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ANALYSIS

Risk-indexed herbicide taxes to reduce ground and surface water pollution: an integrated ecological economics evaluation

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Abstract

Public policy toward pesticide use in agriculture can benefit from data coming from models that integrate ecological and economic constraints into cropping decisions and pesticide use. Herein we use such a model to focus on the environmental and economic effectiveness of a specific set of tools used to promote sustainable agriculture with less pesticide runoff — incentive-based instruments created by *risk-indexed herbicide input-taxes*. We measure risk by health advisory levels and by an ecological economic simulation model that estimates predicted exposure levels. We explore whether this innovative solution of herbicide input-taxes does better at reducing losses to farm net returns, and surface and groundwater loadings than quantity restrictions. Using the integrated CEEPES model, our results suggest that risk-indexed input taxes by information about individual herbicide exposure levels can be a cost-effective tool to reduce predicted groundwater exposures. No single policy, however, was efficient at simultaneously improving groundwater and surface water quality. Instead we construct an efficient policy set. We find exposure-induced taxes were most efficient for small percentage reductions in overall exposure, bans were efficient for medium reductions, and flat taxes were efficient for high reductions.

Keywords: Pesticides; Input tax; Integration; CEEPES; Groundwater; Surface water

1. Introduction

A major research question in ecological economics is 'what regulatory or incentive-based instruments are most appropriate for assuring sustainability?' (Costanza et al., 1991, p. 15). This question is especially relevant when considering the goal of sustainable agriculture, in which high

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productivity levels are maintained by many actions including pest control. People control pests such as weeds, diseases, nematodes and insects because they are constraints that reduce net returns in agricultural production. They maintain financial returns by controlling pests through pesticides including herbicides, insecticides, and rodenticides.

Pesticides are estimated to be a good investment — $1 spent on a pesticide yields an average $3–6 savings in reduced crop damage (see Headley (1968) and Carrasco-Tauber (1990)). People like good investments, as revealed by the estimate that about 2.5 million tons of 55 000 pesticide products are applied annually worldwide (Pimentel et al., 1992), in which 80% are used in developed economies like Canada and the United States.[1] The United States Department of Agriculture (1991) estimates that people use pesticides on about 92% of the corn acres, 95% of the acres in the six largest cotton states, and 95% of the soybean acres. Use of the popular pesticide atrazine during the 1980s, for example, was estimated at nearly eighty million pounds of active ingredient, accounting for about 12% of total herbicide use (United States Environmental Protection Agency, 1990). For corn alone, atrazine use is estimated at nearly 60 million pounds of active ingredient over 64% of all treated acres (Center for Agricultural and Rural Development, 1993).

But pesticides are also perceived to pose a risk to human and environmental health including toxicity to non-target organisms such as pollinators and wildlife, environmental contamination of soil, water, and air affecting ecosystem functions such as nutrient cycles, selection of resistant pests, and acute and chronic toxicity to humans. Pesticides are best viewed as part of an overall pest control strategy aimed at providing abundant food at reasonable prices — an objective with which few would disagree given the broader goal of sustainable agriculture. But when pesticides threaten the sustainability of human and environmental health due to their persistence, mobility, and toxicity to non-target species, the public often asks policymakers to rethink how these inputs are used.[2] The public push for a more sustainable agriculture asks these decision makers to better understand the nature of pesticide risks to humans and natural resources, how people perceive and react to these risks, and how specific public policy tools can help or hinder private actions. Understanding which policy tools work best for risky choices under both economic and ecology constraints can provide additional information to help policymakers promote effective sustainable agricultural — more food and less pesticide risk for more people.

The World Health Organization (1990) estimated that over 3 million cases of acute pesticide poisoning occur annually worldwide, including 735 000 cases of long-term chronic impacts, and 37 000 cases of cancer. In the U.S., pesticides as a source of non-point pollution have also been accused of damaging an estimated 16% (206 179 miles) of the rivers in 40 surveyed states and 20% (5.4 million surface acres) of lakes. In addition, one or more of 46 pesticides have been detected in the groundwater of 26 states. Based on this evidence, the final report of the U.S. Congress on section 319 of the Clean Water Act states that '...information indicates very clearly that non-point source pollution has caused severe damage to aquatic communities nationwide and has destroyed the aesthetic values of many of our treasured recreational waters' (United States Environmental Protection Agency, 1992, 1–2).

Effective policies to promote sustainable agriculture through reduced pesticide pollution require information on how alternative policy tools affect the economic and environmental relationships involved in crop or livestock production. In general, three general management tools exist to address the human and environmental risks asso-

[1] In 1985, herbicide use in Canada increased to about 23 million hectares (51% of all cultivated land), up from about 8.6 million hectares in 1970, while 1985 insecticide use increased to 4.6 million hectares (10% of cultivated land) from about 900 000 hectares in 1970. In the United States, the use of pesticides has increased tenfold from 1945 to 1989, nearly tripling in crop production between 1964 and 1985 (MacIntyre 1987; Pimentel et al., 1991, 1992, and Pimentel et al., 1993).

[2] See, for example, Wargo (1997) and the papers in Russell and Shogren (1993).

ciated with pesticide use: technological restrictions such as improved pesticide application methods or biological pest control; cooperative institutions to share information on costs, damages, and technology; and economic incentives to change producer behavior and raise revenues.

But monitoring difficulties and random weather shocks complicate the task of deciding which instrument to select to such a degree that the typical emission-based policies promoted by economists are often impractical. Pesticides are a non-point source of pollution because there are many diffuse sources of pollution that are extremely costly to identify or monitor. Designing an incentive to alter a producer's pollution control strategy requires a significant amount of information on the marginal costs and benefits of control, including the environmental fate and transport systems and the value of life. Often a producer has private information on his own costs of pollution control or choice of control strategies, and if the regulator is uninformed, the producer can take advantage of this asymmetry to gain additional net returns. First-best incentive systems — those that would meet both economic and environmental objectives of efficiency and effectiveness — still require a significant amount of information on behavior that might not be feasible due to high costs or political unacceptability or both (see Hanley et al., 1997; Shortle and Abler, 1999).

In response, pragmatic policymakers and researchers have offered up a novel second-best policy tool to promote the goal of sustainable agriculture — *input taxes indexed by riskiness* as measured by health advisory levels or ecological economic simulation models that construct 'nonpoint production functions' to predict the fate and transport of pollutants. The open question is whether these indexed input charges can outperform more traditional policy options like command and control quantity restriction.[3] Herein we examine how on-farm economic and water quality indicators are impacted by a set of corn and sorghum herbicide input taxes indexed by either: (1) the U.S. Environmental Protection Agency's (EPA) human Health Advisory Level (HAL) benchmark; (2) the EPA's aquatic advisory benchmark; (3) predicted chronic exposure values for 1.2 m (meter) groundwater; and (4) predicted chronic exposure values by tillage–herbicide combinations for 1.2 meters (m) groundwater. Focusing on corn and sorghum in the Iowa region, we compare the indexed input taxes to the baseline policies of an atrazine ban, a triazine ban, and a flat input tax.[4]

We use the Comprehensive Environmental Economic Policy Modeling System (CEEPES) to construct these alternative input taxes that target herbicide characteristics and tillage practices. We generate trade-off frontiers to compare the effects of each policy tool on producer net returns and measures of groundwater and surface water quality.[5] Our results suggest that no policy tool is globally efficient for simultaneously improving both groundwater and surface water quality. While input taxes indexed by 1.2 m groundwater exposures is an effective tool to improve groundwater quality, an atrazine ban is equally effective for medium reductions in herbicide loadings. Since no one policy works for all goals, we define an efficient policy set that shows which policies achieve different levels of water quality improvement cost-effectively. Giving equal weights to improvements in groundwater and surface water quality, the exposure-based taxes are most efficient to produce small improvements in water quality; flat taxes are most efficient for larger improvements; and an atrazine ban is most efficient for intermediate improvements.

[3] See Griffin and Bromley (1982) and Shortle and Dunn (1986).

[4] Recent examples of other second-best nonpoint pollution solutions based on simulation models include Johnson et al. (1991), Mapp et al. (1994), and Lakshminarayan et al. (1996). Johnson et al. use a biophysical model to examine nitrate-reducing policies such as restriction on nitrogen applied, restrictions on total nitrate leachate, an input tax, and a Pigouvian tax on predicted leachate. Mapp et al. also use a biophysical model to explore both broad-based and targeted restrictions of nitrogen use; no tax policies were considered. Similarly, Lakshminarayan et al. compared the environmental and economic effects of herbicide bans and restrictions on herbicide use to improve water quality. Again no tax policies were considered.

[5] Bouzaher et al. (1992, 1993, 1995) and Bouzaher and Shogren (1997) supply the details on the CEEPES modeling system.

