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Information Problems in the Design of

Nonpoint Source Pollution Policy*

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Information Problems in the Design of

Nonpoint Source Pollution Policy

I. *Introduction*

Large industrial and municipal emissions were the focus of first-generation environmental policies of the 1970's. Twenty years later, with much success in cleaning up industrial and municipal sources, the focus has changed. The problems of the moment include hazardous wastes, solid wastes, auto pollution, nutrient pollution, pesticide pollution, and sedimentation. These problems, by and large, are caused by many small polluters--such as users of weed sprays, motorists, farmers, and generators of household trash.

A common denominator of the contemporary problems is a high degree of difficulty in keeping track of individual pollution sources. There are so many sources that monitoring all of them would be prohibitively expensive. Furthermore, in many cases, the pollution is not a distinctive discharge, but rather is a diffuse side effect of complex activities (e.g., farming operations). This also hampers monitoring at the source. In addition, it is generally difficult to infer the pollution originating from any individual source from observations of ambient pollution levels, since ambient levels are determined by the combined activities of many polluters as well as random factors over which polluters have no control. The second generation pollutants also have complex environmental fates. For example, dissolved fertilizers break down into several chemical forms as they move into surface and ground waters. Dislodged soil particles move around in space creating flooding, degrading fish habitat, and increasing water treatment costs. With the potential to affect several media, these pollutants may be the focus of multiple policy objectives. The appropriate policy tactics may change over space and time.

We have, then, a set of complicated environmental problems that do not fit neatly into the first generation mold. Rather, the second generation problems involve polluters who are difficult to identify, emissions that are virtually impossible to monitor, and environmental fates that are multifaceted and uncertain. Information problems are at the root-of all of these difficulties.

These information problems greatly complicate the selection and implementation of policies to control second generation pollutants. The common and most direct prescription for controlling pollution, namely taxing or regulating emissions, is not a viable option for controlling these second generation pollutants. Instead, indirect policies applied to something other than emissions must be used. Examples include up-line policies (such as taxes or regulation) applied to input use and down-line policies (taxes or liability) based on ambient pollution levels. The question is then whether these indirect instruments can serve as perfect substitutes for direct control of emissions.

In general, these indirect policies are likely to be imperfect substitutes for direct emissions control. Some of the same information problems that prevent the use of direct emission control policies also imply imperfections in using indirect policies. For example, pollution-related inputs that are not easily monitored are not amenable to taxation or regulation. Likewise, the use of ambient taxes or liability can be hampered by possible information problems such as identifying the actual or probable contribution of individual sources.

If no single direct or indirect policy instrument can ensure efficient abatement of these second generation pollutants, then what recourse do environmental economists and policy makers have? Clearly, one approach is simply to live with the imperfections and analyze individual policies in a second-best context. While this seems to be a common approach of economists,[1] policy makers seem to have chosen an alternative approach. Rather than searching for the “best” (in a second-best sense) single instrument, policy makers appear instead to be searching for a combination of indirect instruments to control these second generation pollutants. In a world of first-best instruments, simultaneous use of instruments is at best redundant and at worst counter-productive. However, when information problems prevent the use of first-best instruments, a theoretical rationale for combining instruments may exist. In searching for a combination of instruments, policy makers may in fact be ahead of the theory of efficient pollution control, which has focused almost exclusively on single-instrument approaches.

Examples of multiple instrument approaches are easy to find. As illustrations: A host of initiatives, from litter laws and beverage container deposits to recycling programs and mandated use of recyled paper by government agencies, are aimed at reducing the disposal of solid wastes. The prevention of pesticide contamination is the objective of complex licensing and labelling standards, food safety standards, and, potentially, products liability. And, the abatement of sedimentation is promoted through erosion control standards, government subsidies for erosion control, and, potentially, nuisance or tort law remedies.

In this paper, we consider the choice of environmental policies under incomplete information, with special reference to nonpoint source pollution (NSP) and the types of policy instruments that could be used in this context. In considering nonpoint source pollution, we will focus particularly on pollution from agricultural land uses. [2] Agricultural NSP is widespread and of current concern in many countries. It certainly is prone to many of the challenges noted above: the many, dispersed sources are difficult to identify; monitoring is nearly impossible because of the diffuse sources; several media are affected; and the occurance and impacts of agricultural NSP are nearly impossible to predict because of the importance of stochastic weather and production variables.

We begin by examining in greater detail the particular information problems of agricultural NSP and explore some of the implications for policy design. Next, we focus on the simultaneous use of multiple instruments as a means of compensating for information problems. Using a very simple model, we show that multiple instruments can be an efficient response to imperfections in single instruments due to information problems, although care must be taken that the instruments that are combined are complementary rather than contradictory.

II. Information Problems with Nonpoint Source Pollution

Figure 1 represents a general relationship between sources and ultimate impacts of pollution for two firms whose emissions combine to determine ambient environmental quality at a particular location. The figure depicts the various steps in the production process of a firm, from its initial input/technology choice to its products, emissions, and ultimate environmental or health impacts. The emissions, perhaps of several types and from multiple sources, may affect several environmental media, sometimes interconnected media (such as ground and surface water often are), and cause contamination. Exposure of susceptible humans, other life forms, or physical systems to the contamination leads to damages being incurred.

The description in Figure 1 seems general enough to fit pollution from both point sources and nonpoint sources. Of interest to us are the informational characteristics of the various steps of the pollution process and the specific information problems of agricultural nonpoint source pollution. We will discuss two classes of information problems: natural variability and problems of monitoring and measurement.

Natural Variability

As indicated symbolically in Figure 1, pollution processes are affected by various natural sources of variability, including weather, mechanical malfunctions, and susceptibility to damages.[3] As a result, a particular policy (or a specific abatement plan) will produce a distribution of outcomes rather than single outcome.

If the outcomes cannot be precisely foreseen, then abatement policies and abatement methods must be evaluated according to their effects on the expected distribution of outcomes, as determined by the distributions of the underlying random variables. This randomness does not, by itself, prevent the attainment of *ex ante* efficiency through the use of standard policies.[4] In other words, if neither the firm nor the social planner knows the values of the random variables at the time that decisions are made, and if both are risk neutral, then the policy maker can structure a tax or regulation that will cause rational private decision makers to act in a way that maximizes the expected value of social surplus. For example, the planner can place a tax on polluting inputs equal to the expected marginal external cost of using the input, thereby ensuring that expected marginal social costs equal expected marginal private costs. In such a case, randomness should not affect the selection of a policy goal, although realization of that goal at any one time will be a random event. However, randomness can affect the relative *ex post* efficiency of different policy instruments, as shown by Weitzman (1974).[5]

Another type of variability has to do with space rather than time. A state or national environmental policy must apply in a variety of local circumstances as well an enduring variation over time. As shown by Kolstad (1987), certain policy instruments may contend with diverse local circumstances more efficiently, in an *ex post* sense, than others. Like Weitzman's (1974) analysis, the relative curvatures of abatement cost and benefit functions determine whether incentive or regulatory instruments are more robust when applied acres local circumstances.

The expected mean value of emissions or ambient contamination in many cases is a sufficient *ex ante* measure of an environmental goal.[6] However, in some instances, deviations around the mean are important as well. For example, variation in pollution outcomes is often incorporated into regulatory policies by setting a threshold level of environmental quality (Q*) and a safety margin, expressed as a maximum acceptable frequency (1-P) of exceeding the threshold (Lichtenberg and Zilberman, 1988; Braden, Larson, and Herricks, 1991):

$$\text{Prob}\,(Q \geq Q^*) \leq 1 - P\ ,$$

where Q is measured environmental contamination and P is the cumulative probability of Q. Such a policy goal calls for abatement measures that will not only affect the mean realizations of abatement (keyed to the threshold), but also the variability of the pollution distribution (in reaction to the safety margin). Unless the mean realization and the variance are correlated, a single policy instrument will not generally achieve the joint goal in an efficient manner. Combining instruments that apply to specific moments of the distribution will often enhance efficiency.[7] For example, in addition to specifying maximum customary rates, emissions regulations frequently specify special rates that apply when background conditions are less able than usual to assimilate pollutants.

To summarize the preceding discussion: Even in the presence of natural variability, policy instruments can be selected to achieve *ex ante* efficiency, although the resulting level of environmental quality will deviate from the *ex post* socially efficient goal. A similar conclusion applies when a single policy must address a problem that varies from place to place. In addition, if damages are affected by higher moments of the distribution of ambient quality, then the use of several policy instruments may enhance efficiency under some circumstances. The lessons for agricultural NSP policies depend on the particular empirical properties of the abatement supply and demand curves, on the spatial variation in the problems, and on the importance of and relationships between moments of the distribution of outcomes.

Empirical research on agricultural nonpoint source pollution has benefitted from simulation models of pollution processes[8] that provide insight into the costs of abating agricultural NSP.[9] Unfortunately, there is virtually no corresponding information on benefits (abatement demand).[10] The cost studies indicate that, at least for sediment, the supply curve begins with very little slope and becomes steeper as abatement goals are raised. Illustrative abatement supply curves for sediment, taken from a study of Central Illinois conditions by Braden *et al.* (1989), are reproduced in Figure 2.[11]

With little information on abatement demand, we can only speculate about the ranking of incentive and regulatory policies. If demand and supply intersect at low levels of abatement, then the demand curve would almost certainly be steeper than the very flat supply curve and an abatement standard set to achieve the expected pollution level would probably minimize the *ex post* losses in economic surplus. At the other extreme, the steep portion of the cost function would almost certainly be steeper than an intersecting demand curve, in which case an incentive instrument would minimize the *ex post* losses.[12]

On the matter of spatial variation, at least with respect to abatement costs, the empirical literature provides more to go on. Park and Shabman (1982) analyze the value of regional "targeting" (differentiated policies) while Braden *et al.* (1989) analyze the value of micro-targeting within a watershed. Both indicate that spatially uniform policies are inefficient. The finding of significant *ex post* inefficiency underscores the merit of locally differentiated or flexible policies rather than uniform policies. However, as between uniform taxes and regulations applied to simulated erosion rates, Miltz, Braden, and Johnson (1988) suggest that it may not be possible to draw general conclusions about which is more efficient. Their results indicate that taxes achieve modest reductions in simulated sedimentation at a lower cost while regulations achieve extreme reductions at a lower cost.

Finally, higher moments of the distribution of outcomes are environmentally important for several agricultural pollutants. For example, extreme concentrations of some agricultural chemicals can be acutely toxic while average concentrations have no effect. Sediment is also illustrative--average loads are relevant to depletion of reservoir storage capacity while extreme loads play a major role in flood damages. As noted above, several instruments maybe needed to abate most efficiently the multifaceted damages.

Imperfect Monitoring and Measurement

In addition to natural variability, various aspects of pollution production processes are subject to imperfect monitoring and measurement. Many elements cannot be easily monitored. Others are likely to be monitored only occasionally, so unusual occurances may go undetected. In addition, malfunctioning or insufficiently sensitive testing equipment can provide misleading information.

With imperfect monitoring and measurement, policy enforcement will also be imperfect (Russell, Harrington, and Vaughan 1986). Violations may not be detected ("false negatives") or may be spuriously inferred ("false positives"). The social costs of these errors are of two types: 1) the damages (net of abatement costs) that would have been prevented if violators could have been induced to comply with policies, and 2) the excessive abatement costs (net of abatement benefits) resulting from unfounded enforcement actions. These potential costs must be weighed against the costs of more complete testing and more precise measurement. Vaughan and Russell (1983) illustrate the use of statistical quality control measures to devise an optimal monitoring regime.

Imperfections in monitoring and enforcement are not only potentially costly, they also create opportunities for polluters to influence the information that becomes available to enforcement officials. For example, the enforcement of many pollution control laws is based, in the first instance, on self-monitoring data reported by polluters. These reports are periodically verified through pre-announced site visits by government officials. If it wished, the polluting firm could falsify its reports and misrepresent the typical plant operations during the periodic site visits.

The potential for cheating makes measurement error, in part, an endogenous consequence of the choice of abatement policies. The incentives to cheat can be influenced through more intensive and/or less predictable monitoring and through penalties for misrepresentation as well as for violations (Polinsky and Shaven 1979).[13]

Cheating is one manifestation of a more general problem--information asymmetry. Information on actual production practices, emissions, and costs is available to a polluting firm but often unavailable to a regulatory agency. Information asymmetries can take two basic forms: moral hazard (inability to observe inputs) and adverse selection (inability to observe technology or type). A number of studies have examined the implications of asymmetric information regarding pollution control[14].

In the context of agricultural NSP, information problems related to imperfect monitoring arises in at least three ways: (1) the inability to observe emissions, (2) the inability to infer emissions from observable inputs, and (3) the inability to infer emissions from ambient environmental quality. While no one of these by itself necessarily prevents the design of an efficient pollution control instrument, the combination of the three makes policy design in this context particularly challenging.

Unobservability of Emissions. The inability to observe emissions is the single most troublesome characteristic of nonpoint source pollution and the feature that most distinguishes NSP from point source pollution.[15] Monitoring of NSP emissions is impractical, since emissions are by definition diffuse. For example, measuring the amount of soil lost from a particular field or the amount of a chemical leaving the root zone en route to a nearby aquifer would require monitoring over the entire field rather than at a single location in the field. The associated monitoring costs are prohibitively expensive.

The inability to observe emissions impedes the use of the single most common environmental policy instruments--the emission standard. The lack of observability also undercuts the use of emission taxes, complicates the application of liability (Miceli and Segerson 1991), and diminishes accountability for abatement incentives.

Of course, the inability to observe emissions could be circumvented if the level of emissions were perfectly correlated with some other observable part of the production or pollution process, such as an input or ambient quality (Nichols 1984). In this case, a tax or standard on the input or the ambient quality could serve as a perfect substitute for an emissions tax. However, as discussed below, such a close correlation is unlikely. In the absence of close correlation, a policy based on a particular input or ambient condition could diminish efficiency by biasing the selection of inputs or failing to account for differences in emissions.[16]

Unobservable Inputs/Technology Many agricultural nonpoint source pollutants are closely associated with specific, readily observable production inputs. For example, pesticide contamination is closely associated with pesticide use; more particularly, it is associated with the pesticides that are applied to specific crops grown in porous soils over shallow aquifers. The amounts of pesticides purchased, the crops being grown, and the physical circumstances can all be determined by a regulatory agency. Similarly, erosion is closely associated with certain crops, soils, and tillage techniques, and these are readily inspected.

However, agricultural pollution levels are likely to depend not only on these observable inputs, but also on some critical, unobservable inputs. For example, the pollution resulting from a given quantity of pesticide applied may depend not only on the total quantity applied but also on the care with which it is prepared, the timing of application, and where it is applied (such as how close to streambanks or wellheads). While these timing and application inputs are theoretically observable, observations by a regulatory body would require continual monitoring of farm operations, which is impractical.

The unobservability of some key inputs implies that these inputs cannot be subject to direct control through regulations or taxation. In addition, taxing or regulating only the observable inputs will generally distort the chosen input mix and induce inappropriate substitutions.

The inability to control inputs directly is a classic moral hazard problem. The usual prescription is an output-based incentive instrument. With agricultural NSP, such an instrument would have to be based on ambient environmental quality. As we discuss below, this is not an entirely satisfactory solution, since information problems are likely to hamper the efficiency of such policies. Fortunately, however, in some respects, a farmer's personal economic interest may deter environmentally egregious uses of inputs, such as wasteful chemical applications.[17] To the extent that private costs and benefits cause farmers to use timing and application methods that reduce runoff and leaching in order to increase efficacy, the moral hazard problem from unobservability of these inputs is reduced.

Inferring Emissions from Ambient Pollution. Since ambient pollution levels are relatively easy to observe, they can provide information about the extent of polluting activities in the vicinity of a given environmental medium. Unfortunately, however, while it may be relatively easy to observe contamination levels (such as the turbidity of a stream or the level of contamination of an aquifer), attributing that contamination to a given level of emissions at a particular source may be very difficult. For example, determining the origin of particles deposited in a stream is virtually impossible.

The inability to infer emissions from observed ambient pollution is the result of both natural randomness and the influence of other neighboring polluters. If many polluters border a particular stream or overlie a particular aquifer, then the level of contamination is determined by their combined activities. In addition, the effectiveness of abatement measures undertaken by one firm depends on the actions taken by others.

Despite the inability to attribute a given level of ambient pollution to the activities of individual polluters, Segerson (1988) and Xepapadeas (1991) have shown that, at least in theory, an ambient tax/subsidy scheme can provide the correct incentives for individual polluters to undertake socially efficient abatement measures.[18] Under the proposed policy, each polluter (actual or potential) would be required to pay an ambient-based pollution tax (or receive a subsidy) equal to the full marginal social cost (benefit) of the collective level of contamination (abatement).[19] Even with multiple polluters, this approach provides each polluter with the socially efficient marginal incentive to abate.[20] Polluters for whom management changes will have little impact on contamination will have less incentive to abate than those whose management changes will have a large effect. The tax would also encourage the most efficient means of abatement, be it reducing inputs or modifying technology.

