Institut für Lebensmittel- und Ressourcenökonomik

Farm-level impacts of policy instruments targeting plant protection products

Dissertation

zur Erlangung des Grades

Doktor der Agrarwissenschaften (Dr. agr.)

der Landwirtschaftlichen Fakultät

der Rheinischen Friedrich-Wilhelms-Universität Bonn

von

Thomas Böcker

aus

Lingen (Ems)

Bonn, 2018

Referent:	PD Dr. Wolfgang Britz
Korreferenten:	Prof. Dr. Robert Finger (ETH Zürich)
	Prof. Dr. Thomas Heckelei
Tag der mündlichen Prüfung:	19. Juli 2018

Angefertigt mit Genehmigung der Landwirtschaftlichen Fakultät der Universität Bonn

Acknowledgements

At this point, I would like to thank my colleagues, relatives and friends for helping me completing this dissertation. First, I thank Prof. Dr. Robert Finger and PD Dr. Wolfgang Britz for giving me the possibility to do research in a very interesting area at the University of Bonn and now at ETH Zurich. I am very grateful for your constructive feedback, the topical discussions, the freedom I had in doing research and the nice working atmosphere. In addition, I like to thank my colleagues from the Institute of Food and Resource Economics (ILR) for three wonderful years. We had very nice discussions and after-work events. Many credits go to the Mensa Gang for the relaxing and saturating lunch breaks.

Many thanks go also to the co-authors of my articles and project reports, Niklas Möhring and Tobias Dalhaus. It was a pleasure working together. I also thank Ganga Ram Maharjan and Thomas Gaiser from the Crop Science Group at the University of Bonn for providing yield data for my research. Furthermore, I am grateful to Günter Klingenhagen, Reinhard Wiggenhorn, Hermann Klockenbusch, Bernd Rüter (Landwirtschaftskammer Nordhrein-Westfalen) and Markus Mücke (Landwirtschaftskammer Niedersachsen) for their expert contributions. Moreover, I thank Anne Strøm Prestvik of the Norwegian Institute of Bioeconomy Research for providing important information. I also acknowledge the support and the project funding of the Swiss Federal Office for Agriculture.

Finally, I am very grateful to my family and friends that supported me over the years. Without them, I would have not completed this dissertation. Very sincere thanks to Iris. She has been backing me up and standing by me during the past years.

Abstract

Plant protection is necessary to achieve high yields with high quality. The application of pesticides, however, leads to negative external effects to the environment. Society calls for stricter agri-environmental instruments in order to control pesticide use on farms. Intensively discussed topics are the introduction of a specific tax on pesticides and bans on pesticides such as glyphosate. The aim of this dissertation is to analyse potential effects of those two policy instruments on pesticide use on farms.

This dissertation contains four studies. The first two studies deal with potential effects of specific pesticide taxes from a more general perspective. Gaps in literature on whether a pesticide tax can be effective in achieving policy objectives and whether farmers respond to price increases of pesticide products are closed. In study three and four, a bio-economic model is developed in order to analyse potential effects of agri-environmental policy instruments in detail. Furthermore, these two studies close knowledge gaps by analysing economic and environmental effects of a glyphosate ban for the example of silage maize production.

Both policy instruments could contribute to reducing the environmental risks caused by pesticide application. However, in case of a pesticide tax, the design is important and should be in line with policy goals. Differentiated pesticide taxes that impose higher taxes on more toxic products are more suitable than *ad valorem* or per unit taxes. In case of a glyphosate ban, a significant decrease of the pesticide load with respect to environmental toxicity, environmental fate and human health was found. On the other hand, a glyphosate ban would lead to an increase in energy consumption. Introducing measures to substitute pesticide applications by mechanical strategies thus reduce energy efficiency on farms.

Farmer’s costs increase by a pesticide tax as well as a glyphosate ban. In the first case, the own-price elasticity of demand for pesticides was found to be inelastic, meaning that the demand for pesticides decreases relatively little if the price increases. However, demand is heterogeneous for different types of pesticides and herbicide demand was found to be relatively more elastic compared to fungicide and insecticide demand. In the second case, the substitution of glyphosate with mechanical strategies is more expensive and leads to higher labour demand. Moreover, a small but significant yield reduction was found in the analysis, at least in case of a glyphosate ban. On average, those losses do not lead to a significant decrease of the gross margin in this analysis. This is because the optimal control strategy was changed from direct or strip-till sowing with glyphosate application to conservation tillage without glyphosate application, which leads to lower sowing costs. Instead of a full replacement of pesticides because of policy pressure adapting the cropping strategy can thus help to mitigate losses at the farm-level.

Zusammenfassung

Pflanzenschutz ist essentiell, um hohe Erträge mit guter Qualität zu erzielen. Die Anwendung von Pflanzenschutzmitteln (PSM) hat jedoch auch negative Umweltauswirkungen. Weite Teile der Gesellschaft fordern deshalb strengere Auflagen, um den PSM-Einsatz zu reduzieren. In den letzten Jahren wurden u.a. die Einführung einer PSM-Abgabe und ein Verbot des Herbizids Glyphosat intensiv diskutiert. Das Ziel dieser Arbeit ist, die potentiellen Effekte dieser beiden Instrumente auf die Landwirtschaft zu analysieren.

Die Dissertation beinhaltet vier Teilstudien. Die ersten beiden beschäftigen sich mit den Auswirkungen einer PSM-Abgabe. Hierbei wurden die Forschungsfragen beantwortet, ob Abgaben ein effektives Mittel sind, um umweltpolitische Ziele zu erreichen und wie landwirtschaftliche Betriebe auf Preisänderungen reagieren. In der dritten und vierten Studie wurde ein bio-ökonomisches Modell entwickelt, um den PSM-Einsatz detailliert zu untersuchen. Außerdem beantworten diese beiden Studien offene Fragen zu den möglichen ökonomischen und ökologischen Auswirkungen eines Glyphosatverbots am Beispiel des Silomaisanbaus in Nordrhein-Westfalen.

Beide Instrumente könnten dazu beitragen, die ökologischen Risiken des PSM-Einsatzes zu reduzieren. Allerdings ist im Falle einer Abgabe die Ausgestaltung maßgeblich und sie sollte mit den beschlossenen umweltpolitischen Zielen übereinstimmen. Differenzierte Abgaben, die toxischere PSM höher belasten, sind besser geeignet als Wert- oder Mengenabgaben. Im Falle eines Glyphosatverbots wurde einerseits eine signifikante Reduzierung der Umwelttoxizität, des Umweltverhalten und der Belastung für die menschliche Gesundheit festgestellt. Andererseits wurde aber ein Anstieg des Energieverbrauchs ermittelt. Maßnahmen zur Reduzierung des PSM-Einsatzes könnten somit zu einer Senkung der Energieeffizienz führen.

Die Kosten der Betriebe werden sowohl durch eine Abgabe als auch durch ein Glyphosatverbot steigen. Im ersten Fall ist die Eigenpreiselastizität der Nachfrage unelastisch. Bei steigenden Preisen sinkt die Nachfrage nach PSM somit relativ schwach. Allerdings ist die Nachfrage heterogen, was u.a. bedeutet, dass die Nachfrage nach Herbiziden elastischer ist im Vergleich zu Insektiziden und Fungiziden. Im zweiten Fall wird Glyphosat durch teurere und arbeitsintensivere mechanische Unkrautbekämpfungsstrategien ersetzt. Zudem wurde, zumindest im Falle eines Glyphosatverbots, eine geringe aber signifikante Ertragsreduktion festgestellt. Im Durchschnitt führen diese Verluste in dieser Analyse hingegen nicht zu einer Reduktion des Deckungsbeitrags, da die optimale Unkrautbekämpfungsstrategie von Direktsaat- bzw. *strip-till*-Verfahren mit Glyphosateinsatz zu nichtwendender Bodenbearbeitung wechselt. Dies führt in der Simulation zu Einsparungen bei der Aussaat. Anstelle einer vollständigen Substitution von PSM als Folge politischen Drucks kann deshalb eine Anpassung der Anbaustrategie helfen die Verluste landwirtschaftlicher Betriebe gering zu halten.

Farm-level impacts of policy instruments targeting plant protection products

Contents

List of Figures

Chapter 5

List of Tables

Chapter 5

Abbreviations

AS	active substance
Ca	calcium
CSV	comma-separated values
Δ (delta)	difference
DKK	Danish krone
E	input energy (E_d + E_i)
E_{cp}	direct energy consumption for harvest compaction
E_d	direct input energy
E_f	direct energy consumption for fertilisation
E_{fw}	direct energy consumption for harvest field work
E_h	indirect energy consumption for active substance manufacturing
E_{hr}	indirect energy consumption for harvester manufacturing
E_i	indirect input energy
E_{om}	indirect energy consumption for other machinery manufacturing
E_{tm}	indirect energy consumption for tillage machinery manufacturing
E_{tp}	direct energy consumption for harvest transport
E_{tr}	indirect energy consumption for tractor manufacturing
EO	energy output
Eq.	equation
GDD	growing degree-days
GE	gross energy
h	hour
ha	hectare
H_s	Higher heating value
J|KJ|MJ|GJ|TJ	joule | kilojoule | megajoule | gigajoule | terrajoule
K	potassium
kg	kilogramme
l	litre
Λ_{fate} (lambda)	environmental fate load
Λ_{heal} (lambda)	human health load

Λ_{total} (lambda)	total environmental load
Λ_{toxy} (lambda)	environmental toxicity load
m|km	metre | kilometre
N	nitrogen
NA	not applicable
NAP	national action plan
NEO	net energy output
NfE	nitrogen-free extract
NOK	Norwegian krone
NRW	North-Rhine-Westphalia
P_2O_5	phosphorus pentoxide
PLI	Pesticide Load Indicator
r_a	partial risk aversion coefficient
ρ (rho)	density
SAD	standard area dose
SEK	Swedish krona
t|dt	metric tonne | decitonne
T	time of maize emergence compared to weeds
TGAP	*taxe générale sur les activités polluantes*
VAT	value added tax
v_f	volume of diesel for fertilisation
v_{fw}	volume of diesel for harvest field work
v_{tp}	volume of diesel for harvest transport
v_{cp}	volume of diesel for compaction
XP	crude protein content
XL	crude fat content
XF	crude fibre content

Chapter 1

Introduction

Plant protection has always played an essential role in the cultivation of crops (*e.g.* Jaskolla, 2006). High yields with high quality have hardly been achieved without protecting crops from external influences. While some of the well-known plant protection strategies like crop rotation or ploughing are still applied today, other techniques, often labour and capital intensive ones like weeding or collection of insects by hand, were substituted by modern plant protection strategies starting in the mid 20th century (Swinton and Van Deynze, 2017). Modern plant protection is based mainly on applying synthetic plant protection products.[1] Their application increased quickly since they were introduced and the high yield levels that are reached in today's agriculture are amongst others credited to the application of plant protection products (Oerke, 2006).

The downside of achieving higher yield levels are negative external effects on the environment that often go hand in hand with the application of pesticides, even though, in Europe, they go through a detailed risk assessment before they are approved in the European Union (Regulation EC No 1107/2009). Concerns regarding pesticide use are, for example, negative effects on biodiversity (*e.g.* Hallmann *et al.*, 2017), potentially dangerous residues in the ecosystem (*e.g.* Munz *et al.*, 2012) and potentially toxic effects to humans even by very small doses (*e.g.* Vandenberg *et al.*, 2012).

Member states of the European Union but also other countries like Switzerland and Norway have therefore implemented National Action Plans (NAPs) on the sustainable use of plant protection products, aiming to reduce

[1] In the following, the term 'pesticides' is used frequently as a synonym for plant protection products.

the risks caused by their application (Directive 2009/128/EC). Those NAPs include different environmental objectives to be fulfilled by a catalogue of measures. In Germany, the risks that can be caused by pesticide application to the ecosystem shall be reduced by 30% until 2023, compared to the average of 1996-2005. In addition, the maximum residue limits of pesticides on domestic and imported food shall be exceeded in less than 1% of the samples (BMELV, 2013; BMEL, 2017). The objectives shall be reached, among others, by better education (*e.g.* the certificate of competence was introduced as part of the NAP) and promotion of integrated crop protection.

Many critics of pesticide application, not only in Germany, but also in other European countries, argue that the measures of the NAP do not go far enough. Stricter guidelines for application are demanded as well as further measures to reduce overall application of pesticides. Especially after the International Agency for Research on Cancer classified glyphosate as “probably carcinogenic” (Guyton *et al.*, 2015), bans on pesticides with potentially dangerous effects on the ecosystem (*e.g.* the herbicide glyphosate and insecticides of the class of neonicotinoids) and specific incentive taxes on pesticides have been debated (for overviews on the debates see the article in the thesis appendix and Tarazona *et al*., 2017). Sweden, Norway, Denmark and France have already implemented taxes on pesticides and countries such as Germany and Switzerland have been discussing them intensively. During this debate, it was the Swiss Federal Office for Agriculture that commissioned the University of Bonn and ETH Zurich to analyse potential effects of an incentive charge on pesticides in Switzerland (Finger *et al.*, 2016, 2017a, 2017b; thesis appendix). Switzerland aims in its NAP to halve the environmental risks caused by pesticide application until 2027 compared to a reference period of 2012-2015, and introducing an incentive charge was discussed to contribute to fulfilling this aim (Bundesrat Schweiz, 2017). Analytical results of this dissertation have also been used for the project report (see Chapter 2 and Chapter 3).

The project work motivated later parts of this dissertation by unveiling the lack of (public) detailed pesticide application models. By selecting, for example, profit- or utility maximising strategies, such simulation models can be used to analyse pest management decisions in detail, and different scenarios can be analysed by changing the assumed legislative circumstances. Policy measures like introducing a tax on pesticides or banning specific ones can be analysed in detail before they are introduced. Those *ex ante* analyses are necessary to avoid unintended side effects of policy measures. For example, in the discussion about banning the broad-spectrum herbicide glyphosate, critics often argue that a ban would lead to more intense application of alternative herbicides being more toxic to the ecosystem (*e.g.* Fairclough *et al.*, 2017; Schulte *et al.*, 2017). Existing models are, however, not detailed enough for many analyses since they either aggregate pesticides into different groups (*e.g.* herbicides, fungicides or insecticides; *e.g.* Babcock *et al.*, 1992; Kuosmanen *et al.*, 2006) or use the amount of applied pesticides as a measure of efficacy (*e.g.* in litres; Karagiannis and Tzouvelekas, 2012) not accounting for different efficacies of different products. In Chapter 4 and Chapter 5 of this dissertation, a new bio-economic model is developed and presented to analyse weed control decisions. More specifically, a production function-based approach is developed for cultivation of silage maize in the German state of North-Rhine-Westphalia (NRW) depending on weed control inputs (*e.g.* tillage, broad-spectrum herbicide respectively glyphosate application, selective herbicide application).

Plant protection products are typically not applied to increase yield levels but to abate damage from the potential harvest. Thereby, it is often emphasised that plant protection reduces the economic production risk. After describing the research aims of this dissertation, the influence of production risk and the damage abatement characteristic of pesticides are described. In addition, a policy overview is provided with frequently used and potentially introducible measures. This part is followed by an outline and the major contributions of this dissertation. Afterwards, the contributions of this thesis are discussed and future research recommendations are depicted. Finally, general conclusions of this thesis are given. This introduction is followed by four chapters, each of them comprising one original research article, and an appendix, comprising a comment that summarises the project of the Swiss Federal Office for Agriculture.

1.1 Research Aims

The overall aim of this dissertation is to evaluate farm-level effects of policy measures on pesticide application. Furthermore, a specific research objective is pursued in each of the four articles presented, focussing on different problems with respect to policy measures on pesticides. In order to evaluate pesticide taxation schemes, experiences and developments from countries having implemented such a tax can be used. However, firstly, there was no up-to-date and in-depth analysis on the four different tax schemes, and secondly, there was no assessment of the taxation schemes with respect to their effects on pesticide use and their coherence to the aims proposed in the different NAPs. Therefore, the aim of the article presented in Chapter 2 is:

(I) *Present an up-to-date, detailed overview and assessment of the existing pesticide taxation policies in Europe including Sweden, Norway, Denmark and France.*

One crucial factor for the effectiveness and for the efficiency of such tax schemes is the own-price elasticity of demand for pesticides, *i.e.* to what extent does the demand for pesticides change if the price increases by one percent. The estimates of the price elasticity of demand depend on different study characteristics, for example on the time horizon considered, the agricultural system and the pesticide products investigated. Nevertheless, there was no coherent analysis of the different estimates reported in the large body of literature. Because of this large literature collection, a meta-analysis is a reasonable method in order to give reliable estimates for the price elasticity of demand. The aim of the article in Chapter 3 is thus:

(II) *Analyse the own-price elasticity of demand for pesticides in order to give a coherent estimate for the responsiveness of farmers to changes in pesticide's prices.*

Farm models can be used to analyse effects of potential policy measures on agriculture *ex ante*, so that decision support concerning the design and evaluation of policy measures can be given. To reflect pesticide application decisions in a realistic way, such models need to be very detailed. In the application of the developed model, we focussed on another important debate in German and European agriculture: the potential ban of the broad-spectrum herbicide glyphosate. This topic dominated many agri-environmental policy debates throughout 2015 to 2017, and the relicensing of glyphosate for five years in 2017 will not end the debate.[2] The research aim of Chapter 4 is therefore:

(III) *Develop a bio-economic model to analyse different policy measures on plant protection products in detail and apply the model to a potential ban of glyphosate.*

The first study on a potential ban of glyphosate focused on economic factors that could be influenced, for example yield effects, labour demand and changes in weed control costs and the gross margin. Production risk was so far neglected in the analysis. The first objective in the next study was to consider herbicides in the model as influencing production risk. Therefore, the new objective is not the gross margin being maximised but the utility depending on the variations of the gross margin. Moreover, important aspects in debates focus less on economic factors but are more concerned with environmental benefits or drawbacks of a glyphosate ban. Supporters of a ban assert that banning the herbicide would lead to a better protection of nature and human health – irrespective of whether or not glyphosate is carcinogenic. Opponents of a glyphosate ban bring forward the arguments that

[2] The approval of active substances can legally be renewed for a period of up to 15 years according to Regulation (EC) No 1107/2009.

i) alternative and more toxic herbicides would substitute for glyphosate, and ii) a ban would lead to higher energy consumption on farms due to more intense cultivation with different tillage techniques. For those reasons, the aim of Chapter 5 is:

(IV) *Consider production risk in the decision making process and assess environmental effects of policy measures on pesticides by integrating agri-environmental indicators into the model.*

1.2 Background on Pesticide Application and Policy

Several different categories of pesticides are distinguished in relation to the target organisms. Each pesticide contains one or more active substances (AS) and adjuvants, *e.g.* surfactants or emulsifiers to improve mixing characteristics. Herbicides are used to control for herbs and grasses, fungicides are used against fungi, insecticides against insects, molluscicides against molluscs and rodenticides against rodents. Therefore, all pesticides control for potential direct or indirect damage to the crop – except for growth regulators, which are typically applied to steer the crop growth in a specific direction, *e.g.* they shorten the length of the internodes and/or thicken the stems. With this purpose, pesticides are not applied to increase yields but to protect the crop from damage. Hence, they are characterised as damage abating or damage control inputs (similar to hail nets, frost-protection sprinkling or drainage) (Lichtenberg and Zilberman, 1986). On the other hand, productive inputs like fertiliser or seed lead to high yields per hectare, and of course, some inputs may have both a productive and a damage-controlling component, for example, tillage practices on specific soils.

1.2.1 Pesticide Application in German Agriculture

Table 1.1 presents average applied pesticides in Germany for selected crops. The application index depicts the number of applied products on a field weighted with the authorised amount to apply, *i.e.* the standard area dose.

The application frequency represents the spraying passes on a field. It can be seen that potatoes and apples are highly dependent on fungicides. Cereals such as wheat and barley depend both on fungicides and on herbicides. Pesticide application in maize, which is analysed in detail in Chapter 4 and Chapter 5, is generally lower than in other crops and it relies almost exclusively on herbicide application. Insecticide application, *e.g.* against the European corn borer (*Ostrinia nubilalis*) or against the Western corn rootworm (*Diabrotica virgifera*), plays only a minor role.

Table 1.1. Average application of pesticides in selected crops in Germany measured with two application indices

	Winter wheat	Winter barley	Rape seed	Potatoes	Maize	Apples
Application index (number of pesticide products applied per hectare):						
Pesticides total	5.5	4.1	6.8	12.3	1.9	32.2
Fungicides	2.2	1.4	1.9	8.9	0.0	26.0
Herbicides	1.7	1.5	2.0	2.3	1.9	0.9
Insecticides	0.7	0.4	2.7	1.1	0.0	4.6
Other	0.9	0.8	0.1	0.0	0.0	0.7
Application frequency (number of sprayer passes per hectare):						
Pesticides total	4.3	3.5	5.7	9.3	1.4	22.1
Fungicides	2.4	1.7	2.8	7.2	0.0	20.2
Herbicides	1.7	1.6	2.3	2.4	1.4	0.7
Insecticides	0.7	0.5	2.7	1.1	0.0	5.5
Growth regulators	1.6	1.3	-	0.0	0.0	1.3

Reference: Julius-Kühn-Institut (2017).

The amount of sold pesticides in Germany is depicted in Fig. 1.1. The figure shows both the overall volume of sold AS and the average amount of applied AS per hectare by dividing the volume by the agricultural area in Germany. One can see both an increase in total sales volume and in the amount applied per hectare, especially since the mid-2000s. However, this increase is only partly the cause of higher ‘classical’ pesticide sales, such as herbicides or fungicides. One further reason is that the ‘other AS’ were sold

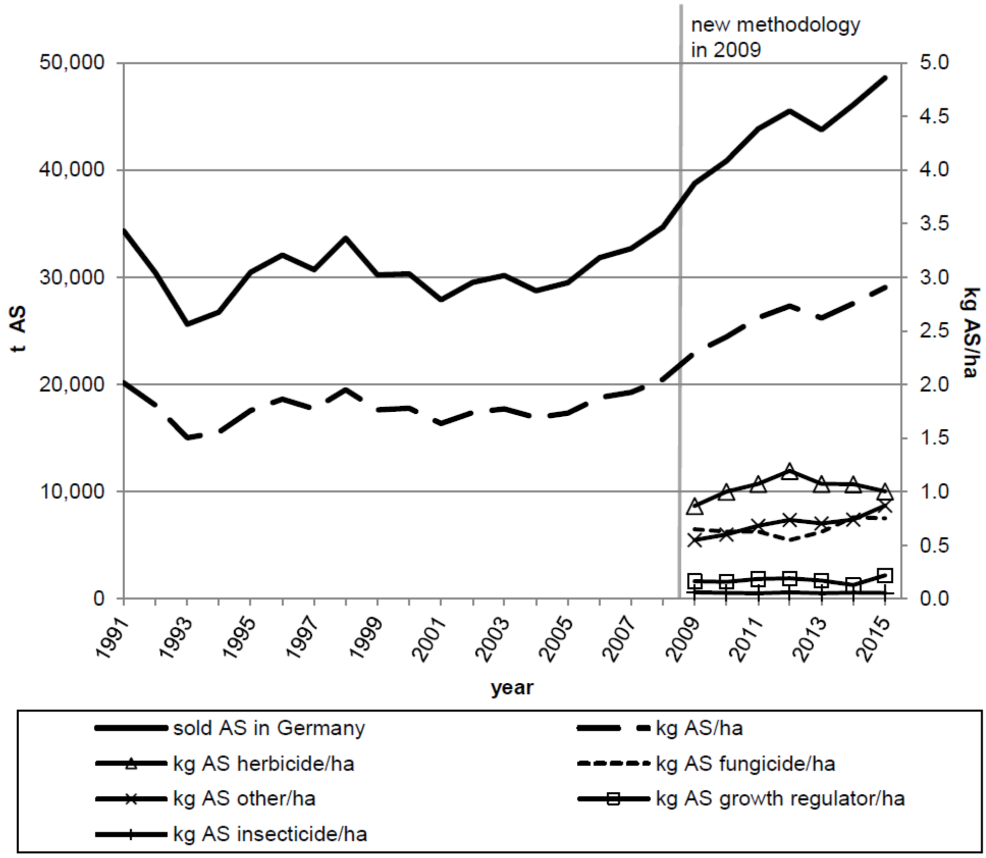

Fig. 1.1. Sold active substances (AS) in Germany as total volume and average amount applied per hectare agricultural area [kg AS/ha].

Note: The amount applied per hectare includes also pesticides that are used in non-agricultural areas. The values are, however, relatively small. Other ASs include, among others, mineral oils, vegetal oils, soil sterilants (incl. nematicides) and rodenticides.
References: Eurostat (2017), Statistisches Bundesamt (1991-2016).

at higher volumes. This group contains, on the one hand, nematicides and rodenticides, which use should be relatively constant in Germany (see Table 1.1 for average applied pesticides on different crops), but on the other hand, also mineral and vegetable oils are included (Eurostat, 2017). The latter are applied especially in organic farming. For example, rapeseed oil can be used as an insecticide against potato beetles and aphids and paraffin can be used as an acaricide against spider mites and as an insecticide against aphids (BVL, 2017). It is likely that the use of both oil types increased due to the growth of organic farming in Germany (Statistisches Bundesamt, 1991-2016). The vast variety of pesticides described above points already to the fact that the quantity of AS does not describe the adverse environmental effects very well. Pesticide indicators that include the biochemical characteristics of each AS are needed (see also the subsequent sections of the Introduction as well as Chapter 5).

1.2.2 Plant Protection and Risk Management

Since pesticides are used to control damage from crops, they are often seen as a substitute for crop insurance (Carlson, 1979; Smith and Goodwin, 1996), which means that they are an input to reduce the production risk, *i.e.* an input that reduces the variation of yield (*e.g.* Carpentier and Weaver, 1997; Skevas *et al.*, 2014). In this case, risk averse decision makers, which farmers typically are (*e.g.* Maart-Noelck and Musshoff, 2014; Meraner and Finger, 2017), would apply an amount of plant protection products exceeding the profit maximising level. However, under certain conditions pesticides can also be risk-increasing, meaning that the variation of income may increase by applying pesticides (Horowitz and Lichtenberg, 1993, 1994). The risk-increasing effect was found in empirical analyses, for example, on arable farms in Switzerland, Italy and the Netherlands (Regev *et al.*, 1997; Di Falco and Chavas, 2006; Gardebroek *et al.*, 2010). It seems to be relevant at unfavourable environmental conditions, when pesticides are applied, but do not lead to improved yields by damage control (Saha *et al.*, 1997). Nevertheless, the risk-effect of pesticides depends also on the indicator used (Möhring *et al.*, 2017). When applying a quantity indicator (the authors use kg of product per hectare), a risk-increasing effect of fungicides and herbicides is found. However, when applying a pesticide load indicator measuring the toxicity of the products (see Chapter 5 for a description of the indicator), they find a risk-reducing effect of fungicides and herbicides. This means that risk averse farmers apply more toxic products than risk neutral or affine farmers do.

Both the property of damage control and the pesticides' effect on income variation are important to take into account when analysing policy measures on pesticides. Ignoring the risk-effect of pesticides could lead to over- or underestimated input use in agricultural economic analyses or to wrong interpretations of farmer's behaviours. For example, introduction of crop insurance, which was said to substitute for pesticides, has not led to a reduction in pesticide use (Horowitz and Lichtenberg, 1993). Ignoring the damage abating effect in a production function can lead to an overestimation of the pesticide's productivity (Lichtenberg and Zilberman, 1986). Among other reasons, this is because classical production function forms such as Cobb-Douglas define inputs to be strictly positive (Saha *et al*., 1997: 774). However, zero use of pesticides is a legitimate scenario in specific cases. Higher than optimal amounts of pesticides would thus be calculated.

1.2.3 Policy Instruments to Regulate Pesticide Application

Pesticide application is highly regulated in developed countries and there is a large variety of policy instruments. Those instruments comprise measures in the category of i) information, persuasion and awareness, ii) voluntary arrangements by stakeholders, iii) private law instruments, iv) economic instruments, v) technical or institutional change, and vi) regulation. An overview of specific measures that can be introduced is given in Table 1.2. Several measures of the six categories are already in force in Germany. There are many regulatory measures regarding the application and mixing the pesticides. Training and education on pesticide application is obligatory for professional applicants since 2015 (certificate of competence; see *Pflanzenschutz-Sachkundeverordnung* 2013). Biological pest control is promoted by the Plant Protection Authorities (*e.g.* against *Ostrinia nubilalis*). Firms develop techniques to reduce drift when applying pesticides and farmers need a certification of the technical inspection association for their boom sprayers (see *e.g.* the German Plant Protection Act and the Plant Protection Products Regulation).

Table 1.2. Instruments for environmental policy on pesticides

Policy category	Regulatory instrument
Information, persuasion and awareness	Training and education Decision support systems for pest management (good agricultural practice) Labelling of farm produce
Arrangements	Agreements by farmers and retailers Voluntary agreements for water protection
Private law instruments	Reduced use clause in land lease contracts Contracts between water suppliers and farmers
Economic instruments	Tax or levy on pesticides Marketable rights or permits Premiums to prevent water contamination Stimulation of alternative pest management Insurance against yield risk Adjusting agri-environmental measures of Common Agricultural Policy (Greening)
Technological or institutional change	Breed resistant cultivars (also genetic modification) Promotion of biological pest control Promotion of extensification Promotion of long-term set aside areas Abolishing price support systems Improvements in application technology Remove residues from water
Regulation	(Re-)registration of pesticides Permission and recording of trade Labelling of pesticide products Disposal of containers and residuals (Water) protection zones Fixed quota of pesticides per farm or ha Restricted access to (high risk) pesticides Restriction of spraying methods Monitoring of residues

Calls for introducing a specific tax on pesticides in Germany and Switzerland

→ **Chapter 2 and Chapter 3**

Bio-economic model necessary to analyse policy instruments

Calls for not re-licensing/banning glyphosate in the EU

→ **Chapter 4 and Chapter 5**

Reference: adapted from Oskam *et al.* (1997)

Economics instruments are established predominantly via the Common Agricultural Policy of the European Union, although there are many possibilities. Only selected countries have established a tax scheme on pesticides and France is even testing a quota/permit system (Belassen, 2015). A reason for the minor relevance of economic instruments could be the uncertainty of success and a lack of knowledge about them. Regulatory measures, such as a ban of glyphosate, are more straightforward at first glance, because the major outcome is often defined by the measure itself – in case of a glyphosate ban, the AS is not applied anymore. However, side effects of such measures have to be taken into account as well. For example, are other herbicides applied more intensively if glyphosate is banned? Biologic and economic models can contribute to taking into account those effects. In the following section, the study outline and major contributions of this thesis are described.

1.3 Study Outline and Major Contributions

Each of the following four chapters of this dissertation focuses on one of the postulated research aims that were previously defined. In Chapter 2, the analysis of policy measures starts with comparing existing pesticide taxation schemes in Europe. Sweden, Denmark, Norway and France have implemented a special tax on pesticides. All countries have different schemes and tax levels. The country's proposed aims on pesticides in their NAPs are compared to the effects of introducing the tax. We found that taxes on pesticides can be effective in fulfilling the objectives of the NAPs, but are highly dependent on the design of the scheme. Because those objectives often comprise a reduction of environmental risks caused by pesticide application, differentiated tax schemes, *e.g.* according to an eco-toxicological indicator such as in Denmark and Norway, seem to be more suitable to meet the national objectives than *ad valorem* or per unit taxes.

The price elasticity of demand for pesticides is investigated in a meta-analysis in Chapter 3. With a median of -0.28 the average price elasticity of demand was found to be inelastic over all included studies. A price increase of 10% would thus lead to an average reduction of 2.8% of pesticide use.

However, differences in elasticity with respect to variability of inputs, the production system and the time of publication were found. More specifically, long-term demand is more elastic than short-term demand due to the higher variability in input use (*e.g.* changes in crop rotations are possible). With regard to the agricultural system, pesticide demand in special crops was found to be more inelastic than demand in arable farming. This can be explained by the lower flexibility of orchard fruits, the higher value per hectare, more application of preventive pesticides, and fewer available substitutes. The type of pesticide applied is found to be relevant as well. Particularly herbicides have a more elastic demand than fungicides or insecticides. One reason could be that mechanical alternatives exist, especially for pre-sowing weed control (see also Chapter 4). Moreover, we found that the price elasticity of demand decreased over time and that peer-reviewed studies tend to find less elastic demand.

In Chapter 4, the bio-economic model is developed and presented to analyse policy measures on plant production products in detail. The model consists of a first part, in which a production function for herbicide application is developed, and of a second part, in which profit-maximising weed control strategies are selected, depending on economic and spatially explicit environmental conditions. The focus is on herbicide application in cultivation of silage maize in NRW and the model is applied to a potential ban of glyphosate. Glyphosate application is found to be the optimal strategy on light soils and on heavy soils under current conditions. This means that the highest gross margin per hectare can be reached. Banning glyphosate would lead to a small but significant yield loss and an increase of labour demand on farms. A significant increase in the application of more selective herbicides was not found and glyphosate application was mainly substituted by mechanical weed control strategies. Furthermore, a ban of glyphosate does not lead to significant reductions of the gross margin in most regions. The reason is that some cost savings in sowing are assumed in the model if several passes of mechanical weed control are applied.

In Chapter 5, production risk is implemented into the model, which means that the utility per hectare is maximised by taking into account variability of the gross margins over time. In addition, two agri-environmental indicators are incorporated. One focuses on the environmental load of the applied pesticides, the so-called Pesticide Load Indicator (PLI), and the second one consists of a process energy balance. On the one hand, the results of this analysis reveal that a ban of glyphosate would lead to a reduction of the environmental load with respect to all three dimensions of the PLI, *i.e.* a reduction in pesticide toxicity, environmental fate and human health load. On the other hand, the results show an increase in direct energy demand and a loss of energy efficiency mainly due to an increase of diesel consumption and a yield loss. Thus, a goal conflict exists between policy objectives to protect nature and human health and those to reduce energy consumption.

1.4 Discussion of Thesis Contributions

In this dissertation, stricter economic and regulatory instruments were analysed and a tool was supplied to analyse such policy instruments. A specific tax on pesticides is found to be effective in reducing the environmental risks caused by pesticide application if it is properly designed in relation to the aims defined in the NAP. Such a tax does not necessarily lead to a reduction in total pesticide application (in litre or kg of AS) but to a substitution with less toxic products or non-chemical plant protection strategies. When introducing a tax on pesticides, it is important to consider the inelastic demand for pesticides. In order to achieve agri-environmental policy targets by a pesticide tax, it has to be either high enough or additional reinforcement measures need to be introduced. Reinforcement possibilities include labels of the tax level to raise awareness or re-funding of the tax revenues into measures that further reduce the pesticide’s risks. For example, promoting investments in new spraying equipment with nozzles having little drift and save pesticides or investments in better information systems for early warning of insects or fungi to prevent unnecessary applications.

The own-price elasticity of demand for pesticides has been estimated by many studies and we analysed the results coherently in a meta-analysis. In general, the elasticity of demand is inelastic. However, many differences were found between the studies as outlined above. When implementing policy measures such as pesticide taxes the findings should be taken into account. For example, special crops often rely highly on pesticide application (Table 1.1), so that regions with many special crops probably also face more negative external effects of pesticide application. However, if the farmer's response to higher prices due to a tax on pesticides is small, the diminishment of negative external effects will be small as well. Furthermore, a simple tax decreases herbicide use more than fungicide or insecticide use, although the latter pesticide groups are often more toxic to the environment (*e.g.* Kovach *et al.*, 1992; Ørum and Sommer Holtze, 2017: 47). This relatively high elasticity for herbicides was also found in the application of the bio-economic model, mainly because mechanical substitutes are available (*e.g.* chisel ploughing instead of applying glyphosate; see also Chapter 4). The reduction of negative environmental effects could therefore be smaller than expected. This highlights the need for a differentiated tax scheme if policy makers decide to introduce a tax.

Bio-economic models such as the presented one are valuable tools to analyse policy measures on farm-level in detail. Especially for pesticide application analysis, such models can be used to account for the large heterogeneity of the products. In addition, the heterogeneity of different regions can be included by geographic information systems. Of course, not all details of an agri-environmental system can be described in a model, but it can be attempted to approximate closely if it is reasonable. If the model simulates spatially explicit policy measures, the modeller has to weigh up the scale of the geographical focus region. He or she could focus, for example, on a county or on a national level. A county level allows analysing on a small scale by including many details of a region. In addition, a smaller size of the regional unit can be chosen, *e.g.* a 1x1 km raster. A larger scale often leads to some aggregation of parameters and thus to a loss of details, but regions can be compared with different biological and economic conditions. In our modelling approach, the regional focus was set to the state of North-Rhine-Westphalia (NRW) and the regional units are the municipalities. On this regional level, we can still account for many regional differences such as yield levels, soil texture and weed pressure. The developed model could theoretically also be used on a smaller scale, *e.g.* at a 1x1 km raster. However, in order to analyse effects of policy measures for the whole state of NRW, this would make the evaluation of results more cumbersome (NRW has a size of 34,110 km², compared to 396 municipalities) and the simulation process would have taken much more time.

1.5 Future Research Recommendations

While society and policy discuss about implementing administrative instruments to reduce the environmental risks caused by pesticide application, scientists and researchers in agriculture and engineering work on smart, technical solutions to reduce pesticide input on farms (see Walter *et al.*, 2017). Digitalisations of agricultural machinery and of farm management are two important dimensions where computer systems can assist farmers in reducing pesticide use. Visions for such precision farming techniques include fleets of robots that physically control for weeds, precise application of pesticides so that only areas are sprayed where weeds, pests or disease occur, and better farm management methods to identify where alternative crop protection strategies are economically optimal (*e.g.* Slaughter *et al.*, 2008). Data collected by humans, machinery, drones or satellites can support those techniques. Adoption of precision farming techniques offer individually tailored technical solutions to generate sustainable reductions of the environmental risks caused by pesticide application.

The approach of the developed bio-economic model could be implemented in such smart farming techniques. The model code is available with open access in Böcker *et al.* (2017). In its current version, silage maize is the only crop in the model. Since the production factor ‘land’ is thus fixed, only short- to mid-term results are obtained. Future research could adapt this approach to other crops, other pesticide categories and/or include livestock production. For example, the increase in labour demand may influence the whole farm management. Furthermore, the objective in the optimisation model could be altered, for example to a safety threshold approach. With such an approach a certain yield has to be reached (*e.g.* because the cattle has to be fed). Thus, such an approach can partly be used to avoid creating a whole farm model.

The bio-economic model can be used to analyse both bans of specific herbicides, as we did for glyphosate, or to analyse taxes on herbicides. Different types of taxes might be analysed: *ad valorem* taxes where a surcharge is added as percentage to the value of the product, quantity taxes where the amount of AS in a product is taxed and differentiated taxes where a tax is calculated based on the biochemical characteristics of a product. Since there was also a debate on introducing taxes on pesticides in Germany, future applications of the model could analyse potential changes in weed control after introducing a tax on pesticides.

Technically, the model could also be used for market research on herbicides. Questions in this field are whether a new herbicide product can compete with existing ones or what has to be done so that a product has an advantage to existing ones (*e.g.* reducing the herbicide’s price).

1.6 Thesis Conclusions

Reducing environmental risks caused by pesticide application is one of the core goals of current agri-environmental policy. With analysing the potential effects of a specific pesticide tax and a potential ban of glyphosate, two different agri-environmental measures were brought into focus in this dissertation. Both of the measures have been discussed intensively in the public, but none of them was implemented so far in Germany, also because their effectiveness and efficiency have been open questions. Glyphosate has been relicensed by the European Commission for additional five years at the end of 2017.

Both of the analysed policy instruments could contribute to reducing the environmental risks caused by pesticide application (see Chapter 1 and Chapter 5). In case of the glyphosate ban, we found a significant decrease of the PLI with respect to all three dimensions of the indicator, environmental toxicity, environmental fate and human health load. On the other hand, pesticides contribute to higher energy efficiency of agricultural systems (Chapter 5). Introducing measures to reduce pesticide application thus may lead to higher energy demand on farms, especially more direct energy consumption in form of diesel. Which of the effects is finally predominating and steering the debate has to be defined by policy makers. As in the case of glyphosate, compromise solutions are often likely if no decision can be or no decision wants to be made.[3]

Farmer's costs will increase by both a pesticide tax and a glyphosate ban (at least of those who use pesticides or glyphosate). In the first case, the own-price elasticity of demand for pesticides was found to be inelastic, meaning that the demand for pesticides decreases relatively little if the price

[3] In Germany, the Federal Minister of Food and Agriculture, Christian Schmidt, announced in an interview with the Rheinische Post to ban glyphosate for private users and to introduce an obligation to report for professional users if glyphosate is applied in cereals before harvest.

increases (Chapter 3). In the second case, glyphosate is substituted by more expensive mechanical strategies. This substitution goes hand in hand with higher labour demand per hectare. Such a labour demand increase is also a likely consequence of a pesticide tax if it leads to substitution of herbicides by more time consuming mechanical techniques. Furthermore, a small but significant yield reduction was found in the analysis – at least in case of a glyphosate ban (Chapter 4). However, in our simulation of the glyphosate ban, those losses do not lead to a significant decrease of the gross margin on average. This is because the optimal, *i.e.* the profit maximising, control strategy was changed from direct or strip-till sowing with glyphosate application to conservation tillage without glyphosate application, which leads to savings in sowing costs in the simulation. Instead of fully replacing pesticides because of political pressure, adapting the cropping strategy can help to mitigate losses at farm-level.

1.7 References

Belassen, V. (2015). Les Certificats D'économie de Produits Phytosanitaires: Quelle Contrainte et Pour qui. Working Paper 2015/4, Centre d'Economie et Sociologie appliquées à l'Agriculture et aux Espaces Ruraux. Dijon: CESAER INRA.

BMEL – Bundesministerium für Ernährung und Landwirtschaft (2017). Nationaler Aktionsplan zur nachhaltigen Anwendung von Pflanzenschutzmitteln. Date: June 2017. Online: http://www.bmel.de/SharedDocs/Downloads/Broschueren/NationalerAktionsplanPflanzenschutz.pdf?__blob=publicationFile [2 November 2017].

BMELV – Bundesministerium für Ernährung, Landwirtschaft und Verbraucherschutz (2013). Nationaler Aktionsplan zur nachhaltigen Anwendung von Pflanzenschutzmitteln. Date: April 2013. Online: https://ec.europa.eu/food/sites/food/files/plant/docs/pesticides_sup_nap_germany_de.pdf [2 November 2017].

Böcker, T., Britz, W. and Finger, R. (2017). Bio-economic model on weed management in maize production. Online: https://doi.org/10.3929/ethz-b-000184083 [17 January 2018].

Bundesrat Schweiz (2017). Aktionsplan zur Risikoreduktion und nachhaltigen Anwendung von Pflanzenschutzmitteln – Bericht des Bundesrates. Schweizerische Eidgenossenschaft. Online: https://www.blw.admin.ch/blw/de/home/nachhaltige-produktion/pflanzenschutz/pflanzenschutzmittel/aktionsplan-pflanzenschutzmittel.html [27 October 2017].

BVL – Bundesamt für Verbraucherschutz und Lebensmittelsicherheit (2017). Zugelassene Pflanzenschutzmittel – Auswahl für den ökologischen Landbau nach der Verordnung (EG) Nr. 834/2007. Stand: Oktober 2017. Online: https://www.bvl.bund.de/SharedDocs/Downloads/04_Pflanzenschutzmittel/psm_oekoliste-DE.pdf?__blob=publicationFile&v=37 [15 November 2017].

Carlson, G. A. (1979). Insurance, Information, and Organizational Options in Pest Management. *Annual Review of Phytopathology* 17: 149-161.

Carpentier, A. and Weaver, R. D. (1997): Damage Control Productivity: Why Econometrics Matters. *American Journal of Agricultural Economics* 79 (1): 47-61.

Di Falco, S. and Chavas, J.-P. (2006). Crop genetic diversity, farm productivity and the management of environmental risk in rainfed agriculture. *European Review of Agricultural Economics* 33 (3): 289-314.

Eurostat (2017). Pesticide sales. Path: aei_fm_salpest. Online: http://ec.europa.eu/ [18 December 2017].

Fairclough, B., Mal, P. and Kersting, S. (2017). Die wirtschaftliche Bedeutung von Glyphosat in Deutschland. Lüdinghausen: Kleffmann Group.

Finger, R., Böcker, T., Möhring, N. and Dalhaus, T. (2016). Ökonomische Analyse des Einsatzes von Pflanzenschutzmitteln – Risikoaspekte und Lenkungsabgaben. Bericht zu Händen des Bundesamts für Landwirtschaft. Zürich: ETH Zürich and University of Bonn.

Finger, R., Möhring, N., Dalhaus, T. and Böcker, T. (2017a). Revisiting Pesticide Taxation Schemes. *Ecological Economics* 134: 263-266.

Finger, R., Böcker, T., Möhring, N. and Dalhaus, T. (2017b). Lenkungsabgaben auf Pflanzenschutzmittel. *Agrarforschung Schweiz* 8 (5): 176-183.

Gardebroek, C., Chavez, M. D. and Oude Lansink, A. (2010). Analysing Production Technologies and Risk in Organic and Conventional Dutch Arable Farming using Panel Data. *Journal of Agricultural Economics* 61 (1): 60-75.

Guyton, K. Z., Loomis, D., Grosse, Y., El Ghissassi, F., Benbrahim-Tallaa, L., Guha, N., Scoccianti, C., Mattock, H. and Straif, K. (2015). Carcinogenicity of tetrachlorvinphos, parathion, malathion, diazinon, and glyphosate. *The Lancet. Oncology* 16 (5): 490-491.

Hallmann, C. A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., Stenmans, W., Müller, A., Sumser, H., Hörren, T., Goulson, D. and de Kroon, H. (2017). More than 75 percent decline over 27 years in total flying insect biomass in protected areas. *PLoS ONE* 12 (10): e0185809.

Horowitz, J. K. and Lichtenberg, E. (1993). Insurance, Moral Hazard, and Chemical Use in Agriculture. *American Journal of Agricultural Economics* 75 (4): 926-935.

Horowitz, J. K. and Lichtenberg, E. (1994). Risk-reducing and risk-increasing effects of pesticides. *Journal of Agricultural Economics* 45 (1): 82-89.

Jaskolla, D. (2006). Der Pflanzenschutz vom Altertum bis zur Gegenwart – Ein Leitfaden zur Geschichte der Phytomedizin und der Organisation des deutschen Pflanzenschutzes. Berlin: Biologische Bundesanstalt für Land- und Forstwirtschaft.

Julius-Kühn-Institut (2017). Statistische Erhebungen zum Einsatz von Pflanzenschutzmitteln in der Praxis. Online: http://papa.julius-kuehn.de/ [19 December 2017].

Kovach, J., Petzoldt, C., Degnil, J. and Tette, J. (1992). A method to measure the environmental impact of pesticides. *New York's Food and Life Sciences Bulletin* No. 139. Geneva, NY: New York State Agricultural Experiment Station.

Lichtenberg, E. and Zilberman, D. (1986). The Econometrics of Damage Control: Why Specification Matters. *American Journal of Agricultural Economics* 68 (2): 261-273.

Maart-Noelck, S. C. and Musshoff, O. (2014). Measuring the risk attitude of decision-makers: are there differences between groups of methods and persons? *The Australian Journal of Agricultural and Resource Economics* 58 (3): 336-352.

Meraner, M. and Finger, R. (2017). Risk perceptions, preferences and management strategies: evidence from a case study using German livestock farmers. *Journal of Risk Research* (in press). Online: http://dx.doi.org/10.1080/13669877.2017.1351476 [18 September 2017].

Möhring, N., Bozzola, M., Hirsch, S. and Finger, R. (2017). The role of pesticide indicators in the analysis of risk effects of pesticides – evidence from Swiss wheat production. Selected paper presented at the XV EAAE congress, 29th August – 1st September 2017, Parma.

Munz, N., Leu, C. and Wittmer, I. (2012). Pestizidmessungen in Fliessgewässern. *Aqua & Gas* 11: 32-41.

Oerke, E. C. (2006). Crop losses to pests. *The Journal of Agricultural Science* 144 (1): 31-43.

Ørum, J. E. and Sommer Holtze, M. (2017). Bekæmpelsesmiddel-statistik 2015. Orientering fra Miljøstyrelsen nr. 17. Copenhagen: Miljøstyrelsen.

Oskam, A. J., Vijftigschild, R. A. N. and Graveland, C. (1997). *Additional EU Policy Instruments for Plant Protection Products*. Wageningen: Wageningen Agricultural University (Mansholt Institute).

Regev, U., Gotsch, N. and Rieder, P. (1997). Are fungicides, nitrogen and plant growth regulators risk reducing? Empirical evidence from Swiss wheat production. *Journal of Agricultural Economics* 48 (1-3): 167-178.

Saha, A., Shumway, C. R. and Havenner, A. (1997). The Economics and Econometrics of Damage Control. *American Journal of Agricultural Economics* 79 (3): 773-785.

Schulte, M. C., Theuvsen, L., Wiese, A. and Steinmann, H.-H. (2017). Die ökonomische Bewertung von Glyphosat im deutschen Ackerbau. In Britz, W., Bröring, S., Hartmann, M., Heckelei, T. and Holm-Müller, K. (eds), *Agrar- und Ernährungswirtschaft: Regional vernetzt und global erfolgreich*. Schriften der Gesellschaft für Wirtschafts- und Sozialwissenschaften des Landbaues 52. Münster: Landwirtschaftsverlag, 29-41.

Skevas, T., Stefanou, S. E. and Oude Lansink, A. (2014). Pesticide use, environmental spillovers and efficiency: A DEA risk-adjusted efficiency approach applied to Dutch arable farming. *European Journal of Operational Research* 237 (2): 658-664.

Slaughter, D. C., Giles, D. K. and Downey, D. (2008). Autonomous robotic weed control systems: A review. *Computers and Electronics in Agriculture* 61: 63-78.

Smith, V. H. and Goodwin, B. K. (1996). Crop Insurance, Moral Hazard, and Agricultural Chemical Use. *American Journal of Agricultural Economics* 78 (2): 428-438.

Statistisches Bundesamt (1991-2016). Statitisches Jahrbuch. Wiesbaden: Statistisches Bundesamt.

Swinton, S. M. and Van Deynze, B. (2017). Hoes to Herbicides: Economics of Evolving Weed Management in the United States. *The European Journal of Development Research* 29 (3): 560-574.

Vandenberg, L. N., Colborn, T., Hayes, T. B., Heindel, J. J., Jacobs, Jr, D. R., Lee, D.-H., Shioda, T., Soto, A. M., vom Saal, F. S., Welshons, W. V., Zoeller, R. T. and Peterson Myers, J. (2012). Hormones and Endocrine-Disrupting Chemicals: Low-Dose Effects and Nonmonotonic Dose Responses. *Endocrine Reviews* 33 (3): 378-455.

Walter, A., Finger, R., Huber, R. and Buchmann, N. (2017). Smart farming is key to developing sustainable agriculture. *PNAS* 114 (24): 6148-6150.

Chapter 2

European Pesticide Tax Schemes in Comparison: An Analysis of Experiences and Developments[4]

Abstract

Policy measures are needed to reduce the risks associated with pesticides' application in agriculture, resulting in more sustainable agricultural systems. Pesticide taxes can be an important tool in the toolkit of policy-makers and are of increasing importance in European agriculture. However, little is known about the effects of such tax solutions and their impacts on the environment, farmers, and human health. We aim to fill this gap and synthesize experiences made in the European countries that have introduced pesticide taxes, *i.e.*, France, Denmark, Norway, and Sweden. The major findings of our analysis are: (1) overall, the effectiveness of pesticide taxes is limited, but if a tax on a specific pesticide is high enough, the application and the associated risks will be reduced significantly; (2) in all countries, hoarding activities have been observed before a tax introduction or increase. Therefore, short-term effects of taxes are substantially smaller than long-term effects; (3) differentiated taxes are superior to undifferentiated taxes because fewer accompanying measures are required to reach policy goals; (4) tax scheme designs are not always in line with the National Action Plan targets. Low tax levels do not necessarily lead to a reduction of pesticide input and differentiated taxes do not necessarily lead to fewer violations of water residue limits.

---

[4] This chapter is written in American English and published in the journal *Sustainability* as: Böcker, T. and Finger, R. (2017). European Pesticide Tax Schemes in Comparison: An Analysis of Experiences and Developments. *Sustainability* 8(4): 378/1-22.

Keywords: pesticide tax; national action plan; pesticide risk indicator; integrated pest management; Sweden; Denmark; Norway; France.

2.1 Introduction

Policy measures are required in order to reduce the risks and negative external effects associated with pesticides' application in agriculture, resulting in more sustainable agricultural systems. Among others, pesticide taxes are an important tool in the toolkit of policy-makers and these taxes are of increasing importance. Hereby, pesticide taxes could foster the agro-ecological transition to integrated pest management practices via reducing pesticide use and substituting chemical inputs for biological and mechanical ones. Particularly with the implementation of National Action Plans (NAP) in Europe (Directive 2009/128/EC), pesticide taxes are an often discussed instrument in various European countries. For instance, a pesticide tax has recently been discussed intensively in Belgium, Switzerland, the Netherlands, and Germany (ARCADIS Belgium, 2014; EAER, 2014; Hof *et al*., 2013; Möckel *et al*., 2015). Despite the fact that Sweden, Norway, Denmark, and France have introduced pesticide taxes, little is known about the effects of such tax solutions and its impacts on the environment, farmers, and human health. Thus, an overview and assessment of the different taxation schemes as well as experiences made is topical and of high relevance for both researchers and policy-makers.

Only few reviews on this topic have been provided (Oskam *et al*., 1997; Hoevenagel *et al*., 1999; ECOTEC *et al*., 2001; Gregoriou *et al*., 2009; Skevas *et al*., 2013). This literature, however, has some limitations: firstly, the mentioned studies deliver outdated information due to changed policies. This limitation is particularly important because pesticide taxation policies have lately been revised completely in countries such as Denmark. A recent article by Lefebvre *et al.* (2015) gives a short, up-to-date description of the different tax schemes, but an in-depth analysis was not in the scope of their paper. Secondly, descriptions and comparisons across all four countries that introduced a pesticide tax are lacking. Thirdly, none of these papers provides an assessment of the pesticide taxation schemes with respect to its effects on pesticide use and risk indicators as well as its coherence to the recent changes in NAPs. We aim to fill these gaps by presenting an up-to-date, detailed overview and assessment of the existing pesticide tax policies in Europe, including Sweden, Norway, Denmark, and France. Our assessment particularly focuses on the effects these policies have on farmers' pest management practices and the associated environmental and health risks. In addition, we summarize recent debates and depict future developments on pesticide taxes in other European countries. Moreover, the current fiscal pesticide policies are evaluated, also regarding their coherence to the targets of the recent NAPs of the four countries. To this end, different indicators are explained and analyzed.

The remainder of this article is structured as follows: in Section 2.2, we present frameworks for the evaluation of policies on pesticide use in general and pesticide taxation schemes especially. Next, fiscal instruments used in European countries are introduced and their effects on pesticide application and associated risks are assessed. Subsequently, the different fiscal policies are integrated in the presented framework for evaluation. Finally, the existing tax schemes are discussed and the conclusions are drawn.

