., 2001; Hoevenagel et al., 1999; Oskam et al., 1997). However, concerns are raised over their exact rate and the differentiation procedure in the light of inadequate information on pesticides' indirect effects and the lack of an accepted methodology for hazard ranking.

Empirical evidence from the introduction of an ad valorem tax in arable farming in South Central Texas and Alabama using aggregate state level data, reveals considerable decreases in pesticide usage but output supply is affected in different ways (Shumway and Chesser, 1994; Chen et al., 1994).[1] Wossink et al. (2001) examine the effects of a pesticide tax for the total Dutch arable farming sector and report that reductions differ considerably among different types of pesticides. Despite the importance of a farm-level approach on primary policy analysis, little empirical research has been done on investigating the impact of different economic instruments on farm income, pesticide use, and environmental spillovers. Falconer and Hodge (2001) examine the linkages between the multidimensionality of ecological problems and the complexities associated with policy design by using farm-level data for a typical arable farm in the UK. Their modeling framework considers four economic incentive-driven policy instruments in an effort to identify the possible trade-offs between reductions in

[1] In South Central Texas, a 25% tax on pesticides yields considerable decreases in output supply while an 1% tax in Alabama yields mild impacts.

environmental damages and the income of farmers.[2] Their results find that different pesticide tax schemes have different impacts in terms of both magnitude and direction, which suggests that compromises need to be made in environmental policy or additional policy instruments should be introduced.

The aim of this study is to assess the effectiveness of different fiscal measures in reducing pesticide use and environmental spillovers by using detailed farm level data from Dutch arable crop production. An important feature of this work is the examination of subsidies, price penalties on pesticides' environmental spillovers, and quotas as additional or an alternative pesticide use policy tool. Another important aspect of this study is that it employs a dynamic perspective addressing the current production impact (through reducing crop damage in the current period) and the future impact through pesticides' environmental spillovers (e.g., impacting the farm biodiversity which alters the future production environment). Hall and Norgaard (1973) and Talpaz and Borosh (1974) were the first to introduce the concept of damage-abatement input, which suggests that pesticides have an indirect effect on output in future years arising from pesticide resistance in addition to a direct yield-increasing effect in the current period. Feder and Regev (1975) developed a theoretical dynamic pest management model incorporating entomological knowledge in their model specification. Their results show that the absence of information on pest and predator populations' growth and the absence of a market price for the environmental effects of pesticides may drive farmers to ignore them in their production process. Using a more comprehensive bio-economic model of optimal pesticide use that employed detailed entomological information, Regev et al. (1983) show that pesticide use does not only decrease pest populations but also increases pesticide resistance. Drawing inspiration from these bio-economic models as well as on early analysis of self-insurance through expenditure in loss-reducing agents initiated by Ehrlich and Becker (1972), Lichtenberg and Zilberman (1986) developed an output damage-abatement specification for estimating pesticide productivity that treats damage abatement inputs in a different manner than regular inputs and serves as the foundation for an extensive range of empirical applications (Babcock et al., 1992; Carrasco-Tauber and Moffit, 1992; Lin et al., 1993; Chambers and Lichtenberg, 1994; Oude Lansink and Carpentier, 2001; Oude Lansink and Silva, 2004, and Guan et al., 2005). With none of these studies explicitly accounted for the impact of pesticides'

[2] Falconer and Hodge (2001) focus on an ad valorem tax, a fixed levy per spray unit, a levy per kilogram of active ingredient, and a levy-based on pesticide hazard.

environmental spillovers on output realization, this paper uses a model that explicitly incorporates the asymmetric effect of pesticides' environmental spillovers on crop production and tests whether economic incentives can alter pesticide decisions at the farm level such that environmental spillovers of pesticides are reduced.

The structure of the paper is as follows. Section 2 presents the theoretical model of optimal pesticide use. Section 3 addresses specification issues and presents the empirical model followed by a description of the applied pesticide quotas and tax and levy schemes. Data description takes place in section 4. Results are presented in Section 5 and discussion and conclusions in Section 6 and 7, respectively.

4.2 Theoretical model

The production technology is expressed by the following separable specification motivated by Lichtenberg and Zilberman (1986):

$$y = f(x_{p_t}, q_{k_t}) * m(Z_t, PI_t) \qquad (1)$$

which indicates output, y, is the product of the production technology, *f(·)*, and the abatement technology, *m(·)*.

The specification reflects that a single output is produced, *y*, using multiple variable inputs (x_p), fixed inputs (q_k) and damage-abatement inputs (*Z*, pesticides). Pesticides are separated into two categories, $Z=(Z_l, Z_h)$, where subscripts "l" and "h" indicate low toxicity and high toxicity pesticides, respectively. The Pesticide Impact (*PI*) variable reflects impacts of pesticide use on water organisms and is related to pesticide use as:

$$PI_t = g(Z_{h_{t-1}}, Z_{l_{t-1}}) \qquad (2)$$

where the beginning of the year PI_t is a result of pesticides used in the preceding year. With last year's pesticide use impacting production of the current year, the importance of PI on the farm decision environment follows from the potential for biodiversity to control pest populations and increase production through crop pollination.[3] The producer maximizes profits over two time periods subject to the production technology reflecting the damage abatement and the equation of motion linking last year's pesticide use to this year's pesticide impact on water organisms.

4.3 Application

4.3.1 Specification issues

The production technology in (1) is specified as:

$$f(x_{p_t}, q_{k_t}) = e^{c_0} x_{1_t}^{\alpha_1} x_{2_t}^{\alpha_2} q_{1_t}^{\beta_1} q_{2_t}^{\beta_2} q_{3_t}^{\beta_3} \tag{3}$$

Different specifications for the damage abatement function are available in the literature. Among the specifications that can accommodate the output-reducing nature of damage abatement (i.e., by constraining the value of the abatement function to the [0, 1] interval) are the exponential, Weibull, Pareto, and logistic.[4] Guan et al. (2005) provide an extensive discussion of the properties and problems associated with different damage abatement specifications. In this study we employ the exponential damage abatement specification:

$$m(Z_t, PI_t) = 1 - \exp(-A) = 1 - \exp(-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \gamma_4 PI_t)) \tag{4}$$

[3]Among the aquatic insects can be Coleoptera (e.g., beetles), Diptera (e.g., flies), Lepidoptera (e.g., moths), Hymenoptera (e.g., wasps) and other orders (Williams and Feltmate, 1992). Coleoptera and Diptera are considered to be primitive pollinators while most Lepidoptera and many Hymenoptera feed extensively on floral nectar (Kevan and Baker, 1983).

[4] Carrasco-Tauber and Moffitt (1992) use the exponential, Weibull and logistic damage abatement specifications to obtain pesticide productivity estimates and find that an exponential damage abatement specification led to substantially different estimates of pesticide productivity than the alternative specifications. However, a statistical test for identifying the most appropriate distribution could not discriminate the exponential as superior to alternative specifications.

This specification is used often in the literature (Lichtenberg and Zilberman, 1986; Carrasco-Tauber and Moffit, 1992; Oude Lansink and Carpentier, 2001) and allows for the regular interpretation of the percentage reduction of damage. The pesticides' interaction term is used to address the issue that pesticide categories may be perfect substitutes.

After defining the production and damage-abatement function, we conceptualize the problem of profit maximization as:

$$\underset{y_t, x_{p_t}, Z_{l_t}, Z_{h_t}}{Max} \; p_t y_t - w_{p_t} x_{p_t} - w_{Z_t}(Z_{l_t} + Z_{h_t}) + \rho[\, p_{t+1} y_{t+1} - w_{p_{t+1}} x_{p_{t+1}} - w_{Z_{t+1}}(Z_{l_{t+1}} + Z_{h_{t+1}})] \qquad (5)$$

s.t. (1) and (2) and ρ is the discount factor.

Each production period starts off with a specific biodiversity status which has been shaped from previous period's pesticide decisions. Then producers decide on the optimal use of pesticides taking into account the effect in the current period and the future periods. The solution to this optimization problem leads to the optimal x_1 and x_2:

$$x_1 = \left(\frac{w_1}{pe^{(c_0 + \sum_i^{N-1} c_i * id_i)} \alpha_1 x_{2_t}^{\alpha_2} q_{1_t}^{\beta_1} q_{2_t}^{\beta_2} q_{3_t}^{\beta_3} (1 - e^{-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \gamma_4 PI_t)})} \right)^{1/(\alpha_1 - 1)} \qquad (6)$$

$$x_2 = \left(\frac{w_2}{pe^{(c_0 + \sum_i^{N-1} c_i * id_i)} x_{1_t}^{\alpha_1} \alpha_2 q_{1_t}^{\beta_1} q_{2_t}^{\beta_2} q_{3_t}^{\beta_3} (1 - e^{-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \gamma_4 PI_t)})} \right)^{1/(\alpha_2 - 1)} \qquad (7)$$

and the optimal pesticide use is:

$$\begin{aligned} & - w_{Z_t} + (\gamma_i + \gamma_3 Z_{i_t}) p_t e^{(c_0 + \sum_i^{N-1} c_i * id_i)} x_{1_t}^{\alpha_1} x_{2_t}^{\alpha_2} q_{1_t}^{\beta_1} q_{2_t}^{\beta_2} q_{3_t}^{\beta_3} e^{-(\gamma_1 Z_{l_t} + \gamma_2 Z_{h_t} + \gamma_3 Z_{l_t} Z_{h_t} + \gamma_4 PI_t)} \\ & + \beta(\gamma_i + \gamma_3 Z_{i_{t+1}}) p_{t+1} e^{(c_0 + \sum_i^{N-1} c_i * id_i)} x_{1_{t+1}}^{\alpha_1} x_{2_{t+1}}^{\alpha_2} q_{1_{t+1}}^{\beta_1} q_{2_t}^{\beta_2} q_{3_{t+1}}^{\beta_3} e^{-(\gamma_1 Z_{l_{t+1}} + \gamma_2 Z_{h_{t+1}} + \gamma_3 Z_{l_{t+1}} Z_{h_{t+1}} + \gamma_4 PI_{t+1})} = 0 \end{aligned} \tag{8}$$

The subscript i indexes each farm and N is the number of farms. Expression (8) implies that the current cost of applying another unit of pesticide, equals the discounted flow of marginal profit arising from current period pesticide use.

4.3.2 Empirical model

With no closed form solution for Z_l and Z_h, the econometric estimation focus on the optimality conditions in (6) and (7) for the variable inputs, the production technology in (1) [using (3) and (4)] and the instruments for modelling the pesticide decisions. This system is estimated using 3SLS recognizing the endogenous variables are y, x_1, x_2, Z_l, and Z_h, and the instruments are the fixed inputs (q_1, q_2, q_3), output and input price indices, and quadratic terms of these variables. We also allow for fixed farm effects. Lastly, the pesticide impact function, in (2) is specified by the quadratic expression:

$$PI_t = c + \delta_1 Z_{h_{t-1}} + \delta_2 Z_{l_{t-1}} + \delta_3 Z_{h_{t-1}} Z_{l_{t-1}} + \delta_4 Z_{h\,t-1}^2 + \delta_5 Z_{l\,t-1}^2 \tag{9}$$

The computation of the output elasticities of pesticides' use reflects the impact of pesticide applications on the current period output and next period's output through the pesticide impact component of the damage abatement function. Therefore, the overall elasticity for example of low toxicity pesticides is composed of the direct elasticity:

$$\frac{\partial y_t}{\partial Z_{l_t}} \cdot \frac{Z_{l_t}}{y_t} = \frac{\partial \ln m(Z_{h_t}, Z_{l_t}, PI_t)}{\partial \ln Z_{l_t}} \tag{10}$$

and the future elasticity:

$$\frac{\partial y_{t+1}}{\partial Z_{l_t}} \cdot \frac{Z_{l_t}}{y_{t+1}} = \frac{\partial m(Z_{h_{t+1}}, Z_{l_{t+1}}, PI_{t+1})}{\partial PI_{t+1}} \cdot \frac{\partial g(Z_{h_{t+1}}, Z_{l_{t+1}})}{\partial Z_{l_{t+1}}} \tag{11}$$

Expression (11) indicates that the effect of present pesticide use is transmitted to future production through next year's pesticide impact on water organisms. Following the econometric estimation, a dynamic optimization of the model in (5) is performed.

4.3.3 Pesticide taxes, levies and quotas

After dynamic optimization takes place, three pesticide taxing options are explored: a) the same tax rate on both types of pesticides; b) a tax rate that differentiates according to pesticides' toxicity; and, c) a price penalty on pesticide impact on water organisms. The first type of tax is achieved by increasing proportionally the price of both types of pesticides and involves price increases of 20, 80, and 120%. A differentiated tax rate places a greater price penalty on high toxicity chemicals. The scenario explored under this tax scheme places a 100% tax on high toxicity products and no tax on low toxicity products. Introducing a greater tax rate on high toxicity products is expected to encourage farmers to reduce their use and increase the use of low toxicity chemicals. A price penalty on pesticide impact on water organisms penalizes farms with higher hazard score for water organisms. Penalizing pesticide impacts on water organims is not as straightforward as the previous taxing options due to the absence of prices for the pesticides' environmental spillovers. However a suitable price for pesticide impact on water organisms can be retrieved through comparison of the tax revenues from different scenarios.[5]

Under a levy scheme, tax revenues can be redistributed back to farmers in the form of subsidies. The idea is to affect farmers' production decisions concerning pesticide use (i.e., to reduce pesticide use and their negative impacts) without decreasing farm income. Three

[5] A suitable price is the one that when penalizing pesticide impacts on water organisms yields comparable penalty/tax revenues with the previous tax schemes.

different subsidies are examined in this study: a) subsidy on the use of low toxicity pesticides (Z_l), b) subsidies on research and development (R&D) of low toxicity alternatives, and c) subsidies on R&D of more environmental friendly pesticides. Subsidies on the use of low toxicity pesticides is proxied by reducing their price in the simulation process. This price reduction can be financed by tax revenues and is expected to trigger producers to increase the use of low toxicity products. A 20% price decrease is tested in this case. The second type of subsidy concerns R&D of low toxicity products and is proxied by improving low toxicity pesticides' productivity. As the simulation model of this study spans the short run and R&D impacts productivity in the long run (Alston et al., 2009), a 2% productivity increase will be tested.[6] Thirtle et al. (2008) in a study that examines the relationship between total factor productivity (TFP) and public and private research, report that public and private R&D in conjunction with farm size explain 2% of the TFP variance. Improving low toxicity pesticides' productivity is expected to lead to lower use of low toxicity products and even trigger farmers to substitute some high toxicity with low toxicity products. Tax revenues can be also directed to the development of more environmental friendly products. This type of subsidy is expected to yield significant reductions in pesticides' environmental burden. Finally, scenarios for pesticide quotas are also employed in this study with producers facing 10 and 20% reductions in pesticide use.

4.4 Data

Panel data for 2003-2004 from 55 farms, provided by the Dutch Agricultural Economics Research Institute (LEI), are used in the simulation model. Variable definitions and summary statistics are provided in Table 1, with one output and 8 inputs being specified. The output consists of wheat, potatoes, sugar-beet, onions, and carrots and is measured as total revenue deflated to 2005 prices using an index of prices from Eurostat. The inputs are classified as productive inputs and damage-abating inputs. The productive inputs are separated into variable inputs which consist of fertilizer and other crop-specific inputs, as well as fixed inputs which include land, capital and labour. Land is measured in hectares, capital includes the replacement value of machinery, buildings and installations deflated to 2005 using a Tornqvist index, and labour is measured in annual work units (AWU[7]). Fertilizers were measured as expenditures deflated to 2005 prices using a fertilizer price index while "other inputs" variable includes

[6] This is achieved by decreasing the initial level of low toxicity pesticides by 2%.
[7] One AWU is equivalent to one person working full-time on the holding (EC, 2001).

expenditures on energy, seeds and other crop-specific costs deflated to 2005 using a Torngvist index. The damage-abating inputs include pesticides, measured as expenditures deflated to 2005 using a pesticide price index. Pesticides are separated into low toxicity and high toxicity products based on their impact on water organisms.[8] The pesticide impact variable provided by the Dutch Centre for Agriculture and Environment (CLM), reflects pesticide impacts on water organisms (mainly aquatic insects). It is expressed in impact points and takes into account the toxicity of pesticide used and the spray drift [9] to watercourses which depends on the application technique (CLM, 2010). Originally, the impact points are expressed for an application of 1 kg/ha (i.e., standard application). To calculate the application specific impact on water organisms the environmental impact points under a standard application are multiplied by the actual applied quantity per hectare (CLM, 2010). The final farm-specific impact on water organisms is computed by summing up the impact points of the individual pesticide applications.

Table 1. Summary statistics (in EUR 1,000, deflated to 2005 prices)

Variable	Symbol	Number of observations	Mean	S.D.
Output	y	231	301.91	191.91
Fertilizers	x_1	231	16.77	9.75
Other inputs	x_2	231	91.61	51.59
Labour	q_1	231	1.89	0.92
Capital	q_2	231	419.03	260.76
Land	q_3	231	94.91	57.07
Low toxicity pesticides	Z_l	231	21.81	11.99
High toxicity pesticides	Z_h	231	11.65	7.32
Impact of pesticides on water organisms	PI	231	0.88	0.82

[8] Pesticides that exceed the acceptable level (under a standard application) for water organisms set by CTB (Dutch board for the authorization of pesticides) were characterized as highly toxic pesticides.
[9] In arable farming the percentage spray drift is 1% (CLM, 2010).

4.5 Results

4.5.1 Pesticide contribution to output and water organisms

Parameter estimates from estimating the production and pesticide impact functions are reported in Tables 2 and 3. [10] [11] The coefficient estimates α_1-β_3 are interpreted directly as elasticities while the Z_l, Z_h, and PI output elasticities are computed at their respective sample means.

Table 2. Estimated coefficients of 3SLS system of equations

Parameter	Estimate	p-value
α_1	0.042***	0.000
α_2	0.252***	0.000
β_1	0.062**	0.014
β_2	0.112	0.102
β_3	0.480**	0.001
γ_1	0.494**	0.031
γ_2	0.025**	0.001
γ_3	0.0036**	0.004
γ_4	1.617**	0.036

α_1 denotes fertilizers and α_2 other inputs; β_1 to β_3 denote labour, capital, and land, respectively; γ_1 to γ_4 denote high toxicity pesticides, low toxicity pesticides, their interaction term, and pesticide impact on water organisms respectively; (**) and (***) indicate that the estimate is significantly different from zero at the 5 and 1 per cent significance level, respectively.

The coefficients in the damage abatement function (γ_1 through γ_4) of Table 2 individually are not directly interpretable but elasticity responses are reported in Table 4 which provide further insight on the output response to each input. Pesticide output elasticities are 0.002 and 0.0004 for high toxicity and low toxicity products, respectively. The contribution of pesticides to output realization through the reduction of crop damage appears minimal. However, their preventative role in the damage abatement process and their capacity to reduce output variability can explain why farmers keep using them.

