Evaluating alternative policies to reduce pesticide groundwater pollution in Dutch arable farming

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This article develops a dynamic optimal control model of farmers' production decisions and applies it to panel data from Dutch arable farms to assess the effectiveness of different policy interventions in reducing pesticide groundwater pollution. Three different policy measures are examined in turn: namely, a flat tax on pesticides; a groundwater contamination tax; and a quantity restriction on pesticide use. The examined policies are compared against both quantitative and qualitative criteria drawn from the pesticide policy literature. Results show that the groundwater contamination tax is the most preferred policy for reducing pesticide groundwater pollution. A way to apply such a tax is proposed and discussed.

Keywords: groundwater pollution; pesticides; accumulating pollutant; environmental policy; dynamic optimization

1. Introduction

Groundwater is an important natural resource since it accounts for over 97% of all available freshwater in the world (excluding ice caps and glaciers) (European Commission 2008; IGRAC 2017). In several parts of the world, human communities depend more on groundwater than surface water for their water needs. For instance, in the European Union (EU) about 75% of the population depends on groundwater for its water supply (European Commission 2008). Groundwater is also important for agriculture, as it is used to irrigate crops and water livestock (Pfeiffer and Lin 2014). For example, in the US, agriculture accounts for 66% of total groundwater withdrawals (OECD 2015). Groundwater is also used for industrial purposes, such as cooling water for electricity plants and food processing (Bergkamp and Cross 2006). Apart from being an important resource for drinking water, agriculture, and industry, groundwater has a significant influence on the ecology of surface water systems. Groundwater provides the base flow of rivers and streams (Gunn 2007), which are often used as drinking water sources and play a key role in the health of aquatic ecosystems. Therefore, it is important that groundwater is managed efficiently and protected for future generations.

Despite efforts to protect groundwater resources, groundwater depletion and pollution are growing at alarming rates in many parts of the world. Agriculture is one of
the main contributors to decreased groundwater quantity and quality. Several studies have pointed to agricultural withdrawals as the main reason for aquifer depletion around the world (Ahmed and Umar 2009; Guzman-Soria et al. 2009; Konikow and Kendy 2005; Pfeiffer and Lin 2014; Rodell et al. 2009; Scanlon et al. 2007; Wang et al. 2009). Concerning groundwater quality, nitrates and pesticides are found increasingly in groundwater aquifers (Andrade and Stigter 2009; Chowdary et al. 2005; Gilliom et al. 2006; Sjerps et al. 2017 Szoege 1996), while over-pumping has also shown to contribute to decreased groundwater quality of coastal aquifers through increased salt-water intrusion (Lambrakis and Kallergis 2001). Since groundwater moves slowly through the aquifer, the impact of contaminants generated from human activities may persist for a long time (European Commission 2008). Therefore, groundwater policies should focus on preventing pollution rather than addressing it after it has occurred.

Groundwater pollution prevention has been the focus of agricultural policies in developed countries. For instance in the US, the federal government operates voluntary programs, such as the Environmental Quality Incentives Program, that offer eligible landowners and farmers technical and financial assistance to encourage adoption of best management practices (BMPs) to protect environmental quality (including groundwater quality) (USDA 2014). In EU, the Water Framework Directive (WFD) (Directive 2000/60/EC), and especially the subsequent Groundwater Directive (Directive 2006/118/EC) established a framework that requires Member States to set environmental objectives for groundwater chemical status and take measures to prevent or limit inputs of pesticides and nitrates into groundwater. A recent evaluation of the EU water policies highlights the need for using economic instruments to internalize the external costs from degradation of ecological and chemical status of water bodies (European Commission 2012). However, to date, only a few EU countries have adopted economic incentive-based policies to protect the environment, including groundwater (these countries are Sweden, Denmark, Norway, and France). With groundwater bodies being increasingly at risk (Sjerps et al. 2017; Urresti-Estala et al. 2016; Van Grinsven, Tikta and Rougoor 2016), stricter national policies would be needed to enhance groundwater quality. In order for policy makers to use economic incentives or command and control regulations as part of their effort to reduce groundwater pollution from agrochemicals, information on how such measures would affect farmers’ economic performance and groundwater quality are needed. Yet farm-level instrument-specific impact assessments on this issue remain underexplored in the literature.