2.2 Methodology for Pesticide Policy Analysis

During the 1990s, several studies evaluated political measures in order to reduce the pesticides' application and/or the environmental risk possibly related to the use of certain pesticides. In particular, the studies of Reus *et al.* (1994), Oskam *et al.* (1997), and Falconer (1998) have presented theoretical foundations for the evaluation of economic instruments such as taxes for the reduction of environmental and human health risks associated with pesticides. The different criteria for the analysis are presented in Table 2.1. In all three studies, six criteria were applied. A combination of those criteria is also used in this article for the evaluation of existing pesticide tax schemes.

Table 2.1. Criteria used in pesticide policy evaluations

Reus *et al.* (1994: 77f.)	**Oskam *et al.* (1997: 176)**	**Falconer (1998: 49)**
Effectiveness	Effectiveness	Effectiveness
Efficiency	Efficiency	Efficiency
Feasibility and maintain-ability	Enforceability	Maintainability
Polluter pays principle	Homogeneity	Polluter pays principle
Economic consequences for farmers	No income and property rights disturbance	Economic consequences for farmers
Support among farmers	Acceptability	Ability to differentiate policies

Effectiveness refers to the ability of a political instrument to achieve its desired objective. *Efficiency* describes the costs of an instrument in relation to its objective achievement. *Feasibility and maintainability*, together with *enforceability* and *maintainability*, consider possibilities of control and fraud. The *polluter pays principle* stands for the justification of an instrument and the person responsible for pollution being charged. Private and societal benefits have to be balanced with private and societal costs (see *e.g.*, Popp *et al.*, 2013; Zilberman and Millock, 1997). The criteria *homogeneity* focuses on the additional financial burdens among farmers and their distribution. For example, fruit and vegetable growers, as well as potato growers, generally need to apply more pesticides than maize growers or grassland farmers and will, therefore, be taxed higher. In addition, the measure *economic consequences for farmers* and *no income and property rights disturbance* describe if losses occur due to a political instrument and how high those costs are. Finally, *support among farmers* and *acceptability* are overall criteria specifying to what extent policy measures are supported by farmers and their organizations. These criteria will be used to coherently assess the effects of the different pesticide taxation schemes.

In order to actually be able to measure the effectiveness of an instrument, the reduction objective needs to be specified. In principle, there are three possibilities to specify and measure such policy targets (Eurostat, 2008). Firstly, pesticide use indicators measure the quantity of sold or applied active substances (AS). Those indicators are straightforward and the necessary data are easy to collect. For example, the treatment frequency measures the calculated number of pesticide applications under the assumption of a given standard area dose (SAD) (Bol *et al.*, 2003). Secondly, pesticide risk indicators aim to measure the load of a pesticide such as its risks on the environment or human health. It can consist of several sub-indicators which can be created *e.g.*, by using hazard statements (H-phrases or R-phrases, respectively), bee hazards (*e.g.*, in Germany the B-scores), the half-life, the deadly dose for non-target organisms, or measures like the concentration that affects 50% of the test organisms, or the concentration at which no effect between the control and test group can be observed. However, in order to measure the overall environmental load reduction due to the introduction of a policy instrument, very detailed data are necessary over a long period of time (Bol *et al*., 2003). Thirdly, pesticide impact assessment systems aim to evaluate the effective impacts of pesticides on the environment, *e.g*., the influence of a pesticide on non-target organisms or biodiversity. In contrast to pesticide risk indicators, pesticide impact assessment systems are, to a larger extent, based on expert judgments than on chemical analyses (Eurostat, 2008).

The specific indicators used by the four different countries that introduced pesticide taxes are introduced in the subsequent sections.

2.3 Fiscal Instruments Established in Europe

When analyzing fiscal instruments, a differentiation between special taxes or levies on pesticides (use), the general taxation of pesticides (for example by the value added tax, VAT), and special charges on pesticides' registration has to be made. All instruments are in force in Europe. Fees for registration exist *e.g.*, in the UK (Plant Protection Products (Fees and Charges) Regulations), in Germany (*Pflanzenschutz-Gebührenverordnung*), and in Sweden (*Förordning (2013:63) om bekämpningsmedelsavgifter*). Usually, those fees have to be paid by the developer and/or distributor. A VAT on pesticides is collected in all European countries, but the rate differs considerably: in most countries of the EU, the regular (*i.e.*, not reduced) rate is charged ranging between 17% in Luxembourg and 27% in Hungary. As exceptions, Cyprus, Poland, Portugal, Slovenia, and Spain charge reduced VAT rates for pesticides (European Commission, 2015). Additionally, in Switzerland, a reduced VAT on pesticides of 2.5% exists (Art. 25 VAT Act). As these systems thus give quasi-subsidization to pesticides, adjustments towards full VAT taxation in Europe would be in line with current policy discussions on pesticide use. France has abandoned reduced VAT rates for pesticides in 2012, but now applies a combined system with the reduced rate on pesticides being allowed in organic farming and the regular rate on other pesticides (*Art. 278 bis Code général des impôts*). Until 2007, Finland had a special system, where the pesticides' producing or retailing sector was levied by a percentage rate to cover the registration and administration costs (with a total revenue of € 2 million/year; OECD, 2009). Except for France, the major purpose of those two fiscal instruments (*i.e.*, VAT and registration fees) is not to reduce pesticide risks or give incentives to adjust pesticide use.

In contrast, we focus on special taxation instruments that aim to especially reduce pesticide use or risks that are associated with pesticide use. In Table 2.2, possible combinations in the design of a pesticide tax are presented with regard to (i) the tax base and the tax rate of the charge; (ii) the imposition point; and (iii) the use or refunding of the revenues. The tax base for specific or for all pesticides can, for example, be a price, a mass/weight, or an indicator. Basically, the tax rate can be fix or differentiated and either a specific monetary value or a percentage. Note that for a wider organizational level (for example EU) also other combinations exist. The special taxations in the four countries will be described using some of the aspects of this framework.

Table 2.2. Variations of taxation on pesticides on state level

Charge		**Imposition point**	**Use/refunding of revenues**	
Tax base	**Tax rate**		**Organization**	**Target**
- wholesale price, - retail price, - active substances, - environmental risk, - human health risk	- fix, - differentiated	- industry, - wholesalers, - retailers, - farmers	- states, - federal states, - agricultural sector, - farmers involved, - other organizations	- state budget/ deficit reduction, - Common Agricultural Policy - direct payments/ha, - crop premiums, innovation programs for industry and agriculture, - supporting alternative techniques, - other
- tax on all pesticides, - specific pesticides	- tariff level either high, medium, or low, - percentage, - flat			

Reference: Following Hoevenagel (1999).

2.3.1 Sweden

As the first country worldwide, Sweden introduced a special flat tax on pesticides based on the volume sold in 1984. Initially, the tax was introduced with SEK 4/kg AS and was increased stepwise to currently SEK 34/kg AS (~€ 3.64/kg) (Notisum AB, 2015). The last increase from SEK 30 to 34 took place at the beginning of August 2015 *(§ 2 Lag (1984:410) om skatt på bekämpningsmedel*). In addition to the pesticide tax, a price regulation fee was charged between 1986 and 1992, which ranged between SEK 29–46/SAD. The fee was used for supporting the export of agricultural goods, but abolished in the course of the EU accession (ECOTEC *et al.*, 2001).

Until 1995, the financial means of the tax were used for agri-environmental programs aiming to reduce pesticide application and to promote integrated pest management (ECOTEC *et al.*, 2001). After 1995, the revenues have been directly allocated to the state's treasury. The revenues are expected to be about SEK 70 million in 2015 and SEK 75 million in 2016 (~€ 7.5 million and ~€ 8 million, respectively) assuming that the sales quantity stays constant.

The first Swedish NAPs that were adopted during the 1980s focused on the reduction of overall pesticide use and application (Landsbygdsdepartementet, 2013). The present NAP still aims to reduce the use of pesticides, but more important is the reduction of the environmental risk that may be associated to the application of pesticides. These goals are, for example, the reduction of residues in surface water or food and the establishment of farming techniques that are less dependent on chemical pesticides (Landsbygdsdepartementet, 2013). Fig. 2.1 shows that the absolute sales of AS in Sweden have been reduced by more than 50% since the 1980s, even though statements about sold amounts AS have to be treated with caution, because the amount of AS does not reveal any information about environmental quality. In the last two decades, however, tax increases have not led to further reductions. In contrast, slight increases of several indicators are revealed in Fig. 2.1 and Fig. 2.2. Regarding residues in water, detection of very high values (greater than 0.5 µg/L) declined from 1987 to 2014. On the other hand, more residues between 0.1 and 0.5 µg/L were found (Larsson *et al*., 2014). Focusing on the share of wells with a minimum of one AS greater than 0.1 µg/L, a reduction can be seen for the period 2010–2014 compared to the decade before (Larsson *et al*., 2014). Nevertheless, fewer samples were taken in this period, so that overall ambivalent effects can be observed.

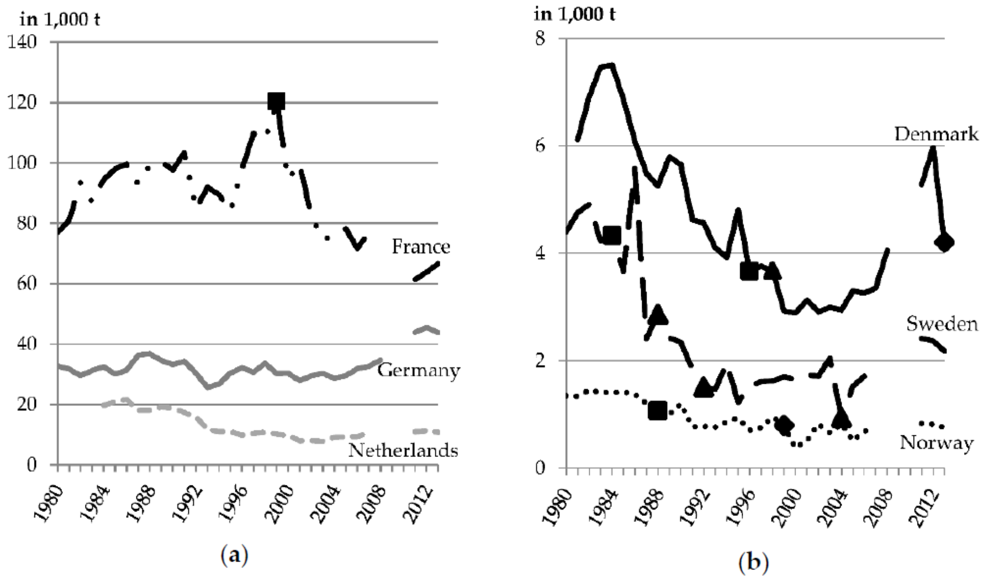

Fig. 2.1. Sold active substances (1,000 t) in selected European countries: (a): France, Germany, and the Netherlands; (b): Denmark, Sweden, and Norway.

Note: A "■" symbolizes the introduction of a pesticide tax, a "▲" symbolizes a tax increase, and a "♦" a change of the tax scheme. Please note the different scales in the diagrams. Gaps in 2009 and 2010 are due to methodological changes in data collection. For Germany, until 1991, data are for West-Germany. For Norway, until 1990, data are estimated according to Spikkerud (2005). Data reference: Eurostat (2016a)

Fig. 2.2 also shows the development of the aggregated Swedish pesticide risk indicator indexed to the year 1988 (index value of 100; disaggregation of the index is not possible). The health and environmental risk indicators are calculated by a point system and a set of scores. Among others, the environmental score, the application method score, the persistence score, and the operator toxicity score are used (see Bergkvist, 2004, for the exact calculation). Fig. 2.2 indicates that especially the human health risk decreased sharply in the beginning of the 1990s and is now relatively constant at a level between 20% and 40% compared to the 1988 level. In contrast, the environmental risk indicator shows a less clear pattern with levels between 50% and 80% if compared to 1988. Thus, positive developments coincide with the introduction of the tax. However, it is unlikely that the pesticide tax is

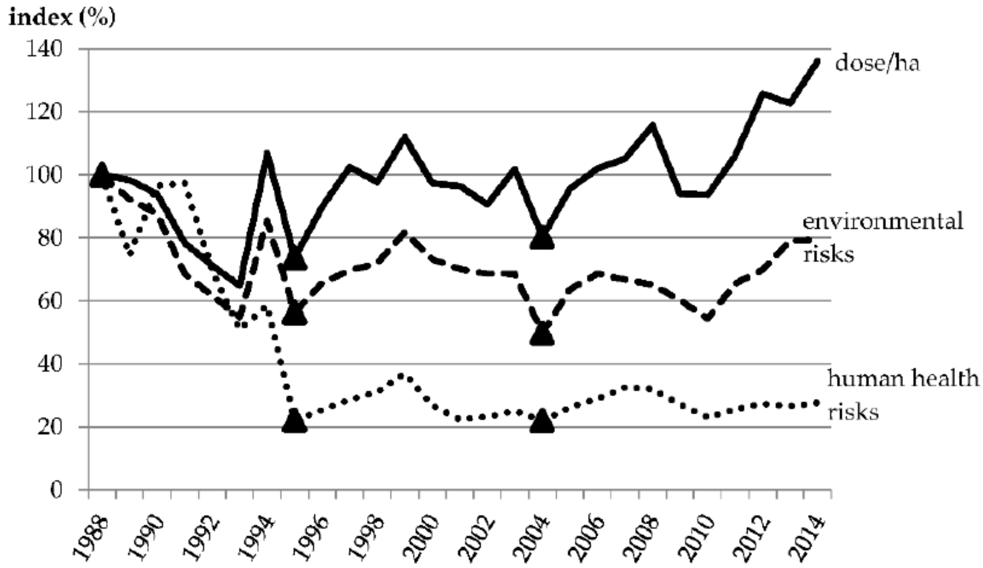

Fig. 2.2. Development of Swedish Pesticide Risk Indicator with respect to environmental and human health risks.

Note: In addition, the average dose/ha is shown as an index. The Pesticide Risk Indicator is indexed to the year 1988 (index value of 100). The "▲" symbolizes a tax increase.
Data reference: Naturvårdsverket (2015)

the only determinant for the decrease of sales and risk. Other factors also contributed to these reductions. For instance, this development was caused by a consulting policy aiming at an integrated pest management, stricter permissions for the registration and application of pesticides, and the introduction of AS with low doses in the 1980s, *e.g.*, by an increased application of seed dressing (Bergkvist, 2004). Overall, the pesticide tax was only a small part of the bundle of additional financial burdens that were introduced in the 1980s: a tax on artificial nitrogen fertilizer and cadmium/phosphorus was in place from 1984 until 2010 and the above mentioned price regulation charge was applied to fertilizers until 1992 as well (ECOTEC *et al.*, 2001; Statskontoret, 2010). Those taxes potentially contributed to a reduction of pesticides sold and their application as high fertilization rates increase pest and disease pressure and *vice versa* (for example a high-nitrogen fertilization might lead to a higher mildew and weed pressure; Baeumer, 1992). Moreover, the value of the marginal product (the added value of one additional unit of input) decreases by reducing fertilizer input and thus causes lower optimal pesticide application levels. However, since both political instruments were introduced at the same time, it is difficult to identify the major influencing factor.

2.3.2 Norway

As the second European country, Norway introduced a tax on pesticides in 1988. First, the tax was designed as an *ad valorem* tax as a percentage of the import value (Spikkerud, 2005). In 1999, the tax was changed into a differentiated scheme and now consists of a base rate and an additional rate. Pesticides are sorted into seven different categories. The base rate is a tax per ha, which is calculated by multiplying NOK 25/ha times a specific factor being associated with the category of a certain pesticide (ranging from 0.5 to 150). The categories are assessed by two sub-categories: (i) risks for human health and (ii) environmental risks. All pesticides for professional use are tested according to several criteria and then categorized in a low, medium, or high risk (*§ 28 Forskrift om plantevernmidler*). The human health-criterion is based on the intrinsic properties (according to R-phrases) and the exposure during application and mixing (Spikkerud *et al*., 2005). The environmental criterion is compounded by eight sub-scores. They measure effects on earthworms, on bees and other arthropods, on birds, on aquatic organisms, the leaching potential, the persistence, the bioaccumulation, and the formulation type (Spikkerud *et al*., 2005). The categorization of the factors can be seen in Table 2.3. The additional rate is calculated via the SAD (Norwegian: *normert arealdose NAD*). The SAD refers to the highest possible application dose (in ml or g per ha), which is recommended for the main crop in the field of application (Spikkerud, 2005; for the product list see Mattilsynet, 2014a). Using the example of liquid products, the final tax is calculated as follows:

$$\text{tax in } \frac{\text{NOK}}{\text{l}} = \text{base rate} * \text{additional rate}$$
$$= \frac{\text{NOK } 25}{\text{ha}} * \text{factor}_\text{i} * \frac{1000\,\frac{\text{ml}}{\text{l}}}{\text{SAD } \frac{\text{ml}}{\text{ha}}} \tag{2.1}$$

Table 2.3. Norwegian factor categorization for base rate calculation

Tax category i		1	2	3	4	5	6	7
Pesticide characteristics	Human health risks / Environmental risks	Both risks low	One low and one medium risk	One low and one high risk or both risks medium	One medium and one high risk	Both risks high	Concentrated products for hobby use	Ready-to-use products for hobby use
Factor i (* NOK 25/ha)		0.5	3	5	7	9	50	150
Tax (NOK/ha)		12.5	75	125	175	225	1,250	3,750

References: *§28 Forskrift om plantevernmidler* (as at 2015); Strøm Prestvik *et al.* (2013)

Products that are allowed in organic farming are exempted from the tax (*§§ 27–28 Forskrift om plantevernmidler*). Producers and importers have to pay the tax to the authorities. The government estimates to earn about NOK 50 million in 2015 (~€ 5.8 million) (Stortingets administrasjon, 2015). An in-depth analysis of the tax scheme was presented by Spikkerud (2005). There are various disadvantages associated with such an assessment that is based on different categories. For instance, products, which are close to the threshold at several criteria, are classified, *e.g.*, as low-risk products. In contrast, products that exceed the threshold value for one criterion but are far below the threshold for the other criteria are categorized as high- or medium-risk pesticides. Therefore, relatively large tax differences can occur for products that may actually differ little in their riskiness to human health or the environment (Spikkerud, 2005). Furthermore, the SAD measure is problematic because a low SAD leads to a higher taxation and *vice versa*. The underlying assumption is that pesticides with a high application dose are less risky (even though environmental toxicity is additionally accounted for in specific factors). In total, this could lead to a higher total application of pesticides, while the human health and environmental risk decreases (Spikkerud, 2005). Finally, the usage of the maximum recommended dose for a specific main field of application is critical. For some pesticides, this determination is not easily feasible. For instance, for vegetables and fruits different doses are usually recommended per crop type and per production system (for example field-grown *vs* greenhouse production) so that probably an incorrect or inappropriate tax base is used (Spikkerud, 2005).

The quantity of pesticides' sold reduced slightly since the introduction of the tax (Fig. 2.1). After changing the tax to a differentiated scheme in 1999, the sold quantity stayed constant except for a break shortly after the change. One reason for the latter might be that the taxation of some low risk products actually was reduced when switching to a differentiated tax scheme since before already an *ad valorem* tax on pesticides was established. The greater popularity of no-till cultivation and the accompanied application of glyphosate also contributed to the non-reduction of the sales quantity of pesticides (Gianessi *et al*., 2009). The two Norwegian NAPs from 1998 to 2008 aimed to reduce the health and environmental risk of pesticides by 50% (Landbruksdepartementet, 2004). The pesticide risk indicator being used for the assessment of this target is divided into a human health risk indicator and an environmental risk indicator (Norwegian Agricultural Inspection Service, n.d.). Analyzing those two pesticide risk indicators, a small to medium reduction can be observed (Fig. 2.3 and Strøm Prestvik *et al.*, 2013). The figure presents the development of the annual sales data of the retailers and the development of the pesticide risk indicator. The marked peaks represent large increases of pesticides sold in advance of the introduction or of changes in the tax scheme. Note that the retailers' behavior is reflected more than the farmers' behavior, although it is likely that also farmers hoarded pesticides to save tax payments in future periods. For this reason, the human health and environmental criteria should only be analyzed in the long-term. Furthermore, Strøm Prestvik *et al.* (2013) observe a decline in the range of highly taxed products (categories 4 and 5) and an increase in the range of category 1 and 2 pesticides. In 2014, one product of the tax category 5 was registered in Norway (Mattilsynet, 2014a). At the single crop level, Strøm Prestvik *et al.* (2013) show that in 2011, hardly any fungicides of category 4 are used anymore in cereal production and fungicides of category 3 have been substituted by products of category 1 or 2 in potato production. Along these lines, in 2001, almost no category 1 products were applied, but they were used in 50% of the applications in 2011 (Strøm Prestvik *et al.*, 2013). However, due to the hoarding activities of farmers (before 1999 at the change of the tax scheme and before 2005 due to minor revisions of the scheme; Fig. 2.3), it took several years until the tax became effective and a more constant reduction of the pesticide risk indicator could be observed. The latest NAP was in place from 2010 to 2014, but did not specify any concrete reduction targets like the two preceding NAPs. Rather, it was aimed to decrease the dependency of chemical pesticides and to increase the share of farmers that produce according to integrated farming practices specified in the good agricultural practice (Landbruks- og matdepartementet, 2009). More specifically, a particular goal is to avoid violations of threshold values of standards for groundwater, surface water, and food. Regarding residues on food products, however, most cases of threshold violations occur in products that are imported (Mattilsynet, 2014b). Moreover, the recent developments show indeed that fewer violations of threshold values could be observed across space (for different regions) and across various AS (Bechmann *et al.*, 2014). However, the analyses show that many violations already declined before introducing the differentiated tax. Measures that contributed to this are for example stricter application guidelines and better spraying techniques to avoid point source contamination and drift. Additionally, the overall number of detected residues (*i.e.*, not exceeding the threshold) has not decreased (Øgaard and Skaalsveen, 2015) and challenges with newer AS appear (*e.g.*, increasing residues of prothioconazole, imidacloprid, and aclonifen; Bechmann *et al.*, 2014). Therefore, no clear pattern is observable whether the tax contributed to these improvements. Note that integrated pest management is also mandatory for countries in the EU since 2009 due to the Directive 2009/128/EC (to achieve the sustainable use of pesticides) and the Regulation (EC) No. 1107/2009 (concerning the placing of plant protection products on the market).

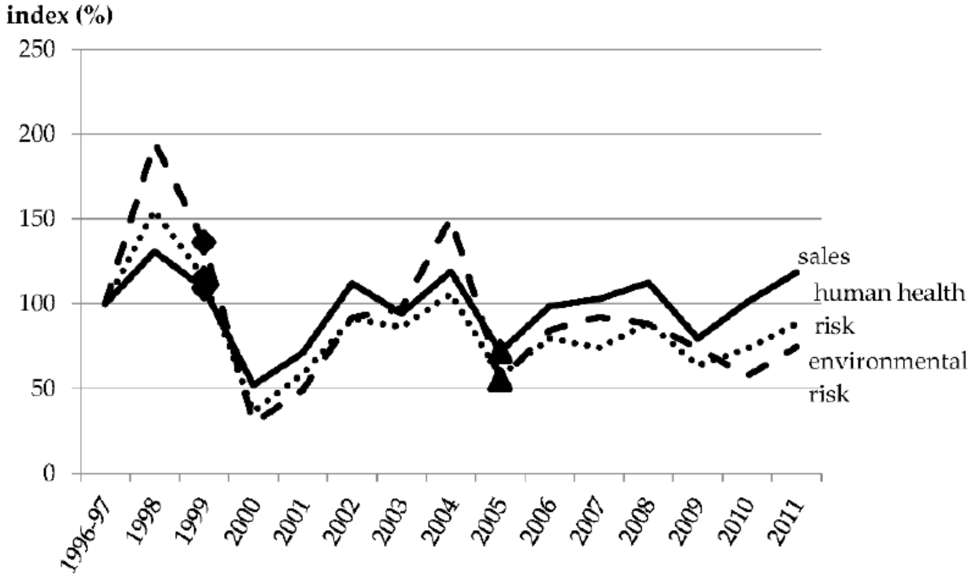

Fig. 2.3. Development of the Norwegian Pesticide Risk Indicator with respect to environmental and human health risks based on retailers' sales data.

Note: The average annual sales are shown, too. The index year is the average of 1996 and 1997. A "♦"symbolizes a change of the tax scheme and a "▲" a tax increase. Data reference: Statistics Norway and the Norwegian Food Safety Authority Strøm Prestvik (2015)

2.3.3 Denmark

In 1996, Denmark introduced an *ad valorem* pesticide tax on the highest existing wholesale price. This tax was differentiated by the pesticide's category. For example, for insecticides a tax rate of 35% was charged and for herbicides, fungicides, and growth regulators a rate of 25%. In 2013, the tax scheme was changed into a more differentiated one, because the treatment frequency (measured by the AS' sales, see also Fig. 2.1) and the pesticides' load re-increased (Miljøministeriet, 2012). Similar to other countries, the latest increase in AS' sales have been induced by the larger relevance of no till practices and the associated application of glyphosate (Gianessi *et al*., 2009). Moreover, high output prices and the corresponding higher value of marginal product contributed to increases in pesticide use (Pedersen *et al*., 2015). In the new tax scheme, each single pesticide product receives its specific tax rate (*LBK nr 232 Bekendtgørelse af lov om afgift af bekæmpelsesmidler* as at 26 February 2015). Note that the old scheme is kept for biocides but with increased tax rates of 40% on insecticides (before: 35%) and 30% on herbicides and fungicides (before: 25%). The new differentiated tax for pesticides is a combination of a pesticide use and a pesticide risk indicator and is calculated as follows (presented for a liquid product):

$$\begin{aligned} \text{tax in } \frac{\text{DKK}}{\text{l}} &= \text{exposition tax} + \text{toxicity tax} \\ &= \frac{\text{DKK 50}}{\text{kg AS}} * \text{X} \frac{\text{kg AS}}{\text{l}} + \sum_{\text{i}=1}^{3} \frac{\text{DKK 107}}{\text{l}} * \text{factor}_{\text{i}} \text{ L} \end{aligned} \tag{2.2}$$

The exposition tax takes into account the amount of AS of the pesticide product (X kg AS/l) and multiplies it times DKK 50. The toxicity tax is calculated with the help of a pesticide risk indicator, the so called Pesticide Load Indicator. This indicator comprises of three different factors (categories) measuring the environmental load of a pesticide: (1) environmental toxicity load; (2) environmental fate and behavior load; and (3) human health load. The Danish scheme, thus, extends the scope of the Norwegian scheme by adding the dimension of environmental fate. The load score of each factor is defined by several sub-indicators and the score is then multiplied by DKK 107. The “human health load” is assessed by the R-phrases of a plant protection product and by the exposure during application (Miljøministeriet, 2012, 2013). The “environmental fate load” is measured by the AS’ degradability, potential for bioaccumulation, and its leaching potential. “Environmental toxicity load” consists of sub-scores for birds, mammals, fish, daphnia, algae, aquatic plants, earthworms, and bees. Additionally, there is a higher tax rate for seed treatment products at the factor “environmental effects” (Miljøministeriet, 2012, 2013).

The tax introduction was accompanied with the implementation of measures to compensate farmers. In particular, the property tax on agricultural land was reduced by DKK 62–72/ha, depending on the county (ECOTEC *et al*., 2001). Moreover, tax revenues were used to support organic farming and for administrative services (ECOTEC *et al*., 2001; Hansen, 2013; Schou and Streibig, 1999). According to the current legislation, the revenues of the tax first flow into the states treasury (§ 1 LBK nr 232) but are then returned for agricultural and environmental purposes. In 2013, these revenues were DKK 659 million (0.23% of the state's budget; ~€ 88.4 million). For 2015, revenues of DKK 600 million are estimated (Skatteministeriet, 2015). From that, DKK 250 million are designated for the agricultural fund (*promilleafgiftsfonden for landbrug*), which supports different measures concerning the Danish agriculture (Natur- og Landbrugskommissionen, 2012). About DKK 175 million are destined for green growth measures – of which some are related to the NAP – and about DKK 75 million are used for administrative purposes. The current Danish NAP for the period 2013 until 2015 aims to reduce the total load of pesticides by 40% until the end of 2015 (Miljøministeriet and Ministeriet for Fødevarer, Landbrug og Fiskeri, 2013). The reduction is measured with the Pesticide Load Indicator. The differentiated tax has a main role in achieving this objective.

Due to the short time span since the implementation of the new tax scheme, no conclusion can be drawn whether the objective is fulfilled. However, the new design of the Danish tax implies large burdens for some products. The heterogeneity of tax levels is higher compared to other taxation regimes. For example, the insecticide Cythrin 500 containing 500 g/L cypermethrin was taxed with DKK 7709/L (DKK 617/ha, ~€ 83/ha respectively at a recommended dose for rapeseed of 0.08 L/ha) and the insecticide Gamma C containing 60 g/L gamma-cyhalothrin is taxed with DKK 6009/L (DKK 361/ha, ~€ 48/ha, respectively, at a recommended dose for rapeseed of 0.06 L/ha; see Table 2.4 for further examples). Even though allowing more flexibility than bans of pesticides, high taxation levels could lead to the disappearance of those highly hazardous products (mostly insecticides) from the market. Critics are concerned that, as a consequence, problems of resistances might be enlarged (Jørgensen and Ørum, 2013), which could lead to a more intense application of other, cheaper pesticides. It is, therefore, possible that a single application leads to a lower environmental load under the new tax scheme, but when summing up all applications, a load similar to the one under the old tax scheme or without tax is reached. In order to counter this effect, one opportunity would be to simplify the pesticide registration process, but since it is an EU policy matter, this is not easily possible. In contrast, the breeding and use of more resistant varieties is a positive side effect of the tax and would support the reduction of pesticide use. Further criticism indicates that farmers face extra burdens and have become less competitive compared to other European producers (Pedersen *et al*., 2015; Jørgensen and Ørum, 2013; Nørring, 2013). However, this depends on the crop that is produced since some pesticides are now burdened lower compared to the old tax scheme (Hansen, 2013).

Table 2.4. Comparison of the taxation of different pesticides (selection)

Type	Product name (depending on country)	Active substance	Sweden		Norway		Denmark		France	
			€/kg or l (SEK/kg or l)	**€/ha (SEK/ha)**	**€/kg or l (NOK/kg or l)**	**€/ha (SEK/ha)**	**€/kg or l (DKK/kg or l)**	**€/ha (DKK/ha)**	**€/kg or l**	**€/ha**
Fungicide	Acanto® 250 SC/Aproach® (SAD in NO = 1000 ml)	Picoxystrobin 250 g/l	0.91 (8.50)	0.91 (8.50)	14.45 (125.00)	14.45 (125.00)	13.54 (101.00)	13.54 (101.00)	0.50	0.50
Fungicide	Amistar® (SAD in NO = 1000 ml/ha)	Azoxystrobin 250 g/l	0.91 (8.50)	0.91 (8.50)	8.67 (75.00)	8.67 (75.00)	5.50 (41.00)	5.50 (41.00)	1.28	1.28
Fungicide	Comet® (SAD in NO = 1000 g/ha)	Pyraclostrobin 250 g/l	0.91 (8.50)	0.91 (8.50)	8.67 (75.00)	8.67 (75.00)	13.00 (97.00)	13.00 (97.00)	1.28	1.28
Fungicide	Stereo® 312.5 EC (SAD in NO = 1500 ml/ha)	Cyprodinil 250 g/l Propiconazole 62.5 g/l	1.14 (10.63)	1.71 (15.94)	9.63 (83.33)	14.45 (125.00)	20.91 (156.00)	31.37 (234.00)	-	-
Fungicide	Switch® 62,5 WG (SAD in NO = 500 g/ha)	Cyprodinil 375 g/kg Fludioxonil 250 g/kg	3.64 (34.00)	1.82 (17.00)	28.90 (250.00)	14.45 (125.00)	14.34 (107.00)	7.17 (53.50)	1.25	0.63
Fungicide	Talius® (SAD in NO = 250 ml/ha)	Proquinazid 200 g/l	-	-	57.80 (500.00)	14.45 (125.00)	-	-	1.02	0.26
Growth regulator	Moddus® M/Moxa® (SAD in NO = 400 ml/ha)	Trinexapac-ethyl 250 g/l	0.91 (8.50)	0.36 (3.40)	3.61 (31.25)	1.45 (12.50)	4.42 (33.00)	1.77 (13.20)	0.50	0.20
Herbicide	Ally® Class 50 WG/Allié® Express (SAD in NO = 50g/ha)	Metsulfuron-methyl 100 g/kg Carfentrazone-ethyl 400 g/kg	1.82 (17.00)	0.09 (0.85)	28.90 (250.00)	1.45 (12.50)	-	-	1.00	0.05
Herbicide	Basagran® SG (SAD in NO = 1600 g/ha)	Bentazone 870 g/kg	3.17 (29.58)	5.07 (47.33)	5.42 (46.88)	8.67 (75.00)	15.15 (113.00)	24.24 (180.80)	1.74	2.78
Herbicide	Boxer® (SAD in NO = 5000 ml/ha)	Prosulfocarb 800 g/l	2.91 (27.20)	14.56 (136.00)	2.89 (25.00)	14.45 (125.00)	16.62 (124.00)	83.11 (620.00)	-	-
Herbicide	Express® Gold SX/CDQ® SX (SAD in NO = 34 g/ha)	Tribenuron-methyl 222.2 g/kg Metsulfuron-methyl 111.1 g/kg	-	-	42.50 (367.65)	1.45 (12.50)	16.76 (125.00)	0.57 (4.25)	-	-
Herbicide	Gratil® 75 WG (SAD in NO = 60 g/ha)	Amidosulfuron 750 g/kg	2.73 (25.50)	0.16 (1.53)	144.51 (1250.00)	8.67 (75.00)	-	-	1.50	0.09

Table 2.4. *(continued)*

Type	Product name (depending on country)	Active substance	Sweden €/kg or l (SEK/kg or l)	Sweden €/ha (SEK/ha)	Norway €/kg or l (NOK/kg or l)	Norway €/ha (SEK/ha)	Denmark €/kg or l (DKK/kg or l)	Denmark €/ha (DKK/ha)	France €/kg or l	France €/ha
Herbicide	Hussar® OD	Iodosulfuron-methyl-sodium 100 g/l	-	-	14.45	1.45	6.30	0.63	-	-
	(SAD in NO = 100 ml/ha)				(125.00)	(12.50)	(47.00)	(4.70)		
Herbicide	MaisTer®	Foramsulfuron 300 g/kg	1.13	0.17	9.63	1.45	11.13	1.67	-	-
	(SAD in NO = 150g/ha)	Iodosulfuron-methyl-sodium 10 g/kg	(10.54)	(1.58)	(83.33)	(12.50)	(83.00)	(12.45)		
Herbicide	Roundup® Max/Roundup® 680	Glyphosate 680 g/kg	2.48	4.95	0.72	1.45	9.79	19.57	1.36	2.72
	(SAD in NO = 2000 g/ha)		(23.12)	(46.24)	(6.25)	(12.50)	(73.00)	(146.00)		
Insecti-cide	Biscaya® OD 240	Thiacloprid 240 g/l	0.87	0.35	36.13	14.45	16.09	6.43	1.22	0.49
	(SAD in NO = 400 ml/ha)		(8.16)	(3.26)	(312.50)	(125.00)	(120.00)	(48.00)		
Insecti-cide	Calypso® 480 SC	Thiacloprid 480 g/l	1.75	0.35	101.16	20.23	-	-	2.45	0.49
	(SAD in NO = 200 ml/ha)		(16.32)	(3.26)	(875.00)	(175.00)				
Insecti-cide	Confidor® 70 WG	Imidacloprid 700 g/kg	2.55	0.51	7.23	1.45	5.23	1.05	-	-
	(SAD in NO = 200 g/ha)		(23.80)	(4.76)	(62.50)	(12.50)	(39.00)	(7.80)		
Insecti-cide	Karate® 5 CS/Karate® Foret	Lambda-cyhalothrin 50 g/l	0.18	0.03	96.34	14.45	-	-	0.26	0.04
	(SAD in NO = 150 ml/ha)		(1.70)	(0.26)	(833.33)	(125.00)				
Insecti-cide	Steward® 30 WG	Indoxacarb 300 g/kg	1.09	0.27	57.80	14.45	102.01	25.50	1.53	0.38
	(SAD in NO = 250 g/ha)		(10.20)	(2.55)	(500.00)	(125.00)	(761.00)	(190.25)		

Note: Hectare cost values were calculated in all countries on the basis of the Norwegian SAD to guarantee comparison of the results. It could be that the SAD in Norway differs in the other countries. The different pesticide products were selected on the basis of: 1) availability in Norway, 2) relevance of product, 3) different categories of products (risk and use class), 4) availability in other countries. Corresponding exchange rates in the first half of 2015 were as follows: EUR : SEK | NOK | DKK = 1 : 9.34 | 8.65 | 7.46 (Eurostat, 2015).

References: author’s own compilation; lists of registered products and/or the corresponding taxes on them can be found at: Kemikalieinspektionen, 2015 (Sweden); Mattilsynet, 2014a (Norway); MEDDE, 2015 (France); Miljø- og Fødevareministeriet, 2015 (Denmark); SEGES P/S, 2015 (Denmark).

2.3.4 France

France has introduced a volume tax on pesticides in 2000. First, the tax was introduced as the *taxe générale sur les activités polluantes (TGAP)*, which was valid until 2009. Pesticides were divided into seven taxation categories (based on the eco-toxicological and toxicological properties according to the R-phrases) and the tax had to be paid by the pesticide distributors (Aubertot *et al*., 2005; MAP, 2006). Category 1 was tax free, category 2 was taxed at € 381/t AS, category 3 at € 610/t, category 4 at € 838/t, category 5 at € 1067/t, category 6 at € 1372/t, and category 7 was taxed at € 1677/t AS. Initially, the values were in Francs, but then converted into Euro at the fixed exchange rate.

In 2009, the TGAP was replaced by a fee on the non-point agricultural pollution (*redevance pour pollutions agricoles diffuses*). Contrary to the TGAP, only three different pesticide categories are established. Pesticide products being based on mineral AS are charged at the lowest level (€ 0.9/kg AS). Pesticides that are considered to be dangerous to the environment are charged at € 2/kg AS. Pesticides that are mutagenic, carcinogenic, or hazardous to reproduction are charged at the highest level, € 5.10/kg AS (*Art. L213-10-8 and Art. R213-48-13 Code de l'environnement*). The new fee has to be paid at the retail level by the customer. The distributors have to state the fee on the invoice in order to create consciousness for the aim of reducing environmental or health risks of pesticides (OECD, 2011; for more information about awareness raising see Oskam *et al*., 1997). The total revenues of the fee amounted up to € 60 million in 2012 and 2013 (République Française, 2014). Half of these revenues are dedicated to water utility and sewage treatment operators in dependence of the regional pesticide contamination in the water. The remaining tax revenues are invested in other measures of the NAP (OECD, 2011). However, 50% of the fee’s revenues cannot cover the expenses of the water operators for the cleaning of the pesticide contamination, which are estimated to be between € 50–100 million per year (Bommelaer and Devaux, 2012). Additionally, the OECD proposed to internalize further external costs, *e.g.*, costs for biodiversity loss (OECD, 2011). Therefore, the OECD evaluates the new fee as too low.

The French NAP *écophyto 2018* lasts from 2008 until 2018 and aims to reduce the total pesticide usage by 50% (MAP, 2008). This NAP was revised in 2015 and the extended NAP *écophyto II* lasts from 2015 to 2025 still with a reduction goal of 50% compared to 2015 levels (MAAF and MEDDE, 2015). When assuming that the applied quantity is equal to the sales quantity, the sales quantity serves as a simple pesticide use indicator. This means that under écophyto 2018, a reduction from about 80,000 t AS in 2008 (Fig. 2.1) to 40,000 t in 2018 has to be achieved. Since the introduction of the tax, the sold amounts decreased sharply to about 66,700 t AS in 2013. Nevertheless, it should be mentioned that the overall pesticide sales in France are quite volatile and that other factors also influence the amount of pesticides applied. Similar to Sweden, the substitution to low dose AS might be relevant, *e.g.*, the market share of copper and sulfur ingredients decreased by 40% from 2001 to 2004 (Aubertot *et al.*, 2005). For this reason, pesticide volume reduction targets in the NAPs are criticized because less hazardous, high-dose products are replaced by more hazardous, low dose products (Barzman and Dachbrodt-Saaydeh, 2011). The new NAP écophyto II is also of special interest, because a quota system is established and tested (*certificats d'économie de produits phytopharmaceutiques*), which is, to our knowledge, the first one worldwide. For more information, see (Belassen, 2015).

2.3.5 Tax Discussions in Other European Countries

Recently, several other European countries have been discussing about implementing pesticide taxes. The Netherlands, for example, have had several pesticide tax debates which led to the denial of a proposed taxation in the beginning of the millennium (Hof *et al*., 2013; Tweede Kamer, 2003a). Arguments against a fiscal instrument were the relatively high organizational effort (Tweede Kamer, 2003b), the low elasticity of demand, and the higher burdens for domestic producers as well as leakage through import (Hof *et al*., 2013; Boon *et al*., 2012; for the elasticity see also Skevas *et al*., 2012). The aims proclaimed in the NAP until 2010 were reached at least partially: large parts of targets regarding environmental load and residue limits in water supply and food have been reached (Eerdt *et al*., 2012). Looking at the overall sold quantity of pesticides, it can be seen that the sales numbers reduced by almost fifty percent since the 1980s (Fig. 2.1). This decrease was mainly due to stricter obligations for soil fumigants, of which in 1985 10,800 t AS were used and in 2005 only 1,400 t AS (CBS *et al*., 2015).

Belgium is currently undertaking further research if a tax on pesticides would be useful and constructive (ARCADIS Belgium, 2014). In the 1990s, even a draft for a law was formulated in which selected pesticides should have been charged, but the law did not pass (ECOTEC *et al*., 2001). Meanwhile, stakeholders of the agri-food chain, which includes pesticide producers and distributors, have to finance the Belgian Federal Agency for the Safety of the Food Chain by a yearly fee depending on the number of authorized plant protection products that are sold (*Art. 3 Loi relative au financement de l'Agence fédérale pour la Sécurité de la Chaîne alimentaire* of 9 December 2004 and the periodically amended *Royal Decree Arrêté royal fixant les contributions visées à l'article 4 de la loi du 9 décembre 2004 relative au financement de l'Agence fédérale pour la Sécurité de la Chaîne alimentaire* of 10 November 2005).

Additionally, in Switzerland, there has been an ongoing debate about pesticide taxes since the 1990s. So far, the Swiss Agency for the Environment, Forests and Landscape argued against pesticide taxes due to the assumed higher effectiveness of other measures (*e.g*., cross-compliance, registration guidelines, or agri-environmental measures), insufficient knowledge about the tax effects, and too high burdens for Swiss farmers. However, there is some public and political pressure to further promote a reduced application of pesticides. In this context, the necessity of a NAP was analyzed and also the effects of a possible tax are re-analyzed (EAER, 2014).

In Germany, the state minister of agriculture from Schleswig-Holstein proposed to introduce a tax on pesticides in October 2015. The proposal is based on a study of Möckel *et al.* (2015). The suggestion is a tax scheme which is related to the Norwegian one. A base rate of € 20/ha shall be multiplied by a human risk indicator. Additionally, ready-to-use products and pesticides that are on the EU list for substitution shall be levied at higher rates. The tax is proposed to be paid by the industry or the wholesalers. Short-term and long-term reductions of pesticide use are expected to be about 20% and 35% in arable farming. The tax would increase the costs of pesticides by 40% to 50% per hectare (Möckel *et al.*, 2015).

2.4 Analysis of Implementation and Objective Achievement

While Norway, Denmark, and France established differentiated tax schemes, Sweden sticks to a fixed tax scheme, which has not changed since the 1980s apart from raising the tax rates and abolishing the additional price regulation charge. In Table 2.5, we present an overview of the different schemes based on the design variations introduced in Table 2.2. Norway, Denmark, and France use a pesticide risk indicator as well as the amount of AS for the calculation of the differentiated tax. Sweden only uses the amount of AS for the calculation of the flat tax.

Table 2.5. Overview of the different pesticide tax schemes currently in place

State	Charge		Imposition point	Use/refunding of revenues	
	Tax base	**Tax rate**		**Organization**	**Target**
Sweden	active substances	fix	industry, importers/ wholesalers	Swedish state	state budget
	all pesticides	low/medium tariff, flat tax			
Norway	active substances, environmental risk, human health risk	differentiated	industry, importer/ wholesalers	Norwegian state	state budget
	all pesticides	low – medium – high tariffs			
Denmark	active substances, environmental risk, human health risk	differentiated	wholesalers/ importers	Danish state – different ministries	state budget, agricultural fund, green growth measures, administration
	all pesticides	low – medium – high tariffs			
France	active substances, human health risk, (environmental risk)	differentiated	retailers/ distributor	agricultural and environmental sector water utility and sewage treatment operators	measures of the NAP, cleaning of water
	all pesticides	low – medium tariffs			

All of the countries established NAPs in which different objectives for the reduction of pesticides' application or the reduction of pesticides' risks are defined. Building upon the framework derived above, we now analyze whether the different established tax schemes are consistent with the defined goals of the NAPs. The mentioned indicators for the analysis were applied to show the advantages and disadvantages of each scheme. The summarized results of this analysis are presented in Table 2.6. Moreover, a comparison of tax levels of selected plant protection products in the four countries is given in Table 2.4 and serves as decision support for the evaluation.

The main advantage of the Swedish tax scheme is its simplicity. The transaction costs that occur due to the administrative effort of such a scheme are low. Essentially, Sweden could reach its NAP goals by this tax, but the tariffs of the scheme are relatively low and, consequently, not very effective due to the low price elasticity of demand for pesticides. In Sweden, herbicides have the highest share of sold AS. For the period 1950–1989, Gren (1994a, 1994b) reports a relatively high elasticity between -0.93 and -0.97 for herbicides. Therefore, this is in line with the reductions of sold AS in the 1980s. In the last two decades, however, the elasticity of herbicides seems to be lower, probably due to high reductions in previous years but also due to the increased popularity of conservation tillage (see Section 2.3.2 and 2.3.3). Furthermore, no differentiation between pesticides with different loads takes place. This may have led to a reduction of some selected pesticides, which need a high dose, have relatively high price elasticities, or effective substitutes. Nevertheless, the tax increases in the past years (*e.g.*, from SEK 20 to SEK 30/kg AS in 2003) have not led to a permanent reduction of the human health and environmental risk indicators, and the dose/ha even increased. For water threshold violations, only ambiguous results can be observed.

Norway has defined two main objectives in the NAP. The established tax scheme has relatively high tariffs on more hazardous pesticides and the tax scheme promotes farmers to use pesticides with a lower environmental load. This has led to a substitution of pesticides, but farmers' reliance on pesticides remains relatively high. Therefore, this scheme is not fully effective in reaching the objective of increasing farmers' compliance with integrated pest management, which has resulted in increased doses being applied, although of less hazardous pesticides. Farmers are burdened by the tax when they substitute to pesticides of the lower risk categories as well. This has a restricting effect to not increase the use of less hazardous pesticides too much. Whether the objective of fewer violations of maximum residue limits can be achieved by such a tax scheme, remains unclear because of difficulties in observing effects that go along or are correlated with the tax introduction. At least, it seems that the tax has not led an increase of water threshold violations, as in most regions and for many AS a reduction can be found (Bechmann *et al*., 2014). However, other accompanying measures that are defined in the NAP are also relevant for this development. These comprise, for instance, better information techniques or the promotion of improved and more precise spraying techniques. The tax calculation in the Norwegian scheme is sometimes not straightforward, because the SAD is determined based on the recommended pesticide dose in one crop only. In addition, in the calculation it is assumed that the higher the SAD, the lower the risk of a certain pesticide, which is not always the case. This is also revealed by some big tax differences for single products between the Norwegian and the Danish scheme (for example for the herbicides Boxer® and Roundup® Max, Table 2.4), where the latter system accounts for loads explicitly.

Table 2.6. Advantages and disadvantages of existing pesticide taxation schemes. The range of this table is as follows: a "–" is a disadvantage, a "+" is an advantage, and a "±" means that no specific effect can be observed

	Country's tax scheme			
Criteria for analysis (section 2.2)	**Sweden** fix, SEK 34/kg AS	**Norway** differentiated, 7 categories	**Denmark** differentiated, individual tax	**France** differentiated, 3 categories
Main objective of NAP	NAP 2013-2017 (1) No violations of residue limits (2) Reduce pesticides' input	NAP 2010-2014 (1) 70% of farmers apply integrated pest management (2) No violations of residue limits	NAP 2013-2015 40% load reduction	NAP 2008-2018 50% use reduction from 2008 to 2018 and from 2015 to 2025
Effectiveness	± (1, 2) in principle possible but tax level too low to cause large reductions, no further long-term reductions after the last tax increases	(1) + farmers use less hazardous pesticides (2) ± farmers use same amount or more pesticides	+ very high taxes on high load pesticides (use reduction)	±/+ overall relatively low taxes for all three categories but nevertheless use reduction since implementation of tax
Efficiency	– (1, 2) additional burden for farmers but no reduction in use	(1) ± Less hazardous pesticides are relatively cheap (2) – more costs but same amount used	± farmers can choose for low taxed pesticides; some products may disappear – potential resistance problems	±/+ relatively low additional costs for farmers but use reduction is achieved

Table 2.6. *(continued)*

Criteria for analysis (section 2.2)	Country's tax scheme			
	Sweden	**Norway**	**Denmark**	**France**
Feasibility, maintainability and enforceability	+ easy to enforce	–/± for some products complicated tax determination	± rather complicated scheme	+ easy to enforce
Polluter pays principle, ability to differentiate taxation	– fix tax scheme, only choice to not pay is not using pesticides at all	+/± seven different categories; disputes about tax calculation	+ individual taxation, almost no tax for products with low load	–/± only three categories; few choices can be made; revenues for water operators
No economic consequences for farmers, homogeneity	± relatively low tax per ha, especially for low dose products; only few intensive pesticide users in Sweden (few fruit and vegetable farming)	– also when choosing less hazardous pesticides a tax is charged, probably less effective plant protection; no return to the sector	± reduction of property tax on agricultural land; revenues returned to sector; high tax when some products have to be used, maybe production losses or changes otherwise, hereby potential for tax savings compared to old tax scheme	± low tax per ha; tax revenues flow only partly back into the agricultural sector (*e.g.* via environmental programs)
Support among farmers, acceptability	– despite relatively low tax burden, some cost increases occur; no tax in most other European countries	– tax burden also when choosing for less hazardous pesticides; no tax in most other European countries	– some products may be too expensive to use, *e.g.* insecticides; no tax in most other European countries	– despite relatively low tax burden, some cost increases occur; few categories; no tax in most other European countries

In the Danish tax scheme, an individual tax is calculated according to the chemical, environmental, and application-specific characteristics of each pesticide. Similar to the Norwegian scheme, more hazardous pesticides are taxed at higher rates. Hereby, the tax differences between single products can be very high (Table 2.4). Focusing on the taxes per ha of single products, it can be observed that the tax range is much higher compared to the Norwegian scheme. The revenues are, to a large extent, designated for agricultural and environmental measures. The aim of the Danish Government is to decrease the load of pesticides by 40% until the end of 2015. Such a scheme is in line with the polluter pays principle. However, since some products (often insecticides) are taxed very high, they may disappear from the market (like the above mentioned Cythrin 500; SEGES P/S, 2015). This also implies potentially large cost increases for some farmers, for example when growing cereals (Hansen, 2013). In contrast, other cropping systems (maize or grassland) are less affected or even have potential to save tax payments, at least when comparing to the old tax system (Hansen, 2013). When the cropping system cannot be changed, critics argue that a consequence could be an increasing use of cheaper pesticides (Nørring, 2013) probably without considering the AS' classification to avoid resistances, *e.g.*, according to the chemical classification of the Herbicide/Insecticide/Fungicide Resistance Action Committees.

France has two tier tax schemes to foster a change in the use of pesticides. Firstly, synthetic pesticides are taxed with the regular VAT rate. Pesticides being allowed in organic agriculture are charged with the reduced VAT, giving a comparative advantage to organic farming. Secondly, a three-category differentiated scheme was introduced. On the one hand, pesticides that are mutagenic, carcinogenic, or hazardous to reproduction are taxed relatively high in this scheme. On the other hand, pesticides that are allowed in organic farming and those that are less hazardous are taxed at a lower rate. It appears that this scheme gives incentives for a reduction of products that are levied at the high rate. However, the overall tax that has to be paid by a farmer also depends on the dose per ha of a product. Pesticides of the high tax categories often need a relatively small dose and, therefore, the tax burden per ha might be low (see Table 2.4 for examples). In contrast, less hazardous, low-levied pesticides get relatively cheaper at a first glance, but these products often have to be applied at a higher dose (*e.g.*, a 20% copper sulfate product is taxed at € 0.18/kg, but their dose is 25 kg/ha, which results in a tax/ha of € 4.5). Thus, in some cases this policy gives incentives to switch to low-dose pesticides. For this reason, the French policy objective of reducing the overall pesticide use is consistent with the French tax but not fully consistent with the targets of a differentiated scheme, which is to reducing the load caused by pesticides. Although the additional costs for farmers are relatively low compared to Norway and Denmark, a use reduction (of AS) is achieved. Even though other measures contributed to this development, the relatively high reduction effects induced by a relatively low tax was confirmed in a simulation study by Jacquet *et al.* (2011), in which a 20% reduction is reached by a 16% *ad valorem* tax.

2.5 Discussion and Conclusions

Four European countries implemented special taxes in order to control the use of pesticides. Several other countries recently discussed such an instrument. In this article, the tax schemes of Sweden, Norway, Denmark, and France were presented and analyzed. European countries defined different objectives in their NAPs that comprise goals to promote agro-ecological transition, among others, the application reduction, the reduction of the pesticides' load, the increase of integrated pest management, and the non-violation of maximum residue limits. Not all of the established tax schemes were found to be in line with the goals defined in the NAPs. A highly-differentiated tax scheme which leads to a reduction of highly-hazardous pesticides can result in an increase of the application of less hazardous pesticides if suitable alternatives are available. Those pesticides often need a higher dose per hectare so that the overall sold volume is not reduced. If the environmental behavior of pesticides is not included adequately in the tax calculation, those pesticides can also be transported into bodies of water. Thus, violations of threshold values can still occur, although shifted to less hazardous products. Nevertheless, it can be stated that when the reduction objective is well defined and the chosen indicators are well developed, differentiated taxes can be an effective environmental policy instrument in the long-term and a contribution to integrated pest management. However, in order to reduce the reliance on pesticides significantly (and not making them dependent on less hazardous ones), a tax scheme has to go hand in hand with accompanying measures promoting preventive measures of integrated pest management. In the short-term, no environmental and human health benefits will be observed due to large hoarding activities by retailers and farmers (peaks in Fig. 2.1 – Fig. 2.3), and low price elasticities (Skevas *et al.*, 2012; Falconer and Hodge, 2000). So far, no clear indication could be found in the literature whether a tax is efficient or not: for example, Reus *et al.* (1994) find no notable or only a moderate effectiveness, whereas Oskam *et al.* (1997) and Falconer (1998) notice a high/positive effectiveness. Our analysis shows that taxes are a potentially effective instrument, but appropriate economic, political, and environmental circumstances have to be given.

