Balancing Bio-Energy Cropping Benefits and Water Quality Impacts: A Dynamic Optimization Approach

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The relationship between bio-energy feedstock production and water quality has received little attention from economists. Here, an optimal control model is used to determine the optimal amount of land to convert to the production of energy feedstocks, specifically ethanol corn, taking into account potential impacts on water quality. Based on comparative static analyses of an optimal control model, and a numerical application, we find that the optimal proportion of land to shift into bio-energy production from a baseline use, such as the Conservation Reserve Program, depends on key model parameters, specifically the rate of degradation of the pollutant and the link between the intensity of bio-energy feedstock production and the rate of change in the pollutant stock. Yet, there is a limit to how much land should optimally be converted as society must trade-off its desire to mitigate climate change against its willingness to accept a decline in water quality.

Le lien entre la production de matières premières bioénergétiques et la qualité de l'eau a attiré peu d'attention de la part des économistes. Nous avons utilisé un modèle de contrôle optimal pour déterminer les superficies optimales à convertir à la production de matières premières énergétiques, en particulier le maïs destiné à la production d'éthanol, en tenant compte des répercussions potentielles sur la qualité de l'eau. D'après une simulation numérique et des analyses de statique comparative obtenues à l'aide d'un modèle de contrôle optimal, nous en sommes venus à la conclusion que les superficies optimales à convertir à la production de matières premières bioénergétiques à partir d'un instrument de référence, tel que le Conservation Reserve Program *(Programme de réserve des terres sous conservation), dépend des paramètres clés du modèle, particulièrement du taux de biodégradation des polluants et du lien entre l'intensité de la production de matières premières bioénergétiques et le taux de variation du stock de polluants. Il existe tout de même une limite quant aux superficies à convertir étant donné que la société doit faire un choix entre son désir d'atténuer le changement climatique et son acceptation d'une diminution de la qualité de l'eau.*

INTRODUCTION

There is now a great deal of interest in producing fuels and electricity from biomass. Prices of petroleum-based fuels rose to their highest level during 2008 and, although they have subsequently fallen as a result of the global financial crisis, are expected to rise again as economies recover. At the same time, European and American policy makers are concerned about increasing dependence on sometimes unreliable foreign suppliers of petroleum and natural gas. Interest in the potential of biofuels has also been driven by concerns about climate change, the lure of untold numbers of new jobs from a developing

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Table 1. Net climate warming relative to fossil fuel CO_2 savings

Crop	Biofuel	Nitrogen use efficiency 0.4	Nitrogen use efficiency 0.6	50% of N harvested for biofuels production replaces crops that need N fertilizer
Rapeseed (canola)	Bio-diesel	1.0–1.7	0.7–1.2	0.5–0.9
Maize (corn)	Bio-ethanol	0.9–1.5	0.6–1.0	0.4–0.7
Sugar cane	Bio-ethanol	0.5–0.9	0.4–0.6	0.3–0.4

Source: Derived from Crutzen et al (2008).
Note: Climate warming occurs if values exceed 1.0. Current nitrogen-use efficiency is around 0.4.

bio-energy industry, and the much anticipated technological breakthrough that, within a few years, may enable ethanol production on a commercial scale from cellulosic plant sources (e.g., corn stalks, perennial native grasses, and short-rotation woody crops, such as willow and poplar).

Many issues still need to be addressed in the context of an expanding bio-economy, including macroeconomic and regional economic impacts on employment, expenditures and taxes, impacts on communities that become centers of production in the emerging bio-economy, and environmental impacts. In terms of the environment, the best hope is that new developments in bio-energy production, such as biofuels from cellulosic plant materials, will reduce the carbon footprint. But there is no guarantee that energy crops will actually reduce greenhouse gas emissions. Indeed, Crutzen et al (2008) find that planting crops, such as corn and canola, might actually enhance rather than mitigate global warming (Table 1). Additional negative environmental impacts have also been examined, most notably the extent to which previously uncultivated land will be converted to bio-feedstock production, leading to agricultural expansion onto marginal croplands and grasslands and to deforestation (Searchinger et al 2008).

The link between bio-feedstock production and water quality has not received sufficient attention to date. In particular, little attention has been devoted to water quality impacts following (a) recent policies in North America and the European Union to expand production of alternative renewable fuels, and (b) the increased likelihood that technologies for producing biofuels from cellulosic sources will soon be available commercially. In the United States in 2007, President Bush set an objective of "increasing the supply of renewable and alternative fuels by setting a mandatory fuels standard to require 35 billion gallons of renewable and alternative fuels in 2017—nearly five times the 2012 target now in law" (The White House 2007). More recently, the U.S. Environmental Protection Agency finalized regulations for the National Renewable Fuel Standard Program (RFS2). The final rule revises the annual renewable fuel standards, with required renewable fuel volumes established for each year through 2022 for (1) cellulosic biofuel, (2) biomass-based diesel (bio-diesel), (3) advanced biofuel, and (4) total renewable fuel, with the total renewable fuel requirement set at 36 billion gallons for 2022 (U.S. Environmental Protection Agency 2010a). More broadly, President Obama clearly intends to devote funding and efforts in the direction of clean energy, desiring to "invest $150 billion over ten years in energy research and development to transition to a clean energy economy" (The White House 2009). Regardless of whether these specific objectives are attained, it is clear that an expansion of lands devoted to bio-energy feedstocks likely will continue in the near future.