The literature presents only studies that use either aggregate (region-level), experimental, or representative farm data to assess the effect of economic incentives or command and control regulations on the use of polluting inputs and groundwater quality. Most of these studies focus on groundwater pollution from nitrates (Chowdhury and Laceywell 1996; Fishman et al. 2012; Huang and LeBlanc 1994; Johnson, Adams, and Perry 1991; Lee and Kim 2002; Peña-Haro et al. 2010; Slabe-Erker et al. 2017) and very few examine pesticide groundwater pollution (Anderson, Opaluch and Sullivan 1985; Archer and Shogren 2001; Conrad and Olson 1992; Slabe-Erker et al. 2017). Farm-level studies examining the impact of different policy instruments on pesticide use and its environmental effects exist (Falconer and Hodge 2001; Skevas, Stefanou, and Lansink 2012), but do not specifically focus on groundwater pollution (i.e. a stock-pollutant problem). Against this background, the objective of this study is to assess the effectiveness of different economic incentives and command and control regulations in reducing the risk that agricultural pesticide use poses to groundwater quality. I address this objective by using a dynamic optimal control model of a farmers’ production decisions in the context of
different policy scenarios related to groundwater protection. The empirical application focuses on a sample of Dutch arable farms. The case of Dutch groundwater pollution from agricultural pesticide use is of interest since recent research has shown the presence of pesticides in Dutch groundwater at levels that exceed the EU and Dutch permissible limit for drinking water (Sjerps et al. 2017). The use of farm-level data allows an improved representation of farmers’ decision making and environmental management under groundwater policy scenarios and provides more accurate recommendations for improving the existing groundwater policies or implementing new ones. Another aspect that makes this study different from other studies in the pesticide policy literature is that it focuses on a variety of policy measures, including a flat tax on pesticides, a groundwater contamination tax, and a quantity restriction on the use of pesticides. By doing so it strengthens the knowledge base on the effectiveness of different policy interventions in controlling groundwater pollution, thus assisting water policy makers in achieving their objectives.

The paper continues in Section 2 by presenting the conceptual model. Section 3 discusses the empirical framework and the data are presented in Section 4. The results are given in Section 5, and discussion and conclusions are presented in Section 6.

2. Conceptual framework

2.1. A net private benefit model of agricultural production and groundwater pollution

Consider a farmer \( i \) who uses pesticides \( z_{it} \), other intermediate inputs \( x_{it} \) (e.g. seeds), and fixed inputs \( k_{it} \) (e.g. farm capital) in year \( t \) to produce multiple crop outputs \( y_{it} \). The use of pesticides results in groundwater contamination \( G \) through leaching. Farmers are assumed to know the risk of leaching of different pesticides to groundwater and could potentially choose the least polluting crop protection strategy (see section 4.1 for more details on this assumption). Pesticides move into groundwater where they accumulate or degrade. Following Yadav (1997), the dynamics of the groundwater pollution stock are determined by the leaching rate of surface application of pesticides to the groundwater aquifer (\( c_{zt} \)), and the degradation rate of the existing pesticide concentration in the groundwater aquifer (\( d_{Gt} \)):

\[
\dot{G} = \gamma z_t - \delta G_t
\]

where \( \dot{G} \) is the time derivative \( \partial G/\partial t \), \( \gamma \) is a parameter describing conversion of applied pesticide into groundwater pollution, \( \delta \) is the degradation rate of pesticides at the groundwater level, and \( t \) is time, \( t \in [0, T] \). The equation of motion for groundwater pollution \( G \), relates pesticide use to groundwater pollution by providing the mechanism through which farmers’ choice of the control variable \( z \) can be translated into a specific pattern of movement of the state variable \( G \). This equation further introduces a dynamic interdependency across periods, where each period starts off with a specific groundwater pollution status which has been shaped from previous periods’ pesticide use and natural degradation of the pollutant.

In the absence of an environmental protection agency (EPA) that regulates groundwater pollution, farmers are maximizing their private net benefit function from the use of pesticides in crop production (i.e. farms will undertake zero abatement). Assuming a given initial level of groundwater pollution \( \Psi \), the problem can be expressed as a dynamic optimization of the present value of private
net benefits:

\[ \max_{z_t, x_t} \int_0^T \left( p_1 y(z_t, x_t, k_t) - w_1 z_t - w_2 x_t \right) e^{-\rho t} \, dt \]

s.t.
\[ \dot{G} = \gamma z_t - \delta G_t \]
\[ -z_t \leq 0, -x_t \leq 0 \]
\[ -G_t \leq 0 \]

and \( G_0 = \Psi, G_T = \text{free} \), \((\Psi, T_{\text{given}})\)

where \( p, w_1, \) and \( w_2 \) are output, pesticide, and other intermediate input prices, respectively, and \( \rho \) is a discount factor. The constraints in (2) include the equation of motion (\( \dot{G} \)), the control variable constraints (i.e. non-negativity of \( z \) and \( x \)), a state space constraint (i.e. non-negativity of \( G \)), and a set of initial and terminal conditions for the state variable.

The following subsections (i.e. 2.2–2.4) show how problem (2) changes when assuming the presence of an EPA that introduces different policy measures to reduce groundwater pollution. More specifically, three policy options are explored: (a) a flat tax on pesticides, (b) a groundwater contamination tax, and (c) a quantity restriction on pesticide use. The feasibility of each policy scheme is discussed later in the paper as part of the policy evaluation section (i.e. Section 5.3).

2.2. A flat tax on pesticides

Suppose that an EPA, in an effort to incentivize farmers to use pesticides more judiciously with the ultimate goal of preventing pesticide groundwater pollution, decides to impose a fixed tax \((\phi)\) per unit of pesticide. This tax, which addresses pesticide use intensity and is similar to the fixed levy per spray unit employed by Falconer and Hodge (2001), will increase pesticide price to \( \hat{w}_1 t \) (where \( \hat{w}_1 t = w_1 t + \phi \)). This implies that problem (2) remains the same, except that \( w_1 t \) is replaced by \( \hat{w}_1 t \).