The identification of the effects of taxes is hampered, because often many policy measures are introduced at the same time and farmers may change their behavior so that versatile effects occur. Many of those alternative measures are effective as well. Examples are regulatory measures, increasing information, persuasion and awareness, technological or institutional change, arrangements, or other economic instruments (see Oskam *et al.*, 1997, for a list of measures). For instance, in the Netherlands, banning-alike permissions for soil fumigants led to a 50% reduction of total pesticides sold. Additional insights from micro-studies are, therefore, needed to evaluate the efficiency of pesticide taxes, also accounting for accompanying measures. In addition, an enhancement of the indicators is necessary to better reflect the use and risks of different pesticides on farms. Ideally, as proposed by Benbrook *et al.* (2002), such indicators should consist of both a pesticide use and a pesticide risk indicator, so that both the exposure and the toxicity are covered. This is, for example, done in the model SYNOPS (Gutsche and Strassemeyer, 2007), but also in the calculation of the tax in Norway and Denmark. The feasibility, maintainability, and enforceability should be unproblematic in developed countries with well-established institutions and modern pesticide application techniques. Moreover, all countries implemented taxes on pesticide products at the industry or distributor level and not on pesticide use at the farm level. Problems occur sometimes in the calculation of differentiated tax schemes, because, although they are often based on a pesticide risk indicator, some political judgments have to be made by the authorities.

Taxes can have different targets. For example, the goal can be to generate revenues (either for the sake of general revenues or according to the polluter pays principle) or to create incentives to use fewer or less hazardous pesticides. It is also possible to combine both targets, but sometimes the target pursued by a government remains unclear. The Danish scheme follows the polluter pays principle since every pesticide is taxed by its individual load. The French tax scheme follows both objectives, but by only differentiating three categories, the polluter pays principle is not adequately followed. The Swedish tax does not consider the polluter pays principle. Notwithstanding this, also due to many accompanying measures, decreases of the pesticide risk indicator in the beginning of the 1990s could be observed. While the general structure of the tax is still in line with the objective of the NAPs, the tax seems not in line with current knowledge on the design of pesticide taxation and the developments in other countries. At first sight, the economic consequences for farmers are negative in most schemes since a higher price has to be paid for the products but the pesticide use reductions are small. This is due to the low price elasticity of demand for pesticides, which also limits the effectiveness of taxes with respect to pesticide use reductions. From a polluter pays-point of view, this is in line with policy targets and promotes transition to integrated agricultural systems that are less dependent on chemical inputs, albeit a change towards more organic farming could not be observed in the wake of tax introductions or increases – according to Eurostat data (Eurostat, 2016b). Additionally, no clear pattern could be observed if the transition to more organic farming influences the pesticide use statistics. Negative economic consequences for farmers might be reduced if tax revenues are fed back into the sector, as it is the case in Denmark and partly in France. Hereby, spending the tax revenues for explicit environmental measures (biocontrol, buffer zones, *etc.*) could generate a leverage effect on the effectiveness of tax schemes. In differentiated tax schemes, less hazardous pesticides are also taxed. From an incentive creating-point of view, however, a differentiated tax scheme should offer untaxed or very low taxed, less hazardous pesticides, creating incentives to switch to these products. Nevertheless, the applicability of the polluter pays principle is limited due to the difficulties of calculating all (external) costs and benefits of a pesticide application.

Acknowledgements

The authors thank the Swiss Federal Office for Agriculture for funding and supporting this research. We also thank Anne Strøm Prestvik of the Norwegian Institute of Bioeconomy Research for providing information about the Norwegian pesticide tax scheme. Moreover, we would like to thank two anonymous reviewers for constructive comments and suggestions on an earlier draft of this paper.

2.6 Appendix

The supplementary materials (*i.e.* the data corresponding to the figures) are available online at https://doi.org/10.3390/su8040378.

2.7 References

ARCADIS Belgium (ed.) (2014). Vergroening van de fiscaliteit. Bijlage 2: Groslijst vergroeningsopties. Brussels: ARCADIS Belgium nv/sa, 42-46.

Aubertot, J.-N., Barbier, J.-M., Carpentier, A., Gril, J.-J., Guichard, L., Lucas, P., Savary, S., Voltz, M. and Savini, I. (2005). Pesticides, Agriculture and the Environment—Reducing the Use of Pesticides and Limiting Their Environmental Impact. Rennes and Lyon: INRA and Cemagref.

Baeumer, K. (1992). *Allgemeiner Pflanzenbau*. 3rd ed. Stuttgart: UTB für Wissenschaft – Verlag Eugen Ulmer.

Barzman, M. and Dachbrodt-Saaydeh, S. (2011). Comparative analysis of pesticide action plans in five European countries. *Pest Management Science* 67: 1481-1485.

Bechmann, M., Stenrød, M., Pengerud, A., Grønsten, H. A., Deelstra, J., Eggestad, H. O. and Hauken, M. (2014). Erosjon og tap av Næringsstoffer og Plantevernmidler fra Jordbruksdominerte Nedbørfelt – Sammendragsrapport fra Program for Jord- og Vannovervåking i Landbruket (JOVA) for 1992–2013. Rapport Volume 9. Ås, Norway: Bioforsk.

Belassen, V. (2015). Les Certificats D'économie de Produits Phytosanitaires: Quelle Contrainte et Pour qui. Working Paper 2015/4, Centre d'Economie et Sociologie appliquées à l'Agriculture et aux Espaces Ruraux. CESAER INRA: Dijon.

Benbrook, C. M., Sexson, D. L., Wyman, J. A., Stevenson, W. R., Lynch, S., Wallendal, J., Diercks, S., van Haren, R. and Granadino, C. A. (2002). Developing a pesticide risk assessment tool to monitor progress in reducing reliance on high-risk pesticides. *American Journal of Potato Research* 79: 183-199.

Bergkvist, P. (2004). Pesticide Risk Indicators at National Level and Farm Level - A Swedish Approach. Nr 6/04. Sundbyberg, Sweden: Kemikalieinspektionen.

Bol, V. van, Claeys, S., Debognie, P., Godfriaux, J., Pussemier, L., Steurbaut, W. and Maraite, H. (2003). Pesticide Indicators. *Pesticide Outlook* 14: 159-163.

Bommelaer, O. and Devaux, J. (2012). Le Financement de la Gestion des Ressources en eau en France (Actualisation de Janvier 2012). Études & Documents Nr. 62. La Défense: Commissariat Général au Développement Durable.

Boon, P. E., Donkersgoed, G. van, Noordam, M., Biesebeek, J. D. te, Ven–van den Hoogen, B. M. van den and Klaveren, J. D. van (2012). Evaluatie van de nota Duurzame gewasbescherming – Deelrapport Voedselveiligheid. RIVM Rapport 320038001/2012. Bilthoven: Rijksinstituut voor Volksgezondheid en Milieu.

CBS – Centraal Bureau voor de Statistiek, Planbureau voor de Leefomgeving and Wageningen UR (eds) (2015). Afzet van Chemische Gewasbeschermingsmiddelen in de Land- en Tuinbouw, 1985–2013. Online: http://www.compendiumvoordeleefomgeving.nl/indicatoren/nl0015-Afzet-gewasbeschermingsmiddelen-in-de-land–en-tuinbouw.html?i=11-61 [22 January 2018].

EAER – Federal Department of Economic Affairs, Education and Research (ed.) (2014). Bedarfsabklärung eines Aktionsplans zur Risikoreduktion und nachhaltigen Anwendung von Pflanzenschutzmitteln – Bericht in Erfüllung des Postulates 12.3299 von Frau Nationalrätin Tiana Angelina Moser vom 16. März 2012. Bern: Eidgenössisches Departement für Wirtschaft, Bildung und Forschung.

ECOTEC, CESAM, CLM, University of Gothenburg, UCD and IEEP (eds) (2001). Study on the Economic and Environmental Implications of the Use of Environmental Taxes and Charges in the European Union and its Member States. Brussels, Belgium and Birmingham: ECOTEC Research & Consulting.

Eerdt, M. van, Dam, J. van, Tiktak, A., Vonk, M., Wortelboer, R. and Zeijts, H. van (2012). Evaluatie van de Nota Duurzame Gewasbescherming. PBL-Publicatienummer 500158001. Den Haag: Planbureau voor de Leefomgeving.

European Commission (2015). VAT Rates Applied in the Member States of the European Union – Situation at 1st September 2015. Taxud.c.1. Online: http://ec.europa.eu/taxation_customs/resources/documents/taxation/vat/how_vat_works/rates/vat_rates_en.pdf [7 January 2016].

Eurostat (2008). A common methodology for the collection of pesticide usage statistics within agriculture and horticulture. 2008 ed. Luxembourg: Office for Official Publications of the European Communities.

Eurostat (2015). Exchange Rates – Bilateral Exchange Rates – Euro/ECU Exchange Rates. Luxembourg. Online: http://ec.europa.eu/eurostat/data/database [22 January 2018].

Eurostat (2016a). Agri-Environmental Indicators – Farm Management – Pesticide Sales. Luxembourg. Online: http://ec.europa.eu/eurostat/data/database [12 April 2016].

Eurostat (2016b). Agriculture – Organic Farming – Number of Certified Registered Organic Operators by Type of Operators. Luxembourg. Online: http://ec.europa.eu/eurostat/data/database [12 April 2016].

Falconer, K. E. (1998). Managing diffuse environmental contamination from agricultural pesticides: An economic perspective on issues and policy options, with particular reference to Europe. *Agriculture, Ecosystems and Environment* 69: 37-54.

Falconer, K. and Hodge, I. (2000). Using economic incentives for pesticide usage reductions: Responsiveness to input taxation and agricultural systems. *Agricultural Systems* 63: 175-194.

Gianessi, L., Rury, K. and Rinkus, A. (2009). An evaluation of pesticide use reduction policies in Scandinavia. *Outlooks on Pest Management* 20: 1-7.

Gregoriou, P., Manuneas, T. and Pashardes, P. (2009). Agricultural Support Policies and Optimum Tax and Levy Scheme for Pesticide Use in Farm Production. Economic Analysis Papers No. 03-09. Nicosia: Economics Research Centre, University of Cyprus. Online: https://www.ucy.ac.cy/erc/documents/DOA_03-09.pdf [7 January 2016].

Gren, I.-M. (1994a). Cost efficient pesticide reductions: A study of Sweden. *Environmental and Resource Economics* 4: 279-293.

Gren, I.-M. (1994b). Regulating the farmers' use of pesticides in Sweden. In Opschoor, H. and Turner, K. (eds), *Economic Incentives and Environmental Policies – Principles and Practice*. Dordrecht: Kluwer Academic Publishers, 153-173.

Gutsche, V. and Strassemeyer, J. (2007). SYNOPS - A model to assess the environmental risk potential of pesticides. *Journal für Kulturpflanzen* 59: 197-210.

Hansen, L. F. (2013). New Pesticide Tax and Indicator in Denmark – National Action Plans. Presentation at the International Seminar on a New Danish Pesticide Tax, 30 May 2013. Copenhagen: Danish Environmental Protection Agency.

Hoevenagel, R., van Noort, E. and de Kok, R. (1999). *Study on a European Union wide regulatory framework for levies on pesticides*. EIM Small Business Research and Consultancy. Zoetermeer: Haskoning.

Hof, B., Koopmans, C., Rougoor, W. and Voort, J. van der (2013). Effecten en vormgeving van een heffing op gewasbeschermingsmiddelen. Rapport nr. 2013-39. Amsterdam: seo economisch onderzoek.

Jacquet, F., Butault, J. P. and Guichard, L. (2011). An economic analysis of the possibility of reducing pesticides in French field crops. *Ecological Economics 70*: 1638-1648.

Jørgensen, L. N. and Ørum, J. E. (2013). Farmers' Possibilities for Shifting to Pesticides with Lower Load and Manage Resistance – Fungicides and Insecticides. Presentation at the International Seminar on a New Danish Pesticide Tax, 30 May 2013. Copenhagen: Danish Environmental Protection Agency.

Kemikalieinspektionen (ed.) (2015). Databaser – Bekämpningsmedelsregistret, Sundbyberg, Sweden. Online: http://webapps.kemi.se/BkmRegistret/Kemi.Spider.Web.External/ [22 January 2018].

Landbruksdepartementet – Ministry of Agriculture (ed.) (2004). Action Plan on Reducing Risk Connected to the Use of Pesticides (2004-2008). Publication number M-0727 E, Landbruksdepartementet. Online: https://www.regjeringen.no/globalassets/upload/lmd/vedlegg/brosjyrer_veiledere_rapporter/action_plan_on_reducing_risk_connected_to_the_use_of_pesticides_2004_2008.pdf [7 January 2016].

Landbruks- og matdepartementet – Ministry of Agriculture and Food (ed.) (2009). Handlingsplan for Redusert Risiko ved Bruk av Plantevernmidler (2010-2014), National Action Plan from 18 September 2009. Online: https://www.regjeringen.no/globalassets/upload/lmd/vedlegg/handlingsplan_plantevern_2010_2014.pdf [22 January 2018].

Landsbygdsdepartementet – Ministry of Rural Affairs (ed.) (2013). *National Action Plan for the Sustainable Use of Plant Protection Products for the Period 2013–2017*. Landsbygdsdepartementet: Stockholm.

Larsson, M., Boström, G., Gönczi, M. and Kreuger, J. (2014). Kemiska Bekämpningsmedel i Grundvatten 1986–2014 – Sammanställning av Resultat och Trender i Sverige under tre Decennier, Samt Internationella Utblickar. Havs och Vattenmyndigheten Rapport 2014:15, CKB Rapport 2014:1. Uppsala: Havs och Vattenmyndigheten, KompetensCentrum för Kemiska Bekämpningsmedel.

Lefebvre, M., Langrell, S. R. H. and Gomez-y-Paloma, S. (2015). Incentives and policies for integrated pest management in Europe: a review. *Agronomy for Sustainable Development* 35: 27-45.

MAAF and MEDDE – Ministère de l’Agriculture, de l’Agroalimentaire et de la Forêt and Ministère de l’Écologie, du Développement durable et de l’Énergie (2015). Plan Écophyto II. Paris: République Française.

MAP – Ministère de l’Agriculture et de la Pêche (2008). Écophyto 2018. Paris: République Française.

MAP – Ministère de l’Agriculture et de la Pêche (2016). Avis Relatif à la Liste des Substances Dangereuses du Plan Interministériel de Réduction des Risques Liés aux Pesticides. Journal Officiel de la République Française, NOR: AGRG0602464V. Online: http://www.legifrance.gouv.fr/jo_pdf.do?id=JORFTEXT000000423707 [7 January 2016].

Mattilsynet – Norwegian Food Safety Authority (ed.) (2014a). Miljøavgift for Plantevernmidler (Februar 2014). Online: http://www.mattilsynet.no/language/english/plants/plant_protection_products/taxes_for_plant_protection_products.17427 [7 January 2016].

Mattilsynet – Norwegian Food Safety Authority (ed.) (2014b). National Monitoring Program for Residues of Pesticides in Food. Melding nr. 4/2013, nr. 5/2013, nr. 7/2014. Brumunddal, Norway: Mattilsynet. Online:

http://www.mattilsynet.no/language/english/sok/?search=plantevernmidler+ [22 January 2018].

MEDDE – Ministère de l'Écologie, du Développement Durable et de l'Énergie (2015). Montant par Produit Pour l'année 2015. Redevance Phyto & Traçabilité des Ventes. Online: http://redevancephyto.developpement-durable.gouv.fr/distributeur/montantparproduit [7 January 2016].

Miljøministeriet – Ministry of Environment (ed.) (2012). *The Agricultural Pesticide Load in Denmark 2007–2010*. Environmental Review No. 2/2012. Copenhagen: Miljøministeriet.

Miljøministeriet - Ministry of Environment (ed.) (2013). Background and Content of the New Pesticide Tax. Presentation at the International Seminar on a New Danish Pesticide Tax, 30 May 2013. Copenhagen: Danish Environmental Protection Agency.

Miljøministeriet, Ministeriet for Fødevarer, Landbrug og Fiskeri – Ministry of the Environment, Ministry of Food, Agriculture and Fisheries (eds) (2013). Protect Water, Nature and Human Health - Pesticide Strategy 2013–2015. Copenhagen: Danish Ministry of the Environment Information Centre.

Miljø- og Fødevareministeriet (ed.) (2015). Afgift på Sprøjtemidler – Hent Listen over Afgifter pr. 12. Februar 2015. Copenhagen. Online: http://mst.dk/virksomhed-myndighed/bekaempelsesmidler/sproejtemidler/bruger/pesticidafgift/ [7 January 2016].

Möckel, S., Gawel, E., Kästner, M., Liess, M., Knillmann, S. and Bretschneider, W. (2015). Einführung einer Abgabe auf Pflanzenschutzmittel in Deutschland. In Gawel, E. (ed.), *Studien zu Umweltökonomie und Umweltpolitik*. Vol. 10. Berlin: Duncker & Humblot, 1-305.

Natur- og Landbrugskommissionen (ed.) (2012). Natur- og Landbrugskommissionens Statusreport: Bilag 10 - Pesticidafgifter Opkrævning og Anvendelse. Copenhagen: Natur- og Landbrugskommissionen.

Naturvårdsverket (ed.) (2016). Dataunderlag – Växtskyddsmedel - Dataunderlag - Växtskyddsmedel. Online: http://www.miljomal.se/Miljomalen/Alla-indikatorer/Indikatorsida/Dataunderlag-for-indikator/?iid=139&pl=1&t=Land&l=SE [24 March 2016].

Nørring, N. P. (2013). Pesticide Taxation: Does It Work? Presentation at the International Seminar on a New Danish Pesticide Tax, 30 May 2013. Copenhagen: Danish Environmental Protection Agency.

Norwegian Agricultural Inspection Service (ed.) (n.d.). OECD Survey of National Pesticide Risk Indicators, 1999–2000. Online: http://www.oecd.org/chemicalsafety/pesticides-biocides/1934217.pdf [7 January 2016].

Notisum AB (ed.) (2015). Rättsnätet – Lag (1984:410) om Skatt på Bekämpningsmedel. Online: http://www.notisum.se/rnp/sls/lag/..%5Cfakta%5Ca9840410.htm [7 January 2016].

OECD – Organisation for Economic Cooperation and Development (ed.) (2009). OECD Environmental Performance Reviews: Finland. Paris: OECD Publishing.

OECD – Organisation for Economic Cooperation and Development (ed.) (2011). OECD Economic Surveys: France 2011. Paris: OECD Publishing.

Øgaard, A. F. and Skaalsveen, K. (2015). Resultater fra Program for Jord- og Vannovervåking i Landbruket (JOVA) for 1992–2014. Ås, Norway: Norsk Institutt for Bioøkonomi.

Oskam, A. J., Vijftigschild, R. A. N. and Graveland, C. (1997). Additional EU policy instruments for plant protection products. A report within the second phase of the programme: Possibilities for future EC environmental policy on plant protection products. Wageningen: Wageningen Agricultural University.

Pedersen, A. B., Nielsen, H. Ø. and Andersen, M. S. (2015). The Danish pesticide tax. In Lago, M., Mysiak, J., Gómez, C. M., Delacámara, G. and Maziotis, A.(eds), *Use of Economic Instruments in Water Policy – Insights from International Experience*. Cham, Switzerland: Springer International Publishing, 73-87.

Popp, J., Pető, K. and Nagy, J. (2013). Pesticide productivity and food security. A review. *Agronomy for Sustainable Development* 33, 243-255.

République Française (ed.) (2014). Agences de l'eau – Annexe au Projet de loi de Finances Pour 2015. Online: http://www.performance-publique.budget.gouv.fr/sites/performance_publique/files/farandole/ressources/2015/pap/pdf/jaunes/Jaune2015_agences_eau.pdf [7 January 2016].

Reus, J. A. W. A., Weckseler, H. J. and Pak, G. A. (1994). *Towards a future EC pesticide policy – An inventory of risks of pesticide use, possible solutions and policy instruments*. Utrecht: Centre for Agriculture and Environment (CLM).

SEGES P/S (ed.) (2015). Middeldatabasen. Aarhus. Online: https://www.middeldatabasen.dk/ [31 January 2018].

Schou, J. S. and Streibig, J. C. (1999). Pesticide taxes in Scandinavia. *Pesticide Outlook* 10: 227-229.

Skatteministeriet (ed.) (2015). Afgifter - Provenuet af Afgifter og Moms 2009–2016. Online: http://www.skm.dk/skattetal/statistik/provenuoversigter/afgifter-provenuet-af-afgifter-og-moms-2009-2016 [15 February 2016].

Skevas, T., Stefanou, S. E. and Oude Lansink, A. (2012). Can economic incentives encourage actual reductions in pesticide use and environmental spillovers? *Agricultural Economics* 43: 267–276.

Skevas, T., Oude Lansink, A. G. J. M. and Stefanou, S. E. (2013). Designing the emerging EU pesticide policy: A literature review. *NJAS – Wageningen Journal of Life Sciences* 64-65: 95-103.

Spikkerud, E. (2005). Taxes as a tool to reduce health and environmental risk from pesticide use in Norway. In OECD (ed.), *Evaluating Agri-Environmental Policies: Design, Practice and Results*. Paris: OECD Publishing, 281-290.

Spikkerud, E., Haraldsen, T., Abdellaue, A. and Holmen, M. T. (2005). Guidelines for a Banded Pesticide Tax Scheme, Differentiated According to Human Health and Environmental Risks. Brumunddal, Norway: Norwegian Food Safety Authority – National Centre of Plants and Vegetable Foods.

Statskontoret (2010). Effekter på Priset för Handelsgödsel när Skatten på Kväve i Handelsgödsel Avskaffas – en Delrapport. Stockholm: Regeringen – Jordbruksdepartementet.

Strøm Prestvik, A. (2015). Personal communication. Ås, Akershus, Norway: Norwegian Institute of Bioeconomy Research.

Strøm Prestvik, A., Netland, J. and Hovland, I. (2013). Evaluering av Avgiftssystemet for Plantevernmidler i Norge. Online: http://www.mattilsynet.no/planter_og_dyrking/plantevernmidler/mattilsynet_evaluerer_avgiftssystemet_for_plantevernmidler.12116 [31 January 2018].

Stortingets administrasjon (ed.) (2015). Statsbudsjettet for 2015 – Saldert Budsjett Vedtatt i Stortinget Høsten 2014. Oslo: Stortingets Administrasjon – Konstitusjonell Avdeling.

Tweede Kamer (ed.) (2003a). Fiscaliteit, landbouw- en Natuurbeleid, Kamerstuk 28207 nr. 3, Lijst van Vragen en Antwoorden, Vergaderjaar 2002–2003. Den Haag. Online: https://zoek.officielebekendmakingen.nl/dossier/28207/kst-28207-3?resultIndex=12&sorttype=1&sortorder=4 [7 January 2016].

Tweede Kamer (ed.) (2003b). Fiscaliteit, Landbouw- en Natuurbeleid, Kamerstuk 28207, nr. 5, Verslag van een Notaoverleg, Vergaderjaar 2003–2004. Den Haag. Online: https://zoek.officielebekendmakingen.nl/dossier/28207/kst-28207-5?resultIndex=10&sorttype=1&sortorder=4 [7 January 2016].

Zilberman, D. and Millock, K. (1997). Financial incentives and pesticide use. *Food Policy* 22: 133-144.

Chapter 3

A Meta-Analysis on the Elasticity of Demand for Pesticides[5]

Abstract

There is an increasing policy interest in pesticide taxation schemes as a measure to reduce harmful effects of pesticide use. The effectiveness of such tax depends, however, on the price elasticity of demand for pesticides. Moreover, information on these demand elasticities and their determinants is of crucial relevance for policymaking and normative modeling approaches. In this article, we present a meta-analysis based on studies that have estimated pesticide demand elasticities in Europe and North America. Our meta-analysis reveals that the own-price elasticities of demand for pesticides are, with a median of -0.28, significantly smaller than zero, but also significantly larger than -1, *i.e.* to be inelastic. We find that the demand for pesticides for special crops is less elastic than that for arable and grassland. In addition, the demand for herbicides is more elastic than for other pesticides. Studies that consider only short-term horizons and little flexibility for farmers to adjust to price changes generate significantly less elastic pesticide demands. The results also indicate that more recent studies identify lower pesticide price elasticities of demand. Furthermore, we find that peer-reviewed studies tend to find more inelastic results compared to grey literature.

Keywords: Agricultural policy; demand analysis; meta-analysis; own-price elasticity of demand; pesticide tax; pesticides; robust regression.

---

[5] This chapter was published in the *Journal of Agricultural Economics* as:

Böcker, T. G. and Finger, R. (2017). A Meta-Analysis on the Elasticity of Demand for Pesticides. *Journal of Agricultural Economics* 68 (2): 518-533.

3.1 Introduction

Reductions of harmful effects of pesticide use are high on agricultural policy agendas. For instance, national action plans have been established in countries of the European Union and pesticide taxation schemes have been established in some European countries (*i.e.* France, Sweden, Norway, Denmark) (Böcker and Finger, 2016; Lefebvre *et al*., 2015). There have also been recent discussions on an introduction of a taxation scheme in other European countries such as Belgium, the Netherlands and Germany (ARCADIS Belgium, 2014; Hof *et al*., 2013; Möckel *et al*., 2015a,b). The effectiveness of such taxes depends on the demand response of farmers to higher pesticide prices. More specifically, low own-price elasticities of demand have been claimed to be a major hurdle for effective regulation of pesticide use via taxes (*e.g*. Falconer and Hodge, 2000; Fernandez-Cornejo *et al*., 1998). Overviews of estimates of pesticide demand elasticities in developed countries have been provided in studies by Capalbo and Vo (1988), Falconer and Hodge (2000), Fernandez-Cornejo *et al*. (1998) and Hoevenagel *et al*. (1999). These overviews are cited frequently and serve as important references for assumptions made in normative modeling approaches (*e.g*. Möckel *et al*., 2015a) and for comparison with positive studies (*e.g*. Muñoz Piña and Avila Forcada, 2004; Skevas *et al*., 2012).

Despite the importance of these overviews for policy analysis and policy recommendations, there are several shortcomings and unexplored potentials. Most obviously, the existing overviews on pesticide demand elasticities are now outdated – the most recent being 2000.[6] Further limitations result from the fact that no comprehensive statistical meta-analyses on pesti-

[6] More recently, Skevas *et al*. (2013) presented an overview based on Fernandez-Cornejo *et al*. (1998), Hoevenagel *et al*. (1999) and Falconer and Hodge (2000), without adding further studies beyond 1997.

cide demand elasticities have been conducted.[7] Specifically, no distinction between short- and long-term elasticities has been made that accounts for farmers' flexibility to adjust to price changes, especially over longer time horizons. However, this distinction reflects crucial information for normative policy analysis of the effects of pesticide taxation (*e.g*. Möckel *et al*., 2015a). Moreover, there has been no systematic analysis of differences of pesticide demand elasticities across different agricultural systems and types of pesticides. In addition, no distinction between methodological approaches underlying demand elasticity estimates has been considered, despite the fact that a large variety of methodological approaches has been used to derive elasticities (*e.g*. econometric vs. optimisation models). Finally, no distinction between peer-reviewed and grey literature has been made. In this note, we aim to fill these gaps and provide a meta-analysis on pesticide demand elasticities. We focus on results from developed countries of the northern hemisphere to include estimates from comparable agricultural systems.

3.2 Methodology and Data

To identify relevant studies, we used bibliographic databases such as Scopus and Google Scholar, and the key words 'pesticide', 'pesticide tax/levy/charge' together with 'elasticity' (singular and plural). In total, 31 studies were identified that report original pesticide demand elasticities. We provide detailed argumentation on the inclusion and exclusion of studies in Appendix 3.A.

From the selected studies, we obtained the following information, as available: (i) the year of publication, (ii) the period of analysis/data collection period, (iii) the country/ region that was the subject of the analysis, (iv) the cropping system investigated, with classification into arable and grassland farming, special crops (horticulture, fruit production, viticulture), and

[7] The study by Bergh *et al*. (1997) presents a structured meta-analysis of some studies on pesticide demand elasticities, not covering the addressed points here empirically.

an aggregate bundle when no specific production type was analysed, (v) the type of pesticide investigated, *i.e*. herbicides, fungicides, insecticides and pesticides in general, (vi) the considered flexibility/time span, *i.e*. short-term/fix or long-term/variable,[8] (vii) the estimated demand elasticity for pesticides, (viii) min–max values of these demand elasticity estimates, (ix) the method used, with distinction between econometric and normative models,[9] and (x) information on whether the study was peer-reviewed. Note that publications can be listed multiple times if they present demand elasticities, for example, for different regions, agricultural systems, types of pesticides and/or use different methodologies.

We use a non-parametric Wilcoxon–Mann–Whitney test to identify differences across categories, *i.e*. differences between the size of the price elasticities. More precisely, we test if pesticide demand elasticities (i) are different from zero, (ii) differ across regions (Europe and North America), (iii) differ between long- and short-term horizons considered, (iv) differ across agricultural systems (arable and grassland, special crops and an aggregated sector), (v) differ between peer-reviewed and grey publications, and (vi) differ across methodological approaches (comparing econometric and normative models).

We conduct a multiple linear regression analysis to assess the most important determinants of pesticide demand elasticity estimates (η) and to estimate marginal effects:

$$\eta(y, L, S, H, E, P, M) = \beta_0 + \beta_1 y + \beta_2 L + \beta_3 S + \beta_4 H + \beta_5 E + \beta_6 P + \beta_7 M + \varepsilon \tag{3.1}$$

[8] If time span and flexibility are not explicitly defined (self-declared) in the studies, the flexibility of variable inputs serves as a measure for long-term and short-term production. Therefore, longer periods of analysis or data collection periods can cover a short-term horizon if the variables in the models are assumed to be fixed.

[9] Normative approaches comprise mainly optimisation models such as linear programming approaches (profit maximisation or cost minimisation).

Specifically, we consider the following variables as potential determinants of the estimates: the middle year of the analysis or data collection period with the base year being defined as 1900 (*y*).[10] We include dummy variables for long-term horizon considered (*L*), special crops (as agricultural systems with particularly high pesticide use levels) (*S*), focus on herbicides (*H*), European study (*E*), peer-reviewed publication (*P*), and the type of methodology (using a dummy for econometric analysis) (*M*). *b0, ..., b7* are the regression parameters that need to be estimated and e represents the error term. Due to the considerable number of outlying observations in our study (see Fig. 3.1), we use MM-regression, a robust regression technique.[11] To account for the fact that some studies report more than one observation for some variables, cluster-adjusted variance-covariance matrices are used to derive standard errors. If a study reported a range or an interval for the same measure, mean values are used in both the regression analysis and the Wilcoxon–Mann–Whitney test, to avoid double counting (Table 3.1). All statistical analyses are conducted with R (R Core Team, 2015).

[10] In addition, we also employed the year of publication, which produced similar results.

[11] The MM-estimator combines a high breakdown point S-estimator and an efficient M-estimator to obtain a high robustness (avoiding biases in coefficient estimates) against outlying observations by maintaining a high level of efficiency (*i.e*. maintaining low standard errors) (see, for example, Finger, 2010). The interpretation of coefficient estimates remains similar to a standard OLS regression.

Table 3.1. Studies with pesticide demand elasticities

Author (publication year)	Land/region	Pesticide type (general or: f/fungicide, h/herbicide, i/insecticide (if appl. min. to max.)	Flexibility (long-term/ variable, short-term/fix)	Period of analysis/ data collection
Brown and Christensen (1981)	USA	-0.19 (-0.20 to -0.19)	Long-term	1947-1974
Schulte (1983)	Germany/Rhineland	f: -0.32 (-0.45 to -0.19)	Short-term	1978-1980
Schulte (1983)	Germany/Rhineland	f: -0.32 (-0.45 to -0.19)	Long-term	1978-1980
Schulte (1983)	Germany/Bavaria	f: -0.17 (-0.33 to 0.00)	Short-term	1978-1980
Schulte (1983)	Germany/Bavaria	f: -0.50 (-0.67 to -0.33)	Long-term	1978-1980
Schulte (1983)	Germany/Schleswig-Holstein	f: -0.21 (-0.27 to -0.14)	Short-term	1978-1980
Schulte (1983)	Germany/Schleswig-Holstein	f: -0.53 (-0.80 to -0.25)	Long-term	1978-1980
Schulte (1983)	Germany/South Oldenburg	f: -0.32 (-0.44 to -0.20)	Short-term	1978-1980
Schulte (1983)	Germany/Hessen	f: -0.63 (-0.76 to -0.50)	Long-term	1978-1980
Antle (1984)	USA	-0.19	Long-term	1910-1946
Antle (1984)	USA	-0.25	Long-term	1947-1978
Dubgaard (1987)	Denmark	-0.30	Long-term	1971-1985
Dubgaard (1987, 1991)	Denmark	h: -0.69	Long-term	1971-1985
Capalbo (1988)	USA	-0.19	Long-term	1948-1983
Capalbo (1988)	USA	-0.88	Long-term	1948-1983
Capalbo (1988)	USA	-0.41	Long-term	1948-1983
Capalbo (1988)	USA	-0.47	Long-term	1948-1983
Capalbo (1988)	USA	-0.61	Long-term	1948-1983
Capalbo (1988)	USA	-0.70	Long-term	1948-1983
Elhorst (1990)	Netherlands	-0.29	Short-term	1980-1986
Aaltink (1992), Master’s thesis, cit. from Oskam *et al.* (1992)	Netherlands	-0.13 (-0.23 to -0.03)	Short-term	unknown
Aaltink (1992), Master’s thesis, cit. from Oskam *et al.* (1992)	Netherlands	-0.39 (-0.55 to -0.23)	Long-term	unknown
Aaltink (1992), Master’s thesis, cit. from Oskam *et al.* (1992)	Netherlands	-0.39 (-0.55 to -0.23)	Long-term	unknown
McIntosh and Williams (1992)	USA/Georgia	-0.11	Long-term	1950-1986
Oskam *et al.* (1992)	Netherlands	-0.21	Short-term	1970-1988
Oskam *et al.* (1992)	Netherlands	-0.22	Medium-term	1970-1988

Table 3.1. *(continued)*

Author (publication year)	Land/region	Pesticide type (general or: f/fungicide, h/herbicide, i/insecticide (if appl. min. to max.)	Flexibility (long-term/ variable, short-term/fix)	Period of analysis/ data collection
Oskam *et al.* (1992)	Netherlands	-0.22	Long-term	1970-1988
Oskam *et al.* (1992)	Netherlands	-0.25	Short-term	1970-1988
Oskam *et al.* (1992)	Netherlands	-0.26	Medium-term	1970-1988
Oskam *et al.* (1992)	Netherlands	-0.29	Long-term	1970-1988
Rude (1992)	Denmark	-0.20 (-0.22 to -0.17)	Short-term	1987-2004
Rude (1992)	Denmark	-0.24 (-0.27 to -0.20)	Medium-term	1987-2004
Rude (1992)	Denmark	-0.28 (-0.32 to -0.23)	Long-term	1987-2004
Villezca-Becerra and Shumway (1992)	USA/California	-0.09	Long-term	1951-1982
Villezca-Becerra and Shumway (1992)	USA/Florida	-0.17	Long-term	1951-1982
Villezca-Becerra and Shumway (1992)	USA/Iowa	-0.04	Long-term	1951-1982
Villezca-Becerra and Shumway (1992)	USA/Texas	-0.21	Long-term	1951-1982
Fernandez-Comejo (1993)	USA/Illinois	-0.10	Short-term	1949-1982
Fernandez-Comejo (1993)	USA/Illinois	-0.10	Short-term	1949-1982
Fernandez-Comejo (1993)	USA/Illinois	-0.12	Long-term	1949-1982
Fernandez-Comejo (1993)	USA/Illinois	-0.38	Long-term	1949-1982
Fernandez-Comejo (1993)	USA/Indiana	-0.08	Short-term	1949-1982
Fernandez-Comejo (1993)	USA/Indiana	-0.08	Short-term	1949-1982
Fernandez-Comejo (1993)	USA/Indiana	-0.09	Long-term	1949-1982
Fernandez-Comejo (1993)	USA/Indiana	-0.60	Long-term	1949-1982
Chambers and Lichtenberg (1994)	USA	-1.53	Long-term	1949-1990
Chambers and Lichtenberg (1994)	USA	-1.50	Long-term	1949-1990
Chambers and Lichtenberg (1994)	USA	0.05	Long-term	1949-1990
Chen *et al.* (1994)	USA/Alabama	-2.42	Long-term	1949-1986
Gren (1994a)	Sweden	h: -0.97	Long-term	1950-1989
Gren (1994b)	Sweden	h: -0.93	Long-term	1948-1989
Oude Lansink (1994)	Netherlands	-0.12	Short-term	1970-1988

Table 3.1. *(continued)*

Author (publication year)	Land/region	Pesticide type (general or: f/fungicide, h/herbicide, i/insecticide (if appl. min. to max.)	Flexibility (long-term/ vari-able, short-term/fix)	Period of analysis/ data collection
Papanagiotou *et al.* (1994)	Greece	-0.28	Short-term	1961-1990
Papanagiotou *et al.* (1994)	Greece	-0.28	Short-term	1961-1990
Shumway and Chesser (1994)	USA/Texas	h: -0.70 (-2.00 to +0.60)	Long-term	1972-1986
Bauer *et al.* (1997)	Germany/Hessen	f: -0.02	Short-term	1991-1993
Carpentier and Weaver (1997)	France/Ile-de-France, Centre,	-1.55 (-1.97 to -1.13	Long-term	1987-1990
Falconer (1997), Dissertation cit. from Falconer (1998)	UK/East Anglia	-0.20 (-0.30 to -0.10)	unknown	unknown
Komen *et al.* (1997)	Netherlands	-0.14	Short-term	1990
Komen *et al.* (1997)	Netherlands	-0.11	Short-term	1990
Komen *et al.* (1997)	Netherlands	-0.25	Long-term	1990
Oude Lansink and Peerlings (1997)	Netherlands	-0.48	Long-term	1970-1992
Russell *et al.* (1997)	UK/North West	-1.11 (-1.12 to -1.09)	Short-term	1989-1993
Jacquet *et al.* (2011)	France	-0.77 (-1.25 to -0.28)	Combined	2002-2007
Ivanova *et al.* (2012)	Bulgaria	-0.11 (-0.12 to -0.09)	Long-term	2002-2008
Ivanova *et al.* (2012)	Bulgaria	-0.16 (-0.19 to -0.12)	Long-term	2002-2008
Ivanova *et al.* (2012)	Portugal	-0.19 (-0.22 to -0.16)	Long-term	2002-2008
Ivanova *et al.* (2012)	Portugal	-0.20 (-0.23 to -0.16)	Long-term	2002-2008
Skevas *et al.* (2012)	Netherlands	-0.02 (-0.03 to -0.0003)	Short-term	2003-2004
Fadhuile *et al.* (2015)	France	h: -0.63	Long-term	2001
Fadhuile *et al.* (2015)	France	h: -0.68	Long-term	2006
Femenia and Letort (2016)	France/Meuse	-0.17 (-0.24 to -0.10)	Long-term	2007-2012

3.3 Results

Table 3.1 lists the studies that are included in the meta-analysis and some selected variables. Note that the tables with the complete information collected, as well as the codes for statistical analyses, are available in Appendices 3.B to 3.C. Fig. 3.1 presents the elasticities across all studies by the publication year of the studies and type of pesticides considered.

The median elasticity across all studies is -0.28. Using the Wilcoxon-Mann-Whitney test, we can reject the null-hypotheses that the elasticity is equal or >0 at the 1% level. Fig. 3.1 shows that the vast majority of studies appeared in the 1990s, though more recent studies exist for Europe, which reflects the re-emerging policy relevance of the pesticide demand regulation. Despite the fact that some studies report on particular types of pesticides or even on individual products (*e.g*. Shumway and Chesser, 1994), the majority of studies present elasticities for pesticides in general so that no patterns can be observed.

Fig. 3.2 presents the distribution of demand elasticities by continent. European studies (22 studies, 64 observations) dominate over North American studies (nine studies, 30 observations, only USA) in quantity. The median of demand elasticities is lower in the US (-0.20) than in Europe (-0.30). However, the variability within the US studies is larger, with an interquartile range of 0.51 compared to 0.30 for Europe. The null-hypothesis of no differences across continents cannot be rejected.

Fig. 3.3 presents the distribution of demand elasticities with respect to input flexibility and time horizon assumed in the studies. We find that the majority of studies analyse long-term effects, accounting for various possibilities to adjust the production system and techniques. As expected, the longer-term perspectives result in more elastic pesticide demand estimates (median = -0.39) compared to short-term perspectives (median = -0.18). The null-hypothesis of no differences across time horizons is rejected at the 1% level of significance.

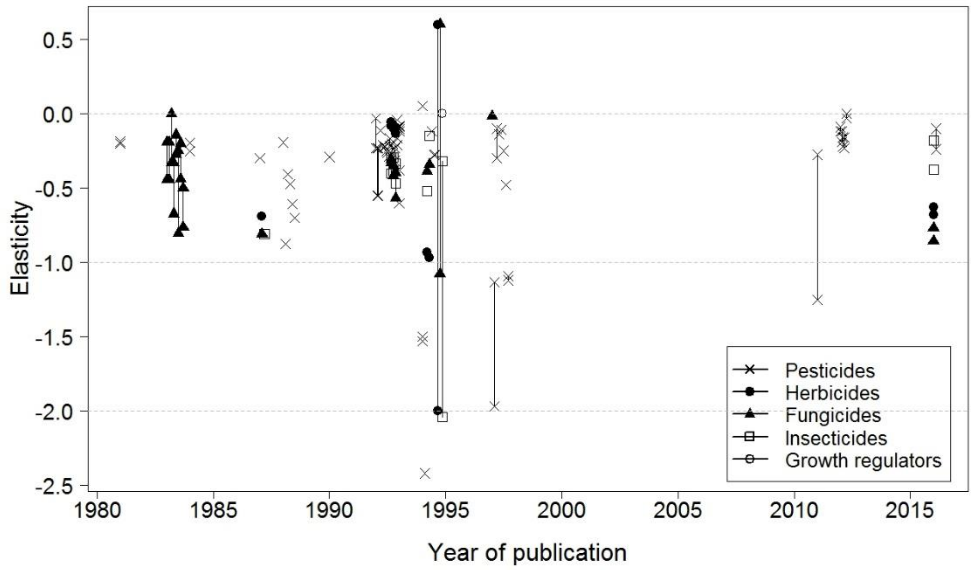

Fig. 3.1. Pesticide's own-price elasticities of demand following the year of publication.

Note: The lines between two observations refer to an interval because of different elasticity values for different levels of price changes.

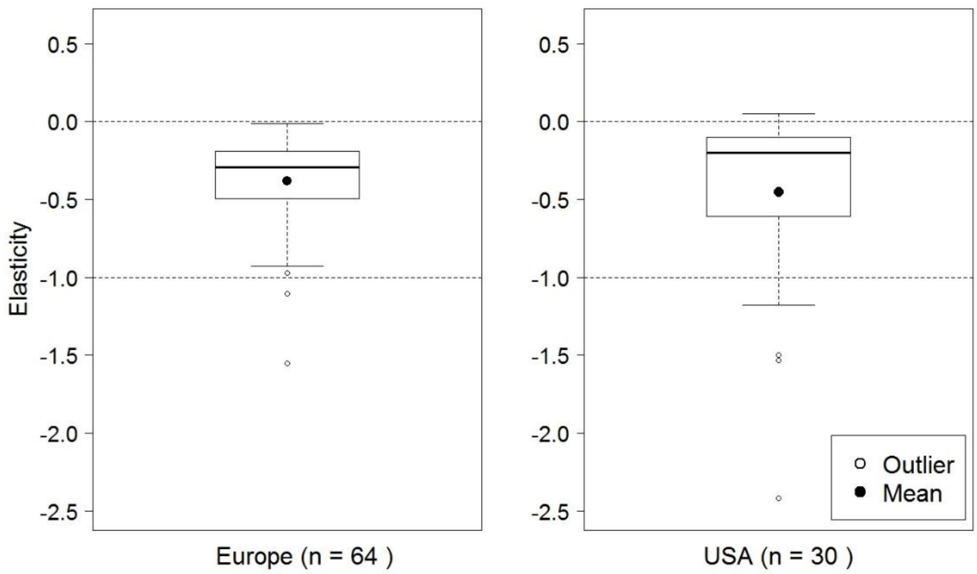

Fig. 3.2. Pesticide's own-price elasticities of demand for Europe and the USA.

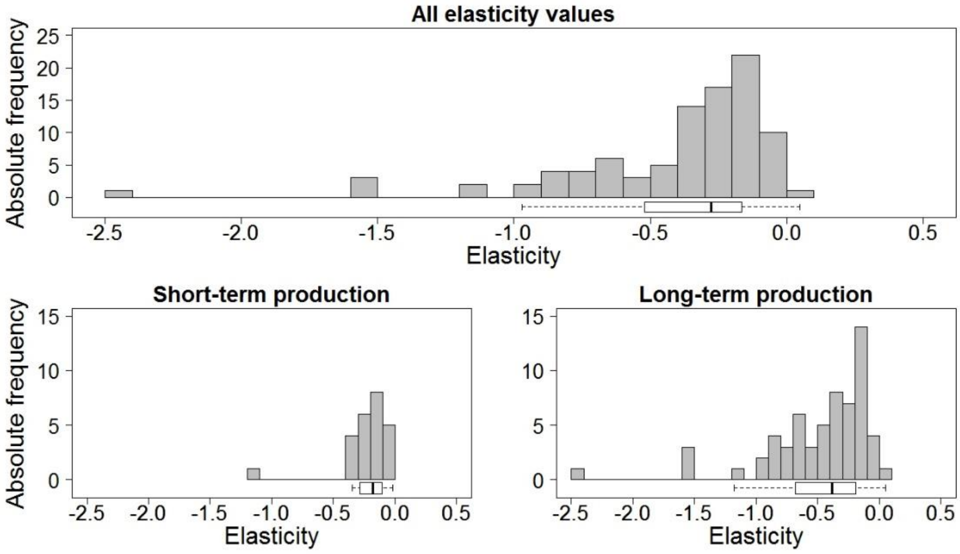

Fig. 3.3. Pesticide's own-price elasticities of demand arranged by flexibility of production.

Fig. 3.4 shows demand elasticity estimates by agricultural system investigated. We find that the demand for pesticides in special crops is less elastic (median = -0.19) than for arable and grassland production (median = -0.30) and pesticide use in aggregate (*e.g.* across the entire farm, median = -0.28). Comparing the demand elasticities in special crops with arable and grassland production, we reject the null-hypothesis of equal demand elasticities at the 5% level of significance. Moreover, a significant difference can also be found between the aggregate use and special crops (5% level), but not between aggregate use and arable production.

Concerning the analysis of the methodologies underlying pesticide demand elasticity estimation, we find 38 observations for econometric approaches (median = -0.30) and 39 observations for normative approaches (median = -0.21). There is a tendency towards lower elasticities in normative approaches with a significance level of 5%. Regarding the publication channel, we find that demand elasticities reported in peer-reviewed studies indicate on average a less elastic pesticide demand (33 observations, median = -0.21) compared to non-peer-reviewed publications (61 observations, median = -0.29). However, this difference is not significant using a Wilcoxon–Mann–Whitney test.

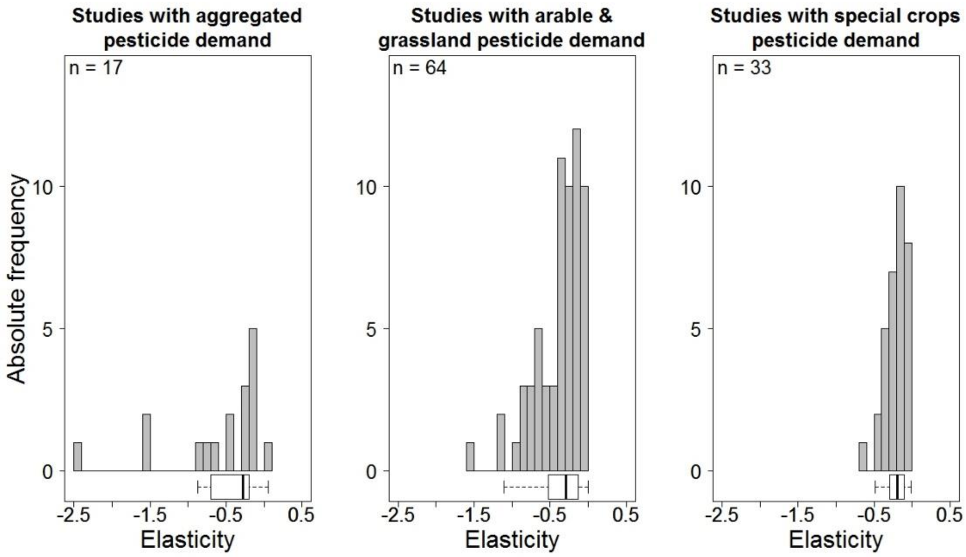

Fig. 3.4. Subject of analysis in studies estimating pesticide's own-price elasticities of demand.

Finally, we conducted the regression analysis including all of these factors. We find that the explanatory variables 'middle year of the analysis period' and 'peer-review' are not independent of each other. More specifically, peer-reviewed studies are more often observed to be more recent studies with more recent middle years of the analysis. Thus, we present both models with either the middle year of analysis or a dummy for peer-review being included. In addition, the consideration of the methodology used (dummy Econometric analysis) results in several missing observations. As sensitivity analysis, we therefore estimate our models with and without the consideration of this variable. Coefficient estimates of the four models are presented in Table 3.2. We find pesticide demand to be less elastic when published in peer-reviewed journals. Along these lines, we also find evidence that studies analysing more recent data obtain significantly inelastic demand. Note that the intercepts in Model 1 and 3 refer to the year 1900. Moreover, studies considering a long-term horizon result in more elastic demand elasticity estimates. We find the demand for pesticides in special crops to be less elastic compared to other agricultural sectors, and demand for herbicides is more elastic than for other pesticides. European studies tend to result in more elastic demand estimates, though significance tests are ambiguous. Even though econometric studies tend to result in less elastic demand elasticity estimates, we do not find a significant effect of the methodology used.[12]

Table 3.2. Parameter estimates of the regression models (marginal values)

Parameter	Model 1	Model 2	Model 3	Model 4
β_0 (intercept)	–0.627 (–11.016)***	–0.216 (-5.068)***	–0.420 (–2.381)**	–0.201 (–4.819)***
β_1 (middle year of analysis)	0.007 (12.241)***	–	0.004 (1.738)*	–
β_2 (long-term)	–0.173 (-5.610)***	0.212 (–7.232)***	–0.177 (–5.198)***	–0.215 (–7.743)***
β_3 (special crops)	0.106 (2.279)**	0.119 (3.316)***	0.086 (1.789)*	0.108 (4.022)***
β_4 (herbicides)	–0.416 (–7.555)***	–0.406 (–5.906)***	-0.262 (-1.955)*	–0.308 (–3.179)***
β_5 (Europe)	–0.207 (–4.854)***	–0.044 (–1.146) (n.s.)	–0.200 (–2.693)***	–0.058 (–1.495) (n.s.)
β_6 (peer-review)	–	0.184 (6.908)***	–	0.194 (5.029)***
β_7 (econometric method)	0.042 (1.587) (n.s.)	0.030 (0.979) (n.s.)	–	–
Degrees of freedom	60	60	74	74

Notes: Values in parentheses are z-values. n.s. denotes not significant. *, ** and *** denote significance at the 10%, 5% and 1%, level, respectively. The package 'MASS' was used for MM-estimation. The package 'multiwayvcov' was used for the derivation of the cluster-adjusted standard errors. The R-code for the regressions can be found in Appendix 3.D.

Note that the few studies that report the uncertainty attached to pesticide demand elasticity estimates (*e.g.* as standard deviation) often report

[12] As there might be a bias towards econometric analyses among peer-reviewed studies, we estimated the model without the dummy for either peer-review or econometric studies, but find no change in coefficient estimates or levels of significance.

large levels of variability of these estimates (see Appendix 3.C). Besides the considerable degree of across study-heterogeneity, there is also large within-study heterogeneity. Thus, generalising from point estimates needs careful reflection and highlights the importance of analysing different pesticide types and products or different areas of application.

3.4 Discussion and Conclusions

Our meta-analysis reveals that own-price elasticities of demand for pesticides are, with a median of -0.28, significantly smaller than zero, but also to be significantly larger than -1, *i.e*. are inelastic. Moreover, demand elasticity estimates are found to be highly heterogeneous across different studies, ranging from complete inelasticity to relatively high elasticity, reflecting the fact that various parameters affect pesticide application. Other developments, such as the development of new pesticides or the occurrence of new pests might even be more important than price developments. For example, Vijftigschild and Oskam (1994) show for the Netherlands that between 1980/81 and 1991/92, despite pesticide prices increased up to 32.5%, the pesticide demand increased by 20%, which would indicate a simple and crude elasticity of +0.63. In a similar vein, pesticides taxes have not necessarily led to sharp decreases in pesticide use, as observed in Sweden and Denmark under the former tax scheme, but, rather, to a substitution of products, as observed in Norway (Böcker and Finger, 2016).

Nevertheless, our results show that demand elasticities are significantly smaller than zero so that, *ceteris paribus*, there will be a quantity reduction if pesticide prices are increased. This indicates that taxation of pesticides will have an effect of reducing pesticide use. However, the finding of inelastic demand for pesticides also indicates that large levels of taxation are required to realise substantial quantity reductions. The meta-analysis also identifies specific agricultural production environments, where the elasticity of the demand for pesticides is almost zero (*e.g*. Skevas *et al*., 2012), and where the effectiveness of taxation will be limited. This finding highlights the importance of country-specific ex-ante analyses of policy measures. Our meta-analysis reveals a large heterogeneity of the magnitude of these effects. More specifically, we find that price elasticities of demand are more inelastic in special crops (horticulture, fruit production and viticulture) compared to arable and grassland farming. This is especially relevant since special crops usually need the largest amount of pesticides per hectare – particularly more fungicide and insecticide but less herbicide. The lower elasticity can be explained due to the fact that (i) fruit production and viticulture are spatially less flexible in production than other crops, (ii) the harvest is more valuable and quality aspects are more important (*e.g.* Weston and Barth, 1997), (iii) pesticides are applied more often preventively, and (iv) fewer substitutes exist for those sectors. This finding also highlights that a pesticide tax could imply non-uniform income effects on farms, *e.g.* comparing farmers growing special crops and arable farmers. Our results show that herbicide demand is more elastic than fungicides and insecticides. Thus, taxation might be particularly suited to reducing herbicide use. Next to the heterogeneity across crops and pesticide type, also a considerable heterogeneity of demand elasticities across time was revealed in our analysis. In the short-run, the median demand elasticities are significantly less elastic than in the long-run, which is in line with Le Chatelier's principle generalised by Samuelson (1983). We find that estimated demand elasticities significantly decreased over time (middle year of the analysis period) and with peer-review. Even though it is not possible in our analysis to identify causalities and determinants for this development, the finding is in line with qualitative observations made in Skevas *et al*. (2012) and Fadhuile *et al*. (2016). Skevas *et al*. (2012) argue that demand elasticities are lower due to increasing importance of pesticides in current agricultural production practices in comparison to earlier decades. Fadhuile *et al*. (2016) argue that stricter policies result in smaller numbers of available pesticides, *i.e*. a decreasing number of substitutes, which leads to more inelastic demand. Thus, in these situations, pesticide taxes may have a limited effectiveness. However, pesticide taxes may also promote changed behaviour towards more agronomical, biological or mechanical plant protection measures.