[10] Parameter estimates come from a longer panel (2003-2007) that includes the data used in the simulation model.
[11] The Translog is more flexible than the Cobb Douglas, but is not used in this study as a statistical test has shown that that the parameter estimates of the Translog function are jointly not significantly different from zero.

Table 3. Estimated coefficients of pesticide impact function (PI)

Parameter	Estimate	p-value
δ_1	0.012*	0.087
δ_2	0.011*	0.092
δ_3	-0.0007	0.326
δ_4	0.0005	0.4767
δ_5	-0.00004	0.863

Note: δ_1 denotes high toxicity pesticides (Z_h), δ_2 low toxicity pesticides (Z_l), δ_3 the interaction term of the two pesticide categories, and δ_4 and δ_5 the squared terms of Z_h and Z_l respectively; (*) indicate that the estimate is significantly different from zero at the 10 per cent significance level.

Upon evaluating the pesticide impact on biodiversity variable, we find that an increased pressure on water organisms (i.e., through increased use of pesticides that impact these organisms negatively) increases output as these organisms may cause some crop damage. This is contrary to our assumption that water organisms may contribute positively to crop production through control of pest populations or increased pollination. However, the impact of water organisms on output [12] is quite small and both types of pesticides have a negative but small impact on water organisms. High toxicity pesticides have a slightly higher contribution to water organisms compared to low toxicity pesticides (Table 3). This is in line with the conventional view that high toxicity pesticides may cause more damage to biodiversity.

[12] A model that takes into account both the symmetric and asymmetric effect of pesticides' environmental spillovers on output has been tested, but the effect of the pesticide impact on water organisms variable on both variable inputs was found to be insignificant.

Table 4. Production elasticities of inputs

	Elasticities			p-value
	Direct	Future	Overall	
Fertilizer	0.042	-	0.042	0.000
Other inputs	0.252	-	0.252	0.000
Labour	0.062	-	0.062	0.014
Capital	0.112	-	0.112	0.102
Land	0.480	-	0.480	0.001
HT pesticides	0.002	$2*10^{-6}$	0.002	0.039
LT pesticides	0.0004	$2*10^{-7}$	0.0004	0.005
PI	0.0003	-	0.0003	0.045

Note: HT and LT denote high and low toxicity pesticides, respectively. *PI* denotes pesticide impacts on water organisms.

4.5.2 Pesticide tax scenarios

Table 5 presents the base scenario ("policy-off" scenario), which demonstrates the crop production decisions of profit-maximizing decision makers, and six different tax scenarios where tax rates are applied both to different pesticide categories and impacts. A general conclusion upon comparing the different tax scenarios with the optimal solution is that pesticide demand is highly inelastic.[13] This is in line with findings of price demand elasticity for the Netherlands reported in the literature (Oskam et al., 1992; Oude Lansink and Peerlings, 1996).

[13] The demand elasticity for low toxicity and high toxicity pesticides is -0.03 and -0.0003 respectively. These elasticities are much smaller than former estimates for The Netherlands. Oude Lansink and Peerlings (1996) and Oskam et al. (1992) report pesticide demand elasticities of -0.48 and -0.12, respectively. The large differences among the estimates obtained under this study and the pre-mentioned studies can be attributed to the use of different modelling framework (dynamic vs static, farm level data vs aggregate data) and the increasing importance of pesticides in Dutch agriculture in comparison to a decade ago.

Table 5. Simulation results

Scenarios	Profit	Z_{HT}	Z_{LT}	Fertilizer	Other inputs	PI		
Policy-off scenario[a]	160.56	10.56	18.84	8.12	45.65	0.71		
				Δ (%)				
	Profit	Z_{HT}	Z_{LT}	Fertilizer	Other inputs	PI	Total Z	Tax revenues[b]
20% tax on both Z_{HT} & Z_{LT}	-4.04	-0.09	-0.60	-1.66	-1.44	-0.12	-0.69	38.36
80% tax on both Z_{HT} & Z_{LT}	-16.03	-0.38	-2.54	-6.74	-6.46	-0.52	-2.92	57.15
120% tax on both Z_{HT} & Z_{LT}	-21.81	-0.51	-3.39	-9.35	-9.13	-0.67	-3.90	69.65
100% tax on Z_{HT} & 0% on Z_{LT}	-6.96	-0.03	-0.21	-0.79	-0.76	-0.04	-0.25	22.81
€10 price penalty on PI	-6.09	-0.10	-0.71	-0.04	-0.25	-0.15	-0.81	47.66
€25 price penalty on PI	-12.56	-0.22	-1.22	-1.74	-2.36	-0.28	-1.44	59.54

Note: Z_{HT} and Z_{LT} stand for high and low toxicity pesticides respectively; PI is the next period pesticide impact on water organisms, measured in impact points; The changes in profit and all inputs are mean values of the average farm per year.
[a] Mean values in € 1000.
[b] Average in € 1000 for the two-year period.

More explicitly, a 20% flat tax on both types of pesticides reduces overall pesticide use less than 1%. Increasing the tax rate (for both high toxicity and low toxicity products) to 80 and 120%, total pesticide use is decreased by almost 3 and 4%, respectively. These scenarios show that even high taxes are not able to achieve significant reductions in pesticide use. Moreover, high taxes decrease farm revenues as the 4% pesticide decrease is accompanied by a 22% decrease in farm revenue. Producers' rigidity in reducing pesticide use, thus avoiding the tax burden, may be attributed to the damage preventing role of pesticides and their capacity to reduce output variability. A differentiated tax rate for high toxicity and low toxicity products did not reveal any substitution between the two types of pesticides, which is contrary to the hypothesis formed in section 4. The absence of low toxicity alternatives may explain farmers' rigidity in switching to these products.

Price penalties on pesticide impact on water organisms seem to yield small decreases in overall pesticide use. This is because in practice pesticide impact on water organisms may be reduced not only from decreased pesticide applications but also from a series of measures that a farmer can adopt such as more precise application techniques or being adjacent to water aquifers buffer zones. A price penalty on pesticide impact on water organisms of €10 and €20 reduces revenues by 6% and 12%, respectively.

In all tax scenarios, fertilizers and other inputs are also decreased with the pesticide tax application. Concerning fertilizers, the 80 and 120% flat taxes yield considerable decreases in fertilizer use (approximately 7 and 9%, respectively), while other inputs' reductions range from 0.2 to 9%. Interestingly, these results suggest that there are potential trade-offs between pesticide use and the use of productive inputs. A pesticide tax may incentivize farmers in finding ways to reduce pesticide use, such as switching to crops that are less input intensive for both pesticides and productive inputs. Also, farmers may reduce the use of fertilizers to make pesticides more effective; for example, a reduction of N-fertilizer makes herbicides more effective (Oude Lansink and Silva, 2004). Reduced or more precise fertilizer applications may be viewed as a pesticide reduction factor as excessive fertilizer use may result in increased presence and growth of non-target species (e.g., weeds). Moreover, reduced use of pesticides or fertilizers leads to less use of spraying equipment and as a result less use of energy. Another common characteristic of the presented scenarios is the small reductions in pesticide impact on water organisms, which can be explained by the small decreases in the use of both high toxicity and low toxicity products' observed in all tax and levy scenarios. Finally, the direction of changes in pesticide use is the same for both pesticide categories. A potential explanation is that different pesticides need to be applied in combinations for better control of crop damage. Therefore, reducing the use of low toxicity pesticides may also involve reductions in the use of high toxicity products.

4.5.3 Levy schemes

Table 6 presents different subsidy schemes which can be financed by tax revenues. When subsidizing low toxicity products by decreasing their price by 20%, then the use of these products increases by 0.6% while high toxicity use does not change significantly. Total pesticide use is increased less than 1 per cent and pesticide impact on water organisms increases marginally. In practice, low toxicity and high toxicity pesticide use are linked as farmers use combinations of different toxicity pesticides to reducing crop loss.

Table 6. Subsidies

Scenarios	Profit	Z_{HT}	Z_{LT}	Fertilizer	Other inputs	PI	
Policy-off scenario[a]	160.56	10.56	18.84	8.12	45.65	0.71	
				Δ (%)			
	Profit	Z_{HT}	Z_{LT}	Fertilizer	Other inputs	PI	Total Z
Decrease of Z_{LT}'s price by 20%	1.63	0.09	0.64	1.72	1.69	0.13	0.73
Increase of Z_{LT}'s productivity by 2%	-0.24	-0.01	-2.02	-0.12	-0.13	-0.35	-2.03
Z_{HT} & Z_{LT} contribute 10% less to PI	-0.41	-0.01	-0.05	-0.12	-0.34	-11.05	-0.06
Z_{HT} & Z_{LT} contribute 20% less to PI	-0.49	-0.03	-0.09	-0.21	-0.48	-24.15	-0.12

Note: Z_{HT} and Z_{LT} stand for high and low toxicity pesticides respectively; PI is the pesticide impact on water organisms measured in impact points; The changes in profit and all inputs are mean values of the average farm per year.[a] Mean values in €1000.

An R&D-led increase of low toxicity pesticide's productivity of 2% leads to a 2% decrease in the use of low toxicity products while the use of high toxicity products does not change significantly. The increased productivity of low toxicity pesticides enables farmers to use less of these products in their effort to tackle crop damage. With the use of high toxicity products being hardly affected suggests that farmers are not willing to substitute them with low toxicity pesticides which may be still considered as more effective in reducing crop damage. Small changes are also observed for farm profit, productive inputs and pesticide impact on water organisms. Finally, a 10 and 20% R&D-led decrease in the impact of high toxicity and low toxicity products on water organisms, causes insignificant changes in the use of pesticides, variable inputs and profit. However, pesticide burden on water organisms is decreased by 11 and 24%.

4.5.4 Quotas

The effects of a 10 and 20% cut in pesticides use on farm profit, variable inputs and pesticide impact on water organisms are presented in Table 7. Both quotas yield marginal decreases in farm profit (0.8% for the 10% quota and 1.1% for the 20% quota). Concerning the use of variable inputs, a 10% quota contributes to the reduction of fertilizer and other inputs by 1.8 and 2%, respectively. When reducing pesticide use to 20%, the use of fertilizers is reduced by 2.3% while other inputs' use decreases by 2.5%. As farmers are faced with reductions in pesticide use, they can be encouraged to apply less fertilizers to prevent the growth of non-target species that then require increased pesticide applications. The decreased use of other inputs under a pesticide quota can be attributed to less use of spraying equipment and therefore decreased use of energy. Both quotas yield considerable reductions in pesticides' contribution to water organisms (2.2 and 4.4% under a 10 and 20% cut in pesticides use, respectively). The fact that pesticide impact decreases are higher under the introduction of quotas in comparison to most of the pre-tested tax and levy schemes is attributed to the high reductions in pesticide use (10 and 20% for both high toxicity and low toxicity pesticides). The use of especially high toxicity pesticides is hardly affected in most of the tax and levy schemes. Therefore, quotas can be viewed as a suitable instrument for reductions in pesticide impacts on water organisms. Oude Lansink (1994) examines the effects of a 10% pesticide quota on input quantities and profit using data from specialized Dutch arable farms, and reports minor decreases in profit which is in line with our findings. A decrease of 5% is reported for fertilizer use which is higher than our finding, while the use of other inputs is increased by around 1% which is contrary to our results. A cut in pesticide use may encourage some farmers to increase the use of other inputs by investing in pest resistant seeds. Such a substitution effect is absent from this study.

Table 7. Pesticide quotas

	Δ (%)					
	Profit	Z_{HT}	Z_{LT}	fertilizers	Other inputs	PI
Pesticide quota (-10%)	-0.84	-10.00	-10.00	-1.86	-2.06	-2.28
Pesticide quota (-20%)	-1.12	-20.00	-20.00	-2.28	-2.49	-4.41

Note: Z_{HT} and Z_{LT} stand for high and low toxicity pesticides respectively; PI is the pesticide impact on water organisms, measured in impact points; The changes in profit and all inputs are mean values of the average farm per year.

4.6 Discussion

Economic incentives are absent from many European countries' pesticide policies, including the Netherlands. This study provides empirical evidence from the application of pesticide tax and levy schemes in Dutch arable farming and proposes potential policy considerations. Several studies on the effects of agricultural input taxes in the EU demonstrate that high tax rates should be applied to attain a desirable reduction of pesticide use (Oskam et al., 1997; Nam et al., 2007). The dilemma inherent in pesticide taxation is that the use of pesticides may be so essential for some crops or regions that tax rates would have to be very high to impact pesticide use. This could result in a major reduction in farm income as depicted through the pesticide tax scenarios presented in this work. This study provides new information on the impacts of different tax and levy systems to Dutch and EU policy makers in the absence of empirical research in the Netherlands on the effectiveness of different economic instruments on pesticide use at the farm level. Results show that even high (and politically challenging) tax rates would result in a small reduction in the use of pesticides due to the rigidity of Dutch farmers in reducing pesticide use. As pesticides are non-homogeneous goods, the ideal taxation requires classification according to toxic contents. The differentiated tax scheme simulated in this study finds that a higher tax rate on high toxicity products does not yield any substitution between low toxicity and high toxicity products. Detailed data on pesticides' environmental spillovers at the farm level are recently available and the findings coming from their use in empirical work require further evaluation before they applied to pesticide policy. The small decrease in pesticide use resulting from the different tax schemes and the absence of substitution effects between high toxicity and low toxicity products yield minor decreases in pesticide impact on water organisms. Therefore, a pesticide tax as a stand-alone measure is ineffective, when taking into account the small decreases in environmental pressure and the fact that high and possibly politically problematic tax rates are needed to achieve considerable pesticide reductions. An example in the literature coming to similar conclusions is Falconer and Hodge (2001) who find that an ad valorem tax in UK arable farming is ineffective in achieving pesticide use or hazard reduction goals. Other instruments such as bans on some high toxicity pesticides or improved farmers' training on more precise applications can play an important role in reducing pesticides' environmental spillovers. While pesticide taxes are not effective in reducing pesticide use and indirect effects, they have secondary environmental advantages arising from decreased fertilizer use that can lead to fertilizer contamination reductions. Moving to the penalties on the environmental impacts of pesticides, this study shows that high penalties on pesticide impacts on water organisms should be applied to achieve considerable pesticide reductions. However, penalties on pesticide impact on water organisms may encourage farmers to increase the precision of pesticide applications or to avoid spraying the adjacent to aquifers strips.

The use of tax revenues is often subject to considerable public discussion. Different subsidies can have different impacts in pesticide use and environmental spillovers. When the primary policy objective is to reduce pesticide's indirect effects, R&D on more environmental friendly products can decrease pesticides' environmental burden significantly. A decrease of low toxicity pesticides' price and an R&D-led increase in these pesticides' productivity yield insignificant changes in the use of high toxicity products. Farmers' rigidity in reducing the use of high toxicity products or substituting them with low toxicity alternatives is a common feature in all the tax and subsidy scenarios examined in this study. Farmers' reluctance to reduce the use of high toxicity products may be explained by a) their beliefs about the effectiveness of high toxicity products in preventing crop damage and reducing output variability, b) the lack of low toxicity alternatives, and c) the fact that potatoes in the case study employed here is one of the most profitable and pesticide-intensive crops.[14] [15]

An important question is whether any pesticide policy tool out-performs the others. In terms of total reductions in pesticide use and environmental spillovers, pesticide quotas perform better than taxes or subsidies.[16] But pesticide taxes and subsidies can also play an important role in pesticide policy frameworks. Taxes can have positive environmental side effects (decrease fertilizer use) while tax revenues are important for funding research and extension. Subsidies on the development of more environmental friendly products can reduce pesticides' environmental damage considerably. In general, it is unlikely that a single instrument will solely address any set of pesticide policy goals. The effectiveness of single economic instruments may be improved by education and extension. For instance, extension can render taxes that differentiate according to toxicity more effective by informing farmers on the use of low toxicity substitutes; also the availability of substitutes for pesticides and the application of new crop varieties that are less susceptible to diseases can make economic instruments more effective. Falconer and Hodge (2000) argue that education and training should coexist with economic incentives in an effective pesticide policy. Archer and Shogren (2001) assess the effectiveness of different policy tools in reducing pesticide runoff and point that risk-indexed taxes can be an effective tool in reducing groundwater exposure. As in our case, this study finds that no single policy tool dominated the other options and proposes a set of policy tools including different tax schemes and bans that can lead to the desired policy goals. A package of measures can enable policy makers to tackle and adjust individual measures' infeasibilities.

One of the shortcomings of this work is that this modelling framework does not account for the effect of pesticide impact on water organisms variable on other farms' production environment. Pesticide decisions in individual farms may impact biodiversity populations on farms that

[14] Potatoes account for 56 and 67 per cent of total pesticide applications and HT applications, respectively.
[15] An important question is whether the character of the empirical results vary when investigations employ greater variation in farms according to their size or other criteria. Results are not expected to vary significantly, as the farms employed in this study are relatively homogenous in that these are all arable farms and their main crop is potatoes which is one of the most profitable and pesticide intensive crops.
[16] Except in the case where R&D subsidies on environmental friendly products result in the highest decrease of pesticides' burden for water organisms.

operate in the same region. This is not possible to be captured in our model as the pesticide impact on water organisms values are farm specific and have no measurable impact on other farms. Another shortcoming of this work is that we do not account explicitly for the effect of pest resistance in our model due to the absence of detailed information on pest populations. In our case, we draw on the presence of the environmental variable (PI) which reflects impacts on biodiversity, and we expect the impact of pest resistance to be reflected in the evolution of this variable.

4.7 Conclusions

This study presents a simulation model of Dutch cash crop producers that explicitly accounts for the effect of pesticides' environmental spillovers on crop production and examines the impact of different economic instruments in pesticide use and environmental spillovers. The empirical results indicate that pesticide taxes as a single instrument can be characterized as ineffective since they yield small decreases in pesticide use and environmental spillovers. Differentiated pesticide taxes do not yield substitution of high toxicity with low toxicity products, pointing either to the importance of high toxicity products in agricultural production or the lack of effective low toxicity alternatives. However, the importance of taxes in a pesticide policy relies on their capacity to raise tax revenues that can finance subsidy schemes. Subsidies on low toxicity pesticides hardly affected the use of high toxicity products while R&D of more environmental friendly products contributed to considerable hazard reductions. These findings provide new information to EU policy makers by showing that no single tax or levy instrument can lead to a substantial reduction of pesticide use. A pesticide policy combining different economic incentives may better address the desired policy goals. Command-and-control measures can also have a share in a pesticide policy framework as this study has shown that pesticide quotas are more appropriate in reducing pesticide use and environmental spillovers in comparison to most of the employed pesticide tax and levy schemes. As EU pesticide policy looks to move toward the use of economic incentives, policy makers can benefit from research on the effectiveness of different economic instruments in different EU countries or regions where agronomic and environmental characteristics vary significantly.