2.3. A groundwater contamination tax

Alternatively, if the EPA applies a tax on emissions from pesticide application to groundwater (i.e. a groundwater contamination tax)\(^2\), problem (2) remains the same, except that the objective function incorporates an additional term: \( \max_{z_t} \int_0^T \left( p_1 y(z_t) - w_1 z_t - \eta G_t \right) e^{-\rho t} \, dt \), where \( \eta G_t \) is the groundwater pollution cost to society resulting from pesticide use. The groundwater contamination tax is essentially a Pigouvian tax (i.e. a charge per unit of emissions generated by a firm (Spulber 1985)).

Both the flat and the groundwater contamination tax are expected to induce farmers to reduce pesticide use and therefore groundwater pollution. However, whether these taxes have the same or a different effect in attaining pollution reduction is a question for empirical investigation.

2.4. A quantity restriction on pesticide use

Suppose that the EPA does not have exact knowledge of the treatment cost of pesticide contaminated drinking water (i.e. marginal social damage) and thus, is unable to
introduce a tax that internalizes the external cost of pesticides. It therefore decides
to introduce a quantity restriction on the use of pesticides (i.e. a command and con-
trol measure). Such a restriction introduces an additional control variable constraint
in problem (2), namely $z_t \leq Q$, where $Q$ is the maximum permissible level of pesti-
cide use after the restriction takes effect. This command and control measure can
be viewed as a quota regime, under which farmers are facing a certain reduction in
pesticide use.

Quantity restrictions on pesticide use are expected to reduce farm-level pesticide
applications and, as a result, reduce groundwater pollution. Meanwhile, such a restric-
tion may have a negative effect on farm profitability in high pest infestation years,
where increased pesticide use may be required to combat pest damage. Whether a
quantitative restriction on pesticide use would cause a significant decrease in ground-
water pollution but at the same time impact negatively on farm income is thus a ques-
tion that warrants empirical research.

The next section discusses the criteria that will be used to determine the best policy
option for addressing pesticide groundwater pollution.

2.5. A framework for the evaluation of the examined policies

Researchers have reported several criteria for evaluating the performance of different
policy options in reducing pesticide use and its potential adverse effects on human health
and the environment (Falconer 1998; Oskam et al. 1998; Reus, Weckseler, and Pak
1994). These criteria are the following: (a) effectiveness, (b) efficiency, (c) feasibility
and maintainability, (d) polluter pays principle, (e) institutional homogeneity, (f) eco-
nomic consequences for farmers, and (g) acceptability. Effectiveness refers to the extent
to which a policy instrument achieves its objectives. It is assumed that the main object-
ive of the examined policies is to reduce pesticide groundwater pollution. Efficiency
refers to the cost of achieving the desired goal. Feasibility and maintainability refer to
the extent to which a policy instrument is practical or possible to implement. The pol-
luter pays principle refers to the requirement that the cost of pollution is borne by the
generator or generators of pollution. Institutional homogeneity refers to the requirement
that the policy tool is compatible with particular policy principles embedded in other
governmental programs (Oskam et al. 1998). For instance, in the case of an EU member
state introducing a pesticide policy, institutional homogeneity implies no conflict
between national regulations and EU pesticide policies. Economic consequences for
farmers refer to economic losses related to the introduction of policy instruments that
can be used to oppose policy interventions. Finally, acceptability refers to the desirability
of policy interventions to involved stakeholders (e.g. farmers and farm organizations).

The nature of this study does allow one to empirically evaluate the performance of
the examined policy instruments based on the criteria of effectiveness and economic
consequences for farmers. However, a critical discussion of each policy’s performance
relative to all the criteria mentioned above will be provided.

The next section discusses specification details that enable me to obtain a quantita-
tive solution of the optimization problems described in sections 2.1–2.4.
3. Empirical framework

3.1. Specification issues

The production technology \( y \) in (2) is represented by the following Cobb-Douglas\(^3,4 \) production function:

\[
y_t = e^{a_0} x_t^{a_1} z_t^{a_2} \prod_{j=1}^{J} k_j^{a_3} \tau_t^{a_4} (f^2)^{a_5}
\]  

(3)

where \( y, z, x, \) and \( k \) are firm and time \( (t) \) varying output, pesticides, other intermediate inputs, and a vector of fixed inputs, respectively. Subindice \( j (j = 1 \ldots J, \text{ with } J = 3) \) denotes the index set for fixed inputs. More specifically quasi-fixed inputs include labour \( (k_1) \), capital \( (k_2) \), and land \( (k_3) \). The alphas and betas are parameters to be estimated. Finally, a time trend is included to capture technological progress, while its squared term allows for non-linear effects.