Despite the fact that policy-makers often aim to reduce the use of toxic pesticides and not necessarily the quantity of all aggregated pesticides, little information about demand differences with respect to the toxicity of the pesticides was found in our research: only Skevas *et al*. (2012) show that the demand elasticity is smaller for more toxic pesticides (-0.0003 vs. -0.03 for less toxic ones). Future research should focus on this aspect, because for policy implications, it motivates a taxation approach that accounts for the toxicity of different pesticide products, as implemented in Norway, Denmark and France.

Acknowledgements

The authors would like to thank the two anonymous referees and the Editor for constructive comments and suggestions on an earlier version of this paper. Moreover, the authors thank the Swiss Federal Office for Agriculture for funding and supporting this research.

3.5 Appendix

Additional Supporting Information may be found in the online version of this article: https://doi.org/10.1111/1477-9552.12198.

Appendix 3.A:

Additional notes to the selection of studies

More studies were found which simulate price increases on pesticides, but elasticities could not be derived, because the pre-tax costs are unknown (*e.g.* by a tax: Archer and Shogren (2001) simulate different tax systems). Dubberke and Schmitz (1993) report a price elasticity of -0.777, which however is not significant at the 10% level. Moreover, Zeddies *et al.* (1992) simulate for winter wheat in Germany that a relative price increase of 1.5 would lead to zero pesticide use. However, the results are criticised by themselves as not realistic in practice. During the literature analysis, also some studies outside the North America and Europe were found giving examples for pesticide elasticities (*e.g.* the analyses of Antle and Pingali (1994) and of Tjornhom *et al.*, 1998 for the Philippines). Those studies are not included.

In comparison to other studies, we have not included the following publications: We were not able to find any (original) information about price elasticities of demand in Lichtenberg *et al.* (1988), Pettersson *et al.* (1989) and Carpentier (1994) and thus excluded them from the analysis. The studies of Hazilla and Kopp from 1986, Johnsson from 1991, and ECOTEC from 1997 were not found in the literature search. The preliminary study of the Dutch DHV and LUW from 1991 was replaced by Oskam *et al.* (1992). The latter is included in the analysis.

Appendix 3.B:

Data collected from studies as CSV-file (Online)

Appendix 3.C:

Table with studies reporting a standard deviation of elasticity results

Table A3.1. Studies reporting a standard deviation of elasticity results

Author	Elasticity	SD	SD/Elasticity
Antle (1984)	–0.194	0.354	–1.82
Antle (1984)	–0.254	0.204	–0.80
McIntosh and Williams (1992)	–0.112	0.034	–0.30
Oskam *et al.* (1992)	–0.210	0.150	–0.71
Oskam *et al.* (1992)	–0.220	0.150	–0.68
Oskam *et al.* (1992)	–0.220	0.150	–0.68
Oskam *et al.* (1992)	–0.250	0.190	–0.76
Oskam *et al.* (1992)	–0.260	0.190	–0.73
Oskam *et al.* (1992)	–0.290	0.190	–0.66
Oude Lansink (1994)	–0.120	0.080	–0.67
Papanagiotou *et al.* (1994)	–0.279	0.163	–0.58
Papanagiotou *et al.* (1994)	–0.276	0.157	–0.57
Fadhuile *et al.* (2016)	–0.629	0.027	–0.04
Fadhuile *et al.* (2016)	–0.856	0.009	–0.01
Fadhuile *et al.* (2016)	–0.374	0.025	–0.07
Fadhuile *et al.* (2016)	–0.681	0.029	–0.04
Fadhuile *et al.* (2016)	–0.769	0.009	–0.01
Fadhuile *et al.* (2016)	–0.182	0.033	–0.18

Appendix 3.D:

R code for the regression analysis

```
# Load file, but adjust filename and header names before
elasticity <- read.csv("C:/filename.csv", header=T)

#--------------------------------------------------------------------------
# REGRESSION ANALYSIS
#--------------------------------------------------------------------------

attach(elasticity)

### Performing robust MM-regression:

# load package 'MASS'
library(MASS)
elasticity$ESY <- (elasticity$EndY + elasticity$StartY)/2
rfit <- rlm(Elas ~  ESY + Longterm + Special + Herbicides + EU + Journal +
Econometric, data=elasticity, method="MM")
summary(rfit)


### Estimate cluster-adjusted variance-covariance matrices to derive:

standard errors
# Load packages 'multiwayvcov' and 'lmtest'
library(multiwayvcov)
library(lmtest)
# rfit is the rlm element
# Author is the ID, which is used for clustering
m1.vcovCL<-cluster.vcov(rfit,Author)
# Apply new z tests
coeftest(rfit, m1.vcovCL)
```

3.6 References

Antle, J. M. (1984). The structure of US agricultural technology, 1910–78. *American Journal of Agricultural Economics* 66 (4): 414-421.

Antle, J. M. and Pingali, P. L. (1994). Pesticides, Productivity, and Farmer Health: A Philippine Case Study. *American Journal of Agricultural Economics* 76 (3): 418-430.

ARCADIS Belgium (ed.) (2014). Vergroening van de fiscaliteit. Bijlage 2: Groslijst vergroeningsopties. Version 4/2014, Working paper BE0114000073. Brussels: ARCADIS Belgium nv/sa.

Archer, D. W. and Shogren, J. F. (2001). Risk-indexed herbicide taxes to reduce ground and surface water pollution: an integrated ecological economics evaluation. *Ecological Economics* 38 (2): 227-250.

Bauer, S., Hoppe, U. and Hummelsheim, S. (1997). Decision support system for controlling pesticide use in Hessen. In Oskam, A. J. and Vijftigschild, R. A. N. (eds), *Proceedings and discussions, Workshop on pesticides, Wageningen, August 24th-27th, 1995*. Wageningen: Wageningen Agricultural University, 293-312.

Bergh, J. C. J. M. van den, Button, K. J., Nijkamp, P. and Pepping, G. C. (1997). Effectiveness of pesticide price policies in agriculture. In van den Bergh, J. C. J. M., Button, K. J., Nijkamp, P. and Pepping, G. C. (eds), *Meta-Analysis in Environmental Economics*. Dordrecht: Springer, 163-174.

Böcker, T. and Finger, R. (2016). European Pesticide Tax Schemes in Comparison: An Analysis of Experiences and Developments. *Sustainability* 8 (4/378): 1-22.

Brown, R. S. and Christensen, L. R. (1981). Estimating Elasticities of Substitution in a Model of Partial Static Equilibrium: An Application to U.S. Agriculture, 1947 to 1974. In Berndt, E. R. and Field, B. C. (eds), *Modeling and measuring natural resource substitution.* Cambridge, MA, USA and London: MIT Press, 209-229.

Capalbo, S. M. (1988). A Comparison of Econometric Models of U.S. Agricultural Productivity and Aggregate Technology. In Capalbo, S. M. and Antle, J. M. (eds), *Agricultural Productivity – Measurement and Explanation*. Wasching-ton, D.C.: Resources for the Future, 159-188.

Capalbo, S. M. and Vo, T. T. (1988). A Review of the Evidence on Agricultural Productivity and Aggregate Technology. In: Capalbo, S. M. and Antle, J. M. (eds), *Agricultural Productivity – Measurement and Explanation*. Washington, D.C.: Resources for the Future, 96-137.

Carpentier, A. (1994). A pesticide ban in the context of intensive cropping technology: the case of the French crop sector. In Michalek, J. and Hanf, C.-H. (eds), *The economic consequences of a drastic reduction in pesticide use in the EU, Revised papers of a workshop held in Tannenfelde (Schleswig-Holstein) November 13th-14th, 1993*. Kiel: Wissenschaftsverlag Vauk Kiel KG, 281-306.

Carpentier, A. and Weaver, R. D. (1997). Damage Control Productivity: Why Econometrics Matters. *American Journal of Agricultural Economics* 79 (1): 47-61.

Chambers, R. G. and Lichtenberg, E. (1994). Simple Econometrics of Pesticide Productivity. *American Journal of Agricultural Economics* 76 (3): 407-417.

Chen, P. C., McIntosh, C. S. and Epperson, J. E. (1994). The Effects of a Pesticide Tax on Agricultural Production and Profits. *Journal of Agribusiness* 12 (2): 125-138.

Dubberke, H. and Schmitz, P. M. (1993). Ökometrische Schätzung von Elastizitäten. In Schmitz, P. M. and Hartmann, M. (eds), *Landwirtschaft und Chemie.* Kiel: Wissenschaftsverlag Vauk Kiel KG, 169-189.

Dubgaard, A. (1987). Anvendelse af afgifter til regulering af pesticidforbruget. Rapport nr. 35, Copenhagen: Statens Jordbrugsøkonomiske Institut.

Dubgaard, A. (1991). Pesticide Regulation in Denmark. In Hanley, N. (ed.), *Farming and the Countryside: An Economic Analysis of External Costs and Benefits*. Wallingford, UK: C·A·B International, 48-58.

Elhorst, J. P. (1990). De inkomensvorming en de inkomensverdeling in de Nederlandse landbouw verklaard vanuit de huishoudproduktietheorie. Dissertation. The Hague: Landbouw-Economisch Instituut.

Fadhuile, A., Lemarié, S. and Pirotte, A. (2016). Disaggregating the Demand for Pesticides: Does it Matter? *Canadian Journal of Agricultural Economics* 64 (2): 223-252.

Falconer, K. E. (1998). Managing diffuse environmental contamination from agricultural pesticides: an economic perspective on issues and policy options, with particular reference to Europe. *Agriculture, Ecosystems and Environment* 69 (1): 37-54.

Falconer, K. E. and Hodge, I. (2000). Using economic incentives for pesticide usage reductions: responsiveness to input taxation and agricultural systems. *Agricultural systems* 63 (3): 175-194.

Femenia, F. and Letort, E. (2016). How to significantly reduce pesticide use: An empirical evaluation of the impacts of pesticide taxation associated with a change in cropping practice. *Ecological Economics* 125: 27-37.

Fernandez-Cornejo, J. (1993). Demand and Substitution of Agricultural Inputs in the Central Corn Belt States. Technical Bulletin No. 1816. Washington, D.C.: U. S. Department of Agriculture – Economic Research Service.

Fernandez-Cornejo, J., Sharon, J. and Smith, M. (1998). Issues in the economics of pesticide use in agriculture: a review of the empirical evidence. *Review of Agricultural Economics* 20 (2): 462-488.

Finger, R. (2010). Revisiting the Evaluation of Robust Regression Techniques for Crop Yield Data Detrending. *American Journal of Agricultural Economics* 92 (1): 205-211.

Filippini, M. (2011). Short-and long-run time-of-use price elasticities in Swiss residential electricity demand. *Energy Policy* 39 (10): 5811-5817.

Gren, I.-M. (1994a). Cost Efficient Pesticide Reductions: A study of Sweden. *Environmental and Resource Economics* 4: 279-293.

Gren, I.-M. (1994b). Regulating the Farmers' use of Pesticides in Sweden. In Opschoor, H. and Turner, K. (eds), *Economic Incentives and Environmental Policies – Principles and Practice*. Dordrecht: Kluwer Academic Publishers, 153-173.

Hoevenagel, R., van Noort, E. and de Kok, R. (1999). Study on a European Union wide regulatory framework for levies on pesticides. Zoetermeer: EIM Small Business Research and Consultancy/Haskoning.

Hof, B., Koopmans, C., Rougoor, W. and van der Voort, J. (2013). Effecten en vormgeving van een heffing op gewasbeschermingsmiddelen. Rapport nr. 2013-39. Amsterdam: seo economisch onderzoek.

Ivanova, N., Stoyanova, Z. and Mishev, P. (2012). Estimation of Optimal Tax Level on Pesticides Use and its Impact on Agriculture. *Bulgarian Journal of Agricultural Science* 18 (5): 641-650.

Jacquet, F., Butault, J.-P. and Guichard, L. (2011). An economic analysis of the possibility of reducing pesticides in French field crops. *Ecological Economics* 70 (9): 1638-1648.

Komen, M. H. C., Oskam, A. J. and Peerlings, J. (1997). Effects of reduced pesticide application for the Dutch economy. In Oskam, A. J. and Vijftigschild, R. A. N. (eds), *Proceedings and discussions, Workshop on pesticides, Wageningen, August 24th-27th, 1995*. Wageningen: Wageningen Agricultural University, 344-359.

Lefebvre, M., Langrell, S. R. H. and Gomez-y-Paloma, S. (2015). Incentives and policies for integrated pest management in Europe: a review. *Agronomy for Sustainable Development* 35 (1): 27-45.

Lichtenberg, E., Parker, D. D. and Zilberman, D. (1988). Marginal Analysis of Welfare Costs of Environmental Policies: The Case of Pesticide Regulation. *American Journal of Agricultural Economics* 70 (4): 867-874.

McIntosh, C. S. and Williams A. A. (1992). Multiproduct production choices and pesticide regulation in Georgia. *Southern Journal of Agricultural Economics* 24 (1): 135-144.

Möckel, S., Gawel, E., Bretschneider W. Kästner, M., Liess, M. and Knillmann, S. (2015a). Eine Abgabe auf Pflanzenschutzmittel für Deutschland. *Natur und Recht* 37 (10) 669-677.

Möckel, S., Gawel, E., Kästner, M., Liess, M., Knillmann, S. and Bretschneider W. (2015b). Einführung einer Abgabe auf Pflanzenschutzmittel in Deutschland. In Gawel, E. (ed.), *Studien zu Umweltökonomie und Umweltpolitik*. Vol. 10. Berlin: Duncker & Humblot, 1-305.

Muñoz Piña, C. and Avila Forcada, S. (2004). Effects of an environmental tax on pesticides in Mexico. *Industry and Environment* 27 (2), 33-36.

Oskam, A. J., van Zeijts, H., Thijssen, G. J., Wossink, G. A. A. and Vijftigschild, R. (1992). *Pesticide use and pesticide policy in the Netherlands*. Wageningen: Wageningen Agriculture University.

Oude Lansink, A. G. J. M. (1994). Effects of input quotas in Dutch arable farming. *Tijdschrift voor sociaalwetenschappelijk oderzoek van de Landbouw* 9 (3): 197-217.

Oude Lansink, A. G. J. M. and Peerlings, J. H. M. (1997). Farm-specific impacts of policy changes on pesticide use in Dutch arable farming. In Oskam, A. J. and Vijftigschild, R. A. N. (eds), *Proceedings and discussions, Workshop on pesticides, Wageningen, August 24th-27th, 1995*. Wageningen: Wageningen Agricultural University, 414-428.

Papanagiotou, E., Alexandridis, K. and Melfou, K. (1994). Some implications of pesticide ban policies in Greece. In Michalek, J. and Hanf, C.-H. (eds), *The economic consequences of a drastic reduction in pesticide use in the EU, Revised papers of a workshop held in Tannenfelde (Schleswig-Holstein) November 13th-14th, 1993*. Kiel: Wissenschaftsverlag Vauk Kiel KG, 307-317.

Pettersson, O., Pettersson, J., Johansson, V., Fogelfors, H., Sigvald, R. and Johnsson, B. (1989). Minskad kemisk bekämpning i jordbruket: möjligheter och konsekvenser. *Journal of the Royal Swedish Academy of Agriculture and Forestry* 128 (4-5): 379-396.

R Core Team (ed.) (2015). R: A language and environment for statistical computing. R Foundation for Statistical Computing. R Core Team: Vienna. Online: https://www.R-project.org/.

Rude, S. (1992). Pesticidforbrugets udvikling – landbrugs- og miljøpolitiske scenarier. Rapport nr. 68. Copenhagen: Statens Jordbrugsøkonomiske Institut.

Russel, N. P., Smith, V. H. and Goodwin, B. K. (1997). The Effects of CAP Reform on the Demand for Crop Protection in the U.K. In Oskam, A. J. and

Vijftigschild, R. A. N. (eds), *Proceedings and discussions, Workshop on pesticides, Wageningen, August 24th-27th, 1995*. Wageningen: Wageningen Agricultural University, 397-413.

Samuelson, P. A. (1983). *Foundations of Economic Analysis*. Enlarged Ed. Cambridge, MA, and London: Harvard University Press.

Schulte, J. (1983). Der Einfluß eines begrenzten Handelsdünger- und Pflanzenbehandlungsmitteleinsatzes auf Betriebsorganisation und Einkommen verschiedener Betriebssysteme. Dissertation. Bonn: Rheinische Friedrich-Wilhelms-Universität Bonn – Faculty of Agriculture.

Shumway, C. R. and Chesser, R. R. (1994). Pesticide Tax, Cropping Patterns, and Water Quality in South Central Texas. *Journal of Agricultural and Applied Economics* 26 (1): 224-240.

Skevas, T., Oude Lansink, A. G. J. M. and Stefanou, S. E. (2013). Designing the emerging EU pesticide policy: A literature review. *NJAS – Wageningen Journal of Life Sciences* 64-65: 95-103.

Skevas, T., Stefanou, S. E. and Oude Lansink, A. (2012). Can economic incentives encourage actual reductions in pesticide use and environmental spillovers? *Agricultural Economics* 43 (3): 267-276.

Tjornhom, J. D., Norton, G. W. and Gapud, V. (1998). Impacts of price and exchange rate policies on pesticide use in the Philippines. *Agricultural Economics* 18 (2): 167-175.

Vijftigschild, R. A. N. and Oskam, A. J. ‘Pesticide use in the Netherlands: extent, problems and policy. In Michalek, J. and Hanf, C.-H. (eds), *The economic consequences of a drastic reduction in pesticide use in the EU, Revised papers of a workshop held in Tannenfelde (Schleswig-Holstein) November 13th-14th, 1993*. Kiel: Wissenschaftsverlag Vauk Kiel KG, 135-171.

Villezca-Becerra, P. A. and Shumway, C. R. (1992). State-Level Output Supply and Input Demand Elasticities for Agricultural Commodities. *Journal of Agricultural Economics Research* 44 (1): 22-34.

Weston, L.A. and Barth, M.M. (1997). Preharvest Factors Affecting Postharvest Quality of Vegetables. *HortScience* 32 (5): 812-816.

Zeddies, J., Fuchs, C. and Schanzenbacher, B. (1992). Marktwirkungen von Öko-Steuern und EG-Agrarreform. In Agrarsoziale Gesellschaft e.V. (ed.), *Schriftenreihe für ländliche Sozialfragen: Öko-Steuern als Ausweg aus der Agrarkrise? Ergebnisse der internationalen Tagung vom 15. Bis 17. Juni 1992 in Stuttgart-Hohenheim*. Göttingen: Agrarsoziale Gesellschaft e.V., 137-149.

Chapter 4

Modelling the Effects of a Glyphosate Ban on Weed Management in Silage Maize Production[13]

Abstract

A bio-economic model is developed that allows a detailed representation of optimal weed control decisions. It implements an output damage control approach for German silage maize production, considering almost eighty mechanical and herbicide based weed control options against over thirty weeds, working with detailed data on weed abundance and yields for more than three hundred municipalities in the federal state of North-Rhine-Westphalia. We apply the model to simulate economic optimal weed control over two growing periods under current environmental standards and under the scenario of a glyphosate ban as recently discussed after glyphosate was classified as probably carcinogenic to humans. Considering different levels of weed pressure, we find that adjustments in the intensity of mechanical pre-sowing strategies are an optimal response to a glyphosate ban, causing yield reductions of about 1%. In contrast, we find little evidence for a substitution towards selective herbicides post-sowing. On average, the aggregated economic impacts of a glyphosate ban are small, *i.e.* at about € 1–2/ha, but single farms may face higher losses at about € 10/ha.

Keywords: Output damage control; pest management; herbicide; maize; glyphosate; Germany.

---

[13] This chapter was published in the journal *Ecological Economics* as:
Böcker, T., Britz, W. and Finger, R. (2018). Modelling the Effects of a Glyphosate Ban on Weed Management in Silage Maize Production. *Ecological Economics* 145: 182-193.

4.1 Introduction

Reducing risks caused by pesticide application is a crucial component of current agri-environmental policy debates in Europe. Different measures are proposed to control pesticide use and the connected risks for the environment and human health, resulting in more sustainable agricultural systems (Lefebvre *et al*., 2015). The proposed measures comprise banning specific pesticides (*e.g.* neonicotinoids and glyphosate; Gross, 2013; Schulte and Theuvsen, 2015) or introducing pesticide taxes (Böcker and Finger, 2016; Finger *et al*., 2017). Especially the renewed licensing or banning of the broad-spectrum herbicide glyphosate in the EU provoked heated discussions after the International Agency for Research on Cancer classified glyphosate as “probably carcinogenic to humans” (Guyton *et al*., 2015). Ex-ante information on health and environmental risks reduction and on the impacts on farmer's income is needed to inform the debate on policy measures targeting pesticides (Falconer, 1998). As substitution effects with other herbicides are likely if specific products are targeted, potential changes in farm management must be depicted in detail. In the debate on banning glyphosate, however, there is a large uncertainty about those effects (Schulte and Theuvsen, 2015; see also the position paper of Steinmann *et al*., 2016). In this paper, we develop a tool for such detailed impact assessment of environmental standards or other policy measures affecting specific pesticides and apply it to assess a potential ban of glyphosate.

In available assessments on pesticide application behaviour of farmers, mainly econometric and optimisation modelling approaches or combinations of both are applied (see Böcker and Finger, 2017). Econometric applications are usually based on historical data, for instance of pesticide applications, and are used to explain historical developments or to make recommendations on decision making. Optimisation and simulation models presume, for example, optimal decision making based on more or less detailed production function approaches combined with an economic objective such as profit maximisation. They can hence be used for what-if-analyses even if observations are missing (Grovermann *et al*., 2017). Existing approaches of the latter group are, however, not detailed enough to assess measures addressing individual pesticides, such as glyphosate in our application. For example, Guan *et al*. (2005) work with a monetary aggregate over fungicides, herbicides and other pesticides; but, higher total costs for pesticide applications do not necessarily lead to a better weed treatment and vice versa. Babcock *et al*. (1992) and Kuosmanen *et al*. (2006) use the total amount of active substances (AS) of fungicides respectively insecticides as an indicator for pesticide use in apple production respectively cotton, neglecting any differences in risk between different AS. Karagiannis and Tzouvelekas (2012) measure insecticide application in olive orchards based on litres of insecticides, and Jacquet *et al*. (2011) model five different alternatives (intensive, recommended by extension services, 2× integrated practices and organic farming practices), both ignoring the diversity of existing AS.

In this paper, we extend the literature studying policy effects on pest management by i) making use of the output damage function approach (*e.g*. Karagiannis and Tzouvelekas, 2012), and ii) differentiating in detail a larger set of pre-sowing and post-sowing weed control options with regard to their yield impact. Specifically, we consider for each strategy both costs and efficacy of controlling individual weeds. Moreover, we develop a framework that is site-specific and allows investigating weed management over time and space. Our empirical analysis focusses on silage maize, one of the most relevant crops in Germany, where pest management mainly relies on herbicide application (Julius Kühn-Institut, 2016). We apply the model to the Federal State of North-Rhine-Westphalia (NRW), Germany, and account for the spatial heterogeneity of weed pressure and yield potential at municipality level. The model identifies economically optimal herbicide strategies in silage maize in each municipality at given pesticide and crop prices as well as specifications and regulations of pesticide use. We apply this model to study the impact of a ban of glyphosate on herbicide use and/or mechanical weed control measures and related costs compared to the current situation. At the moment, there are no alternative chemical herbicides approved to replace glyphosate for pre-sowing application (Kehlenbeck *et al*., 20151). Thus, mechanical weed control is the only alternative which removes all potential risks from herbicides before sowing. However, as claimed in some discussions on the topic, selective herbicides could potentially be used at higher rates after sowing, even increasing the overall health and environmental risks.

The remainder of this article is structured as follows: section 2 presents the damage control approach, the production function and its parameterisation. The data used in the model is depicted in section 3. The following section presents results, starting with some descriptive results before testing several hypotheses. Afterwards, both the model and the results are discussed and, finally, conclusions drawn.

4.2 Methodology

We develop a bio-economic weed control model for silage maize in *m* regions, *i.e*. 377 silage maize producing municipalities in NRW. A two-year cropping period is considered where maize is grown in each of the two years *t*, a standard farming practise. The expected gross margin $E(\pi)$ in year *t* for different pre- (index *b*) and post-sowing (index *h*) weed control strategies is defined as:

$$\mathrm{E}(\pi_{m,t,b,h}) = [y^*_{m,t,b,h} \cdot \mathrm{E(P)} - c(b) - c_s(b) - c(h) - c_f(y) - c_o], \tag{4.1}$$

where $y^*_{m,t,b,h}$ is the expected yield, $E(P)$ is the expected price for maize, $c(b)$ and $c(h)$ are the pre- and post-sowing weed management (and tillage) costs for a certain strategy and $c_s(b)$ are variable costs for sowing depending on the pre-sowing strategy (the more expensive direct precision drill is needed for some types of conservation tillage). $c_f(y)$ are costs for fertiliser depending on the yield and c_o are other costs (*e.g*. proportionate costs for rating and liming). Harvest costs are not included because maize is sold ex field such that the buyer performs the harvest, which is also reflected in lower output prices.

4.2.1 The Damage Control Approach

An output damage function is used to determine the expected yield y^* (Fox and Weersink, 1995; Guan *et al.*, 2006, 2005; Hall and Norgaard, 1973; Oude Lansink and Carpentier, 2001; Pannell, 1990; Talpaz and Borosh, 1974). It depicts first the effect of the damage control input(s) on the population of the damaging organism and from there the resulting yield reduction from surviving damaging organisms (Karagiannis and Tzouvelekas, 2012). We follow Guan *et al.* (2005) and distinct in the production function $y=G(x,D(h))$ between productive (x) and damage-controlling inputs (h) where $D(h)$ is the multiplicative damage controlling effect on the interval [0,1]. h is, for example, the efficacy of a herbicide against a specific weed. If $D(h)$ is equal to unity, no losses due to pests, diseases or weeds occur. Besides chemical inputs, also mechanical inputs such as hoeing or ploughing can be considered as damage-controlling, which somewhat challenges a clear distinction between h and x. Different proposals regarding the functional form of $D(h)$ have been made (see *e.g.* Carrasco-Tauber and Moffitt, 1992; Fox and Weersink, 1995; Kuosmanen *et al.*, 2006; Lichtenberg and Zilberman, 1986). We follow Guan *et al.* (2005) and use the exponential form because it is particularly suited to represent the underlying biological processes:

$$D(h) = 1 - e^{-(\beta_0+\beta_1\cdot z(h))^2}, \quad \beta_0, \beta_1 \geq 0. \tag{4.2}$$

This functional form implies decreasing marginal damage control in input use, a reasonable assumption as, *e.g.*, additional efforts in weed control on an almost weed free field will not lead to much higher damage control. Parameters β_0 and β_1 quantify the effects of inputs on damage control; their estimation is explained in the next sections. The decision variable in our model is *z(h)*, the chosen level of damage control.

4.2.2 Specification of the Damage Controlling Effect

We consider the 32 most important weeds for the case study region in our analysis (see Table 4.1 in section 4.3). Each plant protection strategy is characterised by its weed specific damage control effect, *i.e.* a column vector h with j 1×32 entries ranging between 0 and 1, allowing to represent how specific herbicides and mechanical strategies differ in their impact on individual weeds. Often, an herbicide strategy comprises several herbicide products. The resulting control success is typically not additive since the comprised herbicides usually have a similar spectrum of action. More likely is the case that the maximum suppression effect of any herbicide is crucial for the success. Also, we add a multiplier a_i to each weed $w_{m,i}$ to differentiate yield depression effects by weed, depicted by the average abundance (a_i) which measures the affected area share when that weed occurs (Table 4.1). Finally, in order to quantify the site-specific damage controlling effect of specific herbicides, a weed-row vector w with size i 32×1 depicts for each municipality m the probability that a weed occurs. The three vectors – probability of weed occurrence w, affected share a, and damage control for each weed h – define jointly the control success z for each herbicide strategy j in the different municipalities m:

$$\mathrm{z_{m,j}} = \sum_{\mathrm{i}}^{32} \mathrm{w_{m,i}} \cdot \mathrm{a_i} \cdot \mathrm{h_{j,i}} \,. \tag{4.3}$$

Eq. (4.3) presents the post-sowing weed controlling effects. Since we use probabilities for the determination of the damage controlling effect, the equation is dimensionless. In a similar manner, a vector $v_{m,j}$ can be constructed that accounts for pre-sowing weed management effects (denoted as $b_{j,i}$):

$$\mathrm{v}_{\mathrm{m,j}} = \sum_{\mathrm{i}}^{32} \mathrm{w}_{\mathrm{m,i}} \cdot \mathrm{a}_{\mathrm{i}} \cdot \mathrm{b}_{\mathrm{j,i}} \,. \tag{4.4}$$

4.2.3 *Choice of Functional Form and Implementing the Damage Controlling Effect*

Inserting the damage control success expression from Eq. (4.3) in Eq. (4.2) yields the following specification:

$$\mathrm{D}_{\mathrm{m,j}} = 1 - \mathrm{e}^{-(\beta_0 + \beta_1 \cdot \Sigma_{\mathrm{i}}^{32} \mathrm{w}_{\mathrm{m,i}} \cdot \mathrm{a}_{\mathrm{i}} \cdot \mathrm{h}_{\mathrm{j,i}})^2}, \quad \beta_0, \beta_1 \geq 0. \tag{4.5}$$

One of the remaining issues is to determine the form of the production function. We follow Bosnić and Swanton (1997), Swinton *et al*. (1994), Swinton and King (1994) as well as Kropff *et al*. (1992), and use the rectangular hyperbolic approach of Cousens (1985) and Cousens *et al*. (1987), which allows accounting for biological effects such as time of emergence. Thus, the yield function in relation to weed control is defined as follows:

$$y^*_{m,t,b,h} = y^a_{m,t} \cdot \left[1 - I \cdot \frac{D_{m.j}}{100 \cdot \left(e^{C \cdot T} + I \cdot \frac{D_{m,j}}{A}\right)}\right]. \tag{4.6}$$

In this function, y_a is the attainable yield when no weeds are present, I is the percent yield loss as $D_{m,j}$ approaches 0 (*i.e*. $D_{m,j}$ is not yet 0)[14], A is the percent yield loss as $D_{m,j}$ approaches infinity, T is the time of crop emergence and growth in relation to the weed emergence and growth, measured in growing degree days, which is the sum of the average temperature of each day and C is the rate at which the yield loss I increases as T becomes smaller (*i.e*. weed pressure increases). Fungi and insects are of limited relevance in German maize production or can be controlled by seed dressing or resistant varieties such that except for herbicides usually no other pesticides

[14] Originally, Cousens (1985) used average plants per m². We will use the abundance of a single weed for our purpose, in relation to Table 4.1.

are applied (JKI, 2016). Thus, the attainable yield y^a is defined as the potential water-limited yield under given climatic and soil conditions, *i.e.* under no nutrient stress. The shape of the function is illustrated in Fig. 4.1. Nevertheless, using solely the yield term (4.6) neglects pre-sowing weed controlling practices depicted by $v_{m,j}$. Accounting for that, the expected yield y^* for a specific strategy becomes:

$$y^*_{m,t,b,h} = \underbrace{\left(1 - e^{-(\alpha_0+\alpha_1 \cdot v_{m,j})^2}\right)}_{\text{Pre-sowing}} \cdot y^a_{m,t} \cdot \underbrace{\left[1 - I \cdot \frac{e^{-(\beta_0+\beta_1 \cdot z_{m,j})^2}}{100 \cdot \left(e^{C \cdot T} + I \cdot \frac{e^{-(\beta_0+\beta_1 \cdot z_{m,j})^2}}{A}\right)}\right]}_{\text{Post-sowing}}. \qquad (4.7)$$

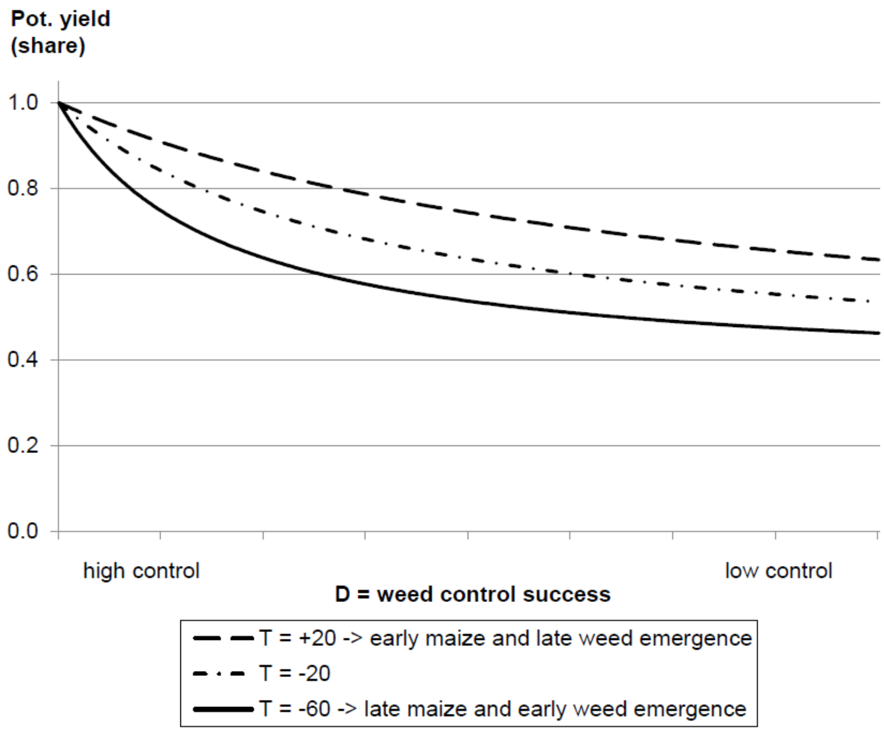

Fig. 4.1. Shape of the rectangular hyperbolic yield function from Eq. (4.6) for different levels of *T*.

4.2.4 Parameterisation and Pesticide Application Restrictions

In order to calibrate the model and to parameterise the production function, we conducted expert interviews with the senior herbicide consultant and three regional herbicide consultants of the chamber of agriculture from NRW who identified strategies in regions of NRW differentiated by soil types.[15] Furthermore, we collected data on the observed yield $\bar{y}_{m,t}$ in each municipality m which should reflect the current weed control practise (IT NRW, 2016). In order to estimate the parameters of interest (α_0, α_1, β_0 and β_1), we determine the parameter values which minimise the error term between the observed yields and the yields simulated with the most frequently used control strategies in selected municipalities where a clear assignment between expert knowledge on strategies used and weeds occurring could be made, *i.e.* municipalities which have homogeneous soil types but different potential yields:[16]

$$\min \varepsilon = \sum_{m}^{8} \left(y^{*}_{m,t^{*},b^{*},h^{*}} - \bar{y}_{m,t} \right)^2 \tag{4.8}$$

[15] Regarding herbicide strategies, three major soil types can be distinguished in NRW: sandy soils where herbicides against *Panicoideae*-varieties are applied, clayey soils where strategies against *Alopecurus myosuroides* are preferred and good loamy soils where simple and cheap strategies are used. Eight municipalities were selected where current weed control strategies are known (4x sandy soils due to the relevance in maize production, 2x good loamy soils and 2x clayey soils). We decided against the alternative way of estimating different parameters for different soil types because municipalities where a mix of soil types is observed accordingly also apply a mix of strategies. Additionally, data on yields for different soil types would be required which are not available. Applying different production functions per soil type would force us to decide for one specific function, although a mix of soil types is predominant.

[16] Another possibility is to use expert knowledge for the parameter estimation as done by Femenia and Letort (2016). In a study by Deen *et al.* (1993), the parameter value is assumed with respect to the efficacy of the herbicide.

As outlined in section 4.2.3, further restrictions are considered during the non-linear estimation process to identify the parameters. Once the parameters are identified and inserted into the production function, optimal strategies can be determined for each *m* and *t* according to Eq. (4.1), *i.e.* profits can be maximised for each municipality and year by choosing pre- ($S_{b,m,t}$) and post-sowing ($S_{h,m,t}$) shares for the control strategies:

$$E(\pi_{m,t}) = \sum_{b=1}^{24}\sum_{h=1}^{55} E(\pi_{m,t,b,h}) \cdot S_{b,m,t} \cdot \varphi_{b,glyphosate} \cdot S_{h,m,t}\,, \qquad S_{b,m,t} \text{ and } S_{h,m,t} \in [0,1] \tag{4.9}$$

$$\max\ \pi = \sum_{m=1}^{377}\sum_{t=1}^{2} E(\pi_{m,t}). \tag{4.10}$$

φ is the information matrix whether glyphosate is allowed in the analysed scenarios. $S_{b,m,t}$ and $S_{h,m,t}$ are the shares of the selected control strategies of the farmers for pre- (*b*) and post-sowing (*h*) weed management and $E(\pi_{m,t,b,h})$ is the profit for each strategy which reflects the expected yield, related fertiliser and other costs including the costs for weed control.

Some further details need to be reflected during estimation and simulation. Firstly, we assume that the strategy needs to be changed from year to year to avoid building up resistance against specific AS in the weed population. More specifically, we classified the strategies based on the Herbicide Resistance Action Committee (HRAC, 2005) into groups and added a constraint, which prevents that strategies from the same groups are used in two consecutive years (Eq. (4.11). Second, special requirements for nicosulfuron-containing strategies have to be included since this AS is only allowed to be applied every second year by law (code NG327 for the use of plant protection products; Eq. (4.12). Those restrictions can be implemented as:

$$\sum_{t=1}^{2}\left(\sum_{b=1}^{24}\left(S_{b,m,t} \cdot \eta_{b,HRAC}\right) + \sum_{h=1}^{55}\left(S_{h,m,t} \cdot \eta_{h,HRAC}\right)\right) \leq \frac{|t|}{2} \quad \forall\, m \wedge HRAC \tag{4.11}$$

$$\sum_{t=1}^{2} S_{h,m,t} \cdot \varphi_{h,nicosulfuron} \leq \frac{|t|}{2} \quad \forall\, m\,, \tag{4.12}$$

where φ and η are information matrices of whether post-sowing strategy j contains nicosulfuron and is classified as a certain *HRAC* [0,1].

The model is written in General Algebraic Modelling System (GAMS) code (Appendix 4.A). We simulate optimal herbicide strategies under a baseline where glyphosate can be applied throughout the two periods and a counterfactual where glyphosate is banned. We conduct sensitivity analyses with regard to the attainable yield (the attainable yield is increased by 10% in t_1 and 15% in t_2), the green maize price ex field P (€ 2.80/dt, € 3.30/dt and € 3.80/dt, dt = deciton) and the difference between weed and crop occurrence T (40 to -90), so that effects of higher or lower prices and higher or lower weed pressure can be seen. The latter is depicted by earlier or later maize emergence compared to weeds, *e.g.* T=0 means that maize and weeds emerge at the same time, T=-50 means that weeds have an advantage in emergence of, on average, five days with an average temperature of 10°. Of course, farmers try to keep the level of T large (*i.e.* close to zero or even positive) to increase the competitiveness of maize, and pre-sowing weed management is therefore done close to sowing. Based on model results derived across different municipalities and scenarios, we test the following five hypotheses: H1) average post-sowing strategies change in case of a glyphosate ban, H2) costs for weed management increase in case of a glyphosate ban, H3) working force demand increases in case of a glyphosate ban, H4) the gross margin decreases in case of a glyphosate ban, and H5) yields significantly decrease in case of a glyphosate ban.[17] To this end, t-tests and Wilcoxon-Mann-Whitney tests are used.

[17] For the hypotheses, we use the average of the periods t_1 and t_2.

4.3 Data

We focus on the most important weeds in maize cultivation for our case study region (defined as more than 10% degree of presence, following the samples of Mehrtens *et al*. (2005) and Mol *et al*. (2015). Additionally, *Digitaria ischaemum* and *Mercurialis annua* were included; weeds which are of importance in specific regions of NRW as they are also listed in the agricultural recommendations (see resulting list in Table 4.1). Information on the occurrence of weeds is taken from the 2.88x2.75km distribution raster of Germany's pteridophytes and flowering plants (NetPhyD and BfN, 2013; see also www.floraweb.de), and mapped via GIS operations to municipality areas. We included only the 377 municipalities of NRW which reported maize cultivation in recent years. Each municipality receives weed specific occurrence probabilities which reflect the area weighted average of raster cells where each weed was observed (see data for *Alopecurus myosuroides, Digitaria ischaemum, Setaria viridis* and *Chenopodium spp.* in Fig. 4.2). Information on the average abundance, *i.e*. the share of affected area when a weed is observed and not controlled, is used from long-term field trials (Table 4.1).

We consider those herbicides (combinations) that are recommended by the Chamber of Agriculture of North Rhine Westphalia (LWK NRW, 2015a) and the Bavarian State Research Centre for Agriculture (LfL, 2016). These recommendations are widely used in agricultural extension and also published in agricultural magazines. Because of lack of data on how different doses affect weed control, we use the recommended dose in each strategy instead of trying to also solve for an optimal rate (Pannell, 1990). However, these doses may vary between strategies comprising the same AS. In total, 55 different post-sowing herbicide strategies were defined, where one reflects zero control, 6 are mechanical only and the remaining 48 apply herbicides once or twice (see Appendix 4.B). For each of those 55 strategies, data by the LfL (2016) and the LWK NRW (2015a) define the suppressing efficiency against each of the 32 weeds in the interval [0,1]. A value of 1 characterises total eradication, a value of zero indicates no impact on the weed, and a value between zero and one was assigned if part of the population is removed. To cope with gaps in this database, we use manufacturer information (obtained from product brochures) to specify herbicide efficacy. Thereby, in general three categories are displayed: well or very well controllable, sufficiently controllable and not sufficiently controllable. For the first category, we assume an efficacy of 0.90, for the second category 0.33 and for the third category null efficacy.

Table 4.1. Maize grass-weeds and weeds implemented in the output damage function approach

Name	**Average abundance (%)**	**Name**	**Average abundance (%)**
Grass-weeds:		*Fumaria officinalis*	2.0
Alopecurus myosuroides	21.3[+]	*Galinsoga parviflora*	12.0
Digitaria ischaemum	21.3[+]	*Galium aparine*	7.0
Echinochloa crus-galli	22.0	*Geranium pusillum*	6.0
Elymus repens	21.3[+]	*Lamium spp.*	6.0
Poa annua, P. trivialis	2.0	*Matricaria spp.*	13.0
Setaria viridis	40.0	*Mercurialis annua*	6.8[+]
Broad-leaved weeds:		*Persicaria lapathifolia*	11.0
Amaranthus retroflexus	13.0	*Persicaria maculosa*	3.0
Atriplex patula	1.0	*Polygonum aviculare agg.*	3.0
Brassica napus	18.0	*Rumex obtusifolius*	4.0
Capsella bursa-pastoris	5.0	*Solanum nigrum*	3.0
Chenopodium spp.	20.0	*Sonchus spp.*	2.0
Cirsium arvense	4.0	*Stellaria media agg.*	6.0
Convolvulus arvensis	2.0	*Thlaspi arvense*	3.0
Equisetum arvense, E.	6.8[+]	*Veronica spp.*	2.0
Fallopia convolvulus	12.0	*Viola arvensis*	5.0

Note: Abundance-values marked with a [+] are estimates according to mean values of grass weeds or broad-leaved weeds. Data on year to year variation of the abundance were not found. *Brassica napus* was included for potential extension of the model by other crops (crop rotation). Source: Meinlschmidt *et al*. (2008).

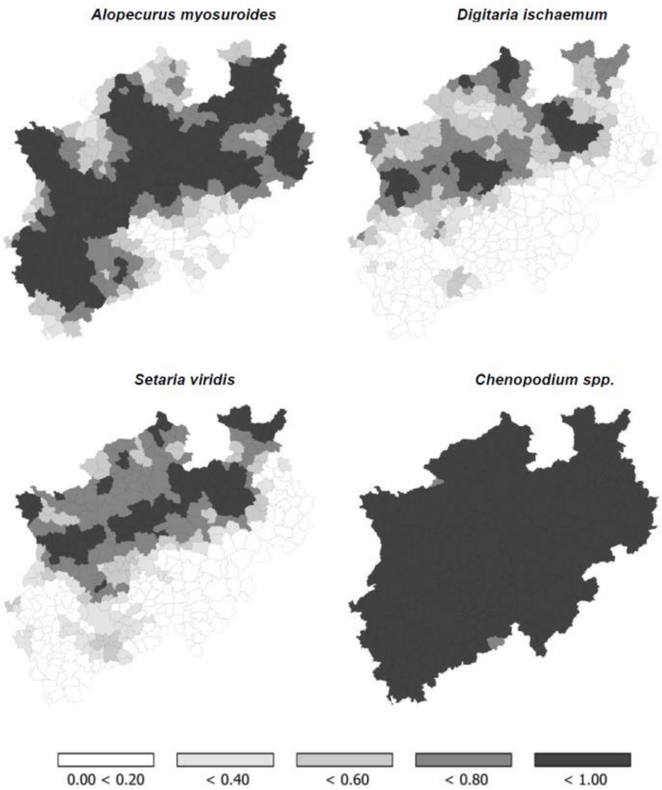

Fig. 4.2. Spread, measured as probability of occurrence, of four selected weeds in NRW.

Reference: NetPhyD and BfN (2013), raster data converted to municipality borders.

To quantify the efficacy of the mechanical strategies, we combine information from extensive or organic farming systems with expert knowledge (Jacquet *et al.*, 2011; Oskam, 1997). Data on mechanical post-sowing techniques could be found in Kees (1984, unpub., cit. from Hoffmann, 1990). Additionally, we consulted the organic farming expert of the Chamber of Agriculture from Lower Saxony for information on the mechanical harrowing and hoeing frequency, and their effect on specific weeds.[18] There are 24 different pre-sowing plant protection strategies in our

[18] On average, an effective mechanical weed management in maize is reached with three runs of harrowing and two runs of hoeing. If maize emerges early, fewer runs are necessary, and more if maize emerges late. In general, root weeds are more troublesome to control than other ones. However, blind harrowing can also be effective against root weeds.

model, consisting of mouldboard ploughing, different chisel ploughing and harrowing combinations and of glyphosate combinations.[19] Except for glyphosate, no other herbicides are allowed before sowing (Kehlenbeck *et al.*, 2015). We could not find unambiguous data about the yield increasing or decreasing effect of different tillage systems. Therefore, with respect to the weed controlling capacity of conventional and conservation tillage, both strategies have almost the same yield potential. Conventional tillage has only slight advantages in weed control.

Data about actual yields are available at county-level (53 counties in NRW; IT NRW, 2016), and $\bar{y}$ is the five year average of the actually observed yield from 2011 and 2015 (Fig. 4.3). A 5% increase of the expected yield is assumed for the second year t_2. Oerke (2006) estimated a 5% yield loss from weeds in Western European maize production with usual weed control strategies ($y^a = 1.05\ \bar{y}$). For information about maximum losses under zero control (scalar A in Eq. (4.7), we draw on field trials by Söchting and Zwerger (2012). Maize yields with herbicide treatment were up to 63.8% higher compared to the untreated control group (A = 63.8%). For I and C in Eq. (4.7), we rely on Bosnić and Swanton (1997), who estimated I = -0.3% and C = 0.017. Further restrictions of the estimation model are that the no-till pre-sowing strategy with no herbicide application has to achieve a yield level between 86% and 90% and that the ploughing strategy has to be larger than 95% (Gehring *et al.*, 2012). The zero control post-sowing strategy is fixed at 86% for normal weed emergence (in relation to the field trials of Söchting and Zwerger, 2012). Based on this data, the estimates from Eq. (4.8) are as follows: ε has a value of 0.8–4.0% of $E(y)$ depending on the municipality in the parameterisation. The best fit parameter values are $\alpha_0=1.304$, $\alpha_1=0.770$, $\beta_0=0.724$ and $\beta_1=0.244$ (estimated at T=0).

Herbicide's costs are based on 2015 recommended retail prices from a German agricultural trader (Roth Agrarhandel, 2015; see Appendix 4.B).

[19] As glyphosate containing product, we chose Roundup® PowerFlex.

For labour costs, € 17.5/h are assumed. In our study region, organic fertiliser is no limiting production factor (see Gömann *et al*., 2010 for details) so that we assume that slurry is for free. The most relevant cost parameters are presented in Table 4.2.

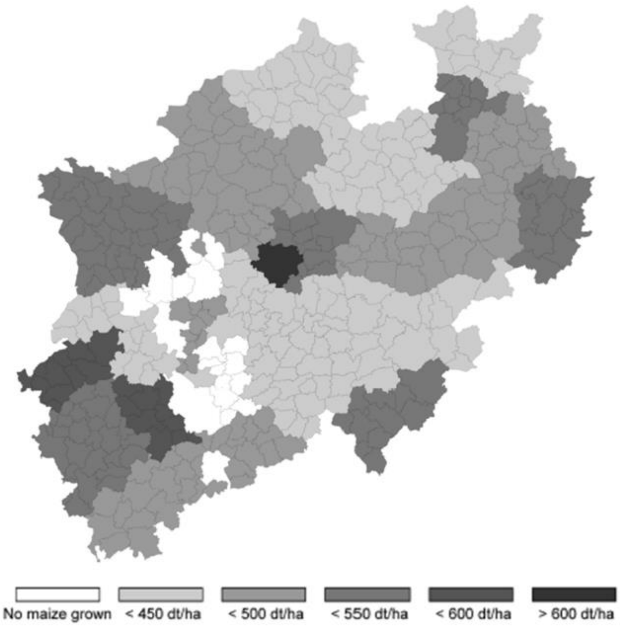

Fig. 4.3. Silage maize yields in different counties of NRW (five-year average of 2011–2015).

Reference: IT NRW (2016).

Table 4.2. Machinery costs and other inputs related to maize growing

Activity	**Sub-activity**	**Work hours (h/ha)**	**Fix and variable machinery costs**	**Other inputs**
		Weed control-related activities:		
Chisel plough/Cultivator (4.5m)		0.44	24.17	
Mouldboard plough and packer (1.4m)		1.73	66.97	
Pesticide sprayer (24m)		0.17	6.90	
Harrow (9m)		0.17	11.09	
Hoe (6m)		0.72	30.03	
		Other activities:		
Inspection (share, every 5th year)		0.04	0.26	-
Manure application (25 m³/ha)		0.74	50.23	-
	Manure	-	-	€ 0.00/ha
Precision drill (6m-		0.53	41.72	-
Direct precision drill (59% increase to normal precision drill, 20% discount on light soils)		0.53	66.31	
	Seed			€ 233.20/ha
Mounted fertiliser spreader (amount depends on *E(y)*)		0.00–0.29	0.00–6.14	
Liming (share, every 3rd year)		0.19	12.47	
	N			€ 1.10/kg
	P_2O_5			€ 0.87/kg
	K			€ 0.77/kg
	Ca			€ 0.05/kg
No harvest cost, sell ex field		-	-	-

Note: Diesel consumption for mouldboard ploughing is assumed to be 30% higher/lower on heavy/light soils (for chisel ploughing 20% higher/lower). References: Achilles *et al.* (2016), fertiliser prices from LfL (2016), weight-shares from LWK NRW (2015b).

4.4 Results

The results are presented in two sections. In the first section, the results from the different simulations and scenarios are shown and described. The second one addresses the hypotheses testing.[20]

[20] The results of the sensitivity analysis with regard to water limited potential yields are available as Appendix 4.C, the most important findings are discussed in the following where appropriate.

4.4.1 Descriptive Results

Fig. 4.4 presents for three price levels[21] the chosen pre-sowing strategies as a share of municipalities where they are applied, on average of the two years *t*. The application of glyphosate in a strategy is found to be on average optimal in about 5% to 25% of the municipalities (depending on prices and weed pressure expressed as *T*). In the other municipalities, conservation tillage with mechanical strategies consisting of one or two chisel ploughings and/or one to three harrowing passes is the most profitable. Glyphosate containing strategies are more profitable when applied closer to maize emergence, *i.e.* close before sowing or even close after sowing. The later maize emerges compared to weeds, the less glyphosate is applied. In case of a ban, the above mentioned mechanical strategies are used throughout, but mouldboard ploughing is not used in any year. As conservative mechanical control suppresses weeds not as effectively as non-selective herbicides, glyphosate use is higher in t_2 where the attainable yield is assumed higher. Only mechanical control is observed under a ban since no alternative herbicides are licensed for pre-sowing application.

[21] We use the price levels of € 2.80/dt, € 3.30/dt and € 3.80/dt. For the region of NRW, the price levels of € 3.30/dt or even € 3.80/dt are under current conditions more realistic than the lower price of € 2.80/dt.

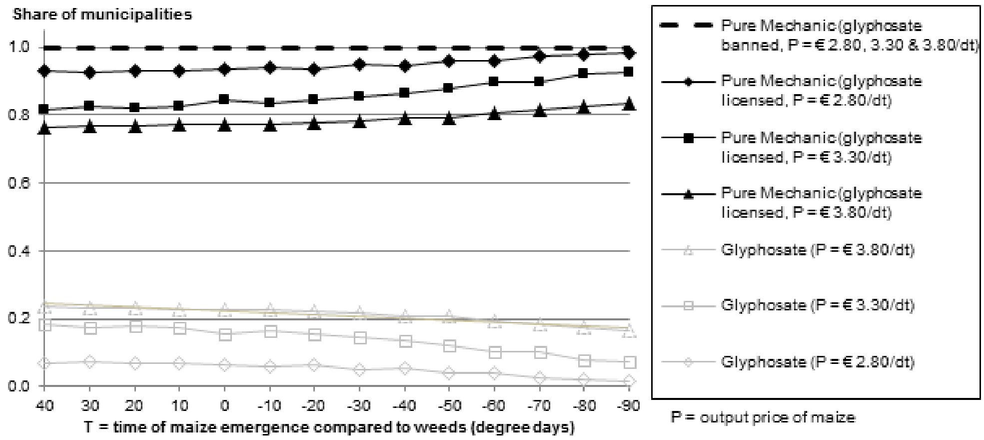

Fig. 4.4. Shares of used pre-sowing strategies (average of t_1 and t_2 of each municipality).

Note: A scenario in which glyphosate is licensed is compared to a scenario with a glyphosate ban. The figure shows the results for the three analysed price levels at different weed pressure levels (expressed as *T*).

Regarding selective herbicide use after sowing, we observe that with a later emergence of silage maize compared to weeds, *i.e*. a higher weed pressure reflected by a more negative *T*, more expensive herbicide strategies get more profitable. This implies that the share of mechanical strategies decreases (Fig. 4.5). Higher silage maize prices reinforce this. Comparing the change in *T* from +40 to -90, for example, implies an increase in weed control costs from € 78/ha up to € 115/ha at P=€ 3.80/dt, compared to an increase from € 66/ha to € 95/ha at P=€ 2.80/dt (Appendix 4.C). The composition of the chosen strategies as a function of *T*, *i.e.* maize relative to weed emergence, is summarised in Fig. 4.6 for the glyphosate licensed-scenario and an output price of € 3.80/dt. In both scenarios, *i.e*. for glyphosate being licensed and banned, the most profitable AS shift from nicosulfuron, prosulfuron and S-metolachlor to terbuthylazine, mesotrione, pethoxamid, flufenacet, foramsulfuron, iodosulfuron and thiencarbazone.

Assuming higher potential water limited yields let the model simulate higher shares of glyphosate usage (up to 21% at P = € 3.30/dt and 26% at P = € 3.80/dt) and also to faster changes in the use of the different AS (*i.e*. higher shares of terbuthylazine, flufenacet, foramsulfuron, iodosulfuron and thiencarbazone can be observed) (Appendix 4.C). Additionally, spending for weed management increase (averagely at a price of € 3.80/dt an increase from € 88/ha at T=-30 to € 122/ha at T=-90).