During the 1990s, a limited number of studies assessed the potential environmental impacts, particularly changes in runoff and soil erosion, from growing various biomass energy crops (Perlack et al 1992; U.S. Congress, Office of Technology Assessment 1993; Graham et al 1996; Green et al 1996; McLaughlin and Walsh 1998; Crutzen et al 2008). Yet, there remains much to be learned about changes in soil erosion and other environmental impacts that could occur if substantial policy-induced alterations in patterns of land use were to occur in response to expanding production of bio-feedstocks. As Graham et al (1996, p. 475) point out: "little research has analyzed the environmental impacts of land-use changes that will occur if biomass energy systems, and associated energy crops, are adopted on a large scale."

While it is recognized that expansion of bio-feedstock production may have significant implications for water quality, many scientists agree on the need for further research to examine these impacts and the importance of accounting for them in designing policies, such as biofuel incentives. Recently, a national panel of experts concluded that "among the possible challenges to biofuel development that may not have received appropriate attention are its effects on water and related land resources" (National Research Council [hereafter NRC] 2008, p. 1). One of the conclusions of that panel was that, "as total biofuels production expands to meet national goals, the long-term sustainability of the groundwater and surface water resources used for biofuel feedstocks and production facilities will be key issues to consider. . . . From a water quality perspective, it is vitally important to pursue policies that prevent an increase in total loadings of nutrient and sediments to waters" (NRC 2008, p. 56).

Recent analyses of water quality impacts have been limited, conducted primarily as a part of the Regulatory Impact Analysis of the U.S. EPA's recently completed RFS2. For example, a case study of the Upper Mississippi River Basin estimated that the renewable fuel requirements under RFS2 would increase the annual average outflows of nitrogen (N), phosphorus (P), and sediment into the Basin by an expected average of 16 million lb, 0.8 million lb, and 16 million lb, respectively, over the years 2010, 2015, 2020, and 2022 for which estimates were developed (U.S. Environmental Protection Agency 2010b).[1] Concerns regarding negative implications of bio-energy for water quality have been addressed by only a handful of studies over the past three years (Secchi and Babcock 2007; Simpson et al 2007; Donner and Kucharik 2008; Powers et al 2008; U.S. GAO 2009).

The water quality impacts of expanding or changing bio-feedstock production patterns are a function of several factors, including primarily the nature of the shift from an existing or baseline land use to a proposed bio-feedstock, such as a forage crop to corn, wheat to short-rotation woody plants, or corn to a perennial grass. The impact of converting land to the production of a bio-feedstock will clearly depend on the land's previous use. If the baseline land use was corn production, conversion to a perennial grass (or woody crop) that is harvested for energy will likely lower inputs of both fertilizers and pesticides. This is because fertilizer and pesticide inputs for corn are an order of magnitude higher than for low-input high diversity (LIHD) mixtures of perennial grasses (Tilman et al 2006). Converting land from corn cultivation to LIHD grasses likely reduces runoff and infiltration of pollutants, such as P, N, and pesticides. On the other hand, if forestry or pasture is the baseline land use then conversion to a dedicated energy crop, such as LIHD grasses, will likely increase runoff of pollutants and infiltration of pollutants into surface and ground water.

In addition to the baseline land use and the bio-feedstock to be cultivated, baseline management practices are important determinants of water quality. Tillage practices, the quantities and timing of input applications, the use of best management practices (BMPs) for soil and water conservation, and other factors affect water quality. Further, there is an array of site-specific characteristics that influence the changes in water quality brought about by changes in land use. These include slope, soil type, precipitation, characteristics of the subsurface and underlying aquifers, and so on.

We are not aware of studies that have addressed the economically optimal balance between bio-feedstock cropping and alternative existing land uses while accounting for potential impacts on water quality. The objectives of the current study are to (1) specify an optimal control framework for determining the desired amount of land within a particular geographic area to devote to the production of a bio-feedstock, (2) derive the comparative static results for the model, and (3) present an illustrative numerical application of the model to one context of current interest, specifically society's trade-off between increased bio-feedstock production and increased agricultural water pollution from converting Conservation Reserve Program (CRP) land to the cultivation of bio-feedstocks. The model that we use is a standard one-control, one-state infinite horizon optimal control model in continuous time. This type of model has been employed in the literature numerous times in a number of topic areas in applied microeconomics and macroeconomics. However, our application of the approach to the trade-off between bio-feedstock production and water quality is new and represents our primary contribution. The application is important because public policy needs to address this trade-off, and there is a dearth of studies regarding its economic aspects.

Our focus is on developing an approach flexible enough to treat several bio-energy feedstock sources that may be employed now or in the future. We take into account differences in per-acre net revenues that may be expected between the baseline land use and the prospective bio-feedstock, as well as the economic value of the flow of damages that may be anticipated from a change in water quality due to bio-feedstock production. The state variable is the stock of agricultural pollution, while the intensity of bio-feedstock production serves as a control variable.

The optimization approach can be applied to any crops that impact water quality, but here we focus only on bio-energy crops because of our interest in the impact of public policies designed to reduce greenhouse gas emissions and increase energy independence. As a starting point, we take the current crop regime as a baseline land use with corresponding water quality impacts. We then consider the impact of expanding bio-energy cropping as a movement away from the baseline.

In the next section, we describe the optimal control model, and derive the steady-state conditions and comparative static results. We then develop numerical solutions to the model (with sensitivity analyses) to obtain sharper insights into the trade-offs and pitfalls facing decision makers. Because there is a concern that CRP lands may revert back to crop production if the potential monetary payoffs from growing crops for biofuels increase, we also solve the model numerically to determine the proportion of CRP land that might optimally be converted to energy crops. The model can be used to analyze other scenarios as well.