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Chapter 5

Pesticide Dynamics, Biodiversity, and Production Uncertainty: Measuring Performance of Dutch Arable Farms

This chapter is in review at *European Journal of Operational Research* as:
Skevas, T., A., Oude Lansink, and S.E., Stefanou. "Pesticide dynamics, biodiversity, and production uncertainty: measuring performance of Dutch arable farms."

Abstract

Pesticides' dynamic effects and production uncertainty play an important role in farmers' production decisions. Pesticides have a current production impact through reducing crop damage in the current period and a future impact through impacting the farm biodiversity which alters the future production environment. This study presents the difference in inefficiency arising from models that ignore the dynamic effects of pesticides in production decisions and the impact of production uncertainty. A dynamic Data Envelopment Analysis model is applied to outputs, inputs, and undesirables of Dutch arable farms over the period 2003-2007. A bootstrap approach is used to explain farmers' performance, providing empirical representations of the impact of stochastic elements and the state of nature on production. These empirical representations are used to adjust outputs, inputs and undesirables to account for the effect of production uncertainty. Finally, the dynamic DEA model is applied to adjusted outputs, inputs and undesirables. We find that efficiency increased dramatically when a production technology representation that considers both pesticides' dynamic impacts, and production uncertainty is adopted.

Keywords: Data envelopment analysis; pesticides; biodiversity; systems dynamics; production uncertainty.

5

5.1 Introduction

Agricultural production is a dynamic process that takes place under a stochastic decision environment. The dynamics of agricultural production technologies are impacted by pesticide use, as pesticides may impact production in the current period by reducing pest damage and in next period through their negative impact on beneficial for the farm organisms. Among the current period on-farm benefits of pesticide use are the improved shelf life of the produce, reduced drudgery of weeding by freeing labor for other tasks, and reduced fuel use for weeding and invasive species control (Cooper and Dobson, 2007). But these benefits can be off-set to some degree by the off-farm costs imposed by pesticides on the environment and human health, such as contamination of surface and ground water, soil, food, biodiversity and human poisonings (Pimentel et. al., 1992; Pimentel and Greiner, 1997; Wilson & Tisdell 2001). Pesticides influence biodiversity by negatively impacting water organisms (Fairchild & Eidt, 1993), birds (Boatman et al. 2004), non-target beetles (Lee et al., 2001) and bees (Brittain et al., 2009), thus depriving the farm from beneficial organisms' productive and damage-abating functions. More specifically, beetles and birds can control pest populations while pollinators like wild bees can increase plant seed set and output quality (Roldan Serrano and Guerra-Sanz, 2006; Morandin and Winston, 2006).

Unpredictable or extreme climatic conditions can lead decision makers to make different production choices. O' Donnell et al. (2010) find efficiency evaluation may lead to biased efficiency estimates when production uncertainty is not taken into account whether based on data envelopment analysis (DEA) or stochastic frontier (SFA) models. Chambers et al. (2011) employ a DEA model incorporating climatic variables to account for the stochastic nature of agricultural production, showing that efficiency results change dramatically when acknowledging stochastic elements. As unpredictable weather conditions and pesticides' environmental spillovers can cause crop losses and/or reduce the quality of output, farmers often use risk management tools to secure farm profit, including production-oriented risk management techniques (e.g., crop diversification, fencing, windbreaks and protective nets) and market-oriented tools (e.g., crop premiums covering farm risks such as flood, fire, third-party liability and crop loss).

In an era of increasing awareness on pesticides' environmental spillovers and ways of minimizing them, information on the environmental efficiency of polluting inputs is useful in the context of maintaining output levels while improving environmental quality. Several attempts have been made in the literature to measure efficiency in the presence of undesirable outputs. Among the employed methods are parametric output and input distance functions (Färe et al. 1993; Coggins and Swinton, 1996; Hailu and Veeman, 2000) and DEA methods (Färe et al. 1989; Ball et al., 1994; Färe et al., 1996; Tyteca, 1997; Boyd and McClelland, 1999; Reinhard et al., 2000; Hailu and Veeman, 2001; Oude Lansink and Silva, 2003; Färe et al., 2004). Färe et al. (1989) proposed an approach allowing for an asymmetric treatment of desirable and undesirable outputs. Undesirable outputs are treated as weakly disposable while desirable outputs are strongly disposable (Färe et al., 1989). Weak disposability means that reducing (increasing) undesirable outputs (inputs) is not a costless procedure. On this basis, a number of studies have proposed the use of directional distance functions as a tool for modelling production in the presence of undesirables (Chung et al.,1997; Ball et al, 2001). A directional distance function efficiency measure allows for a simultaneous expansion of desirable outputs and reduction of inputs and/or undesirable outputs based on a given direction vector (Chung et al., 1997).

There are a several studies in the literature focusing primarily on assessing environmental and/or technical efficiency of pesticides using DEA (Oude Lansink and Silva, 2004; Wossink and Denaux, 2006).[1] However, none of these studies address the dynamic effects of pesticides and the stochastic nature of production.

This study aims to investigate the performance of Dutch arable farms by using a Russell type of measure to identify technical inefficiency and pesticides' environmental inefficiency specifying the environmental impacts of pesticides simultaneously as undesirable inputs and outputs. A dynamic perspective is employed addressing pesticides' current production impact

[1] Oude Lansink and Silva (2004) present a non-parametric production analysis of pesticides use on specialized cash crop farms in the Netherlands and show that pesticides are in general under-utilised. Wossink and Denaux (2006) assess technical, environmental and cost efficiency of pesticides for transgenic and conventional cotton growers in North Carolina by means of DEA and investigate the determinants (farm characteristics and environmental variables) that explain efficiency using a Tobit regression. However their study to explore the factors that might explain efficiency is an invalid approach as DEA efficiency estimates are serially correlated as shown by Simar and Wilson (2007).

and future impact through pesticides' environmental spillovers. We implement the Simar and Wilson (2007) double-bootstrap procedure to explain technical inefficiency using socioeconomic and environmental variables, thus providing empirical evidence for the design of pesticide policy measures and use the results of the double-bootstrap procedure to adjust firms' outputs and inputs to incorporate production uncertainty in efficiency evaluation.

The rest of the paper continues with Section 2 containing the methodology, while Section 3 contains the definition and sources of the data used. In Section 4, the empirical results from the analysis are presented and discussed, and finally Section 5 concludes.

5.2 Methodology

5.2.1 Measuring inefficiency

Let M, V, and F be the maximum number of outputs, variable [2] inputs, and fixed inputs respectively, used in each year t of the production process. Considering a set of firms I, the production process at time t, P_t uses variable inputs x_{iv}^{t} and fixed inputs q_{if}^{t} to produce outputs y_{im}^{t}. To produce y_{im}^{t} an indirect effect related to pesticide use EI_{ij}^{t}[3] is also produced with j denoting the index set for environmental impacts. The indirect effect EI_{ij}^{t} reflects the impact of pesticides on different biodiversity categories and is assumed to impact the production process P in the next year ($t+1$) as beneficial for the farm organisms can decrease pest damage through the control of pest populations and increase crop pollination. Figure 1 illustrates the dynamic production technology corresponding to the basic dynamic technology proposed by Färe et al., (2007), where variable and fixed inputs in year t are used to produce output in year t and environmental impacts that will be taken into account in the production technology of year $t+1$. We can assume that the dynamic effects influence the target periods only (i.e., no compound effect exists). [4]

[2] Includes other inputs, fertilizers, and pesticides.
[3] The acronym *EI* denotes environmental impacts of pesticides.
[4] The full depreciation assumption is actually imposed just by the construction of *EI* variables where a state of nature impact is absent.

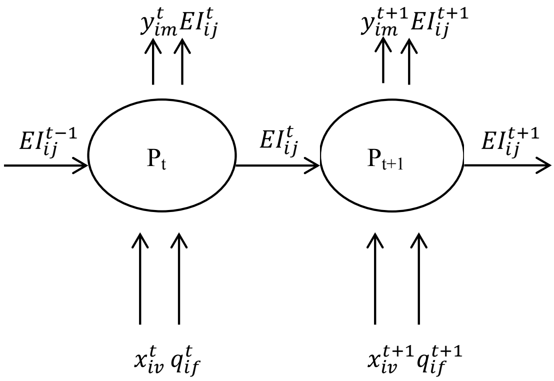

Figure 1. Dynamic production network

The directional technology distance function in the presence of undesirable outputs seeks to increase the desirable outputs while simultaneously reducing the undesirable outputs and variable inputs:

$$\vec{D}_o(x, q, y, EI^t, EI^{t-1}; g) = sup\{\beta: (y, EI^t, EI^{t-1}, x) + \beta g \in \Psi\} \qquad (1)$$

where g is the vector of directions in which desirable and undesirable outputs and variable inputs can be scaled. Expression (1) seeks for the maximum attainable expansion of desirable outputs in the g_y direction and the largest feasible contraction of undesirables and variable inputs in $-g_{EIt}$, $-g_{EIt-1}$ and $-g_x$ direction, respectively (Chung et al., 1997). The latter are negative to reflect that undesirables and inputs are being reduced. In this study undesirable outputs and inputs are modelled as weakly disposable outputs and inputs, implying that reducing pesticides' environmental spillovers is not costless. Assuming weak disposability of undesirables (outputs and inputs) and fixed inputs, a model that decomposes technical inefficiency of the different inputs and outputs for each firm i, i = 1, … ,N, is as follows:

$$\vec{D}^t_o\big(x, q, y, EI; g_y, -g_{EI^t}, -g_{EI^{t-1}}, -g_x\big) = Max_{\beta^t \lambda^t,}\{\beta^t_1 + \beta^t_2 + \beta^t_3 + \beta^t_4\}$$

s.t.

$$\sum_{i=1}^{I^t} \lambda_i^t y_{im}^t \geq y_m^t + \beta_1^t g_y^t \quad \text{(i)}$$

$$\sum_{i=1}^{I^t} \lambda_i^t EI_{ij}^t = \sigma(EI_j^t - \beta_2^t g_{EI_{ij}^t}^t) \quad \text{(ii)}$$

$$\sum_{i=1}^{I^t} \lambda_i^t x_{iv}^t \leq x_v^t - \beta_3^t g_x^t \quad \text{(iii)} \qquad (2)$$

$$\sum_{i=1}^{I^t} \lambda_i^t EI_{ij}^{t-1} = \sigma(EI_j^{t-1} - \beta_4^t g_{EI_{ij}^{t-1}}^t) \quad \text{(iv)}$$

$$\sum_{i=1}^{I^t} \lambda_i^t q_{if}^t = \sigma q_f^t \quad \text{(v)}$$

$$N1'\lambda = 1\ ,$$

$$\lambda_i^t \geq 0\ ,$$

$$0 < \sigma \leq 1$$

Each computed value of β provides the maximum expansion of desirable outputs and contraction of undesirable outputs if a firm has to operate efficiently given the directional vector g. The interaction between time periods comes through the *EIs*. A separate intensity vector is calculated for each year, indicating the role that each observation i plays in determining the set frontier. Free disposability of crop outputs and variable inputs throughout the production process is imposed through constraints (i) and (iii). Constraints (ii), (iv), and (v) reflect weak disposability of environmental impacts and fixed inputs. λ is a N*1 vector of intensity variables (firm weights), while the constraint $N1'\lambda = 1$ allows for a variable returns to scale technology (VRS). The scaling parameter σ is selected such that there is a feasible solution of the DEA problem with weakly disposable fixed inputs and undesirables under variable returns to scale.

The Russell type of model presented above, aggregates both output and input inefficiencies in the framework of a radial measure, thus accounting simultaneously for the inefficiency in both inputs and outputs. In Figure 2, four farms are observed represented by points A, B, C, and D. The DEA technology is the set of all inputs and outputs bounded by the line AB and the horizontal extensions from A and B. The Pareto-Koopmans' efficient subset is represented by the line AB. The farms C and D produce inside the frontier and are technically inefficient. In the case of farm D its projection lies within the Pareto-Koopmans' efficient subset while farm C projects at point E which is outside the efficient subset AB. Therefore, on optimality a radial measure would produce a slack (EA) that is different from zero.

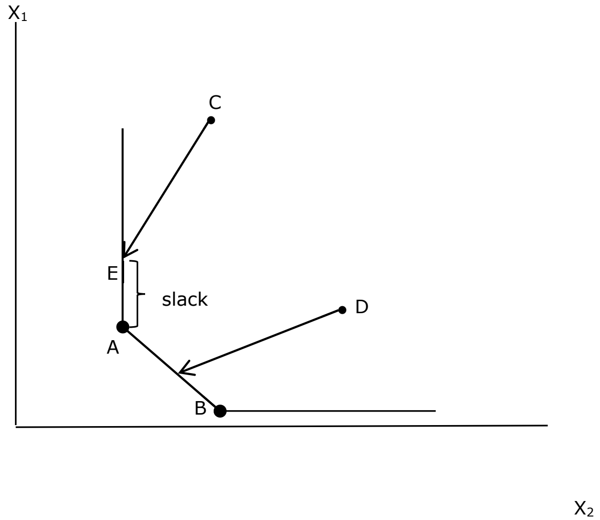

Figure 2. Russell graph measure of technical efficiency.

In an effort to identify the importance of including the *EI*s in the DEA model and the dynamic nature they introduce to the arable production framework, a model which ignores *EI*s is estimated and compared with the initial model in (2). The model specification is as follows:

$$\vec{D}_o^t(x, q, y, EI; g_y, -g_x) = Max_{\beta^t \lambda^t,}\{\beta_1^t + \beta_2^t\}$$

s.t.

$$\sum_{i=1}^{I^t} \lambda_i^t y_{im}^t \geq y_m^t + \beta_1^t g_y^t \quad \text{(i)}$$

$$\sum_{i=1}^{I^t} \lambda_i^t x_{iv}^t \leq x_v^t - \beta_2^t g_x^t \quad \text{(ii)} \qquad (3)$$

$$\sum_{i=1}^{I^t} \lambda_i^t q_{if}^t = \sigma q_f^t \quad \text{(iii)}$$

$$N1'\lambda = 1 ,$$

$$\lambda_i^t \geq 0 ,$$

$$0 < \sigma \leq 1$$

Traditional DEA models that ignore the dynamics of a production process can provide a biased indication of resource efficiency. The comparison of the results of the initial model in (2) with those of the model in (3), can provide further insight into the magnitude of the bias of the inefficiency results.

5.2.2 Factors that influence arable farmers' performance

We make use of the Algorithm 2 procedure proposed by Simar and Wilson (2007), based on truncated regression and bootstrapping techniques, to explain output, undesirables and variable inputs inefficiency. Using γ_i to denote the inefficiency score of farm i, and z_i to denote the vector of producer-specific and environmental variables, a regression can be specified as:

$$\gamma_i = \delta z_i + \varepsilon_\iota \qquad \iota=1\ldots,I, \qquad (4)$$

where δ is the vector of parameters to be estimated and ε is an error term. The unobserved γ_i in (3) is replaced by its bootstrap-based, bias corrected estimate, denoted $\hat{\gamma}_i$ obtained in stage one. Given the directional distance function approach in (2) and the fact that both sides of (4) are bounded by zero, the distribution of ε is restricted by the condition $\varepsilon_\iota \geq 0 - \delta z_i$. The distribution of ε_i is assumed to be truncated normal, with zero mean, unknown variance, and left truncated at point $0 - \delta z_i$ (Simar and Wilson, 2007). Next step requires the use of the following truncated regression model for the stage two analysis:

$$\hat{\gamma}_i = \delta z_i + \varepsilon_\iota \qquad (5)$$

where $\varepsilon_i \sim N(0, \sigma_\varepsilon^2)$. The parameter estimates from (5) and the original estimates are used to construct estimated confidence intervals of δ and σ_ε^2. The selection of variables that explain farms' performance includes farm-specific variables such as age, crop subsidies and rotation, and environmental variables such as soil type, precipitation, temperature, sunshine duration and biodiversity populations.

5.2.3 Adjusted DEA

In this third stage of the analysis, producer's outputs and inputs are adjusted to account for the impact of the different stochastic variables that comprise farms' operating environments and the state of nature (i.e., random statistical noise). We expand on Fried's et al. (2002) approach by taking the results of the double-bootstrap approach used in the previous stage of the analysis to avoid problems of bias in the estimations.[5] This method allows the incorporation in the analysis of the production effects of stochastic elements such as climatic variables and the state of nature (statistical noise), thus accounting for a wide representation of production uncertainty.[6] Adjustment of outputs and inputs takes into account the fact that some producers may operate in relatively unfavorable production conditions, contributing to higher inefficiency scores in the initial DEA evaluation. The extent to which each producer has been disadvantaged by unfavorable production conditions is revealed by the parameter estimates obtained in each truncated regression. The desirable outputs of producers that have been advantaged by each source are adjusted downwards while the undesirables (outputs and inputs) and variable inputs are adjusted upwards. Let weather related variables used in the truncated regression of the double-bootstrap process and their parameter estimates be denoted as $\dot{z}_i$ and $\dot{\delta}$, respectively. Equations (6), (7), and (8) show how producers' adjusted desirable outputs, variable inputs and undesirables are constructed:

$$y_i^A = y_i - \left[max_i\{\dot{\delta}\dot{z}_i\} - \dot{\delta}\dot{z}_i\right] - [max_i\{\varepsilon_i\} - \varepsilon_i] \qquad (6)$$

[5] Simar and Wilson (2007) have noted that Frieds' et al. (2002) approach of regressing radial and non-radial slacks on environmental variables is inappropriate as the dependent variables are functions of estimated efficiencies that are serially correlated.
[6] Cordero et al. (2008) provide a broader picture of the advantages and disadvantages of approaches that incorporate exogenous factors in efficiency evaluation.

$$x_i^A = x_i + \left[max_i\{\dot{\delta}\dot{z}_i\} - \dot{\delta}\dot{z}_i\right] + \left[max_i\{\varepsilon_i\} - \varepsilon_i\right] \tag{7}$$

$$EI_i^A = EI_i + \left[max_i\{\dot{\delta}\dot{z}_i\} - \dot{\delta}\dot{z}_i\right] + \left[max_i\{\varepsilon_i\} - \varepsilon_i\right] \tag{8}$$

where y_i^A , x_i^A, EI_i^A are adjusted desirable outputs, variable inputs and undesirables[7] respectively, while y, x and EI, are observed desirable output, variable input and undesirable quantities, respectively. The first adjustment on the left side of the equations puts all producers on a common operating environment, the least favorable weather conditions observed in the sample, while the second adjustment on the right side of the equations puts all producers into a common state of nature, the worst case situation encountered in the sample. Therefore, producers farming under relatively good production conditions have their desirable outputs (undesirable outputs and inputs, and variable inputs) adjusted downward (upward) by a relatively large amount, while producers experiencing relatively bad production conditions have their desirable outputs (undesirable outputs and inputs, and variable inputs) adjusted downward (upward) by a relatively small amount. The initial DEA model in (2) is re-estimated after replacing the observed output and input data with the adjusted ones. Comparing the results of the adjusted with the initial DEA model can shed light on farmers' performance when unobserved heterogeneity arising from production uncertainty is taken into account in the modelling framework.