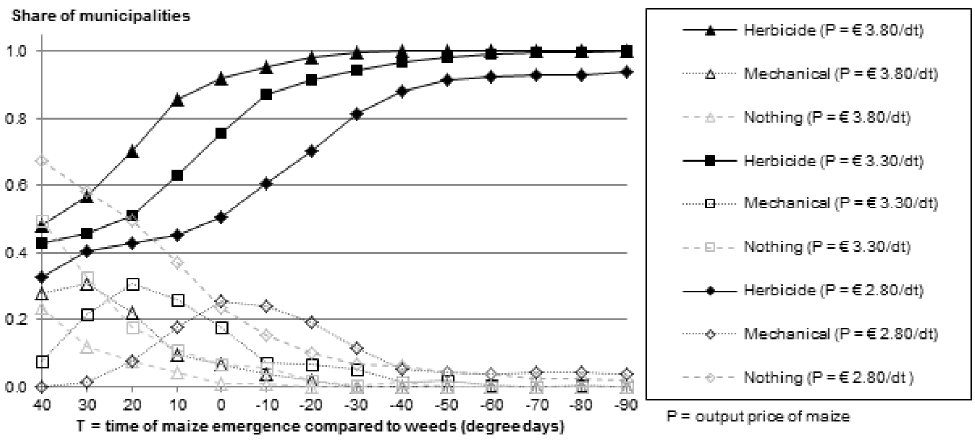

Fig. 4.5. Shares of post-sowing strategies as average of t_1 and t_2 (scenario with glyphosate licensed).

Note: The figure shows the categorisation of the post-sowing strategies in Herbicide, Mechanical and Nothing. Three price levels and different weed pressure levels (expressed as *T*) are analysed.

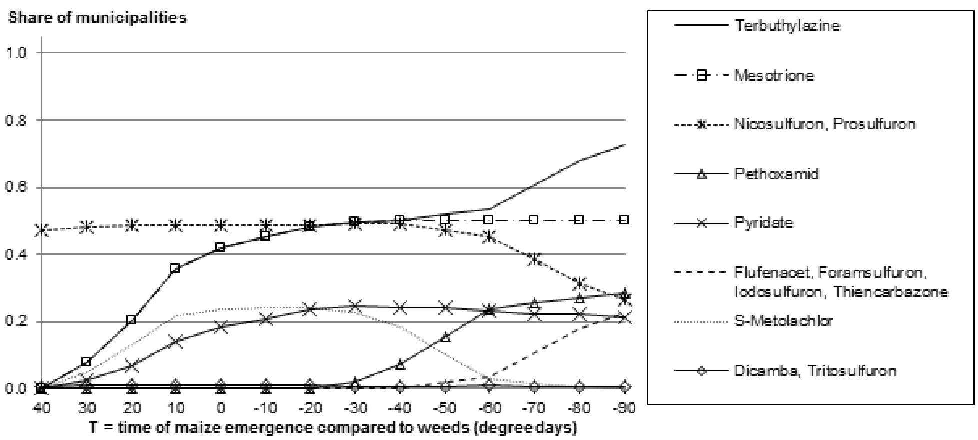

Fig. 4.6. Shares active substances in of post-sowing strategies (glyphosate licensed, P = € 3.80/dt, average of t_1 and t_2).

Note: The figure shows how often an active substance is used on average over the municipalities. For example, a level of 60% for terbuthylazine means that this substance is contained in the weed management strategy, on average of the two years, in 226 of the 377 municipalities each year.

4.4.2 Hypotheses Testing

Table 4.3 shows the results of the hypotheses testing for the price levels of € 2.80/dt and € 3.80/dt. Differences of mean values over all municipalities are given for different levels of *T* and for prices of € 2.80/dt and € 3.80/dt. H1 states changes in post-sowing AS use after a ban. However, the composition of the different AS changes only in few municipalities, but those changes are overall not significant.

We cannot reject H2 that weed control becomes more expensive under a ban. We find that in municipalities where glyphosate was used in the benchmark, a significantly different amount is spent on weed management under a ban (plus € 4–6/ha. The effect decreases with the higher price of € 3.80/dt due to the higher intensity of pre-sowing weed management in the benchmark scenario at the higher price level. The cost increase stems from substituting glyphosate mostly with one or two passes of chisel ploughing. Note that sowing is assumed to be cheaper after two passes compared to only a single pass of chisel ploughing (and also cheaper compared to glyphosate application only). The application of mechanical strategies leads to a significant increase in labour demand (H3). That effect, however, decreases if *T* is lower, *i.e.* the weed pressure after sowing is high. In the latter case, more expensive post-sowing strategies with selected herbicides are used instead.

Generally, expected gross margins vary highly across municipalities reflecting yield differences. Furthermore, the later maize emerges compared to weeds, the lower the gross margin will be. A glyphosate ban causes in our simulation, on average over all glyphosate-using municipalities, decreases of the gross margins (already accounting for higher costs for labour) of about € 1–2/ha, with maximal reductions of € 9/ha (for P=€ 2.80/dt) and € 13/ha (for P=€ 3.80/dt) over the two year growing period. In single years, however, costs can be higher if our assumptions on resistance management are neglected. Thus, the loss of gross margin due to the ban of glyphosate is moderate and the null hypothesis of no change cannot be rejected (see Table 4.3, row H4). The reasons for the moderate gross margin reduction although overall costs increase are again due to the cost savings in precision drilling when glyphosate is substituted by two passes of chisel ploughing. Fig. 4.7 presents the distribution of those losses over the federal state of NRW for different levels of *T* and at P=€ 3.80/dt. First, the differences between the regions are due to the differences in attainable yield levels (Fig. 4.3). Second, differences are dependent on soil types: losses are higher on clayey soils due to relatively high cultivation costs and they are higher on sandy soils due to cost saving when applying direct precision drilling (or strip till). Third, we find that the overall weed pressure ($\sum_i^{32} w_{m,i} \cdot a_i$; *e.g.* by spread of *Panicoideae*-varieties or *Alopecurus myosuroides*, see also Fig. 4.2) is higher in municipalities with lower gross margin losses under a glyphosate ban. This is at first glance a somewhat counterintuitive observation. The reason behind this is that our model simulates already under the benchmark in the light of a high pressure from different weeds a lower pre-sowing weed control intensity, as the gross margin is higher when some yield losses are accepted instead of controlling with high costs multiple weeds. Under a ban of the non-selective herbicide, the shift to mechanical pre-sowing strategies is then also not accompanied by expensive post-sowing use of selective herbicides while the overall already higher weed pressure and thus the attained yield is not much affected.

The reduced plant protection intensity under a ban is reflected in significantly decreased yields by about 0.5–1% (H5), which turns out as more profitable than maintaining the control effort with more expensive strategies (difference is significant at higher levels of *T* and the two presented prices in Table 4.3).

The results of the hypothesis testing do not change much when a higher potential water limited yield is assumed. The most important change is some additional cost increase in weed control.

Table 4.3. Differences between glyphosate-ban-scenario and glyphosate-licensed-scenario (mean across the glyphosate using municipalities) and results of hypotheses testing

	Maize emergence	**T=30**		**T=10**		**T=-10**		**T=-30**		**T=-50**		**T=-70**		**T=-90**	
	Price (€/dt)	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**
H1: share of municipalities (%)	No Herbicide	1.9	4.0	2.0	1.7	6.5	2.9	7.9	-	-	-	-	-	-	-
	Mechanic	-	-	-	1.7	-	2.9	2.6	-	-	-	-	-	-	-
	Nothing	1.9	4.0	2.0	-	6.5	-	5.3	-	-	-	-	-	-	-
	Herbicide	-1.9	-4.0	-2.0	-1.7	-6.5	-2.9	-7.9	-	-	-	-	-	-	-
	Dicamba	-	-	2.0	-	-	-1.8	-	-0.6	-	-	-	0.7	-	-
	Flufenacet	-	-	-	-	-	-	-	-	-	-	-	-0.7	-	0.8
	Foramsulfuron	-	-	-	-	-	-	-	-	-	-	-	-0.7	-	0.8
	Iodosulfuron	-	-	-	-	-	-	-	-	-	-	-	-0.7	-	0.8
	Mesotrione	-	-	-	1.2		1.8	-2.6	2.4	-	-2.5	-	1.5	-	1.6
	Nicosulfuron	-1.9	-4.0	-3.9	-2.9	-6.5	-2.9	-5.3	-1.8	-	2.5	-	-1.5	-	-2.4
	Pethoxamid	-	-	-	-	-	-	-	-	-	-0.6	-	2.2	-	0.8
	Prosulfuron	-1.9	-4.0	-3.9	-2.9	-6.5	-2.9	-5.3	-1.8	-	2.5	-	-1.5	-	-2.4
	Pyridate	-	-	-	1.2	-	-1.2	-	2.4	-	-1.9	-	-0.7	-	0.8
	S-Metolachlor	-	-	-	-	-	2.9	-2.6	-	-	0.6	-	-	-	-
	Terbuthylazine	-	-	-	1.2	-	1.8	-2.6	2.4	-	-2.5	-	0.7	-	2.4
	Thiencarbazone	-	-	-	-	-	-	-	-	-	-	-	-0.7	-	0.8
	Tritosulfuron	-	-	2.0	-	-	-1.8	-	-0.6	-	-	-	0.7	-	-
H2: €/ha	Weed management costs	6.19 ***	5.48 ***	5.83 **	5.24 ***	5.88 **	4.89 ***	4.31	4.80 ***	4.02	4.38 ***	2.17	4.25 ***	8.09	3.76 **

Table 4.3. *(continued)*

	Maize emergence	T=30		T=10		T=-10		T=-30		T=-50		T=-70		T=-90	
	Price (€/dt)	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**
H3: h/ha	Weed management labour demand	0.34 ***	0.34 ***	0.34 ***	0.34 ***	0.34 ***	0.33 ***	0.33 ***	0.33 ***	0.31 ***	0.33 ***	0.27 ***	0.33 ***	0.27 ***	0.31 ***
H4: €/ha	Gross margin per ha	-1.12	-1.99	-1.11	-1.87	-1.09	-1.83	-0.87	-1.68	-0.94	-1.58	-0.73	-1.51	-0.37	-1.28
H5: (%)	Yield difference (model yield minus real yield)	-0.7 ***	-0.4 **	-0.7 **	-0.4	-0.7 *	-0.4	-0.9 **	-0.4	-0.8	-0.4	-0.8	-0.4	-0.2	-0.4

Notes: *, ** and *** represent 5%, 1% and 0.1% significance levels using a t-test. No mark means that no significant difference occurred. Note that for tests on hypothesis H1, a Bonferroni correction was used.

We also tested the hypotheses using a Wilcoxon-Mann-Whitney test, which led overall to similar results. Only for H5, we get more significance values for the lower levels of *T* and also higher levels of significance.

T = growing degree days, which is the sum of the average temperature of each day. T = 30: maize emerges early compared to weeds. T = -90: maize emerges lately compared to weeds.

Hypotheses: H1: average post-sowing strategies change in case of a glyphosate ban (average of t_1 and t_2),
H2: costs for weed management increase in case of a glyphosate ban (average of t_1 and t_2),
H3: working force demand increases in case of a glyphosate ban (average of t_1 and t_2),
H4: the gross margin decreases in case of a glyphosate ban (average of t_1 and t_2),
H5: yields decrease in case of a glyphosate ban (average of t_1 and t_2).

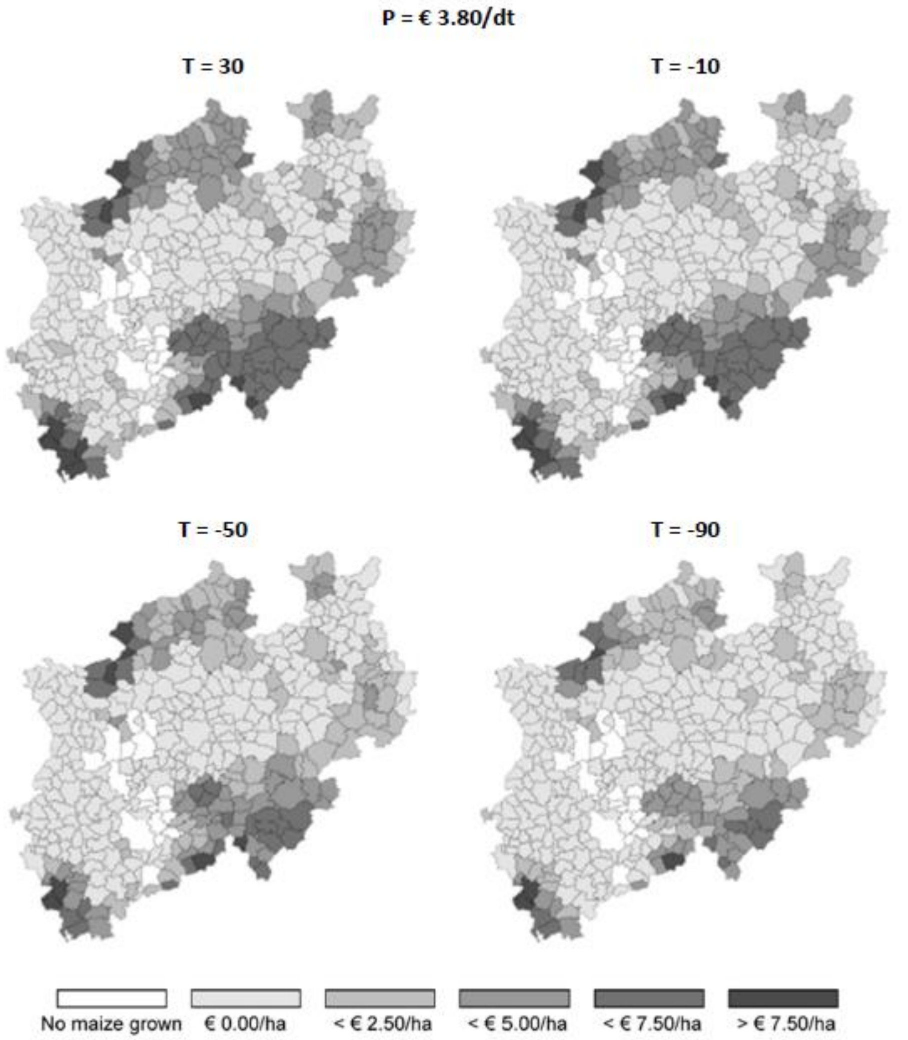

Fig. 4.7. Distribution of the simulated gross margin losses due to a glyphosate ban in NRW, Germany, over a two year maize growing period (for selected levels of *T* and a price of P = € 3.80/dt).

4.5 Discussion

Our model focusses on potential changes in weed management within a single crop. Thus, our results present potential rather short-term effects in herbicide demand for weed control in silage maize production and thus can be used to quantify intensive margin effects of agri-environmental policies targeting single herbicides. Our normative model simulates limited yield losses with some extra costs for farmers under a glyphosate ban, matching the relatively low yield increasing effect of glyphosate reported in literature (Gehring *et al*., 2012). In our model, this leads to a relatively high efficiency and widespread use of alternative (*i.e.* non-glyphosate based) conservation tillage strategies already under the benchmark. Under a glyphosate ban and profit maximising behaviour, overall control intensity and thus the expected maize yield would be somewhat reduced as maintaining the same level of weed suppression and the expected yield is too costly given the available alternative control strategies.

Especially due to the subsidy induced boom in biogas production from silage maize in Germany (Gömann *et al*., 2011; LWK NRW, 2016), silage maize is currently in shortage, being regionally traded at relatively high prices in years with moderate yields. Reducing yields under a glyphosate ban would most probably drive prices further up, such that more costly weed control strategies could become profitable. Farmers might anticipate these impacts and intensify weed control beyond the current profit optimal point to avoid acting as buyers in the short maize markets. If we restrict the model such that a certain yield has to be achieved (a safety threshold to avoid large maize purchases), also more intensive plant protection intensities are used (with costs > € 120/ha) (not shown).

Compared to other studies being based on expert interviews (Kehlenbeck *et al*., 2015; Schulte *et al*., 2016), our results suggest lower additional costs for maize production; though at an overall lower intensity of herbicide use, which decreases somewhat the yields and thus the gross margin. However, our results are in line with findings of Schmitz and Garvert (2012), who estimated also quite limited economic losses due to a glyphosate ban for different crops for northwest Germany of up to 3% of the respective gross margins. In addition, Kehlenbeck *et al*. (2015) estimated that a 75% increase of the glyphosate price would be necessary to reach equivalence of cost between pure mechanical weed management by tillage and weed management including glyphosate use. Even a higher price increase would be necessary in our model to reach equivalence of costs, but alternative mechanical strategies have to be considered as well. A sensitivity analysis with our model shows already under a 10% price increase some reductions in glyphosate use while, under a 30% increase, glyphosate was substituted by mechanical strategies in every municipality (for T=-20 and P=€ 3.80/dt). We presume that these differences to Kehlenbeck *et al*. (2015) reflect more detail in weed control in our model, which additionally considers adjusting yields. More elastic demand for herbicides as found in our analysis is also confirmed by a recent literature review (Böcker and Finger, 2017).

The observed treatment frequency in German maize production, which measures the number of herbicide applications on a field, varied between 1.31–1.47 in 2011–2015, including pre-emergence treatments with glyphosate (JKI, 2016). Our model simulated lower average treatment frequencies over the two years, which are, for instance between 0.57–1.15 at a level of *T=0* and between 0.95–1.21 at *T=-50*. Indeed, pesticide intensities beyond the profit maximising intensity were also found by other authors (Jacquet *et al*., 2011, for France; Skevas *et al*., 2014, for the Netherlands) which could be explained by the risk-reducing effect of herbicides (Skevas *et al*., 2014). This is not reflected in the profit maximising approach used in this paper, but should be addressed in future research (*e.g*. Lehmann *et al*., 2013).

Despite the fact that regional differences in yields and weed pressure prevent an upscaling of our results to silage maize production in Germany at large, some comparison can be made. For example, the JKI also reports the average share of the surveyed German farms which use a specific AS in maize production (JKI, 2016). For example in 2015, 33% of all surveyed farms used an herbicide strategy containing glyphosate, 91% used a strategy containing terbuthylazine, 50% used a strategy containing bromoxynil, etc. Our simulated shares over different levels of *T* differ partly from those values. For example, bromoxynil was not selected at all, but these differences could also root in our regional focus. Still, for selected AS, and depending on *T* and *P*, quite similar shares were calculated, *e.g.* for nicosulfuron, mesotrione, pethoxamid and partly for glyphosate, terbuthylazine, flufenacet, foramsulfuron and iodosulfuron. The survey of Wiese *et al*. (2018) finds with 22% of the farmers applying glyphosate in 2013/14 a similar value compared to our simulated ones.

Herbicide strategies considered in our model were aggregated to some extent, for instance by defining a two-time post-sowing herbicide-application-strategy as one. Future approaches could further refine the strategies such as depicting each single application according to its characteristics and time of application. This would, however, require improved data availability such as research on weed specific impacts on yields. Additional data could also allow including more generally the control impact depending on doses of specific pesticides in the model. So far, reduced doses are only considered in some strategies which use doses below the manufacturers' recommendation. Also, we decided to neglect potential dynamic control impacts, for instance that a conservation tillage strategy might lead to higher weed abundance in the long-term (Ball, 1992; Buhler, 1992; Schwarz and Pallutt, 2014)[22] or that effective control might depress future weed infestation (Pandey, 1989; Swinton and King, 1994), as it is hard to properly account for external weed seed import in a single plot. Here, Hanzlik and Gerowitt (2011) but also Lundkvist *et al*. (2008) find that geographical position and soil conditions have a higher influence on weed species composition compared to previous weed management.

Future research could apply the presented approach to other field crops and implement it into a whole farm context. Other aspects to be covered in future extensions are effects of fertilisation, of preceding or catch crops and of weed control measures in autumn.

[22] Using a chisel plough instead of a mouldboard plough led in field trials to an increase in soil's weed seed bank. Here, a ban of glyphosate could lead to an increase in weed seed banks in the long-term and make more intense pre- and post-sowing strategies necessary.

4.6 Conclusions

We develop a highly detailed, spatially explicit bio-economic model for weed control in silage maize production drawing on an output damage control approach. Here, the raster data of NetPhyD and BfN (2013) on weed occurrence are a key input to analyse weed spread in Germany. Combining this data with expert information on current weed control strategies and comprehensive yield observation data allows us to develop and parameterise our model for mechanical and herbicide based weed control use in silage maize production for 377 municipalities in the state of North-Rhine-Westphalia. Glyphosate as a non-selective herbicide is simulated in about 5 to 25% of the cases as the profit maximal strategy which combines relatively low control cost with high yields. Simulating optimal control strategies under a glyphosate ban, we find that i) economic losses of a ban are limited for farmers currently applying glyphosate, ii) costs slightly increase under a glyphosate ban as mechanical strategies for conservation tillage are used pre-sowing and accepting some yield loss, while switches to more expensive selective herbicides in post-sowing strategies are simulated only in few cases. Therefore, potential risks linked to increased selective herbicide use are for now limited. iii) Rather, somewhat lower yields reflecting decreased weed control intensity turn out as profitable, which, however, could lead to higher regional maize prices. Finally, iv) demand of labour increases due to higher shares of mechanical strategies.

Acknowledgements

We thank the plant protection consultants of the Chamber of Agriculture of NRW and of the Chamber of Agriculture of Lower Saxony for providing valuable information supporting this research. Moreover, we thank two anonymous reviewers for constructive feedbacks on earlier versions of this paper.

4.7 Appendix

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.ecolecon.2017.08.027.

Appendix 4.A:

Weed control strategies included in the model and related parameters (amount/ha, costs, efficacy)

(Excel file, online)

Appendix 4.B:

Average weed management costs (mechanical and chemical) and average gross margins

Table A4.1. Average gross margin (average of t_1 and t_2, all municipalities, also those without glyphosate use)

T	Gross margin (glyphosate licensed, P = € 2.80/dt)	Gross margin (glyphosate banned, P = € 2.80/dt)	Gross margin (glyphosate licensed, P = € 3.80/dt)	Gross margin (glyphosate banned, P = € 3.80/dt)
40	630.93	630.76	1107.89	1106.96
30	621.90	621.74	1095.22	1094.29
20	612.03	611.87	1081.60	1080.70
10	601.20	601.05	1067.09	1066.23
0	589.21	589.07	1051.24	1050.41
-10	576.31	576.18	1033.46	1032.63
-20	562.43	562.31	1013.62	1012.83
-30	547.77	547.68	991.58	990.85
-40	532.02	531.93	967.40	966.71
-50	514.90	514.83	940.82	940.16
-60	496.64	496.58	912.16	911.57
-70	476.97	476.93	881.58	881.03
-80	456.14	456.09	849.23	848.74
-90	434.40	434.38	815.40	814.98

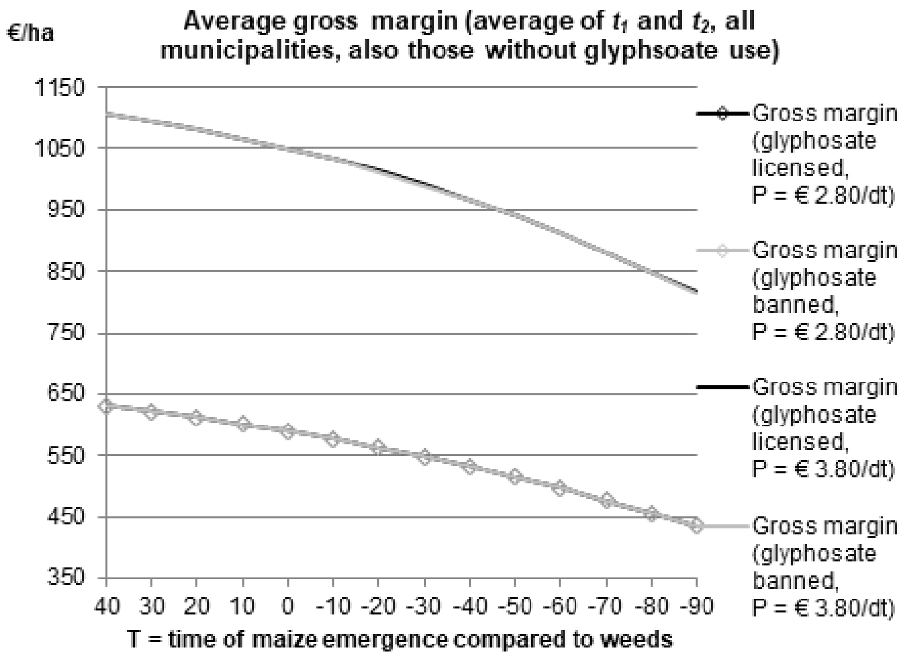

Fig. A4.1. Average gross margin (average of t_1 and t_2, all municipalities, also those without glyphosate use).

Table A4.2. Average costs for weed management (average of t_1 and t_2, all municipalities, also those without glyphosate use)[a]

T	Weed management costs (glyphosate licensed, P = € 2.80/dt)	Weed management costs (glyphosate banned, P = € 2.80/dt)	Weed management costs (glyphosate licensed, P = € 3.80/dt)	Weed management costs (glyphosate banned, P = € 3.80/dt)
40	65.86	66.71	77.76	80.16
30	68.47	69.35	82.73	85.29
20	69.86	70.70	89.33	91.76
10	71.59	72.38	96.27	98.66
0	74.87	75.62	99.11	101.42
-10	80.16	80.88	100.46	102.66
-20	85.14	85.97	101.77	103.90
-30	90.19	90.62	102.58	104.69
-40	92.77	93.38	103.35	105.25
-50	94.65	<u>94.96</u>	105.41	107.24
-60	94.11	94.47	107.32	108.80
-70	<u>94.66</u>	94.79	110.20	111.76
-80	93.71	93.86	113.45	114.77
-90	93.71	94.08	<u>115.47</u>	<u>116.76</u>

[a] Maximum values are underlined.

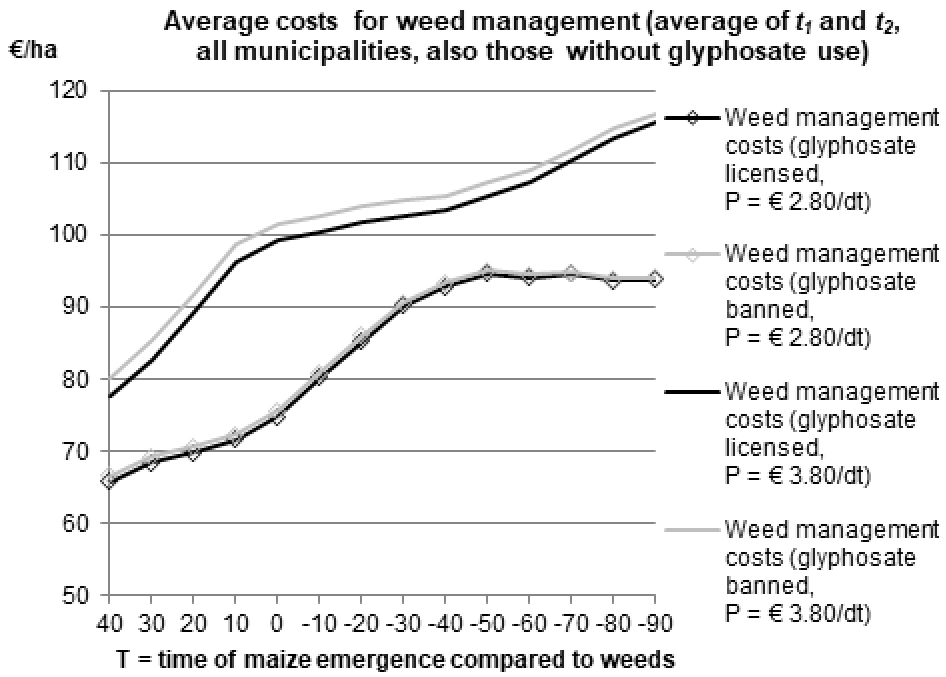

Fig. A4.2. Average costs for weed management (average of t_1 and t_2, all municipalities, also those without glyphosate use).

Appendix 4.C:

Sensitivity analysis with respect to the attainable yield (the attainable yield is increased by 10% in t_1 and 15% in t_2)

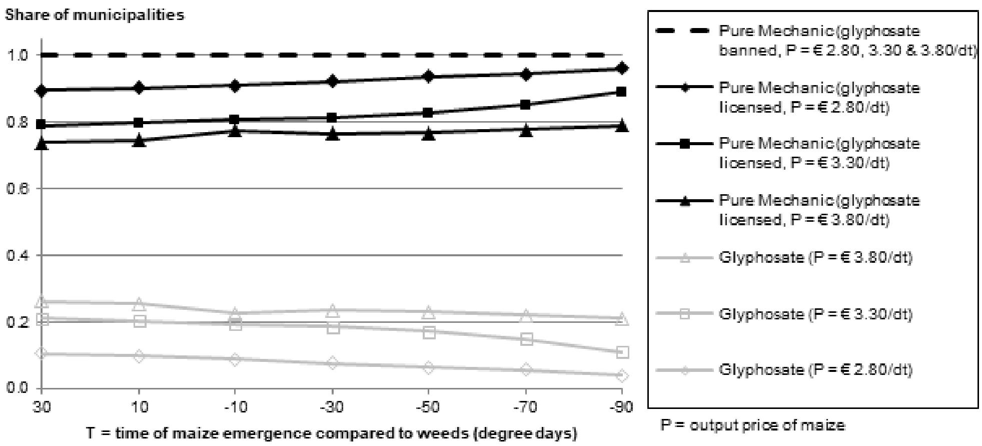

Fig. A4.3. Shares of used pre-sowing strategies (average of t_1 and t_2 of each municipality).

Note: A scenario in which glyphosate is licensed is compared to a scenario with a glyphosate ban. The figure shows the results for the three analysed price levels at different weed pressure levels (expressed as *T*).

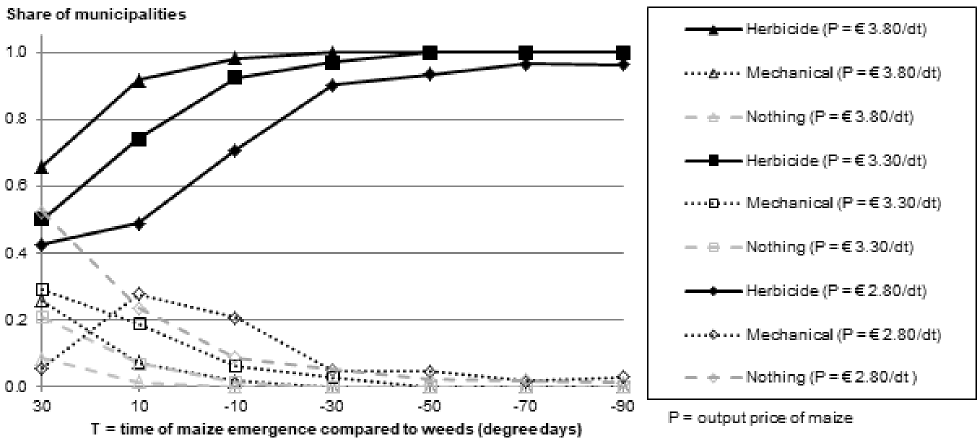

Fig. A4.4. Shares of post-sowing strategies as average of t_1 and t_2 (scenario with glyphosate licensed).

Note: The figure shows the categorisation of the post-sowing strategies in Herbicide, Mechanical and Nothing. Three price levels and different weed pressure levels (expressed as *T*) are analysed.

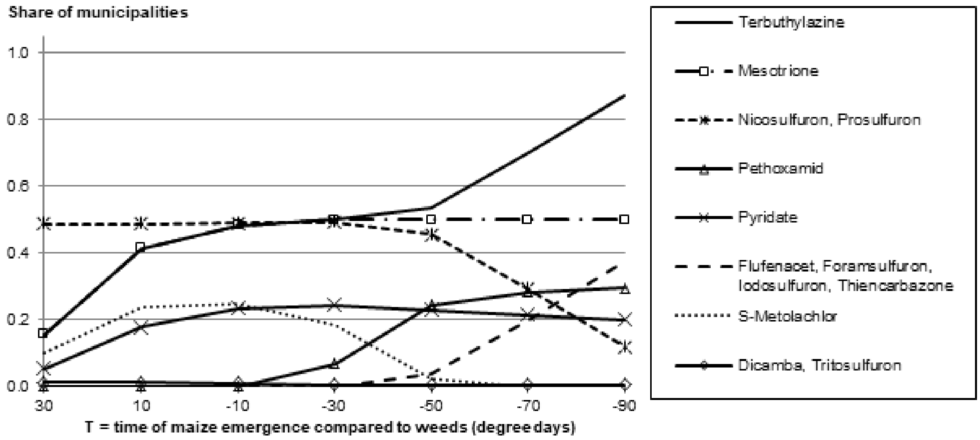

Fig. A4.5. Shares active substances in of post-sowing strategies (glyphosate licensed, P = € 3.80/dt, average of t_1 and t_2).

Note: The figure shows how often an active substance is used on average over the municipalities. For example, a level of 60% for terbuthylazine means that this substance is contained in the weed management strategy, on average of the two years, in 226 of the 377 municipalities each year.

Table A4.3. Differences between glyphosate-ban-scenario and glyphosate-licensed-scenario (mean across the glyphosate using municipalities) and results of hypotheses testing

	Maize emergence	**T=30**		**T=10**		**T=-10**		**T=-30**		**T=-50**		**T=-70**		**T=-90**	
	Price (€/dt)	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**
H1: share of municipalities (%)	No Herbicide	1.3	-	1.4	1.6	13.4	-	6.9	-	6.3	-	-	-	-	-
	Mechanic	-	-	-	1.6	-	-	5.2	-	6.3	-	-	-	-3.4	-
	Nothing	1.3	-	1.4	-	13.4	-	1.7	-	-6.3	-	-	-	3.4	-
	Herbicide	-1.3	-	-1.4	-1.6	-13.4	-	-6.9	-	-	-	-	-	-	-
	Dicamba	-	-	-	-	-4.5	-1.2	1.7	-	-	-	-	-	-	-
	Flufenacet	-	-	-	-	-	-	-	-	-	-	-	-0.6	-	-1.9
	Foramsulfuron	-	-	-	-	-	-	-	-	-	-	-	-0.6	-	-1.9
	Iodosulfuron	-	-	-	-	-	-	-	-	-	-	-	-0.6	-	-1.9
	Mesotrione	-	-	-	-		0.6	-	-0.6	-	-1.1	-	-	3.4	0.6
	Nicosulfuron	-1.3	-	-1.4	-1.6	-9.0	0.6	-8.6	0.6	-6.2	1.2	-	0.6	-3.4	1.3
	Pethoxamid	-	-	-	-	-	-	-	-	-	-0.6	-	-	-	-
	Prosulfuron	-1.3	-	-1.4	-1.6	-9.0	0.6	-8.6	0.6	-6.2	1.2	-	0.6	-3.4	1.3
	Pyridate	-	-	-	-	-	0.6	-	-0.6	-	-	-	-	-	0.6
	S-Metolachlor	-	-	-	-	-	-	-	-	-	-0.6	-	-	3.4	-
	Terbuthylazine	-	-	-	-	-	0.6	-	-0.6	-	-1.1	-	-0.6	3.4	-1.3
	Thiencarbazone	-	-	-	-	-	-	-	-	-	-	-	-0.6	-	-1.9
	Tritosulfuron	-	-	-	-	-4.5	-1.2	1.7	-	-	-	-	-	-	-
H2: €/ha	Weed management costs	4.49 **	7.01 ***	4.23 *	6.62 ***	5.60 ***	6.36 ***	5.74 ***	5.72 ***	6.42 **	5.10 ***	4.64	4.65 ***	0.61	4.09 ***

Table A4.3. *(continued)*

	Maize emergence	T=30		T=10		T=-10		T=-30		T=-50		T=-70		T=-90	
	Price (€/dt)	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**	**2.80**	**3.80**
H3: h/ha	Weed management labour demand	0.33 ***	0.36 ***	0.32 ***	0.36 ***	0.33 ***	0.35 ***	0.34***	0.34 ***	0.35 ***	0.34 ***	0.32 ***	0.33 ***	0.26 ***	0.33 ***
H4: €/ha	Gross margin per ha	-1.22	-2.13	-1.19	-2.12	-1.16	-2.08	-1.16	-2.00	-1.11	-1.85	-0.82	-1.68	-0.73	-1.51
H5: (%)	Yield difference (model yield minus real yield)	-0.8 **	-0.3 *	-0.8 **	-0.3	-0.7 *	-0.3	-0.7	-0.4	-0.7	-0.4	-0.8	-0.4	-1.0	-0.4

Notes: *, ** and *** represent 5%, 1% and 0.1% significance levels using a t-test. No mark means that no significant difference occurred. Note that for tests on hypothesis H1, a Bonferroni correction was used.

We also tested the hypotheses using a Wilcoxon-Mann-Whitney test, which led overall to similar results. Only for H5, we get more significance values for the lower levels of T and also higher levels of significance.

T = growing degree days, which is the sum of the average temperature of each day. T = 30: maize emerges early compared to weeds. T = -90: maize emerges lately compared to weeds.

Hypotheses: H1: average post-sowing strategies change in case of a glyphosate ban (average of t_1 and t_2),
H2: costs for weed management increase in case of a glyphosate ban (average of t_1 and t_2),
H3: working force demand increases in case of a glyphosate ban (average of t_1 and t_2),
H4: the gross margin decreases in case of a glyphosate ban (average of t_1 and t_2),
H5: yields decrease in case of a glyphosate ban (average of t_1 and t_2).

4.8 References

Achilles, W., Eurich-Menden, B., Eckel, H., Frisch, J., Fritzsche, S., Fröba, N., Funk, M., Gaio, C., Grebe, S., Grimm, E., Grube, J., Hartmann, W., Horlacher, D., Kloepfer, F., Meyer, B., Sauer, N., Schroers, J. O., Schultheiß, U. and Wulf, S. (2016). Betriebsplanung Landwirtschaft 2016/17 – KTBL-Datensammlung. 25th ed. Darmstadt: Kuratorium für Technik und Bauwesen in der Landwirtschaft e.V. (KTBL).

Babcock, B. A., Lichtenberg, E. and Zilberman, D. (1992). Impact of Damage Control and Quality of Output: Estimating Pest Control Effectiveness. *American Journal of Agricultural Economics* 74: 163-172.

Ball, D. A. (1992). Weed Seedbank Response to Tillage, Herbicides, and Crop Rotation Sequence. *Weed Science* 40: 654-659.

Böcker, T. and Finger, R. (2016). European Pesticide Tax Schemes in Comparison: An Analysis of Experiences and Developments. *Sustainability* 8: 378/1-22.

Böcker, T. G. and Finger, R. (2017). A Meta-Analysis on the Elasticity of Demand for Pesticides. *Journal of Agricultural Economics* 68: 518-533.

Bosnić, A. Č. and Swanton, C. J. (1997). Economic decision rules for postemergence herbicide control of barnyardgrass (Echinochloa crus-galli) in corn (Zea mays). *Weed Science* 45: 557-563.

Buhler, D. D. (1992). Population Dynamics and Control of Annual Weeds in Corn (Zea mays) as Influenced by Tillage Systems. *Weed Science* 40: 241-248.

Carrasco-Tauber, C. and Moffitt, L. J. (1992). Damage Control Econometrics: Functional Specification and Pesticide Productivity. *American Journal of Agricultural Economics* 74: 158-162.

Cousens, R. (1985). A simple model relating yield loss to weed density. *Annals of Applied Biology* 107: 239-252.

Cousens, R., Brain, P., O’Donovan, J. T. and O’Sullivan, A. (1987). The use of biologically realistic equations to describe the effects of weed density and relative time of emergence on crop yield. *Weed Science* 35: 720-725.

Deen, W., Weersink, A. and Turvey, C. G., Weaver, S. (1993). Weed control decision rules under uncertainty. *Review of Agricultural Economics* 15: 39-50.

De Mol, F., von Redwitz, C. and Gerowitt, B. (2015). Weed species composition of maize fields in Germany is influenced by site and crop sequence. *Weed Research* 55: 574-585.

Falconer, K. E. (1998). Managing diffuse environmental contamination from agricultural pesticides: An economic perspective on issues and policy options, with particular reference to Europe. *Agriculture, Ecosystems & Environment* 69: 37-54.

Femenia, F. and Letort, E. (2016). How to significantly reduce pesticide use: An empirical evaluation of the impacts of pesticide taxation associated with a change in cropping practice. *Ecological Economics* 125: 27-37.

Finger, R., Möhring, N., Dalhaus, T. and Böcker, T. (2017). Revisiting pesticide taxation schemes. *Ecological Economics* 134: 263-266.

Fox, G. and Weersink, A. (1995). Damage Control and Increasing Returns. *American Journal of Agricultural Economics* 77: 33-39.

Gehring, K., Thyssen, S. and Festner, T. (2012). Effects of glyphosate application on succeeding crops. In Nordmeyer, H., Ulber, L. (eds), *Proceedings 25th German Conference on Weed Biology and Weed Control, March 13-15 2012, Braunschweig, Germany*. Quedlinburg: Julius-Kühn-Archiv, 419-426.

Gömann, H., Kreins, P., Münch, J. and Delzeit, R. (2011). Auswirkungen der Novellierung des Erneuerbare-Energien-Gesetzes auf die Landwirtschaft in Deutschland. In Schriften der Gesellschaft für Wirtschafts- und Sozialwissenschaften des Landbaus e.V. (ed.), *Möglichkeiten und Grenzen der wissenschaftlichen Politikanalyse*. Münster: Landwirtschaftsverlag, 189-201.

Gross, M. (2013). EU ban puts spotlight on complex effects of neonicotinoids. *Current Biology* 23: R462-R464.

Grovermann, C., Schreinemachers, P., Riwthong, S. and Berger, T. (2017). „Smart‘ policies to reduce pesticide use and avoid income trade-offs: An agent-based model applied to Thai agriculture. *Ecological Economics* 132: 91-103.

Guan, Z., Oude Lansink, A., Wossink, A. and Huirne, R. (2005). Damage control inputs: a comparison of conventional and organic farming systems. *European Review of Agricultural Economics* 32: 167-189.

Guan, Z., Oude Lansink, A., van Ittersum, M. and Wossink, A. (2006). Integrating Agronomic Principles into Production Function Specification: A Dichotomy of Growth Inputs and Facilitating Inputs. *American Journal of Agricultural Economics* 88: 203-214.

Guyton, K. Z., Loomis, D., Grosse, Y., El Ghissassi, F., Benbrahim-Tallaa, L., Guha, N., Scoccianti, C., Mattock, H. and Straif, K. (2015). Carcinogenicity of tetrachlorvinphos, parathion, malathion, diazinon, and glyphosate. *The Lancet – Oncology* 16: 490-491.

Hall, D. C. and Norgaard, R. B. (1973). On the Timing and Application of Pesticides. *American Journal of Agricultural Economics* 55: 198-201.

Hanzlik, K. and Gerowitt, B. (2011). The importance of climate, site and management on weed vegetation in oilseed rape in Germany. *Agriculture, Ecosystems & Environment* 141: 323-331.

Hoffmann, M. (1990). Mechanische und thermische Unkrautbekämpfung. In Diercks, R., Heitefuss, R. (eds), *Integrierter Landbau – Systeme umweltbewußter Pflanzenproduktion – Grundlagen · Praxiserfahrungen · Entwicklungen*. München: BLV Verlagsgesellschaft, 171-182.

HRAC – Herbicide Resistance Action Committee (ed.) (2005). Classification of Herbicides According to Site of Action. Online: http://hracglobal.com/tools/classification-lookup [09 November 2016].

IT NRW – Information und Technik Nordrhein-Westfalen (ed.) (2016). Erntestatistik. Online: https://www.landesdatenbank.nrw.de/ldbnrw/online/ [10 November 2016].

Jacquet, F., Butault, J.-P. and Guichard, L. (2011). An economic analysis of the possibility of reducing pesticides in French field crops. *Ecological Economics* 70: 1638-1648.

JKI – Julius Kühn-Institut (ed.) (2016). Statistische Erhebungen zur Anwendung von Pflanzenschutzmitteln in der Praxis. Online: http://papa.jki.bund.de/ [09 November 2016].

Karagiannis, G. and Tzouvelekas, V. (2012). The damage-control effect of pesticides on total factor productivity growth. *European Review of Agricultural Economics* 39: 417-437.

Kehlenbeck, H., Saltzmann, J., Schwarz, J., Zwerger, P., Nordmeyer, H. Roßberg, D., Karpinski, I., Strassemeyer, J., Golla, B. and Freier, B. (2015). *Impact assessment of partial or complete abandonment of glyphosate application for farmers in Germany*. Julius-Kühn-Archiv Nr. 451. Quedlinburg: Julius Kühn-Institut.

Kropff, M.J., Weaver, S.E. and Smits, M.A. (1992). Use of Ecophysiological Models for Crop-Weed Interference: Relations Amongst Weed Density, Relative Time of Weed Emergence, Relative Leaf Area, and Yield Loss. *Weed Science* 40: 296-301.

Kuosmanen, T., Pemsl, D. and Wesseler, J. (2006). Specification and Estimation of Production Functions Involving Damage Control Inputs: A Two-Stage, Semiparametric Approach. *American Journal of Agricultural Economics* 88: 499-511.

Lefebvre, M., Langrell, S. R. H. and Gomez-y-Paloma, S. (2015). Incentives and policies for integrated pest management in Europe: a review. *Agronomy for Sustainable Development* 35: 27-45.

Lehmann, N., Briner, S. and Finger, R. (2013). The impact of climate and price risks on agricultural land use and crop management decisions. *Land Use Policy* 35: 119-130.

LfL – Bayerische Landesanstalt für Landwirtschaft (ed.) (2016). Unkrautmanagement in Mais. Online: https://www.lfl.bayern.de/ips/unkraut/033928/index.php [09 November 2016].

Lichtenberg, E. and Zilberman, D. (1986). The Econometrics of Damage Control: Why Specification Matters. *American Journal of Agricultural Economics* 68: 261-273.

Lundkvist, A., Salomonsson, L., Karlsson, L. and Dock Gustavsson, A.-M. (2008). Effects of organic farming on weed flora composition in a long term perspective. *European Journal Agronomy* 28: 570-578.

LWK NRW – Landwirtschaftskammer Nordrhein-Westfalen (ed.) (2015a). Unkraut im Mais: Zweimal spritzen, Gewässer schützen. Online: http://www.landwirtschaftskammer.de/landwirtschaft/pflanzenschutz/ackerbau/mais/spritzfolgen.htm [09 November 2016].

LWK NRW – Landwirtschaftskammer Nordrhein-Westfalen (ed.) (2015b). Düngung. Online: http://www.landwirtschaftskammer.de/Landwirtschaft/ackerbau/duengung/index.htm [07 November 2016).

LWK NRW – Landwirtschaftskammer Nordrhein-Westfalen (ed.) (2016). Biogas in Nordrhein-Westfalen. Online: https://www.landwirtschaftskammer.de/landwirtschaft/technik/biogas/veroeffentlichungen/biogas-in-nrw.htm [14 December 2016].

Mehrtens, J., Schulte, M. and Hurle, K. (2005). Unkrautflora in Mais – Ergebnisse eines Monitorings in Deutschland. *Gesunde Pflanzen* 57: 206-218.

Meinlschmidt, E., Schröder, G., Bär, H., Pittorf, I. and Bergmann, E. (2008). Unkrautbekämpfung in Mais. In Sächsische Landesanstalt für Landwirtschaft (ed.), *Schriftenreihe der Sächsischen Landesanstalt für Landwirtschaft*. 1/2008. Dresden: Sächsische Landesanstalt für Landwirtschaft.

NetPhyD and BfN (eds) (2013). Verbreitungsatlas der Farn- und Blütenpflanzen Deutschlands. Bonn: Netzwerk Phytodiversität Deutschlands e.V. (NetPhyD) and Bundesamt für Naturschutz (BfN).

Oerke, E.-C. (2006). Crop losses to pests. *Journal of Agricultural Sciences* 144: 31-43.

Oude Lansink, A. and Carpentier, A. (2001). Damage Control Productivity: An Input Damage Abatement Approach. *Journal of Agricultural Economics* 52: 11-22.

Oskam, A. J. (1997). The economics of pesticides: an overview of the issues. In Oskam, A. J., Vijftigschild, R. A. N. (eds), *Proceedings and discussions – policy measures to control environmental impacts from agriculture – workshop on pesticides, August 1995*. Wageningen Agricultural University: Wageningen, 360-384.

Pandey, S. (1989). Economics of wild oats control: an application of stochastic dynamic programming model. Paper presented to the Thirty-Third Annual Conference of the Australian Agricultural Economics Society, February 7-9, 1989, Canterbury, New Zealand.

Pannell, D. J. (1990). An economic response model of herbicide application for weed control. *Australian Journal of Agricultural Economics* 34: 223-241.

Roth Agrarhandel (2015). Pflanzenschutz Preisliste 2015. Online: http://www.roth-agrar.de/produkte_1/pflanzenschutz/preislisten/pflanzenschutz-preisliste_2015_1.html [10 November 2016].

Schmitz, P. M. and Garvert, H. (2012). Agro-Economic Analysis of the Use of Glyphosate in Germany. *Journal für Kulturpflanzen* 64: 150-162.

Schulte, M. and Theuvsen, L. (2015). The economic benefit of herbicides in arable farming – with a special focus on glyphosate. *Journal für Kulturpflanzen* 67: 269-279.

Schulte, M. C., Theuvsen, L., Wiese, A. and Steinmann, H.-H. (2016). Die ökonomische Bewertung von Glyphosat im deutschen Ackerbau. In Schriften der Gesellschaft für Wirtschafts- und Sozialwissenschaften des Landbaus e.V. (ed.), *Agrar- und Ernährungswirtschaft: Regional vernetzt und global erfolgreich*. Münster: Landwirtschaftsverlag, 29-41.

Schwarz, J. and Pallutt, B. (2014). Influence of tillage system on the weed infestation in a long-term field trial. In Nordmeyer, H., Ulber, L. (eds), *Proceedings 26th German Conference on Weed Biology and Weed Control, March 11-13, 2014, Braunschweig, Germany*. Quedlinburg: Julius-Kühn-Archiv, 141-148.

Skevas, T., Stefanou, S. E. and Oude Lansink, A. (2014). Pesticide use, environmental spillovers and efficiency: A DEA risk-adjusted efficiency approach applied to Dutch arable farming. *European Journal of Operational Research* 237: 658-664.

Söchting, H.-P. and Zwerger, P. (2012). Weed competition and biomass production of maize and sorghum under different herbicide intensity level. In Nordmeyer, H., Ulber, L. (eds), *Proceedings 25th German Conference on Weed Biology and Weed Control, March 13-15, 2012, Braunschweig, Germany*. Quedlinburg: Julius-Kühn-Archiv, 329-335.

Steinmann, H.-H., Theuvsen, L. and Gerowitt, B. (2016). Rahmenbedingungen für eine künftige Anwendung von Glyphosat im Ackerbau. Agra-Europe 23/16, 6th June 2016: 11-14.

Swinton, S. M., Buhler, D. D., Forcella, F., Gunsolus, J. L. and King, R. P. (1994). Estimation of Crop Yield Loss Due to Interference by Multiple Weed Species. *Weed Science* 42: 103-109.

Swinton, S. M. and King, R. P. (1994). A Bioeconomic Model for Weed Management in Corn and Soybean. *Agricultural Systems* 44: 313-335.

Talpaz, H. and Borosh, I. (1974). Strategy for Pesticide Use: Frequency and Applications. *American Journal of Agricultural Economics* 56: 769-775.

Wiese, A., Schulte, M., Theuvsen, L. and Steinmann, H.-H. (2018). Interactions of glyphosate use with farm characteristics and cropping patterns in Central Europe. *Pest Management Science* 74 (5): 1155-1165.

Chapter 5

An Economic and Environmental Assessment of a Glyphosate Ban for the Example of Maize Production[23]

Abstract

Economic and environmental impacts of a glyphosate ban on silage maize cultivation are simulated using a spatially explicit bio-economic model for 377 municipalities in North Rhine-Westphalia (Germany). For each municipality, a utility maximising mix from 74 pre- and post-sowing weed control strategies is chosen, stage-contingent on stochastic pressure of 32 weeds. The resulting damage control drives the distribution of yield levels. The glyphosate ban is found to slightly reduce gross margins and has only minor effects on maize yields. It leads to significant reductions of the toxicity of pesticide use, but increases tillage intensity, resulting in higher energy consumption.

Keywords: herbicide; glyphosate; maize; output damage control; risk.

5.1 Introduction

The relicensing of glyphosate in European agriculture received massive societal and political attention in the years 2016 and 2017. Those advocating a ban used as arguments potential social costs in the form of human health risks (*e.g.* Guyton *et al*., 2015) and of environmental effects, for instance on biodiversity with regard to a decline in fodder plants for butterflies or with

---

[23] This chapter is under review in the *European Review of Agricultural Economics* as: Böcker, T., Britz, W., Möhring, N. and Finger, R. (2018). An Economic and Environmental Assessment of a Glyphosate Ban for the Example of Maize Production.

regard to accumulation of metabolites (*e.g.* Brower *et al.*, 2012; Helander *et al.*, 2012; see Tarazona *et al*., 2017, for an overview of the ongoing debate). In contrast, arguments for the continued use of glyphosate mainly relate to its private and social economic benefits, such as lower production costs and consequently lower food prices, and potential trade-offs in the environmental and human health dimension if glyphosate is substituted with other forms of weed control, such as more intensive selective herbicide application or increased tillage intensity (see *e.g*. Williams *et al*., 2000; Duke and Powles, 2008). Despite the fact that glyphosate was relicensed by the European Commission for five additional years in the end of 2017, the debate will continue. Additionally, we observe that private actors take action and demand glyphosate-free products from suppliers.[24] However, scientific information on trade-offs between environmental, human health and economic implications of not using glyphosate are limited (cf. Finger, 2018).

We contribute to a more informed debate by focusing on key agronomic and economic aspects of a glyphosate ban in a state-wide case study for a major crop based on a highly detailed bio-economic simulation model. More specifically, we test for the following potential consequences of a glyphosate ban: i) yield losses, ii) a higher tillage intensity and consequently changes in diesel consumption and thus energy efficiency, and iii) increased use of post-emergence herbicides with higher toxicity, and thus stronger adverse effects on the environment than glyphosate. Our paper fills a gap in literature as so far only limited scientific evidence on the consequences of a possible glyphosate ban is available (*cf*. Schulte and Theuvsen, 2015; Böcker *et al*., 2018).

[24] For example, some German and Austrian dairies decided that producers are not allowed to apply glyphosate anymore, and in Switzerland, the integrated production organisation IP-Suisse, representing about one third of all Swiss farms, announced an internal ban of glyphosate on crops marketed under their label (*e.g*. Böcker and Finger, 2018).