MODEL

Consider a situation where there is an interest in expanding the amount of land devoted to the cultivation of a bio-feedstock (a term we use interchangeably with energy crop and bio-energy feedstock). Bio-energy feedstocks include but are not limited to corn for ethanol, soybeans for bio-diesel, a perennial grass, such as switchgrass, a nonnative invasive and unwanted weed (yellow starthistle or perennial pepperweed), or short-rotation trees, such as willow or hybrid poplar. While the expansion of land devoted to the cultivation of energy crops may have a broad suite of impacts (e.g., on food production, wildlife habitat, soil erosion), the present model examines only impacts on water quality. In addition, energy crops, such as corn and soybeans (first generation feedstock crops), are traditionally used as food crops, whereas perennial grasses or woody biomass (second generation feedstock) are not food crops. Despite this distinction, our model does not deal with trade-offs between energy crops and food crops, but, rather, focuses on land use and the difference in water quality impacts between a land use baseline and the conversion of that land to energy crops.

Suppose the benefit to society from employing land to produce energy crops is given by the following relation:

$$U = R(E) - D(S) \tag{1}$$

where R refers to the net revenue from the production of a dedicated energy crop ($R_E > 0$, $R_{EE} < 0$); E is the intensity of energy crop production, with $0 \le E \le \bar{E}$ and $\bar{E}$ the maximum possible intensity of energy crop production; and D refers to the damages to human health and/or the environment as a result of the stock of agricultural pollutants (S) in water, with $D_S > 0$ and $D_{SS} > 0$. Units of the control variable E are left unspecified, but could be area, proportion of land, or could reflect not only spatial aspects but also the intensity of the bio-feedstock production process (e.g., relative units of biomass in energy cropping in terms of plant density, plant cover, and yield per acre).

As the model is intended to be general rather than specific, there is no need to distinguish between pollutants in surface waters and groundwater. Thus, S may comprise a number of different agricultural pollutants, such as sediment, P, N, and/or pesticides. However, since agricultural pollutants persist for a longer time in groundwater than in surface waters, it is intuitive to think of the pollutant stock S as discharge via infiltration to groundwater. We do not consider multiple farms or spatial aspects that might enable us to address nonpoint sources of pollution, but, rather, employ a stylized model applied to a rather straightforward optimal control framework with the objective of assessing the dynamic equilibrium between lands devoted to an energy crop versus other uses.

Two determinants affect the rate of change of the pollutant stock S. First, the implementation of an energy cropping system will generally change the rate at which pollutants are added to the natural water system. If planting and cultivating the energy crop results in more pollutant runoff and leaching than the preexisting (baseline) vegetative regime, initiation of energy cropping augments the stock of pollutants S. This occurs, for example, if CRP land consisting of perennial grasses and shrubs is converted to growing corn for ethanol production. Alternatively, if cultivation of the energy crop results in less pollution than would occur under the baseline land use, the rate of augmentation of S would decline, which could occur if land currently in corn were converted to switchgrass. The impact of increased cultivation of the dedicated energy crop on water quality would depend on a variety of factors, including the energy crop chosen for cultivation, the preexisting (baseline) land use, a number of site-specific variables (e.g., slope, soil characteristics, precipitation patterns), and crop management variables (tillage practices, nutrient and herbicide applications, etc.).

Second, pollutants undergo processes of natural decomposition into nontoxic substances, with the rate of decomposition depending on the pollutant and certain site-specific factors. The rate of change of the pollutant stock S that takes these two factors into account can be written as

$$\dot{S} = \delta E(t) - \alpha S(t) \tag{2}$$

where δ is a parameter linking a change in energy cropping intensity with the change in the pollutant stock; δ is unbounded and may be positive or negative as discussed above. Parameter $\alpha > 0$ is the mean rate of natural decomposition of pollution in the environment.

The mean rate of decomposition of the pollutant stock α will depend on factors that include the characteristics of the pollutants (e.g., types of fertilizers or pesticides applied to the energy crop), soil properties, and the physical, chemical, and hydrological characteristics of the subsurface and groundwater aquifers. The process of decomposition is assumed to be governed by the following relation:

$$x_t = x_0 e^{-\alpha t} \tag{3}$$

where x_0 is the value of x at time $t = 0$.

Next, consider the objective of maximizing social well-being (Equation (1)) in perpetuity. The problem can be formulated as an infinite horizon, constrained optimization problem. The model we employ is analogous to ones used to examine the choice between producing consumer goods and maintaining environmental quality (e.g., Keeler et al 1971; Plourde 1972; Forster 1977). However, rather than applying the model in a generalized fashion to the production-environmental quality trade-off, we apply it specifically to the tension that exists between increasing bio-feedstock production and minimizing the negative impact of that activity on water quality. Our model also is distinctive in that (1) the control variable represents the intensity of bio-feedstock production, where intensity can range from zero to some maximum level in which all available technology for bio-feedstock production is applied to the whole of a given area of land, and (2) an increase in bio-feedstock production intensity may either increase or decrease the rate of change of the pollution stock. The optimization problem is