Following Equation (1), the groundwater contamination function is specified as follows:

\[
G_{t+1} = c z_t - \delta G_t \tag{4}
\]

Parameters \( \gamma \) and \( \delta \) determine the leaching rate of surface pesticide applications to groundwater, and the natural degradation rate of pesticides in the groundwater, respectively.

Based on the specifications of the production technology and the groundwater contamination function, the following subsections present the optimal solution of the farmer’s problem under the policy-off and different policy-on scenarios. Detailed derivation of the optimal solutions of all problems is presented in the Supplementary Material.

3.2. Solving the net private benefit problem

The optimal solution of the net private benefit model (i.e. Equation(2)) is characterized by the following conditions:

\[
A_t^* = 0 \tag{5}
\]

\[
z_t^* = \left( \frac{w_1}{p_1 e^{(x_0 + x_1 z_1 z_2^2) \beta_1} \prod_{j=1}^{J} k_j^{\beta_3} \tau_t^{2 \beta_4}} \right) \left( \frac{1}{r_2 - r_1} \right) \left( \frac{1}{r_3 - r_1} \right) \left( \frac{r_1}{(r_1 - 1) (r_2 - 1)} \right) \tag{6}
\]

\[
x_t^* = \left( \frac{w_2}{p_2 e^{(x_0 + x_1 z_1 z_2^2) \beta_1} \prod_{j=1}^{J} k_j^{\beta_3} \tau_t^{2 \beta_4}} \right) \left( \frac{1}{r_1} \right) \tag{7}
\]

\[
Y_t^* = \left( \frac{\Psi - \frac{\gamma z_t^*}{\delta}}{\delta} \right) e^{-\delta t} + \frac{\gamma z_t^*}{\delta} \tag{8}
\]

The Lagrange multiplier \( A_t^* \) gives the change in the maximized value of profit due to an infinitesimal increase in groundwater pollution. This change is zero here, since farmers are not constrained by pollution regulations.
3.3. Solving the flat tax problem

When a tax is applied on the level of the polluting input \((z)\) then, the optimal solution of the pesticide tax model is similar to that of the net private benefit model (i.e. Equations(5–8)) with the exception that \(w_{1t}\) in (6) is replaced by \(\tilde{w}_{1t}\) (where \(\tilde{w}_{1t} = w_{1t} + \phi\)). This implies that the pesticide price is augmented by the tax rate \(\phi\) to correct the negative externality arising from groundwater pollution.

3.4. Solving the groundwater contamination tax problem

When a tax is applied on groundwater contamination then the optimal solution is characterized by the following conditions:

\[
\frac{K}{C_{3t}} = \frac{Q}{\gamma} \frac{1}{\rho_1} \quad (13)
\]

\[
x_{it} = \frac{w_{2t}}{p_t e^{(x_0 + x_1 + x_2) / \rho_1} \prod_{j=1}^{J} k_{ij}^{\beta_j} (t^2)^{x_2}} \quad (14)
\]

\[
G_{it} = \left( \Psi - \frac{\gamma x_{it}}{\delta} \right) e^{-\delta t} + \frac{\gamma x_{it}}{\delta} \quad (15)
\]

The Lagrange multiplier (\(\Lambda_t^*\)) in this case can be interpreted as the shadow cost of a unit of groundwater pollution.

3.5. Solving the pesticide quantity restriction problem

The optimal solution of the pesticide quantity restriction problem is characterized by the following conditions:

\[
\Lambda_t^* = 0 \quad (13)
\]

\[
x_{it}^* = \frac{w_{2t}}{p_t e^{(x_0 + x_1 + x_2) / \rho_1} \prod_{j=1}^{J} k_{ij}^{\beta_j} (t^2)^{x_2}} \quad (15)
\]

\[
G_{it}^* = \left( \Psi - \frac{\gamma Q}{\delta} \right) e^{-\delta t} + \frac{\gamma Q}{\delta} \quad (16)
\]
\[ \Theta' = p_t e^{(z_0)} x_t^{\beta_1} \beta_2 Q^{\beta_2 - 1} \prod_{j=1}^{J} k_{ij}^{\beta_j} t^{x_{ij}} (J^2)^{\gamma_2} - w_{1t} \] (17)

In the pesticide quantity restriction problem, farmers face a constraint on the quantity of pesticides used. Therefore, \( \Lambda^*_t \), which reflects the change in maximum profit when the groundwater pollution increases by one unit, is zero. On the other hand, \( \Theta^t \) reflects the change in the maximized value of profit when relaxing the pesticide quantity restriction constraint by one unit.

4. Data

4.1. Data on farm practices and groundwater pollution

This study uses a dataset that is similar to the one used by Skevas, Stefanou, and Lansink (2012) and comes from the Farm Accountancy Data Network (FADN). The dataset is a balanced panel of 91 Dutch arable farms for the 2002–2007 period, corresponding to 546 observations. Table 1 provides descriptive statistics for the variables used in the study. One output and five inputs are distinguished.