As real world observations of weed control under a glyphosate ban are not available, we employ a normative modelling approach based on damage abatement functions (Karagiannis and Tzouvelekas, 2012). The methodology presented in this article contributes to the literature by combining state-contingent decisions on pesticide application depending on weed pressure with an expected utility framework using a highly detailed and spatial explicit representation of weeds, weed control strategies and their economic and environmental implications. More specifically, we quantify optimal alternative weed control strategies (including mechanical and chemical pre- and post-sowing strategies) under a glyphosate ban, considering production risk and farmers' risk preferences. To this end, we extend the model and analysis by Böcker *et al*. (2018) in two important directions. First, we account for different sources of risk by incorporating uncertainty with respect to attainable yield levels and weed pressure. Attainable yield levels are unknown in the time of pesticide application, so that the production system contains a high level of uncertainty. For example, in a dry year with low yield levels the benefit of a more intense weed control strategy is rather low because the yield reduction stems from water scarcity and not from weed competition for light or nutrients. By incorporating such risks and farmers' risk preferences in our model we therefore account for a major determinant of pesticide use decisions (Horowitz and Lichtenberg, 1993). Second, we assess the environmental implications of the chosen weed control strategies with different environmental indicators. Environmental consequences of a glyphosate ban are core arguments in the political debate, and should be considered. A pesticide risk indicator is used to evaluate adverse effects of pesticide application on human health and on the ecosystem. Here, we choose the very detailed Pesticide Load Indicator which has been developed and applied in Denmark (Kudsk *et al*., 2018). Furthermore, we quantify the process energy demand of weed control strategies and thus account for all physical material flows. This allows to also assess the environmental effects of potential increased machinery use in response to a glyphosate ban. In summary, our approach allows to conduct a holistic assessment of potential effects of a glyphosate ban, integrating potential economic as well as the most important environmental effects. Revealing and quantifying possible trade-offs between different goal functions is crucial for a well-informed policy debate on glyphosate. The here developed model may further serve as an important building stone for assessments concerning future debates on pesticides.

The model is applied to silage maize cultivation in the federal state of North-Rhine-Westphalia (NRW), Germany (Fig. 5.1). Maize is one of the major crops in Germany, grown on 21% of the arable land and 15% of the agricultural area (2.5 million hectares [1 ha = 10,000 m^2] in 2017; Statistisches Bundesamt, 2017). In our case study region NRW, maize is even the dominant crop with a share of almost 30% of the arable land, mainly used for cattle feeding and for biogas production, either as silage maize or as grain maize or corn-cob-mix (Information und Technik Nordrhein-Westfalen, 2016). Glyphosate is mainly applied before sowing in cultivation of maize (Fig. 5.2).[25] Between 22% and 35% of the maize growing farmers have been found to apply glyphosate, either in combination with mechanical strategies, such as chisel ploughing, or with direct sowing of maize (Julius Kühn-Institut, 2017; Wiese *et al*., 2018). The remaining farmers use mechanical strategies without herbicide application before sowing, for example mouldboard ploughing or one or two passes of chisel ploughing and/or rotary harrowing. Direct sowing in combination with a glyphosate application is especially relevant on light soils where reduced traction need leads to lower costs of sowing (direct sowing is usually more expensive than conventional

[25] Note that genetically modified crops (*i.e.* also herbicide tolerant crops) are not cultivated in Germany and glyphosate is thus used only as pre-sowing strategy. Due to the overall lower intensity of glyphosate use, for example compared to the USA, resistances of weeds against glyphosate (also called superweeds) are not reported on a large-scale. Yet, resistances to other herbicides are relevant and considered in our modelling approach (see section 5.2.2).

sowing). It is also found on heavy soils where direct sowing can have a cost advantage because of high traction needs for mechanical tillage strategies.[26] After sowing, a large share of farmers applies selective herbicides. Mechanical strategies such as hoeing are poorly established.

Our paper is structured as follows: we first present the modelling approach based on an output damage function approach combined with expected utility maximisation. Moreover, we give some detail on the PLI and the energy process analysis and present the data used. Finally, we present and discuss results and conclude.

Fig. 5.1. Location of the case study region NRW in Germany and borders of the geographical units (municipality borders).

[26] On heavy soils, mouldboard ploughing is frequently done in autumn. The purpose is less for weed control but rather for achieving a soil with a coarse surface broken up by the frost in winter.

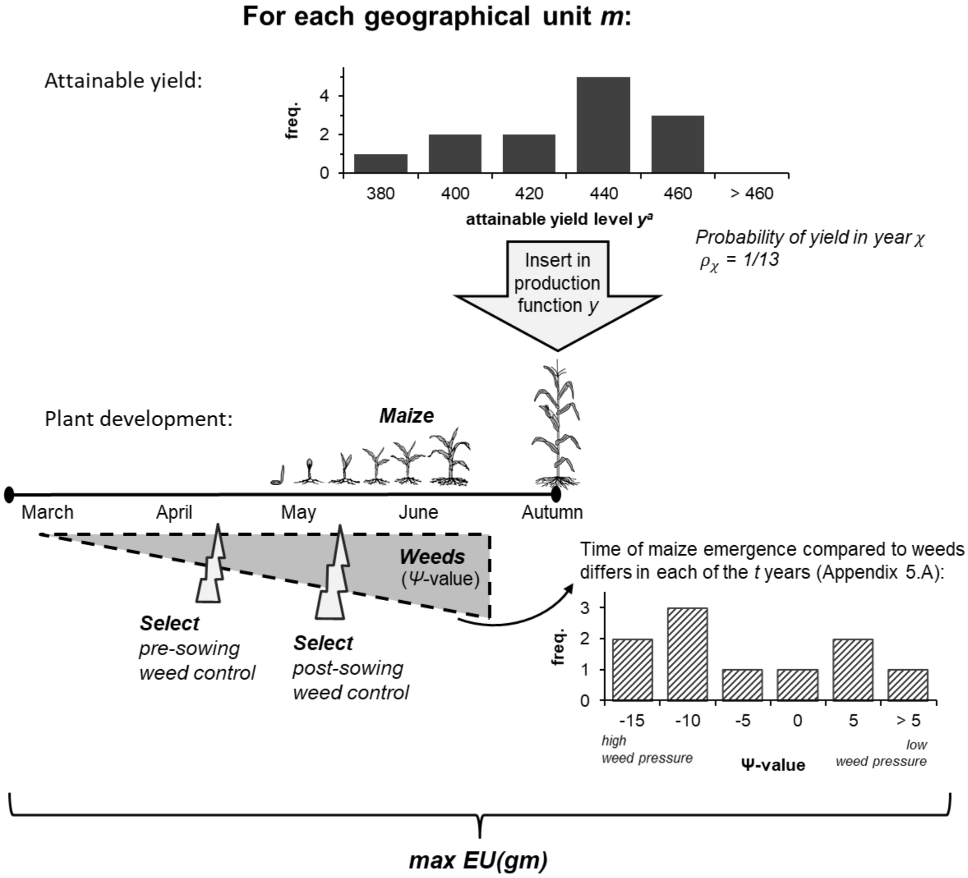

Fig. 5.2. Overview of the methodology. The probability that a certain attainable yield occurs depends on a distribution of yield levels over 13 years per geographical unit. The time of weed emergence compared to maize (Ψ) is estimated for each regional unit *m* and each year *t*.

5.2 Methodology

The spatially explicit optimisation approach applied in this article draws on Böcker *et al*. (2018)[27] where model construction and application are based on two major steps. First, the maize yield is estimated as a function of weed pressure and weed control. Gross margins for each weed control strategy are calculated accounting for output prices and costs, which are also dependent

[27] A detailed model documentation along with the GAMS code related to the model of Böcker *et al*. (2018) is provided online in Böcker *et al*. (2017).

on weed pressure. Second, the optimal pre- and post-sowing weed control strategies are selected. However, the approach of Böcker *et al*. (2018) is deterministic: it is assumed that farmers know for certain both the weed pressure and the realised yield. We expand the approach by accounting for the uncertainty associated with attainable yield distributions and weed pressure, which allows introducing a detailed stochastic production function and accounting for risk preferences of farmers. We differentiate weed pressure and weed control across space to analyse environmental, ecological and economic effects. This output damage control approach is applied to each of the 377 silage maize-producing regional units of NRW, $m = 1, ..., M$ (Fig. 5.1 and Fig. 5.2).

5.2.1 Modelling Structure

The objective function of the model is to maximise farmers' private expected utility based on the distribution of gross margins (*gm*) from silage maize production. Accounting for different pre- (index *g*) and post-sowing (index *h*) weed control strategies, the distribution of the gross margin for a ha of maize $\widetilde{gm}$ is defined as:

$$\widetilde{\mathrm{gm}} = [\tilde{\mathrm{y}} \cdot \mathrm{P} - \mathrm{c(g)} - \mathrm{c_s(g)} - \mathrm{c(h)} - \mathrm{c_c}(\tilde{\mathrm{y}}) - \mathrm{c_o}], \tag{5.1}$$

where $\tilde{y}$ is the distribution of the yield, P is the silage maize price, $c(g)$ and $c(h)$ the pre- and post-sowing weed control (and tillage) costs. $c_s(g)$ are costs for sowing (that depend also on the pre-sowing weed control strategy[28]), $c_c(y)$ are yield-dependent costs for fertiliser, harvest, transport and ensiling, and c_o are other cost. The expected yield $E(y)$ depends on an output damage control approach where weed control strategies decrease damage compared to a yield distribution without weed pressure (Hall and Norgaard, 1973;

[28] For example, higher costs for sowing arise in strategies without seedbed preparation (so called no-till or direct sowing strategies). Lower costs arise if the soil is crumbled and levelled and death plant material is mixed with it.

Lichtenberg and Zilberman, 1986). The production function focussed on the yield effects of weed control is defined as follows:

$$\tilde{y}_{m,t,\chi,g,h} = \underbrace{\left(1 - e^{-(\alpha_0+\alpha_1 \cdot v_{m,j})^2}\right)}_{\text{Pre-sowing}} \cdot \underbrace{\tilde{y}^a_{m,\chi}}_{\text{Attainable yield}} \cdot \underbrace{\left[1 - I \cdot \frac{e^{-(\beta_0+\beta_1 \cdot z_{m,j})^2}}{100 \cdot \left(e^{C \cdot \tilde{\Psi}_{m,t}} + I \cdot \frac{e^{-(\beta_0+\beta_1 \cdot z_{m,j})^2}}{A}\right)}\right]}_{\text{Post-sowing}} \tag{5.2}$$

The production function consists of three parts. The first part accounts for the pre-sowing weed control activities. Secondly, $\tilde{y}^a_{m,\chi}$ is the attainable yield distribution for a regional unit *m* in year χ. The third part accounts for the post-sowing weed control. In the latter, $e^{-(\beta_0+\beta_1 \cdot z_{m,j})^2}$ is the yield loss based on different control strategies (denoted further as *D*), *I* is the percent yield loss as *D* approaches 0, *A* is the percent yield loss as *D* approaches infinity, $\tilde{\Psi}$ is the distribution of the time of maize emergence in relation to weed emergence (in growing degree days; this measure can be seen as an expression for the overall weed pressure in a certain year *t*) and *C* is the rate at which the yield loss *I* decreases as Ψ becomes larger.

Farmers face several uncertainties when deciding upon an optimal weed control strategy (*e.g.* Auld *et al.*, 1987): i) the level of weed infestation, ii) the effectiveness of the weed control strategy, iii) prices, yield improvement and quality effects, and iv) reinvasion, spill-overs on own crop and time-interval effects from delays of receiving benefits. We here focus on i) the level of weed infestation because differing yearly growing conditions, mainly climatic ones, lead to varying weed pressure. In contrast, other aspects are either not important for maize production in our case study (*e.g.* quality effects) or assumed to be less relevant because of the high level of information provided to farmers (*e.g.* effectiveness of weed control strategies; see also section 5.2.2.). Moreover, v) there is uncertainty regarding the attainable yield level, which is of high importance for our application on silage maize.

Weed control usually occurs relatively early in the growing season of maize and accordingly, the same weed control can lead to different yield outcomes. To reflect this uncertainty, the attainable yield $y^{a}_{m,\chi}$ is introduced as a random variable in our model (see Fig. 5.2). We do not account for price risks because silage maize output prices are not characterised by high volatility because the biogas boom in Germany has stimulated the widespread use of long-time supply contracts (Reise *et al.*, 2012; Britz and Delzeit, 2013).

Thus, we introduce risk in our model by accounting for the stochasticity of weed pressure and attainable yield. The attainable yield in each year χ is assumed to be stochastic, *e.g.* due to stochastic weather conditions that are independent of weed pressure and weed control (*e.g.* Tembo *et al.*, 2008). This yield variability is captured by an empirical yield distribution quantified for each regional unit *m*. Concerning weed pressure, farmers face uncertainty with respect to the time of weed emergence Ψ relative to maize, which is a key indicator of weed induced potential yield losses. Herbicide strategies are chosen and applied early in the growing season of maize based on the observed weed pressure. If maize has a growth advantage over weeds in a certain year, cheaper or no herbicide strategies might be favoured. In contrast, strategies with higher efficacy that are often also more complex and expensive might be more promising if weeds have larger growth advantages over maize. An overview of the methodology is also given in Fig. 5.2.

The further parameters in the production function are $v_{m,j}$ and $z_{m,j}$, the pre- and post-sowing weed control effects, *i.e.*:

$$
\begin{aligned}
v_{m,j} &= \sum_{i}^{I} w_{m,i} \cdot a_i \cdot g_{j,i} \\
z_{m,j} &= \sum_{i}^{I} w_{m,i} \cdot a_i \cdot h_{j,i}
\end{aligned}
\tag{5.3}
$$

where $w_{m,i}$ is the probability that a weed i occurs in regional unit m. a_i is the average abundance if a weed is not controlled and acts as a measure for its yield reducing effect, while $g = 1, ..., G$ and $h = 1, ..., H$ are the efficacies of the different weed control strategies j against each weed i [0,1].

The shares of the pre- and post-sowing weed control strategies, *i.e.* $S_{g,m,t}$ and $S_{h,m,t}$ (Eq. (5.4), are the decisions variables. Additionally, a vector $\varphi_{g,glyphosate}$ contains either 0 or 1 reflecting scenarios whether glyphosate is licensed or not:

$$\mathrm{gm}_{m,t,\chi} = \sum_{g=1}^{G}\sum_{h=1}^{H} \mathrm{gm}_{m,t,\chi,g,h} \cdot S_{g,m,t} \cdot \varphi_{g,glyphosate} \cdot S_{h,m,t} \ , \qquad S_{g,m,t}, S_{h,m,t}, \varphi_{g,glyphosate} \in \mathbb{R} = [0,1] \tag{5.4}$$

5.2.2 Goal Function

We use an expected utility (EU) framework to represent production risks, farmers' risk preferences and risk dependent behaviour in our programming model (see *e.g.* Hardaker *et al.*, 1991; Lehmann *et al.*, 2013), based on a power utility function:

$$U(\widetilde{\mathrm{gm}}_{m,t,\chi}) = \frac{1}{1 - r_a} \cdot \widetilde{\mathrm{gm}}_{m,t,\chi}^{1-r_a} \ , \tag{5.5}$$

where r_a represents the partial risk aversion coefficient (Hardaker *et al.*, 2015: 91). A utility function is suited to weigh the variability in yield in a farmer's objective function according to his or her risk preferences. The chosen functional form allows flexible representation of risk preferences and exhibits decreasing absolute risk aversion so that downside risk aversion as a salient pattern of farmers' behaviour can be represented consistently (Chavas and Holt, 1996).[29] This reflects that risk averse farmers aim to avoid low profit events.

We calculate for a strategy and the observed weed pressure Ψ in each year t the distribution of the gross margins, deriving the distribution of the realised yield by correcting the empirical distribution of the attainable yield based on the controlled damage. The distribution of the attainable yield is based on water-limited yields simulated with a crop growth model over 13 years at a 1x1 km raster. The distribution of the gross margin for each strategy is next used to derive expected utility levels and finally, weed control strategies are chosen to maximise *EU* for each regional unit *m*:

$$\max \mathrm{EU}(\widetilde{\mathrm{gm}}_{\mathrm{m}}) = \sum_{\mathrm{t}=1}^{\mathrm{T}} \mathrm{EU}\left(\widetilde{\mathrm{gm}}_{\mathrm{m,t}}\right) \tag{5.6}$$

Considering simultaneously the years *t* for which observations on weed pressure are available allows to reflect that farmers aim to avoid resistances of weeds against herbicides. Specifically, it is assumed that farmers need to change the used active substances. More precisely, the strategies were classified into groups according to the Herbicide Resistance Action Committee and a constraint was added to prevent that strategies from the same group are used in two consecutive years. Furthermore, strategies containing the active substance nicosulfuron are only allowed to be applied every second year.

To address the research questions, we compare a baseline scenario, in which glyphosate is licensed, to a counterfactual scenario, in which glyphosate is banned. We report in the main body of the paper results for slightly risk averse behaviour with $r_a = 0.5$, reflecting recent empirical evidence for

[29] The constant relative risk aversion property of the power utility function enables the here used normalisation of gross margins. Normalisation of expected gross margins in a regional unit *m* in year *t* ($\pi_{m,t,\chi}$) with the maximum income over the range of years (max π_m) is superior to the use of absolute values that differ largely across regions.

German farmers (Maart-Noelck and Mußhoff, 2014; Meraner and Finger, 2017). We conduct additional sensitivity analyses with respect to the partial risk aversion coefficient, considering values of -2.0, 0.0, and 0.8 which reflect risk loving, risk neutral and more risk averse preferences (results presented in the appendix). This is relevant as farmers are found to be on average slightly risk averse, but at the same time a large heterogeneity in the population exists (*ibid*.). Furthermore, we assume expected output prices for silage maize of € 4.00, 4.60 and 5.20/dt (dt denotes deciton, *i.e.* 100 kg), reflecting the range of currently observed silage maize prices. It is assumed that harvesting and ensiling are done by the selling farmer. In addition to the four levels of risk aversion, we also include sensitivity analysis with regard to the expected attainable yield by considering a 10% higher attainable yield level because yield expectations may increase in the future due to new varieties and/or better fertilisation (the results of this analysis can be found in the appendix). In total, we consider therefore 4 (risk aversion) x 3 (maize price) x 2 (attainable yield) = 24 different scenarios. For each scenario, we check for significant differences in gross margins and environmental impacts, using the municipalities as the observation sample as discussed below.

5.2.3 Pesticide Load Analysis

In order to assess potential adverse effects of herbicide use on the environment and human health, we employ the product specific Pesticide Load Indicator which complies with European pesticide regulations. Developed for the Danish Ministry of Environment, it serves as the basis for the Danish pesticide policy goals and pesticide taxation. Pesticide load values are computed individually for each active substance (AS) and are then aggregated to marketed products which can combine different AS. The PLI considers subindicators for Human Health Λ_{heal}, Environmental Fate and Behaviour Λ_{fate} and Environmental Toxicity Λ_{toxy} which in sum define the total load Λ_{total}:

$$\Lambda_{\text{total}} = \Lambda_{\text{heal}} + \Lambda_{\text{fate}} + \Lambda_{\text{toxy}} \tag{5.7}$$

Values for each sub-indicator are computed from a broad range of potential effects on the environment and human health. More specifically, Λ_{toxy} assesses short-term effects on eight different families of animals and plants (birds, mammals, fish, earthworms, bees, daphnia, aquatic plants and algae). Additional long-term effects are taken into account for fish, earthworms and daphnia. Λ_{fate} considers biodegradability, bioaccumulation and mobility in soil. Λ_{heal} is calculated based on Hazard- and Risk Statements with regard to human health of the specific substances as well as product formulation. For a specific pesticide product, the load per kilogram or litre is calculated based on the load of each single AS and its concentration in the product. For details, see Kudsk *et al*. (2018). PLI values for the used products are presented in detail in the data section.

5.2.4 Energy Process Analysis

In order to assess the energy use related to a specific weed control strategy, we use the methodology and definitions of Jones (1989) and of Hülsbergen *et al*. (2001) (for a recent application see Jankowski *et al*., 2015). The aim of that approach is to "trace all the energy inputs into an agricultural system, based on physical material flows" (Hülsbergen *et al*. (2001: 306f.), excluding energy flows from human labour and solar energy (Uhlin, 1999). Direct energy input (E_d) refers in our case study to the consumption of diesel whereas indirect energy inputs E_i quantifies the energy needed to produce the different inputs: seed, mineral fertilisers (we treat manure as waste from livestock production, *i.e*. assigning zero energy content), pesticides and machinery. The overall energy input E is equal to E_d + E_i. The energy output EO is equal to the energy content of the harvested maize minus the inherent energy in seed (which is lower than the energy needed to produce the seed).

The net energy output *NEO* is equal to *EO* – *E*. All energy values are given in calorific values [MJ/ha].[30]

5.2.5 Hypothesis Testing

With respect to the applied herbicides in case of a glyphosate ban, we test for differences in i) weed control costs, ii) yields and iii) the expected gross margin. Furthermore, potential pesticide load decreases are tested according to iv) toxicity, v) environmental fate, vi) human health, and vii) overall pesticide load (sum of all load indicators, see data section). In the energy process analysis, we test the hypotheses that viii) the energy output *EO* decreases (*i.e*. the yield decreases), ix) the net energy output *NEO* decreases, x) more direct energy is used (E_d increases), xi) more indirect energy is used (E_i increases), xii) more energy is used in general (*E* increases), and finally xiii) the energy efficiency decreases (*EO/E*). Wilcoxon-Mann-Whitney tests are used to test our hypotheses on differences between regional unit averages of results over all years *t*.

5.3 Data

The first part of the data section gives an overview of the most important data sources of the above presented bio-economic model. The focus lies on weed control strategies, weed spread and yield data. The subsequent two sections present the data underlying the application of the PLI and the energy balance.

[30] We opted against the focus on CO_2-equivalents due to the large uncertainties in assessing CO_2-equivalent emissions from energy production. Especially for commodity production and related demand of electricity, it depends largely on where factories are located since most countries have a mix of electricity resources and different environmental standards.

5.3.1 Weed Data, Weed Control Strategies and Yield Data

The model consists of $m = 1, \ldots, 377$ regional units, which represent the maize-producing municipalities of NRW (Fig. 5.1). The complete data sources on weed spread, weed abundance, yield losses and herbicide efficacy are documented in Böcker *et al*. (2018). We account spatially explicitly for $i = 1, \ldots, 32$ (grass-)weeds in the model that influence the yield depending on the specific weed. Each regional unit has a certain probability that a specific weed occurs. This data is taken from the FloraWeb database, an open GIS-based platform (NetPhyD and BfN, 2013). In our model, we account for the heterogeneity of weed pressure across space and time. However, we do not account for inter-annual or -regional dynamics of weed abundance because geographical position and soil conditions have been found as more important for the composition of weed species than management factors (Hanzlik and Gerowitt, 2011; Lundkvist *et al*., 2008). In NRW, small to medium size fields are present. Weed seed import is therefore likely, for instance, by wind, unsprayed field edges or machinery, which further motivates to refrain from modelling explicitly weed dynamics which would require besides maize the consideration of all crops in which any of the 32 considered weeds could occur.

For pre-sowing weed control, $g = 1, \ldots, 19$ strategies are considered and for post-sowing $h = 1, \ldots, 55$ (see online Appendix 5.B). This selection includes both the currently dominating strategies and strategies that are currently not yet economically viable but might become relevant under a glyphosate ban. Pre-sowing strategies consist of different combinations of glyphosate application, mouldboard ploughing and non-inverting strategies such as chisel ploughing/rotary harrowing. Post-sowing strategies consist of selective herbicide application (once or twice) and/or mechanical strategies such as harrowing or hoeing. The costs for herbicide application and for machinery are treated as deterministic in the model because farmers know input prices in the moment the weed control decisions are made (see Table A5.2, Appendix 5.C for details).

Whereas Böcker *et al*. (2018) conducted a sensitivity analysis on weed pressure, we now incorporate a distribution of weed pressure in the model, focusing on the distribution of the value describing the difference between the emergence of maize and weeds in a specific year and regional unit, *i.e*. we consider if and to what degree weeds have a growth advantage over maize. Ψ can be positive or negative, depending on the time of maize and weed emergence (see also Fig. 5.2). Here, we consider the period $t = 2006$ to *2015* with yearly, changing values of Ψ depending on the regional unit m. In order to determine this distribution, we make use of the growing degree-day (GDD) concept (McMaster and Wilhelm, 1997) and use spatially and temporally specific information on weather (temperature) and phenology (starting dates of sowing and emergence) data of silage maize[31] for our study region. The weather data is provided by the German Weather Service (*Deutscher Wetterdienst*) from six weather stations in NRW and we assigned each regional unit to the closest weather station.[32] The detailed assessment of the Ψ-values can be found in Appendix 5.A.

[31] Unfortunately, no daily phenological data is available for the different types of weeds. Thus, we suppose emergence at the first of a certain month of a weed's growing period. If a farmer applies successful pre-sowing weed control, the natural emergence patterns of weeds are disrupted. More specifically, we assume that the last weed control measure is done three days before the sowing date, so that the summation of GDD for weeds begins three days before sowing. Of course, if no weed control is done before sowing, also no interruption of weed growth takes place. After weeds are suppressed by pre-sowing measures, they re-emerge after a while. Hence, the summation of GDD starts at that date of re-emergence. Combining GDD models with phenology data were found to be a practical way for describing growing conditions (*e.g*. Dalhaus *et al*., 2018).

[32] If no maize is grown at the weather station directly, as in the case of Düsseldorf, or if phenological data is not available for the municipality of the weather station, information on the phenology from surrounding municipalities was taken.

With regard to the expected attainable yield level, we make use of raster data (1 x 1 km) on water-limited potential yields of silage maize from $\chi = 1999, \ldots, 2011$ that were gratefully provided by Ganga Ram Maharjan and Thomas Gaiser from the Crop Science Group of University of Bonn. The raster data was created by a crop model that is presented and documented in Hoffmann *et al*. (2015) and Zhao *et al*. (2015). This raster data was aggregated to municipality levels. [33] Water-limited yields are chosen as irrigation is basically irrelevant in silage maize production in the region.

The parameters C and I in Eq. (5.2) are taken from Bosnic and Swanton (1997). A is defined as 63.8% which was found to be the maximum potential yield loss in field trials (Böcker *et al*., 2018). The key parameters of the production function in (5.2), α_0, α_1, β_0 and β_1, are estimated by determining those parameter values that minimise the error term between the observed yields and the yields simulated with the control strategies used in current silage maize production (more details can be found in Böcker *et al*., 2018: 184ff., 2017). Expert knowledge from the Chamber of Agriculture of NRW was used in order to get information about the currently used practices of maize cultivation (*ibid*.). In addition, recent Ψ-values were included for the time period 2013 to 2015. The finally estimated parameter values are $\alpha_0 = 1.266$, $\alpha_1 = 0.683$, $\beta_0 = 0.747$ and $\beta_1 = 0.543$.

5.3.2 Pesticide Load

Information about ASs to calculate the PLI is taken from the Pesticide Properties DataBase (Lewis *et al*., 2016) which draws on publicly available sources, such as pesticide admission and regulation procedures. In addition, we obtain complementary information about ASs per product as well as their specific concentration from product specification sheets of the herbicide manufacturers and from herbicide recommendations, such as from the

[33] Böcker *et al*. (2018) instead used average realised yields at the level of 53 regional units, which were used for the municipalities found in one of these units.

Chamber of Agriculture NRW (Landwirtschaftskammer Nordrhein-Westfalen, 2015) and the Bavarian State Research Centre for Agriculture (Bayerische Landesanstalt für Landwirtschaft, 2016). The overall load and the load in the three sub-categories can be found in Table 5.1 for all herbicide products included in our analysis. Note that the selection and weights of the indicators and the indices underlying the calculation of the load values in the PLI reflect the focus on environmental problems and preferences in Denmark (see Appendix 5.D for further details). The PLI covers a broad range of environmental and health effects, it is aligned with European pesticide regulations, is already implemented and has been scientifically tested as a pesticide risk indicator in Denmark for several years (Kudsk *et al.*, 2018). Thus, its application to the German study region seems reasonable, especially as no comparable indicator is available for Germany.

Table 5.1. Values of the Pesticide Load Indicator[a] for the herbicide products included in the model

Herbicide name	Λ_{toxy}	Λ_{fate}	Λ_{heal}	Λ_{total}
Activus	0.109	1.207	0.100	1.416
Arigo	0.215	0.255	0.000	0.471
Arrat	0.064	0.585	0.267	0.916
Aspect	0.199	0.267	0.500	0.966
B 235	0.225	0.003	1.200	1.427
Buctril	0.217	0.002	1.200	1.419
Calaris	0.105	0.220	0.000	0.325
Callisto	0.035	0.041	0.000	0.076
Dash	0.073	0.058	0.675	0.807
Dual Gold	0.116	0.198	0.100	0.414
Elumis	0.048	0.053	0.000	0.101
Gardo Gold	0.086	0.179	0.100	0.366
Laudis	0.014	0.013	0.000	0.027
Lido SC	0.084	0.169	0.150	0.403
MaisTer	0.068	0.014	0.000	0.081
Motivell forte	0.040	0.039	0.000	0.079
Peak	0.707	6.067	0.033	6.808
Roundup PowerFlex	0.024	0.052	0.350	0.426[b]
Spectrum	0.162	0.183	0.000	0.346
Stomp Aqua	0.115	1.269	0.067	1.451
Successor T	0.109	0.129	0.100	0.339
Sulcogan	0.023	0.260	0.800	1.083
Tacco	0.049	0.012	0.000	0.061

[a] The unit is Load per standard treatment. The load values need to be weighted with the application rate and the standard area dose in order to get per hectare values.

[b] The PLI is based on currently available assessments of the environmental and human health effects of pesticides. Thus, glyphosate has modest environmental and health effects among the here listed herbicides (see *e.g*. Gardner and Nelson, 2008).

5.3.3 Energy Balance

Process analysis based on energy balances is a widespread method in agricultural sciences (*e.g*. Deike *et al*., 2008; Jayasundara *et al*., 2014), typically drawing on literature providing general information on energy use in agricultural and industrial processes (*e.g*. Audsley *et al*., 2009; Green, 1987; Hülsbergen *et al*., 2001). For indirect energy consumption of producing and maintaining machinery, Aguilera *et al*. (2017: 341) report for the year 2010 values of $E_{tr} = 156$ MJ/kg of machinery for tractors, $E_{hr} = 102$ MJ/kg for harvesters, $E_{tm} = 72$ MJ/kg for tillage machinery, and $E_{om} = 62$ MJ/kg for other machinery. The gross calorific values of the different production inputs and steps are presented in Table A5.3 (direct energy E_d) and A5.4, Appendix 5.C (material use for indirect energy E_i).

With respect to the energy requirements of herbicide production, we rely on Audsley *et al*. (2009) who, based on data of Green (1987), find that the energy requirements for pesticide production is related to the year of discovery. They fitted the following regression line between the energy requirement of a certain AS (E_{AS}, measured in [MJ/kg]) and the year of discovery as an herbicide (AD_{As}), *i.e*. early developed herbicides require less energy in production:

$$\mathrm{E_{AS}} = -399 + 10.8(\mathrm{AD}_{AS} - 1900). \tag{5.8}$$

Following Jayasundara *et al*. (2014: 83; citing Nagy, 1999), we add 6% to the estimated production requirements to account for formulation, packaging and delivery requirements. The assumed energy requirements used in our analysis are presented in Table 5.2.

Table 5.2. Herbicide active substances[a] in the model and energy requirements for production

Active substance	**Year of discovery (Tomlin, 2006; MacBean, 2012)**	**Estimated production energy in relation to Eq. (5.8) [MJ/kg]**	**Production energy plus 23 [MJ/kg] for formulation, packaging and delivery (Hülsbergen *et al.*, 2001) [MJ/kg]**
Bromoxynil	1963	281	304
Dicamba	1961	260	283
Dimethenamid-P	2000	681	704
Flufenacet	1995	627	650
Foramsulfuron	1995	627	650
Glyphosate	1971	368	391
Iodosulfuron-methyl-sodium	1999	670	693
Mesotrione	1998	659	682
Metosulam	1993	605	628
Nicosulfuron	1990	573	596
Pendimethalin	1974	400	423
Pethoxamid	2001	692	715
Prosulfuron	1993	605	628
Pyridate	1976	422	445
Rimsulfuron	1989	562	585
S-Metolachlor	1996	638	661
Sulcotrione	1991	584	607
Tembotrione (Edenfield and Allen, 2005)	2005	735	758
Terbuthylazine	1966	314	337
Thiencarbazone (Philbrook and Santel, 2007)	2007	757	780
Topramezone	2006	746	769
Tritosulfuron (Schönhammer *et al.*, 2002)	2002	703	726

[a] Please notice the difference between 'herbicide product' (Table 5.1) and 'herbicide active substance'. Herbicide products consist of one or several herbicide active substances plus solvents and adjuvants.

The energy requirements are estimates based on a regression function of Audsley *et al.* (2009) who use values of Green (1987). Thus, these values are only approximations.

5.4 Results

Results are presented as averages over the range of simulation years *T* and over the regional units/municipalities *M*. In the main text, we consider the scenario with moderate risk aversion ($r_a = 0.5$) and the three output price levels of € 4.00, 4.60 and 5.20/dt. The sensitivity analyses with respect to different levels of the the partial risk aversion coefficient and a potential increase of the attainable yield level by 10% are presented in Appendix 5.E. This section first illustrates descriptive results on the weed control strategies used. Following, we present the main results on the trade-offs between economic (yield, gross margin) and environmental consequences (herbicide load and energy process analysis) of a glyphosate ban.

5.4.1 Descriptive Results

Three characteristics mainly influence the choice of pre- and post-sowing weed control strategies in our model: i) the distribution of the gross margin (determined by the output price and depending on the distribution of the attainable yield), ii) the weed occurrence and pressure in a certain region and year, and iii) the soil type influencing the weed control costs. These mechanisms will be described in detail below.

In the base scenario without policy intervention, we find that applying glyphosate is the utility maximising strategy in about 28.5% to 38% of the regional units, depending on the output price (Fig. A5.2, Appendix 5.E). This number is in line with surveys on how many farmers apply glyphosate in Germany (see introduction). A higher price of silage maize, *ceteris paribus*, increases glyphosate application. The major alternative to pre-sowing weed control using glyphosate is mechanical weed control based on two passes of chisel ploughing (or similar machines). Other strategies, such as one pass of chisel ploughing, chisel ploughing in combination with harrowing or conventional tillage with ploughing are less frequent; mouldboard ploughing is not found to be optimal in any scenario. The level of risk aversion only has a minor and insignificant impact on the choice of the pre-sowing and post-sowing weed control strategies.[34] This does not mean that the incorporation of risk effects in our model is obsolete, as the consideration of stochastic weed pressure and attainable yield levels have effects on the optimal decision making of farmers compared to a deterministic approach.

The average number of applied AS/ha per year excluding glyphosate is presented in Table 5.3 and in Fig. A5.4 and Fig. A5.5 in Appendix 5.E. At the low output price level of € 4.00/dt 2.6 AS/ha and year are applied. At the higher output price level of € 5.20/dt, the number increases to 3.0 to 3.1 AS/ha and year on average. At the low price level, nicosulfuron, profulfuron and pyridate have a relatively high share of the applied AS. Terbuthylazine, S-metolachlor, flufenacet, iodosulfuron, foramsulfuron and thiencarbazone gain in importance at higher expected revenues. The model results show that the most frequently applied AS is terbuthylazine, which is in line with observations on farm practices in the case study region (Julius Kühn-Institut, 2017). The use of different ASs is relatively constant over the four different risk aversion levels, with slightly but not significantly lower shares of ASs if a risk affine decision maker is assumed (r_a = -2.0). In case of a glyphosate ban, the composition of the ASs does not change significantly (Table 5.3).

[34] See Fig. 5.3, Fig. A5.2 to Fig. A5.5, Appendix E, for more details.

Table 5.3. Average applied active substances over *m* and *t* [AS/ha and year] post-sowing for both the glyphosate licensed and glyphosate banned scenario (r_a = 0.5)

r_a = 0.5	**Glyphosate licensed scenarios**			**Glyphosate ban scenarios**		
Active Substance:	**€ 4.00/dt**	**€ 4.60/dt**	**€ 5.20/dt**	**€ 4.00/dt**	**€ 4.60/dt**	**€ 5.20/dt**
Flufenacet	0.123	0.176	0.211	0.122	0.174	0.211
Foramsulfuron	0.127	0.178	0.212	0.126	0.176	0.211
Iodosulfuron	0.127	0.178	0.212	0.126	0.176	0.211
Mesotrione	0.411	0.452	0.474	0.410	0.452	0.473
Nicosulfuron	0.371	0.322	0.288	0.372	0.323	0.288
Pethoxamid	0.030	0.062	0.080	0.031	0.062	0.080
Prosulfuron	0.371	0.322	0.288	0.372	0.323	0.288
Pyridate	0.245	0.222	0.206	0.244	0.222	0.207
S-Metolachlor	0.136	0.169	0.187	0.136	0.168	0.187
Terbuthylazine	0.534	0.628	0.685	0.533	0.626	0.684
Thiencarbazone	0.127	0.178	0.212	0.126	0.176	0.211
Others*	0.004	0.001	0.001	0.004	0.001	0.001
Sum	*2.607*	*2.887*	*3.054*	*2.602*	*2.880*	*3.052*

* Other active substances that are applied in the model are dicamba, tembotrione and tritosulfuron.

5.4.2 Economic Consequences

We find that expected revenues (*i.e.* expected output price levels and the expected attainable yield level) are the main drivers of weed control expenditures and the applied AS (Fig. 5.3, Table 5.3, Fig. A5.2–Fig. A5.5, Appendix 5.E). More specifically, the average costs of the optimal weed control strategies increase with higher expected revenues. For example, at lower levels of expected revenues, cheap mechanical weed control which has a low damage control is more frequently used compared to glyphosate even in the baseline scenario without a ban. This is due to lower sowing costs after two passes of chisel ploughing compared to glyphosate application and direct sowing. If glyphosate would be banned, costs for herbicides and herbicide's application decrease significantly at all three output price levels, but we observe a significant but relatively small increase in total weed control costs regardless of the output price. Note that this mainly results from higher costs for mechanical weed control (significant at the 0.01-level), which is the main substitute for glyphosate. The enforced change in weed control also affects the yield level, it would therefore be possible that both revenues and costs increase, but clearly not their difference because such a choice would be an optimum in the reference scenario. In most geographical units, we find a reduction of the yield and consequently of the turnover. This yield reduction is described in the energy output *EO* in section 5.4.4 in detail. The average expected gross margin varies between € 555/ha € 981/ha depending on the output price (without direct payments). On average, this is reduced by € 2 – 3/ha, but in single regional units also higher losses occur of about € 10/ha. The reduction is not only small, but also not statistically significant (Fig. 5.4).

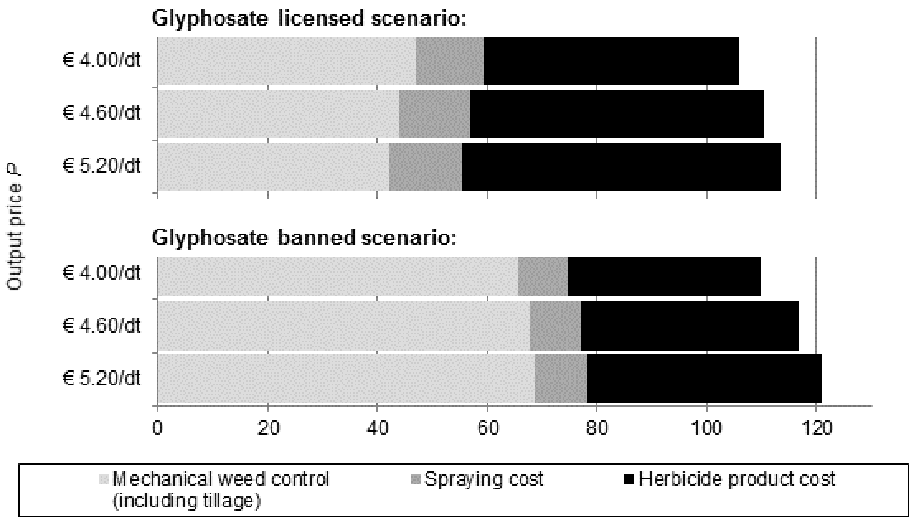

Fig. 5.3. Average costs for pre- and post-sowing weed control (r_a = 0.5; all regional units).

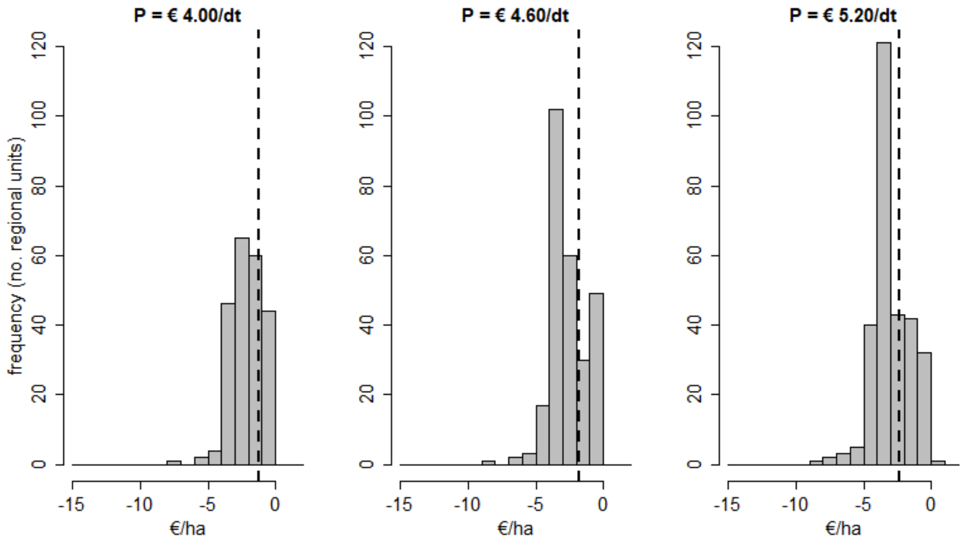

Fig. 5.4. Frequency distributions of the reduction of the gross margin per ha if glyphosate is banned ($r_a = 0.5$).

5.4.3 *Pesticide Load Analysis*

The frequency distribution of the average pesticide load indicator values for the optimal strategies in the different geographical units is illustrated in Fig. 5.5 under moderate risk aversion ($r_a = 0.5$) and the glyphosate licensed-scenario. Pesticide load values differ strongly between the regional units and range from about 0.5 to 3.8 Load/ha. Differences are especially large for the environmental fate load, which is due to high Load values of herbicides containing prosulfuron (*e.g.* contained in the product “Peak”). Load values increase in output prices and expected attainable yield levels. The levels are higher than the average herbicide load for a hectare of maize in Denmark (where the PLI is used in policy analysis and documented), which could be due to higher taxation of products with high load values (Böcker and Finger, 2016).[35]

The analysis of the four different hypotheses regarding the pesticide load is presented in Table 5.4 for $r_a = 0.5$ and Table A5.5, Appendix 5.E, for

[35] For example, the PLI was 0.31 in 2014 (Ørum and Hossy, 2015: 57) and 0.38 in 2015 (Ørum and Sommer Holtze, 2017: 54) (values excluding glyphosate application).

all risk aversion coefficients. We find significant load reductions under a glyphosate ban in all scenarios because chemical weed control is substituted by mechanical control. This holds for the total load indicator, as well as all sub-indicators (*i.e*. environmental toxicity, fate and human health). The decrease is strongest with respect to the human health load. The sub-indicators for environmental fate and toxicity show lower reductions, reflecting the low environmental load of glyphosate-based products (Table 5.3).

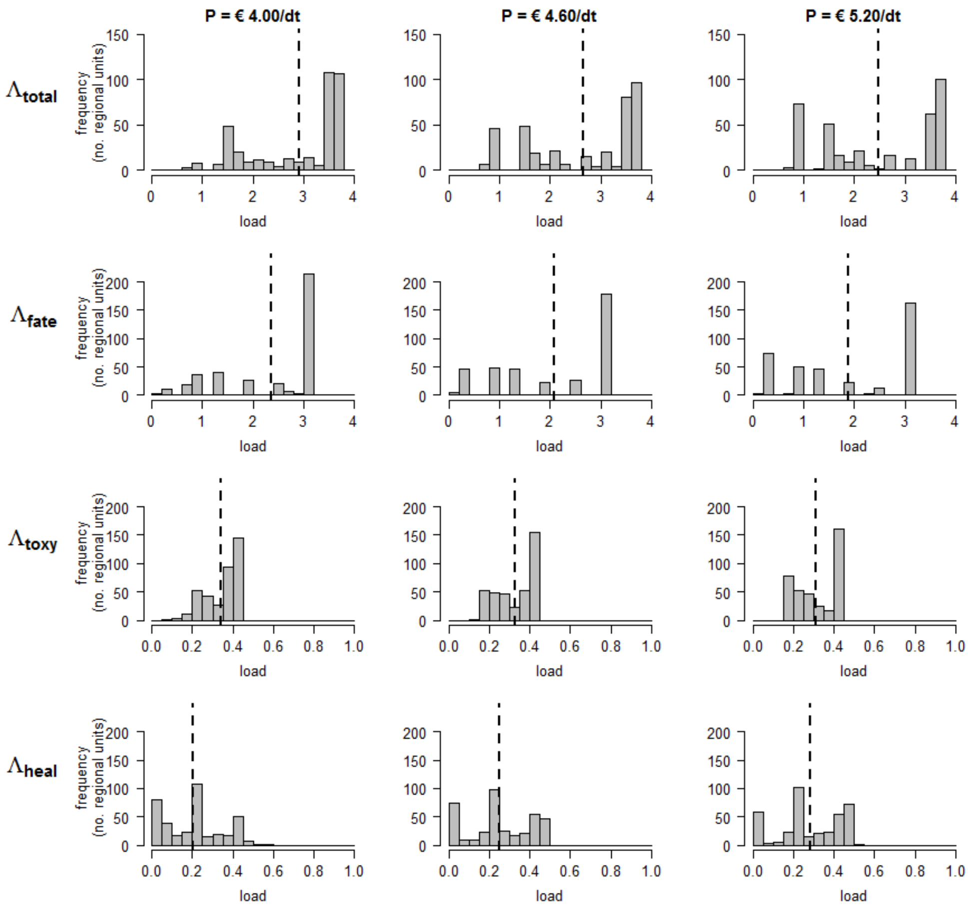

Fig. 5.5. Frequency distribution of pesticide loads w.r.t. total (Λ_{total}), environmental toxicity (Λ_{toxy}) and fate (Λ_{fate}) as well as human health (Λ_{heal}) (over all herbicides) as average over the period 2006-2015 ($r_a = 0.5$; glyphosate licensed scenario).

The spatial distribution of the total potential load reductions in case of a glyphosate ban is presented in Fig. 5.6 (see Fig. A5.6, Appendix 5.E, for the energy consumption plotted against the pesticide load). The load reduction is highest in two types of regions, in which glyphosate is more likely to be used. These are firstly regions with heavy soils (*i.e.* high shares of clay) where mechanical alternatives to glyphosate require higher traction power and thus are more expensive. Here, glyphosate application is under current conditions often the optimal weed control strategy. Secondly, high load reductions are found in regions dominated by high shares of sandy soils where direct tillage or strip-till practices in combination with glyphosate application is relatively cheap. In contrast, on medium soils (*i.e.* with balanced mixtures of clay, silt and sand), glyphosate is applied less frequently in our model. Changes in the expected output price level have small and insignificant impacts on the pesticide load reduction per hectare. However, since we find a glyphosate application optimal in more regional units at higher maize price levels, higher prices would also imply more widespread load reductions under a ban.

Table 5.4. Average absolute change *Δ* in Load/ha by a glyphosate ban in silage maize production (standard deviation in brackets, relative changes in italics)[a,b]

Load: **P [€/dt]:**	$\Delta\Lambda_{toxy}$	$\Delta\Lambda_{fate}$	$\Delta\Lambda_{heal}$	$\Delta\Lambda_{total}$
4.00	-0.011 (0.006) *** *-3.9%*	-0.019 (0.061) *** *-1.4%*	-0.170 (0.023) *** *-62.5%*	-0.200 (0.069) *** *-9.3%*
4.60	-0.011 (0.008) *** *-4.1%*	-0.011 (0.091) *** *-1.3%*	-0.175 (0.015) *** *-58.5%*	-0.196 (0.092) *** *-11.1%*
5.20	-0.012 (0.004) *** *-4.6%*	-0.025 (0.043) *** *-3.3%*	-0.174 (0.011) *** *-56.0%*	-0.211 (0.045) *** *-12.9%*

[a] Difference of Load/ha for the glyphosate licensed scenarios minus the glyphosate banned scenarios. Only municipalities are included in which glyphosate is used in the model under the licensed scenario ($r_a = 0.5$).

[b] *, ** and *** denote significance at the 5%, 1% and 0.1% level respectively based on Wilcoxon-Mann-Whitney-tests.

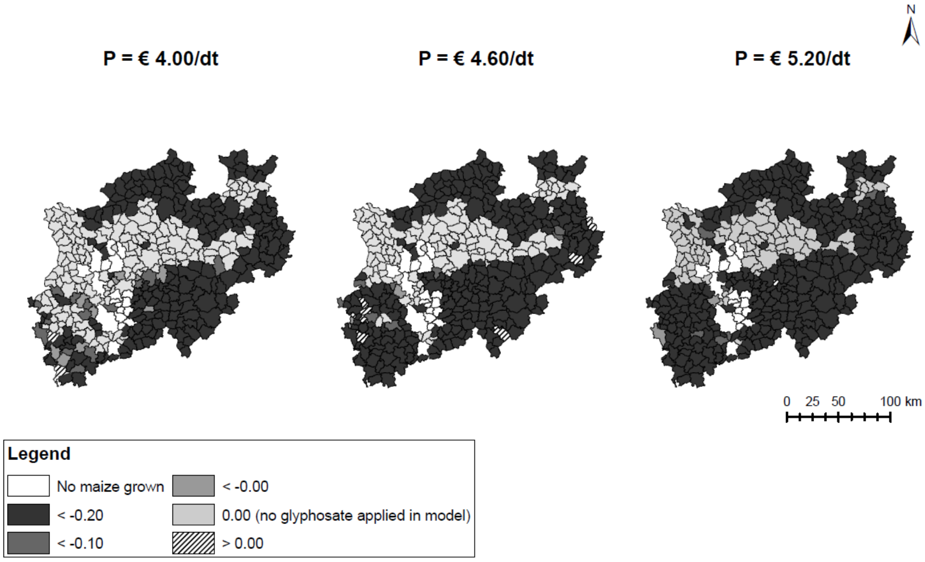

Fig. 5.6. Potential reductions of the total pesticide load for the analysed municipalities under a ban of glyphosate (r_a = 0.5).

5.4.4 Energy Process Analysis

The frequency distribution and the mean values of the energy output and input are presented in Fig. 5.7 for the risk aversion coefficient of 0.5 in the glyphosate licensed-scenario. The expected energy output *EO* across the regional units ranges from 141 GJ/ha to 278 GJ/ha, the energy input ranges from 2.7 to 12.2 GJ/ha. We find higher output prices to increase energy outputs because of increased weed control and consequently higher energy inputs, but also due to the fact that higher maize prices incentivise the use of other yield increasing inputs such as fertiliser. Reflecting slightly decreasing marginal returns, the energy efficiency is highest in the low price-scenario with a ratio of 26.2 compared to 25.3 in the high price-scenario (Fig. 5.7). Similar trends can be observed under other risk aversion coefficients and higher expected attainable yield (not shown).

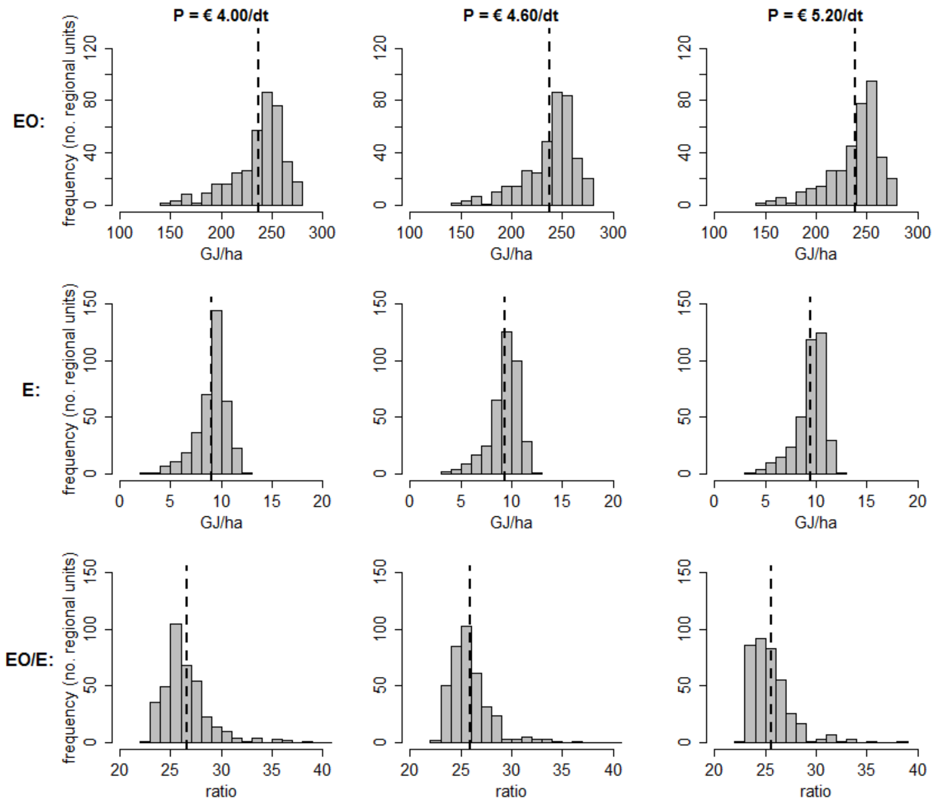

Fig. 5.7. Distribution of energy output, energy input and energy efficiency as average over the period 2006-2015 (r_a = 0.5, glyphosate licensed-scenario). Dashed lines show mean values.

Table 5.5 presents results differences for the energy indicators between the glyphosate licensed and the glyphosate banned-scenarios for the three output price levels. Regarding the energy output *EO*, we observe in most scenarios only a small reduction by about 0.65 – 85 GJ due to lower yields (around -1 dt/ha). However, this difference in *EO* is overall not significant. Similar results are found for net energy output *NEO*. Direct energy use E_d increases significantly by about 300 MJ/ha due to a glyphosate ban. Indirect energy consumption E_i tends to decrease, but differences are not significant. The change in the total energy use (ΔE) is positive, ranging from +91 to +167 MJ/ha. Finally, we find that a glyphosate ban decreases energy efficiency ($\Delta EO/E$), reflecting that mostly mechanical weed control is used as a substitute. This decrease is small and insignificant for the low output price level of € 4.00/dt, but larger and highly significant for the higher price levels of € 4.60/dt and € 5.20/dt.