While in theory the above proposal ensures first-best incentives even in the presence of multiple polluters, it suffers from several practical difficulties. For example, setting each polluter's efficient tax rate requires extensive information on the entire process outlined in Figure 1 for each polluter contributing to the contamination.[21] This presents a serious information burden and maybe impractical, Furthermore, each polluter's tax exposure depends in part on the pollution of others. A uniform tax could be criticized as equal punishment for unequal pollution.

Practical difficulties in monitoring ambient quality may reduce the incentive effects and, hence, the efficiency of an instrument applied to ambient contamination. Ideally, ambient-based taxes would be implemented on the basis of continuous monitoring of environmental quality. The policy signals sent to polluters then could be continually adjusted according to actual circumstances. However, this ideal is far from realistic. A more likely scenario is the periodic taking of samples in a sparse network of monitoring sites. The policy signals would be based on extrapolations to unmonitored sites and times. In such a setting, abatement efforts will have only a tenuous effect on the measured outcomes. Accordingly, polluters will be discouraged from undertaking socially desirable abatement.

An alternative to ambient-based incentives is *ex post* liability for contamination or damages. This approach provides a potential solution where only some unpredictable subset of all emissions cause damages and the transactions costs are modest for seeking compensation for individual episodes.

Liability works only when causality can be established. Thus, information must be available to establish the reality of harm and the responsibility for having caused it. The inability to observe emissions, coupled with the inability to attribute ambient pollution to any individual farm due to natural randomness and the influences of multiple polluters, implies that causality may be difficult to establish or prove in many cases of nonpoint source pollution.[22] As such, even if polluters are theoretically liable for damages under either statute or the law of torts, there is a significant positive probability that they would not actually be held liable. This clearly reduces the incentives for pollution abatement.

Another practical difficulty with liability remedies is that the expected liability for damages as viewed by tortfeasors is likely to be below the expected value of damages The difference is due to the potential to avoid damage claims through bankruptcy, the less than certain likelihood of suits by victims, and the possibility of an inappropriate verdict (Shaven 1984 and Kolstad, Ulen, and Johnson 1990). In the case of agricultural NSP, all three factors seem pertinent, but especially the uncertainty about an appropriate verdict (since farmer liability for environmental damages is only now beginning to be considered) and bankruptcy (since most farms are small enterprises with limited capacity to spread the risk of a damage claim). Under these circumstances, liability alone cannot be counted upon to balance social costs and benefits.

There are certain types of problems, however, for which liability might be effective. One is the case of manufacturer liability for damages due to pesticide contamination of groundwater. Here, bankruptcy is less of a problem since many chemicals are produced by large companies. In addition, liability for damages from products is a well-established field within tort law, suggesting that the legal system has established mechanisms for dealing with such cases. Finally, for many chemicals, a distinctive chemical “fingerprint” removes doubt about the “responsible party”, in terms of the manufacturer. Segerson (1990) establishes that producer liability has consequences equivalent to perfect application of user liability, in that producers will increase the prices of pesticides to fund their expected liability exposure. Thus, holding the manufacturer strictly liable has the same effect as charging the chemical user for damages. The liability will cause the manufacturer to assess the financial exposure and raise the chemical price accordingly. The assessment, and the resulting price increases or users warnings, may even take into account different levels of risk in different physical settings--for example, where soils are more permeable or ground water resources are closer to the surface. Such price increases would discourage use of the chemical just as taxes would. However, if contamination is affected by timing and method of use, manufacturer liability alone may not ensure that these dimensions are efficiently exploited.

III. Multiple Instruments as a Response to Information Problems

With the information problems discussed above and the many facets of agricultural nonpoint source pollution, no single policy instrument is likely to yield efficient pollution abatement decisions. Input taxes applied only to observable inputs will ignore the role of unobservable inputs, thereby distorting input choices. Likewise, while the use of ambient-based policies avoids the need to control input use directly, it is likely to lead to an imperfect internalization of costs due primarily to the inability to attribute ambient pollution to the activities of individual polluters.

Rather than frame the problem as a choice between two imperfect approaches, we suggest that a preferred approach may be to combine policy tools into a policy “package.” While we have made similar suggestions previously (Braden 1990 and Segerson 1990a), we are unaware of any formal analyses of the welfare effects of a multiple instrument approach in the presence of information problems. The use of multiple tools or instruments is redundant in a world of first-best single instruments, but it may have a role to play in improving efficiency when single instruments are imperfect.[23]

In this section we consider a very simple model that illustrates the role that multiple instruments can play in the control of nonpoint pollution. For simplicity, we consider only two information problems: (1) the inability to observe (and thus tax) all pollution-related inputs, and (2) the chance that a responsible party may not be held liable for damages under liability due to difficulties in identifying the source and establishing causation.[24] We show that, while the sole use of an input tax (on the observable input) or liability will not be efficient, combining the two policies may improve social welfare. This result is not guaranteed, however, since in some cases combining policies can actually reduce welfare. The result depends upon the way in which pollution-related inputs interact with each other in both the production and the pollution process. This suggests the need for care in combining policies to ensure complementarity between the individual policies.

Consider a farm that uses two inputs, X and Y, to produce an output. Let the net private benefits from the production process be NB(X,Y), with $NB_x > 0$ and $NB_y > 0$. (Subscripts on functions denote partial derivatives.) NB is assumed to be strictly concave in (X,Y), implying $NB_{xx} \leq 0$, $NB_{yy} \leq 0$, and $NB_{xx}NB_{yy}-(NB_{xy})^2 \geq 0$.

Use of the inputs is also assumed to result in an expected level of damages from ambient pollution, denoted D(X,Y). To the extent that damages are influenced by random variables such as weather, D will depend on both the probability distributions of these random variables and the set of possible outcomes. For simplicity of notation and without loss of generality (given risk neutrality), we subsume these random effects into the D function, which represents <u>expected</u> damages. In addition, if there are multiple polluters, expected damages may also depend on the actions of other farms. In this case, D would have additional arguments reflecting the decisions of other firms. We do not consider the role of other firms explicitly, since doing so would complicate the exposition without changing the basic qualitative conclusions. Finally, damages could result from contamination of several environmental media. For example, D could represent combined impacts on groundwater and surface water (i.e., $D=D^s+D^g$, where D^i denotes damages to media i, with i=s (surface water) or g (groundwater)). We assume that $D_x>0$ and $D_y>0$, i.e., that increases in either of the inputs would increase aggregate damages. This does not imply, however, that tradeoffs between different media do not exist. For example, increases in input X may increase groundwater contamination ($D^g_x>0$) while decreasing surface water contamination ($D^s_x<0$). We simply assume that, on net, the effect is an increase in overall damages. Finally, we assume that damages are convex in (X, Y), i.e. $D_{xx}\geq 0$, $D_{yy}\geq 0$, and $D_{xx}D_{yy}-(D_{xy})^2\geq 0$.

Expected social net benefits from the farm's production process are SNB(X,Y)=NB(X,Y)-D(X,Y). The first-order conditions for the maximization of expected social net benefits are:

(1) $NB_x - D_x = 0$, and

(2) $NB_y - D_y = 0$.

Given the curvature assumptions on NB and D, equation (1) defines the efficient level of X given Y, which we denote X*(Y). Likewise, (2) defines the efficient level of Y given X, Y*(X). Simultaneously, (1) and (2) define the efficient levels of X and Y, (X^*_e, Y^*_e).

We consider three alternative policy approaches that could be used to internalize the farm's external pollution costs. The first approach is the use of input taxes. However, because of information problems, we assume that not all inputs can be monitored and thus subject to direct taxation. In particular, we assume that, while the regulatory agency can easily observe (and thus tax) the X input, it is unable to tax the Y input. Thus, the first policy alternative is simply to impose a per-unit tax on X, with the level of the tax equal to the marginal external damages from use of X, i.e., $t=D_x$. Faced with such a tax, the farmer would choose the levels of X and Y to maximize NB(X,Y)-tX, yielding the following first-order conditions:

(3) $NB_x - D_x = 0$, and

(4) $NB_y = 0$.

Note that (3) is identical to (1). Thus, under the input tax approach, the firm would choose the efficient level of X given Y. However, since (4) differs from (2), it would not choose the efficient amount of Y given X. Let $Y_o(X)$ denote the solution to (4) given X. $Y_o(X)$ and X*(Y) simultaneously determine (X^*_t, Y^*_t), the input choices under the input tax approach, which will be inefficient.

The second policy approach is to use instead an ambient-based policy such as liability for actual damages. Under this approach, if held responsible for contamination, the farmer would expect to pay an amount equal to the resulting damages. However, again because of information problems, there is some probability that parties responsible for pollution will not be easily identified and thus held liable for the associated damages. Let $p<1$ be the probability that the firm will actually have to pay for the expected damages that it creates.[25] Then, under the liability policy, the firm would choose X and Y to maximize NB(X,Y)-pD(X,Y), yielding the following first-order conditions:

(5) $NB_x - pD_x = 0$, and

(6) $NB_y - pD_y = 0$.

Let $X_L(Y)$ denote the solution to (5) given Y, and let $Y_L(X)$ denote the solution to (6) given X. The simultaneous solution of the two equations gives the input choices under the ambient-based policy (X^*_L, Y^*_L). Note that in this case neither X nor Y is chosen efficiently, given the level of the other input.

Finally, policy makers can use a multiple-instrument approach, under which they combine the use of an input tax on X and liability. If policy makers recognize the imperfections in the use of the liability policy, they can add an input tax on X to try to completely internalize the external costs resulting from the use of X. Alternatively, if they recognize that the external costs from using Y are not internalized through the input tax approach, they can add, for example, a liability rule to try to influence indirectly the choice of Y. It should be noted, however, that when the two policies are combined the level of the input tax that will fully internalize the costs of X will no longer equal marginal expected damages, D_x. Since the marginal effect of liability will impose costs of pD_x, the input tax should simply reflect the remaining costs that have not been internalized, i.e., $(1-p)D_x$.

Under this combined approach, the firm will choose X and Y to maximize NB(X,Y)-pD(X,Y)-tX, where $t = (1-p)D_x$ evaluated at the efficient level of X given Y. This yields the following first-order conditions:

(7) $$NB_x - pD_x - (1-p)D_x = NB_x - D_x = 0, \text{ and}$$

(8) $$NB_y - pD_y = 0.$$

Note that, since (7) and (1) are identical, again the firm chooses the efficient level of X given Y. In addition, comparing (8) and (6) implies that it chooses the same level of Y given X that it would have chosen had a liability rule been used alone. However, the combined solution to (7) and (8), (X^*_c, Y^*_c), will in general differ from the input choices when either of the two policies is used alone.

Our objective is to compare expected social net benefits under the single-instrument approaches (input tax alone or liability alone) to the expected social net benefits when the instruments are combined, to determine if the use of multiple instruments improves social welfare.

Consider first the comparison of the tax alone to the tax coupled with liability. Under the tax alone, expected social net benefit is given by $SNB(X^*_t, Y^*_t)$. Likewise, expected social net benefit under the combined approach is $SNB(X^*_c, Y^*_c)$. Note, however, that

$$(9) \qquad SNB(X^*_t, Y^*_t) = SNB(X^*(Y^*_t), Y^*_t) = S\tilde{N}B(Y^*_t), \text{ and}$$

$$(10) \qquad SNB(X^*_c, Y^*_c) = SNB(X^*(Y^*_c), Y^*_c) = S\tilde{N}B(Y^*_c).$$

Thus, to compare the two approaches, we need simply to determine whether $S\tilde{N}B(Y^*_t)$ is greater or less than $S\tilde{N}B(Y^*_c)$.

It can be easily shown that $Y^*_t > Y^*_c > Y^*_e$, i.e., that imposing liability (in addition to the tax on X) will decrease the use of Y, but with $p < 1$ the resulting use of Y will still exceed the efficient level. Furthermore, by definition of $S\tilde{N}B$ and Y^*_e, Y^*_e maximizes $S\tilde{N}B(Y)$. Thus, as illustrated in Figure 3, it must be true that $S\tilde{N}B(Y^*_c) > S\tilde{N}B(Y^*_t)$. In other words, combined use of liability and a tax on X must result in a higher level of social welfare than use of the tax on X alone.

The intuition behind this result is straightforward. With the input tax alone, the tax can be set to ensure the efficient level of X given Y, although it does not ensure the efficient level of Y. Thus, there is only one distortion in the firm's production decision, namely, the distorted choice of Y (too much Y, given X). Adding liability reduces the level of Y, thereby reducing the distortion and improving social welfare.

Unfortunately, the conclusions are not so straightforward when the combined approach is compared to the use of liability alone. In particular, we show next that use of liability alone can in some cases yield higher social welfare than the combined use of liability and an input tax.

When liability is used alone, expected social net benefit is given by

$$(11) \qquad SNB(X^*_L, Y^*_L) = SNB(X^*_L, Y_L(X^*_L)) = \hat{SNB}(X^*_L).$$

Likewise, expected social net benefit under the combined approach is

$$(12) \qquad SNB(X^*_c, Y^*_c) = SNB(X^*_c, Y_L(X^*_c)) = \hat{SNB}(X^*_c).$$

The desirability of the two approaches then depends on whether $\hat{SNB}(X^*_L)$ is greater or less than $\hat{SNB}(X^*_c)$.

As before, we can easily show that $X^*_c < X^*_L$, i.e., that adding the tax on X to a pre-existing liability rule will decrease the use of X since it increases the firm's marginal cost of X. To rank $\hat{SNB}(X^*_L)$ and $\hat{SNB}(X^*_c)$, we then need to determine if $\hat{SNB}(X)$ is increasing or decreasing at X^*_L and X*C. The slope of $\hat{SNB}(X)$ is given by

$$(14) \qquad \hat{SNB}_x = (NB_x - D_x) + [NB_y - D_y](dY_L/dX).$$

However, using (6) (which defines $Y_L(X)$), this can be re-written as

$$(15) \qquad \hat{SNB}_x = (NB_x - D_x) - (1-p)D_y(dY_L/dX).$$

At X^*_c, $NB_x - D_x = 0$ by (7). Likewise, by (5), $NB_x - D_x < 0$ at X^*_L. Thus, to determine the sign of $\hat{SNB}_x$, we need to determine the sign of dY_L/dX.

The definition of $Y_L(X)$ (equation (6)) implies that

(16) $dY_{L'}/dX = -[NB_{xy} - pD_{xy}] / [NB_{yy} - pD_{yy}]$.

From the curvature assumptions, the denominator of (16) is negative. However, the sign of the numerator depends on the interactions between the two inputs both in the production process and in determining ambient pollution. Without empirical information on these interaction effects, the sign of $dY_{L'}/dX$ cannot be determined. We thus consider the two possible cases ($dY_L/dX>0$ and $dY_L/dX<0$) separately.

Suppose $dY_L/dX>0$. In this case, from (15) it is clear that $\hat{SNB}_x<0$ at both X^*_c and X^*_L. Thus, since $X^*_c<X^*_L$, it must be true that $\hat{SNB}(X^*_c)> \hat{SNB}(X^*_L)$. (See Figure 4.) In other words, if $dY_{L'}/dX >0$, then the combined use of liability and an input tax will be preferred to the use of liability alone.

Suppose instead that $dY_L/dX<0$. In this case, $SNB_x>0$ at X^*_c, while SNB_x can be positive or negative at X^*_L. If $SNB_x>0$ at X^*_L, then again we can unambiguously rank the policy alternatives (see Figure 5). However, in this case, the use of liability alone is preferred to the combined policy. In other words, imposing an input tax on X on top of a pre-existing liability rule will unambiguously decrease social welfare. This illustrates a case where the use of multiple instruments would actually be counterproductive.

Alternatively, if $dY_L/dX<0$ but $SNB_x<0$ at X^*_L, then the results are ambiguous. Depending on the levels of X^*_L and X^*_c, it is possible for liability alone to be preferred (see X^*_L' in Figure 6) or for the combined approach to be preferred (see X^*_L" in Figure 6). Thus, the two policy approaches cannot be unambiguously ranked in this case.

The ambiguity that arises when comparing the use of liability alone to the use of liability plus a tax on X can be explained as follows. When liability alone is used and enforcement is imperfect, there are two distortions present. Neither X nor Y is efficient, given the level of the other input. Adding a tax on X (appropriately set to reflect the existing liability rule) will eliminate the distortion on X, thereby reducing the number of distortions to one. However, it is well-known from the theory of the second best that eliminating one distortion in a world of multiple distortions will not always improve welfare. In this case, the effect on welfare depends on how the tax affects the choice of Y (through the effect on X). How Y responds will depend on the nature of the synergisms (both in production and pollution) between X and Y. If Y stayed constant (at its level with liability alone) or decreased and X decreased when the tax was imposed, then the result would be an unambiguous increase in social welfare. However, in general a change in X could result in an increase in Y. If Y decreases also, then the effect on Y reinforces the effect on X and welfare unambiguously increases. However, if Y increases in response to the decrease in X, then taxing X would actually exascerbate the distortion in the choice of Y. If this effect is sufficiently large to offset the gain from the reduction in X, then welfare could actually decrease as a result of imposing the tax.[26]

The above analysis suggests that the ability of a multiple-instrument approach to combat effectively the information problems inherent in nonpoint source pollution hinges on both the nature of the single instruments that are considered and the nature of the interactions between pollution-related inputs. Thus, designing an effective policy package requires empirical information on these interactions. Unfortunately, little attention has been focused on this issue to date.