2. Policies

We address the ecological economic question of what incentive-based instruments are most appropriate for assuring agricultural sustainability by examining the economic and environmental tradeoffs given six different sets of policies: five sets of tax policies, and one set of bans. The bans include an *atrazine ban* and a *triazine ban* (e.g. banning atrazine, cyanazine, and simazine). The tax policies are: *Flat tax* — flat taxes on all corn and sorghum herbicides; *HAL tax* — taxes on each herbicide weighted according to the lifetime health advisory level for that herbicide; *Aquatic tax* — taxes on each herbicide weighted according to the aquatic benchmark for that herbicide; *Exposure tax* — taxes on each herbicide according to the predicted baseline 1.2 m groundwater chronic exposure value for that herbicide; and *Tillage-exposure tax* — taxes for each herbicide and tillage combination weighted according to the predicted baseline 1.2 m groundwater chronic exposure value by tillage for that herbicide.

As a natural first step, we first consider the atrazine and triazine bans as a baseline given several analyses have focused on herbicide restrictions.[6] The U.S. Environmental Protection Agency (EPA) can implement a ban simply by changing the labels of herbicides or by canceling the registrations of herbicides.

Following the insight of Griffin and Bromley (1982) and Shortle and Dunn (1986), we should be able to find input tax policies that more efficiently achieve water quality improvements. We first consider a flat tax policy on all herbicides. We chose this policy because it is straightforward to implement since it is targeted toward pounds active ingredient of each herbicide without requiring additional information about each herbicide. The idea of a flat herbicide tax is to make herbicides with low application rates more attractive than herbicides with high application rates. This should reduce the total pounds of herbicide applied and fulfill the aim of reducing quantities of herbicide loadings reaching water supplies and causing environmental damage.

To see if we can improve upon a flat tax using information about specific herbicides, we consider next the HAL tax and the Aquatic tax policies. These taxes are based on two out of the three environmental benchmarks listed in Table 1. The EPA offers these non-enforceable standards as guidance to interpret risks. The HAL tax is based on the lifetime health advisory level for each herbicide. The lifetime HAL is a measure for comparing the relative health risks of long-term consumption of herbicides in drinking water. Herbicides with a low lifetime HAL present a higher relative health risk. The idea of the HAL tax is to encourage substitutions from herbicides more likely to produce health risks if they reach water supplies to herbicides less likely to produce health risks. Similarly, the Aquatic tax is based on the aquatic benchmark. The aquatic benchmark is a measure for comparing the relative risks of damage to aquatic habitat. Herbicides with a low aquatic benchmark present a higher risk of aquatic habitat damage. The idea of the Aquatic tax is to encourage substitutions from herbicides more likely to damage non-target plants if they

Table 1
Environmental benchmarks for each herbicide (parts per billion)

Herbicide	Lifetime HAL[a]	10 day HAL	Aquatic benchmark
Atrazine	3	100	2
Nicosulfuron	44	44	0.03
Dicamba	9	300	1
Cyanazine	9	100	2
Bromoxynil	140	700	1
Bentazon	20	25	1
Metolachlor	100	100	1
EPTC	175	875	1
Alachlor	2	100	1
Simazine	35	50	500
Pendimethalin	300	1400	1
Propachlor	70	350	1
Glyphosate	700	20 000	60
Butylate	50	2400	1
2,4-D	70	1100	1

[a] HAL, Health Advisory Level.

[6] See, for example, Taylor and Frohberg (1977), Burton and Martin (1987), Osteen and Kuchler (1987), Taylor et al. (1991), NAPIAP (1992), and Lakshminarayan et al. (1996).

reach water supplies to herbicides which are less likely to damage non-target plants.[7]

Finally, to determine if information from simulated fate and transport models can improve the efficiency of input tax policies, we consider an Exposure tax and a Tillage-exposure tax. These policies are based on exposure values, which are unitless measures of predicted groundwater and surface water concentrations normalized using EPA environmental benchmarks. Exposure values capture both the predicted quantities of herbicides reaching water supplies and the relative risks the herbicides pose once they reach water supplies. The exposure value is calculated as:

$$\text{exposure value}_{ij} = \frac{\text{predicted concentration}_j}{\text{environmental benchmark}_{ij}} \quad (1)$$

where i = environmental benchmark and j = herbicide. The tillage-exposure value is calculated as:

$$\text{exposure value}_{ijk} = \frac{\text{predicted concentration}_{jk}}{\text{environmental benchmark}_{ij}} \quad (2)$$

where i = environmental benchmark, j = herbicide, and k = tillage practice. Exposure values larger than unity indicate that the predicted concentration exceeds the environmental benchmark, while exposure values less than unity indicate predicted concentrations less than the environmental benchmark. To compare the relative long-term health risks of herbicides, we divide predicted average groundwater concentrations by the EPA's lifetime health advisory level (HAL) for each herbicide. We denote the resulting exposure value as chronic exposure. To compare the relative short-term health risks, we divide predicted peak groundwater or surface water concentrations by the EPA's 10-day health advisory level for each herbicide. We denote the resulting exposure value as acute exposure. To compare the aquatic habitat risks, we divide predicted peak surface water concentrations by the EPA's aquatic benchmark. The resulting exposure value is denoted as acute aquatic exposure.

For this analysis we only consider tax policies based on the chronic exposure value. This focuses our efforts toward reducing herbicide health risks in groundwater, a priority for us since pesticide contamination of groundwater is a significant concern in Iowa. In a survey of private rural water wells in Iowa, Kross et al. (1992) detected pesticides in approximately 14% of the wells surveyed, with 1.2% of the wells having concentrations exceeding the lifetime HAL. The majority of the wells were thought to be contaminated by non-point sources related to normal agricultural practices.

We chose these policies first to show how tax policies compare to herbicide bans, and second to show what level of herbicide tax targeting provides the most benefit. Unlike previous studies which have focused on geographical targeting, our analysis targets characteristics of specific herbicides.[8] The tax policies we consider show incremental increases in the level of herbicide targeting, with the Flat tax being the least targeted tax. The HAL and Aquatic tax show an increase in herbicide targeting over the flat tax since they incorporate information about the relative risks individual herbicides pose once they reach water supplies. The exposure tax shows an increase in herbicide targeting over the HAL tax and Aquatic tax since it includes information about the likelihood of individual herbicides reaching water supplies, as well as presenting the risks these herbicides pose once they reach water supplies. Finally, the tillage-exposure tax shows an increase in herbicide targeting over the exposure tax since it includes information about the tillage practice used in addition to the information required for the exposure tax.

For each set of tax policies, tax rates are set for atrazine under conventional tillage at $1.00 5.00, 10.00, and 15.00 per pound active ingredient (a.i.). For the flat tax these rates are levied on all herbicides. In initial testing, a $15.00 per pound

[7] We do not compute a tax based on the 10-day HAL since herbicide concentrations in water supplies generally do not reach these levels. However, we do use this benchmark to examine the effects of alternative policies on short-term health risks.

[8] See, for example, Braden et al. (1989) and Mapp et al. (1994).

active ingredient tax for atrazine reduced total pounds of atrazine applied by about 90%, indicating that this tax rate was comparable to an atrazine ban. Also this rate produced changes in predicted groundwater exposure levels comparable to a triazine ban, so this tax rate was chosen as an upper bound. The $1.00, 5.00, and 10.00 levels were chosen as intermediate points between the baseline and the $15.00 per pound tax. Tax rates for all other herbicides are set relative to the atrazine tax rate to ensure that the tax levels under each policy are comparable.

For the HAL and aquatic taxes, the tax rates for each herbicide are proportional to the atrazine tax rate and inversely related to the benchmark for that herbicide:

$$\text{tax rate}_{ij} = \frac{(\text{atrazine benchmark}_j) \times (\text{atrazine tax rate}_j)}{\text{benchmark}_{ij}} \tag{3}$$

where i = environmental benchmark, j = herbicide. For example, if the HAL tax rate for atrazine is $5.00 per pound active ingredient, the corresponding HAL tax rate for alachlor would be (3 × $5.00)/2 = $7.50 per pound active ingredient. Similarly, if the aquatic tax rate for atrazine is $5.00 per pound active ingredient, the corresponding aquatic tax rate for alachlor would be (2 × $5.00)/1 = $10.00 per pound active ingredient. So, herbicides with a benchmark less than the atrazine benchmark (i.e. herbicides that pose a higher risk than atrazine) have a higher tax rate.

For the chronic exposure tax, the tax rates for each herbicide are proportional to the atrazine rate and the predicted 1.2 m groundwater chronic exposure for that herbicide:

$$\text{exposure tax rate}_{ij} = \frac{(\text{predicted exposure}_{ij}) \times (\text{atrazine exposure tax rate}_i)}{\text{atrazine groundwater predicted exposure}_i} \tag{4}$$

where i = environmental benchmark (in this case, lifetime HAL), j = herbicide, and the predicted exposure values are calculated as in Eq. (1). So, herbicides with a predicted exposure greater than the predicted atrazine exposure will have a higher tax rate.