Table 5.5. Absolute average change *Δ* in energy output and input per hectare by a glyphosate ban in silage maize production (standard deviation in brackets, relative changes in italics)[a,b]

Energy: **P [€/dt]:**	**ΔEO[c] [MJ/ha]**	**ΔNEO [MJ/ha]**	**ΔE_d [MJ/ha]**	**ΔE_i [MJ/ha]**	**ΔE [MJ/ha]**	**ΔEO/E**
4.00	-716 (667)	-783 (695)	+277 (51) ***	-181 (59)	+91 (64)	-0.272 (0.282) ‘
	-0.3%	*-0.4%*	*+28.7%*	*-2.8%*	*+1.1%*	*-1.0%*
4.60	-849 (436)	-981 (399)	+300 (40) ***	-159 (58)	+137 (70)	-0.523 (0.282) ***
	-0.4%	*-0.4%*	*+29.9%*	*-2.2%*	*+1.6%*	*-1.9%*
5.20	-648 (364)	-814 (327)	+311 (31) ***	-140 (45)	+167 (65) ‘	-0.564 (0.320) ***
	-0.3%	*-0.4%*	*+30.4%*	*-1.8%*	*+2.0%*	*-2.1%*

[a] Difference of process analysis for the glyphosate licensed scenarios minus the glyphosate banned scenarios: energy output (*EO*), net energy output (*NEO*), direct energy use (E_d), indirect energy use (E_i), total energy use (*E*) and energy efficiency (*EO/E*). Only municipalities are included in which glyphosate is used in the model under the licensed scenario (r_a = 0.5).

[b] ‘, *, ** and *** denote significance at the 10%, 5%, 1% and 0.1% level respectively, based on Wilcoxon-Mann-Whitney-tests.

[c] One deciton of silage maize has an energy content of 671.4 MJ in the model.

Fig. 5.8 includes maps of the increase in direct energy consumption (E_d) and the reduction in energy efficiency (*EO/E*) (see Fig. A5.6, Appendix 5.E, for the scatterplot). The emerging picture resembles the one for the PLI. We find relatively strong increases on heavy soils where mechanical strategies require high energy input, but also increases on light soils due to current application of no till or strip-till practices in combination with glyphosate. The trends of changes are similar in most of the regional units. Energy efficiency gains can only be observed in very few locations.

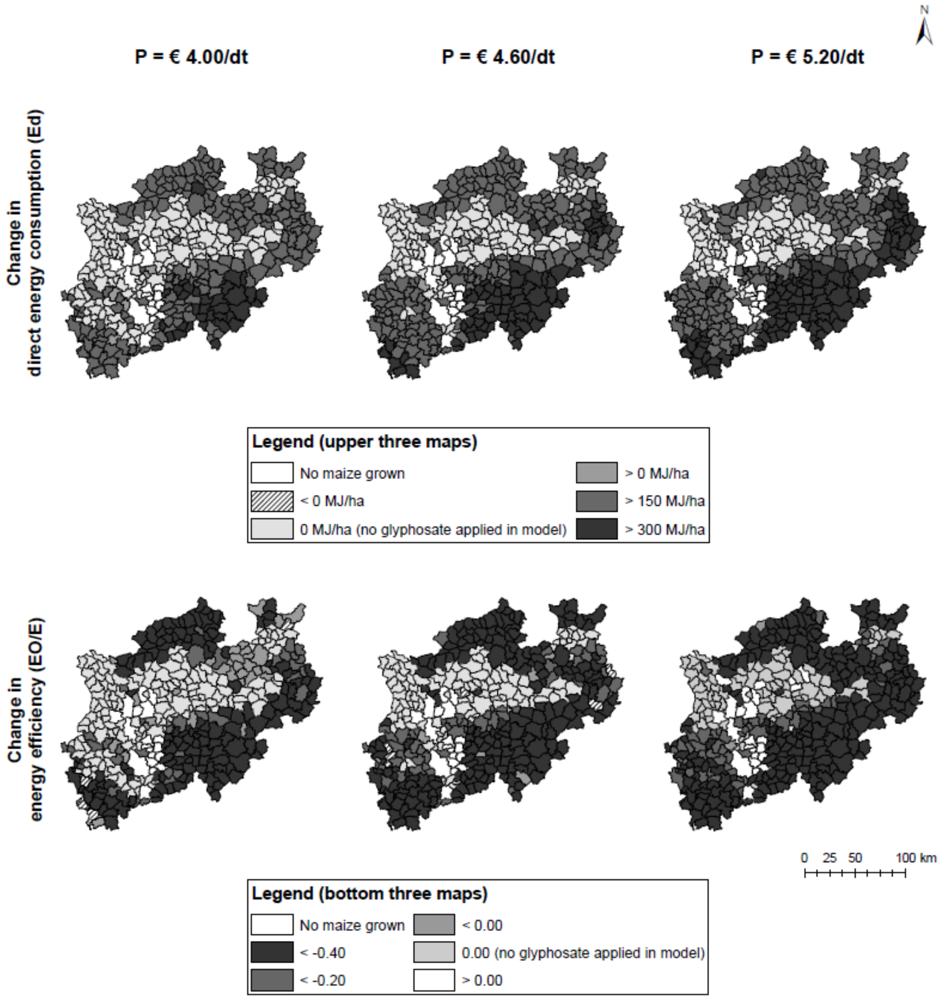

Fig. 5.8. Spatial distribution of increase in direct energy consumption (E_d) and of reduction in energy efficiency (*EO/E*) over the state of NRW at a potential ban of glyphosate (r_a = 0.5).

5.5 Discussion

We find that a glyphosate ban would lead to a significant, but relatively small increase in weed control costs, independently of expected output prices, stemming mostly from increases in more expensive mechanical weed control measures, which outweigh lower herbicide expenditures. Furthermore, we find reduced yields such that a glyphosate ban leads to lower gross margins, on average by € 2 – 3/ha and in some cases by up to € 10/ha. The small change also reflects cost savings in sowing when switching from direct sowing – used mainly in combination in glyphosate – to mulch sowing. This income loss is lower than what was found in other studies. For example, Kehlenbeck *et al*. (2015) found potential losses of € -88 to +22/ha, depending on the scenario, but they also included unprofitable strategies in their calculations. Of course, results depend strongly on cost assumptions, which reflect the assumed machinery and plot size. Larger machinery and plot sizes reduce the per-hectare-costs of weed control and tend to increase the attractiveness of mechanical weed control compared to chemical one. Moreover, reduced herbicide prices decrease the usage of mechanical strategies. Nevertheless, the fact that under current conditions only one third of the farmers apply glyphosate in silage maize production in Germany (Julius Kühn-Institut, 2017), a share close fitting to what is simulated by our model on average over the municipalities under current conditions, supports the result of our analysis that mechanical strategies are a feasible alternative to glyphosate application.

Our results thus indicate no large economic implications of a glyphosate ban in silage maize production in North Rhine-Westphalia. However, our findings point to a trade-off between reducing adverse effects of pesticides on human health and the environment, and on energy consumption as a driver of climate change.

In our case study, we find an average increase of total energy consumption of up to 170 MJ/ha and year, but the change depends strongly on expected output prices. If glyphosate is assumed to be used on 66,500 ha of silage maize in NRW (~190.000 ha silage maize according to Information und Technik Nordrhein-Westfalen, 2016, and 35% of farmers applying glyphosate; Julius Kühn-Institut, 2017), our analysis indicates that the energy demand of silage maize cultivation would increase by 11 TJ in case of a glyphosate ban. Relative to the total final energy consumption of agriculture in the state of NRW (IWR, 2018), this would indicate an increase in energy demand of 0.03% (assuming that 2.2% of the total final energy consumption is consumed by agriculture; Eurostat, 2018). The climate change impact of this additional energy consumption depends on energy sourcing and production. Indirect energy, *i.e*. for factor production, is consumed for the most part in form of electricity; direct energy relates to diesel use. The model also simulates a loss in energy efficiency if glyphosate would be banned, also due to somewhat lower yields. This means that more silage maize will have to be cultivated in case of a glyphosate ban to meet the demand.

We find a significant decrease in the overall toxicity expressed in Pesticide Load Indicator units of, on average, 0.2 Load units (11%) per hectare. Yet, pesticide load levels (per ha) in maize production before and after the ban are low compared with load levels in other crops such as potatoes or vegetables.[36]

The level of risk aversion has only a small influence on the choice of weed control strategies. Note that this does not imply that the introduction of risks in our model was not important for outcomes. It rather reflects the importance of observing the level of weed infestation on the field. Farmers are thus able to apply stage-contingent weed control. This finding is also in line with the ambiguous findings on the risk effects of pesticides in the literature (*e.g*. Möhring *et al*., 2017). A major source of uncertainty is the level of the attainable yield, which leads to a stochastic marginal value product of weed control. This can even create incentives for risk averse decision makers to use less than profit maximising levels of herbicides (*e.g*. Horowitz and Lichtenberg, 1993). Alternatives to the EU framework used by us could provide additional insights. For example, Carpentier (2017) recently applied

[36] For example, the PLI in potato production in Denmark was over all types of pesticides 2.48 Load/ha in 2014 and 6.75 Load/ha in 2015 (Ørum and Hossy, 2015; Ørum and Sommer Holtze, 2017). Vegetables even had higher Load/ha of 6.54 in 2014 and 8.27 in 2015 (Ørum and Hossy, 2015; Ørum and Sommer Holtze, 2017).

prospect theory to pesticide application, to explicitly consider farmer's reference situation to the protected or the unprotected crop.

Our results relate to short- to mid-term, and not long-term changes in weed control strategies in silage maize cultivation under a glyphosate ban. Furthermore, extensive margin and farm effects such as changes in crop rotation or other adjustments at farm level were not included so far, but could be addressed in future research. However, extensive margin and farm effects of a glyphosate ban in cultivation of maize in our study area are likely to be quite limited for three major reasons. Firstly, the simulated changes in gross margins are quite small and glyphosate use in other important crops is even lower than in maize, rendering significant adjustments in cropping patterns under a ban unlikely. Secondly, conservation tillage, *i.e.* non-inverting tillage, without applying glyphosate is feasible in silage maize production. Due to high investment cost for specialised machinery, a problem in increased conservation agriculture without glyphosate could arise, if the necessary machinery is neither already owned by the farmer nor contractors can be found. In our case study region, however, the density of contractors is quite high, so that the problem should not be applicable. Thirdly, total maize production is unlikely to change much as policy measures that promote biogas production from silage maize generate a stable and high demand (Information und Technik Nordrhein-Westfalen, 2017). The slight yield reduction on areas that currently apply glyphosate would decrease feedstock availability for the existing biogas plants facing large sunk cost and thus further intensify the recently observed price increases for silage maize. That would act against the incentive to reduce the maize acreage in response to the slightly decreased gross margin under the ban. Although the short-term economic consequences would be rather small, further research should address possible unwanted ecological and socio-economic side-effects (*e.g.* via changes in land use or trade-patterns) occur in case of bans if pesticides. Stricter regulatory measures or economic incentives to reduce glyphosate application (*e.g.* Finger *et al.*, 2017) are for this reason more suitable than bans. Our results show clear trade-offs and provide a quantitative basis for political debates on glyphosate and reveal the need for flexible and tailor-made agri-environmental policy solutions. This could for example comprise solutions such as pesticide taxation (Finger *et al*., 2017). Pesticide taxation incentivises those farmers to reduce glyphosate use, which are able to substitute glyphosate with little gross margin reduction, while still enabling glyphosate use for those facing higher reductions. Similarly, farmers, which need large amounts of energy to substitute glyphosate use, should be able to further apply glyphosate, while those, which face lower increases in energy use, should be incentivised to reduce glyphosate applications.

Regarding the weed control implementation in the model, additional temporal dimensions could be introduced. More specifically, temporal interdependencies of applied weed control strategies could be considered, *e.g*. if weed pressure spills over different periods. For example, if only conservation tillage strategies without glyphosate application are chosen, the size of the weed seed bank growths (Bàrberi *et al*., 1998). In addition, it could be interesting to include a behavioural algorithm relating to the choice of weed control strategies (*cf*. Hüllermeier, 2005) and farmers experiences. Furthermore, other environmental dimensions could be included in the model. For example, no till practices – possibly economically viable only if glyphosate is licensed – reduce soil erosion especially on hilly grounds (Montgomery, 2007). Moreover, we do not take into account that a change to mechanical weed control maybe has negative effects on ecological soil conditions. Along these lines, problems with nitrogen surpluses might increase due to slightly lower yield levels if glyphosate is banned and manure application is not adjusted. New technologies could be a game changer for the future role of pesticides in agriculture (*e.g*. Finger, 2018). For example, autonomous weeding robots might allow to reduce pesticide use dramatically without facing trade-offs of high emissions or soil compaction (*e.g*. Walter *et al*., 2017). Thus, future modelling attempts shall also incorporate new technological options.

5.6 Conclusions

We develop and employ a detailed bio-economic model focusing on weed damage control in order to analyse the potential effects of a glyphosate ban. The model simulates the optimal choice from a larger set of pre- and post-sowing weed control strategies in a case study for silage maize production of 377 municipalities in North Rhine-Westphalia, Germany. We consider production risks and farmers' risk preferences in an expected utility framework based on a damage control approach for weed occurrence in cultivation of silage maize. The main finding from our analysis is that a glyphosate ban would cause a shift towards more mechanical weed control measures, but not to more pronounced use of selective herbicides. Economic impacts on maize yields and gross margins are small. The weed control strategy set chosen in response to a glyphosate ban is less toxic as expressed with the Pesticide Load Indicator, but more energy intensive based on a detailed energy process analysis. The magnitude of these effects is found to be critically dependent on output price levels and yield expectations. Thirteen different hypotheses are tested with regard to a glyphosate ban (see Table 5.6 for summary results). Our analysis thus quantifies in detail trade-offs between different policy goals and can inform policy debates, as well as regional and private initiatives for alternatives to glyphosate use. Furthermore, the here developed modelling approach could be an important starting point for the assessment of economic and environmental trade-offs in future debates on restrictions of pesticide use.

Table 5.6. Summary of the thirteen tested hypotheses related to a glyphosate ban

Analysis	**Hypothesis**	**Direction/ prefix of results**	**Significance**
Economic change	1) Weed control costs increase	Increase (+)	Significant
	2) The yield decreases	Decrease (–)	Not significant
	3) The gross margin decreases	Decrease (–)	Not significant
Pesticide load	4) The toxicity load decreases ($Λ_{toxy}$)	Decrease (–)	Significant
	5) The environmental fate load decreases ($Λ_{fate}$)	Decrease (–)	Significant
	6) The human health load decreases ($Λ_{heal}$)	Decrease (–)	Significant
	7) The load of applied herbicides decreases ($Λ_{total}$)	Decrease (–)	Significant
Energy balance	8) The energy output *EO* decreases	Decrease (–)	Not significant
	9) The net energy output *NEO* decreases	Decrease (–)	Not significant
	10) More direct energy is used (E_d increases)	Increase (+)	Significant
	11) More indirect energy is used (E_i increases)	Decrease (–)	Not significant
	12) More energy is used in general (E increases)	Increase (+)	Mostly not significant
	13) The energy efficiency decreases (EO/E)	Decrease (–)	Significant

Acknowledgements

We thank Ganga Ram Maharjan and Thomas Gaiser from the Crop Science Group of University of Bonn for providing yield data for this research. Furthermore, we thank the plant protection consultants of the Chamber of Agriculture of NRW and of the Chamber of Agriculture of Lower Saxony for providing valuable information supporting this research.

5.7 Appendix

Appendix 5.A

In addition to the phenology data of maize, we use information on the time of emergence for each weed *i*. However, information on this emergence data is not available on a daily but only monthly resolution (Mehrtens, 2006; Otte, 1990; see Fig. A5.1 and Table A5.1). Thus, we suppose emergence at the first of the indicated month. If a farmer applies successful pre-sowing weed control, the natural emergence pattern of weeds is disrupted. More specifically, we assume that the last weed control measure is done three days before the sowing date, so that the summation of growing-degree days (GDD) for weeds begins three days before sowing. Of course, if no weed control is done before sowing, also no interruption of weed growth takes place. After weeds are suppressed by pre-sowing measures, they re-emerge after a while. Hence, the summation of growing degree-days starts at that date of re-emergence.

To estimate the date of re-emergence, we use data by Vleeshouwers (1997, see Table A5.1). First, we calculate a *GDD*-value for each weather station *ws* and for each weed *i* that emerges earlier than silage maize:

$$\mathrm{GDD}_{\mathrm{ws,i,t}} = \begin{cases} 0 & \forall\left(\bar{\tau}_{\mathrm{ws,t}} - \mathrm{T}_{\mathrm{B_i}}\right) < 0 \\ \sum_{\mathrm{d=1}}^{\mathrm{m.em}} \left(\bar{\tau}_{\mathrm{ws,t}} - \mathrm{T}_{\mathrm{B_i}}\right) & \forall\left(\bar{\tau}_{\mathrm{ws,t}} - \mathrm{T}_{\mathrm{B_i}}\right) \geq 0 \end{cases} \tag{A5.1}$$

$\bar{\tau}_{ws,t}$ is in this case the average daily temperature per weather station in year t and T_{B_i} is the base temperature for each weed (the temperature at which seeds germinate). The summation of GDD starts at the first day of the reported month and ends at the time of maize emergence *m.em*. If weeds emerge later than maize, *i.e.* cases in which maize has a growth advantage, Eq. (A5.1) becomes:

$$\mathrm{GDD}^{*}_{\mathrm{ws,i,t}} = \begin{cases} 0 & \forall \left(\bar{\tau}_{\mathrm{ws,t}} - \mathrm{T}_{\mathrm{B}_{\mathrm{maize}}}\right) < 0 \\ \sum_{\mathrm{d=m.em}}^{\mathrm{i.em}} \left(\bar{\tau}_{\mathrm{ws,t}} - \mathrm{T}_{\mathrm{B}_{\mathrm{maize}}}\right) & \forall \left(\bar{\tau}_{\mathrm{ws,t}} - \mathrm{T}_{\mathrm{B}_{\mathrm{maize}}}\right) \geq 0 \end{cases} \tag{A5.2}$$

In this case, $T_{B_{maize}}$ is the base temperature of maize (we use a value of 6° C; Lewandowski and Böhmel, 2009: 132) and the summation starts at the day of maize emergence *m.em* and ends at the day of weed emergence *i.em*. In a second step, the GDDs of weeds interrupted in growth by pre-sowing weed control are summed again under the condition that $GDD_{ws,i,t} \geq \theta_i$, where θ_i symbolises the necessary GDD sum until re-emergence (see Table A5.1):

$$\overrightarrow{\mathrm{GDD}}_{\mathrm{ws,i,t}} = \begin{cases} 0 & \forall \, \mathrm{GDD}_{\mathrm{ws,i,t}} < \theta_{\mathrm{i}} \\ \sum_{\mathrm{d=1}}^{\mathrm{m.em}} \left(\bar{\tau}_{\mathrm{ws,t}} - \mathrm{T}_{\mathrm{B_i}}\right) & \forall \, \mathrm{GDD}_{\mathrm{ws,i,t}} \geq \theta_{\mathrm{i}} \end{cases} \tag{A5.3}$$

Here, $\overrightarrow{GDD}_{ws,i,t}$ symbolises the modified sum of the GDD. Of course, maize can also get an advantage in emergence compared to weeds (Fig. A5.1). In this case, the base temperature of the weeds turns to the base temperature of maize $T_{B_{maize}}$ and the GDD sums up until the weeds emerge (*w.em*), *i.e*. the necessary GDD sum until re-emergence (θ_i) is reached:

$$\overrightarrow{\mathrm{GDD}}^{*}_{\mathrm{ws,i,t}} = \begin{cases} 0 & \forall \, \mathrm{GDD}_{\mathrm{ws,i,t}} < \theta_{\mathrm{i}} \\ \sum_{\mathrm{d=1}}^{\mathrm{w.\ em}} \left(\bar{\tau}_{\mathrm{ws,t}} - \mathrm{T}_{\mathrm{B}_{\mathrm{maize}}}\right) & \forall \, \mathrm{GDD}_{\mathrm{ws,i,t}} \geq \theta_{\mathrm{i}} \end{cases} \tag{A5.4}$$

After assigning the values of $GDD_{ws,i,t}$, $GDD^{*}_{ws,i,t}$, $\overrightarrow{GDD}_{ws,i,t}$ and $\overrightarrow{GDD}^{*}_{ws,i,t}$ to the different geographical units *m* depending on the weather stations *ws* (the index *ws* becomes *m*), we calculate in a third step one *Ψ*-value for each geographical unit and year *t*. We use the weighted potential damage of each weed *i* in each geographical unit *m* for this:

$$\Psi_{m,t} = \sum_{i=1}^{32}\left(\left(\frac{w_{m,i} \cdot a_i}{\sum_{i=1}^{32}\left(w_{m,i} \cdot a_i\right)}\right) \cdot GDD_{m,i,t}\right) \tag{A5.5}$$

$GDD_{m,i,t}$ is in this case the dedicated GDD-value from Eq. (A5.1) to (A5.5) (see also Fig. A5.1).

Table A5.1. Starting days of emergence, base temperature T_B and growing degree-days (GDD) to re-emerge after weed control for different weeds varieties in the model

Weed variety *i*	**Starting day of emergence (1st of …)[a]**	**GDD to re-emergence after pre-sowing weed control[b]**	**Base temperature T_B**	**Reference of T_B**
<u>Grass-weeds:</u>				
Alopecurus myosuroides	April	88.2	2.5	Storkey (2004)
Digitaria ischaemum	June	45.0	12.0	Fidanza *et al*. (1996)
Echinochloa crus-galli	May	46.3	11.7	Martinkova *et al*. (2006)
Elymus repens/Elytrigia repens	April	85.9	3.0	Bay. Landesanstalt für Landwirtschaft (2017)
Poa annua, P. trivialis	April	92.8	1.5	Storkey (2004)
Setaria viridis	June	67.7	7.0	Lauer (1953)
<u>Broad-leaved weeds:</u>				
Amaranthus retroflexus	May	49.5	11.0	Benvenuti and Macchia (1993)
Atriplex patula	May	99.6	0.0	Nurse *et al*. (2008)
Brassica napus	March	95.9	0.8 (0.4–1.2)	Vigil *et al*. (1996)
Capsella bursa-pastoris	March	95.0	1.0	Storkey (2004)
Chenopodium spp.	March	101.5	1.0	Storkey (2004)
Cirsium arvense	April	81.4	4.0	Liew *et al*. (2012)
Convolvulus arvensis	April	76.8	5.0	Bay. Landesanstalt für Landwirtschaft (2017)
Equisetum arvense, E. palustre	April	83.7	3.5	Assumption (no data found) based on Bay. Landesanstalt für Landwirtschaft (2017)
Fallopia convolvulus	April	81.4	4.0	Storkey (2004)
Fumaria officinalis	April	83.7	3.5 (2.0–5.0)	Lauer (1953)
Galinsoga parviflora	May	67.7	7.0	Lauer (1953)
Galium aparine	March	83.7	3.5	Storkey (2004)

Table A5.1. *(continued)*

Weed variety *i*	**Starting day of emergence (1st of …)**[a]	**GDD to re-emergence after pre-sowing weed control**[b]	**Base temperature T_B**	**Reference of T_B**
Geranium pusillum	April	76.8	5.0	Lauer (1953); Gehring and Thyssen (2011)
Lamium spp.	April	90.5	2.0	Bay. Landesanstalt für Landwirtschaft (2017)
Matricaria spp.	March	90.5	2.0	Storkey (2004)
Mercurialis annua	May	67.7	7.0	Lauer (1953)
Persicaria lapathifolia	May	78.1	3.5 (2.0–5.0)	Lauer (1953)
Persicaria maculosa	May	85.0	3.2	Vleeshouwers and Kropff (2000)
Polygonum aviculare agg.	April	97.3	0.5	Storkey (2004)
Rumex obtusifolius	May	61.8	8.3	Benvenuti *et al*. (2001)
Solanum nigrum	May	61.3	8.4	Benvenuti and Macchia (1993)
Sonchus spp.	May	81.4	4.0	Liew *et al*. (2012)
Stellaria media agg.	April	81.4	4.0	Storkey (2004)
Thlaspi arvense	April	95.0	1.0	Wehsarg (1918) cited from Lauer (1953)
Veronica spp.	March	85.9	3.0	Lauer (1953); Storkey (2004)
Viola arvensis	March	76.8	5.0	Lauer (1953); Gehring and Thyssen (2011)

[a] Exact data for emergence dates are not available. We estimate the dates according to Mehrtens (2006) and Otte (1990).
[b] The values are estimates based on data of Vleeshouwers (1997), who calculated the growing degree-days for three weeds until emergence after the day of cultivation, dependent on the base temperature of each weed. With this information, we derive a linear function of the form $\theta = 4.553 \cdot T_B + 99.589$, where θ symbolises the sum until re-emergence and T_B is the base temperature of the weeds.

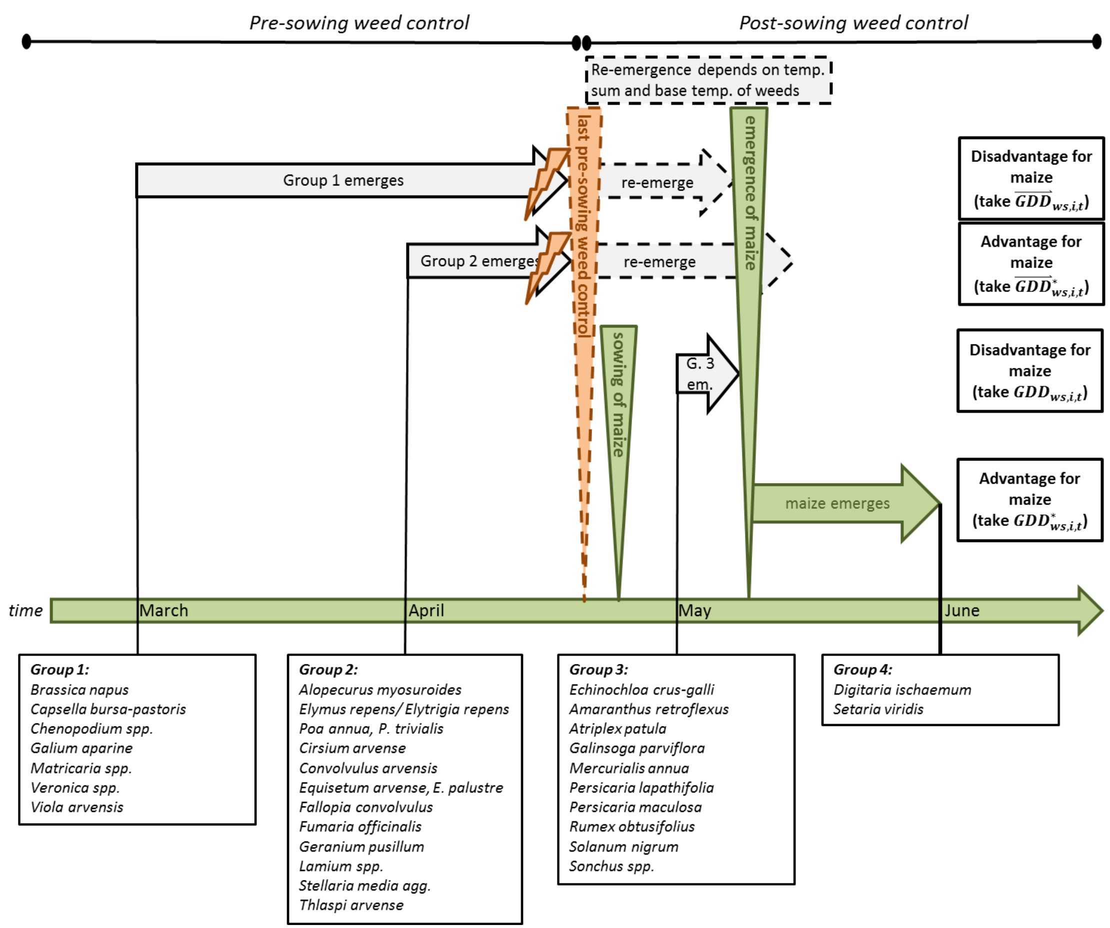

Fig. A5.1. Schematic representation of the calculation of the time of maize emergence compared to weeds (Ψ-values in the production function). Maize emerges on different dates, depending on the weather station's information and the year, and also weeds emerge in different months. Depending on the dates, either weeds or maize has a growth advantage. If weeds already emerged before maize and a weed control activity is done, the growth and the advantage is interrupted. But, weeds re-emerge after a while depending on the necessary growing degree-days and the base temperature to emerge.

Appendix 5.B

Weed control strategies included in the model and related parameters (amount/ha, costs, efficacy)

Excel file, will be available online, see also Böcker *et al*. (2017, 2018)

Appendix 5.C

Table A5.2. Machinery costs and other inputs related to maize growing

Activity	Sub-activity	Work hours (h/ha)	Fix and variable machinery costs	Other inputs
		Weed control-related activities:		
Chisel plough/Cultivator (4.5m)		0.44	24.17	
Mouldboard plough and packer (1.4m)		1.73	66.97	
Pesticide sprayer (24m)		0.17	6.90	
Harrow (9m)		0.17	11.09	
Hoe (6m)		0.72	30.03	
		Other activities:		
Inspection (share, every 5th year)		0.04	0.26	-
Manure application (25 m³/ha)		0.74	50.23	-
	Manure	-	-	€ 0.00/ha
Precision drill (6m-width)		0.53	41.72	-
Direct precision drill (59% increase to normal precision drill, 20% discount on light soils)		0.53	66.31	
	Seed			€ 233.20/ha
Mounted fertiliser spreader (amount depends on y)		0.00–0.29	0.00–6.14	
Liming (share, every 3rd year)		0.19	12.47	
	N			€ 1.10/kg
	P_2O_5			€ 0.87/kg
	K			€ 0.77/kg
	Ca			€ 0.05/kg
Forage harvester (7.5m, dependent on y)		$= 0.4325 + 0.00003y$	$= 119 + 0.02038y$	
Forage transport (dependent on y)		$= 0.7425 + 0.00308y$	$= 13 + 0.13873y$	
Telehandler + weight (dependent on y)		$= 0.2225 + 0.00218y$	$= 0.0487y$	

Note: Diesel consumption for mouldboard ploughing is assumed to be 30% higher/lower on heavy/light soils (for chisel ploughing 20% higher/lower).

Only the yield-dependent fertilisation costs are included in our calculations. If a higher yield is achieved, higher fertilisation costs arise. Other fertilisation costs are excluded, because we do not know the soil nutrient content and we do not know the type of fertilisation (artificial or organic fertiliser). The calculated gross margins are therefore slightly higher than in reality.

Reference: Böcker *et al*. (2018)

Table A5.3. Calculation of material use in [kg/ha] for machines used in maize production

Machine	**Empty weight [kg] (Source: machinery producers)**	**Operating potential (Source: Achilles *et al.*, 2016)**	**Material use per ha[a] [kg/ha]**	**Multi-plier for energy use**
Tractor (133 kW) (add to other processes)	7,790	10,000h	$= \frac{7{,}790[\mathrm{kg}]}{10{,}000[\mathrm{h}]} * \mathrm{t}_{\Omega}\left[\frac{\mathrm{h}}{\mathrm{ha}}\right]$	E_{tr}
Plough (4 mould-boards)	907	2,000ha	$= \frac{907[\mathrm{kg}]}{2{,}000[\mathrm{ha}]}$	E_{tm}
Packer (1.4m)	758	2,550ha	$= \frac{758[\mathrm{kg}]}{2{,}550[\mathrm{ha}]}$	E_{tm}
Chisel cultivator (4.5m)	2,125	2,250ha	$= \frac{2{,}125[\mathrm{kg}]}{2{,}250[\mathrm{ha}]}$	E_{tm}
Sprayer (1,500 l)	615	2,400m³	$= 615[\mathrm{kg}] * \left(\frac{2{,}400[\mathrm{m}^3]}{0.3\left[\frac{\mathrm{m}^3}{\mathrm{ha}}\right]}\right)^{-1}$	E_{om}
>>Boom (24m)	835	9,600ha	$= \frac{835[\mathrm{kg}]}{9{,}600[\mathrm{ha}]}$	E_{om}
Precision drill (6m)	2,300	1,500ha	$= \frac{2{,}300[\mathrm{kg}]}{1{,}500[\mathrm{ha}]}$	E_{om}
Direct precision drill (6m)	3,100	1,500ha	$= \frac{3{,}100[\mathrm{kg}]}{1{,}500[\mathrm{ha}]}$	E_{om}
Harrow (9m)	1,520	4,500ha	$= \frac{1{,}520[\mathrm{kg}]}{4{,}500[\mathrm{ha}]}$	E_{tm}
Hoe (6m)	1,900	3,200ha	$= \frac{1{,}900[\mathrm{kg}]}{3{,}200[\mathrm{ha}]}$	E_{tm}
Slurry tanker (20m³)	5,800	200,000m³	$= 5{,}800[\mathrm{kg}] * \left(\frac{200{,}000[\mathrm{m}^3]}{25\left[\frac{\mathrm{m}^3}{\mathrm{ha}}\right]}\right)^{-1}$	E_{om}
>>Line spreading boom (18m)	1,620	210,000m³	$= 1{,}620[\mathrm{kg}] * \left(\frac{210{,}000[\mathrm{m}^3]}{25\left[\frac{\mathrm{m}^3}{\mathrm{ha}}\right]}\right)^{-1}$	E_{om}
Mounted fertiliser spreader (24m, 1.5m³)	332	5,000t	$= 332[\mathrm{kg}] * \left(\frac{5{,}000[\mathrm{t}]}{\mathrm{m_f}\left[\frac{\mathrm{t}}{\mathrm{ha}}\right]}\right)^{-1}$	E_{om}
Trailed fertiliser spreader (average annual value based on application every 3 years)	2,500	17,500t	$= 2{,}500[\mathrm{kg}] * \left(\frac{17{,}500[\mathrm{t}]}{\mathrm{m_f}\left[\frac{\mathrm{t}}{\mathrm{ha}}\right]}\right)^{-1}$	E_{om}

Table A5.3. *(continued)*

Machine	**Empty weight [kg] (Source: machinery producers)**	**Operating potential (Source: Achilles *et al.*, 2016)**	**Material use per ha[a] [kg/ha]**	**Multi-plier for energy use**
Forage harvester	14,350	3,000h	$= \frac{14{,}350[\mathrm{kg}]}{10{,}000[\mathrm{h}]} * t_{\Omega}\left[\frac{\mathrm{h}}{\mathrm{ha}}\right]$	E_{hr}
>>Header (7.5m)	3,250	3,100ha	$= \frac{3{,}250[\mathrm{kg}]}{3{,}100[\mathrm{ha}]}$	E_{hr}
Forage transport wagon	8,000	121,000t	$= 8{,}000[\mathrm{kg}] * \left(\frac{121{,}000[\mathrm{t}]}{y\left[\frac{\mathrm{t}}{\mathrm{ha}}\right]}\right)^{-1}$	E_{om}
Telehandler + weight	13,500	10,000h	$= \frac{13{,}500[\mathrm{kg}]}{10{,}000[\mathrm{h}]} * t_{\Omega}\left[\frac{\mathrm{h}}{\mathrm{ha}}\right]$	E_{tr}

[a] t_{Ω} depicts the time needed for a certain field work, y is the achieved yield in a certain year and m_f is the amount of fertiliser applied.

Table A5.4. Direct energy inputs and outputs in silage maize production

Activity	Direct energy input via diesel use [l/ha] ≙ [kg/ha] (ρ = 0.84 kg/l)	Indirect energy input energy equivalent GE [MJ/kg] (H_s = 45.4 MJ/kg + 10.5 MJ/kg for upstream energy use, Jayasundara *et al*. (2014: 81))	Reference
Outputs:			
Silage maize (fresh matter, contains 34.8% dry matter)	NA	$Y_{m,t}$ [dt/ha] · 100 [kg/dt] · {23.9 [MJ/kg] · 2.7% XP + 39.8 [MJ/kg] · 1.2% XL + 20.1 [MJ/kg] · 6.7% XF + 17.5 [MJ/kg] · 24.2% NfE}	LUFA Nord-West (n.d.); Kirchgeßner (1995)
Inputs:			
Weed control-related activities:			
Chisel plough/Cultivator (4.5m)	7.50[a] ≙ 6.30	352.17	Achilles *et al*. (2016)
Mouldboard plough and packer (1.4m)	25.55[a] ≙ 21.46	1199.73	*Ibid.*
Pesticide sprayer (24m)	0.95 ≙ 0.80	44.61	*Ibid.*
Harrow (9m)	2.25 ≙ 1.89	105.65	*Ibid.*
Hoe (6m)	3.95 ≙ 3.32	185.48	*Ibid.*
Other activities:			
Rating (average annual value based on application every 5 years)	0.03/5 ≙ 0.03/5	1.41/5	*Ibid.*
Manure application (25 m³/ha)	10.40 ≙ 8.74	488.34	*Ibid.*
Manure	By-product	0.00	*E.g.* Jayasundara *et al*. (2014: 82)
Precision drill (6m-width)	2.55 ≙ 2.14	119.74	Achilles *et al*. (2016)
Direct precision drill (6m width)	2.55·(1+0.59) ≙ 3.41	190.38	Assumption based on Achilles *et al*. (2016: 184)
Seed	NA	35.75 [kg/ha] * 10.38 [MJ/kg]	Graboski (2002: 28)[b]
Mounted fertiliser spreader (amount depends on *E(y)* and x_f = necessary fertiliser)	$V_f = \begin{cases} 0, & x_f \leq 0 \\ 0.543 + 0.001x_f, & x_f > 0 \end{cases} \triangleq \begin{cases} 0, & x_f \leq 0 \\ 0.456 + 0.001x_f, & x_f > 0 \end{cases}$	$E_f = \begin{cases} 0, & y \leq 0 \\ V_f H_s, & y > 0 \end{cases}$	Functions according to data of Achilles *et al*. (2016)

Fig. A5.4. *(continued)*

Activity	Direct energy input via diesel use [l/ha] ≙ [kg/ha]	Indirect energy input energy equivalent GE [MJ/kg]	Reference
Inputs:			
Liming (average annual value based on application every 3 years)	2.25/3 ≙ 1.89/3	105.65/3	Achilles *et al*. (2016)
N	NA	47.10	
P_2O_5	NA	15.80	Biskupek *et al*. (1997: 214)
K_2O	NA	9.28	
CaO	NA	2.12	
Harvest: field work	$V_{fw} = \begin{cases} 0, \ y \leq 0 \\ 10.300 + 0.028y, \ y > 0 \end{cases} \triangleq \begin{cases} 0, y \leq 0 \\ 8.652 + 0.023y, \ y > 0 \end{cases}$	$E_{fw} = \begin{cases} 0, \ y \leq 0 \\ V_{fw}H_s, \ y > 0 \end{cases}$	
Harvest: transport	$V_{tr} = \begin{cases} 0, \ y \leq 0 \\ 7.675 + 0.029y, \ y > 0 \end{cases} \triangleq \begin{cases} 0, \ y \leq 0 \\ 6.447 + 0.024y, \ y > 0 \end{cases}$	$E_{tr} = \begin{cases} 0, \ y \leq 0 \\ V_{tr}H_s, \ y > 0 \end{cases}$	Functions according to data of Achilles *et al*. (2016: 216)
Harvest: compaction[c]	$V_{cp} = \begin{cases} 0, \ y \leq 0 \\ 0.012y, \ y > 0 \end{cases} \triangleq \begin{cases} 0, \ y \leq 0 \\ 0.010y, \ y > 0 \end{cases}$	$E_{cp} = \begin{cases} 0, \ y \leq 0 \\ V_{cp}H_s, \ y > 0 \end{cases}$	

[a] Diesel use for mouldboard ploughing is, in relation to Kalk and Hülsbergen (1999) and Jayasundara *et al*. (2014: 82), assumed to be 30% higher/lower on heavy/light soils. Diesel use for chisel ploughing is assumed to be 20% higher/lower on heavy/light soils.

[b] Graboski (2002) reports a value of 241,554 BTU/Bushel at 12.5% moisture. To convert the value, a weight conversion for corn of 54.08 Pound/Bushel as well as factors of 0.454 Kg/Pound and 0.0011 MJ/BTU are assumed (Murphy, 1993).

[c] Silo material and silage tarps were not included in the energy balance.

Appendix 5.D

Detailed explanation weighing scheme Pesticide Load Indicator

In the following, the calculation of the Pesticide Load Indicator (PLI) is described in detail. Additionally, the specific weights and focus points of the PLI are discussed briefly.

The PLI has the unit Load per kg pesticide product and is calculated in two steps:

a) Computation of the sub-indicators of the *Pesticide Load* per **kg active substance** (AS)

   i. Computation of the Environmental Fate Load

   ii. Computation of the Environmental Toxicity Load

b) Computation of the *Pesticide Load* per **product/kg** from the sub-indicators in a) and the *Health Load*

The detailed steps in the computation are as follows:

a) Computation of the sub-indicators of the *Pesticide Load* per **kg AS** (Miljøministeriet, 2012: 17)

   i. Computation of the Environmental Fate Load

$$\begin{aligned}\text{Fate Load } AS &\\ = \text{Degradation in soil} + \text{Bioaccumulation potential} + \text{Mobility in soil} &\\ = \frac{\text{Soil DT}_{50}}{354} * 2.5 + \left(\frac{\text{BCF}}{5100}\right) * 2.5 + \left(\frac{\text{SCI} - \text{Grow}}{12.5}\right) * 20 &\end{aligned} \tag{A5.6}$$

The Fate Load is computed from the soil half-time (Soil DT_{50}), the potential accumulation in the soil (bio-concentration factor, BCF) and the soil mobility (SCI-GROW Index). The three values are divided by the respective maximum values in the sample, creating a factor $\in [0,1]$ respectively. For maximum values (factor of 1) 2.5, 2.5 and 20 Load units are then allocated.

   ii. Computation of the Environmental Toxicity Load

$$\text{Toxicity Load} = \text{Short term effects}\begin{pmatrix}\text{Birds (1), Mammals(1), Fish(30), Daphnia(30), Algae(3),}\\ \text{Aquatic plants(3), Earthworms(2), Bees(100)}\end{pmatrix} + \text{Long term effects}\left(\text{Fish(3), Daphnia(3), Earthworms(2)}\right) \quad \text{(A5.7)}$$

The *Environmental Toxicity Load* is computed from long- and short-term effects of the respective AS on different families of animals and plants. Short-term effects are measured with LD_{50}/EC_{50} values and are again divided by the respective maximum value in the entire sample and then multiplied with the values in parentheses in the above formula. Long-term effects are measured with No Observed Effect Concentration (NOEC)-values and are normalised by the same approach as used for short-term values and then multiplied by the respective values in parentheses (Miljøministeriet, 2012: 47f.).

a) Computation of the *Pesticide Load* per **kg of product** from the sub-indicators in a) and the *Health Load*

First, the *Health Load* is calculated on a product level and *Fate* and *Toxicity Load* are transformed from AS to product aggregation level:

$$\text{Health Load Product} = \text{Formulation} * \frac{\sum_{i=1}^{n} \text{Risk Score}_i}{\text{max.sum risk scores (350)}}, \quad \text{(A5.8)}$$

where the index *i* denotes single risk (R)- and hazard (H)- statements. Potential effects on human health are measured with a ‘risk-score’ attributed to each R- and H-statement (Miljøministeriet, 2012: 17, for the list of ‘risk scores’). Here, H- and R-statements of the respective product are relevant. All risk scores of the R- and H-statements are the summed up per product and divided by the maximum value of all products in the sample. Finally, the *Health Load* for products that have a higher risk of exposition (powder or liquids that have to be mixed) is multiplied by 1.5.

Fate Load und *Toxicity Load* per product are computed as the weighted sums of the Loads of their AS. Weighting is done by the respective concentration of AS per product:

$$\text{Fate Load Product} = \sum_{i=1}^{n} \text{Fate Load AS}_i * \frac{\text{kg AS}_i}{\text{kg Product}}$$
$$\text{Toxicity Load Product} = \sum_{i=1}^{n} \text{Toxicity Load AS}_i * \frac{\text{kg AS}_i}{\text{kg Product}} \;, \tag{A5.9}$$

where the index *i* indicates single AS per product and the concentration of all AS per product in the range of $[0,1]$. The PLI in Load per kg product is then computed as the sum of the three sub-indicators:

$$\begin{aligned} &\text{Pesticide Load Product} \\ &= \text{Fate Load Product} + \text{Toxicity Load Product} + \text{Health Load Product} \end{aligned} \tag{A5.10}$$

The detailed explanations underline that the PLI covers a broad range of indicators and indices in the three categories environmental toxicity, environmental fate and human health. At the same time, note that specific indicators or indices are weighted stronger than others representing on the one hand a calibration of the indicator to the values appearing in the sample, and on the other hand valuation and priorities. Moreover, the selection of criteria for the sub-indicators already reflects a valuation of most important animal and plant families to be protected. Aligned with European pesticide regulations, the indicator reflects priorities, indicators and indices used in the European registration procedures for pesticides.

Appendix 5.E

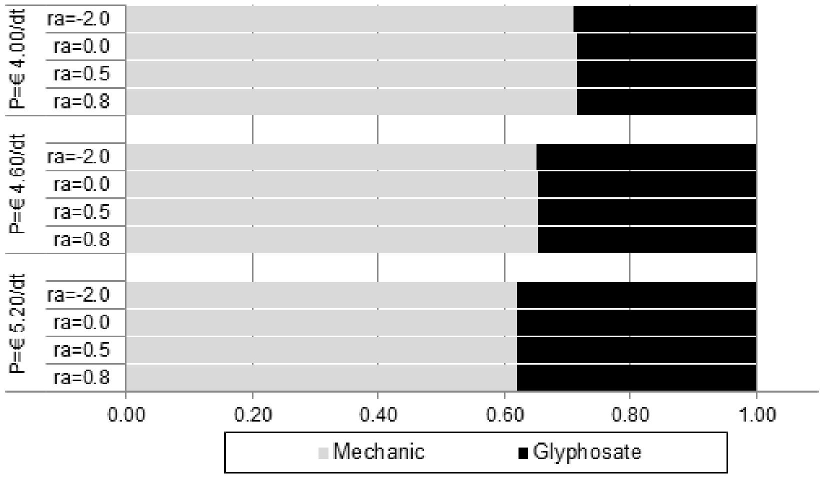

Fig. A5.2. Applied pre-sowing weed control strategies in the model for three price levels (*P*) and four risk aversion coefficients (r_a) (glyphosate licensed-scenarios).

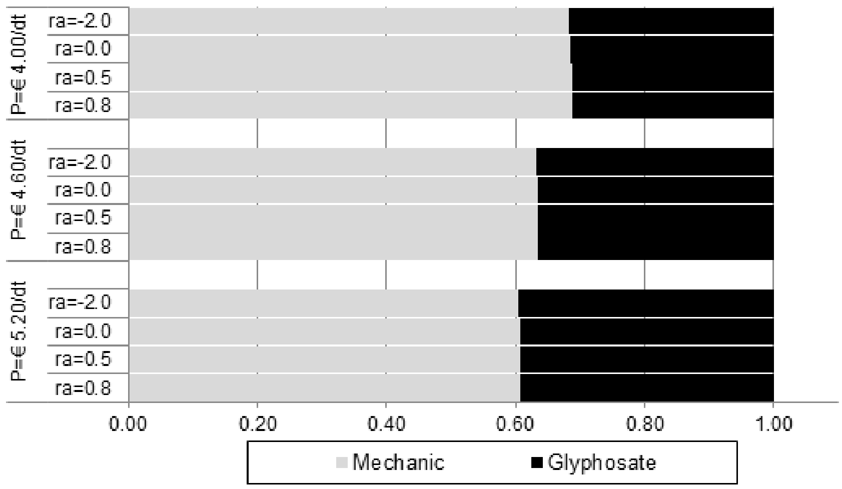

Fig. A5.3. Applied pre-sowing weed control strategies in the model for three price levels (*P*), four risk aversion coefficients (r_a) and the increased attainable yield level by 10% (glyphosate licensed-scenarios).

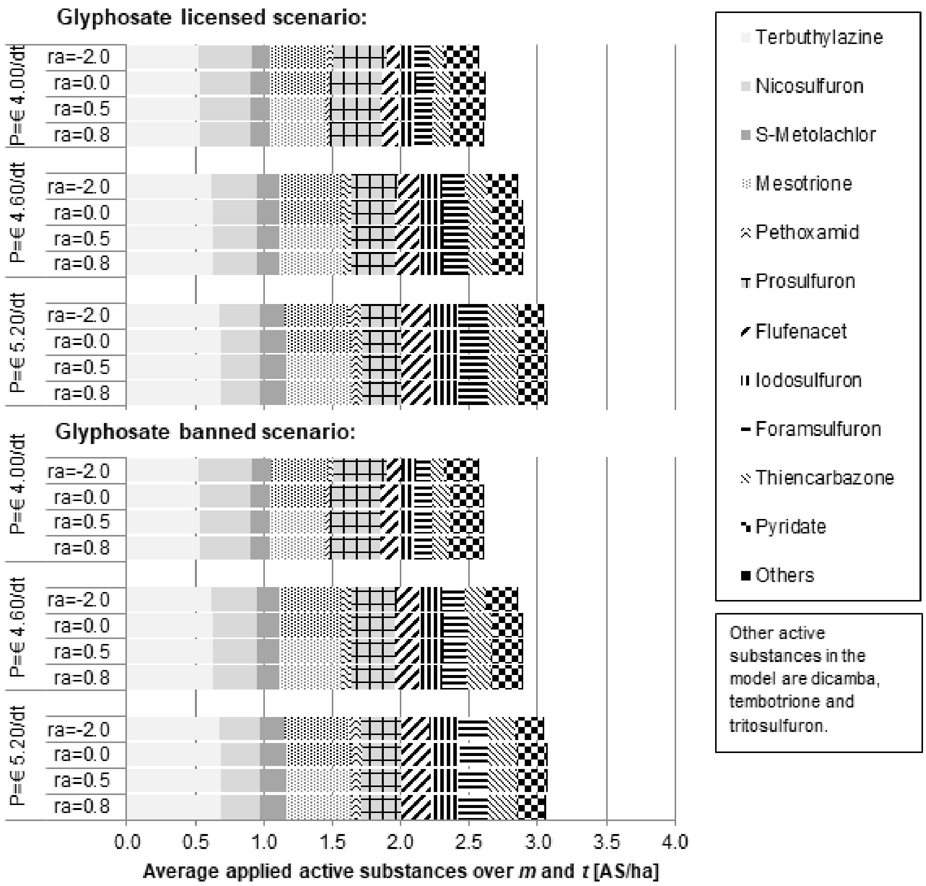

Fig. A5.4. Average applied active substances [AS/ha and year] in the post-sowing weed control strategies for three price levels (*P*) and four risk aversion coefficients (r_a).

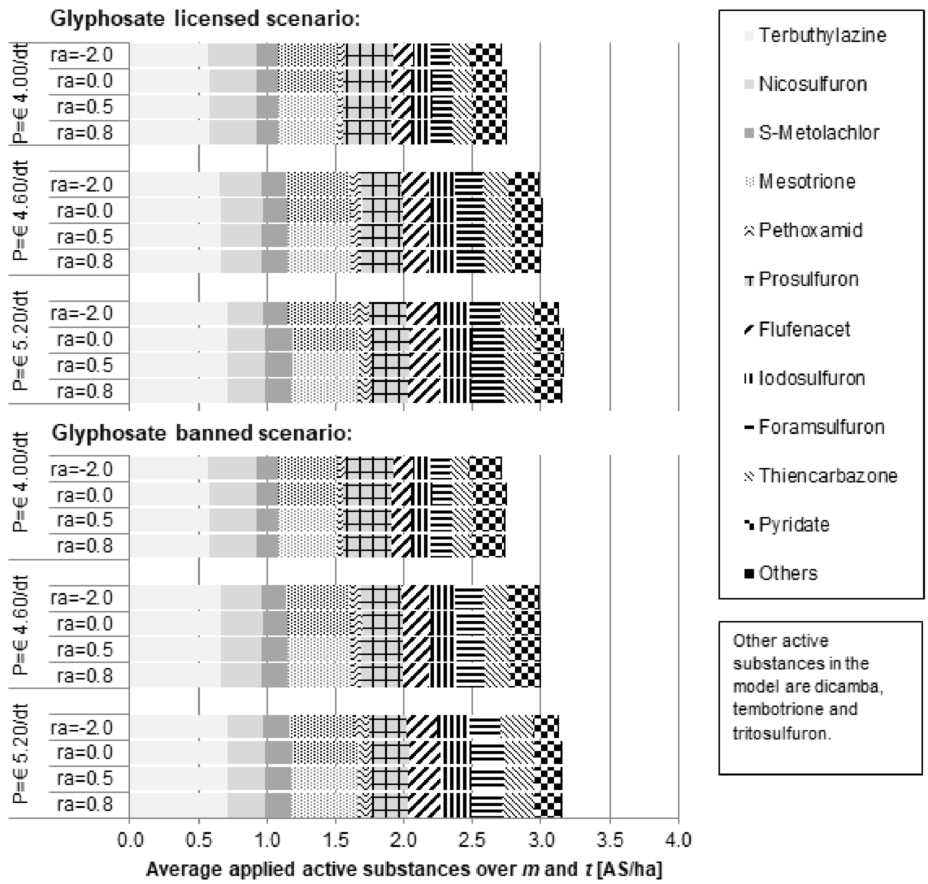

Fig. A5.5. Average applied active substances [AS/ha and year] in the post-sowing weed control strategies for three price levels (P), four risk aversion coefficients (r_a) and the attainable yield with a 10% increase.

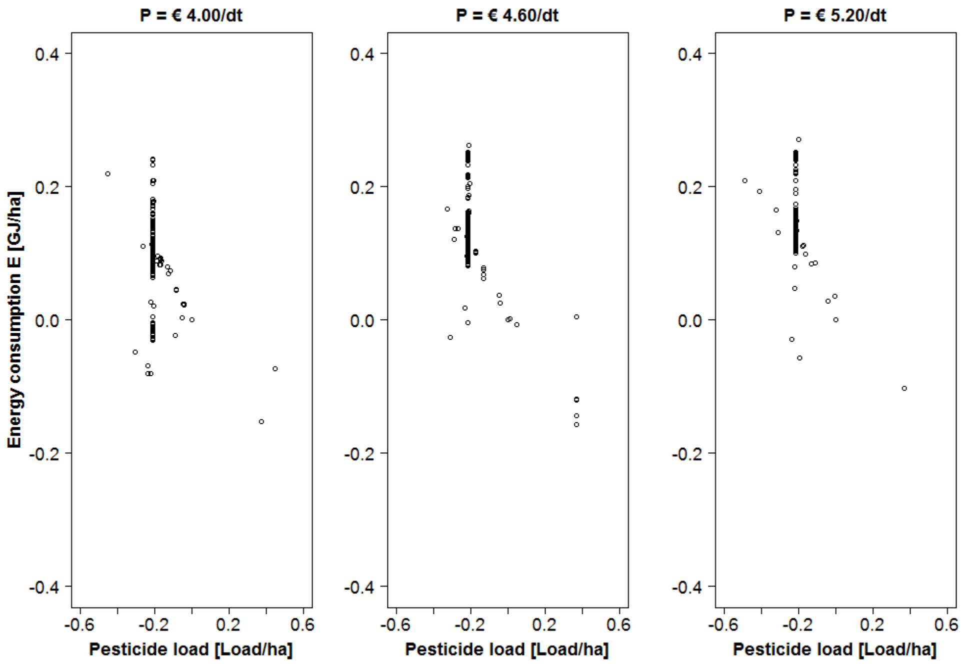

Fig. A5.6. Reduction of pesticide load (x-axis) plotted against the increase in energy consumption *E* (y-axis).