$$\text{Maximize} \int_o^{\infty} [R(E(t)) - D(S(t))]e^{-rt} dt \tag{4}$$

subject to

$$\dot{S} = \delta E(t) - \alpha S(t) \tag{5}$$

$$S(t_0) = S_0, \lambda_1(t_1) = 0 \tag{6}$$

$$0 \leq E \leq \bar{E} \tag{7}$$

where r is the (social) rate of discount, λ_1 is the current value costate variable associated with the current-value Hamiltonian resulting from solving the optimization problem (Equations (4)–(7)), t_1 is the free terminal point (endogenously determined ending time at which λ_1 goes to zero), and E and S are as previously defined. The necessary conditions for a solution are

$$R_E + \lambda_1 \delta \leq 0; \quad E \geq 0; \quad (R_E + \lambda_1 \delta)E = 0 \tag{8}$$

$$D_S = \dot{\lambda} - \lambda_1(r + \alpha) \tag{9}$$

$$\dot{S} = \delta E(t) - \alpha S(t) \tag{10}$$

$$S(t_0) = S_0; \quad \lambda_1(t_1) = 0 \tag{11}$$

$$(\bar{E} - E) \geq 0; \quad \lambda_2 \geq 0; \quad \lambda_2(\bar{E} - E) = 0 \tag{12}$$

where λ_2 is the multiplier associated with constraint (Equation (7)).

Kuhn-Tucker condition in Equation (8) indicates that, if an interior solution, control variable E should be chosen so that the marginal net revenue of energy cropping equals the shadow cost of adding to the pollutant stock by increasing energy cropping intensity by one unit. Equation (9) provides the optimal evolution of the pollutant stock shadow price over time; it is a function of the marginal damage caused by the pollutant, the rate of decomposition of pollutants in the environment, and the discount rate. Equations (10), (11), and (12) are the respective state equation, endpoint conditions, and Kuhn-Tucker conditions associated with the upper bound on E.

To derive the steady-state condition for E, differentiate Equation (8) with respect to time and substitute the result along with Equation (8) into Equation (9):

$$\dot{E} = \frac{R_E(r + \alpha) - \delta\, D_S}{R_{EE}} \tag{13}$$

Setting $\dot{E} = 0$ gives the steady-state condition for E:[2]

$$R_E(r + \alpha) = \delta D_S \tag{14}$$

which, upon totally differentiating, gives

$$dE/dS|_{\dot{E}=0} = \frac{\delta D_{SS}}{R_{EE}(r + \alpha)} \tag{15}$$

This gives the slope of the $\dot{E} = 0$ locus, which is unambiguously negative for $\delta > 0$ because $R_{EE} < 0$ while $D_{SS} > 0$.

Similarly, setting $\dot{S} = 0$ in Equation (10) yields

$$\delta E = \alpha S \tag{16}$$

which, upon totally differentiating, gives

$$dE/dS|_{\dot{S}=0} = \frac{\alpha}{\delta} \tag{17}$$

Equation (17) provides the slope of the $\dot{S} = 0$ locus, which is positive for $\delta > 0$.

Note that Equation (14) can be rewritten as $R_E = \frac{\delta D_S}{(r+\alpha)}$, which states that the conversion of land from its current use to energy crops continues until the marginal net revenue equals the marginal damage costs, adjusted for the effect of energy crops on the pollutant stock, the rate at which pollutants decompose, and the fact that today's energy crop affects the pollutant stock in the next period. The steady state is where the stock S shifts the marginal net revenue curve to equal the bio-physical equilibrium in Equation (16). This is illustrated in Figure 1, which gives the phase diagram for the dynamical system when $\delta > 0$. The dynamics of the system are shown and yield the expected saddle point equilibrium.

COMPARATIVE STATICS OF THE STEADY STATE

The optimal steady-state levels of E and S, which are denoted E^∞ and S^∞, vary across different potential bio-feedstock production landscapes. An array of site-specific characteristics determines the rate of natural decomposition (α) of pollution from cultivating the bio-feedstock. Further, the parameter δ, which gives the change in the stock of pollution that results from a marginal change in the intensity of bio-feedstock production, varies spatially and is a function of the bio-feedstock harvested, hydrological characteristics (e.g., aquifer recharge properties), and other factors discussed previously. The optimal solution is also affected by the discount rate.

Consider how changes in these key parameters (r, δ, and α) influence the steady-state solutions. This may be determined analytically through comparative statics analysis, providing the signs (directions) of the effects of changes in each of the parameters on the optimal values of E and S. Comparative statics also indicates, which effects are ambiguous in sign, a finding that is not always revealed by a single numerical application. The

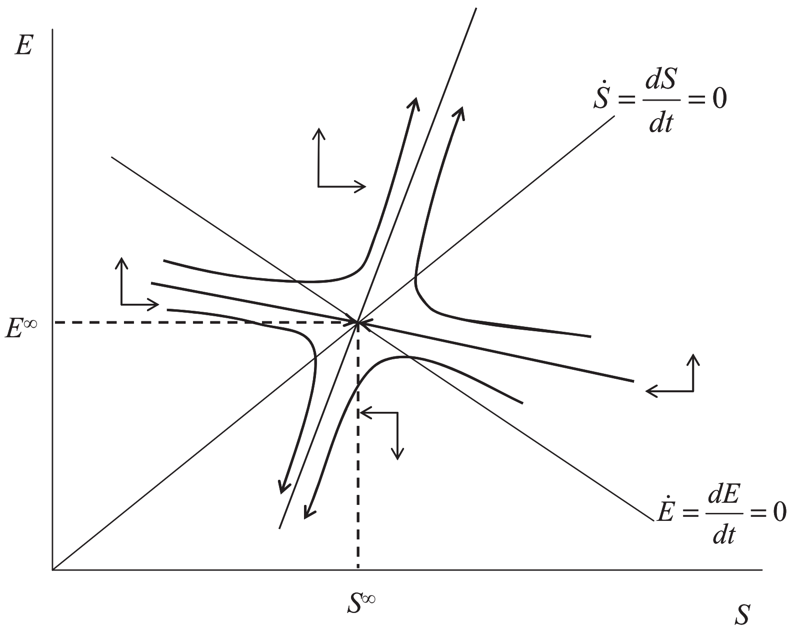

Figure 1. Saddle point equilibrium, $\delta > 0$

magnitudes of impacts due to changes in parameters are examined in the numerical application below, and the findings of any application depend on the values of the parameters employed.