IV. *Information Generation as a Direct Policy Objective*

We have argued that information problems prevent the standard policy tools from individually achieving efficient incentives to control nonpoint pollution and that combining instruments may improve efficiency. At the same time, better information about pollution processes could increase the prospects for efficiency in the choice of policy goals and instruments through better ability to forecast the outcomes of particular policy choices. Measures to improve information may be thought of as distinct policy instruments which may be part of a multiple instrument approach to abatement.

For examples: Data provided by the quintennial Natural Resource Inventories, which were initiated in the late 1970's by the federal Soil and Water Resource Conservation Act, have improved the capacity to direct erosion control subsidies toward environmental needs. Data generated in recent years by the U .S. Environmental Protection Agency's national ground water survey have clarified the extent and nature of contamination problems. Data on chemical use will be enhanced by the provisions of the 1990 Farm Bill, that require participants in various farm subsidy programs to keep detailed records on chemical use. In addition, pesticide registration and licensing regulations typically require the keeping of detailed records on applications of restricted use chemicals. These data may contribute in future years to a better understanding of the factors causing pesticide contamination. Similarly, many states now require extensive monitoring of groundwater quality, particulary around landfills and hazardous waste sites, which will provide early warning of problems as well as data for use in understanding causes.

These and other information discovery policies have economic value insofar as they improve the efficiency with which other policy instruments can be applied. Perhaps the most compelling need is for data that will support the selective application of policies to areas that have or to pollutants that cause especially serious problems. Furthermore, through a more complete understanding of the connections between the sources and fates of contaminants, additional information may make *ex post* liability more compelling and more efficient as a mechanism for promoting abatement of nonpoint source pollution.

V. Conclusions

With nonpoint source pollution, natural variability in pollution processes and imperfections in monitoring and measurement (enforcement) complicate the design and implementation of policies. The single most troublesome information problem is the inability to observe NSP emissions. Without this information, pollution control policies must be indirectly applied, through regulations or incentives on input use or ambient conditions. But, indirect instruments are themselves subject to information problems which limit their capacity to achieve an efficient solution. We have shown above that the use of multiple indirect instruments may promote efficiency where single instruments cannot because of information problems.

Our analysis pertains to the stylized facts of public policies toward a variety of nonpoint source pollutions, especially agricultural NSP. Those facts include the simultaneous use of multiple instruments, such as standards plus cost-sharing for erosion control and input regulations plus liability for pesticides. However, the reinforcing effects of multiple instruments are not guaranteed. An indirect policy applied to one component may push firms toward production systems that are more polluting rather than less-polluting. Greater insight into these interrelationships could enhance the efficiency of a multiple instrument approach. Thus, generating better information can bean important foundation for a pollution control strategy that also includes other instruments.

Figure 1. Information about Pollution Relationships

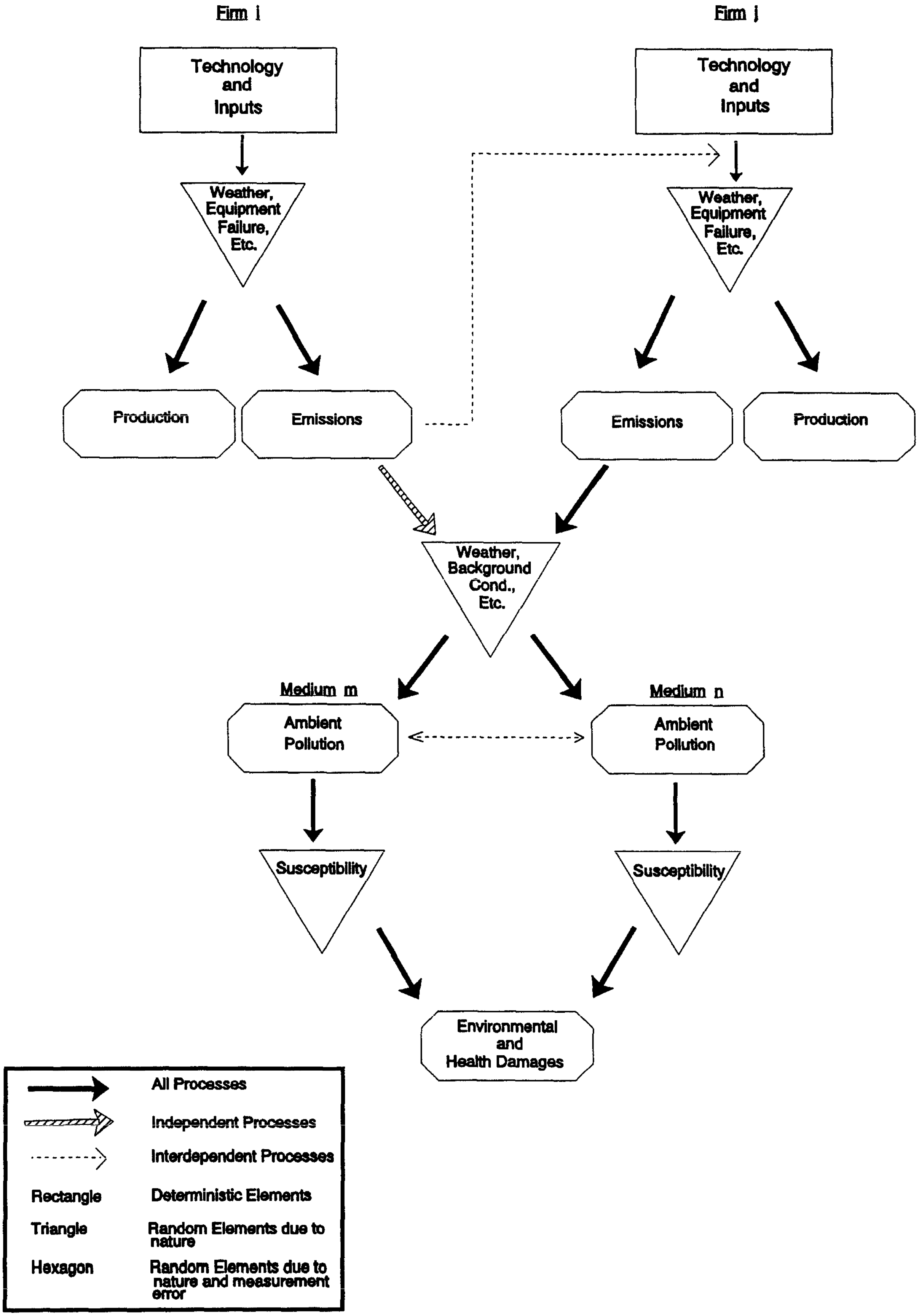
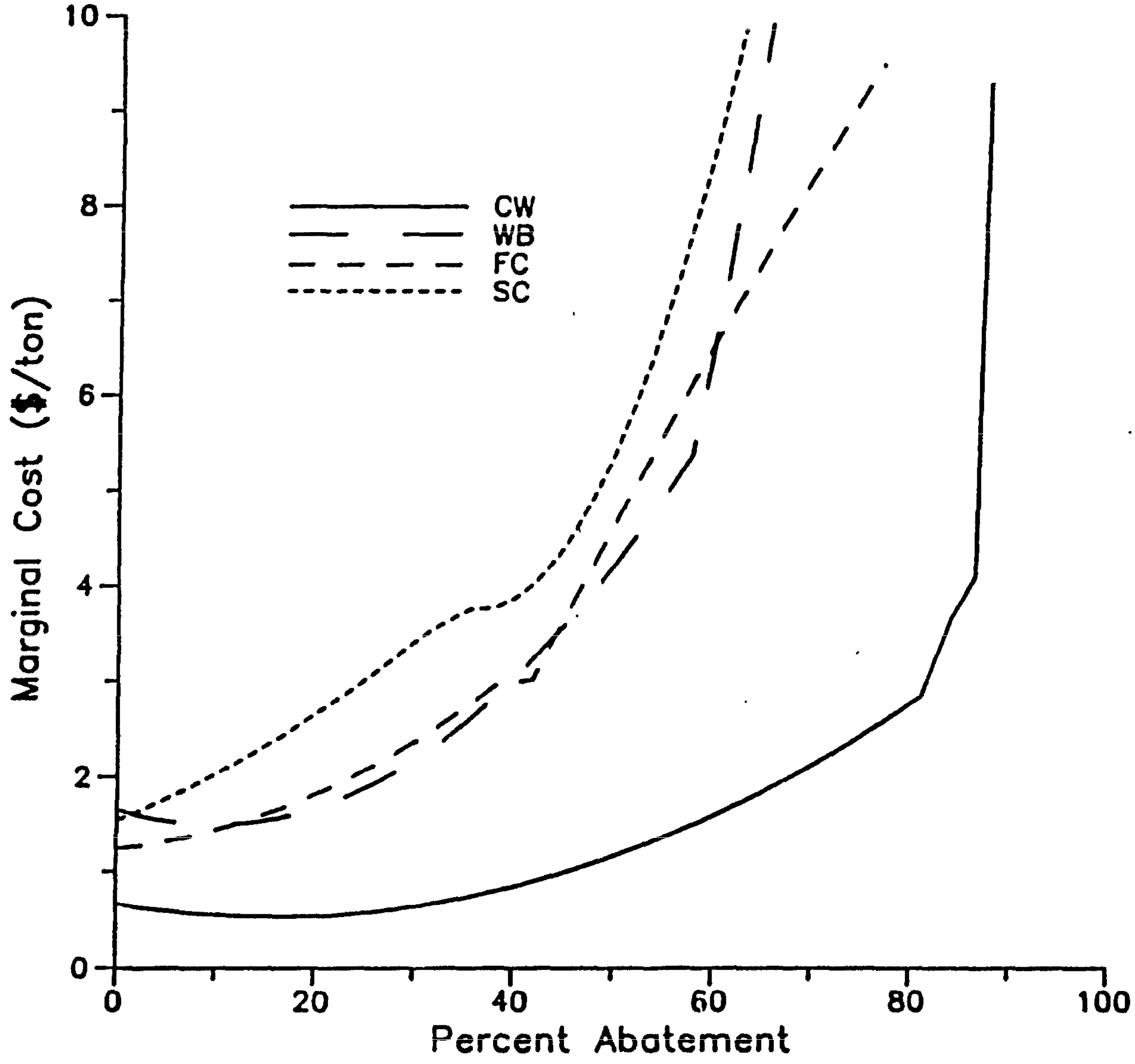


Figure 2. Illustrative Supply Curves for Sediment Abatement



Source: Braden *et al* (1989, p. 410)

Figure 3. Comparison of Input Tax Alone and Tax Plus Liability

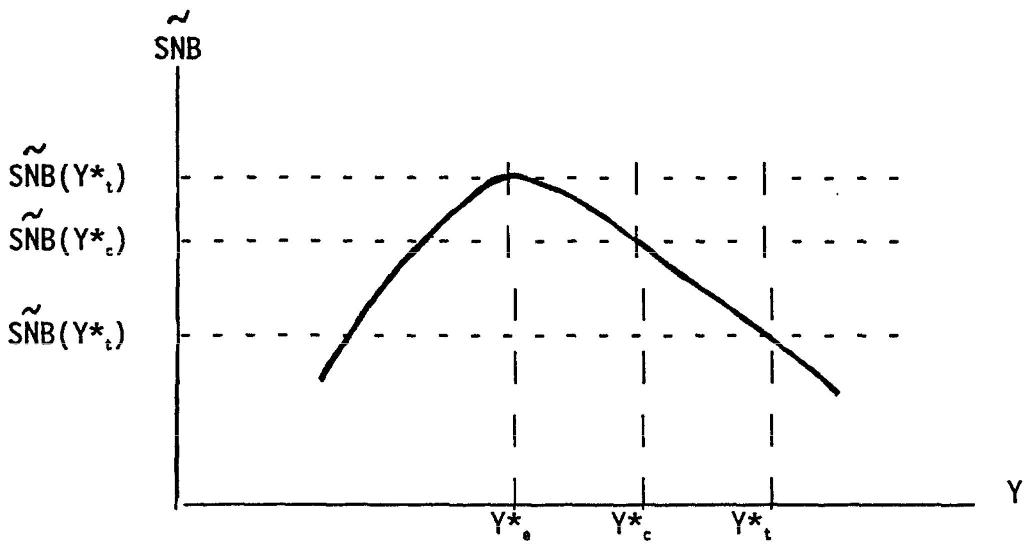


Figure 4. Comparison of Liability Alone and Liability Plus Tax
Case 1: $dY_L/dX > 0$

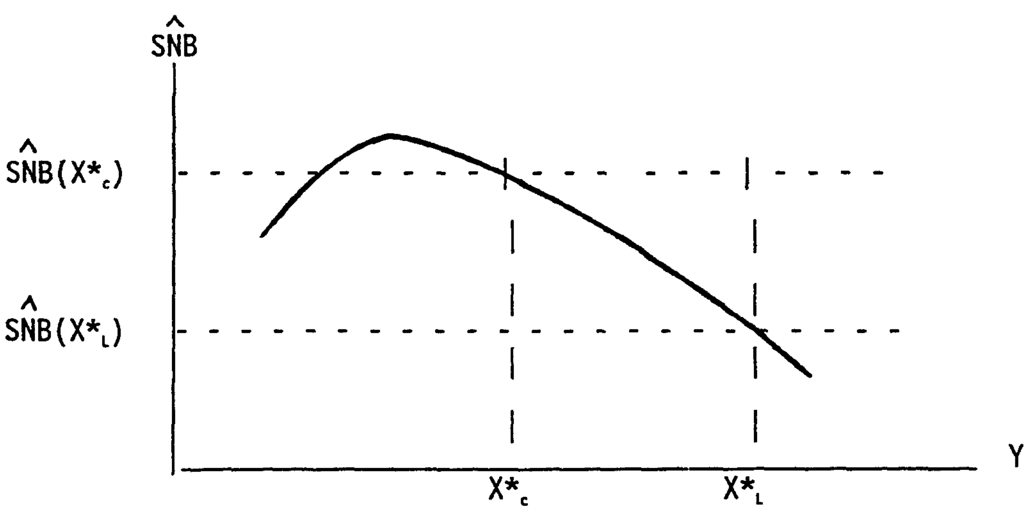


Figure 5. Comparison of Liability Alone and Liability Plus Tax
Case 2a: $dY_L/dX < 0$ and $\hat{SNB}_x > 0$ at X^*_L

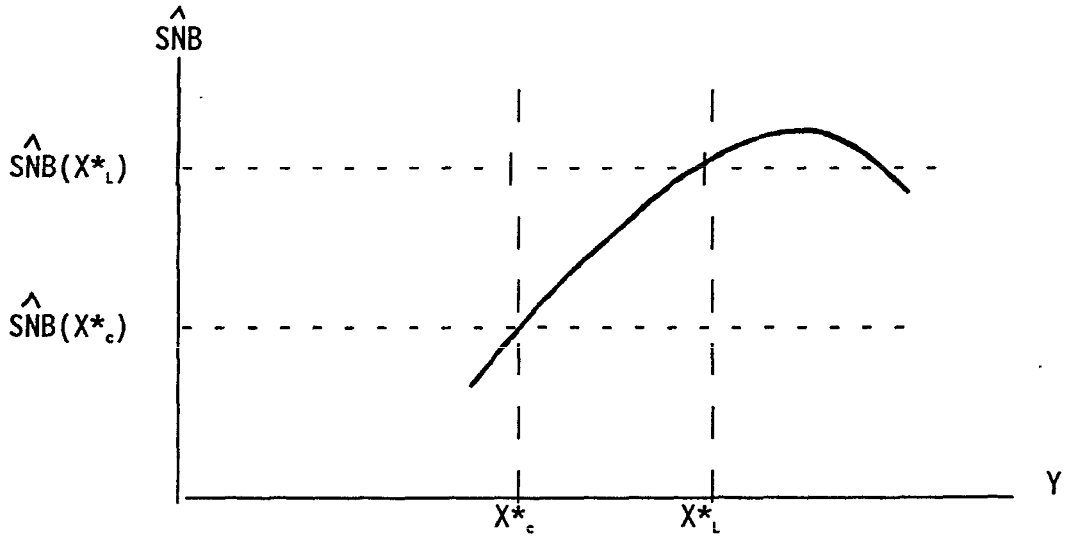
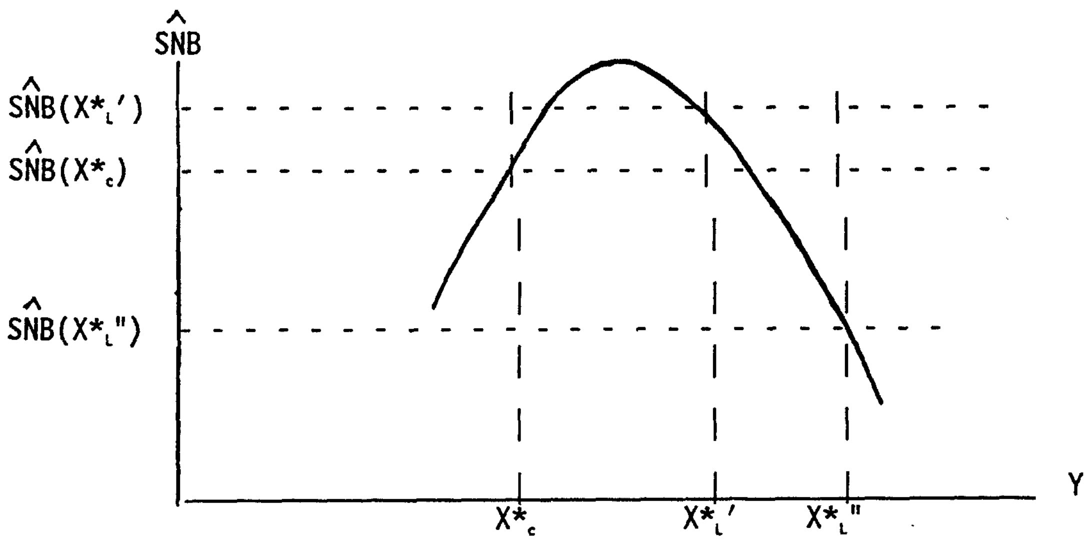


Figure 6. Comparison of Liability Alone and Liability Plus Tax
Case 2b: $dY_L/dX < 0$ and $\hat{SNB}_x < 0$ at X^*_L



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ENDNOTES

1. Examples of the second-best approach can readily be found in the extensive empirical literature on erosion and sediment control. Many studies in this literature evaluate the costs of public policies that clearly would institutionalize inefficiency, such as erosion regulations or mandated tillage practices(e.g., Lovejoy, Lee and Beasley 1985). Even more common in the economic literature is the consideration of policies one by one, rather than in combination. Examples come from our own work: Miltz, Braden, and Johnson (1988) compared the costs of reducing sedimentation via erosion taxes, erosion standards, and a spatially optimal plan. Segerson (1988) analyzed a tax on surface water ambient pollution due to agricultural emissions, and Segerson (1990b) considered liability remedies for ground water pollution by agricultural chemicals.