5.3 Data

Data on specialized arable farms covering the period 2002-2007, were obtained from a stratified sample of Dutch farms which kept accounts on behalf of the farm accounting system of the Agricultural Economics Research Institute (LEI). The panel is unbalanced and on average farms stay in the sample for four to five years. The data set used for estimation contains 703 observations from 188 farms. Table 1 reports the mean values of the data.

[7] Equation 8 reflects the adjustment of undesirable ouputs. Adjustments in undesirable inputs can be obtained by replacing EI with EI^{t-1} in both the left and right side of equation (8).

Table 1. Variables and descriptive statistics.

Variable	Dimension	Mean	S.D.
Output	1000 Euros	191.33	191.35
Fertilizer	1000 Euros	10.52	8.45
Other	1000 Euros	59.11	58.53
Labor	Annual working units	1.74	1.01
Capital	1000 Euros	366.92	384.78
Land	Hectares (ha)	85.72	55.74
Pesticides	1000 Euros	19.15	18.05
EI_w [a]	Impact points	365.42	461.66
EI_b	Kg	43.85	198.86
Rotation	Percent (%) ha	45	21
Age	Years	54.65	9.88
Crop subsidies	1000 Euros	13.71	22.59
Economic size	European Size Units (ESU) [b]	164.99	4.558
Temperature	Mean temperature (°C) of first half year	8.77	1.06
Precipitation	Mean precipitation (mm) of first half year	346.85	75.63
Biodiversity	Number of species	458.04	406.48
	Category	Percent (%)	
Soil type [c]	0	40.43	
	1	59.57	

[a] EI_w and EI_b denote pesticide impacts on water organisms and biological controllers, respectively.
[b] One ESU corresponds to a standard gross margin of €1200.
[c] 0 indicates low quality soil type (sand, peat) and 1 high quality soil type (loess, fluvial or marine clay).

One output, five inputs (fertilizers, pesticides, other variable inputs, labour, capital, and land), and two environmental impacts of pesticides (impacts on water organisms and biological controllers) are distinguished. Output mainly consists of root crops (potatoes, sugar beets, carrots and onions), cereals (wheat, barley, triticale, corn, oats and rye) and other crops (green beans and peas and grasseed) and is measured as total revenue from all products, deflated to 2005 values using a Tornqvist index based on output prices from Eurostat. The inputs are separated into fixed ones which include land, capital and labour, and variable ones which consist of fertilizers, pesticides, and other variable (or specific crop) inputs. Land represents the total area under crops and is measured in hectares, capital includes the replacement value of machinery, buildings and installations, deflated to 2005 using a Tornqvist index based on the respective price indices, and labour is measured in annual work units (AWU).[8] Fertilizers were measured as expenditures deflated to 2005 using the fertilizer price index and pesticides were measured as expenditures deflated to 2005 using pesticide price index. The "other inputs" variable includes expenditures on energy, seeds and other specific crop costs, deflated to 2005 using a Torngvist index for the disaggregated "other inputs" components.

The environmental impact data were obtained from the Dutch Centre for Agriculture and Environment (CLM, 2010). For each pesticide that Dutch arable farmers use, there is an environmental indicator which shows the impact on aquatic, surface water organisms (EI_w), and biological controllers (EI_b). The effects of pesticides on water organisms is known as environmental impact points. [9] The EI_w depends on pesticide toxicity and the amount of spray drift to watercourses. The amount that reaches a watercourse depends on the application technique. The percentage spray drift is 1% for arable farming. Originally the environmental impact points for EI_w are computed for a standard application (i.e., 1 kg/ha). To calculate the application specific EI_w the environmental impact points under a standard application are multiplied by the actual applied quantity per hectare (CLM, 2010). The total farm specific EI_w for one year is computed by summing up the impact points of the individual pesticide applications. EI_w increases when pesticides have a greater impact on the environment. For water organisms, a score of 10 impact points is in line with the acceptable level (AL) set by the Dutch board for the authorization of pesticides (CTB) which reflects the concentration which implicates minor risk for the environment.

The risk for biological controllers (EI_b) (e.g., ladybugs, predatory mites, hymenopteran parasitoids) is indicated in the data ordinally with a symbol. This symbol shows the usability for integrated cropping systems and is a combination of all pesticide effects for individual beneficial organisms. [10] There are four symbols for bio-controllers: symbol ‘A’ indicates that the pesticide is useful in the effort to save beneficial organisms; symbol ‘B’ slightly useful; symbol ‘C’ not useful; and symbol ‘?’ not well known impact. The EI_b variable is a continuous variable that represents the sum of the kilograms of the most hazardous for beneficial organisms

[8] One AWU is equivalent to one person working full-time on the holding (EC, 2001).
[9] This category includes mainly aquatic insects (CLM, 2010).
[10] Direct effects, such as mortality or non-hatching of eggs and pupae, have been taken into account as well as indirect effects, such as reduced fertility, repellency, persistence etc.

applications ("C"). In this way the EI_b variable reflects the magnitude of pressure farmers exert on biological controllers.

Rotation is measured as the share of root crops in the total crops' acreage; crop subsidies are measured in euros. The average age of the sampled farmers was 55 years (Table 1). Farms distinction according to soil type took place after using a simplified soil map of The Netherlands (Hiemstra et al., 2009) which distinguishes soils in six classes (sand, peat, loess, marine clay, fluvial clay, and built-up). The soil type variable is measured as a dummy variable with 0 indicating farms operating under low quality soil (sand and peat) and 1 under high quality soil (loess, fluvial and marine clay). Farms that operate under low quality soil account for around 40% of the total number of farms in the sample while around 60% operate under high quality soil. The mean economic size of the sampled farms is 165 European Size Units (ESU) and is determined on the basis of the overall standard gross margin of the holding.

Meteorological data from 36 weather stations within the Netherlands were obtained from the Royal Netherlands Meteorological Institute (KNMI, 2011). The high number of weather stations in conjunction with detailed location data of the sampled farms enabled us to obtain a highly spatially disaggregated dataset on meteorological variables.[11]

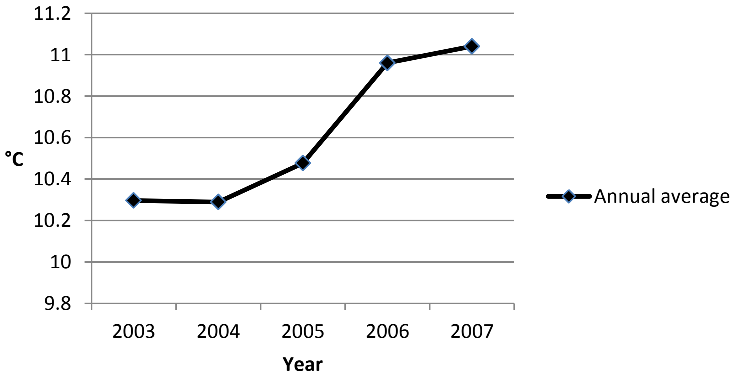

Figure 3. Annual average temperature of first half year in The Netherlands, 2003-2007.
Source: KNMI (2011)

[11] The sampled farms are separated in 33 regions according to a location map provided by the LEI (LEI, 2011).

Averaging the temperatures and precipitation of the first six months of each year, results in the mean temperature and precipitation for the main growing period from 2003-2007 (Reidsma et al., 2009). The mean temperature of the main growing period between 2003-2007 was 10.6 degrees Celsius with the warmest year being 2007 with around 11 degrees Celsius (Figure 3).The average precipitation amount during the same period is 785 mm, with 2007 being the most wet year with 942 mm (Figure 4).

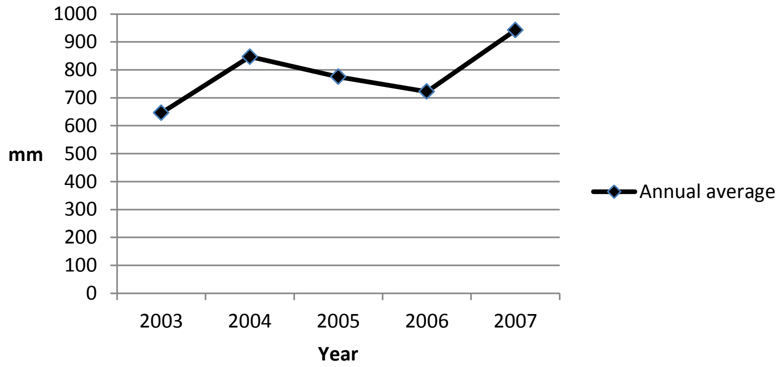

Figure 4. Annual average precipitation of first half year in The Netherlands, 2003-2007.
Source: KNMI (2011)

Finally, biodiversity data were obtained from the Netherlands Biodiversity Information Facility (NLBIF, 2011) including species of both flora and fauna (e.g. arthropods, birds etc.). The biodiversity variable reflects the number of species found in one of the 33 regions that each sampled farm belongs to.

5.4 Empirical results

5.4.1 Inefficiency measures

Technical inefficiency scores for output, variable inputs and undesirables are obtained using the GAMS programming software. Annual averages of technical inefficiency scores under VRS and WD of undesirables and fixed inputs in the years 2003-2007 are found in Table 2. Dutch arable farmers have considerable output technical inefficiency, with annual averages ranging between 15% and 30%. The average technical inefficiency for EI inputs (24%) and EI outputs

(25%) is slightly higher than the average output technical inefficiency score (21%), whereas the annual average of variable inputs technical inefficiency is much lower, ranging between 2% and 4%. EI output inefficiency can be interpreted as the level of farm pressure on the state of the environment, while EI input inefficiency indicates the extent that farmers consider the impact of their current pesticide decisions on next period's production realization. The high EI inefficiency shows that there is a considerable scope for decreasing the environmental impacts of pesticides.

When ignoring the *EI* (reduced form model) most of the inefficiency is picked up by the output variable (output inefficiency scores are overestimated by 56% on average) while variable inputs' inefficiency scores are hardly affected (Table 2). [12] Therefore, ignoring the dynamics may lead to increased output inefficiency thus misleading policy makers.

Table 2. Inefficiency measures of the directional distance function (VRS and WD of *EI*)

	2003	2004	2005	2006	2007
Output	0.23	0.23	0.15	0.16	0.30
EI (output)	0.21	0.27	0.23	0.21	0.30
Variable inputs	0.03	0.03	0.03	0.02	0.04
EI (input)	0.21	0.27	0.26	0.20	0.27
Inefficiency measures when *EI* are ignored					
Output	0.49	0.46	0.51	0.44	0.53
Variable inputs	0.05	0.04	0.04	0.03	0.03

Note: *EI* denotes environmental impacts of pesticides.

[12] This result is somehow expected as the number of efficient decision making units (DMUs) increases (decreases) as more variables are added to (excluded from) the model (Nunamaker, 1985).

5.4.2 Explaining farms inefficiency

Next, we turn to investigating how producer-specific and environmental variables are influencing the inefficiency scores of Dutch arable farmers. Table 3 presents the parameter estimates and their bootstrap-estimated 95% and 90% confidence interval. [13] Concerning output, the closer farmers are to retirement, the less efficient and more risk averse they may be. The effect of age on efficiency is highly debatable across the literature: older farmers are likely to have more experience and hence be less inefficient (Coeli and Battese, 1996). On the other hand, younger farmers may be more efficient as they tend to acquire more easily knowledge on technical advances (Weersink, et al., 1990) and are more motivated in adopting efficiency improving changes in their farms. Crops subsidies have a positive effect on output technical inefficiency as farmers may substitute subsidy income with farm income. The marginal effect of €1000 subsidies on technical inefficiency is 0.0057, implying that the average farm will have an increase of 0.57% in technical inefficiency.

Subsidies can reduce farmers' motivation to produce efficiently as they may decide to trade off market income for subsidy income. The negative impact of subsidies on farms' technical efficiency seems to be a fairly common finding in the literature (e.g. Lambarraa et al., 2009; Bezlepkina et al., 2005; Guyomard et al., 2006; Kleinhanß et al., 2007 and Emvalomatis et al., 2008; Karagiannis and Sarris, 2005; Dinar et al., 2007; Zhu et al., 2008; Zhu and Oude Lansink, 2010; Giannakas et al., 2001). Farming in more fertile soils increases output inefficiency as farmers may rely on a few highly profitable crops (e.g. potatoes) or varieties and hence do not diversify and spread risk spatially (Di Falco and Chavas, 2009). Biodiversity increases inefficiency in output production possibly through increased presence of non-target plants and other pests that cause crop damage. Higher temperatures decrease output inefficiency as they promote crop growth. An increase in precipitation rates increases output inefficiency as land may become increasingly waterlogged in some cases with negative consequences on farm productivity (Chambers et al., 2011). Finally, larger farms are more output inefficient compared to small farms. Larger farms (due to their size) may have difficulties in conducting their operations at the optimal time and thus being less efficient (Amara et al., 1999). Another

[13] When utilising the Simar and Wilson (2007) bootstrap procedure to compute the estimated bias-corrected inefficiency scores in the stage one bootstrap, a choice must be made about the number of replications. The number of replications in the bootstrap procedure has been set equal to 1000.

explanation may be that the share of family labour, that may be more motivated, flexible and committed to the business, is higher in smaller farms while large farms are more dependent on hired labour that needs to be supervised and may be less motivated (Wiggins et al, 2010). [14]

Table 3. Truncated regression. Estimated parameters and bootstrapped confidence intervals.

		95% confidence		90% confidence	
Output	Estimated parameter	Lower bound	Upper bound	Lower bound	Upper bound
Rotation [a]	-0.02307	-0.02834	0.01006	-0.02395	0.00573
Age	0.19710	**0.03849**	**0.86501**	**0.13086**	**0.76937**
Crop subsidies	0.00577	-0.00058	0.01016	**0.00042**	**0.00911**
Soil type [b]	0.11684	**0.05773**	**0.14751**	**0.06861**	**0.13997**
Biodiversity [c]	0.03761	**0.00670**	**0.04916**	**0.01187**	**0.04397**
Temperature [d]	-0.33302	**-0.62819**	**-0.08982**	**-0.57142**	**-0.15321**
Precipitation [e]	0.45266	**0.28565**	**0.57088**	**0.31685**	**0.54369**
Economic size [f]	0.05178	**0.01583**	**0.06455**	**0.02044**	**0.05853**
_cons	-2.09984	-2.75489	-1.37462	-2.63836	-1.4875
EI (output)					
Rotation	-0.04211	**-0.03773**	**-0.01494**	**-0.03483**	**-0.01735**
Age	0.92253	**0.60950**	**1.08654**	**0.66369**	**1.03413**
Crop subsidies	0.00035	-0.00298	0.00367	-0.00220	0.00292
Soil type	0.02196	-0.00662	0.04658	-0.00123	0.04028
Biodiversity	-0.00417	-0.01282	0.01488	-0.00929	0.01177
Temperature	0.28680	**0.15698**	**0.49306**	**0.20063**	**0.45947**
Precipitation	0.06577	-0.04231	0.13186	-0.02389	0.11297
Economic size	0.02897	**0.01614**	**0.04570**	**0.01974**	**0.04191**
_cons	-1.36143	-1.60634	-0.80770	-1.53021	-0.89432

[14] The mean share of hired labour on total labour (hired+own) for the bottom one-third of farms ranked by size for the study period was around 4% while the respective figure for the top quarter of farms was around 16%.

Table 3. (continued)

		95% confidence		90% confidence	
Variable inputs	Estimated parameter	Lower bound	Upper bound	Lower bound	Upper bound
Rotation	-0.04987	**-0.07820**	**-0.02880**	**-0.07267**	**-0.03405**
Age	-0.13650	-0.26088	0.55135	-0.17901	0.45411
Crop subsidies	0.00434	-0.00432	0.00727	-0.00313	0.00603
Soil type	0.07910	**0.01746**	**0.11063**	**0.02701**	**0.10040**
Biodiversity	-0.05982	**-0.09207**	**-0.04488**	**-0.08450**	**-0.04997**
Temperature	-0.16273	-0.35024	0.21528	-0.29340	0.14450
Precipitation	0.52981	**0.26641**	**0.59915**	**0.30018**	**0.56130**
Economic size	-0.03474	**-0.06084**	**-0.01054**	**-0.05430**	**-0.01530**
_cons	-2.05267	-2.74010	-1.16666	-2.52481	-1.29143
EI (input)					
Rotation	-0.03239	**-0.04137**	**-0.01825**	**-0.03821**	**-0.02068**
Age	-0.18075	-0.17872	0.30301	-0.11902	0.24432
Crop subsidies	0.00329	-0.00067	0.00593	**0.00013**	**0.00524**
Soil type	0.04361	**0.02784**	**0.07718**	**0.03211**	**0.07172**
Biodiversity	-0.01392	-0.02363	0.00290	**-0.02040**	**-0.00059**
Temperature	0.08021	-0.19049	0.13993	-0.15997	0.10078
Precipitation	-0.00531	-0.00718	0.16163	**0.01122**	**0.14339**
Economic size	0.04677	**0.02602**	**0.05420**	**0.02984**	**0.05116**
_cons	0.62833	-0.25242	0.57114	-0.16409	0.48650

Notes: i) The regressand is the bootstrap-based bias-corrected DEA estimate of the unobserved inefficiency score of output, *EI* (output), variable inputs, and *EI* (input). ii) Statistically significant confidence intervals are in bold.

[a] Measured as the percentage of root crops in the total crops' acreage.

[b] Measured as a dummy with 0 indicating low quality soil type (sand, peat) and 1 high quality soil type (loess, fluvial or marine clay).

[c] Total number of species in farms' area.

[d] Mean temperature (°C) of first half year.

[e] Mean precipitation (mm) of first half year.

[f] Measured in European Size Units (ESU).

The results of the regression of *EI* output inefficiency suggest that a higher share of root crops (rotation) decreases environmental inefficiency. Older farmers are less efficient in terms of environmental impacts as they may be less informed on the external effects of pesticides or less motivated in adopting environmental friendly innovations. In addition to being beneficial for the growth of target plants, higher temperatures may also favour the growth of non-target species and hence attract more farmland organisms (e.g., arthropods). This may require the intensification of pesticide applications to maintain crop yields, leading to higher environmental inefficiency. Larger farms are less environmental efficient as they use more pesticides leading to higher environmental spillovers.[15]

Concerning variable inputs, a higher share of root crops decreases inefficiency in the use of variable inputs suggesting inputs are used more efficiently in root crops than in other outputs. Farming in more productive soils leads to less efficient use of variable inputs showing that variable inputs are used more efficiently by farmers operating in less productive soils. Operating in regions with higher biodiversity populations leads to more efficient use of variable inputs, possibly through increased pollination and decreased crop damage as pests may encounter difficulties in spreading in a highly non-uniform environment. An increase in precipitation rates increases variable inputs' inefficiency as they promote the growth of non-target species leading to increased yield variability and higher crop specific costs (e.g., mechanical weeding, fuel use). Larger farms are less inefficient in the use of variable inputs by exploiting scale economies (Hallam and Machado, 1996). Coelli and Battese (1996) argue that smaller farms may have alternative income sources and thus put less effort in farming compared with the larger farms.