To derive the comparative statics results, we first totally differentiate Equations (14) and (16) with respect to E, S, r, δ, and α, which yields after rearranging terms:

$$\begin{bmatrix} R_{EE}(r+\alpha) & -\delta D_{SS} \\ \delta & -\alpha \end{bmatrix} \times \begin{bmatrix} dE \\ dS \end{bmatrix} = \begin{bmatrix} D_S\, d\delta - R_E(dr + d\alpha) \\ S\, d\alpha - E\, d\delta \end{bmatrix} \tag{18}$$

Upon rearranging Equation (18) by inverting the matrix, we obtain:

$$\begin{bmatrix} dE \\ dS \end{bmatrix} = \begin{bmatrix} \frac{-\alpha}{|A|} & \frac{\delta D_{SS}}{|A|} \\ \frac{-\delta}{|A|} & \frac{R_{EE}(r+\alpha)}{|A|} \end{bmatrix} \times \begin{bmatrix} D_S\, d\delta - R_E(dr + d\alpha) \\ S\, d\alpha - E\, d\delta \end{bmatrix}$$

$$= \frac{1}{|A|} \begin{bmatrix} \delta D_{SS}(S\, d\alpha - E\, d\delta) - \alpha D_S\, d\delta + \alpha R_E(dr + d\alpha) \\ R_{EE}(r+\alpha)(S\, d\alpha - E\, d\delta) - \delta D_S\, d\delta + \delta R_E(dr + d\alpha) \end{bmatrix} \tag{19}$$

where $|A| = -\alpha R_{EE}\,(r+\alpha) + \delta^2 D_{SS}$, which is unambiguously positive as $R_{EE} < 0$ while $D_{SS} > 0$.

Table 2. Comparative static results and predicted signs

Comparative static effect	$\delta > 0$	$\delta < 0$
$\frac{\partial E^{\infty}}{\partial r} = \frac{\alpha R_E}{\lvert A \rvert}$	+	+
$\frac{\partial S^{\infty}}{\partial r} = \frac{\delta R_E}{\lvert A \rvert}$	+	−
$\frac{\partial E^{\infty}}{\partial \alpha} = \frac{\alpha R_E + \delta S D_{SS}}{\lvert A \rvert}$	+	?
$\frac{\partial S^{\infty}}{\partial \alpha} = \frac{\delta R_E + R_{EE} S(r+\alpha)}{\lvert A \rvert}$	?	−
$\frac{\partial E^{\infty}}{\partial \delta} = \frac{D_S - \delta E D_{SS}}{\lvert A \rvert}$	?	+
$\frac{\partial S^{\infty}}{\partial \delta} = -\frac{\delta D_S + E R_{EE}(r+\alpha)}{\lvert A \rvert}$	?	+

Note: Given $R_E > 0$, $R_{EE} < 0$, $D_S > 0$, $D_{SS} > 0$, $\alpha > 0$.

Equation (19) give the effects of changes in the parameters on the long-run optimal levels of the control and state variables (E^{∞} and S^{∞}) (Table 2). The signs on the partial derivatives reported in Table 2 depend on whether δ is positive (an increase in E causes an increase in $\dot{S}$) or negative (increased E decreases $\dot{S}$). A positive value for δ is expected if expanded production of the bio-feedstock results in increased runoff and infiltration of nutrients or pesticides. This will occur, for example, if baseline CRP grassland is converted to corn production for ethanol. On the other hand, δ is expected to be negative if land used for row crops in the baseline was converted to the production of LIHD mixtures of perennial grasses, which is less likely given that corn has more desirable properties for producing ethanol.

The first two rows in Table 2 indicate the impacts of a change in the social rate of discount. The signs of $\partial E^{\infty}/\partial r$ and $\partial S^{\infty}/\partial r$ are unambiguously positive if $\delta > 0$, but if $\delta < 0$ the sign of $\partial E^{\infty}/\partial r$ is positive but that of $\partial S^{\infty}/\partial r$ is negative. As the discount rate increases, future damages from the stock of pollution are discounted more heavily and thus are smaller relative to the rise in current net revenues from an increase in the intensity of bio-feedstock production. Therefore, E^{∞} will increase and, in turn, lead to a higher level of S^{∞} if $\delta > 0$, but to a lower level of the pollutant stock if $\delta < 0$.

An increase in the decomposition rate of pollutants would increase the intensity of energy production if $\delta > 0$, as $\partial E^{\infty}/\partial \alpha$ is unambiguously positive, but may or may not increase energy production if $\delta < 0$ as the sign of $\partial E^{\infty}/\partial \alpha$ is ambiguous. The sign of $\partial S^{\infty}/\partial \alpha$ is ambiguous if $\delta > 0$ and negative if $\delta < 0$. It is ambiguous in the first case because, although an increase in α leads to a higher optimal level of energy crop production and thereby more pollution, the pollutants will degrade more rapidly in the environment. Thus, the net effect on S^{∞} depends on the values of the parameters. However, it is negative when $\delta < 0$ because less pollution is produced as land is converted from the baseline use to bio-feedstock production.

