A Biotic Control Perspective on Nitrate Contamination of Groundwater from Agricultural Production

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Agronomists consider the continuity and nutrient capturing properties of cover crops as important determinants of nutrient cycling in agricultural systems. Managing for these biotic control functions can help limit nutrient loss and groundwater contamination between main crop harvests. This simulation study highlights the potential role of cover crop management in a welfare economics framework. The objective is to find the optimal combination of nutrient input to the main crop, the extent of off-season cover crops, and crop functional diversity to maximize the sum of benefits from agricultural production and groundwater protection.

In managed agricultural systems, the importance of biotic control over the structure and functioning of ecosystems is increasingly appreciated (Chapin et al. 1997; Vitousek et al. 1997; Schläpfer and Schmid 1999). In particular, agricultural land conversions influence water and nutrient dynamics (Parton, Stewart, and Cole 1988; Matson et al. 1997). Agricultural production, watershed protection, and other processes and properties of ecosystems have been collectively called "ecosystem services" (Ehrlich and Wilson 1991). Based on estimates of restoration costs, proper functioning of these services in the United States provides economic benefits in the order of billions of dollars (National Research Council 1997; Chichilnisky and Heal 1998), and may contribute substantially to farm income if a market for these services could be established (Daily et al. 2000).

The watershed protection function of land vegetation is one of the most critically threatened ecosystem services provided by the world's biotic resources (Nolan et al. 1997; Chichilnisky and Heal 1998; OECD 1998). In particular, nitrate contamination from agricultural systems has stirred widespread health concerns over contaminated drinking water. In response, the protection of groundwater resources has emerged as an important objective of national agricultural policies (USEPA 1990; National Research Council 1997; EPA-USDA 1998; OECD 1998). Currently an intensive effort is underway to assess contingent values for water quality improvements through reduced nitrate concentrations (Edwards 1988; Sun, Bergstrom, and Dorfman 1992; Boyle, Poe, and Bergstrom 1994; Poe 1998; Poe and Bishop 1999).

With identification of groundwater contamination and valuation of groundwater protection becoming established, attention is now turning to the policy and mechanics of solutions. Agricultural research on cropping systems has identified quality, quantity and timing of nutrient inputs (Sexton et al. 1996), frequency and intensity of tillage (Patni et al. 1998), and the continuity of vegetation cover (Meisinger et al. 1991) as critical factors influencing nutrient leaching. However, nitrate contamination remains an important issue even in the presence of well-known and widely available nutrient conserving practices (Guimerà 1998; Lamarrie 1998; Horan, Shortle, and Abler 1999). Better integration of agricultural and ecosystem science, hydrology, economics, and environmental valuation in agricultural and environmental policy could lead to a more diverse and sustainable approach to agricultural ecosystem management (National Research Council 1997).

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This study examines the economic importance of biotic control in limiting nitrate leaching and conserving soil resources in agricultural ecosystems. A welfare function is used to weigh the benefits of agricultural output against the risks of groundwater damage. New features to this modeling approach include the development of continuous functions to capture biotic control effects, and the integration of agronomic relationships with empirical estimates of groundwater damage. Various management options are simulated and evaluated across multiple economic and environmental criteria. Estimation of modeling parameters draws on the best available data from ecology, agricultural ecology, and environmental valuation. The model and results are illustrative rather than case specific. Results point to the need to study these dynamics within the context of specific agricultural systems with site-specific data. The objectives of the current model are to (a) examine the potential role of functional aspects of cover crops in optimizing total benefits from agricultural land use, (b) identify important factors that affect decisions to invest in nutrient-conserving crop properties, and (c) consider a more integrated use of biotic control variables in the design of agricultural systems.

Nitrate Leaching, Cover Crops and Species Functional Diversity

High rates of nitrate leaching from agricultural soils have led to increasing numbers of private and public wells for drinking water supply that fail to comply with governmental water quality standards in North America, Europe, and other areas of the world. Technical end-of-pipe solutions with nitrate removal systems are being developed (Dorsheimer et al. 1997; Kapoor and Viraraghavan 1997). However, the costs, waste products, and public acceptability tend to work in favor of a reduction of the sources of pollution (Stevens et al. 1997; Yadav and Wall 1998). Indeed, most groundwater studies have focused heavily on the effects of different levels of fertilizer input (Botterweg, Bakken, and Romstad 1994; Teague, Bernardo, and Mapp 1997; Randhir and Lee 1997; Yadav and Wall 1998).

Agricultural and resource economic studies of water quality have approached the issue of nitrate contamination from several complementary angles. One aim of research has been to evaluate land and water resource management systems using multiple criteria or attributes (Sexton et al. 1996; Prato et al. 1996) and integrated models such as WAMADSS (Qui et al. 1998; Prato 1999) or ECECMOD (Vatn et al. 1999). A second approach has been to compare the costs and benefits of conventional agricultural management systems in combination with water purifying technology with those using best management practices that maintain groundwater quality (Yadav and Wall 1998). Yet another strand of research attempts to estimate the economic value of suggested improvements of groundwater quality relative to market commodities using contingent valuation (Sun, Bergstrom, and Dorfman 1992; Boyle, Poe, and Bergstrom 1994; Edwards 1988) or conjoint analysis (Stevens 1997).

Partly due to a lack of data on environmental values, environmental objectives have often been considered as separate criteria in the form of environmental constraints to optimization. Or, economic benefits of environmental improvements have been estimated using environmental valuation techniques but only to evaluate discrete management alternatives in benefit-cost frameworks. Environmental objectives have rarely been integrated as damage functions such as the function for nitrate contamination estimated by Poe (1998). Moreover, crop patterns that lead to different leaching rates have been integrated in the form of static best management versus conventional scenarios rather than as continuous choice variables and inputs to damage functions. In the field, crop vegetation plays a key role when tight cycling of nutrients can help ameliorate leaching. Within this context, questions such as the extent of vegetation cover across space and time (de Willigen 1991; Meisinger et al. 1991) and crop plant nutrient cycling properties could be important crop ecosystem management variables (Swift and Anderson 1993; Swift et al. 1998). Cover cropping is an agricultural practice that benefits soil-crop systems by increasing nutrient supply for subsequent crops, reducing soil erosion, improving physical soil properties, and conserving nutrients in the soil-crop system.