Output \((y)\) consists of deflated revenues from sales of root crops (i.e. potatoes, carrots, onions, and sugar beets), cereals (i.e. barley, wheat, triticale, rye, corn, and oats), and other crops (i.e. grass seed, and green peas and beans). These revenues are deflated using a Törnqvist index based on output prices from Eurostat (Eurostat 2017), using 2005 as the base year. Inputs include pesticides \((z)\), other intermediate inputs \((x)\), and fixed inputs \((k)\) (i.e. capital, labour, and land). The pesticide variable is measured as expenditures on plant protection products deflated to 2005 using its own price index from Eurostat. The sample farms use a wide variety of pesticide products to reduce crop damage caused by pests and diseases. The majority of pesticide applications are to potatoes. Potato production requires the use of significant amounts of fungicides to combat Phytophthora infestans, which is the causal agent of one of the most severe and economically important diseases to potato production, late blight. Other intermediate inputs consist of five subcategories of inputs: fertilizer, energy, seeds, water, and other crop-specific costs. A Törnqvist index was constructed using price indices and expenditures for each input subcategory. The total reported value was deflated to 2005 values using the Törnqvist index. Capital includes the replacement value of buildings, installations, and machinery, deflated to 2005 values using a Törnqvist index which is constructed using expenditures and price indices for each capital subcategory. Labour is measured in man-hours and consists of family and hired labour. Land is measured in hectares and includes both owned and rented land.
The groundwater pollution indicator \( G \), which is a measure of the risk of pesticide leaching to groundwater, was calculated using pesticide quantity data and pesticide risk data drawn from the Centre for Agriculture and Environment (CLM 2007). It is measured in impact points and calculated for each farm as the sum of the impact points of individual pesticide applications. The pesticide-specific impact points are computed taking into account the actual applied quantity of active ingredient per hectare, the fields’ organic matter content, the characteristics of the pesticide under consideration (e.g. degradation rate, mobility in soil), and the season of application (which is Spring in this study). Dutch farmers can use the CLM web page to compare pesticides in terms of groundwater pollution and other environmental pressures and choose the least harmful crop protection strategy. This tool can be also used by farmers in other countries to make relative comparisons of pesticides in terms of their environmental impacts (CLM 2007). Finally, the discount rate is assumed to be 0.044, approximating the average bank lending rate in the Netherlands during the period 2003–2013.

### 4.2. The parameter values of the policy scenarios

Some EU countries have adopted economic-incentive based policies in order to reduce pesticide use and its adverse effects on human health and the environment. Sweden has adopted a flat tax on pesticides where a fixed amount is charged per kg of active ingredient. Denmark, France, and Norway have introduced differentiated pesticide tax schemes based both on the intensity of pesticide use and on the environmental and health risks of pesticides (Pedersen, Nielsen, and Andersen 2015; Skevas, Stefanou, and Lansink 2012; UNDP 2017). Based on tax rate increases reported in Swedish and Danish pesticide tax schemes, the flat tax on pesticides is realized by increasing the price of pesticides by 10, 30, and 50%. In the absence of estimates of the external cost of groundwater contamination for the Netherlands, the groundwater contamination tax scenarios employed in this study are based on values of the external cost of pesticides in sources of drinking water for the UK and Germany found in the study by Leach and Mumford (2008). These authors report that the cost of pesticides in sources of drinking water is €9.66 and €5.15 per kg active ingredient for the UK and Germany, respectively. Departing from these estimates, three pesticide emission tax levels are considered in this study: €10, €15, and €20, per impact point. Finally, following Skevas, Stefanou, and Lansink (2012), quantity restrictions on pesticide use involve farms facing 10, 20, and 30% reductions in pesticide use.

### Table 2. Estimated coefficients of the production function.

| Parameter                  | Estimate | p-value |
|----------------------------|----------|---------|
| Other intermediate inputs  | 0.30     | 0.000   |
| Labour                     | 0.10     | 0.000   |
| Capital                    | 0.17     | 0.000   |
| Land                       | 0.32     | 0.000   |
| Pesticides                 | 0.25     | 0.000   |
| Trend                      | 0.01     | 0.264   |
| Trend squared              | −0.04    | 0.002   |

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5. Results

5.1. Production technology of Dutch arable farms and pesticide contribution to groundwater pollution

Parameter estimates from estimating the production and groundwater contamination functions are presented in Tables 2 and 3. Concerning the production technology, the parameter estimates are interpreted as elasticities. All inputs have a significant positive impact on production at the 1% significance level. Other intermediate inputs and land have greater elasticities than the rest of the inputs, implying that these inputs, as expected, play an important role in crop production (Skevas, Stefanou, and Lansink 2012). The elasticity of pesticides is 0.25 and is within the range of pesticide elasticities reported in other studies (see the meta-analysis by Skevas, Lansink, and Stefanou (2013)). The parameter estimate of the time trend squared is negative and significant, suggesting a tendency for production to fall in the long run.

Moving to the estimates of the groundwater contamination function, pesticides have a significant positive contribution to groundwater contamination. This implies that pesticides leach through the soil to the groundwater zone. Finally, the estimated value of the degradation rate, $\delta$, is 0.36 and is significant at the 1 percent level. This implies that groundwater has a self-cleansing mechanism for pesticide pollutants. The groundwater contamination function estimates reported here, are close to the estimates reported in a study by Yadav (1997) that examined the impact of nitrates on groundwater pollution.