Finally, changes in the long-run steady-state equilibrium values E^{∞} and S^{∞} that occur in response to a change in parameter δ are provided in the final rows of Table 2. Both $\partial E^{\infty}/\partial \delta$ and $\partial S^{\infty}/\partial \delta$ are ambiguous in sign when $\delta > 0$, depending on the actual values of the parameters, namely, $\partial E^{\infty}/\partial \delta > 0$ if $D_S > \delta E D_{SS}$ and $\partial S^{\infty}/\partial \delta > 0$ if $\lvert E R_{EE}(r + \alpha) \rvert > \delta D_S$. If $\delta < 0$, $\partial E^{\infty}/\partial \delta$ is positive because the reduction in pollution caused by the conversion from the baseline land use to the energy crop is less per unit of energy crop production (the new land use is better but not quite as good since $|\delta|$ is now lower); thus, greater bio-feedstock production is needed to make up for the reduction in the environmental benefit that the energy crop provides. Further, if $\delta < 0$, $\partial S^{\infty}/\partial \delta$ is positive because a rise in the value of δ now results in more of the pollution going into the pollutant stock.

Actual numerical solutions to the model provide greater insights into the trade-offs and options facing the policy maker. Therefore, we now consider a numerical application.

APPLICATION

There are many different contexts in which the model can be applied. To illustrate one possibility, consider a situation in which a policy maker wants to know what proportion of existing CRP land might be converted to the cultivation of a row crop, specifically corn for ethanol. We examine this choice because there is a concern that CRP lands may be converted to energy crops if the potential monetary payoffs from growing crops for biofuels increase: "Staying the current policy path would likely result in the continued trend of expansion of corn-based ethanol production. . . . If projected future increases in use of corn for ethanol production do occur, the increase in harm to water quality could be considerable. In addition, expansion of corn production on fragile soils or soils that do not hold nutrients can increase both loads of nutrients and sediments" (NRC 2008, p. 57). More specifically related to CRP lands, the same report predicts that "one of the most likely causes of increased erosion in the near term may be the withdrawal of lands from the. . . Conservation Reserve Program. . . as well as expansion of biomass production on non-CRP marginal land. . ." (p. 30), a concern also raised by the U.S. GAO (2009). In one study, Secchi and Babcock (2007) estimated that more than 526,000 ha of CRP land in Iowa would be converted to a rotation of corn and soybeans if corn prices rose to \$5/bushel, a price that was exceeded in 2008.

Our application examines the trade-off between increased net agricultural revenue and greater runoff expected from conversion of CRP lands to energy crop production. We focus on the expected increase in P runoff. Consider an area of 10,000 acres (4,050 ha) that is currently in the CRP. We solve for E^{∞}, or the proportion of the land (given in terms of the number of acres out of 10,000) that optimally should be converted from CRP to the production of corn as an energy crop. We assume that the productivity of the land for growing the bio-feedstock varies spatially across the study area, as described below.

Consider parameter δ. To estimate increases in P losses caused by converting the land to the bio-feedstock, we use data from Green et al (1996), who assessed the environmental impacts of converting cropland to biomass production. They estimate average total runoff for various plant cover types, including corn and trees with a cover crop (trees&cover). We use runoff and sediment loss measures for the trees with a cover crop land type as a proxy for runoff from CRP lands. Comparing runoff and soil loss estimates for these two plant cover types yields a proxy for the percent increase in P loss that would be expected as a result of converting the land to corn for energy. For example, calculations based on the data in Green et al (1996) indicate that the increase in total runoff resulting from conversion of land from trees&cover to corn would be approximately 45%. Similarly, Green et al (1996) calculate that land conversion from trees&cover to corn would increase sediment loss by about 165%.

With regard to P loss due to soil loss (with P bound to soil particles) and loss resulting from surface runoff (with P dissolved in water), we rely on Perlack et al (1992), who use data on annualized P emissions by location and crop type for a set of five study locations. For a baseline, we employ information on P losses from tree crops at Peoria, Illinois: the loss in surface water amounts to 0.54 lb P/acre/year, while loss due to soil erosion is 1.09 lb P/acre/year. These estimates are combined with the percent increases in runoff and sediment loss from Green et al (1996), yielding an estimate of an increase in P loss via soil erosion and surface water of 2.05 lb P/acre/year as a result of converting land from trees&cover to corn. This constitutes the baseline value of the δ parameter.

Now consider parameter α. Reitzel et al (2007) measure the degradation rates of organic P in lake sediment, and provide half-lives of various P compounds in supernatant and precipitate. Half-lives vary across P compounds and range from 1 to 88 years. For our analysis, we use 40 years as a baseline value for the half-life and then conduct sensitivity analyses around this value. For the pollutant decomposition function (Equation (3)), a half-life of 40 years corresponds to $\alpha = 0.017$.

