Ecological Economics 134 (2017) 263–266

Contents lists available at ScienceDirect

Ecological Economics

journal homepage: www.elsevier.com/locate/ecolecon

Revisiting Pesticide Taxation Schemes

Robert Finger [a,*], Niklas Möhring [a], Tobias Dalhaus [a], Thomas Böcker [b]

[a] Agricultural Economics and Policy Group, ETH Zurich, Switzerland
[b] Production Economics Group, University of Bonn, Germany

ARTICLE INFO

Article history:
Received 17 September 2016
Received in revised form 22 November 2016
Accepted 7 December 2016
Available online 26 January 2017

JEL Classification:
Q12
Q52
H23

Keywords:
Pesticide
Tax
Incentive
Environment
Regulation

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.

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.

2. Goals and Effectiveness of Policies

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, Ministère de

* Corresponding author at: Sonneggstrasse 33, 8092 Zurich, Switzerland.
E-mail address: rofinger@ethz.ch (R. Finger).

[1] Moreover, a quota system has been introduced in France in 2016 (Bellassen, 2015). See Möckel et al. (2015) and Finger et al. (2016a) for recent discussions in Germany and Switzerland.

http://dx.doi.org/10.1016/j.ecolecon.2016.12.001
0921-8009/

l'Agriculture, de l'Agroalimentaire et de la Forêt and MEDDE, Ministère de l'Écologie, du Développement durable et de l'Énergie, 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, Miljøministeriet and FVM, Ministeriet for Fødevarer, Landbrug og Fiskeri, 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.

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.

3. Pesticide Taxes as Part of a Coherent Set of Policies

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