The results of the *EI* input inefficiency regression show that farmers that adopt a higher share of root crops are more aware of the production impacts of pesticides' environmental spillovers on future production. Farming in more productive land increases *EI* inputs' inefficiency as farming intensity is higher. Potatoes are one of the most profitable arable crops and are widely cultivated in clay soils. Their production requires intensive use of fungicides that may lead to

[15] A comparison of the bottom (small) and top one-third (large) of farms ranked by economic size reveals that large farms' have spent on average €22.5 thousand more pesticides than the small farms .The average EI_w and EI_b of large farms are 487.9 and 88.45, respectively, while the respective figures for small farms are 239.6 and 18.7, respectively.

higher *EI*. In such a case, the economic benefits from operating in more fertile land may outweigh potential crop losses from pesticides' negative effects. An increase in precipitation during the growing season may render farmers more precise and careful in applying pesticides, considering their potential negative future impacts through increased leaching. Finally, large farmers are less effective in taking into account the impact of pesticide's environmental spillovers in future output, as production in larger economic size farms may be more profit-driven and thus the short-term economic benefits of increased pesticide applications may outweigh their negative long-run production impact. Therefore, large farmers tend to be more myopic decision makers by ignoring the dynamics or future effects of their current production decisions.

5.4.3 Accounting for the impact of variation in the production conditions

The Stage 3 DEA inefficiency scores are presented in Table 4. After adjusting for variation in the production conditions, inefficiency scores decrease. This is consistent with the hypothesis that producers operating under unfavourable production conditions may be disadvantaged in the initial DEA evaluation that does not take this factor into account. More specifically, adjusting performance evaluations for variation in the production conditions results in a decrease in average output, *EI*-output, variable inputs, and *EI*-input inefficiency of around 24%, 50%, 40%, and 46%, respectively.

The highest output inefficiency decrease is observed in 2007 and may be partly related to the fact that this year accounts for the highest precipitation amount in the study period (Figure 4). This is consistent with the hypothesis that increased precipitation can impact negatively farm productivity as in some cases land may become waterlogged. Concerning *EI*-output inefficiency, the highest decrease is observed in 2007 and 2004 as some farms that received relatively low initial performance (i.e. higher inefficiency scores) did so in part due to their relatively unfavourable production conditions such as high precipitation rates that can be responsible for increased pesticide leaching.[16] Variable inputs' inefficiency scores did not change significantly while *EI*-input inefficiency scores follow almost the same trend as the *EI*-

[16] High precipitation rates in conjunction with high temperatures (especially in 2007, Figure 3) can boost not only target crops' but also non-target species' growth, leading to higher use of pesticides and thus greater environmental pressure.

output scores (i.e., high inefficiency changes during the years of relatively extreme weather events).

Accounting for production uncertainty in the evaluation of farmers' performance has resulted in a notable improvement of efficiency scores. The recent empirical literature on this point reports similar findings. Chambers et al. (2011) developed an event specific DEA model that shows how event-specific representation of the production technology can be implemented within a DEA framework and compared its findings with the standard DEA model. Their results show that when stochastic elements that alter the nature of the production technology are ignored, efficiency scores are underestimated up to 50%. Emvalomatis (2011) in a study that applies a dynamic stochastic frontier model to a panel of US electric utilities shows that ignoring unobserved heterogeneity (i.e., firm-specific factors that affect productivity and are not under the control of the firm) leads to higher persistence of inefficiency, as part of the unobserved heterogeneity is interpreted as inefficiency. Greene (2005a, b) argues that the stochastic component of frontier models can be viewed as containing both inefficiency and heterogeneity and shows that accounting for heterogeneity in stochastic frontier models brings significant changes in estimated results.

The initial DEA evaluation has revealed that Dutch arable farmers are for output and undesirables on average around 21% and 24%, respectively, below the production frontier, while with the adjusted DEA model the distance from the frontier is reduced to around 16% and 13%, respectively. Therefore, it can be argued that managerial inefficiency accounts for only around 13-16% while another 5-11% is attributed to production uncertainty. In monetary terms, farmers' profit loss from production uncertainty is on average € 9.57 thousand. The notable amount of profit loss of the sampled farms reveals a need to mitigate the economic damage through risk management tools. Market-oriented risk management tools are widely available but to what extent are economically feasible for farmers that want to cover such a considerable amount of profit loss needs further investigation.

Table 4. Comparison of the DEA initial and adjusted models' results.

Results	Initial	Adjusted	Initial	Adjusted	Initial	Adjusted	Initial	Adjusted	Initial	Adjusted
Year	2003	2003	2004	2004	2005	2005	2006	2006	2007	2007
Output	0.23	0.18	0.23	0.18	0.15	0.13	0.16	0.13	0.30	0.16
EI (output)	0.21	0.11	0.27	0.10	0.23	0.12	0.21	0.11	0.30	0.16
Variable inputs	0.03	0.02	0.03	0.01	0.03	0.03	0.02	0.01	0.04	0.02
EI (input)	0.21	0.13	0.27	0.14	0.26	0.10	0.20	0.13	0.27	0.14

Note: *EI* denotes environmental impacts of pesticides.

5.5 Conclusions

Employing non-parametric methods to compute farmers technical and environmental inefficiency may provide a wrong estimation of where we stand when ignoring the dynamics of the operating environment and the impact of the variability in production conditions. This study uses DEA to compute output, input and undesirables inefficiency in Dutch arable production over the period 2003-2007. Initially, a dynamic DEA model is employed and farms' performance is compared with the results of a standard DEA model that ignores the dynamics of pesticide use. Then a bootstrap procedure is applied to explore the factors that might explain inefficiency in the dynamic model. After the bootstrap procedure has been completed, the original outputs and inputs are adjusted to account for the impact of the variability in production conditions. Then the dynamic DEA model is re-employed after replacing observed output and input data with those adjusted for the impact of variability in production conditions and compared to the initial dynamic DEA model.

Results of the initial DEA evaluation show that Dutch farmers have noticable output inefficiency scores and high EI (both input and output) inefficiencies that reveal a considerable scope for decreasing pesticides' environmental spillovers. The analysis reveals among others that large farms are more output and environmental inefficient both in terms of protecting the status of the environment and taking into account pesticides' future negative effects in their current production decisions. Biodiversity and weather related variables do have a statistically significant effect on farmers' performance. After adjusting outputs and inputs to account for the impact of variability in production conditions, estimates of inefficiency decreased dramatically. The results highlight the degree to which our understanding of efficiency levels can be distorted when using models that ignore the dynamics of production and the effects of variability in production conditions.

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Chapter 6

Pesticide Use, Environmental Spillovers and Efficiency: A Nonparametric Risk-Adjusted Efficiency Approach Applied to Dutch Arable Farming

This chapter is in review at *Environmental and Resource Economics* as:
Skevas, T., S.E., Stefanou, and A., Oude Lansink. ″Pesticide use, environmental spillovers and efficiency: A nonparametric risk-adjusted efficiency approach applied to Dutch arable farming ″.

Abstract

Pesticides are widely used by crop producers in developed countries to combat risk associated with pest and diseases. However, their indiscriminate use can lead to various environmental spillovers that may alter the production environment thus contributing to production risk. This study utilises a nonparametric efficiency approach to measure performance of arable farms, incorporating pesticides' environmental spillovers and output variance as risky outputs in the efficiency analysis and taking explicitly into account the effect of pesticides and other inputs on production risk. This approach is applied to outputs, risk-mitigating inputs, and risky outputs of Dutch arable farms over the period 2003-2007. A moment approach is used to compute output variance, providing empirical representations of the risk-increasing or -decreasing nature of the used inputs. Finally, shadow values of risk-mitigating inputs are computed. We find that pesticides are overused in Dutch arable farming and there is a considerable scope for decreasing pesticides' environmental spillovers.

Keywords: environmental spillovers, Netherlands, pesticides, production risk.

6

6.1 Introduction

Risk constitutes an integral part of the agricultural production environment, playing an important role in farm-level production decision making. Production risk alongside with price risk, technological risk and, policy risk are the main sources of risk and uncertainty that are relevant from the point of view of the agricultural decision maker (Moschini and Hennessy 2001). Production risk refers to the stochastic nature of agricultural production where there is uncertainty on the amount and quality of output resulting from different input choices. This uncertainty may arise from unpredictable weather events and/or sudden increase in destructiveness or population numbers of a pest species in a given area (i.e., pest infestation). Several risk management tools are available to producers in developed countries to manage risk, notably market or financial insurance, price contracts, pesticides, fertilizers, crop rotation, anti-hail protection equipment, and genetically modified crops.

Concerning pesticide use, as pest arrival is an uncertain event and pesticide productivity varies across time and space, there is an uncertainty at the time of application. This uncertainty can lead to overuse of pesticides relative to the private or social optimum. In an effort to avoid crop losses, risk averse farmers apply pesticides at an early stage when the pest population may not be at its peak. This action can induce extra costs as additional pesticide doses are applied. On the other hand, waiting and monitoring the pest population and applying pesticide when full information is available may increase the crop loss at the monitoring stages. Norgaard (1976) states that the major motivation for pesticide application is the provision of some "insurance" against damage. Therefore, uncertainty in the pest-pesticide system leads to a higher and more frequent use of pesticides.

Farmers often lack full knowledge of the relation between pesticides and pest mortality (Feder 1979). Pesticide effectiveness can be influenced by fluctuations in weather conditions such as precipitation and temperature. Changes in weather conditions can impact both pest populations and the effectiveness of pesticides as each chemical product has different durability. Horowitz and Lichtenberg (1994) consider three scenarios of risk or uncertainty: risk or uncertainty about a) crop growth conditions only; b) pest damage only; and c) both growth conditions and pest damage. Their findings support the conventional view that when there is uncertainty due to pest damage, pesticides are likely to be risk-reducing inputs. However, the literature reports mixed findings on the role of risk aversion with some studies finding that pesticides are risk-reducing (Griffiths and Anderson 1982; Saha et al. 1994; Smith and Goodwin 1996) and other risk-increasing inputs (Horowitz and Lichtenberg 1993; Pannell 1995; Gotsch and Regev 1996; Saha et al. 1997). When both pest populations are high and growth conditions are favorable, pesticides will be risk-increasing as they increase the variability of harvests (increase output under good growth conditions). Horowitz and Lichtenberg (1993) have shown that pesticides may be risk-increasing inputs even if a federal government provides crop insurances that act as a substitute for additional pesticide applications.

The relationship between pesticide use and production risk may also be shaped by pesticides' environmental spillovers. Farmland biodiversity can benefit farm productivity (Di Falco and Chavas 2006; Omer et. al 2006; Tilman et al. 2005), reduce environmental risk and yield variability, improve pest control by impeding the evolution of pest populations and consequently reducing pest damages (Priestley and Bayles 1980; Heisey et al. 1997). Therefore, pesticide indirect effects on biodiversity may increase production risk through decrease in beneficial natural predators (Pimentel et al. 1992).

The theory on modelling firms' production risk is well developed (Just and Pope, 1978; Antle, 1987). The Just and Pope (1978) approach to modeling production processes in the face of production risk has been widely used in applied analysis, with the variation in production being influenced by the input levels; some inputs may be variation-increasing, while others are variation-decreasing, where risk is defined as the variance of output. Increasing attention has been given in recent years to risk in agricultural decisions and ways to mitigate it. Output variance and pesticides' environmental spillovers may be considered among the outputs of agricultural production to be minimized (i.e., undesirable outputs). Several attempts using data envelopment analysis (DEA) methods have been made in the literature to measure efficiency in the presence of undesirable outputs (Färe et al., 1989; Ball et al., 1994; Färe et al., 1996; Tyteca, 1997; Reinhard et al., 2000; Hailu and Veeman, 2001; Oude Lansink and Silva, 2003; Piot-Lepetit and Moing, 2007). An approach allowing for an asymmetric treatment of desirable and undesirable outputs was proposed by Färe et al. (1989), where undesirable outputs are treated as weakly disposable while desirable outputs are strongly disposable. Weak disposability means that reducing (increasing) undesirable outputs (inputs) is not a costless procedure. On this basis, a number of studies have proposed the use of directional distance functions as a tool for modelling production in the presence of undesirables (Chung et al.,1997; Ball et al., 2001). A directional distance function allows for a simultaneous expansion of desirable outputs and contraction of inputs and/or undesirable outputs (Chung et al. 1997). The employment of directional distance functions to measure both technical and environmental efficiency of firms that produce both desirable and undesirable outputs has become widespread (Färe et al., 2005; Piot-Lepetit and Moing, 2007; Murty et al., 2007; Kjærsgaard et al., 2009). Undesirables are not always treated as inputs or outputs in a directional distance function approach. Hoang and Alauddin (2011), in a study that measures economic, environmental and ecological performance of agricultural production systems in 30 OECD countries, use a directional distance function approach seeking the optimal input and output combination that minimizes the total amount of nutrient and cumulative exergy balance sent into the environment.

Various attempts have been made to incorporate risk in non-parametric efficiency analysis with the vast majority being in the banking sector. Some studies focus on cost efficiency measures with incomplete price information (Schaffnit et al., 1997; Camanho and Dyson, 2005) and risk-adjusted profit efficiency using a mean variance criterion (Settlage et al., 2009) [1], and others are treating risk as an external factor and employ the methods described in Fried et al. (2002) to adjust efficiency measures for risk (Chang 1999; Chen et al. 2007).

In the context of agricultural production, Chambers et al. (2011) employ a DEA model incorporating climatic variables to account for production uncertainty in the evaluation of farmers' performance, finding that efficiency results change dramatically when acknowledging stochastic elements. Skevas et al. (Chapter 5) have expanded Fried's et al (2002) approach to account for a wide representation of production uncertainty in computing the efficiency of Dutch cash crop farms. Both studies adjust efficiency modelling to reflect risk as an exogenous factor. However, none of these studies explicitly accounted for the risk-increasing or decreasing nature of agricultural inputs and the employment of risky outputs. The objective of this study is to investigate the performance of Dutch arable farms by using a risk-adjusted efficiency

[1] Others include risk as an input in the production process of a bank (Berg et al. 1992) or as an undesirable output (Chang 1999; Park and Weber 2006).

measure to a) identify technical inefficiency and risk-mitigating inputs and risky outputs inefficiency, b) incorporate risky outputs in the efficiency analysis and c) take explicitly into account the extent to which agricultural inputs increase or decrease production risk.

The rest of the paper continues with Section 2 with the presentation of the methodology, while Section 3 describes the definition and sources of the data used. In Section 4, the empirical results from the analysis are presented and discussed, and Section 5 concludes.

6

6.2 Methodology

6.2.1 A risk-adjusted inefficiency model

The inefficiency model is based on a set of observations of farms in a sample that use a vector of variable inputs, fixed inputs, and risk-mitigating inputs to produce a desirable output (*y*) and risky or undesirable outputs (*r*). The directional technology distance function in presence of risky outputs, seeks to increase the desirable output while simultaneously reducing the risky outputs. Assuming weak disposability of risky outputs, and fixed inputs, a model that decomposes technical inefficiency of the different inputs and outputs for each firm *i*, *i* = 1, … ,*N*, is as follows:

$$\vec{D}_o^t\left(q, y, EI; g_y, -g_z, -g_r, -g_c\right) = \underset{\beta^t \lambda^t,}{Max}\{\beta_1^t + \beta_2^t + \beta_3^t + \beta_4^t\}$$

s.t.

$$\sum_{i=1}^{I^t} \lambda_i^t y_{im}^t \geq y_m^t + \beta_1^t g_y^t \qquad \text{(i)}$$

$$\sum_{i=1}^{I^t} \lambda_i^t z_{iv}^t \leq z_v^t - \beta_2^t g_z^t \qquad \text{(ii)}$$

$$\sum_{i=1}^{I^t} \lambda_i^t q_{id}^t = \varphi q_d^t \qquad \text{(iii)}$$

$$\sum_{i=1}^{I^t} \lambda_i^t r_q^t = \varphi(r_q^t - \beta_3^t g_r^t) \qquad \text{(iv)} \qquad (1)$$

$$\sum_{i=1}^{I^t} \lambda_i^t x_{if}^t \leq x_f^t - \beta_4^t g_x^t \qquad \text{(v)}$$

$$N1'\lambda = 1 \qquad \text{(vi)}$$

$$\lambda_i^t \geq 0 \qquad \text{(vii)}$$

$$0 < \varphi \leq 1 \qquad \text{(viii)}$$

where β_1, β_2, β_3 and β_4 are the technical inefficiency scores for the *i*-th farm of desirable output, risk-mitigating inputs, risky outputs, and variable inputs, respectively. Desirable output is represented by y, z are the risk-mitigating inputs (i.e., fertilizer, fungicides, herbicides, insecticides and, other pesticides), q are the fixed inputs (i.e., capital, labour, land), r represents risky outputs (i.e., output variance, and pesticide effects on biodiversity), x are the variable inputs (i.e., other inputs), and λ are the firm weights (intensity variables). Pesticide effects on biodiversity are considered risky outputs as higher pressure on farmland organisms can deprive farms from services such as soil nutrient enhancement and increase production risk through decrease in beneficial natural predators (Pimentel et al. 1992). The vector of directions in which outputs and inputs can be scaled is represented by g. Model (1) seeks for the maximum attainable expansion of desirable outputs in the g_y direction and the largest feasible contraction of risk-mitigating inputs, risky outputs, and variable inputs in $-g_z$, $-g_r$ and $-g_x$ direction, respectively (Chung et al. 1997). The latter are negative to pick up the fact that risky outputs and inputs are being reduced. Constraint (vi) allows for a variable returns to scale (VRS) technology. The scaling parameter φ is selected to ensure a feasible solution of the DEA model with weakly disposable fixed inputs and risky outputs under VRS. Weak disposability of fixed inputs and risky outputs throughout the production process is imposed through constraints (iii) and (iv), respectively. Fixed inputs are specified as weakly disposable (in the short run) as changes in land or capital are processes that involve high costs. The indirect effects of pesticides on biodiversity characterized as risky outputs and also considered weakly disposable as considerable reductions in environmental spillovers may require significant changes in the type and cost of pesticide products used (e.g., purchasing more environmental friendly products may be more expensive than high toxicity products). Moreover, the indirect effects of pesticides may reduce the production possibility set for other farms, resulting in governmental regulations. Therefore, the disposing of this risky output is not a costless activity. Yield variance is also not freely disposable as it is related to weather conditions and changes in pest populations, i.e., variables beyond the control of farmers. On the other hand, risk-mitigating inputs are considered strongly or freely disposable; i.e., changing the levels of these inputs does not involve costs for farms.