To determine the marginal damage D_s from P in the steady-state Equation (14), we employ data from Sano et al (2005) who estimated that the cost of P removal at Lake Okeechobee ranged from \$11 to \$35/lb. Sensitivity analysis is useful to decision makers because the marginal damages from P pollution range widely across different sites, depending on the characteristics governing the fate and transport of P, the receptors (human health, wildlife, etc.), and the service flows eventually affected by the pollution (fishing, swimming, boating, drinking water, etc.). Since removal costs also vary across sites, we use the approximate midpoint (\$23/lb) as a baseline value. Finally, since P removal costs occur at a particular point in time, whereas damages are a flow over time resulting from the stock of pollutants, we calculate the annual damages that would yield a discounted cumulative flow of some \$23/lb. This value is about \$1/lb/year, which we use as our baseline value for D_s. Note that we use the cost of nutrient removal as a proxy for nutrient damages, which may underestimate or overestimate the true damages caused by P. While recognizing the drawback to this approach, we do so because of the paucity of data on the true damages from a given stock of P in the environment. Again, this true value is expected to be site specific and depends on numerous factors related to the site's environmental characteristics, fate and transport parameters, the impacted receptors, and levels of exposure.

The marginal net revenue from energy cropping (R_E) is also an important factor in determining the optimal solution. Consider a geographic area with spatial heterogeneity so that R_E varies across the area in question. This will be the case for areas that are diverse with respect to soil type, topography, and so on. Assume then that $R_{EE} < 0$ with R_E given as follows:

$$R_E = k - (E - {}^{1}\!/_{2}\bar{E})/b \tag{20}$$

where k is the midpoint of the range of marginal net revenue (\$/acre) from production of the bio-feedstock on the land area, $\bar{E}$ is the maximum acreage that may be devoted to the bio-feedstock, and $b = \bar{E}/2k$. If the baseline yield for corn is 160 bushels/acre and net returns per bushel are \$0.50, then net revenue is \$80/acre. These figures obviously vary from place to place, but break-even revenues for corn have ranged between \$1.40 and \$2.00/bushel from 2003 to 2007 in northern Illinois, with the cost of producing corn running from \$550 to over \$600/acre (Fyksen 2007). Assuming that R_E actually varies, over the 10,000-acre area in question, from \$160/acre on the most productive land to \$0/acre on the least productive acre, the following are the base case parameter values for Equation (20): $k = \$80$/acre, $\bar{E} = 10{,}000$ acres, and $b = 62.5$. Finally, we set the base case social discount rate at $r = 0.04$.

The solution to the optimal control problem is derived by solving the following two equations (where $\dot{E} = 0$ and $\dot{S} = 0$) simultaneously:

$$[k - (E - \tfrac{1}{2}\bar{E})/b](r + \alpha) = \delta D_S$$

$$\delta E = \alpha S$$

RESULTS AND DISCUSSION

Our steady-state values of the energy cropping intensity E^{∞} (number of acres out of 10,000 to be put into corn) and the stock of pollutant S^{∞} (P) are provided in Tables 3 and 4 for various values of the respective parameters α (the mean rate of natural decomposition of the pollution in the environment) and δ (change in the stock of pollution resulting from a marginal change in the intensity of bio-feedstock production). For the base case value $\alpha = 0.017$, the optimal level of energy cropping is $E^{\infty} = 7{,}766$ acres—about 78% of the area would optimally be converted to the production of corn for ethanol. As α varies from 0.009 (half-life of 80 years) to 0.069 (half-life of 10 years), E^{∞} rises from 74% to 88% of the land unit (Table 3). This conforms to our unambiguous comparative static result in Table 2 (for $\delta > 0$). The numerical results also show that, as α increases, the optimal pollution stock S^{∞} decreases even though E^{∞} (and thus the rate of production of agricultural pollution) rises. This is the case for the parameter values employed in our example, rather than as a general rule, since the result in Table 2 indicates that this relationship may be either positive or negative for $\delta > 0$.

From Table 4, an increase in the marginal effect of bio-feedstock production intensity on the pollution stock (δ) reduces the optimal intensity of bio-feedstock production, from

Table 3. Sensitivity of optimal energy cropping level (E^{∞}) and pollutant stock (S^{∞}) to changes in the rate of pollutant degradation α[a]

α (1/year)	Half-life of pollutant (years)	E^{∞}(acres out of 10,000)	S^{∞} (lb)
0.069	10	8,825	262,068
0.035	20	8,293	485,467
0.017	40	7,766	918,297
0.012	60	7,516	1,333,316
0.009	80	7,368	1,743,358

Notes: [a]Base case value for $\alpha = 0.017$. Base case values for other model parameters are $\delta = 2.05$ lb/acre/year; $D_s = \$1$/lb/year; mean $R_E = \$80$/acre; $r = 0.04$.

Table 4. Sensitivity of optimal energy cropping level (E^{∞}) and pollutant stock (S^{∞}) to changes in δ (linkage parameter between E and dS/dt)[a]

δ (lb/acre/year)	E^{∞} (acres)	S^{∞} (lb)
0.20	9,782	112,899
2.05	7,766	918,297
5.00	4,549	1,312,556
10	0	0

Notes: [a]Base case value for $\delta = 2.05$. Base case values for other model parameters are $\alpha = 0.017$/year; $D_s = \$1$/lb/year; mean $R_E = \$80$/acre; $r = 0.04$.

some 98% for $\delta = 0.20$ to 45% for $\delta = 5$, and zero for very high levels of δ. For our base case ($\delta = 2.05$), some 78% of land would be used to produce corn for ethanol. Clearly, pollution is a negative side effect of bio-feedstock production that society wants to avoid; as the optimal stock of P increases as a result of a rise in δ, less land should be converted from trees to corn.