Similarly, the chronic tillage-exposure tax rates for each herbicide are proportional to the conventional tillage atrazine rate and the predicted 1.2 m groundwater chronic exposure for that herbicide and tillage combination:

$$\text{exposure tax rate}_{ijk} = \frac{(\text{predicted exposure}_{ijk}) \times (\text{atrazine cv.till. exposure tax rate}_i)}{\text{atrazine groundwater predicted exposure}_{ik}} \tag{5}$$

where i = environmental benchmark (in this case, lifetime HAL), j = herbicide, and k = tillage practice. So, herbicide and tillage combinations with a predicted exposure greater than the atrazine conventional tillage exposure will have a higher tax rate.

Table 2 shows the $1.00 per pound atrazine tax rate schedules for the HAL weighted tax policy, the aquatic benchmark weighted tax policy, and the two exposure weighted tax policies. Separate exposure weighted tax rates are calculated for atrazine applied at a rate greater than 1.5 pounds active ingredient and atrazine applied at a rate less than 1.5 pounds active ingredient based on baseline use. The aquatic benchmark tax does not include a $15.00 per pound a.i. tax since the $10.00 per pound tax resulted in a larger decrease in on-farm net returns than any other policy.

3. Modeling system

We use the CEEPES model to evaluate the effects of alternative policies. CEEPES is an integrated ecological economic system of models designed to evaluate the risks and benefits of alternative policy scenarios (Bouzaher et al., 1995). Under the configuration of CEEPES used for analysis of herbicides policies aimed at reducing non-point source pollution, WISH (Weather Impact Simulator for Herbicides) models the effectiveness of alternative weed control strategies. Fig. 1 shows the information flow in CEEPES.

WISH simulates the cost and effectiveness of 488 weed control strategies for corn and 148 weed control strategies for sorghum using 50 years of daily weather information. Each weed control strategy is composed of a primary strategy which

Table 2
Tax rates per pound active ingredient based on a $1.00 per pound atrazine conventional till atrazine tax

Chemical	Environmental benchmark taxes		Exposure taxes			
	LifetimeHAL tax $/lb a.i.	Aquatic tax $/lb a.i.	Exposure tax $/lb a.i.	Conv. Till Exposure tax $/lb a.i.	Red. Till Exposure tax $/lb a.i.	No Till Exposure tax $/lb a.i.
Atrazine > 1.5 lb a.i.	1.00	1.00	1.000	1.000	2.483	2.210
Atrazine < 1.5 lb a.i.	1.00	1.00	0.936	1.385	2.556	5.584
Nicosulfuron	0.07	66.67	0.000	0.000	0.000	0.000
Dicamba	0.33	2.00	0.005	0.007	0.012	0.015
Cyanazine	0.33	1.00	0.000	0.000	0.000	0.000
Bromoxynil	0.02	2.00	0.000	0.000	0.000	0.000
Bentazon	0.15	2.00	0.120	0.266	0.481	0.605
Metolachlor	0.03	2.00	0.000	0.001	0.000	0.000
EPTC	0.02	2.00	0.000	0.000	0.000	0.000
Alachlor	1.50	2.00	0.000	0.000	0.001	0.001
Simazine	0.09	0.00	0.030	0.066	0.122	0.109
Pendimethalin	0.01	2.00	0.013	13.147	0.000	0.000
Propachlor	0.04	2.00	0.000	0.000	0.000	0.000
Glyphosate	0.00	0.03	0.000	0.000	0.000	0.000
Butylate	0.06	2.00	0.000	0.001	0.000	0.000
2,4-D	0.04	2.00	0.000	0.000	0.000	0.000

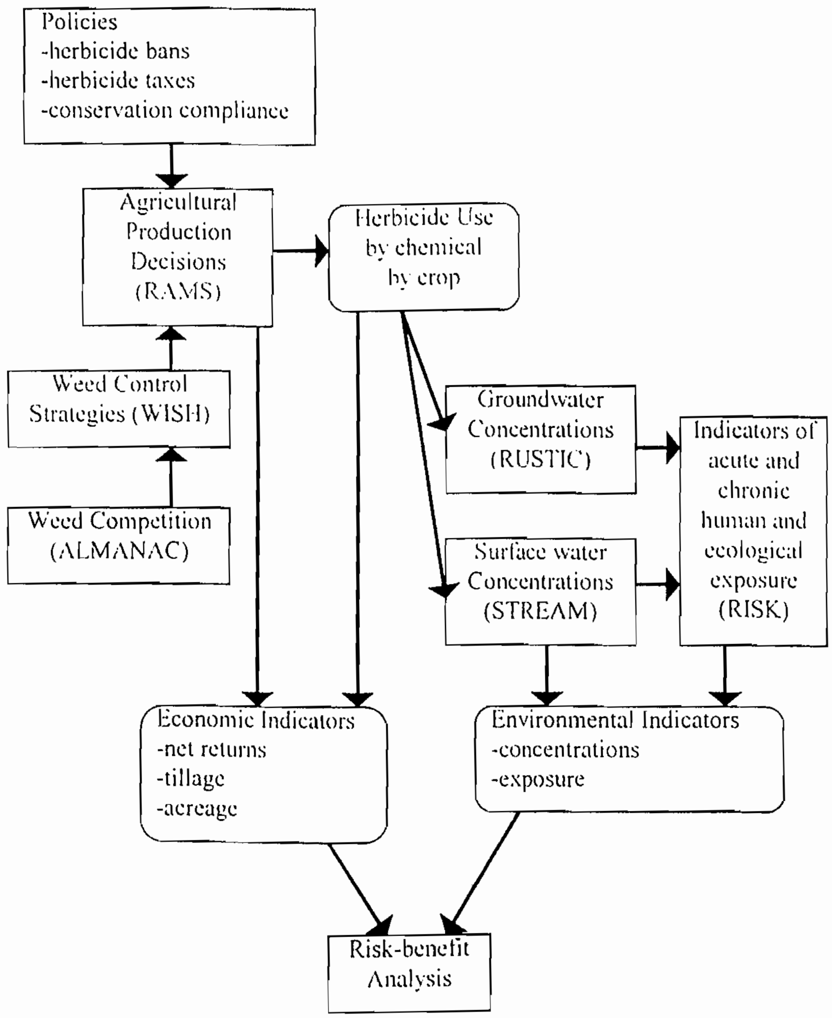

Fig. 1. Information flow in CEEPES.

is used if weather conditions permit; and a secondary strategy which is used if the primary strategy cannot be used or fails. Each strategy is differentiated according to different timings and lengths of application and effectiveness 'windows of opportunity', different associated tillage practices, and different soil types (sand or clay). See Bouzaher et al. (1992) for a detailed description of the WISH model, and see Archer and Shogren (1996) for an application to the theory of self-protection. The use of the WISH model allows us to consider a rich set of herbicide substitution possibilities using a combination of expert opinion and physical simulation. This is a significant improvement over previous policy analyses which have used only limited substitution or relied entirely on surveys or expert opinion.[9]

The effectiveness of each weed control strategy from WISH is converted to a crop yield using ALMANAC (Agricultural Land Management Alternatives with Numerical Assessment Criteria).

[9] See, for example, Taylor and Frohberg (1977), Kania and Johnson (1981), Burton and Martin (1987), Cox and Easter (1990), Knutson et al. (1990), Smith et al. (1990), and Swinton (1991).

ALMANAC simulates the effect of weed competition and interactions of management alternatives on weed growth (Jones and O'Toole, 1986; Kiniry et al., 1991).

The costs and yields for each weed control strategy are then used as input coefficients for the weed control activities in RAMS (Resource Adjustment Modeling System). RAMS is a linear-programming model that chooses the profit-maximizing mix of crop production activities for a representative producer at the producing area (PA) level. One hundred and five producing areas exist in the U.S. This analysis focuses on policies in a single PA (PA41), covering most of Iowa and a few counties from neighboring states.

The output of RAMS is linked to the fate and transport models RUSTIC (Risk of the Unsaturated/Saturated Transport and Transportation of Chemical Concentrations) and STREAM (Surface Transport and Agricultural Runoff of Pesticides for Exposure Assessment) (Dean et al., 1989; Donigian et al., 1986). These models respectively estimate groundwater and surface water concentrations of herbicides. For groundwater, peak and average concentrations are estimated at depths of 1.2 and 15 m. For surface water, only a peak concentration is estimated.

The CEEPES model links WISH and ALMANAC, and RAMS, RUSTIC and STREAM by using metamodel response functions (Bouzaher et al., 1993). Metamodels are regression models explaining the input–output relationships of simulation models. Metamodeling allows us to evaluate several alternatives without running each of the simulation models for each alternative. Without this feature the time required to run each of the simulation models would limit the number of alternatives we could compare. This feature allows us to consider different policies and multiple levels of tax rates.

We measure the effects of alternative policy scenarios by losses in producer net returns, and changes in ground and surface water exposure values. We examine the effects of alternative policies on total pounds of herbicide applied and average application rates.