2. Agricultural nonpoint sources pollution exclude effluents from livestock confinement areas or spills or spills at chemical storage sites; these are generally regarded as point source problems and are regulated accordingly.

3. In addition to pollution processes, random variables also influence output levels. Concern for variation in output can affect the randomness of pollution outcomes. For example, a major concern for farmers is to hedge against the possibility of bad weather and bad crops. Crops, inputs, and farming techniques are chosen in part because of their potential for circumventing various risks--bad weather, bad prices, large fixed investments, and so on. Agricultural pollution stems in part from the choices that are made. Thus, government policies can influence polluting behavior indirectly by influencing the relative risks of different production systems (Kramer *et al.*, 1983; McSweeney and Kramer, 1986).

4. As shown by Weitzman (1974), with variable abatement costs and the need to select a policy instrument before costs are resolved, both incentives and regulations can be set to produce zero expected efficiency losses, but the costs of being wrong can differ greatly depending on the relative slopes of the damage and abatement cost curves.

5. With a flat supply curve and a steep demand curve, a regulatory instrument will tend to minimize the losses. On the other hand, with a steep supply curve and a flat demand curve, an incentive instrument will tend toward smaller losses.

6. While this paragraph makes particular reference to temporal variability, its conclusions also apply when a single policy is applied to a problem that varies through space.

7. Although standard instruments may be used, determining an optimal set of policies may be extremely difficult. Lichtenberg and Zilberman (1988) circumvent the problem by assuming a tractable functional form for damages More generally, however, Beavis and Walker (1983) analyze the case where the firms' discharges are independent and the goal is to limit the sum of realized emissions according to a constraint like (1). They show that, generally, the feasible set is nonconvex, so the first order conditions are not sufficient to ensure an optimum, and the regulatory or incentive measures identified through standard analyses will not necessarily be globally efficient. The search for efficiency would require detailed information and exhaustive analyses of alternatives. Convexity can also arise from interdependence among polluters. See, for example, the studies of sediment abatement by Bouzaher *et al.* (1990) and Braden *et al.* (1989).

8. For a somewhat outdated survey, see DeCoursey (1985).

9. Park and Shabman (1982) studied reductions of phosphorous. Reductions in sediment, phosphorus, and nitrogen were analyzed by Milon (1987). Braden *et al.* (1989) and Miltz *et al* (1988) considered the costs of reducing sedimentation. Braden *et al.* (1991) considered the costs of protecting fish habitat from sediment and pesticide pollution. Yet more numerous are studies of the costs of input restrictions (e.g., fertilizer or pesticide restrictions) without any linkage to environmental contamination or damages. These studies typically employ farm budgeting or linear programming techniques that involve fixed input combinations or tradeoffs. The Universal Soil Loss Equation (Wischmeier and Smith, 1978) is usually used to predict average annual erosion rates--there is no assumption that the rates could actually be observed in the field. Rarely are pesticide or fertilizer use rates transformed into measures of emissions.

10. Miles (1987) estimated the offsite value of erosion reduction to be about $1.00 per ton. Clark *et al.* (1985) estimated total offsite damages of $2.2 billion per annum for sediment in the U.S. If 10 percent to 30 percent of eroded soil accounts for these damages, the damages per ton would be between $0.75 and $2.30. Unfortunately, full-fledged demand (benefit) functions for abatement appear to be absent from the literature.

11. The supply curves in Figure 2 represent different assumptions about the relationship between erosion and sedimentation. The curve labelled CW reflects detailed simulation of the overland transport process including locations where deposition occurs. The curve labelled WB is based on a distance function--the closer source of the erosion, the higher the percentage of eroded soil that enters the water body. The curve labelled FC presumes that each field can be represented by a unique but fixed delivery ratio. This ratio does not change even though surrounding land uses change. Finally, the curve labelled SC is based on a single, fixed, average delivery ratio for the entire area. The curves CW, SC, and FC were calibrated to be directly comparable. The methodology behind WB cannot be directly compared to the others.

12. As evident in Figure 2, different methods of estimating the cost function may produce different curvatures and different conclusions about the type of instrument that will minimize realized errors. Here we have another information problem--limited understanding of pollution abatement options leads to the potential for substantial specification error in the estimation of abatement costs.

13. Russell, Harrington and Vaughan (1986), and more recently Harford (1990), show that monitoring costs and losses due to cheating can be diminished by a “state-dependent enforcement” scheme that makes the likelihood of monitoring and the regulatory standard conditional on a firm’s history of compliance. Those who have cheated and been caught would subsequently face more intensive monitoring and a tougher standard, and these threats help to induce compliance by former cheaters and potential cheaters alike.

14. See generally Besanko and Sappington (1987). Concerning environmental problems, Spulber (1988) analyzes an adverse selection problem while Shortle and Dunn (1986), Segerson (1988), and Xepapadeas (1991) consider a moral hazard problem. In Spulber's case, pollution abatement costs (which define the “type” of firm) are private information. An optimal policy entails abatement contracts which make polluters indifferent between falsifying their costs in an effort to gain a more lenient policy and truthful revelation of costs leading to a socially efficient standard. This is done by paying information rents. Such a policy is worthwhile only if the social benefit of less pollution is greater than the sum of the information rents plus the reduced economic surplus in the product markets to which the pollution is connected. In the other studies, the emissions or abatement efforts of individual polluters cannot be observed. Under these circumstances, an optimal policy involves a combination of fines and subsidies. One of the

instruments induces the optimal marginal incentives to abate while the other transfers income to counteract long-run distortions that can arise when the marginal incentives are determined by collective rather than individual emissions.

15. While this difficulty seems especially pronounced for nonpoint source pollution, indirect instruments also are used for point source applications in some instances when the costs of emissions monitoring would be prohibitive. This explains, for example, the use of design standards rather than emissions standards for various types of industrial fumes that are not easily captured in collection systems.

16. Holterman (1976) shows in a deterministic setting that efficient correction of an externality caused by a multiple input production process generally requires an instrument applied to each pollution-related input. Use of fewer instruments generally will lead to inefficient solutions. Nichols (1984) shows in a stochastic setting that input-based instruments will be more prone to inefficiency when the covariation between input use and external impacts is low; furthermore, the nature of the covariation can make incentive instruments superior to regulations (or *vice-versa)* in terms of efficiency.

17. This may be one area in which agricultural NPS is different from littering or other nonagricultural problems. For example, with waste disposal, cost-minimization promotes midnight dumping and other deceptive methods.

18. In contrast, the absence of measured emissions would undercut the use of a pure regulatory instrument applied to contamination. There would be no means of translating a contamination standard into abatement actions by individual polluters. However, such a translation could be achieved by combining an ambient standard with input or design restrictions, based on simulated predictions that the restrictions would achieve the standard. Alternatively, taxing contamination in excess of the standard, along lines suggested by Segerson (1988), could provide the necessary incentives.

19. Of course, the analysis could also be conducted for an abatement subsidy. A subsidy could achieve short-run efficiency but would boost agricultural profitability and encourage more of the polluting activity in the long run.

20. See Miceli and Segerson (1991) for an analysis of similar incentive problems in the context of liability for "joint torts", where the actions of several parties combine to determine a single level of expected damages.

21. Where the pollutant transport process causes physical interdependencies in the pollution transport process (Braden *et al.*, 1989), the efficient response of one party to a particular tax rate on ambient quality may depend on the actions of others . Under these circumstances, efficiency cannot be achieved through a fully decentralized price system.

22. The doctrine of joint and several liability offers a way around the need to determine the involvement of potential polluters. Miceli and Segerson (1991) discuss the efficient application of this doctrine to liability for environmental damages.

23. The use of multiple instruments to improve efficiency can be justified on other bases as well. For example, if there are multiple pollution-related inputs, then an input tax approach would require use of multiple input taxes (one for each input). Likewise, if there are multiple environmental media that are affected by a firm's activities, then an ambient-based approach would require use of multiple ambient-based taxes (or liability applied to multiple types of

damages). These examples do not, however, justify combined use of two different approaches such as the simultaneous use of input taxation and liability, which is the topic we consider in this section.

24. While we focus here on liability as a particular form of an ambient-based policy, we could alternatively have formulated the model in terms of ambient taxes. The results would be qualitatively similar, as long as the tax that would be paid by a given firm differs (with some probability) from the damages caused by that firm.

25. Note that the probability of being held liable for damages may differ across different media and different pollution types. For example, it may be easier to identify the source of a particular pesticide found in groundwater than the source of sedimentation in a given stream. For simplicity, we abstract from these issues here. Including them would complicate the notation without changing the basic qualitative results.

26. Of course, this is the same result that would be obtained if the base scenario were no liability and a tax were imposed on only one of the polluting inputs.

Working Draft

REGULATORY/ECONOMIC INSTRUMENTS FOR AGRICULTURAL POLLUTION: ACCOUNTING FOR INPUT SUBSTITUTION

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I. INTRODUCTION

In recent years, economists have explored the properties and relative merits of alternative instruments for the control of agricultural nonpoint-source pollution, such as design standards or incentives applied to farm inputs and performance-based standards or incentives applied to pollutant discharges. For the most part, such explorations have led to the conclusion that it is not practical at present to target either standards or incentives on agricultural nonpoint-source discharges directly, since their measurement is so difficult (e.g. Griffin and Bromley 1982; Dunn and Shortle 1988; Segerson 1988). As a result, much of the discussion regarding potential policies for agricultural pollution abatement has focused on restricting or providing negative incentives for the use of agricultural inputs and practices that yield pollution (such as nutrient fertilizers, pesticides, and erosive tillage systems).

A complicating factor in the design of instruments, however, is that a policy applied to one agricultural input can alter farm management practices and thus the utilization rates of other inputs. Such changes in input mixes have the potential to increase the release of pollutants that are different from the pollutant targeted by the instrument.

One example of potential input substitution involves tillage methods and their relationship to other inputs and practices used in agricultural production. Agricultural experts have long advocated conservation tillage as a best management practice (BMP) for soil consecration and the reduction of surface runoff of dissolved or sediment-bound pollutants from cropland. This BMP generally is defined as any type of tillage system that significantly lowers the erosion of soil by increasing the amount of plant residue (from the previous crop) that remains on the soil surface following planting. For example, where water erosion is the chief problem, consecration tillage is defined to be a system that maintains residue cover over at least thirty percent of the soil surface. The four main tillage types that satisfy this definition are no-till (which leaves the greatest amount of residue), mulch-till, ridge-till and strip-till (Soil Conservation Service 1989). The percentage of total acres farmed with conservation tillage in the United States has risen significantly in recent years, with the highest percentages now occurring in the Appalachian region (36% of total acres), the Northeast (33%), and the Corn Belt (33%) (Conservation Technology Information Center 1990).

Conservation tillage appears to be quite successful in reducing soil erosion and associated surface runoff of pollutants. However, experts have observed that the use of conservation tillage often is accompanied by increased use of nitrogen fertilizer (Crosson 1981), and studies have shown that under some soil conditions conservation tillage can lead to a significant increase in the infiltration of nitrogen into the subsurface (e.g. Alberts and Spomer 1985). In addition, the adoption of conservation tillage generally is associated with increased pesticide use (Crosson 1981; Epplin et al. 1982; Jolly et al. 1983; Duffy and Hanthorn 1984). Herbicide use tends to be higher under reduced tillage because conventional tillage serves to remove weeds In addition, reduced tillage creates moister soil conditions and more crop residue on the surface, both of which foster weed growth. (Residue increases the growth of weeds by isolating some of the applied herbicides.) The need for insecticides also tends to increase under conservation tillage because plant residue creates more favorable conditions for insects.

If an agricultural producer were able to increase sufficiently the use of integrated pest management (IPM) methods at the same time that he adopted conservation tillage, then it might be possible to reduce soil loss and associated surface runoff without significantly increasing the sheer volume of pesticides applied.(IPM decreases the necessity for pesticides through methods such as improved timing of planting and pesticide applications, use of resistant crop types, scouting, crop rotations, and biological pest control (National Research Council 1989).) However, the extent of this potential for IPM is not well understood at present. Furthermore, imperfect information regarding the private costs and benefits of IPM may lead to its suboptimal use. Thus while IPM may play an important role, evidence to date for specific crops clearly points to higher uses of pesticides under conservation tillage systems. Duffy and Hanthorn found this to be true for corn and soybeans in the major producing states, as did Epplin et al. for Southern Great Plains winter wheat.

As a consequence, conservation tillage may increase the potential for nitrogen and pesticides to escape from cropland and particularly to leach into groundwater. Conversely, it is conceivable that restrictions on herbicide use in areas that have exhibited increases in conservation tillage could force farmers to revert to conventional tillage methods, thereby exacerbating soil loss and surface runoff problems (Gianessi et al. 1988). Therefore a possible tension, or tradeoff, appears between the reduction of soil loss/surface runoff on the one hand and the infiltration of agricultural chemicals into groundwater on the other.

This paper presents an exploration of conceptual approaches to the problem of simultaneously managing multiple categories of agricultural pollution. First, the paper shows in simple fashion the way in which the "least-cost" allocation of pesticide abatement in an area may change if a link between pesticides and tillage is considered. (Though there also may be a relationship between tillage and nitrogen discharges, this paper focuses on pesticides.) Next, the paper describes a dynamic model that accounts for the possibly long-term damages that may result when pesticides leach into groundwater. The approach is useful in that it illustrates conceptually the source-specific characteristics that influence variability across areas and farmers in the desired degree of adoption of a BMP (in this case conservation tillage). Clearly, the information required to achieve an "optimal" tradeoff between different BMPs is far beyond reach. Therefore, the paper concludes by offering a few observations on possible ways to move closer to a least-cost approach to agricultural pollution control.

II. CONCEPTUAL APPROACHES

A. Allocating a Reduction in Pesticide Applications

Suppose that an environmental planning agency for a given area were to focus on the objective of reducing the area-wide rate of pesticide applications by a particular amount.This kind of objective would be similar to that established in 1987 for nutrients within the Chesapeake

Bay Agreement. Under that agreement, the U.S. Environmental Protection Agency and the states surrounding the Bay agreed to cut the loading of nutrients into Chesapeake Bay by forty percent by the year 2000. An objective of cutting area-wide pesticide applications, though, would be even simpler than the Chesapeake Bay nutrient objective in that it would not account for the relationship between application of pesticides at a given source and the loadings of pesticides to either surface water or groundwater.

Consideration of such a basic objective may be used to show how a linkage between pesticide application and tillage might affect the desired pattern of efforts to reduce the release of agricultural pollution in a particular area. First, suppose that no account were taken of the link between inputs. If the planning agency wished to achieve at least cost a reduction of a minimum of $\bar{p}$ in total pesticide use, then the problem would be similar in spirit to that which Krupnick (1989) illustrates for allocating the reduction of nutrient loadings to Chesapeake Bay, yet simpler due to the omission of discharge-loading coefficients:

$$\min_{p_j} \sum_j C_j(p_j) + \lambda_1(\bar{p} - \sum_j p_j) \qquad (1)$$

where: p_j = reduction in annual rate of pesticide application at Source j,

C_j = total annual private cost of reducing pesticide application at Source j,

$\bar{p}$ = desired annual reduction in area-wide pesticide application rates,

$C_j'(p_j) > 0,\ C_j''(p_j) > 0.$

The conditions representing the least-cost pattern of pesticide reduction, then, are:

$$C_j'(p_j) - \lambda_1 = 0, \quad \text{for all } j \qquad (2)$$

$$\bar{p} - \Sigma p_j \leq 0, \ \lambda_1 \geq 0, \ \lambda_1(\bar{p} - \Sigma p_j) = 0 \qquad (3)$$

Condition (2) shows that, if the planning agency wished to adopt an approach of restricting pesticide inputs, it could do so at least cost only if the input restrictions were made to vary so that the marginal cost of reducing pesticide use were equal for all sources. Alternatively, as Krupnick points out for nutrient reductions, a permit scheme could be established such that trading would take place until (2) was satisfied.