After solving model (1), a set of dual variables for each observation is obtained, accounting for the effect on inefficiency of a change of each technological constraint. Utilizing the procedure suggested by Ball et al. (1994, 2004), these dual variables can be used to generate the shadow values of each risk-mitigating input. The shadow value of each risk-mitigating input is:

$$SV_{vi} = p * \left[-\frac{\partial\beta_2 / \partial z_{vi}}{\partial\beta_1 / \partial y_i} \right] \qquad (2)$$

Where SV_{vi} is the shadow value of risk-mitigating input v, v=1,...,V for each firm i, i=1,...,N, and p is the output price. The terms $\partial\beta_2/\partial z_{vi}$ and $\partial\beta_1/\partial y_i$ are the shadow costs associated with constraints (i), and (ii) (i.e., on output y, and risk-mitigating inputs v, respectively) from model (1). The extent to which risk-adjusted inputs are over- or under-used is inferred from a comparison of the shadow values with the market prices. Market prices are greater (lower) than shadow values for inputs that are over-used (under-used).

6.2.2 Output variance

The estimation procedure of output variance follows two steps. First, we estimate the first two moments of the output distribution following a sequential estimation procedure as described in Kim and Chavas (2003) which adapts the procedure in Antle (1987). In the first step output is regressed on the input variables as shown in the following model:

$$y_i = f(x_i; \beta_i) + u_i \qquad (2)$$

where y denotes output, x is a vector of production inputs (fertilizers, other variable inputs, fungicides, herbicides, insecticides, other pesticides, labour, capital, and land), u is the identically independently distributed error term, and i =1,...,N denotes individual farmers in the sample.

The jth moment of output conditional on input use is:

$$\mu_j = E\{[y(.)]^j\} \quad (3)$$

Thus, the estimated errors from the regression in equation (2) are estimates of the first moment of output distribution. The estimated errors $\hat{u}$, are then squared and regressed on the same set of explanatory variables as in equation (2):

$$\hat{u}_i^2 = q(x_i; \gamma_i) + \hat{u}_i \quad (4)$$

Consistent estimates of the parameter vector γ are obtained after applying OLS to equation (4). The predicted values $\hat{u}_i^2$ are consistent estimates of the second central moment of output distribution (Antle 1983). The farm specific variance of output is computed as following:

$$v_i = \hat{u}_i^2 - \mu_1^2 \quad (5)$$

In general, it is expected that all inputs increase output, but for the second moment, inputs can be either risk-increasing or risk-decreasing.

6.3 Data

The data are an unbalanced panel of Dutch arable farms covering the period 2003-2007, obtained from the Agricultural Economics Research Institute (LEI). Farms remain in the panel for a maximum of five years. The data set used comprises 493 observations from 119 farms. Table 1 reports the mean values of the data.

Table 1. Summary statistics

Variable	Dimension	Mean	S.D.
Output	1000 Euros	197.89	202.03
Fertilizers	1000 Euros	10.07	8.67
Other inputs	1000 Euros	51.59	54.38
Labour	Annual work units (AWU)	1.67	0.88
Capital	1000 Euros	335.30	389.17
Land	Hectares (ha)	80.96	56.55
Fungicides	1000 Euros	12.76	9.82
Herbicides	1000 Euros	8.35	5.44
Insecticides	1000 Euros	1.74	1.92
Other pesticides	1000 Euros	2.55	3.26
Pesticide impact on water organisms	Impact points	490.01	508.05
Pesticide impact on soil organisms	Impact points	643.09	654.86

One output, 7 inputs (fertilizers, fungicides, herbicides, insecticides, other pesticides, other variable inputs, labour, capital, and land), and two pesticide externalities (impacts on water and soil organisms) are distinguished. Output mainly consists of potatoes, sugar beets, and cereals (wheat, barley, corn) and is measured as total revenue from all products, deflated to 2005 values using a Tornqvist index for the disaggregated output components. The inputs are separated into fixed inputs including land, capital and labour, variable inputs consisting of other variable (or specific crop) inputs, and risk-mitigating inputs including fertilizers, fungicides, insecticides, and other pesticides [2]. Land represents the total area under crops and is measured in hectares, capital includes the replacement value of machinery, buildings and installations, deflated to 2005 using a Tornqvist index based on the respective price indices, and labour is measured in annual work units (AWU[3]). Fertilizers were measured as expenditures deflated to 2005 using a fertilizer price index. All pesticide categories are measured as expenditures deflated to 2005 using pesticide price indexes for each pesticide category from Eurostat. The "other inputs" variable includes expenditures on energy, seeds and other specific crop costs, deflated to 2005 using a Torngvist index for the disaggregated "other inputs" components.

[2] Other pesticides include growth regulators, rodenticides, additives (i.e., mineral oil), ground disinfectants, detergents, sulfur, and, unclassified products.
[3] One AWU is equivalent to one person working full-time on the holding (EC 2001).

The pesticide impact data are obtained from the Dutch Centre for Agriculture and Environment (CLM). There is an environmental indicator expressed in impact points for each pesticide farmers use, capturing its indirect effect on water and soil organisms.[4] Pesticide toxicity and the amount of spray drift to watercourses (1% for arable farming) are taken into account in computing the impact points for water organisms. The impact points for soil organisms, are computed based on the organic matter content (3-6% for the case study farms), pesticide characteristics (degradation rate, and mobility in soil) and pesticide toxicity. The organic matter content in conjunction with pesticide characteristics determine the amount of pesticides that remain in the soil over time. Originally, the environmental impact points (for both water and soil organisms) are expressed for an application of 1 kg/ha (standard application). The impact points under a standard application are multiplied by the actual applied quantity per hectare (CLM, 2010). The final farm-specific impact for water and soil organisms is computed by summing up the impact points of the individual pesticide applications.

The environmental impact points increase when pesticides have a greater impact on the environment. For soil organisms a score of 100 impact points is in line with the acceptable level (AL) set by the Dutch board for the authorization of pesticides (CTB) which reflects the concentration which implicates minor risk for the environment. Since 1995, the AL for aquatic organisms is 10 impact points per application (CLM, 2010).

6.4 Results

6.4.1 Analysis of elasticities and variance

The results of the moment function estimation are summarized in Table 2 in terms of the elasticities of the first two moments of output with respect to each production input using a quadratic functional form specification (which includes inputs in levels, squares and cross variables).[5] Concerning the first moment, all farmer choice factors have a positive and

[4] Water and soil organisms include mainly aquatic and soil insects, respectively (CLM 2010).

[5] Most of the inputs in squares and cross variables of equation (2) where insignificant even at the 10 per cent significance level and were excluded from the estimation. The only significant terms were the squared insecticide variable (parameter estimate: 0.01, p-value: 0.000) and the cross variable "fungicides*herbicides" (parameter estimate: -0.05, p-value: 0.000).

significant impact, except for labour that is also insignificant. This is quite close to our expectation that inputs in the first moment function estimation increase output. The significant elasticities of fertilizer and other inputs indicate that these inputs do play an important role in crop production. Land elasticity is higher in comparison to the rest of the productive inputs, implying that land is a scarce input that constrains the cash crop sector. All pesticide elasticities are significant, pointing to the importance of pesticides in reducing crop damage. Fungicide elasticity is higher in comparison to the rest of pesticide inputs. This is expected as one of the most important crops of the sampled farms is potatoes that requires rigorous fungicide applications to combat oomycete Phytophthora infestants, considered the crops' main enemy (Haverkort et al. 2009).

In the second stage estimation describing the second moment, production inputs can be separated into marginal risk-reducing and marginal risk-increasing inputs. The marginal risk-reducing input category includes fertilizer, land, fungicides, insecticides, and herbicides, but only pesticide inputs play a significant role in decreasing output variance. Fungicides are important in preventing crop damage in potato production that has a high share in the examined farmers' crop basket. Insecticides and herbicides can reduce output variability by reducing the presence of pests, thus optimizing the growth conditions of target plants. Griffiths and Anderson (1982), Saha et al., (1994) and Smith and Goodwin (1996) support the view that pesticides are risk-reducing inputs while Horowitz and Lichtenberg (1993), Pannell (1995), Gotsch and Regev (1996), and Saha et al., (1997) suggest othewise. When both pest populations are high and growth conditions are favorable, pesticides will be risk increasing as they increase the variability of harvests (i.e., increase output under good growth conditions). Among the marginal risk-increasing inputs are other inputs, labour, capital, and other pesticides. Labour and other pesticides are the only significant parameters at the 5% significance level. Assuming that pest damage is independent of other factors affecting output may lead to the conventional view that pesticides are risk-reducing inputs (Horowitz and Lichtenberg 1994).

Table 2. Elasticity of moment with respect to each input

	Moment	
	First	Second
Fertilizer	0.090 (0.001)	-0.014 (0.270)
Other inputs	0.194 (0.000)	0.021 (0.156)
Labour	-0.057 (0.154)	0.041 (0.036)
Capital	0.091 (0.000)	0.018 (0.147)
Land	0.363 (0.000)	-0.022 (0.280)
Fungicides	0.190 (0.029)	-0.029 (0.014)
Insecticides	0.016 (0.016)	-0.014 (0.008)
Herbicides	0.084 (0.040)	-0.026 (0.019)
Other pesticides [a]	0.014 (0.028)	0.009 (0.002)

Note: P-values in parenthesis. P-values for fungicides, insecticides, and herbicides were computed using bootstrap techniques.
[a] Include growth regulators, rodenticides, additives (i.e., mineral oil), ground disinfectants, detergents, sulfur, and, unclassified products.

6.4.2 Technical inefficiency

Technical inefficiency scores for output, risk-mitigating inputs, risky outputs, and variable inputs are obtained using the GAMS programming software. Annual averages of technical inefficiency scores under VRS and WD of risky outputs and fixed inputs in the years 2003-2007 are found in Table 3. Annual averages of output technical inefficiency of Dutch arable farmers ranges between 6% and 13%. The average technical inefficiency for risk-mitigating inputs (6%) is slightly lower than the average output technical inefficiency score (9%), whereas the average technical inefficiency for risky outputs (9%) is at a similar level. The annual average of variable inputs technical inefficiency is quite low, ranging between 2% and 4%. Risky outputs' inefficiency can be interpreted as the level of farm production pressure on farmland biodiversity, and farmers' capacity in reducing output variability. Risk-mitigating inputs' inefficiency indicates how efficient are farmers in using risk-mitigating inputs to manage production risk. The considerable level of risky outputs' inefficiency presents a considerable range (9%) of potential improvement in decreasing the environmental impacts of pesticides.

Table 3. Inefficiency measures (SD [a] of risk-mitigating inputs, WD of risky outputs).

	2003	2004	2005	2006	2007
Output	0.06	0.07	0.07	0.13	0.10
Risk-adjusted inputs	0.05	0.05	0.08	0.07	0.06
Risky outputs	0.08	0.09	0.10	0.09	0.08
Variable inputs	0.03	0.04	0.04	0.03	0.02

[a] SD and WD denote strong and weak disposability, respectively.

6.4.3 Analysis of shadow values

Table 4 presents the shadow values of risk-mitigating inputs which are computed at the sample means, at average output price index 1.12. The average shadow price of fertilizer over the period 2003-2007 is 0.55. A comparison of this shadow price with its market price shows that fertilizers were overused.[6] This finding is consistent with results from Oude Lansink and Silva (2004), and Guan et al. (2005) while Skevas et al. (Chapter 3) report that fertilizers were optimally used from Dutch cash crop producers. A comparison of the average shadow values of fungicides, herbicides, insecticides, and other pesticides with pesticide prices shows that all pesticides were overused. This finding shows that farmers could increase their profitability by decreasing the use of pesticides.

Oude Lansink and Silva (2004) report fungicides, other pesticides, and herbicides [7] were (on average) under-utilized in their study of Dutch arable farms over the period 1989-1992. Underutilization of fungicides, herbicides and other pesticides is also reported by Oude Lansink and Carpentier (2001) for Dutch specialized arable farms over the same period. Guan et al. (2005) report a pesticide shadow value of 1.25 and conclude that pesticides were optimally used at the farm level but they add that this might lead to an overuse if the indirect effects of pesticides are taken into account. Skevas et al. (Chapter 3) use a model with two types of

[6] A statistical test has shown that the shadow value of fertilizers is significantly different from its market price.
[7] Herbicides were over-utilized only in the model that measured efficiency radially in the productive input subspace.

pesticides that differ in terms of toxicity and environmental spillovers of pesticides in their study of Dutch arable farms over the period 2003-2007. Their findings show that both types of pesticides were overused, a result that is in line with our finding.

Table 4. Annual averages of the shadow values of fertilizer and pesticides (SD [a] of risky inputs, WD of risky outputs).

	2003	2004	2005	2006	2007	2003-2007	Input price [b]
Fertilizer	0.49	0.57	0.35	0.43	0.70	0.51	0.98
Fungicides	0.09	0.04	0.09	0.31	0.16	0.14	0.99
Herbicides	0.28	0.27	0.11	0.09	0.13	0.18	1.01
Insecticides	0.43	0.69	0.95	0.56	0.16	0.56	1.02
Other pesticides	0.92	0.86	0.83	0.79	0.26	0.73	1.01

[a] SD and WD denote strong and weak disposability, respectively.
[b] Price index from Eurostat.

6.5 Conclusions

This study uses a non-parametric risk-adjusted inefficiency model of Dutch cash crop producers that explicitly accounts for the effect of production means on output variability to compute output, risk-mitigating inputs and risky outputs inefficiency. Farmers use risk-mitigating inputs to manage risk, with agricultural practices leading to the production of risky outputs, defined as the variance of output and pesticide effects on biodiversity. The first two moments of output distribution are used to compute output variance which is then incorporated into the efficiency modelling framework, thus accounting for the risk increasing or decreasing nature of the employed production inputs.

Results show that fungicides and herbicides are risk-reducing inputs while other pesticides and other inputs are risk-increasing inputs. Dutch cash crop farmers have considerable levels of risky outputs' inefficiency (i.e., 9%) implying that policy makers could focus on reducing pesticides' environmental spillovers. Fertilizer and all types of pesticides (i.e., fungicides, herbicides, insecticides and other pesticides) are overused by Dutch cash crop farmers. The considerable level of risky outputs' inefficiency in conjunction with pesticides' overuse reveals a need to decrease pesticide use and their environmental spillovers. Therefore, pesticide policies aiming at optimal pesticide use may increase farmers' profitability by reducing pesticide use and improve environmental quality through reductions in pesticides' environmental spillovers.

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Chapter 7

General Discussion

7.1 Introduction

In modern agriculture, pesticides feature prominently in growers' arsenal to reduce crop damage caused by various pests and diseases. But their indiscriminate use can harm human health and the environment (Pimentel et. al., 1992; Pimentel and Greiner, 1997; Wilson & Tisdell 2001) and, eventually, impact agricultural productivity negatively. In an era of an increasing public awareness on the external effects of pesticides, the EU intends to update its pesticide policy by establishing tax and levy schemes. Information coming from empirical research on pesticide use and environmental spillovers at the farm level may assist policy makers in introducing optimal pesticide policies.

However, little is known about the impact of pesticides' environmental spillovers on output realization. Moreover, there is little farm-level empirical research investigating the impact of different economic instruments on pesticide use and environmental spillovers. The major objective of this thesis is to assess empirically the effect of pesticide use and environmental spillovers on farmers' production environment with a view toward contributing to the development and implementation of future pesticide policies.

The issues addressed in this research were: a) the composition of an optimal pesticide policy and the information needs for applying such a policy; b) the impact of pesticide use and environmental spillovers on output realization; c) the potential or ability of economic incentives and command and control approaches to alter farm practices by reducing pesticide use and environmental spillovers; d) farmers' technical and pesticides' environmental efficiency under pesticide dynamics and production uncertainty; and e) farmers' technical and allocative efficiency when considering undesirable outputs, pesticides as risk-mitigating inputs and taking explicitly into account the risk-increasing or decreasing nature of production inputs. Numerous implications for policy makers, scientists, and other stakeholders can be derived from these issues. These implications are going to be presented and discussed in this chapter.

The structure of this chapter proceeds as follows. The next section presents the approaches used in this thesis and information on their application. Section 7.3 reports the main results, while policy implications and recommendations for future research are presented in section 7.4 and 7.5, respectively.

7.2 Approach and implementation

The framework of the research comprises four approaches to address the five research questions. A review of the economics of pesticide use literature took place which enabled not only the identification of the contour of an optimal pesticide policy and the information needed for applying such a policy, but also provided the foundations for the development of a dynamic model of optimal pesticide use incorporating pesticides' environmental spillovers and takes into account explicitly the symmetric and asymmetric effect of pesticides' environmental spillovers on production. This model captures the impact of pesticide use and environmental spillovers on output realization, thus providing empirical evidence of the impact of farmland biodiversity on farmers' production environment. A similar dynamic model is used in a simulation process to identify the impact of economic incentives and command and control approaches on pesticide use and environmental spillovers. Data envelopment analysis is employed to measure the performance of farmers after adjusting outputs and inputs to account for the effect of variability in farmers' operating environment using a bootstrap-approach. Data envelopment analysis is also conducted to provide a risk-adjusted efficiency measurement of the performance of Dutch arable farmers, using undesirable outputs, risk-mitigating inputs, and taking explicitly into account the effect of production inputs on risk management. Secondary data are used for this study. These data included panel data of Dutch cash crop farms over the period 2002-2007 obtained from the Agricultural Economics Research Institute (LEI), data on pesticides' environmental spillovers from the Dutch Centre for Agriculture and Environment (CLM), and data on weather variables from the Royal Netherlands Meteorological Institute (KNMI, 2011).

7.3 Overview of findings

To reach the overall goal of this study, five research questions were addressed. The highlights for each research question are presented in this section.

Research question one:

What is the contour of an optimal pesticide policy scheme and what are the knowledge gaps to be addressed to support the design of optimal pesticide policies?

The optimal pesticide policy should involve incentives to achieve environmental and health standards. Concerning the information needs for the introduction of optimal pesticide policy frameworks, overuse or underuse of pesticides depends on the model specification employed and the crops under consideration. A clearer view of pesticide use trends should be obtained through more research on different EU countries and crops. Pesticide demand is inelastic, in general, implying that only high pesticide taxes may alter farmers' practices. Considering that high pesticide taxes may impact farm profit negatively, using a mixture of instruments and regulations can compensate for the deficiencies of each other. Consumers are in general willing to pay to reduce the environmental risks from pesticide use, implying that pesticide policies should inform and encourage farmers on low pesticide production practices. Incentives can be also provided for farmers forming organic or IPM production groups and thus gain from their collective capacity to establish a reputation for their products.

Data on pesticides' environmental spillovers may help policy makers in classifying pesticides according to toxic content, thus supporting the introduction of differentiated pesticide taxes. One of the reasons behind the absence of economic incentive-based pesticide policies that are tied to environmental indicators in several EU countries' pesticide policies may be the lack of data on pesticides environmental spillovers. As agronomic and climatic conditions differ among EU countries, country-specific research on the environmental effects of pesticides may help scientists form robust pesticide environmental indicators. This can enable policy makers to introduce pesticide policy schemes that will better reflect pesticides' potential environmental damage. These schemes can alter pesticide decisions at the farm level such that negative environmental spillovers of pesticides are reduced.