4. Analysis

We present the results in four steps: (1) the economic indicators of changes in net returns and crop yields for each policy; (2) the tradeoff frontiers between economic and environmental indicators of total herbicide use, total atrazine use, application rates, and exposure levels; (3) the tradeoffs between surface and groundwater exposures for each policy; and (4) a tradeoff frontier and 'best' policy set based on a uniform weighting of each environmental indicator.

4.1. Economic indicators

Table 3 shows the change in producer net returns for each scenario compared to the baseline. Net returns for the region are calculated as the total revenue from all crops produced in the region minus the total crop production costs. Per acre net returns are calculated as the total net returns for the region divided by the total acres used for crop production in the region. The \$10.00 aquatic benchmark tax produced the largest reduction in net returns: \$190 million from the baseline or \$8.03 per crop acre. This represents a 4.22% decrease in net returns from the baseline. Table 4 shows changes in corn and sorghum yields for each policy. Corn yield changes were generally small, with the maximum yield change of 2.7% occurring under the \$10.00 aquatic benchmark tax. Yield changes were more significant for sorghum with a 29.3% decrease for a triazine ban. These yield decreases show how the alternative policies affect weed control decisions. Policies that result in larger yield changes force changes toward weed control practices which are less expensive but are more likely to result in weed control failures, while policies that result in smaller yield changes produce substitutions to weed control practices that are more expensive but maintain levels of weed control.

This model will tend to exaggerate the effects of alternative policies on production decisions, since crop prices and herbicide prices are held constant in the RAMS model. This is a reasonable assumption for regional policies, although we would need to include market effects for national policies.

Also, an upper bound is placed on soybean acreage at 125% of the 1992 actual level to calibrate the model to current conditions. To avoid biasing the outcomes, this upper bound was left in place for all policies. The constraint was binding for all policies. As a result there could be no substitution from corn to soybeans as taxes increased. There were no constraints on substitutions to other crops, however, crop acreage did not change from the baseline for any of the policies modeled.

4.2. Tradeoffs between economic and environmental indicators

Examining the results for several different tax levels allows us to construct a policy frontier showing the tradeoffs achievable with each policy tool. These frontiers can then be combined to form an overall efficient frontier of available policies, showing the tradeoffs achievable with this set of tools. These frontiers are similar to Xu et al. (1995) who constructed non-inferior solution frontiers among net returns, soil erosion, and nitrate leaching, with one key difference. Their analysis showed the frontier set of solutions that could be achieved given current production technology. They did not indicate the policies that could be used to reach this frontier. Our approach generates a frontier of solutions that could be achieved given the current production technology and given *a set of policy instruments*. Furthermore, we can indicate which policy instruments are used to reach points on the frontier. For example, Fig. 2 shows all of the solution points for each tax level and each policy tool plotted in terms of decreased net returns and decreased total pounds of herbicide applied. The solution points

Table 3
Change in net returns for each policy

Scenario	Decrease from baseline (million $)	Per acre decrease from baseline	% Change from baseline
Atrazine Ban	9.7	$0.41	0.21
Triazine Ban	73.8	3.11	1.64
$1.00/lb Flat tax	15.1	0.64	0.34
$5.00/lb Flat tax	75.2	3.17	1.67
$10.00/lb Flat tax	135.2	5.71	3.00
$15.00/lb Flat tax	180.0	7.60	4.00
$1.00/lb HAL tax	7.7	0.33	0.17
$5.00/lb HAL tax	33.7	1.42	0.75
$10.00/lb HAL tax	57.6	2.43	1.28
$15.00/lb HAL tax	81.6	3.45	1.81
$1.00/lb Aqua. Bench tax	25.4	1.07	0.56
$5.00/lb Aqua. Bench tax	107.3	4.53	2.38
$10.00/lb Aqua. Bench tax	190.2	8.03	4.22
$1.00/lb Exp. Tax	3.6	0.15	0.08
$5.00/lb Exp. Tax	9.1	0.38	0.20
$10.00/lb Exp. tax	9.7	0.41	0.22
$15.00/lb Exp. tax	9.8	0.41	0.22
$1.00/lb Till-Exp. tax	7.2	0.31	0.16
$5.00/lb Till-Exp. tax	9.9	0.42	0.22
$10.00/lb Till-Exp. tax	10.2	0.43	0.23
$15.00/lb Till-Exp. tax	10.4	0.44	0.23

Table 4
Change in crop yields for each policy

Scenario	% Change in corn grain yield	% Change in corn silage yield	% Change in sorghum grain yield
Atrazine Ban	−0.1	0.0	−9.1
Triazine Ban	−1.4	−0.9	−29.3
$1.00/lb Flat tax	0.0	0.0	0.0
$5.00/lb Flat tax	−0.2	−0.2	−10.6
$10.00/lb Flat tax	−1.7	−0.4	−10.6
$15.00/lb Flat tax	−2.0	−0.4	−10.6
$1.00/lb HAL tax	0.0	0.0	0.0
$5.00/lb HAL tax	−0.1	0.0	−10.6
$10.00/lb HAL tax	−0.1	0.0	−10.6
$15.00/lb HAL tax	−0.1	0.0	−10.6
$1.00/lb Aqua. Bench tax	−0.1	−0.2	0.0
$5.00/lb Aqua. Bench tax	0.7	−0.1	−2.3
$10.00/lb Aqua. Bench tax	−2.7	0.3	−2.3
$1.00/lb Exp. Tax	0.0	0.0	0.0
$5.00/lb Exp. Tax	−0.1	0.0	−13.7
$10.00/lb Exp. Tax	0.0	0.0	−13.7
$15.00/lb Exp. Tax	0.0	0.0	−13.7
$1.00/lb Till-Exp. Tax	0.0	0.0	−0.3
$5.00/lb Till-Exp. Tax	−0.1	0.0	−0.3
$10.00/lb Till-Exp. Tax	−0.1	0.0	−8.6
$15.00/lb Till-Exp. Tax	−0.1	0.0	−8.6

for each policy instrument are joined to identify the set of solutions that are obtained with that particular policy instrument. Points below and to the right of this set are more efficient since they have greater reductions in herbicides applied with a smaller reduction in net returns. Conversely, points above and to the left of this set are less efficient since they would have smaller reductions in herbicides applied with a larger reduction in net returns.

Table 5 lists changes in total pounds of herbicides applied and average application rates for all herbicides together and for atrazine only. Negative values imply that pounds applied or application rates actually increase. The values in Table 5 are plotted in Fig. 2 and Fig. 3 to show the tradeoff frontiers between a decrease in net returns and a decrease in total pounds of herbicide applied for each set of policies. These figures show the effects of alternative policies in reducing the amount of herbicides that are applied to the region as a whole. Looking at Fig. 2, we see that only flat taxes and bans produce significant reductions in total pounds of herbicide applied. This is expected for a flat tax since it makes all herbicide use more expensive. For the bans, this means that eliminating atrazine or all triazines leads either to use of herbicides with lower application rates, or reductions in the number of acres treated. The exposure tax and the tillage-exposure tax policies, denoted by 'Exposure' and 'TExposure' in the legend, produce little change in the amount of herbicide applied, implying any changes in application rates are offset by changes in the number of acres treated. The HAL tax and the aquatic benchmark tax policies lead to increases in total herbicide use, implying producers either substitute toward herbicides which require higher application rates to be effective, or to herbicides which require more rescue applications.

Fig. 3 shows that atrazine use decreases in all of the sets of policies. Even though atrazine is not always the most highly taxed herbicide per pound of application, it does become relatively more expensive than some herbicides. Fig. 3 shows that more substitutions are made away from atrazine toward relatively less expensive herbicides than from relatively more expensive herbicides toward atrazine. The two exposure tax policies and the set of bans can achieve reductions in atrazine use at the lowest losses to net returns. This is expected since these policies focus more narrowly on reducing the use of atrazine.

Fig. 4 and Fig. 5 show the tradeoff frontiers between a decrease in net returns and a decrease in average herbicide application rates. These figures show the effectiveness of alternative policy tools in reducing the intensity of herbicide use. Fig. 4 shows that only a flat tax is effective in reducing average application rates of all corn and sorghum herbicides. All other policies lead to substitutions toward herbicides with higher application rates. Although this is somewhat discouraging, it is not entirely surprising. Other than the flat tax, the tax policies we consider change the relative prices of herbicides per pound active ingredient. These changes in relative prices may make herbicides that tend to be applied at low application rates relatively more expensive than herbicides that tend to be applied at high application rates. This would increase the overall average application rate for the region.