This study focuses foremost on the effects of cover crops on nitrate (NO_3^-) conservation, but also considers the complementary benefits from reducing soil erosion. Nitrate-nitrogen is a difficult nutrient to manage in agricultural systems. Crops depend on a high quantity, either from the soil itself or by fertilization. In the soil, nitrate is highly mobile and moves freely with percolating rain or irrigation water. In humid climates, percolation of water occurs most frequently during fall, winter, and spring when evapotranspiration is low and precipitation exceeds the uptake of water by plants. Therefore, the primary nitrate leaching season is between November and May, typically after the harvest of the main crop. Cover crops sown in the fall can reduce the amount of nitrate leached into the groundwater by reducing the amount of water available for leaching through transpiration, and limiting the soil nitrate pool by converting nitrate, which is continuously released in the mineralization of plant, animal and microbial detritus, into immobile plant organic nitrogen. In addition, a cover crop between harvest crop plantings reduces soil erosion.

The choice of the cover species depends mostly on the cropping system and climate. Various cover crop groups are in use worldwide, each with their specific advantages and disadvantages related to their respective physiology (Meisinger et al. 1991). The effectiveness of cover crops depends strongly on how well subsequent release of nitrogen from cover crops in spring is synchronized with uptake by the following main crop. In general, legume cover crops have limited ability to reduce leaching, with leach reduction averaging 23%. Their main benefit lies in supplying nutrients to the following crop. Among non-legumes, grasses reduce leaching by 60% on average, while brassicas (such as mustard) can average 60 to 75% (Meisinger et al. 1991). However, brassicas are not as winter-hardy as grasses and some release nitrogen more rapidly when killed in the spring. Also, rapid establishment and vigorous growth during the fall is found in some species, while physiological activity around the freezing point in winter may be higher in others.

A species that optimally combines the various favorable traits has not been found to date, but there is considerable evidence that well-designed mixtures of species could complement each other to maximize performance. Cover crop mixtures have been evaluated for N-supply to the following season's crop (Moschler et al. 1967; Mitchell and Teel 1977), and optimal crop mixtures for nutrient retention have been identified as an important area for further agronomic research (Meisinger et al. 1991; Brummer 1998). To date, experimental studies in ecosystem science have found considerable effects of plant species diversity on nitrogen pools below the root zone (Ewel, Mazzarino, and Berish 1991; Tilman, Wedin, and Knops 1996; Hooper and Vitousek 1997 and 1998) and on nitrogen leaching (Scherer-Lorenzen 1999). Cover crop benefits of reducing soil erosion accrue similarly from the increased continuity of the crop cover in fall to spring seasons. The percentage of the soil surface covered with residue has been identified as an extremely important controlling factor in experimental studies (Fryrear 1995).

Although harvest for a profit is possible, cover crop planting has mostly focused on optimizing nitrogen supply for the main crop or limiting erosion. Establishment costs for winter cover crops in the United States are in the range of 25 to 70 dollars per hectare, depending on equipment needs (Meisinger et al. 1991; Roberts et al. 1998). Optimization of the cover crop over space and time involve additional cost considerations. For example, late plowing of cover crops may delay the supply of nutrients from residues to the main crop. Reduced-tillage management allows strips of the cover crop to remain between rows of the main crop during early stages of establishment (Carter 1994). Cover crops may also remain before emergence and after harvest of the main crop to improve soil conservation (Abdin et al. 1997). Additional costs of optimal diversified crops may arise from precision management information, seeding, and sequential planting. In the instances when diversification in a main crop is possible, changing harvest methods could also introduce new costs.

A Simulation Exercise

Resource management has typically been conceptualized as a problem of optimal allocation over time, or dynamic optimization. This study develops an objective function that models welfare from both private income from agricultural production and public benefits from ecosystem services that are jointly produced within a managed agricultural ecosystem. The joint social welfare function is then evaluated across numerous management scenarios, and optimized for nutrient input, cover crop extent, and species functional diversity. For reasons of parameter availability, continuous corn is chosen as the model production system. The modeling exercise is based on the following assumptions about system characteristics.

(a) A soil resource accumulates depending on site conditions (climate, geology, etc.).
(b) The soil resource is decreased by productive land use through erosion, which depends on the crop cover.
(c) The harvest of the main crop (corn in this case) is positively related to fertilizer input and topsoil depth.
(d) The agricultural crop determines local biogeochemical processes such as nutrient cycling. Nitrate leaching into groundwater is further determined by the percentage of ground area in cover crops.
(e) The vegetated portion of the soil surface in winter between harvest and establishment of the new corn crop can be increased from

zero to a maximum of 95% through cover crops.
(f) Nutrient cycling efficiency of cover crops can be increased by specifically designed species mixtures.
(g) Leached nitrate accumulates in the groundwater body. Differences between the inflows of nitrate via leaching and the outflows through water wells and diffusion lead to a change in the nitrate concentration of the groundwater.
(h) Groundwater wells supply drinking water to local households. The subjective perception of the groundwater damage through contamination is reflected in a concave damage function.
(i) A welfare function integrates the profits from corn production with the perceived public costs of reduced drinking water quality.