Research question two:

Are pesticides' impacts on biodiversity affecting agricultural output?

The impacts of pesticides on biodiversity are impacting the farmers' production environment significantly. More specifically, when increasing the pressure on water and soil organisms and biological controllers some output losses are realized as these organisms can have a beneficial impact on output by reducing crop damage through the control of pest populations, enhancing soil nutritional characteristics and contributing to increased crop pollination. The results also show that pesticides are overused on average. Overuse of pesticides may impact beneficial farm organisms negatively, pointing the need to reduce pesticide use and conserve farmland organisms.

Chapter 3 provides a contribution to the economics of pesticide use literature. It is the first time that a dynamic model of optimal pesticide use accounts for both the symmetric and asymmetric effect of the environmental spillovers of pesticides on output. Pesticides do not only protect crops from pests and diseases but also cause environmental damage. The integration of pesticides' environmental spillovers in farmers' production technology is an improvement compared to earlier specifications in terms of richness of the results, thus providing valuable information to policy makers aiming at introducing optimal pesticide policies.

Research question three:

Are pesticide tax and levy schemes effective in reducing pesticide use and environmental spillovers in Dutch arable farming?

No single tax or levy instrument can lead to a substantial reduction of pesticide use. Pesticide taxes as a single instrument can be characterized as ineffective since they yield small decreases in pesticide use and environmental spillovers. Pesticide tax schemes that put higher penalties on high toxicity than low toxicity pesticides do not result in the substitution. Farmers' beliefs on the effectiveness of high toxicity products in preventing crop damage and reducing output variability may explain farmers' reluctance to reduce the use of high toxicity products. However, pesticide taxes can have positive environmental side effects (decrease fertilizer use), raise tax revenues and finance subsidy schemes. Subsidies on low toxicity pesticides did not affect the use of high toxicity products while R&D of more environmental friendly products effectively reduced the environmental spillovers of pesticides. Pesticide quotas are more appropriate in reducing pesticide use and environmental spillovers in comparison to most of the pesticide tax and levy schemes employed in this study.

In conclusion, taxes are not effective in reducing pesticide use. A set of policy tools including both economic incentives and command and control regulations may better address the desired policy goals. The contribution of chapter 4 to the literature on the economics of pesticide use and pesticide policy analysis is threefold. First, the asymmetric effect of pesticides' environmental spillovers on crop production are explicitly incorporated into the analysis to test whether economic incentives can alter pesticide decisions at the farm level such that environmental spillovers of pesticides are reduced. Second, this chapter provides new insights to Dutch and EU policy makers on the impacts of different policy tools. In the absence of empirical research in the Netherlands on the farm-level impact of pesticide tax and levy schemes, the results from such empirical analysis may help policy makers in setting optimal pesticide policies. Third, this study provides a way to classify pesticides according to toxic contents, thus assisting policy makers in developing differentiated pesticide taxes.

Research question four:

What is Dutch farmers' technical and pesticides' environmental inefficiency when considering pesticide dynamic effects on biodiversity and production uncertainty?

The initial DEA evaluation shows that Dutch farmers have noticeable output inefficiency scores (21%) and high pesticide environmental inefficiency (24-25%) that reveal a considerable scope for decreasing pesticides' environmental spillovers. When ignoring pesticide dynamics (i.e., the environmental spillovers of pesticides) output inefficiency increases dramatically (48%). Large farms, operated by older decision makers, with high soil quality, increased presence of farmland biodiversity, and exposed to high precipitation rates tend to be more output inefficient. Concerning pesticides' environmental inefficiency, large farms are more inefficient in protecting the status of the environment and taking into account pesticides' future negative effects in their current production decisions.

After adjusting for the variation in production conditions, farmers' output and pesticide environmental inefficiency scores decrease. More specifically, average output and pesticide environmental inefficiency scores decrease by 24% and 46-50%, respectively. This result provides evidence that producers operating under unfavorable production conditions may be disadvantaged in the initial DEA evaluation not accounting for production uncertainty (i.e., the variation in production arising from climatic events and other random forces). Comparing the initial and the adjusted DEA evaluation we see that farmers' average output inefficiency is reduced from 21% to 16% while pesticide environmental inefficiency decreased from 24% to 13%. This result shows that managerial inefficiency accounts for only 13-16% of the inefficiency while production uncertainty contributes 5-11%. Production uncertainty leads to considerable profit loss for farmers (€ 9.57 thousand) revealing a need to reduce the economic damage through market-oriented risk management tools.

Based on the empirical findings in this section, the following conclusions can be derived. First, inefficiency scores that consider pesticide dynamics and variability in the operating environment reveal a considerable scope for improving the process of output realization and reducing the environmental spillovers of pesticides. Second, ignoring the dynamics of production and the effects of variability in production conditions when measuring farmers' performance may lead to an overestimation of farmers' inefficiency scores. Chapter 5 contributes to the recent literature (Chambers et al., 2011; Emvalomatis, 2011) by providing further evidence that efficiency levels can be distorted when using models that ignore production uncertainty.

Research question five:

What is Dutch farmers' technical and allocative inefficiency when accounting for undesirable outputs and the risk-increasing or decreasing nature of agricultural inputs?

Dutch farmers have considerable average output technical inefficiency (9%) and undesirable outputs' inefficiency score (9%). Results reveal a considerable scope for decreasing output variability and pesticides' environmental spillovers. The average technical inefficiency of risk-mitigating and variable inputs is 6% and 3%, respectively. Fungicides, insecticides, and herbicides decrease output variance while labour and other pesticides are marginal-risk increasing inputs (i.e., increase the variability of harvests under good growth conditions). All types of pesticides (i.e., fungicides, herbicides, insecticides, and other pesticides) are overused on average, indicating that farmers can increase their profitability by decreasing the use of pesticides.

In conclusion, the findings of this chapter suggest that considerable improvements may still be achieved in farmers' profitability and environmental status through reductions in pesticide use and environmental spillovers. More specifically, policy makers can focus on reducing pesticides' environmental spillovers by up to 9% reduction. The major contribution of chapter 6 to literature is the employment of a DEA model that accounts for the risk-increasing or decreasing nature of agricultural inputs. Unlike most of the DEA studies at the agricultural firm-level that account for risk as an exogenous factor (i.e., weather variables or statistical noise) (Chambers et al., 2011; Skevas et al., Chapter 5), this work presents a way to incorporate risk focusing on the risk behavior on the part of the producer. This is realized by including in the modeling framework undesirable outputs and risk-mitigating inputs and taking explicitly into account the impact of those inputs on farmers' risk behavior.

7.4 Policy implications

Pesticide taxes proved to be ineffective in reducing pesticide use and environmental spillovers while higher taxes on high toxicity products, and subsidies on the use of low toxicity products did not affect the use of high toxicity products. However, pesticide taxes should not be excluded from pesticide policy schemes due to their capacity to raise tax revenues that can finance subsidies (e.g. for R&D of more environmental friendly products) and extension and the fact that they have secondary environmental advantages arising from decreased fertilizer use that can lead to fertilizer contamination reductions. Subsidies on the development of more environmental friendly products can decrease pesticides' environmental spillovers considerably. These findings indicate that an optimal pesticide policy should not designed with a single instrument but will involve a mixture of policy tools including economic incentives and command and control approaches. The latter proved to be effective in reducing pesticide use and environmental spillovers, implying that command and control approaches can have a role in any pesticide policy. Education and extension can further enhance the effectiveness of individual policy tools. For instance, informing farmers on low toxicity substitutes may render differentiated according to pesticide toxicity tax schemes more effective. As this is the first Dutch study on the effectiveness of different economic instruments on pesticide use at the farm level and detailed data on pesticides' environmental spillovers at the farm level are recently available, the findings coming from their use in empirical work require further evaluation before they applied to pesticide policy.

The considerable pesticide environmental inefficiency of Dutch arable farmers imply that a reduction of pesticides' environmental spillovers can be achieved with the current technology (e.g. more precise pesticide applications). Alternatively, pesticide policies may trigger farmers in switching to more environmental friendly practices. As many studies have shown that consumers are willing to pay for higher environmental quality, providing incentives to farmers to adopt more environmental friendly practices can enhance environmental quality and protect farmers profit through increasing the number of beneficial farm organisms and/or taking advantage of the price premium of more environmental friendly products. Overuse of pesticides is a common finding of chapters 3 and 6 providing further evidence for the need to decrease pesticide use, thus increasing farmers' profitability and reducing environmental damage. Large farms tend to have a higher output and pesticide environmental inefficiency, suggesting that policies aiming at reducing pesticides' environmental spillovers can initially target those farms.

Another policy contribution of this work is that it provides a way to classify pesticides according to toxicity contents using the official classification from the Dutch Board for the authorization of pesticides(CTB) for pesticide toxicity in the Netherlands. Results coming from pesticide variables using this measure can help policy makers in designing more realistic pesticide policies. Classification of pesticides enables the introduction of differentiated according to toxicity taxes[1], thus better reflecting the potential environmental damage caused

[1] Sweden was one of the first countries to introduce a simple tax scheme based on an environmental levy of 30 SEK (3.25 €) per kg active substance. Norway had also introduced a tax system where the taxation level is banded by health and environmental properties (differentiated tax rates per hectare and standard area doses) (Lesinsky and Veverka,, 2006 ; OECD, 2008).

by pesticides. Instead of taxing pesticides, taxes can be placed on their environmental spillovers. Despite the absence of prices for environmental spillovers the tax can be a monetary value per environmental spillover. Spillovers on biological controllers and water organisms can be taxed at a higher rate as they have a significant negative impact on output realization.

Agricultural producers are operating in a stochastic environment where production dynamics and uncertainty are affecting production decisions. Models that ignore pesticide dynamics and variation in the operating environment can lead to biasing results, thereby possibly misinforming policy makers. As inefficiency due to production uncertainty counts for 5-11% leading to farmers' profit loss of around € 9.57 thousand on average, market-oriented risk management tools can mitigate farmers' economic damage. The considerable amount of profit loss raises concerns on the ability of farmers to cover it through market-oriented risk management tools. In any case, extension services should inform farmers on the economic damage caused by production uncertainty.

Moving from a country specific pesticide policy to an EU wide pesticide policy a mixture of policy instruments including both economic incentives and command and control approaches and accompanied by education and extension can possibly better address any set of policy goals. Differences in agronomic characteristics and employed pesticide products across EU states or even regions within a state should not pose an obstacle to forming an EU-wide pesticide policy. Heterogeneity across states' production characteristics shows that taxes and levies should be country- or region-specific with the overall pesticide policy being coordinated by the EU. Agronomic characteristics and more specifically pesticide use and impacts differ across states or regions. Uniformity of environmental indicators across states may help towards the harmonization of national pesticide policies. The environmental indicators developed in the Netherlands (CLM, 2010) can be an example of developing similar databases in other EU countries. Problems (such as unanimity in tax-related decisions) arising by harmonizing pesticide policies across all EU states may be surpassed by cooperation among groups of states.

7.5 Recommendations for future research

Pesticides' environmental spillovers do not only have farm level impacts but they may affect the production environment of other farms operating in the same region. Regional data on changes in biodiversity populations in conjunction with farm level data on pesticides' environmental spillovers can be included in a modeling framework, thus providing a more comprehensive bio-economic representation of the impact of pesticide decisions on both farms' and neighboring producers' production environment. Taking into account not only the farm-level impact of pesticides but also their impact on other agents' production choices can help researchers design models that better internalize pesticide externalities. Information on the production impacts of pesticides' environmental spillovers and the effect of each farmers' production decisions on biodiversity populations' changes allow for studying incentive systems in which the generator of the impact compensates the affected party. Modeling frameworks using compensation/payment and, in general, ex-post liability rules may enable researchers to test whether farmers can further reduce the use of highly toxic products and/or move towards more precise pesticide applications.

Detailed information on pest populations can also help to explicitly capture the effect of pest resistance on output realization. Although this effect can be reflected in the evolution of the environmental spillover variables (i.e., higher values of environmental spillovers are assumed to reflect higher pesticide applications due to resistance), data on pest populations can better reflect this issue. Incorporating both pesticides environmental spillovers and resistance development in modeling frameworks of optimal pesticide use can provide a better picture of the impact of pesticide practices on production.

Given the dynamic nature of agricultural production, assessing the impact of dynamics in farm-decision making and investigating whether farmers are rational when taking into account these dynamics can provide valuable information to researchers and policy makers. Stochastic as well as structural components can lead to dynamic linkages. Modeling frameworks that integrate dynamic linkages coming from both stochastic and structural components may provide a more overall and realistic representation of the drivers behind farmers' production decisions.

Differences in agronomic characteristics across Europe or even within states reveal a need to apply pesticide use modeling approaches to each country or region separately. Tax or levies as stand-alone measures may not be effective in reducing pesticide use and environment spillovers in the Netherlands, where potatoes is one of the most profitable arable crop requiring high number of preventive fungicide applications. But these measures may be effective in other countries were their agriculture is dominated by different crops. Incorporating environmental spillovers of pesticides at the farm level in modeling approaches is an addition to the literature of the economics of pesticide use. More empirical applications are needed to verify the robustness of the results of this dissertation. More specifically as this research shows that overuse of pesticides is a common finding in modeling frameworks that take into account pesticides' environmental spillovers, further investigation is needed to verify this finding possibly at country level where pesticide use and climatic and agronomic characteristics vary considerably.

Pesticides do not only pose environmental challenges but also affect human health through direct exposure of farm workers during pesticide applications or through the food chain. Research on the economics of pesticide use at the firm-level can be enriched further by simultaneously examining the effect of both environmental and health spillovers of pesticides on output realization. This might not be relevant for the Netherlands where farmers spray pesticides from a closed environment (i.e., tractors) and wear the appropriate protective equipment. However, it can be important in countries where farmers have considerably smaller acreages (i.e., where maybe it is not cost effective to use closed tractors for spraying) and do not use frequently the appropriate protective equipment.

Production uncertainty plays a decisive role in farmers' production choices. This dissertation demonstrated ways to modeling risk either as an exogenous (e.g., adjusting outputs and inputs for changes in weather conditions) or an endogenous impact (i.e., considering risk behaviour on the part of the producer reflected by choices on risk-mitigating inputs and management of undesirable outputs). An interesting research avenue might be to develop a modeling approach that takes into account risk both as an endogenous and exogenous impact. For instance, when detailed meteorological data at the farm-level are available, using the risk-adjusted efficiency measurement developed in chapter 6 for separate groups of farms classified according to the intensity of climatic events may provide more insights of how farmers manage production risk.

Technology diffusion in agriculture is closely related, among others, to pesticide use. For instance, new pesticide products that are more effective in combating pest damage and simultaneously more environmental friendly may appear in the market, and genetically modified crops are related with lower pesticide applications. New agricultural technologies can enhance farm productivity and contribute significantly to pesticide policies that aim to reduce pesticide use and environmental spillovers. But new technologies may imply extra costs due to compliance with policy regulations. The implications of the diffusion of new technologies in the agricultural sector for shifting patterns of production and resource use are providing new pathways in empirical research.

7.6 Conclusions

The major conclusions of this thesis are:

1. Incentives to achieve environmental and health standards should be part of pesticide policies. Economic incentive-based pesticide policies that are tied to environmental indicators can benefit from country specific research on the environmental effects of pesticides, as agronomic and climatic conditions differ among EU countries and regions.
2. Pesticides are overused in Dutch arable farming. Overuse of pesticides implies that farmers could increase their profitability by decreasing pesticide use. Pesticide policy frameworks should aim at reducing pesticide use in order to increase farmers' profitability and reduce environmental damage.
3. Pesticides' effects on biodiversity are affecting agricultural output significantly. Therefore, when increasing the pressure on these organisms some output changes are realized.
4. Pesticide taxes are not effective in reducing pesticide use and environmental spillovers. However, pesticide taxes can be part of pesticide policies due to their capacity to raise tax revenues that can finance subsidies (e.g. for R&D of more environmental friendly products) and the fact that they have secondary environmental advantages (e.g.

decreased fertilizer use that can lead to fertilizer contamination reductions). Pesticide quotas are more effective in reducing pesticide use and environmental spillovers.

5. Inefficiency measurement revealed a considerable scope for decreasing pesticides' environmental spillovers. This implies that a reduction of pesticides' environmental spillovers can be achieved with the current technology, through better management of agricultural practices such as more precise pesticide applications.
6. Technical inefficiency decreased dramatically when accounting for production uncertainty. Agricultural producers are operating in a stochastic environment where production uncertainty affects their decisions. As production uncertainty is an integral part of farmers' production environment, models that ignore it, can lead to erroneous results.

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Summary

During the second half of the last century agricultural firms faced great changes with agricultural intensification being one of the most important ones. Being part of this intensification, pesticides are highly used in agricultural production of developed countries to prevent and combat crop damage, thus securing crop yields. Pesticides have an indirect effect on output rather than a direct yield-increasing effect. Pesticides reduce pest damage and enable farmers to obtain high quality products that can have a positive impact on their revenues. Despite the positive outcomes from the use of pesticides, adverse impacts to human health and the environment are a related consequence. Continuous use of chemical inputs such as pesticides produces significant negative impacts that have been broadly documented in the scientific literature. Negative impacts on flora and fauna, reduced numbers of beneficial pest predators, development of pesticide resistant weeds and pests, aquifers' polution, fishery loss, contaminated products and bee poisonings are some of the adverse effects of pesticides.

As public awareness on the indirect effects of pesticides on the environment and human health is increasing, European Union (EU) seeks to update its pesticide policy by using economic incentives to reduce pesticide use and their spillovers. Currently, a few EU countries, such as Sweden and Norway, have embedded economic incentives in their pesticide policy but it is difficult to separate the impact of taxation on pesticide use from other factors influencing farmers' use decisions (e.g., switch to low dose agents, conversion to organic farming, improved pesticide technologies and management). Although tax and levy systems in these countries may assist policy makers in developing tax and levy schemes, farm-level empirical research on pesticide use and environmental spillovers is needed. Implementation of economic incentive-based pesticide policies require detailed information on pesticide use and environmental spillovers thus pointing to the importance of farm-level approaches in primary policy analysis.

However, little empirical research has been done on the impacts of both pesticide use and environmental spillovers on output realization. Also, little empirical evidence exists on the impacts of pesticide policy tools (such as economic incentives and command and control approaches) on farmers' pesticide use decisions and environmental spillovers. The objective of

this thesis is to examine empirically the impact of pesticide use and environmental spillovers on output realization, thus providing evidence for the implementation of future pesticide policies.

In Chapter 2, a literature review took place to identify the contour of an optimal pesticide policy and the information needs for applying such a policy. These information include knowledge on the production structure (i.e., production function, pesticide demand elasticities), attitudes toward risk and uncertainty related to pesticides application, the value of pesticides to consumers (e.g., the willingness to pay (WTP) for lower pesticide use), and the effects of pesticide use on biodiversity in relation to existing pesticide policies. The literature review produced a number of key findings: more research (at regional or state level) is needed on examining whether pesticides are over- or under-utilized as pesticide use trends depend on the employed model specification and the crops under consideration; pesticide demand is inelastic, implying that high and possibly politically problematic taxes are needed to reduce pesticide use; consumers are willing to pay to reduce pesticides' environmental spillovers; country specific research on pesticides' environmental spillovers can assist policy makers in introducing pesticide policies that will better reflect pesticides' potential environmental damage and alter farmers' pesticide use decisions.