Fig. 5 shows that only the bans and the exposure tax policies are effective at reducing atrazine application rates. Furthermore, the exposure tax policies are only effective at reducing atrazine application rates if the tax is set high enough. But Fig. 3 showed that all of the policies reduced total pounds of atrazine applied. This indicates that most policies result in a decrease in the number of acres treated by atrazine, and that the acres no longer treated by atrazine were receiving low average application rates in the baseline. This indicates that most policies result in a reduced use of atrazine as a rescue treatment, but maintain the use of atrazine as a primary weed control strat-

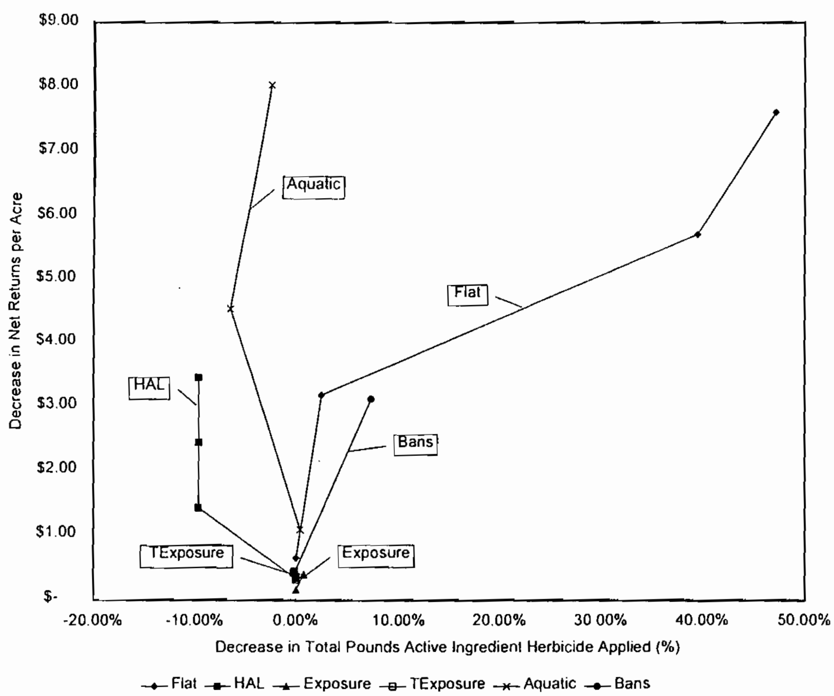

Fig. 2. Tradeoffs between decreased net returns and decreased total herbicide applied.

Table 5
Decrease in pounds applied and average application rates for each policy

Scenario	All Herbicides		Atrazine	
	Decrease in total pounds applied (%)	Decrease in average application rate (pounds per acre) (%)	Decrease in total pounds applied (%)	Decrease in average application rate (pounds per acre) (%)
Atrazine Ban	−0.15	−11.78	100.00	100.00
Triazine Ban	7.19	−9.88	100.00	100.00
\$1.00/lb Flat tax	0.00	0.00	0.00	0.00
\$5.00/lb Flat tax	2.36	−3.32	1.16	−2.57
\$10.00/lb Flat tax	39.51	11.85	71.90	−48.00
\$15.00/lb Flat tax	47.14	20.47	89.03	−130.72
\$1.00/lb HAL tax	0.00	0.00	0.00	0.00
\$5.00/lb HAL tax	−9.70	−27.24	89.97	−130.12
\$10.00/lb HAL tax	−9.70	−27.24	89.97	−130.12
\$15.00/lb HAL tax	−9.70	−27.24	89.97	−130.12
\$1.00/lb Aqua. Bench tax	0.37	−3.96	−0.32	−0.01
\$5.00/lb Aqua. Bench tax	−6.42	−24.88	29.94	2.34
\$10.00/lb Aqua. Bench tax	−2.52	−26.88	51.22	−0.48
\$1.00/lb Exp. Tax	0.00	0.00	0.00	0.00
\$5.00/lb Exp. Tax	0.74	−13.61	92.17	−120.92
\$10.00/lb Exp. Tax	−0.16	−11.68	99.90	45.57
\$15.00/lb Exp. Tax	−0.16	−11.68	99.90	45.57
\$1.00/lb Till-Exp. Tax	0.00	−9.82	53.79	−23.06
\$5.00/lb Till-Exp. Tax	−0.24	−11.85	99.62	−106.34
\$10.00/lb Till-Exp. Tax	−0.15	−11.74	99.99	96.96
\$15.00/lb Till-Exp. Tax	−0.15	−11.74	99.99	96.96

egy. When atrazine becomes relatively more heavily taxed, as in the higher exposure tax policies, we begin to see a reduction in the use of atrazine as a primary weed control strategy reducing both the acres treated by atrazine and atrazine average application rates.

Policies which decrease total pounds of herbicide applied and/or herbicide application rates might be expected to increase water quality. However, these measures do not take into consideration differences in the risks individual herbicides pose once they reach water supplies. Also they do not take into consideration how likely each herbicide is to reach water supplies. We use the EPA benchmarks to account for the differences in risks individual herbicides pose once they reach water supplies. The metamodels for RUSTIC and STREAM are used to predict the transport of herbicides to water supplies. Table 6 lists changes in weighted groundwater and surface water exposure levels for each policy. These values represent a weighted sum of herbicide chronic groundwater exposure values across herbicide, tillage and crop, weighted according to the number of acres treated by each herbicide for each crop and tillage practice:

weighted groundwater exposure value

$$= \sum_{ijk} \frac{(\text{exposure value})_{ijk}(\text{acres})_{ijk}}{\text{total corn and sorghum acres}} \quad (6)$$

where i = crop, j = tillage practice, and k = herbicide. By calculating a weighted sum for these exposures, we are looking at region-wide water quality measures.

Fig. 6 shows the tradeoff frontiers between a decrease in net returns and a decrease in weighted 1.2 m chronic groundwater exposures (column A in Table 6). This figure shows that all of the policies are effective at decreasing 1.2 m chronic exposure values. The two sets of exposure tax policies and the set of bans achieve reductions at the lowest on-farm cost. It is expected that the two exposure policies achieve the reductions efficiently since these policies are targeted specifically at reducing 1.2 m chronic exposure. It is surprising that banning atrazine is also an efficient means of reducing 1.2 m chronic exposure from all herbicides since this policy targets atrazine only. This indicates the significance of atrazine contribution to the overall exposure values. The atrazine ban is equally as effective as high exposure weighted taxes at making large reductions in 1.2 m chronic exposure levels. If the goal is a

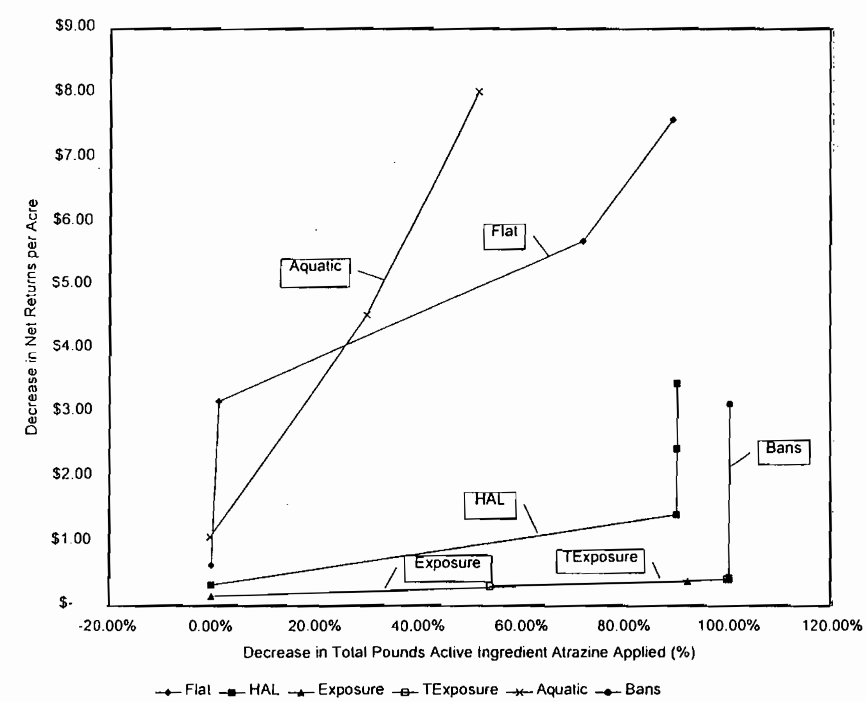

Fig. 3. Tradeoffs between decreased net returns and decreased total atrazine applied.