Table A5.5. Change (*Δ*) in Load/ha by a glyphosate ban in silage maize production (SD in brackets)[a,b]

Att. Yield:		**Given att. yield**				**Att. yield + 10%**			
	Load:	$\Delta\Lambda_{toxy}$	$\Delta\Lambda_{fate}$	$\Delta\Lambda_{heal}$	$\Delta\Lambda_{total}$	$\Delta\Lambda_{toxy}$	$\Delta\Lambda_{fate}$	$\Delta\Lambda_{heal}$	$\Delta\Lambda_{total}$
P [€/dt]:	$\mathbf{r_a}$**:**								
4.00	**0.8**	-0.011 (0.004) ***	-0.02 (0.037) ***	-0.17 (0.024) ***	-0.201 (0.047) ***	-0.011 (0.007) ***	-0.012 (0.077 ***	-0.173 (0.016) ***	-0.197 (0.080) ***
	0.5	-0.011 (0.006) ***	-0.019 (0.061) ***	-0.17 (0.023) ***	-0.2 (0.069) ***	-0.011 (0.006) ***	-0.013 (0.076 ***	-0.173 (0.015) ***	-0.197 (0.077) ***
	0.0	-0.011 (0.008) ***	-0.013 (0.087) ***	-0.171 (0.025) ***	-0.194 (0.094) ***	-0.011 (0.006) ***	-0.016 (0.075 ***	-0.172 (0.017) ***	-0.199 (0.077) ***
	-2.0	-0.009 (0.023) ***	-0.011 (0.263) ***	-0.154 (0.058) ***	-0.173 (0.275) ***	-0.010 (0.020) ***	-0.006 (0.233 ***	-0.170 (0.034) ***	-0.186 (0.237) ***
4.60	**0.8**	-0.011 (0.008) ***	-0.011 (0.089) ***	-0.174 (0.016) ***	-0.197 (0.091) ***	-0.012 (0.005) ***	-0.019 (0.055 ***	-0.174 (0.011) ***	-0.205 (0.057) ***
	0.5	-0.011 (0.008) ***	-0.011 (0.091) ***	-0.175 (0.015) ***	-0.196 (0.092) ***	-0.011 (0.005) ***	-0.018 (0.057 ***	-0.174 (0.014) ***	-0.202 (0.060) ***
	0.0	-0.011 (0.005) ***	-0.019 (0.062) ***	-0.173 (0.016) ***	-0.204 (0.064) ***	-0.011 (0.009) ***	-0.013 (0.098 ***	-0.174 (0.017) ***	-0.198 (0.100) ***
	-2.0	-0.011 (0.028) ***	-0.026 (0.322) ***	-0.161 (0.052) ***	-0.197 (0.328) ***	-0.012 (0.019) ***	-0.030 (0.223 ***	-0.165 (0.043) ***	-0.207 (0.226) ***
5.20	**0.8**	-0.012 (0.005) ***	-0.025 (0.054) ***	-0.174 (0.012) ***	-0.211 (0.056) ***	-0.011 (0.006) ***	-0.018 (0.072 ***	-0.174 (0.012) ***	-0.204 (0.073) ***
	0.5	-0.012 (0.004) ***	-0.025 (0.043) ***	-0.174 (0.011) ***	-0.211 (0.045) ***	-0.011 (0.007) ***	-0.018 (0.080 ***	-0.175 (0.012) ***	-0.204 (0.081) ***
	0.0	-0.012 (0.007) ***	-0.02 (0.076) ***	-0.174 (0.015) ***	-0.206 (0.079) ***	-0.011 (0.008) ***	-0.016 (0.089 ***	-0.174 (0.015) ***	-0.202 (0.090) ***
	-2.0	-0.011 (0.021) ***	-0.017 (0.244) ***	-0.17 (0.036) ***	-0.198 (0.249) ***	-0.012 (0.016) ***	-0.026 (0.178 ***	-0.166 (0.039) ***	-0.205 (0.184) ***

[a] Difference of Load units for the glyphosate licensed scenarios minus the glyphosate banned scenarios. Only municipalities are included in which glyphosate is used in the model under the licensed scenario.

[b] ‘, *, ** and *** denote significance at the 10%, 5%, 1% and 0.1% level respectively. A Wilcoxon-Mann-Whitney test is used for the hypotheses testing. Applying t-tests leads to different results, $\Delta\Lambda_{toxy}$ and $\Delta\Lambda_{fate}$ are not significant and $\Delta\Lambda_{total}$ is significant at the 5% level.

Table A5.6. Change (*Δ*) in energy output and input per hectare by a glyphosate ban in silage maize production (SD in brackets)[a,b]

Att. yield:		**Given att. yield**						**Att. yield + 10%**					
P: [€/dt]	**Energy:** **r_a:**	**ΔEO [MJ/ha]**	**ΔNEO [MJ/ha]**	**$ΔE_d$ [MJ/ha]**	**$ΔE_i$ [MJ/ha]**	**ΔE [MJ/ha]**	**ΔEO/E**	**ΔEO [MJ/ha]**	**ΔNEO [MJ/ha]**	**$ΔE_d$ [MJ/ha]**	**$ΔE_i$ [MJ/ha]**	**ΔE [MJ/ha]**	**ΔEO/E**
4.00	**0.8**	-1101 (509)	-1187 (503)	273 (60) ***	-176 (64)	87 (64)	-0.42 (0.25) *	-1105 (519)	-1201 (495)	290 (36) ***	-178 (64)	103 (72)	-0.37 (0.21) **
	0.5	-716 (667)	-783 (695)	277 (51) ***	-181 (59)	91 (64)	-0.27 (0.28) '	-1107 (506)	-1211 (475)	290 (40) ***	-177 (61)	105 (69)	-0.38 (0.20) **
	0.0	-1208 (452)	-1294 (413)	278 (48) ***	-193 (65)	86 (73)	-0.44 (0.29) *	-1132 (464)	-1243 (416)	291 (39) ***	-179 (52)	111 (68)	-0.40 (0.20) **
	-2.0	-940 (773)	-1026 (704)	271 (69) ***	-148 (144)	85 (142)	-0.42 (0.53) *	-1076 (885)	-1186 (757)	286 (56) ***	-174 (142)	110 (148)	-0.39 (0.37) **
4.60	**0.8**	-841 (445)	-972 (415)	300 (37) ***	-160 (60)	134 (70)	-0.50 (0.28) ***	-815 (445)	-959 (408)	305 (35) ***	-153 (52)	147 (68)	-0.43 (0.23) ***
	0.5	-849 (436)	-981 (399)	300 (40) ***	-159 (58)	137 (70)	-0.52 (0.28) ***	-799 (450)	-944 (414)	305 (38) ***	-151 (51)	148 (68)	-0.44 (0.24) ***
	0.0	-843 (419)	-990 (367)	302 (34) ***	-155 (51)	147 (65)	-0.56 (0.27) ***	-836 (440)	-986 (381)	306 (36) ***	-155 (55)	150 (72)	-0.45 (0.23) ***
	-2.0	-717 (868)	-851 (744)	297 (53) ***	-134 (162)	134 (170)	-0.51 (0.51) ***	-730 (647)	-879 (566)	302 (50) ***	-136 (119)	149 (125)	-0.44 (0.33) ***

Table A5.6. *(continued)*[a,b]

Att. yield:		Given att. yield						Att. yield + 10%					
P: [€/dt]	Energy: r_a:	ΔEO [MJ/ha]	ΔNEO [MJ/ha]	ΔE_d [MJ/ha]	ΔE_i [MJ/ha]	ΔE [MJ/ha]	ΔEO/E	ΔEO [MJ/ha]	ΔNEO [MJ/ha]	ΔE_d [MJ/ha]	ΔE_i [MJ/ha]	ΔE [MJ/ha]	ΔEO/E
5.20	0.8	-652 (359)	-811 (328)	310 (36) ***	-139 (47)	166 (66) '	-0.57 (0.32) ***	-675 (372)	-845 (314)	314 (32) ***	-143 (48)	171 (69)	-0.47 (0.23) ***
	0.5	-648 (364)	-814 (327)	311 (31) ***	-140 (45)	167 (65)	-0.56 (0.32) ***	-652 (376)	-823 (323)	313 (36) ***	-140 (51)	172 (72)	-0.46 (0.24) ***
	0.0	-677 (360)	-844 (302)	310 (35) ***	-144 (52)	166 (70)	-0.58 (0.32) ***	-667 (370)	-839 (309)	314 (35) ***	-141 (52)	172 (71)	-0.47 (0.23) ***
	-2.0	-632 (537)	-794 (448)	309 (41) ***	-134 (117)	162 (127)	-0.57 (0.44) ***	-594 (527)	-766 (449)	310 (49) ***	-128 (99)	172 (112)	-0.47 (0.30) ***

[a] Difference of process analysis for the glyphosate licensed scenarios minus the glyphosate banned scenarios: energy output (*EO*), net energy output (*NEO*), direct energy use (E_d), indirect energy use (E_i), total energy use (*E*) and energy efficiency (*EO/E*). Only municipalities are included in which glyphosate is used in the model under the licensed scenario.

[b] ‘, *, ** and *** denote significance at the 10%, 5%, 1% and 0.1% level respectively. A Wilcoxon-Mann-Whitney test is used for the hypotheses testing. Applying t-tests leads to lower significance levels for the energy efficiency (*ΔEO/E*) at the price levels € 4.00/dt and € 4.60/dt (not significant and significant at the 1% level).

5.8 References

Achilles, W., Eurich-Menden, B., Eckel, H., Frisch, J., Fritzsche, S., Fröba, N., Funk, M., Gaio, C., Grebe, S., Grimm, E., Grube, J., Hartmann, W., Horlacher, D., Kloepfer, F., Meyer, B., Sauer, N., Schroers, J. O., Schultheiß, U. and Wulf, S. (2016). *Betriebsplanung Landwirtschaft 2016/17 – KTBL-Datensammlung*. 25th ed. Darmstadt: Kuratorium für Technik und Bauwesen in der Landwirtschaft e.V. (KTBL).

Adamson, D., Mallawaarachchi, T. and Quiggin, J. (2007). Water use and salinity in the Murray–Darling Basin: A state-contingent model. *The Australian Journal of Agricultural and Resource Economics* 51 (3): 263-281.

Aguilera, E., Guzmán Casado, G. I., Infante, J., Soto, D. and González de Molina, M. (2017). Appendix II – Methodology for the Estimation of Embodied Energy Coefficients. In Guzmán Casado, G. I. and González de Molina, M. (eds): *Energy in Agroecosystems – A Tool for Assessing Sustainability*, Boca Raton, Florida: CRC Press, 317-400.

Audsley, E., Stacey, K., Parsons, D. J. and Williams, A. G. (2009). Estimation of the greenhouse gas emissions from agricultural pesticide manufacture and use. Cranfield: Cranfield University.

Auld, B. A., Menz, K. M. and Tisdell, C. A. (1987). *Weed Control Economics*. London: Academic Press.

Bàrberi, B., Cozzani, A., Macchia, M. and Bonari, E. (1998). Size and composition of the weed seedbank under different management systems for continuous maize cropping. *Weed Research* 38 (5): 319-334.

Bayerische Landesanstalt für Landwirtschaft (ed.) (2016). Unkrautmanagement in Mais. Online: https://www.lfl.bayern.de/ips/unkraut/033928/index.php [19 September 2017].

Bayerische Landesanstalt für Landwirtschaft (ed.) (2017). Pflanzenschutz - Unkrautbekämpfung - Unkraut-Steckbriefe. Online: https://www.lfl.bayern.de/ips/unkraut/034361/index.php [25 September 2017].

Benvenuti, S. and Macchia, M. (1993). Calculation of threshold temperature for the development of various weeds. *Agricoltura Mediterranea* 123: 252-256.

Benvenuti, S., Macchia, M. and Miele, S. (2001). Light, temperature and burial depth effects on Rumex obtusifolius seed germination and emergence. *Weed Research* 41 (2): 177-186.

Biskupek, B., Patyk, A. and Radtke, J. (1997). Daten zur Pflanzenproduktion. In Kaltschmitt, M. and Reinhardt, G. A. (eds), *Nachwachsende Energieträger – Grundlagen, Verfahren, ökologische Bilanzierung*. Braunschweig/Wiesbaden: Vieweg, 167-225.

Böcker, T. and Finger, R. (2016). European Pesticide Tax Schemes in Comparison: An Analysis of Experiences and Developments. *Sustainability* 8 (4): 378.

Böcker, T., Britz, W. and Finger, R. (2017). Bio-economic model on weed management in maize production. Online: https://doi.org/10.3929/ethz-b-000184083 [04 January 2018].

Böcker, T., Britz, W. and Finger, R. (2018). Modelling the Effects of a Glyphosate Ban on Weed Management in Silage Maize Production. *Ecological Economics* 145: 182-193.

Böcker, T. and Finger, R. (2018). Implikationen eines herbizidlosen Extenso-Weizenanbaus in der Schweiz. *Agrarforschung Schweiz* 9 (9): 296-305.

Bosnic, A. C. and Swanton, C. J. (1997). Influence of barnyardgrass (*Echinochloa crus-galli*) time of emergence and density on corn (*Zea mays*). *Weed Science* 45: 276-282.

Britz, W. and Delzeit, R. (2013). The impact of German biogas production on European and global agricultural markets, land use and the environment. *Energy Policy* 62: 1268-1275.

Brower, L. P., Taylor, O. R., Williams, E. H., Slayback, D. A., Zubieta, R. R. and Ramírez, I. (2012). Decline of monarch butterflies overwintering in Mexico: is the migratory phenomenon at risk? *Insect Conservation and Diversity* 5: 95-100.

Carpentier, A. (2017). Risk aversion and pesticide use: further insights from Prospect Theory. Paper presented to the XV European Association of Agricultural Economics (EAAE) Congress, August 29 – September 1. Parma (Italy): EAAE.

Chavas, J.-P. and Holt, M.T. (1996). Economic Behavior under Uncertainty: A Joint Analysis of Risk Preferences and Technology. *Review of Economics and Statistics* 78: 329-335.

Dalhaus, T., Musshoff, O. and Finger, R. (2018). Phenology Information Contributes to Reduce Temporal Basis Risk in Agricultural Weather Index Insurance. *Scientific Reports* 8 (46): 1-10.

Deike, S., Pallutt, B. and Christen, O. (2008). Investigations on the energy efficiency of organic and integrated farming with specific emphasis on pesticide use intensity. *European Journal of Agronomy* 28 (3): 461-470.

Duke, S. O. and Powles, S. B. (2008). Glyphosate: a once-in-a-century herbicide. *Pest Management Science* 64: 319-325.

Edenfield, M. and Allen, J. (2005). Sweet corn and popcorn tolerance to tembotrione/AE 0172747. *North Central Weed Science Society Abstracts* 60: 91.

Eurostat (ed.) (2018). Simplified energy balances - annual data (nrg_100a). Online: http://ec.europa.eu/ [23 April 2018].

Fidanza, M. A., Dernoeden, P. H. and Zhang, M. (1996). Degree-Days for Predicting Smooth Crabgrass Emergence in Cool-Season Turfgrass. *Crop Science* 36 (4): 990-996.

Finger, R., Möhring, N., Dalhaus, T. and Böcker, T. (2017). Revisiting pesticide taxation schemes. *Ecological Economics* 134: 263-266.

Finger, R. (2018). Take a holistic view when making pesticide policies stricter. *Nature* 556 (7700): 174.

Gardner, J. G. and Nelson, G. C. (2008). Herbicides, glyphosate resistance and acute mammalian toxicity: simulating an environmental effect of glyphosate-resistant weeds in the USA. *Pest Management Science* 64 (4): 470-478.

Gehring, K. and Thyssen, S. (2011). *Integrierter Pflanzenschutz - Leitunkräuter in Getreide*. Freising-Weihenstephan: Bayerische Landesanstalt für Landwirtschaft.

Graboski, M. S. (2002). Fossil energy use in the manufacture of corn ethanol. Golden, Colorado: Colorado School of Mines.

Green, M. B. (1987). Energy in pesticide manufacture, distribution and use. In Helsel, Z. R. (ed.): *Energy in Plant Nutrition and Pest Control*. Amsterdam: Elsevier, 165-177.

Guyton, K. Z., Loomis, D., Grosse, Y., El Ghissassi, F., Benbrahim-Tallaa, L., Guha, N., Scoccianti, C., Mattock, H. and Straif, K. (2015). Carcinogenicity of tetrachlorvinphos, parathion, malathion, diazinon, and glyphosate. *The Lancet – Oncology* 16 (5): 490-491.

Hall, D. C. and Norgaard, R. B. (1973). On the Timing and Application of Pesticides. *American Journal of Agricultural Economics* 55 (2): 198-201.

Hanzlik, K. and Gerowitt, B. (2011). The importance of climate, site and management on weed vegetation in oilseed rape in Germany. *Agriculture, Ecosystems & Environment* 141 (3-4): 323-331.

Hardaker, J. B., Pandey, S. and Patten, L. H. (1991). Farm Planning under Uncertainty: A Review of Alternative Programming Models. *Review of Marketing and Agricultural Economics* 59 (1): 9-22.

Hardaker, J. B., Lien, G., Anderson, J. R. and Huirne, R. B. M. (2015). *Coping with Risk in Agriculture – Applied Decision Analysis*. 3rd ed. Wallingford, UK, and Boston, USA: CABI.

Helander, M., Saloniemi, I. and Saikkonen, K. (2012). Glyphosate in northern ecosystems. *Trends in Plant Science* 17 (10): 569-574.

Hoffmann, H., Zhao, G., Bussel, L. G. J. van, Enders, A., Specka, X., Sosa, C., Yeluripati, J., Tao, F., Constantin, J., Raynal, H., Teixeira, E., Grosz, B., Doro, L., Zhao, Z., Wang, E., Nendel, C., Kersebaum, K.-C., Haas, E., Kiese, R., Klatt, S., Eckersten, H., Vanuytrecht, E., Kuhnert, M., Lewan, E., Rötter, R., Roggero, P. P., Wallach, D., Cammarano, D., Asseng, S., Krauss, G., Siebert, S., Gaiser, T. and Ewert, F. (2015). Variability of effects of spatial climate data aggregation on regional yield simulation by crop models. *Climate Research* 65: 53-69.

Horowitz, J. K. and Lichtenberg, E. (1993). Insurance, moral hazard, and chemical use in agriculture. *American Journal of Agricultural Economics* 75 (4): 926-935.

Hüllermeier, E. (2005). Experience-Based Decision Making: A Satisficing Decision Tree Approach. *IEEE Transactions on Systems, Man, and Cybernetics, Part A: Systems and Humans* 35 (5): 641-653.

Hülsbergen, K.-J., Feil, B., Biermann, S., Ranthke, G.-W., Kalk, W. D. and Diepenbrock, W. (2001). A method of energy balancing in crop production and its application in a long-term fertilizer trial. *Agriculture, Ecosystems and Environment* 86 (3), 303-321.

Information und Technik Nordrhein-Westfalen (ed.) (2017). Anbau auf dem Ackerland nach Fruchtarten. Online: https://www.it.nrw.de/statistik/i/index.html [20 April 2018].

IWR - Internationales Wirtschaftsforum Regenerative Energien (ed.) (2018). NRW-Energiestatistik – Ausgewählte Statistiken und Grafiken. Münster: IWR.

Jankowski, K. J., Budzyński, W. S. and Kijewski, L. (2015). An analysis of energy efficiency in the production of oilseed crops of the family Brassicaceae in Poland. *Energy* 81: 674-681.

Jayasundara, S., Wagner-Riddle, C., Dias, G. and Kariyapperuma, K. A. (2014). Energy and greenhouse gas intensity of corn (Zea mays L.) production in Ontario: A regional assessment. *Canadian Journal of Soil Science* 94: 77-95.

Jones, M. R. (1989). Analysis of the Use of Energy in Agriculture—Approaches and Problems. *Agricultural Systems* 29: 339-355.

Julius Kühn-Institut (ed.) (2017). Statistische Erhebungen zur Anwendung von Pflanzenschutzmitteln in der Praxis. Online: http://papa.julius-kuehn.de/index.php?menuid=29 [19 September 2017].

Kalk, W.-D. and Hülsbergen, K.-J. (1999). Dieselkraftstoffeinsatz in der Pflanzenproduktion. *Landtechnik* 54 (6): 332-333.

Karagiannis, G. and Tzouvelekas, V. (2012). The damage-control effect of pesticides on total factor productivity growth. *European Review of Agricultural Economics* 39 (3): 417-437.

Kehlenbeck, H., Saltzmann, J., Schwarz, J., Zwerger, P., Nordmeyer, H. Roßberg, D., Karpinski, I., Strassemeyer, J., Golla, B. and Freier, B. (2015). *Impact assessment of partial or complete abandonment of glyphosate application for* *farmers in Germany*. Julius-Kühn-Archiv Nr. 451. Quedlinburg: Julius Kühn-Institut.

Kirchgeßner, M. (ed.) (1995). Zur Energiebewertung beim Wiederkäuer – Mitteilungen des Ausschusses für Bedarfsnormen der Gesellschaft für Ernährungsphysiologie. *Proceedings of the Society of Nutrition Physiology* 4: 121-123.

Kudsk, P., Jørgensen, L. N. and Ørum, J. E. (2018). Pesticide Load – A new Danish pesticide risk indicator with multiple applications. *Land Use Policy* 70: 384-393.

Landwirtschaftskammer Nordrhein-Westfalen (ed.) (2015). Unkraut in Mais: Zweimal spritzen, Gewässer schützen. Online: http://www.landwirtschafts kammer.de/landwirtschaft/pflanzenschutz/ackerbau/mais/spritzfolgen.htm [04 January 2018].

Larsen, A. E., Gaines, S. D. and Deschênes, O. (2017). Agricultural pesticide use and adverse birth outcomes in the San Joaquin Valley of California. *Nature Communications* 8: 302.

Lauer, E. (1953). Über die Keimtemperatur von Ackerunkräutern und deren Einfluß auf die Zusamensetzung von Unkrautgesellschaften. *Flora (oder allgemeine botanische Zeitung)* 140: 551-595.

Lehmann, N., Briner, S., Finger, R. (2013). The impact of climate and price risks on agricultural land use and crop management decisions. *Land Use Policy* 35: 119-130.

Lewandowski, I. and Böhmel, C. (2009). Zucker- und Stärkepflanzen. In Kaltschmitt, M., Hartmann, H. and Hofbauer, H. (eds), *Energie aus Biomasse – Grundlagen, Techniken und Verfahren*. 2nd ed. Heidelberg: Springer, 115-134.

Lewis, K. A., Tzilivakis, J., Warner, D. J. and Green, A. (2016). An international database for pesticide risk assessments and management. *Human and Ecological Risk Assessment: An International Journal* 22 (4): 1050-1064.

Lichtenberg, E. and Zilberman, D. (1986). The Econometrics of Damage Control: Why Specification Matters. *American Journal of Agricultural Economics* 68 (2): 261-273.

Liew, J., Andersson, L., Boström, U., Forkman, J., Hakman, I. and Magnuski, E. (2012). Influence of temperature and photoperiod on sprouting of Cirsium arvense and Sonchus arvensis root buds. *Weed Research* 52 (5): 449-457.

LUFA Nord-West (ed.) (n.d.). Auswertung zu den Maissilagequalitäten 2015. Online: http://www.lufa-nord-west.de/index.cfm/action/downloadcenter.html [3 May 2017].

Lundkvist, A., Salomonsson, L., Karlsson, L. and Dock Gustavsson, A.-M. (2008). Effects of organic farming on weed flora composition in a long term perspective. *European Journal of Agronomy* 28 (4): 570-578.

Maart-Noelck, S. C. and Musshoff, O. (2014). Measuring the risk attitude of decision-makers: are there differences between groups of methods and persons? *The Australian Journal of Agricultural and Resource Economics* 58 (3): 336-352.

MacBean, C. (ed.) (2012). The Pesticide Manual – A World Compendium. 16th ed. Alton (Hampshire, UK): British Crop Production Council.

Martinkova, Z., Honek, A. and Lukas, J. (2006). Seed Age and Storage Conditions Influence Germination of Barnyardgrass (Echinochloa crus-galli). *Weed Science* 54 (2): 298-304.

McMaster, G. S. and Wilhelm, W. W. (1997). Growing degree-days: one equation, two interpretations. *Agricultural and Forest Meteorology* 87: 291-300.

Mehrtens, J. (2006). Unkraut ist nicht gleich Unkraut – Das Keim- und Auflaufverhalten von Ungräsern und Unkräutern in Mais. *Kurier: das Bayer-Crop-Science-Magazin für moderne Landwirtschaft* 1/06: 6-9.

Meraner, M. and Finger, R. (2017). Risk perceptions, preferences and management strategies: evidence from a case study using German livestock farmers. *Journal of Risk Research* (in press). Online: http://dx.doi.org/10.1080/13669877.2017.1351476 [18 September 2017].

Miljøministeriet (ed.) (2012). The Agricultural Pesticide Load in Denmark 2007-2010. Environmental review no. 2, Miljøstyrelsen. Online: http://www2.mst.dk/udgiv/publikationer/2012/03/978-87-92779-96-0.pdf [7 July 2017].

Möhring, N., Bozzola, M., Hirsch, S. and Finger, R. (2017). The role of pesticide indicators in the analysis of risk effects of pesticides – evidence from Swiss wheat production. Paper presented to the XV European Association of Agricultural Economics (EAAE) Congress, August 29 – September 1. Parma (Italy): EAAE.

Montgomery, D. R. (2007). Soil erosion and agricultural sustainability. *PNAS* 104 (33): 13268-13272.

Murphy, W. J. (1993). Tables for Weights and Measurements: Crops. Online: https://extension2.missouri.edu/G4020 [28 March 2018].

Nagy, C. N. (1999). Energy Coefficients for Agriculture Inputs in Western Canada. Saskatoon, Saskatchewan: Centre for Studies in Agriculture, Law and the Environment – University of Saskatchewan.

NetPhyD and BfN (eds) (2013). Verbreitungsatlas der Farn- und Blütenpflanzen Deutschlands. Bonn: Netzwerk Phytodiversität Deutschlands (NetPhyD) and Bundesamt für Naturschutz (BfN).

Nurse, R. E., Reynolds, W. D., Doucet, C., Weaver, S. E. (2008). Germination Characteristics of the Dimorphic Seeds of Spreading Orach (Atriplex patula). *Weed Science* 56 (2): 216-223.

Ørum, J. E. and Hossy, H. (2015). Bekæmpelsesmiddel-statistik 2014. Orientering fra Miljøstyrelsen nr. 13. Copenhagen: Miljøstyrelsen.

Ørum, J. E. and Sommer Holtze, M. (2017). Bekæmpelsesmiddel-statistik 2015. Orientering fra Miljøstyrelsen nr. 17. Copenhagen: Miljøstyrelsen.

Otte, A. (1990). Die Entwicklung von Ackerwildkraut-Gesellschaften auf Böden mit guter Ertragsfähigkeit nach dem Aussetzen von Unkrautregulierungsmaßnahmen. *Phytocoenologia* 19: 43-92.

Philbrook, B. D. and Santel, H. J. (2007). Thiencarbazone-methyl: a new molecule for PRE and postemergence weed control in corn. *Proceedings of the North Central Weed Science Society* 62: 150.

Reise, C., Liebe, U. and Mußhoff, O. (2012). Farmers' Preferences regarding the Design of Substrate Supply Contracts for Biogas Plants: A Choice Experiment. *German Journal of Agricultural Economics* 61 (3): 162-177.

Schönhammer, A., Pörksen, N., Freitag, J. and Nuyken, W. (2002). Biathlon – Das erste Tritosulfuron-haltige Herbizid zur Unkrautbekämpfung in monokotylen Kulturen. *Mitteilungen der Biologischen Bundesanstalt Land- und Forstwirtschaft* 390: 241-242.

Schulte, M. and Theuvsen, L. (2015). The economic benefit of herbicides in arable farming – with a special focus on glyphosate. *Journal für Kulturpflanzen* 67 (8): 269-279.

Statistisches Bundesamt (ed.) (2017). Land- und Forstwirtschaft, Fischerei – Bodennutzung der Betriebe. Fachserie 3 Reihe 3.1.2. Wiesbaden: Statistisches Bundesamt.

Storkey, J. (2004). Modelling Seedling Growth Rates of 18 Temperate Arable Weed Species as a Function of the Environment and Plant Traits. *Annals of Botany* 93 (6): 681-689.

Tarazona, J. V., Court-Marques, D., Tiramani, M., Reich, H., Pfeil, R., Istace, F. and Crivellente, F. (2017). Glyphosate toxicity and carcinogenicity: a review of the scientific basis of the European Union assessment and its differences with IARC. *Archives of Toxicology* 91 (8): 2723-2743.

Tembo, G., Brorsen, B. W., Epplin, F. M. and Tostão, E. (2008). Crop input response functions with stochastic plateaus. *American Journal of Agricultural Economics* 90 (2): 424-434.

Tomlin, C. D. S. (ed.) (2006). *The Pesticide Manual – A World Compendium*. 14th ed. Alton (Hampshire, UK): British Crop Production Council.

Uhlin, H.-E. (1999). Energy productivity of technological agriculture-lessons from the transition of Swedish agriculture. *Agriculture, Ecosystems & Environment* 73: 63-81.

Vigil, M. F., Anderson, R. L. and Beard, W. E. (1996). Base Temperature and Growing-Degree-Hour Requirements for the Emergence of Canola. *Crop Science* 37 (3): 844-849.

Vleeshouwers, L. M. (1997). Modelling weed emergence patterns. Dissertation. Wageningen: Wageningen Agricultural University.

Vleeshouwers, L. M. and Kropff, M. J. (2000). Modelling field emergence patterns in arable weeds. *New Phytologist* 148 (3): 445-457.

Walter, A., Finger, R., Huber, R. and Buchmann, N. (2017). Smart farming is key to developing sustainable agriculture. *PNAS* 114 (24): 6148-6150.

Wehsarg, O. (1918). *Verbreitung und Bekämpfung der Ackerunkräuter in Deutschland*. Arbeiten der Deutschen Landwirtschafts-Gesellschaft. Berlin: Paul Parey Verlag.

Wiese, A., Schulte, M., Theuvsen, L. and Steinmann, H.-H. (2018). Interactions of glyphosate use with farm characteristics and cropping patterns in Central Europe. *Pest Management Science* 74 (5): 1155-1165.

Williams, G. M., Kroes, R. and Munro, I. C. (2000). Safety Evaluation and Risk Assessment of the Herbicide Roundup and Its Active Ingredient, Glyphosate, for Humans. *Regulatory Toxicology and Pharmacology* 31: 117-165.

Zhao, G., Hoffmann, H., Bussel, L. G. J. van, Enders, A., Specka, X., Sosa, C., Yeluripati, J., Tao, F., Constantin, J., Raynal, H., Teixeira, E., Grosz, B., Doro, L., Zhao, Z., Nendel, C., Kiese, R., Eckersten, H., Haas, E., Vanuytrecht, E., Wang, E., Kuhnert, M., Trombi, G., Moriondo, M., Bindi, M., Lewan, E., Bach, M., Kersebaum, K.-C., Rötter, R., Roggero, P. P., Wallach, D., Cammarano, D., Asseng, S., Krauss, G., Siebert, S., Gaiser, T. and Ewert, F. (2015). Effect of weather data aggregation on regional crop simulation for different crops, production conditions, and response variables. *Climate Research* 65: 141-157.

Appendix of Thesis

Revisiting Pesticide Taxation Schemes

This comment summarises the project on pesticide taxes funded by the Swiss Federal Office for Agriculture. It was published in the journal *Ecological Economics* as:

Finger, R., Möhring, N., Dalhaus, T and Böcker, T. (2017). Revisiting Pesticide Taxation Schemes. *Ecological Economics* 134: 263-266.

Abstract

The risks caused by pesticide use for human health and nature are one of the major challenges for agricultural policies. Despite their high potential to contribute to better policies, economic instruments such as pesticide taxes are rarely used in the current policy mix. In this essay, we combine current discussion on pesticide policies in European countries with new insights from recent economic research to provide an outline for better pesticide policies to policy makers and stakeholders. We show that differentiated taxation schemes have a high potential to reduce risks caused by pesticide use and that the targeted re-distribution of tax revenues in the agricultural sector is crucial to create leverage effects on pesticide use and to increase the acceptability of pesticide taxes.

Keywords: Pesticide, Tax, Incentive, Environment, Regulation.

A.1 Introduction

Plant protection is essential for the provision of high quality food in adequate quantities (*e.g*. Oerke, 2006). However, especially the use of pesticides often induces possible negative effects for the environment and human health (*e.g*. Gilden *et al*., 2010; Pimentel, 2009; Travisi and Nijkamp, 2008).

The risks for human health and nature caused by pesticide use are one of the major challenges for agricultural policies and have caught large attention in recent public debates, such as on the potential ban of glyphosate in Europe. In response to these challenges, various European countries have introduced National Action Plans on pesticide use (*e.g.* due to the Directive 2009/128/EC). Economic instruments such as pesticide taxes can be efficient components of an optimal pesticide policy (Skevas *et al.*, 2013). Yet, these instruments are rarely used. For example, pesticide taxation schemes are established only in four European countries, *i.e.* in France, Sweden, Denmark and Norway - an introduction, however, is discussed in various other countries (*e.g.* Belgium, Switzerland, the Netherlands and Germany) (see Böcker and Finger, 2016, for an overview).1 Despite the higher allocative efficiency than other policy instruments that are frequently used, such as bans or regulation, little progress has been made to overcome stakeholders' preconceptions and concerns with respect to pesticide taxes (*e.g.* Zilberman and Millock, 1997). In a similar vein, current policies and policy proposals are often not aligned with the current state of research. This essay aims to contribute to bridge new insights from recent economic research and from current discussions on pesticide taxation in different European countries to provide an outline for better pesticide policies to policy makers and stakeholders.

A.2 Goals and Effectiveness of Policies

A.2.1 Definition of Goals of a Pesticide Tax

In order evaluate pesticide policy measures, crucial criteria are i) the effectiveness and efficiency of the measures, ii) the polluter pays principle and iii) the acceptability of the measure among stakeholders including the effects of policy measures on farmers' income (see *e.g.* Falconer, 1998). We will use these criteria as guiding principles to combine recent policy discussion and scientific evidence.

Important for the evaluation of the effectiveness and efficiency of measures such as a pesticide tax is the specification of policy targets, which varies substantially across countries. Reductions of physical quantities of pesticides used dominate public and policy debates, especially because these are easy to communicate and easy to measure. For example, the French policy defines a 50% reduction target for the total quantity of pesticides used from 2015 to 2025 (MAAF and MEDDE, 2015). However, such policy targets not necessarily internalize external effects. Pesticides differ strongly with regard to their properties, *i.e.* average quantities applied, intensity of application, risks of applied products for human health and the environment. A reduction of applied pesticide quantities could for example be easily achieved by the substitution of oils, normally used in great quantities but with low risks for human health and the environment, through smaller quantities of pesticides with potentially high risks (*e.g.* Böcker and Finger, 2016). Thus, risk-based indicators (*e.g.* based on H-phrases or R-phrases or impact assessment systems) should preferably be used to formulate policy targets as these better reflect external effects. For example, the Danish government planned to achieve a risk reduction of 40% between 2013 and 2015 (MIM and FVM, 2013), and the current proposal for a national action plan on pesticides in Switzerland postulates a reduction of risks caused by pesticide use by 50% (FOA, 2016).

The definition of risk-based policy targets has implications for the optimal design of pesticide taxes, which should internalize external costs of pesticide use to contribute to welfare increases. In order to reflect social marginal costs, taxes should not be uniform across pesticides in terms of ad-valorem or per unit taxes. In contrast, the potential mismatch between quantities of pesticide used and associated risks outlined above motivates differentiated pesticide taxes, so that more risky pesticides are taxed at higher rates. This creates incentives to substitute towards less toxic pesticides and non-chemical plant protection strategies. Along these lines, evidence from European taxation schemes shows that despite the fact that taxes have not reduced total quantities of pesticide use, they have led to the targeted reductions of risks caused by pesticide use (Böcker and Finger, 2016). In contrast, non-differentiated taxation schemes might create unintended consequences. For instance, quantity reductions can be caused by the substitution towards more toxic products resulting in higher risks for humans and the environment. Moreover, taxing only specific products at high levels keeps the average tax burden for pesticides low. If the high taxation of specific products, however, causes plant protection gaps, a dynamic fiscal scheme, as proposed by Martin (2015), should be adopted.

A.2.2 The Effectiveness of Pesticide Taxes

An important requirement for effectiveness and efficiency of pesticide taxes is that the demand for pesticides is price sensitive. The inelastic demand structure for pesticides was claimed in policy debates as a major reason for not introducing a pesticide tax (*e.g.* Hof *et al*., 2013, for the Netherlands). It is also used as a key argument in the policy debate such as by the German farmers' union in response to a recent proposal for a pesticide tax to be introduced in Germany (DBV, 2015). A recent meta-analysis shows that the median of pesticide demand elasticities reported in studies in North America and Europe is − 0.28 (Böcker and Finger, 2017). Thus, there is – on average – a significant change in pesticide use due to the introduction of a tax to be expected. However, this response is inelastic. Elasticity levels reported by individual studies differ remarkably. Skevas *et al*. (2012), for instance, report elasticities between − 0.03 and − 0.0003 for pesticides in Dutch crop production. In contrast, Chen *et al*. (1994) report elasticities of − 2.42 for mixed farms in Alabama (USA). The specific structure of demand elasticities has important implications for pesticide taxation schemes. In that respect, three observations from the study by Böcker and Finger (2017) are especially important. First, elasticities differ largely across agricultural systems. For example, special crops show less elastic demand structures. Thus, also pesticide use reductions and tax burdens will differ across agricultural systems. Second, the demand for pesticides is in the short-run substantially less elastic than in the long-run. In the long-run, crop rotations and production technologies can be adjusted. Thus, pesticide taxes should be evaluated only in the longer run. This is further emphasized by the observation that before the introduction or increase of a pesticide tax, hoarding activities were observed regularly in Sweden, France, Denmark and Norway (Böcker and Finger, 2016). Thus, a clear communication of the non-short term time horizon of targeted effects is indispensable. Third, elasticities differ across types of pesticides. More specifically, herbicides are found to be more elastic, also because mechanical alternatives are available. Thus, lower tax rates are required to reduce herbicide use.

A.3 Pesticide Taxes as Part of a Coherent Set of Policies

A.3.1 The Use of Tax Revenues Is Crucial

Increasing pesticide prices due to a tax could, especially in the short run, result in lower farm incomes. However, some recent studies suggest that income reduction due to reduced pesticide applications could be small. For example, Pedersen *et al*. (2012) show for a sample of 1164 Danish farms that one third of these farms is not operating cost-oriented but rather apply pesticides to maximize yields. In a similar vein, Nielsen (2005) argues that massive reductions in pesticide use have been achieved in Denmark without observing losses in aggregate agricultural incomes or production levels. For Dutch arable farms, Skevas *et al*. (2014) show that – if comparing with profit maximizing levels – 100% of the farms overuse herbicides, 86% overuse fungicides and 67% overuse insecticides. Jacquet *et al*. (2011) show that a

30% reduction of pesticide use would be possible without income losses for French arable farming systems.

However, potential income reductions for farmers and use of the tax revenues remain an important aspect of policy debates (see *e.g*. Bahrs and Back, 2016, for Germany). In addition, revenues of the pesticide tax in Sweden and Norway are not specifically used for agricultural or related purposes. In France, part of the tax revenues is used to internalize external effects of pesticide use, *i.e*. is used to clean water from pesticide residues (Art. L213-10-8 *Code de l'environnement*). The remaining revenues are allocated to the general budget. Earlier research has argued to support research activities with proceeds of pesticide taxes (*e.g*. Zilberman and Millock, 1997). Despite the fact that these solutions fulfil the polluter pays principle, the income reduction in the agricultural sector caused by a tax is one of the key hurdles for acceptance of such measure. Moreover, by not re-distributing tax revenues opportunities to create leverage effects are missed. We argue that a complete re-distribution of tax revenues to the sector shall be envisaged. Transaction costs of existing taxation schemes are very small, so that large parts of tax revenues are available for such re-distribution. For instance, in the Norwegian system transaction costs represent only about 1% of the tax revenues, with only about 10% of these costs incurring at the public administration level (Vatn *et al*., 2002). Tax revenues should be used to finance measures that create leverage effects with respect to reductions of risks caused by pesticide use. Those might comprise measures such as i) support of extensification (switch to organic or low pesticide production techniques), ii) support of new spraying material and new equipment related to pesticide use, iii) support of independent extension and advisory services, iv) support of biological plant protection strategies.

First, subsidizing organic production can lead to overall decreases in pesticide use. However, reduced production increases concerns of effectiveness if measuring environmental effects per unit of output and reduced domestic production causes problems of leakage. Other alternatives are subsidies to production systems that explicitly exclude specific pesticides from the production. For example, in Switzerland the production of cereals, rapeseed, sunflowers and beans not using all types of pesticides but herbicides and seed treatment is supported with an ecological direct payment (Finger and El Benni, 2013). Similar programs also exist in other countries (*e.g.* Baylis *et al*., 2008). Despite the higher intensity of these systems if compared to organic agriculture, smaller output levels are observed compared to intensive agricultural systems. Second, improved equipment and spraying technology can reduce emissions from pesticide use. For example, Aubert and Enjolras (2014) show in an analysis for French wine producer that the age of the equipment is significantly increasing the likelihood of over-application of pesticides. Investment in better technologies remains an important policy measure even in developed countries. Precision agriculture and the development of smart farming systems will offer new potentials in this field for the future. Support of these measures can compensate for tax burdens and does not lead to negative production effects. Third, independent management advice and extension service with particular focus on plant protection can contribute to the reduction of risks caused by pesticide use. Wiebers *et al*. (2002) show that support by advisors resulted in a higher pesticide use of farmers, which they argue stems from the profit maximization rationale of advisors who also sell pesticides to farmers. Thus, better and independent advisory and extension services might reduce pesticide use and increase farmers' income. Fourth, the support of biological prevention and control strategies contributes to reduce pesticide use and to maintain production levels (Cullen *et al*., 2008). These measures may comprise the creation of habitats for natural enemies or the use of biological crop protection agents.

Providing and supporting a wide set of targeted measures and strategies to reduce the risk caused by pesticide use is necessary to account for the large diversity of agricultural systems, environmental and human health impacts of pesticide use and finally also the large diversity of farm and farmers' characteristics. The revenues of a pesticide taxation scheme can provide the financial means required to realize these measures. More general, pesticide taxes are not efficient if used as stand-alone measure, but should be used in a coherent set of policies and measures aiming to reduce risks of pesticide use.

A.3.2 Aligning Agricultural Policies

Pesticide policies and pesticide taxes need to be aligned with other policy instruments. Four aspects are highlighted here that are overlooked in current policy designs. First, land use (*i.e*. extensive margin) effects of agricultural policies are potentially more relevant than policies at the intensive margin. The quantity of pesticide use as well as the resulting risks for the environment and humans differs substantially across crops. For instance, German promotion of biogas has created land use effects towards higher acreage under maize (Britz and Delzeit, 2013). Thus, a coherent framework is required that accounts also for extensive margin effects of pesticide taxes and other policies. Second, policies on fertilizer and pesticide use have to be assessed jointly. Empirical studies highlight the important interaction of fertilizer application and pesticide use. More specifically, higher levels of nitrogen fertilizer use have been reported to increase herbicide, insecticide and fungicide use, caused by positive effects on, for instance, weed growth, aphids and mildew (*e.g*. Bürger *et al*., 2012). In addition, the value of the marginal product of pesticides increases for fields with high yield potentials due to high fertilization levels. Thus, regulating nitrogen fertilizer use also affects pesticide use. However, such regulation via fertilizers cannot account for the risks for humans and the environment by specific pesticides. Third, pesticide policies and taxes can have important spill-over effects towards other policy goals. The above presented finding of the demand for herbicides being more elastic, implies that a pesticide tax may also affect tillage intensities, potentially diminishing benefits from minimal soil disturbance (Hobbs *et al*., 2008). The relevance of this interdependence between policy goals to restrict pesticide use and increase soil conservation has been highlighted in Europe's debate on the ban of glyphosate. Fourth, effects of pesticides on income risks and farmers' risk preferences can be crucial for policy design. For instance, recent shifts in agricultural policies towards the provision of better risk management instruments and subsidizing insurances (*e.g.* El Benni *et al*., 2016) may affect pesticide use and the effects of a taxation scheme. If pesticides are risk reducing, risk averse farmers would use larger than profit maximizing quantities of pesticides. However, pesticides have often been found to even have risk increasing effects (*e.g.* Horowitz and Lichtenberg, 1994). This is one of the reasons why yield and revenue insurances do not necessarily substitute for pesticide use. In contrast, empirical studies show that insurance uptake often leads to higher farm-level pesticide use (*e.g.* Finger *et al*., 2016b). Especially the effects of insurance on land use decisions, *i.e.* extensive margin effects, matter (*e.g.* Wu, 1999). Insurance solutions are often more attractive for intensive-input crops. Providing better insurance opportunities might thus lead to a land allocation towards these crops and increase total pesticide use at the extensive margin. Based on this background, insurance mechanisms should be developed that are specifically targeted for reductions of pesticide use (*e.g.* targeted index solutions, or insurances supporting low-pesticide production systems, *e.g.* Norton *et al*., 2016; Serra *et al*., 2008) and avoid land allocation effects.

A.4 Conclusions

Based on the above findings, we conclude the following for the design of pesticide policies and pesticide taxation schemes:

1) Differentiated pesticides taxes can effectively reduce risks for human health and the environment, induced by pesticides use. Substitution to less risky pesticides and non-chemical plant pro-

tection strategies can be incentivized when pesticides are taxed according to their potential riskiness and thus linking taxation more closely to external effects than an ad-valorem or per unit tax. Such a system keeps the average tax burden for pesticides low and can be adjusted to country specific policy goals.

2) Pesticide policies, including pesticide taxes have potentially large interdependencies with other policy targets and instruments with respect to risk management, fertilizer use and conservation agriculture that need to be accounted for in policy design.

3) The small but significant price elasticity of demand for pesticides implies that tax rates for highly toxic pesticides have to be large to generate a relevant decrease in their demand.

4) Low transaction costs for pesticide taxation allow re-distribution of tax revenues.

5) A reimbursement of tax revenues to the agricultural sector helps prevent income effects. Leverage effects on pesticide use can be created if this reimbursement is made via instruments which further reduce risks of pesticide use. The introduction of a tax scheme on pesticides should only be part of a portfolio of coherent accompanying policy measures, focusing on measures that do not imply reductions of production levels (*e.g.* better application technologies and better non-chemical plant protection) in order to avoid leakage problems. This especially applies for the case of production systems with small price elasticities of demand for pesticides.

6) Pesticide taxes do not have considerable short-run effects on pesticide use, because the elasticity of demand is small in the short-run and strong hoarding activities are induced. Pesticide

taxes, however, incentivize the long-run reduction of the risks caused by pesticide use for human health and nature.

A.5 References

Aubert, M. and Enjolras, G. (2014). The Determinants of Chemical Input Use in Agriculture: A Dynamic Analysis of the Wine Grape-Growing Sector in France. *Journal of Wine Economics* 9: 75-99.

Bahrs, E. and Back, H. (2016). Pflanzenschutzmittelsteuer – Das wird teuer. Top-Agrar 9/2016: 32-36.

Baylis, K., Peplow, S., Rausser, G. and Simon, L. (2008). Agri-environmental policies in the EU and United States: A comparison. *Ecological Economics* 65: 753-764.

Bellassen, V. (2015). Les certificats d'économie de produits phytosanitaires : quelle contrainte et pour qui ? Working Paper 2015/4. Dijon: Centre d'Economie et Sociologie appliquées à l'Agriculture et aux Espaces Ruraux.

Böcker, T. and Finger, R. (2016). European Pesticide Tax Schemes in Comparison: An Analysis of Experiences and Developments. *Sustainability* 8 (4/378): 1-22

Böcker, T. G. and Finger, R. (2017). A Meta-Analysis on the Elasticity of Demand for Pesticides. *Journal of Agricultural Economics* 68: 518-533.

Britz, W. and Delzeit, R. (2013). The impact of German biogas production on European and global agricultural markets, land use and the environment. *Energy Policy* 62: 1268-1275.

Bürger, J., de Mol, F. and Gerowitt, B. (2012). Influence of cropping system factors on pesticide use intensity–A multivariate analysis of on-farm data in North East Germany. *European Journal of Agronomy* 40: 54-63.

Chen, P.-C., McIntosh, C. S. and Epperson, J. E. (1994). The Effects of a Pesticide Tax on Agricultural Production and Profits. *Journal of Agribusiness* 12 (2): 125-138.

Cullen, R., Warner, K. D., Jonsson, M. and Wratten, S. D. (2008). Economics and adoption of conservation biological control. *Biological Control* 45 (2): 272-280.

DBV (ed.) (2015). Steuer auf Pflanzenschutzmittel ohne Lenkungswirkung. Deutscher Bauernverband (DBV). Online: http://www.bauernverband.de/steuer-auf-pflanzenschutzmittel-ohne-lenkungswirkung [26 January 2018].

El Benni, N., Finger, R. and Meuwissen, M. (2016). Potential effects of the Income Stabilization Tool (IST) in Swiss agriculture. *European Review of Agricultural Economics* 43: 475-502

Falconer, K. E. (1998). Managing diffuse environmental contamination from agricultural pesticides: an economic perspective on issues and policy options, with particular reference to Europe. *Agriculture, Ecosystems & Environment* 69 (1): 37-54.

Finger, R. and El Benni, N. (2013). Farmers' Adoption of Extensive Wheat Production - Determinants and Implications. *Land Use Policy* 30 (1): 206-213.

Finger, R., Böcker, T., Möhring, N. and Dalhaus, T. (2016a). Ökonomische Analyse des Einsatzes von Pflanzenschutzmitteln – Risikoaspekte und Lenkungsabgaben. Bericht zu Händen des Bundesamts für Landwirtschaft. Zürich: ETH Zürich und Universität Bonn.

Finger, R., Möhring, N., Dalhaus, T. and Enjolras, G. (2016b). Crop Insurance and Pesticide Use. 156th EAAE Seminar, Prospects for agricultural insurance in Europe, Wageningen, 3-4 October 2016.

FOA (ed.) (2016). Aktionsplan zur Risikoreduktion und nachhaltigen Anwendung von Pflanzenschutzmitteln. Swiss Federal Office for Agriculture (FOA): Bern. Online: http://www.blw.admin.ch/themen/00011/00075/02001/index.html?lang=de [22 August 2016].

Gilden, R. C., Huffling, K. and Sattler, B. (2010). Pesticides and health risks. *Journal of Obstetric, Gynecologic, & Neonatal Nursing* 39 (1), 103-110.

Hobbs, P. R., Sayre, K. and Gupta, R. (2008). The role of conservation agriculture in sustainable agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363 (1491): 543-555.

Hof, B., Koopmans, C., Rougoor, W. and van der Voort, J. (2013). *Effecten en Vormgeving van een Heffing op Gewasbeschermingsmiddelen*. Rapport nr. 2013-39. Amsterdam: seo Economisch Onderzoek.

Horowitz, J. K. and Lichtenberg, E. (1994). Risk-reducing and risk-increasing effects of pesticides. *Journal of Agricultural Economics* 45 (1): 82-89.

Jacquet, F., Butault, J.-P. and Guichard, L. (2011). An economic analysis of the possibility of reducing pesticides in French field crops. *Ecological Economics* 70 (9): 1638-1648.

MAAF (Ministère de l’Agriculture, de l’Agroalimentaire et de la Forêt) and MEDDE (Ministère de l’Écologie, du Développement durable et de l’Énergie) (2015). Plan Écophyto II. Paris: République Française.

Martin, E. (2015). Should we internalise inter-temporal production spillovers in the case of pest resistance? *European Review of Agricultural Economics* 42 (4): 539-578.

MIM (Miljøministeriet) and FVM (Ministeriet for Fødevarer, Landbrug og Fiskeri) (eds) (2013). Protect Water, Nature and Human Health – Pesticide Strategy 2013-2015. Copenhagen: Danish Ministry of the Environment Information Centre.

Möckel, S., Gawel, E., Kästner, M., Liess, M., Knillmann, S. and Bretschneider W. (2015). Einführung einer Abgabe auf Pflanzenschutzmittel in Deutschland. In

Gawel, E. (ed.), *Studien zu Umweltökonomie und Umweltpolitik*. Vol. 10. Berlin: Duncker & Humblot, 1-305.

Nielsen, H. (2005). Danish Pesticide Use Reduction Programme – To Benefit Environment and the Health. London: Pesticide Action Network Europe.

Norton, M., van Sprundel, G.-J., Turvey, C. G. and Meuwissen, M.P.M. (2016). Applying weather index insurance to agricultural pest and disease risks. International *Journal of Pest Management* 62 (3): 195-204.

Oerke, E. C. (2006). Crop losses to pests. *The Journal of Agricultural Science* 144 (1): 31-43.

Pedersen, A. B., Nielsen, H. Ø., Christensen, T. and Hasler, B. (2012). Optimising the effect of policy instruments: a study of farmers' decision rationales and how they match the incentives in Danish pesticide policy. *Journal of Environmental Planning and Management* 55 (8): 1094-1110.

Pimentel, D. (2009). Environmental and economic costs of the application of pesticides primarily in the United States. In Peshin, R. and Dhawan, A. K. (eds), *Integrated Pest Management: Innovation-Development Process*. Springer Science+Business Media: Dordrecht, 89-111.

Serra, T., Zilberman, D. and Gil, J. M. (2008). Differential uncertainties and risk attitudes between conventional and organic producers: the case of Spanish COP farmers. *Agricultural Economics* 39 (2): 219-229.

Skevas, T. Stefanou, S. E. and Oude Lansink, A. (2012). Can economic incentives encourage actual reductions in pesticide use and environmental spillovers? *Agricultural Economics* 43 (3): 267-276.

Skevas, T., Oude Lansink, A .G. J. M. and Stefanou, S. E. (2013). Designing the emerging EU pesticide policy: A literature review. *NJAS – Wageningen Journal of Life Sciences* 64-65: 95-103.

Skevas, T., Stefanou, S. E. and Oude Lansink, A. (2014). Pesticide use, environmental spillovers and efficiency: A DEA risk-adjusted efficiency approach applied to Dutch arable farming. *European Journal of Operational Research* 237: 658-664.

Travisi, C. M. and Nijkamp, P. (2008). Valuing environmental and health risk in agriculture: A choice experiment approach to pesticides in Italy. *Ecological Economics* 67 (4): 598-607.

Vatn, A., Kvakkestad, V. and Rørstad, P. K. (2002). Policies for Multifunctional Agriculture – The Trade-off between Transaction Costs and Precision. Report no. 23. Ås, Norway: Agricultural University of Norway – Department of Economics and Social Sciences.

Wiebers, U. C., Metcalfe, M., Zilberman, D., Hall, D. C. and Moffitt, L. J. (2002). The influence of pest management advice on pesticide use in California tomatoes. In Hall, D. C. and Moffitt, L. J. (eds), *Economics of pesticides, sustainable food production, and organic food markets*. Amsterdam: Elsevier Science, 81-98.

Wu, J. (1999). Crop insurance, acreage decisions, and nonpoint-source pollution. *American Journal of Agricultural Economics* 81 (2): 305-320.

Zhengfei, G., Oude Lansink, A., Wossink, A. and Huirne, R. (2005). Damage control inputs: a comparison of conventional and organic farming systems. *European Review of Agricultural Economics* 32 (2): 167-190.

Zilberman, D., & Millock, K. (1997). Financial incentives and pesticide use. *Food Policy* 22 (2): 133-144.