5.2. Policy scenarios

Table 4 presents the production decisions of profit-maximizing farms (i.e. net private benefit or policy-off scenario) and the average effect of different policy scenarios on farmers’ profitability, pesticide, and other intermediate inputs’ use, and groundwater pollution.

A 10% rise in the price of pesticides reduces pesticide use by around 12%. Increasing the tax rate to 30% and 50% reduces pesticide use by 30% and 35%, respectively. The use of intermediate inputs experiences smaller decreases (i.e. 5–16%) as a result of increasing the price of pesticides. Groundwater pollution is also reduced by the introduction of different flat taxes. The degree of groundwater pollution reduction is in the range of 10–20%. Economic (i.e. farm profit) losses from the introduction of flat taxes on pesticides are not negligible (i.e. in the range of 16–19%).

Previous research on the effect of flat taxes on pesticide use and environmental risks is inconclusive, with some studies showing a significant decrease in pesticide usage or environmental risks (Archer and Shogren 2001) and others failing to do so (Falconer and Hodge 2001; Skevas, Stefanou, and Lansink 2012).

| Parameter$^a$ | Estimate | $p$-value |
|---------------|----------|-----------|
| $\gamma$      | 0.27     | 0.032     |
| $\delta$      | 0.36     | 0.000     |

Note: $^a$ $\gamma$ and $\delta$ denote the leaching rate of surface pesticide applications to groundwater, and the degradation rate of pesticides in the groundwater aquifer, respectively.
Concerning the groundwater contamination tax, a €10 (per impact point) tax on groundwater pollution reduces pesticide use by around 29%. Increasing the tax rate to €15, and €20, reduces pesticide use by 37% and 42%, respectively. Smaller but significant reductions in other intermediate inputs’ use are caused by the different pollution taxes. More specifically, a €10, €15, and €20 tax on pollution reduces the use of other intermediate inputs by 22, 30, and 35%, respectively. Pollution taxes also yield significant reductions in groundwater pollution. More specifically, a €10, €15, and €20 tax on pollution reduces groundwater pollution by 13, 22, and 27%, respectively. These reductions are larger than those under a flat tax on pesticide use, a finding that is in line with theoretical expectations (Perman 2003). The considerable reductions in groundwater pollution do not occur without cost. The large reductions in pesticide use and other intermediate inputs cause large decreases in farm profitability — under the three pollution tax scenarios farm profit is decreased in the range of 26–34%. The shadow cost of a unit of groundwater pollution is €-13, €-20, and €-27 impact point⁻¹, in the €10, €15, and €20 tax scenario, respectively.

Moving to the quantity restriction on pesticide use scenarios, a 10, 20, and 30% cut in pesticide use reduces other intermediate inputs’ use by 19, 22, and 26%, respectively. Farm profit falls by 9–12% when introducing pesticide cuts in the range of 10–30%. Concerning groundwater pollution, reducing pesticide use by 10% reduces groundwater pollution by around 8%. Increasing the cut to 20 and 30% reduces groundwater contamination by 25 and 34%, respectively. Skevas, Stefanou, and Lansink (2012) also find that pesticide quotas are effective in reducing pesticide environmental spillovers. The value of the Lagrange multiplier (\(\Theta\)), is €12, €13, and €14 for the 10, 20 and 30% cut in pesticide use, respectively. These estimates reflect the marginal profit of relaxing the quantity restriction constraint by one unit.

Under all policy interventions, decreases in pesticide use are accompanied by reductions in the use of other intermediate inputs. Since farmers face restrictions in pesticide use, they may reduce fertilizers, as the latter may boost the growth of non-target plants that then require increased use of herbicides. Another explanation may be that reductions in pesticide use lead to reductions in the use of spraying equipment.

Table 4. Results of policy scenarios.

| Scenarios                          | Profit | Z     | X     | G     |
|-----------------------------------|--------|-------|-------|-------|
| Policy-off scenario               | 127.46 | 20.37 | 52.67 | 10.27 |
| Flat tax on pesticides            |        |       |       |       |
| 10% price increase                | -15.74 | -11.93| -4.92 | -9.88 |
| 30% price increase                | -17.05 | -29.55| -12.97| -12.95|
| 50% price increase                | -18.70 | -35.49| -15.97| -20.25|
| Groundwater contamination tax     |        |       |       |       |
| € 10/impact point                 | -26.09 | -28.62| -22.21| -13.34|
| € 15/impact point                 | -30.35 | -37.46| -30.21| -22.30|
| € 20/impact point                 | -33.59 | -42.07| -35.35| -26.97|
| Quantity restriction on pesticide use |        |       |       |       |
| 10% cut                           | -9.58  | -10.01| -18.74| -7.79 |
| 20% cut                           | -10.46 | -20.03| -22.46| -25.12|
| 30% cut                           | -11.63 | -30.00| -26.45| -34.47|

Note: Z, X and G denote pesticides, other intermediate inputs, and groundwater pollution respectively.
and therefore decreased use of energy (e.g. motor fuels). A decrease in the use of other intermediate inputs when farmers face restrictions in the use of pesticides is not an uncommon finding in the literature (Oude Lansink 1994; Skevas, Stefanou, and Lansink 2012). However, there are cases where a substitution effect is present (i.e. a cut in pesticides leads farmers to increase the use of other inputs) (Oude Lansink 1994). Such an effect may be the result of farmers’ efforts to substitute pesticides with pest resistant seeds or mechanical weeding (that leads to increased use of energy).