Society is caught in a dilemma when it comes to producing energy crops. While it is important to rely more on biofuels as a means of mitigating climate change and reducing reliance on foreign suppliers of petroleum, there are other costs. A cost that is most frequently discussed concerns the diversion of forestlands and croplands into energy production. This might be a positive undertaking if the baseline land use is less environmentally friendly, although this is unlikely to occur because the underlying reasons for cropping in an environmentally damaging way are more likely to be exacerbated under policies to encourage production of bio-feedstock.

Our findings indicate that bio-energy policies will indeed cause land to be shifted out of the CRP, and that, given these policies, this would be socially optimal. However, there is a limit to how much land can optimally be converted to producing bio-feedstock. Society is forced to make a trade-off between its desire to pursue mitigation of climate change and its willingness to accept a decline in water quality. The trade-off depends on technical factors as well as human preferences.

Several topics are important for future research. These include the impact of increased bio-feedstock production on water quantities, the spatial aspects of bio-feedstock production and its resulting environmental impacts, and the development of more and better scientific data on the parameters in our model. On the first point, experts currently recognize that, as with the link between bio-feedstock production and water quality, the implications for water availability are not well understood (NRC 2008). Conversion of land to bio-feedstocks, even grassland to switchgrass or willow, may increase water requirements. This, in turn, may have implications for competing water uses, such as irrigated agriculture (food crops and forage), municipal uses, recreational and ecological values from in-stream flows, and even water quality as less pollution can be absorbed. These potential implications have not yet been fully addressed by agronomists and economists.

The second and third research areas are interrelated. Future analyses that apply a spatial modeling framework to examine bio-feedstock cropping implications for water quality would be useful. However, this creates a tremendous challenge due to the nonpoint source nature of agricultural pollution. Similarly, bio-economic modeling efforts are constrained by limited scientific data on parameters describing pollutant emissions from bio-feedstock production, pollutant fate and transport, and the impacts of receptor exposures to pollution. Since our model indicates sensitivity to such parameters, better data in the future would allow for more accurate estimates of optimal policies.

It is also useful to mention policy incentives that potentially could allow for a more efficient expansion of bio-energy cropping and account for the water quality impacts. First, it is possible to implement variable subsidies for biofuels. This could be done by lowering biofuel subsidies during times when biofuel is relatively profitable on its own, and then using the savings to mitigate negative water quality impacts from bio-feedstock production (NRC 2008). Alternatively, the magnitude of the per unit biofuel subsidy could be tied spatially to the expected scale of negative water quality impacts. For example, a multitiered subsidy scale could address factors, such as baseline land use, expected chemical inputs to produce the bio-energy crop, and potential soil erosion or runoff.

Second, incentives could be developed to encourage the transition from existing feedstocks (e.g., corn ethanol) to next-generation sources, such as cellulosic feedstocks. For example, biofuel subsidies could be higher for cellulosic sources that have lower expected negative impacts on water resources.

Third, there is a broad suite of existing and potential policies and programs that provide incentives for agricultural BMPs to reduce water quality impacts.[3] Existing programs include the CRP, the Environmental Quality Incentives Program, and the Conservation Security Program. These programs will continue to be relevant as pressures increase to use lands for energy crops. Nonetheless, there is room to enhance existing incentives that would minimize potential negative water quality impacts of expanded energy cropping. For example, incentives could target watersheds where soil and nutrient loss potentials are greatest (NRC 2008), or cross-compliance could be used to attain further reductions in agricultural nonpoint source water pollution (U.S. General Accounting Office 2003). Cross-compliance could result in a producer forfeiting support payments if water quality BMPs were not implemented on lands with high potential for soil erosion or nutrient runoff. Interestingly, the possibility of extending approaches like Total Maximum Daily Load regulation from point sources to agricultural nonpoint sources has been raised as well in the context of biofuels (NRC 2008). Again, there are many challenges to implementing such approaches in a nonpoint context. Further advances in nonpoint monitoring and modeling are expected to help address such challenges.

CONCLUSIONS

In the current research, we developed an optimal control framework for determining the socially desirable amount of land within a given area to use for the production of a bio-feedstock, derived comparative static results, and presented a numerical application of the model to an issue of current interest. Our application involved the trade-off between increasing bio-feedstock production and minimizing P pollution in water from converting CRP land to corn. Using base case parameter values, we found that about 78% of a given land area would optimally be converted to the production of the bio-feedstock. Sensitivity analysis with respect to changes in the assumed half-life of P in water yielded optimal bio-feedstock cropping levels that ranged from 74% to 88% of the given land area. Results were even more sensitive to changes in the relation between bio-feedstock cropping and the rate of change of the pollutant stock in the environment, with optimal cropping proportions ranging from 0% to 98% of the given land area. Given current uncertainty regarding the true parameter values as well as natural variation in those values across sites, our findings indicate the need for more and better research on the site-specific values of key bio-physical parameters.

NOTES

[1]These are U.S. EPA's estimated increases relative to those estimated for a pre-RFS2 baseline projection of renewable fuel use of 13.56 billion gallons per year by 2022, as developed by the U.S. Energy Information Administration's 2007 Annual Energy Outlook (U.S. Department of Energy 2007).

[2]Equation (14) could be obtained in a more straightforward fashion, without the need to differentiate Equation (8), by setting $\dot{\lambda}_1 = 0$ in Equation (9), solving for λ_1 and substituting the result in Equation (8).

[3]Examples of agricultural practices to reduce the water quality and quantity impacts of bio-energy feedstocks are summarized elsewhere (U.S. GAO 2009, pp. 42–43).

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