These assumptions outline a conceptual framework within which an evaluation of basic management scenarios, welfare maximization, and sensitivity analysis is performed. The problem is to find the optimal combination of nitrogen fertilizer input (I), extent (ground cover percentage) of off-season cover crops (V), and crop functional diversity (S) (number of locally interacting species with distinct physiologies related to nutrient uptake). In the optimization exercise, the objective is to maximize the discounted sum of net benefits (NPV) over time (from $t = 0$ to T) of agricultural production:

(1)
$$\max \quad NPV = \sum_{t=0}^{T}[P(t) - D(t) - C](1 + \rho)^{-t},$$

where $P(t)$ is the annual net revenue from agricultural production (not considering costs of cover crop management), $D(t)$ is the estimated environmental cost of increased nitrate concentration in the groundwater, C is the time invariant cost of cover crop management, and ρ is the discount rate (set at 5% in the baseline scenarios). Dropping the time argument, the expressions for P, D, and C are functions of the three choice variables (I, V, and S) and the following relationships for resource dynamics, agricultural output, nitrate leaching, benefit and cost functions, and groundwater damage.

Soil Resource Dynamics

Soil formation per year, G, is a function of accumulation of available minerals and organic material and is modeled on a soil formation rate (a):

(2) $$G = 0.0615a.$$

Assuming a representative soil mass per volume of 1.45 g/cm^3 yields the conversion factor of 0.0615 to obtain soil depth (in millimeters) from tons of soil per hectare. In subsequent simulations a = 0.65 tons per hectare, a value representative of soils in Europe, North America or China (Scheffer and Schachtschabel 1992). Thus, equation (2) implies a soil formation rate of 0.04 mm/ha/yr. Although the speed of soil formation is known to depend on the activity of diverse organisms (Naeem et al. 1994; Parton, Stewart, and Cole 1988), there are no empirical data available to integrate effects of vegetation diversity into this function.

Soil loss through erosion is usually modeled on crop and residue biomass, such as in the well-known EPIC model (Williams and Renard 1985). However, unlike some cash crops, cover crops may produce little biomass despite providing good soil protection. Thus, for cover crops, soil erosion may be more directly related to the proportion of total ground cover than to crop biomass (Fryrear 1995). An erosion rate (b) of 8.4 tons per hectare per year, an estimated long-term mean of soil erosion by water in the U.S. corn belt (Lee, Phillips, and Dodson 1996), is assumed for a rotation without cover crops. Assuming that the corn crop provides full ground cover during part of the year, erosion is estimated based on the off-season crop cover, which includes the early, erosion-prone stages of the main crop. Assuming a linear functional form and using the conversion factor of 0.615 mm/t to obtain a measure of soil depth, soil loss per year is described as:

(3) $$E = 0.0615b(1 - V),$$

where V is the average proportion of the off-season vegetation cover, which can vary between 0 and 0.95. Implied reduction of erosion corresponds with empirical data on the effect of winter cover crops on reducing yearly erosion in continuous corn rotations by 96% (Wendt and Burwell 1985).

The difference equation for the soil resource stock (R) thus becomes:

(4) $$R(t) = R(t - 1) + G - E.$$

The resource base (R) is measured directly as the depth of the A soil horizon (USDA 1993) which holds most of the available plant nutrients. Whether a change in the resource stock will result in a reduction of crop yield depends on the initial depth of R at time t.

Agricultural Output

Agricultural production ($A(t)$ in t/ha/yr) as a function of nutrient input (I in kg N/ha/yr) is modeled as an empirical function obtained from a corn production system in Minnesota (Sexton et al. 1996). The relationship between corn yield and topsoil depth follows Xu et al. (1997), where yield on severely eroded soil (A-horizon < 15 cm) was reduced by about 30% relative to the yield of a fully productive soil (with an A-horizon of at least 30 cm). For R between 15 and 30 cm, the erosion impact was linearly interpolated (equation (7) below). For soil depth above and below this interval, yield was assumed to be unrelated to topsoil depth. This implies the following functions for corn yield response to fertilizer application:

(5) $A(t) = 1.71 + 0.0355I - 0.0000709I^2$ when $R > 30$ cm

(6) $A(t) = 0.7\ (1.71 + 0.0355I - 0.0000709I2)$ when $R < 15$ cm

(7) $A(t) = (0.4 + 0.6R(t)/300)(1.71 + 0.0355I - 0.0000709I^2)$ when $15 \text{ cm} < R < 30$ cm

Such discontinuous relationships are common in biological systems when large ranges of parameter values are considered (Schmid et al. 1994). This specification of yield does not consider nitrogen carry-over effects, implying that the harvested cover crop contains the plant-available nitrogen otherwise leached.

Nitrate Leaching

Nitrate leaching (L in kg/ha/yr) is composed of two additive components:

(8) $$L = L_{mc} + L_{bc}$$

(9) $$L_{mc} = 21.5 - 0.132I + 1.17I^2$$

(10) $$L_{bc} = 100 - 60V - 60V(1 - (0.52 + 0.72e^{-0.41S}))$$

The first component estimates leaching from the corn crop (L_{mc}) as a function of nutrient input to the main crop based on experimental results of Sexton et al. (1996). The second component describes leaching from the soil between two corn crops (L_{bc}). This winter leaching L_{bc} is expressed as the winter leaching rate without cover crop (100 kg N/ha, a round average of the literature data in Meisinger et al. [1991]) minus the leaching reductions due to both cover crop extent and diversity. Following Meisinger et al. (1991) who estimate reduction of nitrogen leachate by various cover crop species for the United States, the management of a single-species (grass) winter cover crop is assumed to reduce leaching by 60%, This reduction is effective on the proportion of ground planted to cover crops (V). Special management practices to increase ground cover are not modeled explicitly. Further reduction of leaching due to increased plant diversity (S), which is the number of competitively interacting species with complementary nutrient use, of the cover crop is modeled using the relationship obtained in a large recent experiment with grassland species in a seminatural grassland in Minnesota (Tilman, Wedin, and Knops 1996; Tilman et al. 1997a and 1997b). The experiment provides a relationship for the reduction of nitrates in the rooting zone of plant mixtures relative to the nitrate concentration in average single-species vegetation.[1] The use of this function within the range of nitrate concentrations in question requires assuming that soil nitrate is leached into the groundwater body proportionately to its availability in the rooting zone (Hansen et al. 1991; Tietma et al. 1997). Leaching coefficients are season averages. For simplicity it is assumed that there is instant connectivity between surface conditions and groundwater contamination. Consistent with empirical findings for several cover species, a suitable cover is assumed to not interfere with the corn crop (Abdin et al. 1997).