Finally, a sensitivity analysis has been conducted to investigate the impact of higher taxes and pesticide use restrictions on farm-level decision-making and groundwater pollution. The results obtained, which are available upon request, do not alter the ranking of the various policy options (in terms of reductions in all variables) shown in Table 4. However, further decreases in both farm profit and groundwater pollution (for all policy scenarios) are reported. These findings make it clear that using very high taxes or quotas on pesticide use as stand-alone measures will be politically problematic, since they shrink farm income. It has to be mentioned that higher tax rates and quota restrictions than the ones used in the present study are probably not very realistic, since they are larger than typical tax increases and restriction targets reported in EU countries that have introduced fiscal instruments to reduce pesticide use and environmental spillovers (Danish Ecological Council 2015; UNDP 2017).

### 5.3. Policy evaluation

In Table 5, an evaluation of the examined policies based on the criteria introduced in section 2.5 is provided. The effectiveness of all policies is relatively high (i.e. ++ sign in Table 5), since they all lead to considerable reductions in groundwater pollution and pesticide use. The flat tax is the most efficient of the examined policy interventions (i.e. +++ sign) given its low administrative cost. The remaining policy tools entail higher administrative costs and therefore their efficiency is low (i.e. – sign).

The flat tax is also easy to implement and control (it only requires implementation and control at the level of pesticide suppliers). Therefore, its feasibility and

| Criteria                        | Flat tax on pesticides | Groundwater contamination tax | Quantity restriction on pesticide use |
|---------------------------------|------------------------|-------------------------------|-------------------------------|
| Effectiveness                   | ++                     | ++                            | +++                           |
| Efficiency                      | +++                    | –                             | –                             |
| Feasibility/                     | +++                    | –                             | –                             |
| Maintainability                 | –                      | –                             | –                             |
| Polluter pays principle         | –                      | +++                            | –                             |
| Institutional                   | –                      | +++                            | ++                            |
| homogeneity                     | –                      | –                             | –                             |
| No economic consequences        | –                      | –                             | –                             |
| for farmers                     | –                      | –                             | –                             |
| Acceptability                   | –                      | ++                            | –                             |

Notes: The symbols denote as follows: +++: very high/very good, ++: high/good, –: low/bad, –: very low/very bad.
maintainability are characterized as very high. It is also argued that a version of the groundwater contamination tax (which is convenient for practical implementation) could be potentially implemented at the level of pesticide suppliers. Using the CLM environmental impact data, an impact point value per kg could be assigned to each pesticide (i.e. active substance) allowed to be traded in the market. This process will classify pesticides according to their groundwater pollution potential and provide the base for applying a groundwater contamination tax. The groundwater contamination tax, which is essentially a price penalty per impact point, will make pesticides that cause the highest groundwater pollution the most expensive. This is essentially a penalty on farmers’ potential rather than actual groundwater pollution, which is difficult to measure. This version of the groundwater pollution tax resembles the current Danish pesticide differentiated tax system where taxes are based on quantities and properties (i.e. environmental and health effects) of sold pesticides (Danish Ecological Council 2015).

In contrast, the feasibility and maintainability of the pesticide use quota is described as low; this is because its implementation requires collecting farm-level data on pesticide use which are not easily available. However, administrative apparatus that would have been potentially able to collect pesticide use data exist in developed countries. For instance, EU farmers receive direct payments after reporting farm-level information, such as the exact location and size of their holdings, land use, and crops grown (European Commission 2017). These apparatus could be further developed to include the collection of pesticide use data. Such data could, for instance, help to apply a pesticide quota (e.g. by including a desired reduction in the total use of pesticides as an additional criterion for receiving direct payments).

The groundwater pollution tax is in line with the polluter pays principle, since the more a farmer purchases highly polluting pesticides the higher the tax, he/she pays. On the other hand, the flat tax scheme does not consider the polluter pays principle (i.e. — sign in Table 5). Concerning the institutional homogeneity criterion, the groundwater contamination tax has the highest performance since it is in line with existing differentiated pesticide tax schemes (i.e. the Danish, French, and Norwegian schemes) and recent EU water policy evaluations (i.e. Blueprint to Safeguard Europe’s Water Resources) that highlight the need for using economic instruments to internalize the external costs of polluting activities. Restrictions on polluting inputs’ use are also part of the tools that the WFD and the Plant Protection Products Directive identified as having the potential to be effective in reducing pesticide use and related risks. The flat tax is the worst performer with respect to institutional homogeneity, since it is not in line with current knowledge on the design of pesticide taxes (i.e. differentiated taxes).