Table 6
Decreases in groundwater and surface water weighted sum exposure values[a]

Scenario	(A) Decrease in 1.2 m chronic exposure	(B) Decrease in 1.2 m acute exposure	(C) Decrease in 15 m chronic exposure	(D) Decrease in 15 m acute exposure	(E) Decrease in stream acute exposure	(F) Decrease in stream acute aquatic exposure
Atrazine Ban	0.0835	0.0268	4.83E-04	1.48E-04	0.1619	1.7880
Triazine Ban	0.0835	0.0242	4.86E-04	1.66E-04	1.2583	−36.0989
$1.00/lb Flat tax	0.0016	0.0035	1.98E-05	1.89E-05	−0.1331	−12.0135
$5.00/lb Flat tax	0.0028	0.0039	7.11E-05	2.86E-05	−0.1257	−6.1468
$10.00/lb Flat tax	0.0648	0.0220	3.74E-04	1.34E-04	1.3083	58.4604
$15.00/lb Flat tax	0.0740	0.0253	4.19E-04	1.50E-04	1.4054	55.8361
$1.00/lb HAL tax	0.0016	0.0035	1.98E-05	1.89E-05	−0.1331	−12.0135
$5.00/lb HAL tax	0.0721	0.0107	3.99E-04	−4.00E-06	−0.2351	24.0890
$10.00/lb HAL tax	0.0721	0.0107	3.99E-04	−4.00E-06	−0.2351	24.0890
$15.00/lb HAL tax	0.0721	0.0107	3.99E-04	−4.00E-06	−0.2351	24.0890
$1.00/lb Aqua. Bench tax	−0.0006	−0.0008	−3.10E-06	−1.52E-06	−0.0128	−0.7714
$5.00/lb Aqua. Bench tax	0.0145	−0.0238	1.08E-04	−1.28E-04	−0.7885	6.9543
$10.00/lb Aqua. Bench tax	0.0331	−0.0235	2.04E-04	−1.22E-04	−0.6401	19.1591
$1.00/lb Exp. Tax	0.0016	0.0035	1.98E-05	1.89E-05	−0.1331	−12.0135
$5.00/lb Exp. Tax	0.0759	0.0265	4.32E-04	1.54E-04	0.2487	12.4273
$10.00/lb Exp. Tax	0.0825	0.0267	4.76E-04	1.47E-04	0.1615	1.8597
$15.00/lb Exp. Tax	0.0825	0.0267	4.76E-04	1.47E-04	0.1615	1.8597
$1.00/lb Till-Exp. Tax	0.0554	0.0192	2.86E-04	1.12E-04	0.1682	3.0461
$5.00/lb Till-Exp. Tax	0.0831	0.0268	4.39E-04	1.45E-04	0.1595	1.7164
$10.00/lb Till-Exp. Tax	0.0835	0.0268	4.82E-04	1.48E-04	0.1618	1.7849
$15.00/lb Till-Exp. Tax	0.0835	0.0268	4.82E-04	1.48E-04	0.1618	1.7849

[a] Note: Exposure values are unitless values calculated as concentrations (ppb) divided by EPA environmental benchmarks (ppb).

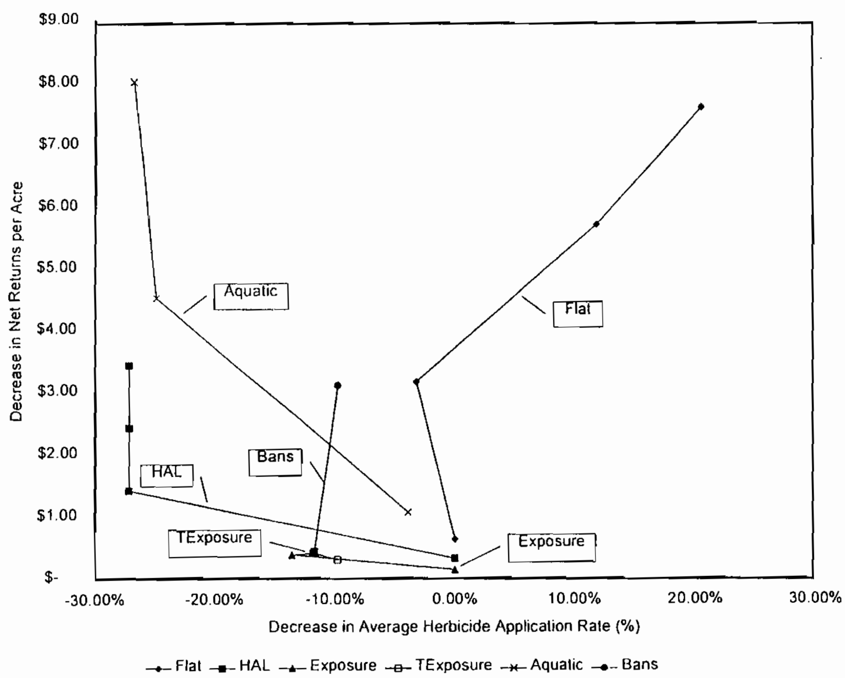

Fig. 4. Tradeoffs between decreased net returns and decreased average application rates.

smaller reduction at a smaller cost, the exposure taxes are most efficient, achieving 0% to nearly 100% reductions in 1.2 m chronic exposure levels at the lowest costs. There appears to be no significant advantage to targeting herbicide applications by tillage practice than targeting individual herbicides without regard to tillage, even though the RUSTIC and STREAM metamodels indicate that herbicide fate and transport differs significantly for different tillage practices (Bouzaher et al., 1993). This suggests that even though different tillage practices can affect the fate and transport of herbicides, the characteristics of the individual herbicides themselves are more important in determining groundwater and surface water exposure values.

4.3. Tradeoffs among environmental indicators

Up to this point we have considered tradeoffs between net returns and one single physical effect. We now consider the tradeoffs between pairs of physical effects. This helps us to see that individual policies have several simultaneous physical effects as well as an economic effect, and that choosing among policies also involves understanding tradeoffs among physical effects. Fig. 7 shows the tradeoffs between a decrease in the weighted sum acute stream aquatic exposure (column F in Table 6) and a decrease in the weighted sum 1.2 m chronic exposure. This shows tradeoffs between drinking water risks and aquatic habitat risks. In this figure, policies in quadrant I are improvements in both 1.2 m groundwater quality and aquatic surface water quality (win–win policies). Policies in quadrants II and IV show mixed effects with an improvement in one water quality indicator and a decline in the other water quality indicator (win–lose policies). Policies in quadrant III show a decline in both water quality indicators (lose–lose policies). All policies except the tillage specific exposure taxes at some point increase acute aquatic exposure values, however, all of the tax policies do reduce acute aquatic exposures as tax rates increase. Targeting individual herbicides becomes less important as tax rates increase. An atrazine ban decreases acute aquatic exposure slightly, but going to a triazine ban increases acute aquatic exposure over 20% above the baseline level. The $10.00 per pound and $15.00 flat taxes produce the largest decrease in acute aquatic exposures and simultaneously decrease 1.2 m chronic exposure by 70–90% from the baseline. The aquatic tax policies show a trend of decreasing both acute aquatic exposure and 1.2 m chronic exposure, but greater reductions are seen from the flat tax with a smaller decrease in net returns. The flat tax achieves the reductions more efficiently even though the aquatic tax is targeted at reducing aquatic habitat risk.

Fig. 8 shows the tradeoffs between a decrease in stream acute exposure (column E in Table 6) and a decrease in 1.2 m chronic exposure. This shows tradeoffs among groundwater and surface water drinking water exposures. Similar to the acute aquatic exposure, all of the tax policies except the tillage-exposure taxes initially increase stream acute exposure. The ban policies produce decreases in both the stream acute exposure and the 1.2 m chronic exposure. Also similar to the acute aquatic exposure, the $10.00 and 15.00 per pound flat taxes produce the greatest decrease in stream acute exposure. However, the triazine ban results in nearly as large a decrease in stream drinking water exposure while reducing 1.2 m chronic exposure by nearly 100% and producing a smaller decrease in net returns. All of the aquatic taxes and HAL weighted taxes increase stream acute exposures. Recall that these policies target individual herbicides based on the risks they present when they reach surface waters. Since these policies do not account for the likelihood of individual herbicides reaching surface waters, they are not effective at reducing stream acute exposures.

4.4. Choosing 'best available' policies

We construct a single efficient tradeoff frontier between losses to net returns and water quality to choose a set of 'best available' policies from the taxes and bans. Therefore, we combine the exposure values into one measure using the relative importance of each individual exposure value.

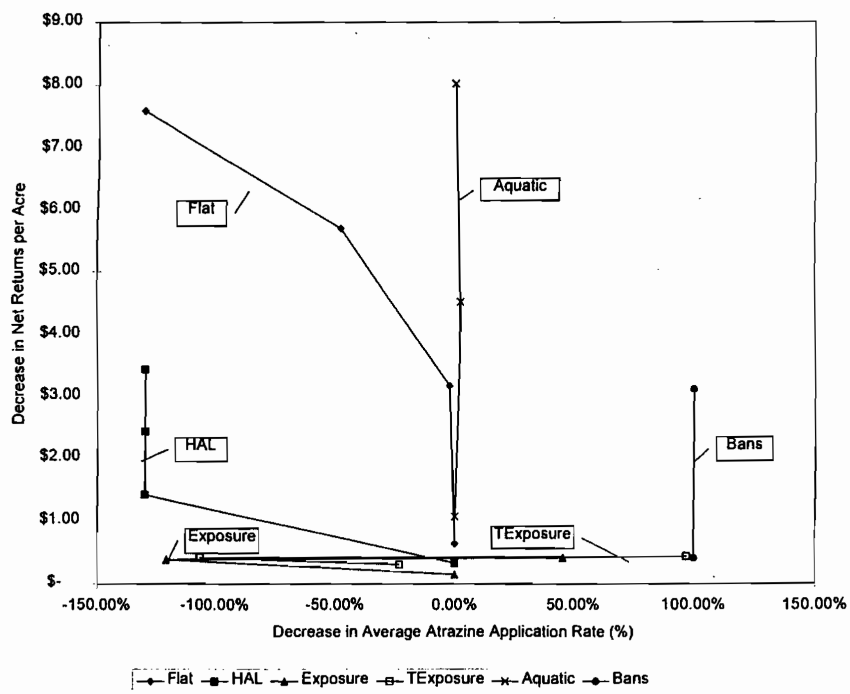

Fig. 5. Tradeoffs between decreased net returns and decreased average atrazine application rates.