For simplicity, the agricultural land surface in the model (u) is assumed to correspond with the groundwater catchment area. Nitrogen accumulation in the groundwater body is modeled by the mass balance equation:

(11) $$Z(t + 1) = Z(t) + (uL - mZ(t))/r,$$

where $Z(t)$ is the nitrate concentration in the groundwater in year t (in mg N/liter), u is land surface in hectares, and L is the leaching rate (in kg N/ha/yr) which is assumed constant for any specific management. The parameter m is the yearly volume of water (in m^3) that leaves the groundwater body by wells or by diffusion (corresponding to the recharge through water percolation of 0.5 m/yr), and r is the volume of the groundwater body (in m^3) in the catchment area u.

Benefit and Cost Equations

Annual net revenue from corn production (not considering cover crop investments) is given by

[1] The function for the relative effect of diversity in the last part of equation (10) was modified from the original fit (with $R^2 = 0.22$, $n = 147$, $P < 0.001$) to obtain effects relative to one-species vegetation.

(12) $$P(t) = u\pi A(t),$$

where the parameter π is the revenue per-ton of corn.

The cost of nutrient-conserving cover crop management is estimated by

(13) $$C = (V^2 g + V^2(S - 1)^2 d)u$$

and represents the additional production costs of high ground-cover, high-diversity cover cropping systems. Although planting time, method, and seeding rate have major impacts on fall growth and N uptake (Meisinger et al. 1991) empirical data for cover crop costs as a function of effective ground cover could not be obtained. Per hectare investments are assumed to increase with the square of the off-season vegetation cover V on the basis that a low ground cover may occur spontaneously while nearly full ground cover requires increasingly precise timing and special management techniques such as inter-seeding within the standing crop (Meisinger et al. 1991). Thus, for instance, assuming a cost parameter g (costs per hectare for establishment and maintenance throughout the off-season) of \$70/ha, a 60% off-season ground cover would cost \$25/ha, an 80% ground cover would cost \$45/ha, and a 90% ground cover would cost \$56/ha for a single-species cover crop. This range of costs for increasingly effective cover crops corresponds with \$25–50/ha establishment costs for grass or brassica cover crops in the United States (Meisinger et al. 1991).

The parameter d for costs of cover diversity is again difficult to estimate without a site specific study. Cover crop diversity is assumed to require additional investments that are a quadratic function of species number S with a base-case parameter d of \$10/ha. Sensitivity to this parameter will be addressed in the results section.

Damage Function

During the past decade a body of contingent valuation (CV) studies on ground water quality improvements has emerged (see Boyle, Poe, and Bergstrom 1994; Poe 1998). These studies provide important policy relevant information. Yet, most surveys involve strongly hypothetical situations that do not confront the respondents with detailed information on the current quality of their water supply. Contingent values collected under these conditions may not reliably predict willingness-to-pay (WTP) for a population actually experiencing contamination (Poe and Bishop 1999). From a modeling perspective, a further disadvantage of many past studies is that they were not designed to estimate continuous damage functions that allow comparison of benefits and costs over a range of exposure levels. Past research has mostly collected information about the valuation of specific hypothetical water quality improvements.

Poe (1998) was the first study based on both actual exposure levels and informed respondents to estimate a nitrate damage function. Poe estimated the WTP for a community-wide groundwater protection program in which "all wells . . . will definitely be kept below the government health standard of 10 mg/l." WTP can be interpreted as an option price for a reduction in future health risk exposure. If the concentrations resulting from current agricultural management will *not* definitely be kept below the standard then a subjective damage is perceived which can be averted by investment into the program. For the purpose of modeling, "will definitely" in the Poe survey is interpreted as "for the next 100 years." Thus for agricultural management specifications that are expected to violate the standard within the next 100 years (i.e., past the 10 mg/l before $t = 99$ in the simulation), Poe's nitrate exposure model implies a damage function of:

(14) $$D(t) = (180.75Z(t)^{0.352} - 146.63)h$$

The variable h represents the number of households with water supply from the local groundwater. Subjective damage is set at zero when land management is effective at keeping nitrate concentrations below 10 mg/l. Individual WTP survey responses cannot be aggregated to a total damage function where individuals with different initial exposure are operating at different expected utility surfaces (Poe and Bishop 1999). This concern was not addressed.