The socially efficient allocation of pesticide reduction in the area, however, would differ from that described by (2) if the total costs of reducing pesticide application included external costs not borne by the individual sources. That is, some producers might react to an instrument for pesticide reduction (be it an input restriction, tax, or tradable permits scheme) by substituting tillage operations for pesticides as a pest control method. A producer who does this will incur increased costs of labor and equipment necessary to conduct tillage operations. (Depending on the marginal effect of increased tillage, the producer also will incur a cost of foregone future productivity due to soil loss from farmland.) In addition, though, an increase in the surface runoff of pollutants will represent an external cost to society.

Ideally, the planning agency would develop a package of instruments that simultaneously takes into account the potential for surface runoff, the leaching of agricultural chemicals into the subsurface, and possible linkages between production practices that influence the magnitudes of both kinds of pollution. It is helpful, though, to consider initially the kind of problem the agency might face if it were to concentrate on the formulation of a policy for reducing one of the two kinds of pollution, say pesticide leaching. In this case, it might seek to attain the desired pesticide reduction objective while not causing changes in tillage practices that in turn would increase total surface runoff in the area above a predetermined acceptable increment.

This approach clearly would require some knowledge regarding the expected effect of a pesticide instrument on farm-level variables, including yield. In an empirical study, Gianessi et al. (1988) estimated the negative effect of a hypothetical local ban of a particular herbicide on farm production, and the associated consumer and producer welfare effects, for the Chesapeake Bay region. Because of the complexity of estimating input linkages, that study understandably did not attempt to examine the possible effect of such a ban on tillage practices and a consequent countervailing increase in yieldIt is useful for the present purpose of conceptually exploring the pesticide-tillage link to make a simplifying assumption about per-acre yield. Specifically, consider the problem the planning agency would face if it anticipated that producers would respond to an agency action by attempting to keep per-acre yields constant. (In what follows, the relaxation of this assumption would simply involve the introduction of an additional constant term denoting the expected percentage decrease in yield after accounting for anticipated changes in tillage.)

If producers wished to maintain per-acre yields constant, then in general a positive p_j at Source j would require that the producer substitute either or both of the following for the volume of pesticides applied: (1) an increase in tillage operations, or (2) IPM techniques that would allow the farmer to maintain yield while reducing pesticide applications and not increasing the intensity of tillage. Let:

$$\text{Level of Conservation Tillage Used at Source } j = T_j \qquad (4)$$

where: $0 \leq T_j \leq \tilde{T}$

$T_j = 0$ represents full conventional tillage,

$T_j - \tilde{T}$ represents complete no-till farming,

and where intermediate values of T_j represent low-till systems, with increasing values for T_j reflecting higher "percent residue" levels. (Percent residue cover is an accepted way of comparing tillage methods.) It is assumed for simplicity that, for each source j, the method of tillage is homogeneous across all acres at Source j.

The change in T_j that is brought about by the pesticide instrument may be given as $t_j(p_j)$, where $\tau_j < 0$ represents a shift away from conservation tillage and toward conventional tillage, with $\tau_j'(p_j) < 0$. The value of $\tau_j'(p_j)$ is farm-specific and depends on the extent and nature of pest problems; the cost of managing pest problems with innovative IPM approaches; and factors that affect the desirability of conservation tillage, including soil productivity, perception of soil erosion (Gould et al. 1989), and operator tenure status (Hinman et al. 1983).

Define the change in the rate of surface runoff at Source j following a pesticide reduction policy as r_j, where $r_j > 0$ represents an increase in surface runoff. The change in runoff is a function of the change in tillage intensity:

$$r_j = \alpha_j \tau_j(p_j) \quad , \quad \alpha_j < 0 \tag{5}$$

where a_j denotes physical characteristics at Source j that influence the marginal effect of a change in tillage intensity on change in surface runoff. These characteristics would be those represented by the variables that appear in the Universal Soil Loss Equation, e.g. soil erodibility, precipitation, cropping, and farmland slope. Large absolute values for α_j would reflect conditions such as highly erodible soils, high rainfall, and steep land slopes. For simplicity a is assumed to be a constant, although in reality it might vary with the type of tillage employed.

Given these expected relationships, the planning agency could define its pesticide reduction problem as:

$$\underset{p_j}{\mathrm{Min}} \sum_j C_j(p_j) + \lambda_1(\bar{p} - \sum_j p_j) + \lambda_2[\bar{r} - \sum_j(\alpha_j \tau_j(p_j))] \tag{6}$$

where: $\bar{r}$ = maximum area-wide increase in annual surface runoff that the planning agency wishes to allow,

and: $\lambda_1 \geq 0$, $\lambda_2 \leq 0$,

with the following necessary conditions:

$$C_j'(p_j) - \lambda_2 \alpha_j \tau_j'(p_j) = \lambda_1 \ , \quad \text{for all } j \tag{7}$$

$$(\bar{p} - \Sigma p_j) \leq 0 \ , \ \lambda_1 \geq 0 \ , \ \lambda_1(\bar{p} - \Sigma p_j) = 0 \tag{8}$$

$$[\bar{r} - \Sigma(\alpha_j \tau_j(p_j))] \geq 0 \ , \ \lambda_2 \leq 0 \ , \ \lambda_2[\bar{r} - \Sigma(\alpha_j \tau_j(p_j))] = 0 \tag{9}$$

Condition (7) shows that, as under the earlier agency problem, an interior solution requires that at every farm the marginal cost of reducing the application of pesticides should be set equal to the marginal benefit of doing so. Unlike in the earlier problem, the marginal cost of pesticide reduction now includes a term representing the environmental cost of an increase in surface runoff that is expected as producers respond by adjusting their tillage practices.

Condition (7) indicates the general way in which simultaneous incentives for pesticide reduction and conservation tillage should vary across areas so as to yield a least-cost solution to the agency's problem. Since tillage practices are observable by the agency (data exist already) and a tax/permit for pesticide use may be enforced, albeit imperfectly, at time of purchase, incentives targeted on these inputs would appear to be relatively practical from an enforcement standpoint.

B. Intertemporal Differences in Environmental Damages

1. A Simple Dynamic Model

While useful, the simple conceptual approaches above do ignore important aspects of the problem. First, they are based on the agency's objective of attaining at least cost those levels of pesticide application and soil erosion/surface runoff that it somehow has deemed to be acceptable. Under this kind of problem, the planner does not account for site-specific links between discharges at a given source and the environmental damages that are thereby generated. To take this into consideration, it would be necessary to develop some sort of damage function that would have as arguments the different kinds of agricultural discharges of interest.

Second, there may be interesting and important differences in the intertemporal patterns of damages generated by surface runoff and infiltration of pollution. For example, the persistence of pollutants can differ markedly depending on whether they leave farmland via surface runoff or leaching. Some toxic pesticides, for example Aldicarb, degrade rapidly in surface waters. Its degradation in groundwater, however, is much slower (Anderson et al. 1985). More generally, the degradation rates of many pollutants tend to be slower in groundwater due to the lack of sunlight, lower levels of oxygen, lower temperatures, and other physical, chemical, and biological conditions that are unfavorable to important degradation processes. Dynamic models that account for multiple pathways for pollution from a given source have been presented and simulated for an industrial waste stream (Eisworth 1988) and presented for agricultural pollution (Crutchfield and Brazee 1990). Krupnick (1989) uses a dynamic model to analyze damages from agricultural sources affect (which groundwater) and municipal treatment plants (which are assumed not to affect groundwater).

A useful way to incorporate the above elements is to consider the problem a planning agency would face if it wished to account for differences across sources in the link between production practices and environmental damages. Doing so provides insights on the way in which the desired degree of adoption of BMPs, such as conservation tillage and reduced use of persistent pesticides, may vary among geographic areas or producers. It is possible at the same time to incorporate dynamic factors. Consider, for example, the case in which the infiltration of pesticides into groundwater were to cause damages over a much longer period of time than pollutants carried from farmland in surface runoff. This is not to say that surface runoff cannot yield a long-lived flow of damages. However, it is fruitful conceptually to explore the extreme case where the environmental damages resulting at any point in time from the operation of a farm may be thought of as a function of: (1) the stock of existing pesticides that has built up in the subsurface due to pesticide applications in previous periods, and (2) the flow of pollutants that currently is escaping from the source via surface runoff. If the planning agency were interested in minimizing these environmental damages, then its instantaneous "utility function" for a given pollution source (farm) could be written as:

$$\text{Agency Function} = f(R,S)\ , \qquad (10)$$

where: R = flow of surface runoff of pollutants from the farm,

S = stock of pesticides in groundwater resulting from applications on the farm,

$f(R,S) \leq 0,\ f_R < 0,\ f_{RR} < 0,\ f_S < 0,\ f_{SS} < 0,$

and where for simplicity the function is assumed to be additively separable so that $f_{RS} = f_{SR} = 0$. (With this kind of function, the agency's "utility", which is the negative of environmental damages, is always less than zero but may be increased by lowering surface runoff or the amount of pesticides in groundwater.)Next, let surface runoff at time t be a function of tillage method at time t:

$$\text{Surface Runoff} = R(T_t)\ ,\ R_T < 0, \tag{11}$$

where T is as defined above but no longer carries the subscript j because the level of analysis is now the individual source (farm). The sign and value of R_{TT} are dependent on physical conditions and presumably vary from farm to farm.

A portion of the pesticides applied to the farmland may be assumed to infiltrate into the saturated zone of the subsurface, and once there to undergo processes of natural decomposition into non-toxic substances. A general equation describing change over time in the stock of pesticides in the groundwater would be of the form:

$$\dot{S} = \beta Z_t - \alpha S_t \tag{12}$$

where: Z = rate of pesticide application,

α = mean rate of natural decomposition of pesticides in the subsurface, $\alpha > 0$,

β = proportion of total pesticides applied that migrate to groundwater, $0 < \beta < 1$,

and where β is dependent on factors such as soil permeability, rainfall, and depth to groundwater. The mean rate of decomposition, a, depends upon the characteristics of the chemicals applied and on physical conditions in the subsurface such as temperature, moisture, and chemical and hydrological characteristics.

Suppose that the agency is interested in encouraging the adoption of conservation tillage on farms in the area. How might the authority want the pattern of conservation tillage to vary spatially? The agency realizes that a shift toward conservation tillage may cause some producers to increase the intensity of pesticide use, but that the magnitude of such an effect would vary appreciably across farms. Given this, the agency might be interested in examining the way in which source-specific characteristics influence the desired level of conservation tillage at a given farm.

In order to do this most accurately in practice, it would be necessary to use a full model of agricultural production to estimate the response of all important variables, including the level of agricultural production, to an instrument that would require or encourage conservation tillage. Again, however, it is instructive to consider a much simpler model that focuses on the tension between minimizing damages from the surface runoff of pollutants and the infiltration of persistent chemicals. Assume therefore that in response to the agency's encouragement of conservation tillage, a given producer attempts to keep the rate of production q constant at $\bar{q}$. While this is a simplifying assumption, it may not be unreasonable for conceptual purposes, as some studies of input mixes under alternative tillage practices suggest that farmers who change tillage methods attempt to change the use of other inputs so as to keep per-acre yield at approximately the same level. (For example, Duffy and Hanthorn (1984) find differences in pesticide volumes and mixes, but no significant differences in per-acre yields, across different tillage practices for corn.)

A standard production model would have as inputs labor, capital, materials (such as pesticides and fertilizers) and land (e.g. number of acres and depth of soil). To examine the tillage-pesticide linkage, one may consider without loss of insight a partial production function such as:

$$q_t = q(Z_t, T_t) \tag{13}$$

where $q_Z > 0$ and $q_T < 0$. If the producer is assumed to maintain $q(Z,T) = \bar{q}$, then Z may be expressed as a function of T, with $Z_T > 0$. The magnitude of Z_T will indicate several farm-specific characteristics, including the extent to which greater adoption of IPM would allow this particular producer to move toward no-till without applying a greater volume of pesticides. Though one might suspect that $Z_{TT} > 0$, its sign is not readily apparent and could vary across farms. An intertemporal model is made much more tractable by allowing Z_T to be a constant, and therefore let:

$$Z(T) = \delta T + \bar{Z} \, , \tag{14}$$

where: $\delta > 0$, $\bar{Z} > 0$

and where $\bar{Z}$ represents the rate of pesticide application under conventional tillage.

Given such anticipated behavior of the producer, the planning agency might reasonably set as its goal the maximization of (10) net of the producer's expected costs of pesticide application and tillage operations, abstracting from other production costs such as those for seed and fertilizer. Such a planner's problem would be:

$$\max_{T_t} \int_0^\infty \{[f(R(T_t),S_t) - C_1(T_t) - C_2(Z(T_t))]e^{-rt}dt\} \tag{15}$$

$$\text{s.t.: } \dot{S}_t = \gamma T_t + \beta\bar{Z} - \alpha S_t \tag{16}$$

$$S_0 = \bar{S} \tag{17}$$

$$0 \le T_t \le \bar{T} \tag{18}$$

where: $C_1(T_t)$ = total private cost, at time t, of labor, fuel and repair, and machinery necessary for tillage operations; $C_{1T} < 0$; $C_{1TT} > 0$,

$C_2(Z(T_t))$ = total private cost, at time t, of labor, chemicals, fuel and repair, and machinery necessary for pesticide application; $C_{ZZ} > 0$; $C_{ZZZ} > 0$,

r = rate of discount,

$\gamma = \beta\delta > 0$.

The necessary conditions for this problem are:

$$f_T - C_{1T} - \delta C_{ZZ} + \gamma\lambda_1 \le 0,\ T \ge 0,\ T(f_T - C_{1T} - \delta C_{ZZ} + \gamma\lambda_1) = 0 \tag{19}$$

$$\dot{\lambda}_1 = \lambda_1(r + \alpha) - f_S \tag{20}$$

$$\dot{S} = \gamma T + \beta \bar{Z} - \alpha S \tag{21}$$

$$(\bar{T} - T) \geq 0, \ \lambda_2 \geq 0, \ \lambda_2(\bar{T} - T) = 0 \tag{22}$$

where for ease of notation time no longer explicitly appears as a subscript.

Condition (19) says that, for an interior solution, T should be set such that the marginal benefits of conservation tillage (reduced environmental damages from surface runoff plus reduced costs of labor, fuel and repair, and capital employed for tillage operations) equals the marginal costs of conservation tillage (an increased stock of pesticide in groundwater plus increased costs of labor, chemicals, fuel and repair, and machinery for the application of pesticide). Condition (20) shows that the optimal rate of change of the shadow price of the stock of pesticide in the groundwater depends on the instantaneous marginal damage caused by the pesticide stock, the rate of pesticide degradation, and the discount rate. Conditions (21) and (22) are the state equation and the Kuhn-Tucker conditions relating to the upper bound on T (complete no-till).