In Chapter 3, a dynamic model of optimal pesticide use is used to assess the impact of pesticide use and environmental spillovers (i.e., impacts on farmland biodiversity) on output realization. Two pesticide categories are used (i.e., high and low toxicity pesticides) and both the symmetric and asymmetric effect of pesticides' environmental spillovers on output is taken into account. Including environmental spillovers of pesticides on farmers' production technology is an improvement compared to earlier studies in terms of richness of the results. Results show that pesticide impacts on biodiversity have a significant effect on agricultural output and both types of pesticides are overused in Dutch arable farming. The importance of farmland biodiversity in output realization in conjunction with pesticide overuse show a need to reduce pesticide use and protect farmland organisms. This chapter contributes to the economics of pesticide use literature by internalizing effectively in a dynamic model of optimal pesticide use the environmental spillovers of pesticides.

In Chapter 4, a dynamic model of optimal pesticide use is developed and estimated econometrically. After the econometric estimation, a simulation model of optimal pesticide use is employed to investigate the impact of different economic incentives and command and control measures on pesticide use and environmental spillovers. The economic incentives employed in this study include uniform and differentiated taxes on high and low toxicity products, subsidies on the use of low toxicity products and R&D of more environmental friendly products while the command and control measures include pesticide quotas. The results of the simulation analysis indicate that taxes as stand-alone measures are not effective in reducing pesticide use and environmental spillovers. Differentiated taxes and subsidies on the use of low toxicity products did not affect the use of high toxicity products. Farmers' rigidity in reducing the use of high toxicity products or substitute them with low toxicity ones may be explained by high toxicity products' capacity to prevent crop damage and reduce output variability. R&D of more environmental friendly products can reduce pesticides' environmental spillovers while pesticide quotas are more effective in reducing pesticide use and their environmental impacts in comparison to most of the employed instruments. A pesticide policy framework including both economic incentives and command and control approaches may be more effective in achieving pesticide policy targets. This chapter contributes to the pesticide policy literature by providing new evidence to Dutch and EU policy makers on the impact of pesticide policy tools on pesticide use and environmental spillovers.

In chapter 5, a dynamic Data Envelopment Analysis (DEA) model is used to investigating the performance of Dutch arable farms when taking into account pesticide dynamics and production uncertainty (i.e., variability in production conditions due to weather events and the state of nature). Technical inefficiency is explained using the Simar and Wilson (2007) double-bootstrap procedure with socioeconomic and environmental variables, thus providing empirical representations of the impact of stochastic elements and the state of nature on production. Firms' outputs and inputs are adjusted using the results of the double-bootstrap procedure to account for the impact of variability in production conditions. Finally, the dynamic DEA model is applied to adjusted outputs, inputs and undesirables (i.e., pesticides environmental spillovers). The results of the initial DEA evaluation show that Dutch arable farmers have noticeable output inefficiency scores (21%) and high pesticide environmental inefficiency (24-25%). Large farms have higher pesticide environmental inefficiency; i.e., are more inefficient in protecting the status of the environment and taking into account pesticides' future negative effects in their current production decisions. When adjusting outputs and inputs for the variation in production conditions farmers' output and pesticide environmental inefficiency scores decrease; average output and pesticide environmental inefficiency scores decrease by 24% and 46-50%, respectively. The initial DEA evaluation shows that Dutch arable farmers are for output and undesirables on average around 21% and 24%, respectively, below the production frontier, while with the adjusted DEA model farmers' inefficiency is reduced to around 16% and 13%, respectively. This results shows that managerial inefficiency accounts for only around 13-16% while another 5-11% is attributed to production uncertainty. This amount of production uncertainty is translated to farmers' profit loss of around € 9.57 thousand on average, revealing a need to mitigate it through market-oriented risk management tools. This chapter provides evidence that efficiency levels can be distorted when using models that ignore production uncertainty.

In chapter 6, a risk-adjusted DEA model is used to measure technical and allocative inefficiency of Dutch arable farms. The DEA model uses undesirable outputs and risk-mitigating inputs and takes explicitly into account the risk-increasing or –decreasing effect of production inputs on output realization. Results show that fungicides, insecticides, and herbicides are marginal-risk decreasing inputs while labour and other pesticides are marginal-risk increasing inputs. Results further indicate that Dutch arable farmers have considerable output technical inefficiency (9%) and undesirables inefficiency score (9%). These findings show that pesticides' environmental spillovers can be reduced by up to 9%. Fungicides, herbicides, insecticides, and other pesticides are overused on average, revealing a considerable scope for increasing farmers' profitability through decreases in pesticide use. This chapter contributes to the literature by providing a risk-adjusted efficiency model that takes explicitly into account the impact of production inputs on output variability.

Numerous conclusions and policy implications can be drawn from this dissertation. First, as farmland biodiversity plays an important role in farmers' production environment and pesticides are overused in Dutch arable farming, pesticide policies should try to conserve farmland organisms and reduce pesticide use. Second, tax and levy schemes are not effective in reducing pesticide use and environmental spillovers but they can still have a share in pesticide policies as they can finance subsidies (as subsidies on more environmental friendly products

can reduce the environmental spillovers of pesticides) and decrease fertilizer use leading to fertilizer contamination reductions. Pesticide quotas are more effective in reducing pesticide use and their indirect impacts on biodiversity in comparison to most of the examined economic incentives. Therefore, pesticide policies need not entail a single policy tool but should involve a mixture of measures including both economic incentives and command and control approaches. Third, Dutch farmers considerable environmental inefficiency shows that a reduction of pesticides' environmental spillovers can be achieved with the current technology. This may be realized by better management of pesticide use such as higher precision of pesticide applications. Fourth, modeling frameworks that ignore pesticide dynamics and variation in the operating environment can lead to erroneous results and conclusions. Fifth, the differences in agronomic and climatic characteristics of different countries and regions require the application of country or region specific pesticide modeling frameworks. Such frameworks can be benefited from collection of country or region specific pesticide impact data as biodiversity, employed crops and production practices may differ considerably among different countries or regions.

Samenvatting

In de tweede helft van de vorige eeuw ondergingen landbouwbedrijven grote veranderingen in de intensiteit. De grotere intensiteit ging samen met een groter gebruik van pesticiden in de landbouw van ontwikkelde landen, ter preventie van schade en om oogsten zeker te stellen. Pesticiden hebben eerder een indirect effect op output in plaats van een direct opbrengst verhogend effect. Pesticiden verminderen de schade als gevolg van ziekten en plagen en maken het mogelijk dat agrarische ondernemers een goede kwaliteit kunnen afleveren en hoge revenuen kunnen genereren. Naast positieve effecten van pesticiden, zijn er ook negatieve consequenties voor de humane gezondheid en het milieu. Een voortdurend gebruik van pesticiden heeft negatieve effecten die goed zijn gedocumenteerd in de literatuur. Voorbeelden van negatieve effecten zijn: effecten op flora en fauna, minder natuurlijke vijanden, ontwikkeling van pesticiden resistentie, vervuiling van grondwater, residuen in voedsel en vergiftiging van vissen en bijenpopulaties.

De toename van het bewustzijn van burgers van de negatieve effecten heeft geleid tot meer beleid vanuit de EU om de effecten te beteugelen. Daarbij onderzoekt de EU ook de mogelijkheden van economische incentives. Op dit moment hebben Zweden en Noorwegen economische incentives opgenomen in hun pesticiden beleid. Het exacte effect van economische incentives vis-à-vis het gebruik van lage doseringen, verbeterde toedieningstechnieken en biologische landbouw, is echter moeilijk vast te stellen. Empirisch onderzoek is nodig om te onderzoeken of belastingen en heffingen kunnen bijdragen aan een terugdringing van het gebruik van pesticiden. Implementatie van pesticiden beleid gebaseerd op economische incentives vraagt gedetailleerde informatie over pesticiden gebruik en milieu effecten. Empirisch onderzoek op het niveau van landbouwbedrijven is daarbij essentieel. Desalniettemin is er tot nu toe weinig onderzoek geweest naar de effecten van pesticiden en milieu spillovers op de gerealiseerde output van landbouwbedrijven. Ook zijn er weinig empirisch onderzoeksresultaten over de effecten van pesticiden beleid op beslissingen op bedrijfsniveau over het pesticiden gebruik en milieu spillovers. Het doel van deze thesis is om empirisch het effect te bepalen van pesticidengebruik en milieu spillovers op de gerealiseerde output om daarmee inzicht te verschaffen in de mogelijkheden van economische incentives in het pesticidenbeleid.

In Hoofdstuk 2 is een literatuur review uitgevoerd om de contouren te schetsen van een optimaal pesticiden beleid en om vast te stellen wat de daarbij behorende informatiebehoefte is. De informatiebehoefte omvat kennis van de productiestructuur (productiefunctie, vraag naar pesticiden), houding ten opzichte van risico en onzekerheid over pesticidengebruik, de willingness to pay (WTP) van consumenten voor een lager pesticiden gebruik en de effecten van pesticidengebruik op biodiversiteit. De belangrijkste resultaten van de literatuur review zijn: meer onderzoek (op regionaal of nationaal niveau) is nodig om vast te stellen of er sprake is van overmatig dan wel een te laag gebruik van pesticiden; trends in het gebruik van pesticiden hangen af van de gebruikte model specificatie en de gewassen die worden meegenomen in de analyse; de vraag naar pesticiden is inelastisch, wat impliceert dat hoge belastingen of heffingen nodig zijn om het gebruik te verminderen; consumenten zijn bereid om te betalen voor een lager pesticiden gebruik en vermindering van milieu spillovers. Onderzoek dat specifiek voor individuele landen wordt uitgevoerd kan behulpzaam zijn bij het ontwerpen van pesticiden beleid waarin milieu schade beter wordt meegenomen en dat leidt tot daadwerkelijke veranderingen in de beslissingen van boeren over pesticiden gebruik.

In Hoofdstuk 3 wordt een dynamisch model van optimaal pesticiden gebruik ingezet om de impact te bepalen van pesticidengebruik en milieu spillovers op de gerealiseerde output. Het model omvat twee categorieën van pesticiden (hoge en lage toxiciteit) en zowel de symmetrische en asymmetrische effecten van milieu spillovers op output worden meegenomen. Het meenemen van milieu spillovers van pesticiden op de productie technologie van bedrijven is een verbetering van modellen die gebruikt worden in de huidige literatuur. De resultaten laten zien dat de impact van pesticiden op biodiversiteit een significant effect heeft op de output. Ook laten de resultaten zien dat beide categorieën van pesticiden overmatig worden gebruik op Nederlandse akkerbouwbedrijven. Het belang van biodiversiteit op de gerealiseerde output in samenhang met het overmatig gebruik van pesticiden laten zien dat een reductie van pesticiden nodig is, ter bescherming van biodiversiteit. Dit hoofdstuk draagt bij aan de literatuur over de economie van pesticiden gebruik door de milieu spillovers op te nemen in een dynamisch model.

In Hoofdstuk 4 wordt een dynamisch model van optimaal pesticiden gebruik ontwikkeld en econometrisch geschat. Met behulp van de geschatte parameters wordt een simulatie model geconstrueerd waarmee de effecten van verschillende economische incentive mechanismen op pesticden gebruik en milieu spillovers worden onderzocht. De economische incentives in dit onderzoek zijn uniforme en gedifferentieerde belastingen op pesticiden met een hoge en lage toxiciteit, subsidies op het gebruik van pesticiden met een lage toxiciteit en R&D investeringen in meer milieu vriendelijke producten. Daarnaast worden pesticiden quota onderzocht. De simulaties geven aan dat belastingen nauwelijks effect hebben op het pesticiden gebruik en de milieu spillovers. Gedifferentieerde belastingen op hoge toxiciteit pesticiden en subsidies op het gebruik van pesticiden met een lage toxiciteit hadden geen effect op het gebruik van de hoge toxiciteit pesticiden. Deze resultaten suggereren dat hoge toxiciteit pesticiden zeer effectief zijn in het voorkomen van gewasschade. R&D in meer milieuvriendelijke producten kan leiden tot lagere milieu spillovers. Pesticiden quota zijn het effectiefst van alle onderzochte instrumenten in het reduceren van het pesticiden gebruik en spillovers. Pesticiden beleid dat economische incentives combineert met quota kan daarom effectiever zijn in het behalen van de beleidsdoelstellingen. Dit hoofdstuk draagt bij aan de literatuur over pesticiden beleid door nieuwe inzichten te verschaffen aan Nederlandse en EU beleidsmakers over het terugdringen van het pesticiden gebruik en milieu spillovers.

In Hoofdstuk 5 wordt een dynamisch Data Envelopment Analysis (DEA) model gebruikt om de prestaties van Nederlandse akkerbouwbedrijven te meten. Het model houdt rekening met de dynamische effecten van pesticiden en onzekerheid over de productie omstandigheden (b.v. als gevolg van het weer). Technische inefficiëntie wordt verklaard door een regressie op sociaal-economische en omgevingsvariabelen met behulp van een dubbele bootstrap procedure. De inputs en outputs van bedrijven worden aangepast met behulp van de resultaten van de dubbele double-bootstrap procedure om te corrigeren voor onzekerheid over de productie omstandigheden. Het dynamische DEA model wordt tenslotte toegepast op de aangepaste inputs, outputs en milieu spillovers. De resultaten van de DEA laten een grote inefficiëntie zien (21%) en een hoge milieu inefficiëntie (24-25%). Grote bedrijven hebben een hogere milieu inefficiëntie; ze zijn dus minder goed in staat om het milieu te beschermen en nemen de negatieve effecten van pesticiden op de toekomstige gerealiseerde output ook minder mee in hun huidige beslissingen. Na aanpassing van inputs en output voor onzekerheid worden de output milieu inefficiëntie scores lager; de gemiddelde output en milieu inefficiëntie scores verminderen met respectievelijk 24% en 46-50%. Na aanpassing voor onzekerheid liggen de output en milieu inefficiëntie scores rond de 16 en 13% onder de productie frontier. Deze resultaten laten zien dat inefficiëntie die kan worden toegerekend aan management slechts 13-16% is, terwijl 5-11% inefficiëntie kan worden toegerekend aan productie onzekerheid. Deze productie onzekerheid leidt tot verliezen van gemiddeld 9.57 duizend euro; dit geeft aan dat er behoefte is aan risico management tools om de consequenties van onzekerheid te verminderen. Dit hoofdstuk toont aan dat inefficiëntie schattingen kunnen worden verstoord door onzekerheid over de productie omstandigheden.

In Hoofdstuk 6 wordt een risico aangepast DEA model gebruikt om de technische en allocatieve inefficiëntie van Nederlandse akkerbouwbedrijven te meten. Het DEA model onderscheidt ongewenste outputs and risk-verminderende inputs en neemt expliciet het risico-vergrotende of verminderende effect van inputs op de gerealiseerde output in acht. De resultaten laten zien dat fungiciden, insecticiden, en herbiciden marginaal risico verminderende inputs zijn, terwijl arbeid en overige pesticiden marginaal risico-verhogende inputs zijn. De resultaten laten verder zien dat Nederlandse akkerbouw bedrijven een technische output inefficiëntie hebben van 9% en dat de ongewenste output inefficiëntie eveneens 9% is. Deze resultaten laten zien dat milieu spillovers kunnen worden gereduceerd met 9%. Fungiciden, herbiciden, insecticide en overige pesticiden worden overmatig gebruikt, wat impliceert dat bedrijven hun winstgevendheid kunnen verbeteren via een afname van het pesticiden gebruik. Dit hoofdstuk draagt bij aan de DEA literatuur door een risico-aangepast DEA model te ontwikkelen en expliciet het effect van inputs op de variabiliteit van output mee te nemen.

Verschillende conclusies en beleidsimplicaties kunnen worden ontleend aan deze thesis. Ten eerste, beleid moet zich richten op het terugdringen van pesticiden gebruik en het behoud van biodversiteit, aangezien biodiversiteit een belangrijk aspect is van de productie omgeving van, en pesticiden overmatig worden gebruikt, op Nederlandse akkerbouwbedrijven. Ten tweede, belastingen en heffingen zijn niet geschikt als middel voor het terugdringen van het pesticiden gebruik en milieu spillovers. Echter, heffingen kunnen wel gebruikt worden voor de financiering van subsidies op milieuvriendelijke middelen en kunnen leiden tot een lager gebruik van kunstmest. Pesticiden quota zijn effectiever dan andere economische incentives (heffingen, belastingen, subsidies) in terugdringen van het pesticiden gebruik en de indirect gevolgen op biodiversiteit. Pesticiden beleid moet daarom niet bestaan uit één enkele tool, maar moet een mix zijn van economische en command en control mechanismen. Ten derde, de aanzienlijke milieu inefficiëntie van Nederlandse boeren suggereert dat een reductie van het pesticiden gebruik en de milieu spillovers kan worden bereikt met de huidige technologie. Dit kan worden gerealiseerd door een beter management van het pesticidengebruik, b.v. door betere applicatie technieken. Ten vierde, model raamwerken die de dynamische effecten van pesticiden en onzekerheid over de productieomstandigheden niet meenemen kunnen tot foutieve resultaten en conclusies leiden. Ten vijfde, de verschillen in agronomische en klimatologische omstandigheden in vragen om een land- en regio specifieke modellering van het pesticiden gebruik en spillovers. Om deze modellen op een zinvolle manier te kunnen toepassen moeten in de toekomst gegevens worden verzameld in verschillende regio's of landen over de impact van pesticiden, zoals biodiversiteit, gewassen en productieomstandigheden.

Training and Supervision Plan

Name of the activity	Department/Institute	Year	ECTS*
Project related competences			
Irreversibilities Uncertainties and Real Option Values	MG3S	2008	4
Dynamic Efficiency and Productivity Analysis	MG3S	2008	4
Advanced Production Economics AEREC 534	Penn State University, USA	2009	6
Advanced Agricultural Business Economics BEC30306	BEC	2009	6
General research related competences			
Advanced Econometrics AEP50806	AEP	2008	6
Econometrics I	Penn State University, USA	2009	6
Mansholt Introduction course	MG3S	2008	1,5
'Do farmers internalize external impacts of pesticides in production?'	EAAE Seminar, Chania, Greece	2010	1
'Using economic instruments in pesticide policy - A case study of Dutch cash-crop farmers'	EAAE Congress, Zurich, Switzerland	2011	1
Career related competences/personal development			
PhD Competence Assessment	WGS	2009	0.3
Teaching methodology and skills for PhD students	DO	2009	1.8
Total			37.6

*1 ECTS on average is equivalent to 28 hours of course work.

Colophon

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