Simulation Results

Table 1 details the baseline parameter set, including initial soil stock and groundwater nitrate, used to simulate three and optimize two scenarios for the system outlined by equations (1) through (14). The base case for comparison is the "conventional crop" that is calculated under a production-as-usual assumption. The hydrological parameters were chosen to yield a nitrate concentration in leachate from the "conventional" crop that corresponds with long-term data from sensitive agricultural areas (e.g., BUWAL 1993). Initial soil depth (A-horizon thickness) corresponds with a soil on the boundary between "depositional phase" (>30 cm) and "slight erosion" (20–30 cm) (USDA 1993). The three simulation scenarios—conven-

Table 1. Base-Case Parameter Values

Parameter	Description	Units	Value
a =	Growth rate of the soil stock	tons/ha/yr	0.65
b =	Soil erosion rate (bare soil)	tons/ha/yr	8.4
d =	Species diversity cost parameter	\$/ha	10
ρ =	Real annual discount rate	/yr	0.05
π =	Net revenue from agric. product	\$/ton	70
g =	Cover cropping cost parameter	\$/ha	100
h =	Potential water consumers	households	1000
m =	Groundwater recharge in area u	m^3/yr	$5 \cdot 10^6$
r =	Groundwater volume in area u	m^3	10^8
u =	Agricultural land/watershed area	ha	1000
R_0 =	Initial soil stock	mm	300
Z_0 =	Initial groundwater nitrate	mg/l	5
T =	Considered time horizon	years	100

tional, cover crop, and diversified cover crop—use pre-set choice variables to represent a range of different agricultural management types (with equal N-input) to examine resulting average annual leaching rates, cumulative nitrate contamination level, soil resource stock, and total net present value. The two optimization scenarios maximize total net present value for the cases with 0 and 1000 households potentially consuming local well water. Results are tabulated in table 2.

In each of the management simulations and optimizations, groundwater nitrate concentration increased across the 100-year time horizon from the initial value of 5 mg/l, highlighted in figure 1. The increase in the conventional crop simulation amounted to about 10 mg/l in the first ten years, then asymptotically approached 25 mg/l due to the compensating effect of nitrate removal through water wells and diffusion. Management of a 90% cover crop had a strong effect on contamination levels, reducing the asymptote to 15 mg/l. In the third simulation, diversified cover crops (i.e. three selected species with complementary resource use patterns) further reduced the increase of groundwater nitrate concentrations. However, despite cover crop management, national regulatory safety standards of approximately 10 mg/l are not met due to high nitrogen input to and leaching rates from the main crop.

The two optimized scenarios generated very different choice variable sets depending on inclusion of stakeholders. In the zero-stakeholder case (h = 0), benefits from cover crops only come in the form of reduced erosion. In this case, a nutrient input of 250 kg/ha/yr was optimal, with only a marginal investment into one species of cover crop. The NPV maximizing combination of choice variables for the 1,000-household case included nitrogen input of 154 kg, an effective cover crop over 95% of the soil surface area between main crop rotations, and an optimal cover crop diversity of 2.4. This optimized stakeholder scenario reduces leaching by nearly two-thirds over the zerostakeholder optimum. This results in sustained nitrate concentrations below the 10 mg/l policy threshold. Compared to the three non-optimal management scenarios, the stakeholder optimum results in a considerably higher overall benefit due to compliance with the health-standard nitrate concentration and nullification of groundwater damages.

Table 2. Base-Case Simulation Results

	Simulation Results							
Scenario	I N Input (kg/ha)	V Cover crop (%)	S Plant diversity index	L Leaching rate (kg N/ha/yr)	Z (99) Nitrate concentration at t = 99 (mg/l)	R (99) Soil resource at t = 99 (mm)	Total NPV h = 1000 (Mill. \$)	Total NPV h = 0 (Mill. \$)
Conventional	200	0	1.0	131.9	26.2	252.8	5.391	11.669
Cover crops	200	90	1.0	75.3	15.0	298.8	5.703	10.745
Diverse cover	200	90	3.0	64.4	12.8	298.8	5.276	10.029
Max. NPV (h = 1000)	154	95	2.4	50.2	10.0	301.8	9.290	—
Max. NPV (h = 0)	250	8	1	146.5	29.1	257.8	—	12.030

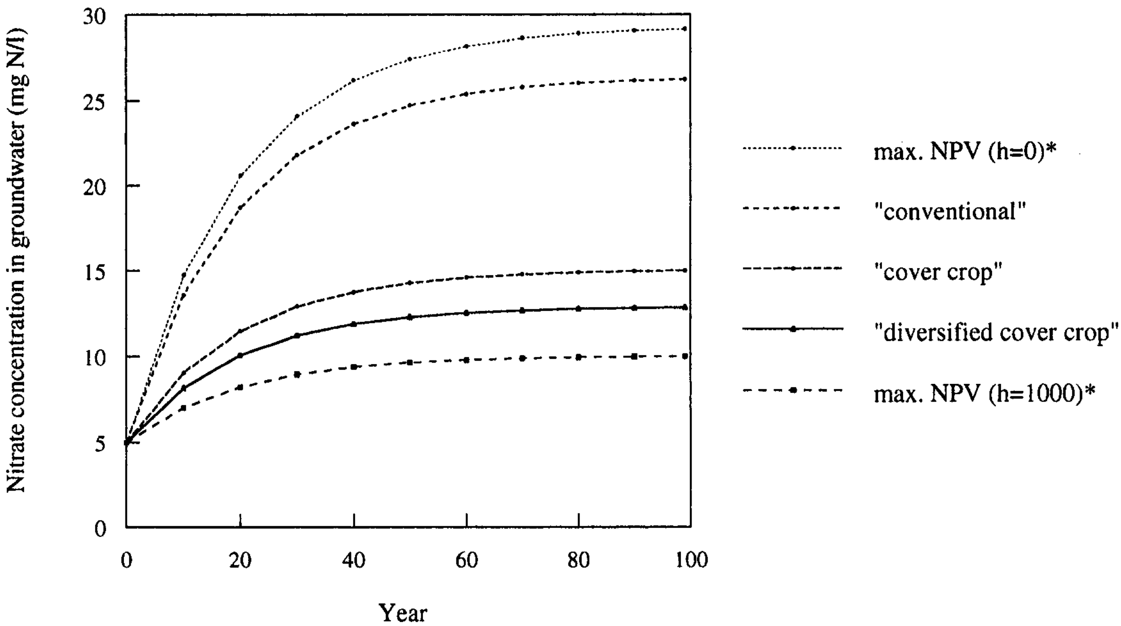

Figure 1. Development of nitrate-N concentrations in the groundwater for five basic management scenarios (see table 2 for scenario descriptions) (*h: number of groundwater consumer households)

Sensitivity Analysis

The base-case parameter set outlined in table 1 resulted in optimal nitrogen input (I), crop cover (V), and species diversity (S) of 154 kg/ha, 95%, and 2.42 species. This section examines the affect on these optimal values when certain critical parameters and model assumptions are varied. Specifically, the effects of the number of stakeholders, initial pollutant concentration, net revenue from the main crop, costs for improved cover crop composition, discount rate, and nitrate compliance uncertainty are examined.