The dynamically optimal level of conservation tillage is given by the simultaneous solution of (21) and the steady-state condition for T. This condition is found by differentiating (19) with respect to time and substituting the result and (19) into (20), which after simplification gives:

$$\dot{T} = [(C_{1T} + \delta C_{ZZ} - f_T)(r + \alpha) - \gamma f_S]/[C_{1TT} + \delta^2 C_{ZZZ} - f_{TT}] \tag{23}$$

In steady state, then:

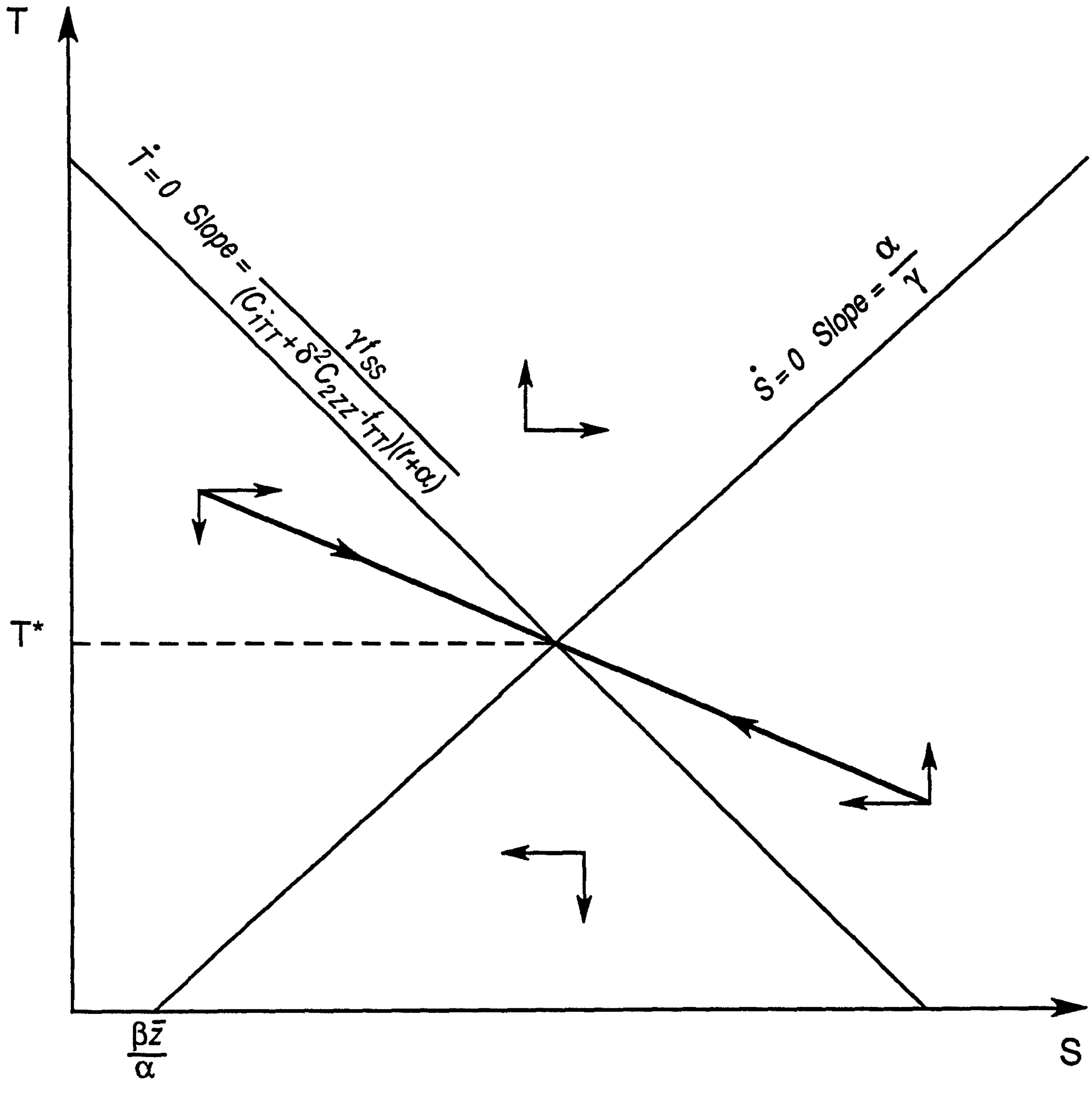
$$(C_{1T} + \delta C_{ZZ} - f_T)(r + \alpha) = \gamma f_S \tag{24}$$

Total differentiation of (24) shows that, as long as f_{TT} is either negative or, if positive, is less than $(C_{1TT} + \delta^2 C_{ZZZ})$, then the steady-state locus for T will slope downward in T-S space as shown in the phase diagram of Figure One. (The standard assumption is that f_{TT} is negative, which represents diminishing returns, in the form of reduced environmental damages from surface runoff, to conservation tillage.) As Figure One shows, a saddle point equilibrium exists for this problem.

One conceptual benefit of this model is that comparative statics analysis can show how changes in site-specific characteristics influence the optimal level of T. As an example, one may determine how the "desired" level of conservation tillage might vary from farm to farm according to variation in δ (the anticipated farm-specific link between tillage practice and the rate of pesticide application) and β (the proportion of applied pesticides that are expected to leach into groundwater). In this simple model, $y = \delta\beta$. The effect of a change in y is given by:

$$\partial T/\partial \gamma = -(\alpha f_S + \gamma f_{SS} T)/\{[-\alpha(r + \alpha)(C_{1TT} + \delta^2 C_{ZZZ} - f_{TT})] + \gamma^2 f_{SS}\} \tag{25}$$

Inspection shows that (25) is unambiguously negative, which is completely intuitive. In relation to δ, this means that for a farm at which one would expect to see a relatively high rate of substitution of pesticides



T
$\dot{T}=0$ Slope $= \frac{\gamma f_{ss}}{(C_{1TT}+\delta^2 C_{2ZZ}-f_{TT})(r+\alpha)}$
$\dot{S}=0$ Slope $= \alpha/\gamma$
T*
$\frac{\beta\bar{z}}{\alpha}$
S

Figure One

for tillage operations as a pest control method, the optimal level of conservation tillage would be relatively low, all else equal. With regard to β, this means that at a site exhibiting physical conditions that favor pesticide infiltration, the optimal level of T again will be relatively low, all else equal, as expected. The magnitude by which changes in δ and β would affect the desired degree of adoption of conservation tillage depends upon the private cost functions for pesticide application and tillage operations; the instantaneous damage function for surface runoff and pesticides in groundwater; the persistence of the pesticides; the rate of discount; and the values of y and T. Though the results are not shown here, one can use comparative statics analysis in this model to show the effects of changes in the other parameters on the desired level of conservation tillage.

2. Additional Considerations

The planner's problem shown above neglects an important consideration that may influence the pattern of adoption of conservation tillage that the agency wishes to encourage. One of the impacts of soil erosion is to reduce the agricultural productivity of land. That is, conservation tillage yields benefits to the agricultural producer in that it allows him to avoid the costs of foregone production that are imposed by soil erosion. The producer, however, may not take full account of this in his production decisions due to tenure status or misperceptions of erosion (Hinman et al. 1983; Gould et al. 1989). In addition, it would not be correct simply to fold this factor into the cost term $C_1(T)$, since a reduction in the intensity of tillage at time t yields benefits to the producer over all future time periods.

Instead, an appropriate agency problem that would capture this consideration would be:

$$\underset{T_t}{\text{Max}} \int_0^{\infty} \{[f(R(T_t),S_t) - C_1(T_t) - C_2(Z(T_t)) - C_3(E_t)]e^{-rt}dt\} \tag{26}$$

where: E_t = cumulative erosion, or soil loss, at time t;

$$E(\bar{t}) = \int_0^t g(T_t)dt \;, \; g_T < 0;$$

$g(T_t)$ = rate of erosion at time t;

$C_3(E_t)$ = total cost incurred at time t from lost agricultural production due to cumulative soil loss,

and where maximization of (26) would be subject to the same constraints as before plus an additional one:

$$\dot{E}_t = g(T_t) \tag{27}$$

With this objective function, the condition which maximizes the Hamiltonian with respect to T (for an interior solution) would differ from (19) only by the term denoting the addition to cumulative erosion:

$$f_T - C_{1T} - \delta C_{ZZ} + \gamma\lambda_1 + \lambda_3 g_T = 0 \tag{28}$$

where: λ_3 is a multiplier associated with the new constraint.

Such a framework could allow the agency to take account of the dynamic effects of lost productivity due to soil erosion. This would be most important for cases in which farmers do not perceive or take account of the full cost of foregone future productivity.

Lastly, uncertainty associated with parameter values clearly is a defining characteristic of the problems posed above. Sensitivity analysis therefore would be an important component of an attempt to simulate a dynamic model of tillage choice for a given site. Alternatively, uncertainty could be introduced explicitly by using a stochastic model of optimal control (e.g. Pindyck 1980; Kamien and Schwartz 1981).

C. Possibilities for Tailored Incentives

For any given source, there generally are large knowledge gaps regarding the kinds of parameters and functions featured in the conceptual approaches above. Furthermore, the expected values of and uncertainties associated with key parameters and functions vary appreciably across geographic regions and crop types. Policy clearly needs to account for such variation when addressing the tension between abatement practices for different pollution pathways. An important question, then, involves how this might be possible given constrained data on several counts and a limited understanding of pollution fate and transport processes, particularly in the subsurface.

Ideally, of course, planning agencies should like to implement a bundle of instruments that would bring about a least-cost movement to the "optimal" levels of different categories of pollution. In a less than ideal world, the agency might hope to develop instruments that would produce “charges and standards” results (Baumol and Oates 1975) for multiple pollution categories. These could consist, for example, of simultaneous instruments designed to achieve predetermined environmental quality changes through soil consecration (as a proxy for that class of surface runoff problems positively correlated with erosion) as well as chemical input use reductions.

In developing incentives for the control of multiple pollutants and pathways, a planning agency need not be concerned with tailoring the incentives according to producers’ private costs and benefits of abatement, since producers will account for those factors in deciding how to respond to incentives. The key lies in accounting for variation across areas, producers and crops in the external effects of BMPs that policies encourage. If incentive (fee, subsidy or permit) schemes were implemented simultaneously for both pesticide use and tillage practice, then the total costs of reducing environmental damage would be lowered by varying the incentives spatially according to area-specific parameters and functions such as α, β, δ, R(T), and f(R,S). (In an expanded model allowing for the "containment" of pollution in addition to the reduction of discharges (Braden et al. 1989), the total costs of damage reduction could be lowered even further.)

Given limited information, a practical approach to the development of tailored incentives might involve identifying a small number of specific ranges into which key parameter values may fall. Then, an agency could proceed to build a taxonomy that identifies, by crop and spatial location, the expected place that each key parameter is thought to occupy in the classification. For some key parameters, the information necessary to characterize their expected ranges is already available. For example, good information by location is available for the variables of the Universal Soil Loss Equation. This means that though the precise specification of runoff as a function of tillage may be difficult, one certainly can draw general conclusions about the way in which the relationship varies across areas and producers. Information on other factors, of course, is less available. The effect of different agricultural production practices on water quality, for example, is not well understood at present. Research has been underway on these factors, and plans for new studies currently are being developed by the U.S. Geological Survey and the U.S. Department of Agriculture (Burkart et al. 1990). This kind of research should enhance the base of knowledge regarding spatial variation in factors such as δ, α and β and the marginal damages associated with the discharge of pollutants from cropland.

Even in the presence of incomplete information on factors such as pollutant fate and transport in groundwater and the effect of farming practices on various discharges, it is possible with current knowledge to make general distinctions among areas. This is demonstrated quite well by Crutchfield et al. (1991) through their classification of the vulnerability of groundwater to pesticide and nitrate leaching from cotton production in different states. Their work estimates the percentages of cotton cropland in the major producing states that fall into four distinct categories of vulnerability to pesticide leaching, running from “most vulnerable” to “little or no likelihood” of leaching. The same is done for nitrates, with three categories corresponding to high, moderate, and low vulnerability. These kinds of estimates could provide useful input to tailor the magnitudes of incentives for BMP adoption according to the agricultural production and environmental characteristics of different geographic areas.

III. SUMMARY

A policy designed to decrease the use of an agricultural input that causes pollution can lead farmers to alter their management practices and thus the overall input mix. This may lead in turn to an increase in the discharge of pollutants different from those targeted by the policy. One example involves substitution between tillage operations and the application of pesticides. A policy to decrease pesticide pollution by lowering the rate of pesticide application may cause an increase in erosive tillage practices and thus soil loss and associated surface runoff. Alternatively, a policy designed to increase conservation tillage may yield higher damages from pesticides.

This paper has explored conceptual approaches to the management of agricultural nonpoint-source pollution that take account of substitution between tillage operations and pesticides. Under the simple objective of reducing the total discharge of pesticides in an area by a given amount, the least-cost allocation of abatement changes if input substitution is accounted for and a constraint on surface runoff is imposed. The allocation of pesticide reductions would change according to farm-specific factors such as soil erodibility and productivity, rainfall, cropping, farmland slope, severity of pest problems, the potential and cost of "integrated pest management," farmer perception of soil erosion, and farm operator tenure status. This paper also presented a dynamic model of a planning agency's choice of tillage at the farm level that accounts for the potentially long-term damages that may result when pesticides leach into groundwater. The approach illustrates the tradeoff between reducing surface runoff of pollution and the leaching of pesticides, and shows how cross-farm variability in key parameters would influence the desired degree of adoption of conservation tillage.

In the conceptual approaches of this paper, it is assumed that farmers respond to an instrument targeted at an agricultural input by altering other inputs so as to maintain constant per-acre yields. Though some data on agricultural practices suggest that farmers may attempt this, a more realistic approach would relax this assumption to allow for a decline in yield. Useful further work also would include explicit treatment of uncertainty regarding parameters; consideration of a range of pesticides with varying effectiveness, toxicity and persistence, among which farmers may choose; and an exploration of the impact of integrated pest management techniques on the extent of input substitution.

Since tillage practices are observable and disincentives for pesticide use may be applied at the time of purchase, instruments targeted directly at these inputs are relatively practical. Empirical application of conceptual approaches for even a few agricultural sites could help to determine a ranking of priorities for fine tuning a package of instruments according to local agricultural and environmental factors. For a given set of multiple environmental objectives, one could determine how relatively sensitive an efficient solution is to variation in different parameters and functions (e.g. β vs. $R(T)$), and thus identify the most critical "driving" characteristics upon which the tailoring of instruments might be based. This in turn would increase the efficiency of pollution control efforts by directing future research toward those parameters that are found to be most important.

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(preliminary version)

NON POINT SOURCE POLLUTION CONTROL , INFORMATION ASYMMETRY, AND THE CHOICE OF TIME PROFILE FOR ENVIRONMENTAL FEES

by

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1. INTRODUCTION

Water quality management specialists have long ago emphasized the practical difficulties of relying on “conventional” end-of-pipe control techniques when dealing with non-point pollution (NPP) problems, and, hence, the need to prevent pollutant loads as far as possible.

The application of such a “preventive approach” through effective and possibly efficient, regulatory schemes, may, however, involve number of problems, which to a large extent arise from the difficulty, and sometimes the technical impossibility, of monitoring non-point emissions at source. This may be due either to the mode of conveyance of pollutant flows, or to the intermittent nature of emissions or to the fact that pollutants originate over a broad area [Vigon, 1985]. The relative role played by each of these factors in preventing the monitoring of emissions on a continuous and widespread basis may vary according to the specific pollutant at hand.

Due to the difficulty of applying *sic et simpliciter* emission-based policy instruments when dealing with NPP, attempts have been made to find alternatives, with respect to actual emissions at source, as a basis for establishing regulatory schemes. In this respect, basically two recommendations can be found in economic literature dealing with NPP control.

The first consists of selecting incentives defined with respect to ambient pollutant levels, According to its proponents, the proposal would have the attractive property of allowing regulatory bodies to rely on "...an incentive mechanism based on the observable variable (ambient pollutant levels) to induce certain unobservable actions [pollutant abatement at source]" [Segerson, 1988, P.89].

The second policy strategy, which may be termed an "indirect approach", suggests regulatory bodies should grant political legitimacy to NPP mathematical models which make predictions about either emissions at source or ambient pollutant levels, and hence define appropriate incentives accordingly. As the proponents state, ". . . while such models will never provide [..] a perfect substitute for accurate monitoring of actual flows, they can serve as an important tool for diminishing the uncertainty about nonpoint loadings . . . . furthermore, predictions obtained from such models offer an alternative to actual flows as a basis for the application of policy instruments [Shortle-Dunn, 1986, p.668].

The two above-mentioned approaches will be briefly reviewed and commented on in section 2. While there are no a *priori* decisive arguments in favor of one or the other, we suggest that the "indirect" one should be preferred whenever there are no indications that the suspected polluters possess better information about the "technology" of pollutant abatement. Or, more generally, whenever it is believed that the cost of acquiring information about the implications of productive decisions in terms of ambient pollutant levels are prohibitively high for the private economic agents involved.

It is worthwhile stressing, however, that granting political legitimacy to a NPP mathematical model does not constitute, *per* *se,* a panacea. In fact, from a regulatory point of view, it simply implies that the availability of adequate information about the parameters needed to feed the model, rather than monitoring of actual emissions, becomes the key issue when establishing policy instruments.

Such parameters are usually represented by productive decisions taken by suspected polluters, such as the use of (potentially) polluting inputs (eg. nitrogen fertilizers) and the physical characteristics of the production site (eg. soil water retention capacity). In addition, models conceived as tools for providing not only estimates of potential pollutant flows but also pollutant transport rates, typically require information about the hydrological structure of the watershed surrounding the water body which is thought to receive a fraction of estimated emissions at field level.

Acquiring such information may not constitute a serious problem for regulatory bodies operating in Countries or regions with long-standing traditions of land classification and where management practices are monitored on a continuous and widespread basis. However, in Countries which do not have such traditions the application of the “indirect approach” may be problematic and lead to unsatisfactory results, unless regulatory bodies are either prepared to invest resources in collecting the required information directly, or to “extract” it from suspected polluters through appropriately defined incentive mechanisms.

The paper concentrates on the application of the policy strategy which has been referred to as the “indirect approach”. In particular, we try to make a step forward with respect to a previous work where we attempted to provide, through a static, discrete “adverse-selection” framework, a broad characterization of NPP incentive schemes when suspected polluters possess private information about their production site’s physical characteristics (Dosi and Moretto 1990). Here, on the other hand, we shall assume that at the time when the regulatory scheme is designed, the suspected polluters do not possess such private information. However, the assumption that the relevant physical characteristics of the production site (“soil quality”) do not vary over time, will be relaxed,

In fact, it appears more realistic to assume that a number of physical characteristics affecting the extent of pollutant loads (as well as, often, suspected polluters’ productive performance) may vary over time. Changes may either occur because of “exogenous shocks” or because of actions taken by firms or both. In the paper we concentrate on non-monitorable actions undertaken by firms (“maintenance decisions”), but it will be assumed that the “maintenance technology”, known by both parties, is affected by a certain degree of uncertainty.