Moving to the economic consequences for farmers, all measures result in considerable reductions in farm profitability. Finally, the acceptability of flat taxes on pesticide consumption is usually low among farmers (Oskam et al. 1998). Quantity restrictions on pesticide use are also unlikely to receive support among farmers, since they may reduce farmers' competitiveness in producing high pesticide demanding crops where preventive pesticide applications are very important (e.g. potatoes, fruit, and vegetables). The acceptability of the groundwater contamination tax is characterized as good. This is because farmers and other agricultural stakeholders are more likely to accept a tax that is based on the environmental load of pesticides rather than on pesticide consumption. This is the case in Denmark, where farmer associations and the food industry were in agreement with the introduction of differentiated pesticide taxes according to health and
environmental risks (Danish Ecological Council 2015). Overall, it can be concluded that the groundwater contamination tax possesses the most advantages compared to the rest of the examined policy interventions.

6. Discussion and conclusions

Despite improvements in the quality of both surface and groundwater in some rural areas across the world, groundwater pollution from agriculture remains a cause for concern. Addressing this concern would require, among other things, the effective integration of different environmental policies (e.g. groundwater parts of the WFD and legislation in the agricultural sector), and the use of economic instruments for internalizing the external costs of polluting activities (European Commission 2012). By assessing the effectiveness of different policy instruments in reducing pesticide groundwater pollution, this study provides policy implications toward improving groundwater quality in Dutch (and other countries’) arable farming. Three policy instruments are examined, namely, a flat tax on pesticides, a groundwater contamination tax, and a quantity restriction on pesticide use.

Using evaluation criteria defined in the pesticide policy literature, it was found that the groundwater contamination tax performs better than the rest of the examined policy tools. Such a tax is in line with the polluter pays principle, recent developments in pesticide taxation (i.e. differentiated taxes), and the demands of the WFD. Following the Danish example, a groundwater contamination tax can be levied on pesticide sales after assigning a groundwater load value to each marketed pesticide product.

Although this research has identified a groundwater contamination tax as the preferred policy option for reducing pesticide groundwater pollution, it does not suggest that such a tax be applied as a stand-alone measure. In an effective pesticide policy, the groundwater contamination tax should coexist with other measures designed to reduce groundwater pollution from pesticide use. An example of such measures is education and training of farmers on secure pesticide use (e.g. education on the timing of pesticide applications in relation to rainfall) and awareness of the environmental and health hazards of pesticide applications (Falconer and Hodge 2001). These measures should, in fact, be in the forefront of any policy package designed to reduce pesticide groundwater pollution.

Moreover, implementing a broad array of policy interventions may better match different rationales for pesticide use and improve the effect of the overall policy (Pedersen et al., 2012). Of course, agricultural groundwater quality is not only affected by pesticide leaching but also by pollution caused by nitrates from agricultural sources (e.g. livestock waste and fertilizer use). Future research should assess the joint effectiveness of multiple policy interventions aiming at ensuring groundwater quality. Future work should also account for farmers’ heterogeneity in terms of education and training on secure pesticide use and environmental awareness of pesticide hazards and how this heterogeneity would influence their response to pesticide policy interventions. Finally, pesticide use can adversely affect not only groundwater quality but also other ecological processes and features of agricultural landscapes (e.g. bird habitat and distribution). Future pesticide policy studies should attempt to incorporate additional environmental parameters when modeling the effectiveness of pesticide policy measures in achieving environmental improvement.
Notes
1. To simplify notation, the subscript \( i \) is dropped throughout the remainder of this article.
2. Pollution taxes can be applied on the levels of polluting inputs, on emissions, or on output of
the final product (Perman 2003). I focus here on taxation of polluting inputs and emissions.
3. Both the Cobb-Douglas and a Translog production function were estimated. The likelihood
ratio test was used to compare these alternative models. It was found that the restrictions
of the Cobb–Douglas cannot be rejected (LR Chi² = 5.64; \( p \)-value = 0.431) implying that
the Cobb-Douglas function is favorable relative to the Translog function.
4. A Clarke test (Clarke 2007) was used to compare the model in (3) with the following
damage-abatement model:
\[
y_{it} = e^{(a_0)}x_{it}^{b_1}(1-e^{(-z_0+b_1)}) \prod_{j=1}^{k} q_{ijt}^{(a_2)}(i^2)^{b_2}
\] Clarke test results supported the selection of model (3) against the alternative damage-abatement model
(Clarke statistic = 199, \( p \)-value = 0.000).
5. Differences in the sample size between the current study and that of Skevas, Stefanou, and
Lansink (2012) are due to the use and availability of different environmental indicators
(i.e. pesticide impact on aquatic insects in Skevas, Stefanou, and Lansink (2012) versus
impact on groundwater here).

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Supplementary material
Supplemental data for this article can be accessed here.

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