Figure 2 demonstrates the impact on optimal nutrient input, percent cover crop, and functional diversity when the number of stakeholders is varied from 0 to 1000 households. With no stakeholders for groundwater quality, optimizing net revenue yields limited investment into biotic control and required high nitrogen input (248 kg/ha/yr). Even under the assumption of no damages from poor water quality, biotic control reduces soil erosion which benefits agricultural yield. However, the discounted long-term benefits are low compared to the immediate cost of yearly cover crop management. As the number of stakeholders increases to 100 households, a small investment of 12% cover and a negligible reduction of nitrogen input (to 246 kg) becomes optimal. At 400 households, optimal choice variables reach a level in compliance with the 10 mg/l nitrate standard. Due to groundwater damages, over this range of households the optimal NPV decreases from $12 million to $4.8 million.

There was only a slight effect of the initial groundwater concentration, $Z(0)$, on the variable choice set. As $Z(0)$ increased from 0 to 15 mg/l, optimal nitrogen input decreased from 154 to 153 kg/ha/yr. Crop cover and diversity remained at 95% and 2.4 species, which help attain contamination levels just below 10 mg/l by $t = 99$.

Costs of improved cover crop combinations are difficult to estimate. Figure 3 demonstrates the impact of this parameter on the optimal variable set. The investments required for cover crop diversity did not affect the optimal extent of the cover crop. This is due to higher marginal benefits from investments into cover crop area than cover crop diversity. However, at full cover (95%), diversification of the cover crop was efficient in reducing costs of groundwater damage. Increasing costs in crop diversity from $5 to $20 yielded a reduction in optimal diversity from 3.1 to 2 species.

An increase in crop net revenues per ton of product caused a sharp increase in optimal fertilization levels from a zero input at $0 per ton net revenue to 131 kg/ha input at $20 per ton. Over the same interval, optimal plant diversity decreased from 2.4 to 1.6, and optimal vegetation cover increased from 85 to 95%. As crop net revenue further increased from $20 to $200 per ton, optimal nitrogen

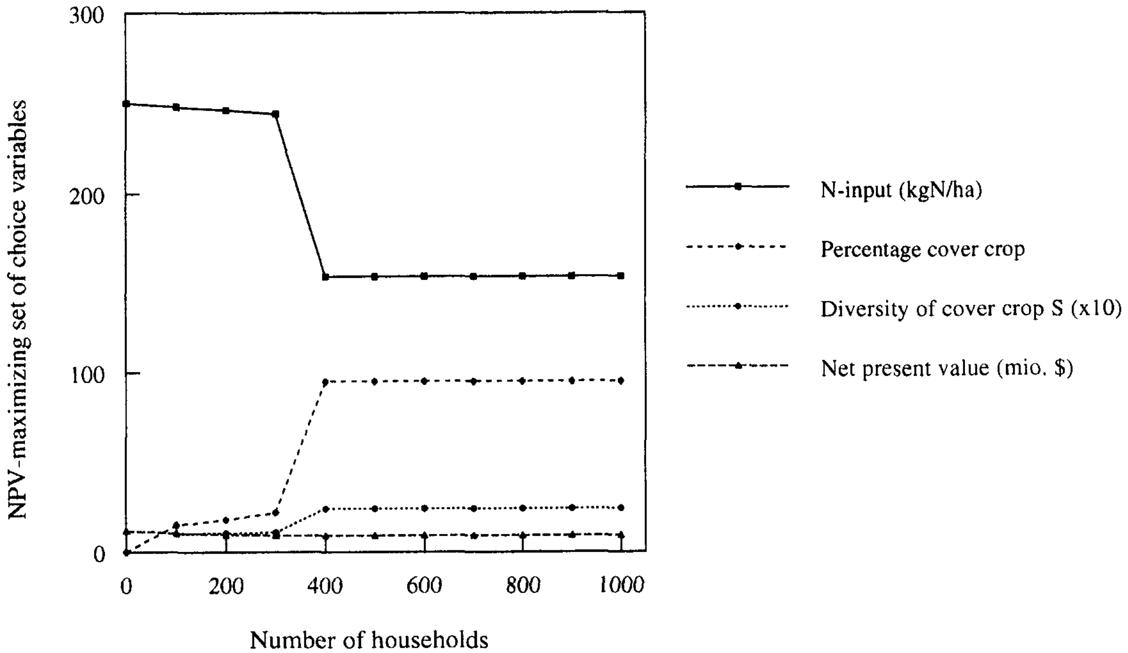

Figure 2. Sensitivity of optimal management to the number of local stakeholders of groundwater quality

input increased from 131 kg to 164 kg, crop cover remained at its maximum, and the benefits of diversity take effect, increasing from 1.6 to 3.0 species. Each of these management scenarios maintained the 10 mg/l health standard.

Varying the discount rate had little effect on optimal management. Rates between 0.01 and 0.1 resulted in similar management practices. This is partly due to the fact that, in contrast to soil erosion effects, both groundwater benefits and operation costs of cover cropping occur near the same time, with only a minor delay of the costs of groundwater damage.