Furthermore, we account for the possibility that, even if the social planner (hereafter the agency or the (p)rincipal) perceives the existence of detrimental externalities due to the presence of unregulated non-point pollutant sources, he might consider the opportunity of delaying the introduction of “environmental fees”. Such fees will be assumed to take the form of an increase in the market price of the variable input which is believed to contribute to emissions, and, hence, water contaminations with an intensiveness which depends on the production site's physical characteristics. Moreover it will be assumed that suspected polluters grant enough credibility to the agency's announcement concerning the time profile of environmental charges.

The implications of such a delay as well as its optimal choice characterization constitute the paper's central issues.

Section 3 explores the implications of the announcement of alternative delays upon management decisions adopted by the suspected polluter(s) -hereafter the firm or the (a)gent)- and on environmental damage. In the same section we also analyze the action of uncertainty about future realizations of the soil quality index, with regard to the firm's maintenance pattern and its "market value", as well as with regard to the consequent (expected) environmental damage.

Section 4, on the other hand, will be devoted to the optimal choice characterization of the time profile for environmental fees, by assuming the perspective of a utilitarian agency which, over the entire planning horizon, takes care of (expected) estimated social environmental damages as well as of the firm's welfare, and receives a utility from tax collection. The optimal choice will be derived by looking for a "perfect equilibrium" within a two player game in which the firm chooses management practices after observing the agency's time profile decision. The equilibrium is then obtained by working backward: the agency foresees that the firm will react optimally to whatever time delay for environmental fees is announced. That is, the agency should solve the firm's optimization problem before taking its own decision.

2. THE RATIONALE FOR AN "INDIRECT" NPP CONTROL POLICY APPROACH

2.1 Before turning to the paper's main issues, let us briefly review the two general NPP control policy strategies mentioned in the previous section.

As a point of reference, let us start by assuming that the firm's fixed-capital output per unit (say bushels of corn per acre) is given by the following production function:

[1] $$Q = Q(\theta, x)$$

where x represents a (potentially) polluting variable input (eg. chemical fertilizer) and θ represents an index for fixed-capital quality (eg. soil water retention capacity).

Let us distinguish between (unobservable) pollutant emissions at source, R, and pollutant levels actually found in a given water body, P. Assume, for the time being, that R depends only on x, R= R(X), P is linked to R by a one-to-one relationship, P = P(R), and social damages associated with P are evaluated according to D = D(P).

Given such relationships, a variety of policy options are, at least theoretically, open to the agency. Such options range from restrictions on the permissible x level the firms will be allowed to choose, to incentives defined over observable ambient pollutant levels. Placing an incentive over P (or R, since it may be easily inferred from observing P), however, appears to be the most attractive option from an "administrative" point of view. Moreover, if the firm possesses better information about the "technology" of pollutant abatement, placing an incentive over P (or R) should induce the selection of cost-minimizing pollutant abatement strategies.

However, if this condition is not met, or, more generally, if the firm does not possess a priori information about R(.) and P(.), the agency should transmit all the relevant information, but in this case we do not see any reason why the agency should not "convey" such information either through appropriately defined incentives over x or through mandatory measures.

Whether or not defining standards in terms of permissible x will provide results allocatively equivalent to management practice incentives will depend on whether or not the agency possesses adequate information about θ, a state variable which, for the time being, we assume only affects the firm'smarketable output. Setting aside transaction costs, if the parties share the same information about θ, the two policy instruments would lead to allocatively similar results, since an optimal incentive scheme over x should induce a profit maximizing firm to choose the same variable input level which the agency would choose as a management practice standard. However, if the firm has better information about θ when the agency implements the policy, such equivalence breaks down, and the pricing mechanism is preferable to standards since it has the advantage of relieving the principal from the problems posed by the definition of a standard on x in conditions of uncertainty regarding θ.

To summarize, two considerations follow from the stylized “technical” relationships which have been assumed up to now.

Firstly, if the firms do not possess a *priori* information either on R(.) or P(.), whilst the agency does, a regulatory approach directly defined over the production decisions affecting pollutant flows appears more appropriate than defining incentives over actual ambient pollutant levels (or on R), since the theoretical advantages of assigning the firms the role of choosing the best abatement strategy can not be exploited.

Secondly, if the firms possess private information about θ, management practice incentives appear to be allocatively superior to management practice standards. According to Shortle and Dunn [1986], in conditions of imperfect monitorability of the firms' “typology”, such incentives would not only outperform all the alternative policy instruments but, at least in the special case of a single (suspected) polluter, they would lead to achievement of a first-best solution: a result which, however, if our interpretation is correct, crucially depends on the assumption that θ does not enter R(.) (or P(.)).

2.2 Let us now modify the technical relationships p(.) and R(.) so as to make them a little closer with “reality”.

First of all, let us specify P(.) in a way which formalizes the NPP attribute consisting of the difficulty ofinferring, emissions at source, without errors, from observable ambient pollutant levels. This can be done either by incorporating in the argument a random variable representing the imperfect knowledge about pollutant transport mechanisms:

$$P = P(R,\tau)$$

or by allowing for the existence of multiple sources:

$$P = P(R_1, \ldots\ldots\ldots, R_n)$$

or both:

$$P = P((R_1,\tau), \ldots\ldots\ldots, (R_n,\tau))$$

How does recognition of the existence of a more complex relationship between ambient pollutant levels and each source's emissions affect the conclusions previously drawn when assuming to deal with a one-to-one relationship?

A first consideration should be made with regard to the need to distinguish between emissions at source and ambient pollutant levels. If the agency wishes to improve social welfare, and not to reduce emissions as such, accounting for a more complex and articulated relationship between R and P clearly emphasizes the need to implement policy instruments which take ambient pollutant levels rather than emissions at source as their point of reference. In this respect, we entirely share the view expressed by Segerson [1988], according to whom economic incentives concentrating on the latter tend to ignore "...the important distinction between "discharges" and the resulting pollutant levels which determine damages" [p.87].

However, whether or not this objective may be better accomplished by relying on incentives defined on actual ambient levels or through management practice incentives defined according to NPP mathematical models (providing predictions about P) will depend, again, on how plausible we believe is the assumption that private agents possess, or may easily acquire, information about the implications of their management practices in terms of ambient pollutant level. It is clear, however, that the more complex the P(.) relationship is, the less this assumption appears to be plausible.

Further complications arise if we assume that also the relationship between the firm's productive decisions and emissions at source is more complex that the one assumed up to now. For example, R might be influenced not only by use of the potentially polluting input as such, but also by the physical characteristics of the site in which x is used, i.e.

$$R = R(x,\theta)$$

Incorporating θ in the argument of R(.) appears, in fact, more consistent with the technical literature dealing with NPP, which suggests that climatic, pedological and topographic parameters may play an important role in determining the extent of pollutant quantities potentially affecting surface and underground water quality.

Unfortunately, once a more complex P(.) function is combined with a more complex relationship between R and management practices, it becomes even more difficult to share the optimism expressed, for example, by Segerson [1988] that, "... since firms are in a better position to determine the abatement strategy that will be most effective for them", the selection of incentives defined over actual ambient pollutant levels would ensure that "...any given level of abatement is achieved at the lowest possible cost" [Segerson, 1988, P.86].

This optimism may be justified if we assume we are dealing with relatively "simple" and, at least in certain Countries, long experienced phenomena such as erosion, but is less convincing when dealing with inherently more complex, and less “perceptible”, pollution phenomena such as nitrate and pesticide leaching.

Moreover, it should be pointed out that the difficulties met by individual polluters in identifying the relationship between their productive decisions and the level of pollution appearing in (often distant) water bodies, and, then, in conjecturing their own responsibility, increase with the number of sources: in fact, even with respect to the same pollutant, within the same watershed point and non-point sources may be contemporaneously present, and, among the latter, differentiated urban, industrial and agricultural polluting activities. This is, for example, the case of the watershed surrounding the Venice lagoon, a water body in which worrying phenomena of algae-bloom have occurred repeatedly in recent years: this densely populated area of about 180,000 hectares; located in Northern Italy, presents an extraordinary mix of industrial and agricultural activities which, together with the urban sector, emit quantities of nitrate and phosphorus estimated, respectively at approx. 9,000 and 1,300 tons per year [Regione del Veneto, 1989].

In such conditions, it appears, in our view, more suitable to rely on what has been termed the"indirect approach", and hence convey information to firms about NPP mathematical model-based ambient pollutant predictions through appropriate incentives directly placed on management practices.

2.3 The paper thus focuses on this policy approach, and concentrates on some issues related with its adoption.

We assume that at the time when the regulatory framework is decided, firms do not hold private information about the production site's physical parameter(s), θ, entering the NPP mathematical model which has been granted with political legitimacy. It will be assumed, however, that the soil quality index, characterizing each production unit, may vary over time. In fact, if the agency is assumed to take a sufficiently long planning horizon, accounting for the possibility that θ will evolve with respect to its initial status, appears more realistic than assuming it remains invariable.

Let us take the example of "nitrate emissions" from cultivated soils, a phenomenon often considered, at least in EEC Countries, as one of the most relevant problems among NPP. Technical literature suggests that "discharges" are undoubtedly positively correlated with fertilizer use; however, leaching of available nitrates may significantly increase due to high rates of water movement through the soil. In turn, high water movement may be due to (more or less) unpredictable heavy rainfall conditions which cannot be prevented. However, farmers may contribute to reducing very high water movement, for example, by takingactions designedto maintain the soil's organic content, since soils rich in terms of organic matter have, generally, relatively high water retention capacity, and are therefore liable to experience lower losses of available nitrates [OECD, 1986]. Again, however, the performance of such actions in terms of maintaining (or not depleting, or increasing) organic content, and, then, the consequences in terms of final ambient pollutant levels, may be affected by a certain degree of uncertainty) depending on a number of (more or less) unforeseeable factors [Regione Veneto, 1990].

On the grounds of this example, two considerations are in order.

Firstly, it would be inappropriate to disregard the possibility of a variation over time of the soil quality index θ which enters the NPP mathematical model upon which ambient pollutant predictions are based, and, hence, on which regulatory schemes are defined. To go back to our example, whether this index refers to “soil water balance” or to “organic content of the soil”, θ is unlikely not to vary over time.

Secondly, θ may vary both because of exogenous “shocks” and actions undertaken by firms. It follows that, even if we concentrate on the latter, it appears convenient to assume that “maintenance decisions” are undertaken in conditions of uncertainty about future realizations of thesoil quality index which the firm (the agency) wishes to alter in order to improve its profits (social welfare).

An attempt to deal formally with such issues is made in the paper. It is assumed that the firms’ “maintenance expenditures” are not monitorable by the agency, but that both parties share the same information about maintenance technology and uncertainty with regard to future realizations of θ. The assumption of identical information about the structure of the maintenance technology function appears not too unrealistic, since agencies themselves may provide the firms with all the relevant information they possess about the possible performance of maintenance actions.

As far as the “form” of uncertainty is concerned, θ is assumed to move randomly in continuous time according to the following stochastic differential equation:

[2] $$d\theta_t = \left[f(\theta_t, m_t) - \delta \right]\theta_t dt + \sigma\, \theta_t dz_t$$

where f(.) stands for the effect of maintenance expenditure, m, δ is a constant soil quality "depreciation" rate, and dz is the increment of a Wiener process, or Brownian motion, with zero mean and unit variance (i.e. $dz_t = \varepsilon_t \sqrt{dt}$, while ε is a serially uncorrelated and normally distributed random variable)[1].

Equation [2] implies that the future realizations of θ are uncertain with a variance which grows linearly with the time horizon, Thus, although information is obtained over time, future soil quality status is always uncertain to the firm.

We assume that:

$f_m > 0$ (i.e. maintenance expenditure has a positive influence on θ),

$f_{mm} < 0$ (i.e. this influence diminishes as m increases),

$f_\theta < 0$ (i.e. for a given amount of maintenance expenditure the improvement of "low quality soil" is greater than for "high quality soil"),

$f(0,\theta) = 0$,

$\delta > 0$ (i.e., if the firm decides not to spend money on maintenance, the expected value of θ deteriorates at the constant exponential rate δ).

It is assumed that the firm wishes to maximize its "market value", i.e. its discounted (expected) cash flows over the planning horizon $[0, \infty)$. According to [1] and [2], and setting output price equal to one, the firm's objective function in the absence of public intervention is described by:

$$E_0\left\{ \int_0^{\infty} \left[Q(x_t,\theta_t) - \omega\, x_t - m_t \right] e^{-rt} dt \right\}$$

where ω indicates the input market price faced by a competitive, representative firm and r is a constant discount rate,

As, according to our hypothesis, the agency might announce the decision to delay the introduction of “environmental fees”, the firm’s objective function with regulation becomes:

$$[3] \quad V(\theta_0;T) = E_0\left\{ \int_0^{T} \left[Q(x_t,\theta_t) - \omega\, x_t - m_t \right] e^{-rt} dt + \int_T^{\infty} \left[Q(x_t,\theta_t) - \omega\, x_t - D(P(\theta_t))x_t - m_t \right] e^{-rt} dt \right\}$$

where T represents the time lag “granted” to firms before introducing a tax which is assumed to take the form of an increase in the price of the variable input x. The amount of this increase will depend on the social damage, D(P), attributed to ambient pollutant levels, evaluated according to a mathematical model, $P(\theta)$, which is assumed to provide variable input (x) predictions per unit.

3. THE EFFECT OF TIME PROFILE ON FIRM'S MANAGEMENT PRACTICES AND THE ROLE OF UNCERTAINTY

3.1 Let us assume the production function [1] has the following properties:

$$[4]\qquad \begin{cases} Q_x > 0\ , & \text{for}\quad x \le \bar{x} \\ Q_x < 0\ , & \text{for}\quad x > \bar{x} \\ Q_{xx} \le 0\ ,\ Q_\theta > 0\ ,\ \bar{x}_\theta > 0 & \\ Q(0,\theta) = Q(x.0) = 0\ ,\quad Q_x(0,\theta) = \infty & \end{cases}$$

Since x may be freely adjusted, the firm's optimal variable input level can be derived from the usual first order condition:

$$[5]\qquad \begin{aligned} &Q_x - \omega = 0 && \text{for}\quad 0 \le t < T \\ &Q_x - \omega - D = 0 && \text{for}\quad t \ge T \end{aligned}$$

If we set, for simplicity, $\omega = 0$, according to [4] and [5] the optimal input level will be[2]:

$$[6]\qquad \begin{aligned} &x^{*}_{(a)t} = \bar{x}(\theta_t) && \text{for}\quad 0 \le t < T \\ &x^{**}_{(a)t} = Q_x^{-1}[D(\theta_t)] && \text{for}\quad t \ge T \end{aligned}$$

where, for a given θ_t, $x^{*}_{(a)t} \ge x^{**}_{(a)t}$.

BY substituting [6] in [3] the latter reduces to:

[7] $$V(\theta_0;T) = E_0\left\{ \int_0^T \left[Q(\bar{x}(\theta_t), \theta_t) - m_t\right]e^{-rt}dt + \right.$$

$$\left. + \int_T^{\infty} \left[Q[Q_x^{-1}(D(\theta_t)),\theta_t] - m_t - D(\theta_t)Q_x^{-1}(D(\theta_t))\right]e^{-rt}dt \right\}$$

To keep the problem mathematically tractable, we shall assume that:

[8]
$$\begin{aligned}
Q(x,\theta) &= h(\theta)x^{\alpha} &&, \quad 0 < \alpha < 1 \\
h(\theta) &= \theta^{\nu} &&, \quad \nu > 0 \\
D(P(\theta)) &= \theta^{-\beta} &&, \quad \beta > 0 \\
\bar{x}(\theta) &= \theta^{\psi} &&, \quad \psi > 0 \\
f(\theta,m) &= m^{\xi}\,\theta^{-\gamma} &&, \quad 0 < \xi < 1, \quad \gamma > 0
\end{aligned}$$

Then the firm's maximization problem becomes:

[9] $$\max_m V(\theta_0;T) \equiv E_0\left\{ \int_0^T \left[\theta_t^{\varphi} - m_t\right]e^{-rt}dt + \int_T^{\infty}\left[C_{(a)}\theta_t^{\phi} - m_t\right]e^{-rt}dt\right\}$$

where $C_{(a)} = (1-\alpha)(\alpha)^{\alpha/(1-\alpha)} < 1$

$\phi = \frac{1}{1-\alpha}\nu + \frac{\alpha}{1-\alpha}\beta$

$\varphi = \nu + \alpha\psi$

The maximization is subject to equation [2], the constraint $m \geq 0$, and θ_0 is given. Moreover we assume that the sample path of {z,} contains all the information relevant to the firm's problem, and $E_0\{\cdot\}$ denotes conditional expectation taken, at time zero, over the distribution of $\{z_t\}$ and $\{\theta_t\}$ processes. While the former is exogenous to the firm's problem, the latter is determined endogenously by the optimal maintenance pattern.

According to [9], the firm's maximization problem can be set as a two-stage optimal control problem, where the integral assumes different forms in each stage. In the second stage the firm maximizes its expected discounted cash flows, defined as the difference between "operational profits" and environmental fees. Then, in the first stage, the firm will maximize its discounted operational profits, with the constraint that at time T the firm's market value will coincide with the (discounted) value calculated in the second stage.