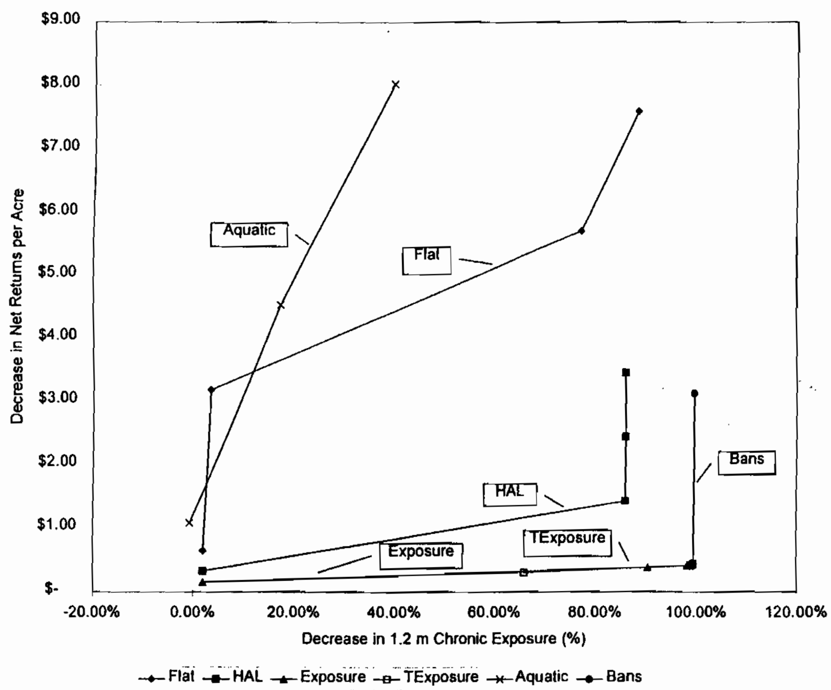

Fig. 6. Tradeoffs between decreased net returns and decreased weighted sum 1.2 m chronic groundwater exposure.

Ideally, the relative weights should take into consideration social preferences and the relative damages implied by each exposure measure. To our knowledge no empirical evidence exists for determining the relative weights to place on groundwater, surface water, and aquatic habitat water quality measures.

Lacking an empirical basis for setting relative water quality weights, we illustrate how an efficient policy tool would be chosen giving equal weighting to proportional reductions in 1.2 m chronic exposure, acute stream exposure, and acute aquatic exposure. An overall exposure index is calculated by dividing the 1.2 m chronic exposure value, the acute stream exposure value, and the acute aquatic exposure value for each policy by the corresponding baseline exposures. The single overall exposure index is the average of these three values. Using this index, a 10% reduction in 1.2 m chronic exposure is equally as important as a 10% reduction in acute aquatic exposure.

Table 7 shows the overall exposure index for each policy. Index values greater than one indicate a decrease in overall water quality compared to the baseline, while index values less than one indicate an improvement in overall water quality compared to the baseline. For all of the tax policies except the tillage-exposure tax, the overall exposure index increases at the $1.00 per pound a.i. tax level. As the tax level increases, the overall exposure index eventually decreases for each of the tax policies. The $15.00 per pound a.i. flat tax leads to the largest decrease in the overall exposure index.

Fig. 9 shows the tradeoffs between producer net returns and the exposure index. The dashed line shows an overall efficient tradeoff frontier. The overall frontier includes the $1.00 and 5.00 per pound a.i. exposure tax policies, the $1.00 per pound a.i. tillage-exposure tax, the triazine ban and the $10.00 and 15.00 per pound a.i. flat taxes. The rest of the exposure and tillage-exposure taxes and the atrazine ban are just off of the efficient frontier, having decreases in net returns within $0.06 per acre and decreases in overall exposure index within 1.25% of the $5.00 per pound a.i. exposure tax policy.

In general, we cannot choose a single overall efficient policy. Instead we have an efficient policy set. Different policies achieve different levels of water quality improvement efficiently. This supports the Malik et al., (1992) point that no single policy is likely to be effective in reducing all non-point source pollution. The exposure and tillage-exposure tax policies are most efficient at achieving 0–37% reductions in the exposure index. The bans are most efficient at achieving 35–46% reduction in the exposure index, and flat taxes are most efficient at achieving up to 62% reductions in the exposure index. It appears that modeling exposure is useful for developing tax policies if the goal is to fine tune water quality improvements. However, if an improvement greater than a 37% reduction in the exposure index is desired, the added information is not necessary and bans or flat taxes should be used. Using this information, a policy maker can determine the losses to net returns to achieve different levels of water quality improvement, and which policy tools can be used to achieve these improvements.

Finally, one benefit of tax policies is that the tax revenue raised can be used to offset some of the costs of administering the policies. Table 8 shows the tax revenues raised by each policy. The tax revenues are also expressed as a percentage of the decrease in net returns that occurs as a result of the policy. For each policy, this shows the portion of the decrease in net returns that can be directly attributed to the tax. The remainder of the cost is incurred indirectly through changes in production practices. Note the decreases in net returns for the exposure weighted tax policies result largely from changes in production practices. As these targeted taxes increase, tax revenues decrease dramatically as a percentage of the change in net returns. These taxes cause producers to make substitutions toward herbicides that are not taxed. Producers would rather absorb the yield losses than pay the tax since the marginal cost of the yield loss is less than the marginal cost of the tax. Alternatively, the flat taxes, HAL taxes, and aquatic taxes apparently do not induce such substitutions. These taxes are too broad-

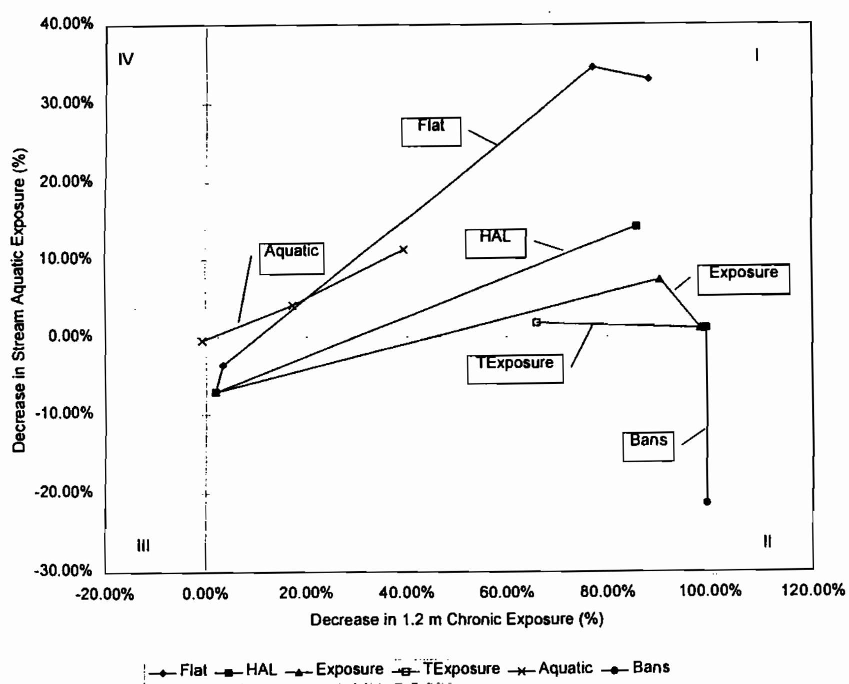

Fig. 7. Tradeoffs between decreased acute stream aquatic exposure and 1.2 m chronic exposure.