Groundwater quality at $t = 99$ (which can be forecasted) is the criterion for defining compliance and non-compliance regarding the 10 mg/l national health standard. In the basic model no preference is assumed for a reduction of nitrates below 10 mg/l. Scenarios with $Z(t = 99)$ approaching 10 mg/l usually appeared optimal. In a separate model, damage according to eq. 14 is assumed to be perceived in even those agricultural management scenarios that are expected to comply with the standard into the future (effectively dropping the assumption of no preference for a reduction below 10 mg/l). Figure 4 outlines this scenario over the range of 0 to 5,000 households. In contrast to the basic model, a large number of stakeholders of water quality necessarily drives benefits below zero. There is no management option available to reduce contamination at a reasonable cost when damages from nitrate pollution are perceived at any level above zero.

Discussion

Watershed-scale time series of groundwater quality under different cover-crop regimes are not currently available (Dabney 1998). Insights into the benefits and costs of biotic control currently rely largely on mathematical models, and their necessary assumptions and simplifications. The critical methodological assumptions of this study include: (1) nitrate management options are limited to a single cash crop, fertilizer level, and cover crop extent and diversity; (2) other techniques such as perennial grassland plantings, no-tillage management, reforestation, or nitrate removal by filtration may be viable nutrient management alternatives that were not studied; (3) the effects of crop species diversity critically depend on empirical relationships obtained from a small number of best-available studies; (4) system optimization is viewed from the standpoint of a social planner with a fixed discount rate and time horizon; and (5) benefit functions from contingent valuation (Poe

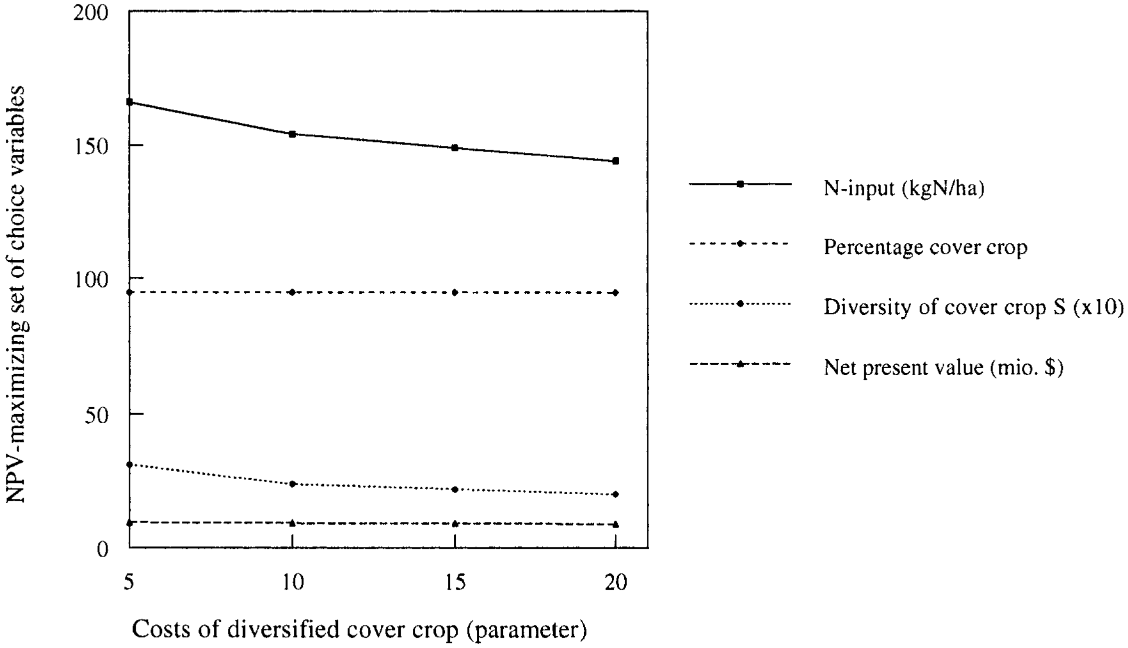

Figure 3. Sensitivity of optimal management to the costs of diversified cover crops

1998) are assumed to be transferable to other locations and situations.

The model represents a generic agricultural situation with the major cause-effect relationships of policy interest. This study pieces together information from a number of research projects, however, provides a framework to gather biological, hydrological, agronomic, and welfare economic relationships for site-specific research. Variations of the assumed constellation of leaching and soil erosion

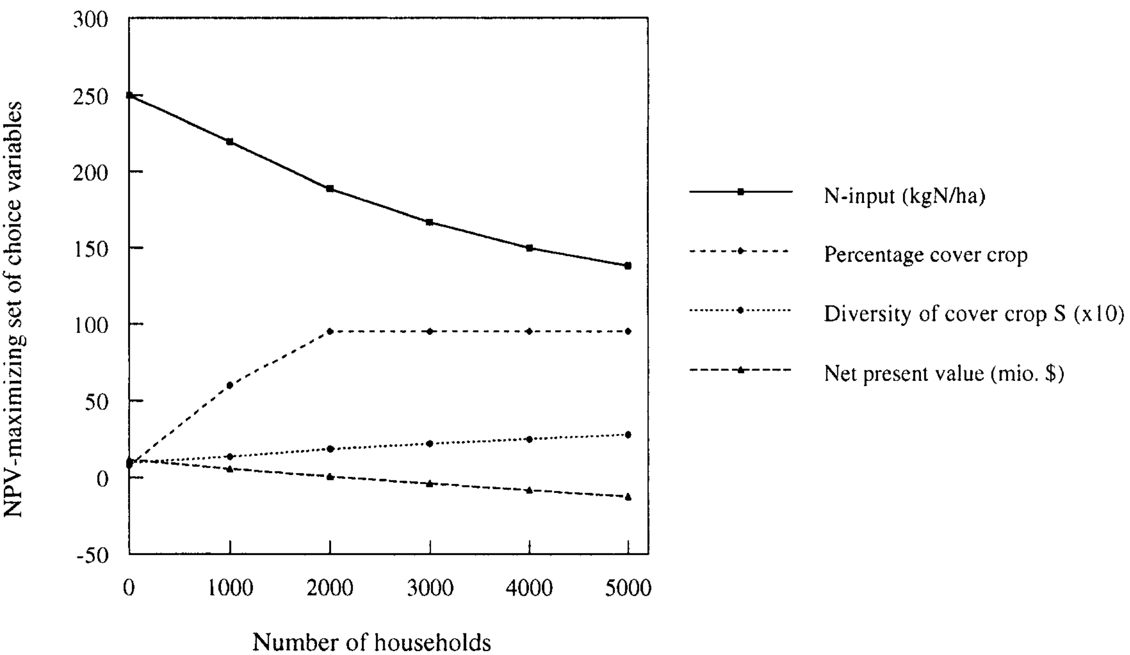

Figure 4. Sensitivity of optimal management to the number of local stakeholders with compliance uncertainty (damage function independent on forecasted water quality)

proneness, aquifer properties and consumer preferences (WTP) implied by the model parameters may exist in a number of sensitive watersheds throughout the United States and Europe. However, for important parameters such as those of the damage function there are few empirical data available. It is difficult to judge how broadly the high significance of cover management applies. For site-specific applicability of the modeling approach, parameters would need to be estimated on a case-study basis.

Agronomic and agricultural economics studies often distinguish few discrete management alternatives, such as "rotation with cover crop" versus "rotation without cover crop" or "conventional" versus "best management." Modeling results point toward the choice of cover crop extent and diversity as significant agricultural management variables in their own right, particularly in the presence of high damage to groundwater stakeholders. When contamination levels exceeded the policy standard of 10 mg/l and the number of stakeholders was increased above the base case of 1000 households, the overall net present value of agricultural production was quickly driven to high negative values, implying high social costs of fertilizer-intensive agriculture. One implication is that highly effective leaching control solutions may be required to render productive agriculture acceptable in the vicinity of urban or other densely populated areas.

The results of the sensitivity analysis raise some important questions regarding the interpretation of the damage function derived in Poe (1998).[2] First, how should the damage function be interpreted when perceived nitrate damages are accounted for below the government health standard level of 10 mg/l? Second, how should managers respond if negative net values imply that agricultural activities should be further restrained, even if public health standards are being achieved? And third, should public perceptions of nitrate damages be regarded as valid above the government health standard level, yet ignored below that level? Given the original contingent valuation, there seems to be some validity in the initial simulation assumption. However, the perceived costs were clearly sensitive to actual exposure, even in the range below the health standard. In this light, the standard may not be relevant when there is a demand for even higher water quality.

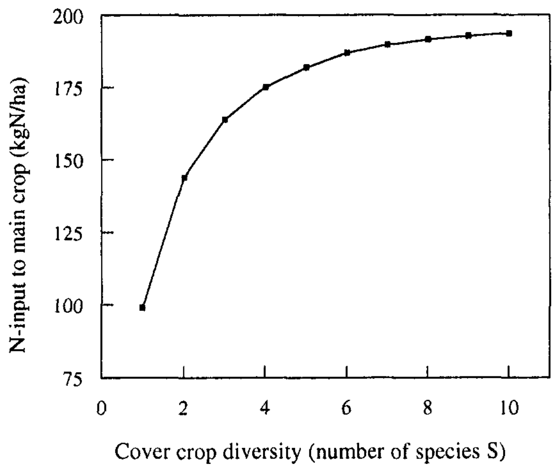

Figure 5. Combinations of N-input to the main crop and species functional diversity in the off-season cover crop that maintain groundwater nitrate just below 10 mg N/l

From a farm management perspective, the compensating erosion control benefits of biotic control help offset the social costs of nitrogen fertilization. Optimal sets of the choice variables include a full (95%) cover crop across a wide range of parameters. Figure 5 plots combinations of nitrogen input and cover crop diversity that keep nitrate concentrations in the groundwater below 10 mg/l when crop cover is at its maximum. For a specific policy level of contamination, an appropriate increase of cover crop functional diversity allows up to double the nitrogen input to the main crop due to increased nutrient use efficiency. Modeled soil erosion effects on agricultural productivity were not large and immediate enough to expedite large investments into resource conservation. However, benefits to water quality mattered as soon as a relatively modest number of stakeholders were present.

Conclusions

Accounting for production as well as biotic control services of crops may be a useful approach to agricultural policy design. Assuming that groundwater contamination through leaching of nitrates in agriculture has high social costs, the extended use of cover crops may be highly desirable, even from a somewhat limiting and simplifying welfare economic perspective. Empirical models estimated in local situations can provide further useful information for site-specific nutrient management plans (EPA-USDA 1998; Ribaudo and Kuch 1999). De-

[2] We are grateful to a reviewer who pointed us to these important issues; although we perhaps raise more questions than provide definitive answers to them.

velopment of transfer payment schemes from urban to rural households may have positive welfare effects (Loehman and Randhir 1999). Ground cover parameters, as implemented in the model, could serve as pragmatic criteria for such green payments when direct measurement of environmental effects is not feasible (Horan, Shortle, and Abler 1999).

If the cost of planting mixtures of cover crop species is not prohibitively high (<$10/ha for assumed base-case parameters), then increasing the functional diversity of cover crop species may also be cost-effective. A biotic control perspective may stimulate additional research into the development of optimal crop combinations and crop rotations for reducing nutrient losses to groundwater. There may also be spill-over benefits on other off-site effects such as surface water pollution and sedimentation. The present model modestly suggests that the recognition of crops as both commodities and drivers of ecosystem function will give impetus to improved agricultural management practices that push even high-input, industrial agriculture toward a more sustainable basis.

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