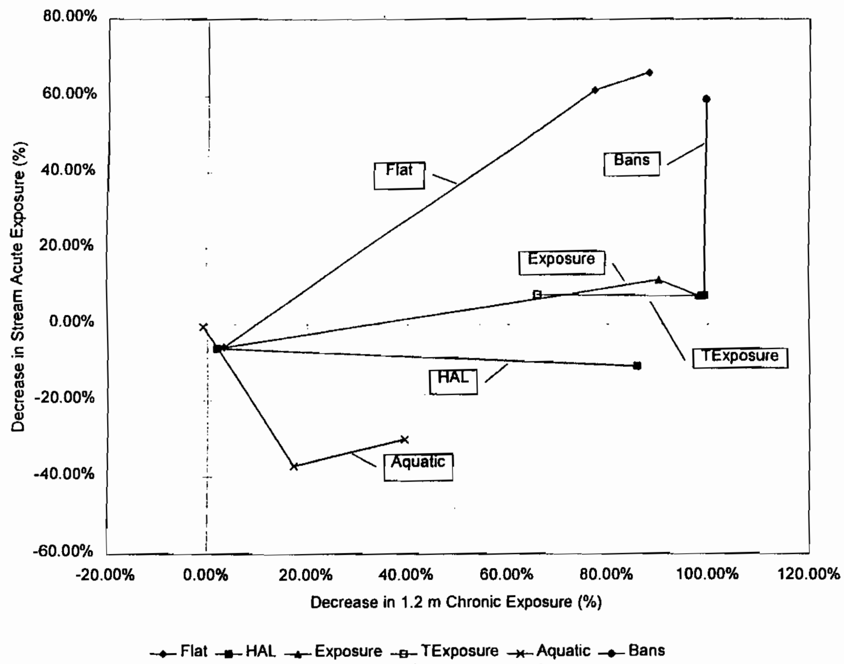

Fig. 8. Tradeoffs between decreased acute stream exposure and decreased 1.2 m chronic exposure.

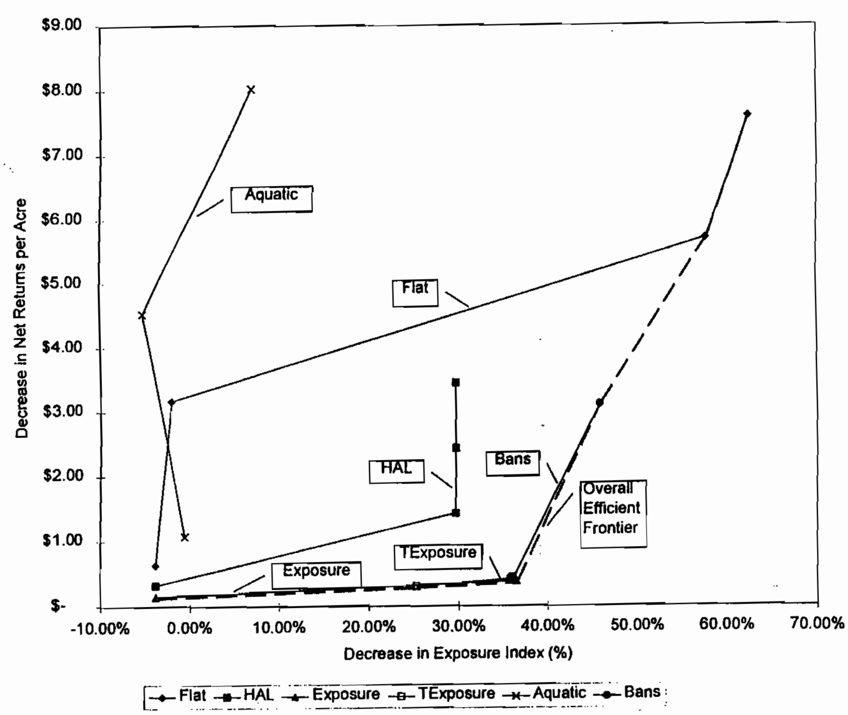

Fig. 9. Tradeoffs between decreased net returns and decreased overall exposure index.

based. Producers cannot avoid the taxes without reducing overall herbicide use. It takes a bigger stick to induce substitutions away from herbicide use to mechanical weed control than substitutions to alternative herbicides. The exposure-weighted taxes are effective because they make the specific herbicides that are likely to cause environmental damage relatively more expensive than the herbicides that are less likely to cause environmental damage. Producers can largely avoid the tax by substituting to the less damaging herbicides.

5. Conclusion

Ecological economic principles promote the ideal of global sustainability. Such broad principles become more concrete from specific case studies that explore how to make the abstract operational on the ground. Herein we examine one such case study — the design of environment-indexed incentive policies to reduced pesticide use in agriculture. We use the integrated economic-ecological CEEPES model to consider how alternative *risk-indexed incentive policies* affect economic and environmental indicators of

Table 7
Overall exposure index for each policy[a]

Scenario	Overall exposure index
Baseline	1
Atrazine Ban	0.63976
Triazine Ban	0.54211
\$1.00/lb Flat tax	1.03831
\$5.00/lb Flat tax	1.02087
\$10.00/lb Flat tax	0.42213
\$15.00/lb Flat tax	0.3755
\$1.00/lb HAL tax	1.03831
\$5.00/lb HAL tax	0.7033
\$10.00/lb HAL tax	0.7033
\$15.00/lb HAL tax	0.7033
\$1.00/lb Aqua. Bench tax	1.006
\$5.00/lb Aqua. Bench tax	1.0527
\$10.00/lb Aqua. Bench tax	0.93139
\$1.00/lb Exp. Tax	1.03831
\$5.00/lb Exp. Tax	0.63528
\$10.00/lb Exp. Tax	0.64376
\$15.00/lb Exp. Tax	0.64376
\$1.00/lb Till-Exp. Tax	0.74758
\$5.00/lb Till-Exp. Tax	0.6419
\$10.00/lb Till-Exp. Tax	0.6398
\$15.00/lb Till-Exp. Tax	0.6398

[a] Note: The exposure index is the average of the 1.2 m chronic exposure index, acute aquatic exposure index, and acute stream exposure index. Index values greater than one indicate a decrease in overall water quality compared to the baseline, while index values less than one indicate an increase in overall water quality compared to the baseline.

Table 8
Tax revenues raised by each policy

Scenario	Total tax revenues	% of Decrease in net returns
\$1.00/lb Flat tax	15 132 503	100
\$5.00/lb Flat tax	73 874 574	98
\$10.00/lb Flat tax	91 535 846	68
\$15.00/lb Flat tax	119 993 443	67
\$1.00/lb HAL tax	7 744 269	100
\$5.00/lb HAL tax	23 989 153	71
\$10.00/lb HAL tax	47 978 306	83
\$15.00/lb HAL tax	71 967 458	88
\$1.00/lb Aqua. Bench tax	25 353 897	100
\$5.00/lb Aqua. Bench tax	86 742 784	81
\$10.00/lb Aqua. Bench tax	160 413 158	84
\$1.00/lb Exp. Tax	3 589 027	100
\$5.00/lb Exp. Tax	1 530 955	17
\$10.00/lb Exp. Tax	197 976	2
\$15.00/lb Exp. Tax	296 964	3
\$1.00/lb Till-Exp. Tax	2 696 407	37
\$5.00/lb Till-Exp. Tax	344 820	4
\$10.00/lb Till-Exp. Tax	490 829	5
\$15.00/lb Till-Exp. Tax	736 244	7

well-being, and to explore how these indexed policies compare to a traditional command and control *herbicide ban*. The risk-indexed incentive tax targets specific herbicides and herbicide–tillage practices based on predicted groundwater exposure levels from the CEEPES model. Our results suggest that indexed taxes can be an effective and cost-efficient tool to reduce predicted groundwater exposure. We also find no significant advantage exists to fine tune the index to include herbicide–tillage combinations: the results are similar regardless of whether we target herbicides alone or we target herbicide–tillage combinations. This occurs because individual herbicides have more effect on groundwater quality than do tillage practices. Finally, we observed that environmental advisory benchmarks alone are not useful to construct effective tax policies.

No single policy tool dominated the other options for reducing groundwater exposure, surface water acute exposure, and surface water aquatic exposure. Different tools were more effective than others depending on the context. We used this data to construct an efficient policy set describing the best tool for the context using an equal weight on decreases in 1.2 m groundwater chronic exposure, surface water acute exposure, and surface water aquatic exposure. With the efficient policy set, we show both the achievable tradeoffs, and the policies that can achieve specific exposure levels. Our results indicate that exposure tax policies are most cost-efficient to achieve small percentage reductions in overall exposure, bans are most cost-efficient for moderate reductions, and flat taxes are most cost-efficient for high reductions. This suggests that the usefulness of information about predicted exposures depends on the desired reductions in exposure levels. For large reductions, for instance, crude tools like bans or flat taxes are most cost-effective.

Several possible extensions to our work may be useful for future policy analysis. First, our results for combined surface water and groundwater effects are subject to change depending on the relative weights placed on changes in groundwater and surface water exposure values. Future research should focus on developing weighting schemes based on social or policy preferences. Second, our framework allows for the comparison of any number of non-point pollution policies. Policies that could be added to our analysis include targeting based on measures of surface water quality and targeting based on geographical characteristics. Finally, our analysis was restricted to the corn and sorghum weed control decisions modeled in WISH. Adding additional crops to the WISH model could expand our approach to cover all herbicide use in the region.

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