Let us then solve the optimal control problem at II-stage, formally expressed as follows:

$$[10] \quad \max_{m} e^{-rT} V^{II}(\theta_T) \equiv e^{-rT} E_T\left\{ \int_T^{\infty} \left(C_{(a)}\theta_t^{\phi} - m_t\right) e^{-r(t-T)} dt \right\}$$

The maximization is subject to equation [2], m $\geq$ 0, and θ_T given. If the firm's maximum market value at the II-stage is differentiable, then $V^{II}(.)$ has to be a solution of the following dynamic programming equation:

$$[11] \quad rV^{II} = \max_{m} \Big[\left(C_{(a)}\theta_t^{\phi} - m_t\right) + V_{\theta}^{II} \left(m_t^{\xi}\, \theta_t^{-\gamma} - \delta\right)\theta_t + $$
$$+ \frac{1}{2}\sigma^2\, \theta_t^2\, V_{\theta\theta}^{II} \Big]$$

where V_{θ}^{II} and $V_{\theta\theta}^{II}$ are derivatives of $V^{II}(.)$ with respect to 8.

Equation [11] is the Hamilton-Jacobi-Bellman equation of the stochastic version of the optimal control theory. By differentiating the r.h.s of [11] with respect to m_t , we obtain:

$$[12] \qquad m_t = \left(\xi \, V_\theta^{II} \, \theta_t^{1-\gamma} \right)^{1/(1-\xi)} \qquad \text{for} \quad t \geq T$$

which represents the first order condition for optimality of the firm's maintenance pattern.

Equations [11] and [12] together can be expressed as a non-linear second order differential equation of parabolic type in V^{II}. As pointed out, for example, by Freedman (1964), Merton (1975), such a differential equation, in general, can not be solved explicitly. However, if some restrictions on the coefficients of the production, damage, and maintenance technology function are imposed, it may be possible to find a solution in a closed analytical form. In particular, if we set[3]:

$$\xi = \frac{1}{2} \quad , \quad \gamma = \frac{1}{2}\phi$$

the solution for the firm's market value is (see appendix A):

$$[13] \quad V^{II}(\theta_t) = M^{II}\theta_t{}^{\phi} \qquad \text{for } t \geq T$$

where
$$M^{II} = \frac{B - \sqrt{B^2 - 4AC_{(a)}}}{2A}$$

$$A = \frac{1}{4}\phi^2$$

$$B = r + \phi\delta - \frac{1}{2}\phi(\phi-1)\sigma^2 \quad > 0$$

$$C_{(a)} = (1-\alpha)(\alpha)^{\alpha/(1-\alpha)}$$

According to [13], the firm's optimal market value at the

II-stage is an increasing function of the state variable θ with elasticity equal to ϕ, which, in turn, depends on the production function's parameters α and ν, and on β, the elasticity of the social damage function with respect to θ.

From [13] the optimal maintenance policy can be derived:

$$[14] \qquad m_t = \frac{1}{4}\phi^2 \left[M^{II}\right]^2 \theta_t^{\phi} \qquad \text{for} \quad t \geq T$$

which implies the stochastic differential equation [2] reduces to:

$$[15] \qquad d\theta_t = \left[\frac{1}{2}\phi\, M^{II} - \delta\right]\theta_t dt + \sigma\, \theta_t dz_t \qquad \text{for} \quad t \geq T$$

From [14], the optimal maintenance policy at the II-stage depends on the current realization of the state variable θ. Since, in turn, θ is described by a stochastic process, also m will be a stochastic process. In other words, the firm can not decide on maintenance expenditure before looking at the "performance" of θ achieved through past maintenance decisions.

We can now examine maximization of the firm's market value at the I-stage, on condition that it coincides, at time T, with the discounted value described by [13], which, in this optimal two-stage control problem, takes on the sense of "scrape-value". Formally the firm's expected discounted profit or loss of at time zero, described by [9], becomes:

$$[16] \qquad \max_{m} V(\theta_0;T) \equiv E_0\left\{ \int_0^T \left(\theta_t^{\varphi} - m_t\right) e^{-rt}dt + e^{-rT} M^{II}\theta_T^{\phi} \right\}$$

Now the firm faces the problem of maximizing [16] by choosing a maintenance expenditure policy in the interval [0,T) with a terminal constraint at time T. In our case, for $\theta \geq 0$ and a generic $0 \leq t < T$, [16] can be rewritten as:

$$[16] \quad \max_{m} V(\theta_t, t; T) \equiv e^{rt} E_t \left\{ \int_t^T \left(\theta_s{}^{\varphi} - m_s \right) e^{-rs} ds + e^{-rT} M^{II} \theta_T{}^{\phi} \right\}$$

Hence $V(\theta_t, t; T)$ is the maximum profit at time t if the soil quality index at that time is θ multiplied by e^{rt},

Again the maximization is subject to equation [2], $m \geq 0$, and θ_t given. The procedure for solving [16'] is the same as that used for the II-stage. In other words, if the market value function of the firm V is differentiable, then $V(\theta_t, t; T)$ has to be a solution of the following dynamic programming equation:

$$[17] \qquad -V_t + rV = \max_{m} \left[\left(\theta_t{}^{\varphi} - m_t \right) + V_\theta \left(m_t^{\xi} \, \theta_t^{-\gamma} - \delta \right) \theta_t + \right.$$

$$\left. + \frac{1}{2} \sigma^2 \theta_t^2 V_{\theta\theta} \right] \qquad , \text{ for } \; 0 \leq t \leq T$$

with the following constraints:

$$V(\theta_T; T) = M^{II} \theta_T{}^{\phi}$$

$$V(0; t) = 0$$

where V_t is the partial derivative of V(.) with respect to t.

By differentiating the r.h.s of [17] with respect to m, we get:

[18] $$m_t = \left(\xi\, V_\theta\, \theta_t^{1-\gamma}\right)^{1/(1-\xi)} \qquad \text{for} \quad 0 \le t < T$$

Again, to obtain a solution in a close analytical form, we need to impose some restrictions on the "technical" coefficients. In particular, if we set[4]:

$$\xi = \frac{1}{2}, \qquad \gamma = \frac{1}{2}\varphi \left(\equiv \frac{1}{2}\phi\right)$$

the solution for the firm's market value at I-stage is (see appendix 1):

[19] $$V(\theta_t,t;T) = M(t;T)\,\theta_t^{\varphi} \equiv M(t;T)\,\theta_t^{\phi} \qquad , \text{for} \quad 0 \le t < T$$

where:

$$M(t;T) = \frac{M_2^I - M_1^I \left(\dfrac{M^{II} - M_2^I}{M^{II} - M_1^I}\right) \exp\left(\sqrt{B^2 - 4A}\,\right)(T-t)}{1 - \left(\dfrac{M^{II} - M_2^I}{M^{II} - M_1^I}\right) \exp\left(\sqrt{B^2 - 4A}\,\right)(T-t)}$$

$$M_1^I = \frac{B - \sqrt{B^2 - 4A}}{2A} \quad , \quad M_2^I = \frac{B + \sqrt{B^2 - 4A}}{2A}$$

It is easy to show that $M(T;T) = M^{II}$ and the following limits hold:

$$\lim_{T\to\infty} M(t;T) = M_1^I$$

$$\lim_{T\to 0} M(t;T) = M^{II}$$

with $M_1^I \le M^{II}$ as indicated in fig.1. In addition, since

$\frac{M^{II} - M_2^{I}}{M^{II} - M_1^{I}} > 0$ (see appendix B) if the introduction of environmental fees were postponed forever, the firm's maximum expected value would be reached at t = 0, and would decrease over time. On the other hand, if no delay were conceded, the firm's maximum value would be that obtained in the II-stage solution.

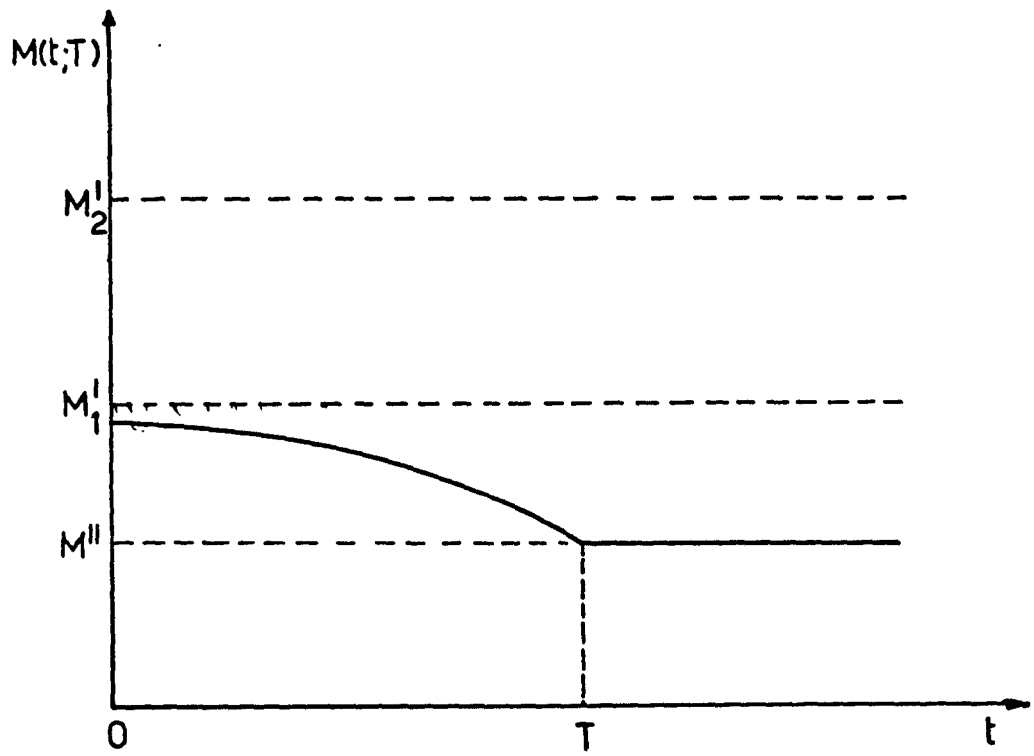

fig.1

The optimal maintenance expenditure pattern during the period preceding introduction of the environmental fees can be derived from [19]:

$$[20] \qquad m_t = \frac{1}{4} \phi^2 \left[M(t;T) \right]^2 \theta_t^{\phi} \qquad \text{for} \quad 0 \le t < T$$

while the stochastic differential equation [2] reduces to:

[21] $$d\theta_t = \left[\frac{1}{2}\phi\, M(t;T) - \delta \right]\theta_t dt + \sigma\, \theta_t dz_t \ , \quad \text{for} \quad 0 \le t < T$$

Again, m_t appears to be a stochastic process, and the firm can not decide on the optimal maintenance expenditure in advance before looking at the current realization of the state variable θ.

3.2 On the basis of the results obtained above, let us now explore: (i) the relationship between the time profile announced by the agency, T, and the expected maintenance expenditure pattern; (ii) the relationship between T and expected total damage; (iii) the action, at equal T, of uncertainty with regard to future realizations of soil quality parameter θ on the (expected) maintenance expenditure, the firm's market value and environmental damage.

3.2.1 Since m is a stochastic process, the expected value of its rate of variation can be derived by applying the **Itô's** Lemma to [20]:

[21] $$\frac{1}{dt} E_t\left(\frac{dm_t}{m_t}\right) =$$

$$= \begin{cases} 2\dfrac{\dot{M}(t;T)}{M(t;T)} + \phi\left[\dfrac{1}{2}\phi\, M(t;T) - \delta\right] + \dfrac{1}{2}\phi(\phi-1)\sigma^2 \ , & 0 \le t < T \\ \phi\left[\dfrac{1}{2}\phi\, M^{II} - \delta\right] + \dfrac{1}{2}\phi(\phi-1)\sigma^2 \equiv r - \sqrt{B^2 - 4AC_{(a)}} \ , & t \ge T \end{cases}$$

By solving the differential equation [21] taking the expectation at time zero, we obtain:

$$
[22] \quad E_0(m_t) = \begin{cases} m_0 \exp\left(\int_0^t 2\frac{\dot{M}(s;T)}{M(s;T)} + \phi\left[\frac{1}{2}\phi\, M(s;T) - \delta\right] + \frac{1}{2}\phi(\phi-1)\sigma^2 ds\right) & , \ 0 \le t < T \\ E_0(m_T) \exp\left(r - \sqrt{B^2 - 4AC_{(a)}}\right)(t-T) & , \ t \ge T \end{cases}
$$

where $m_0 = \frac{1}{4}\phi^2(M(0;T))^2\ \theta_0^{\ \phi}$

Moreover, in the case where T = 0, since $M(t;T) = M^{II}$ and $\dot{M}(t;T) = 0$, the equation [22] reduces to:

$$
[23] \quad E_0\left[m_t\right] = m_0 \exp\left(r - \sqrt{B^2 - 4AC_{(a)}}\right)t \quad , \ t \ge 0
$$

where $m_0 = \frac{1}{4}\phi^2(M^{II})^2\ \theta_0^{\ \phi}$

On the other hand, if we let $T \to \infty$, [22] reduces to:

$$
[24] \quad E_0\left[m_t\right] = m_0 \exp\left(r - \sqrt{B^2 - 4A}\right)t \quad , \ t \ge 0
$$

where $m_0 = \frac{1}{4}\phi^2(M_1^I)^2\ \theta_0^{\ \phi}$

Since $\sqrt{B^2 - 4A} < \sqrt{B^2 - 4AC_{(a)}}$, if the introduction of environmental fees were delayed for ever, $(T = \infty)$, the expected maintenance expenditure pattern would be higher than in the case where the firm is charged from the beginning of the planning horizon, More generally, under our assumptions, the expected maintenance expenditure is positively correlated with T as shown in fig.2:

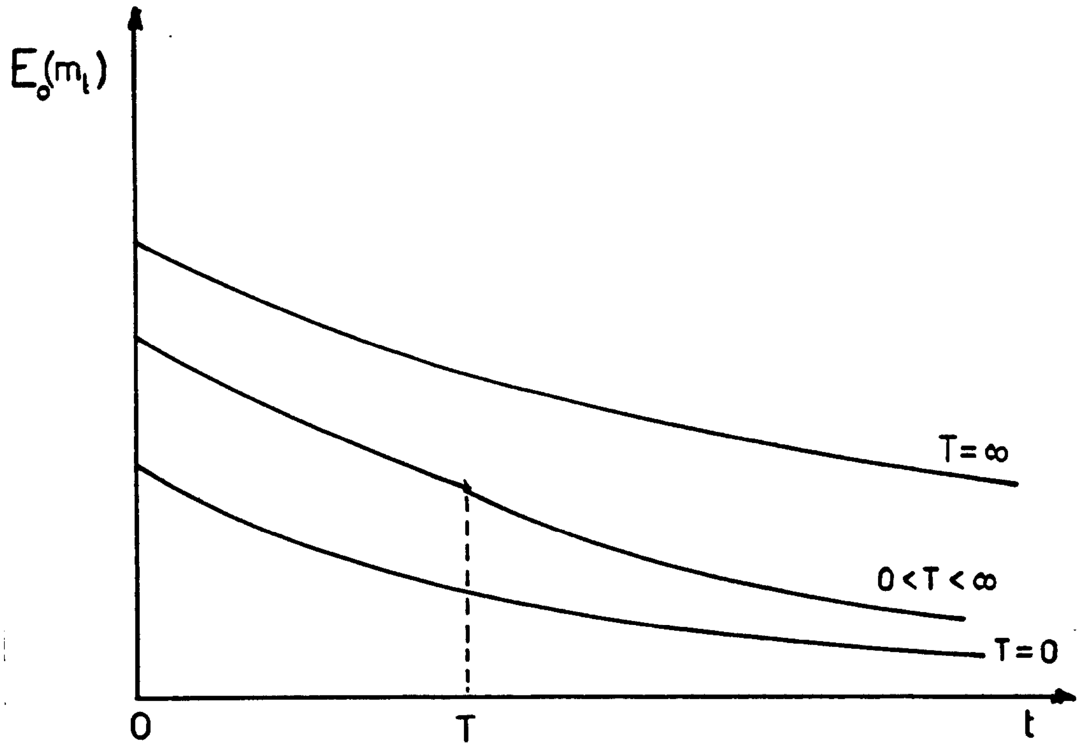

fig.2 Optimal expected maintenance expediture pattern under alternative time profile and $(r < \sqrt{B^2 - 4A})$

3.2.2 Let us now turn to the relationship between T and total expected environmental damage. Total damage attributed to each firm's unit of land is defined as follows:

$$D_{TOT} \equiv D(\theta_t)x_t = \begin{cases} \theta_t^{\Phi} & 0 \le t < T \\ (\alpha)^{1/(1-\alpha)} \theta_t^{\Phi} & t \ge T \end{cases}$$

Again by applying the **Itô's** Lemma it is possible to derive the expected rate of variation of D_{TOT}, and therefore:

$$[25]\quad E_0\left[\, D_t x_t \right] = \begin{cases} D_0 x_0 \exp\left(\int_0^t \phi\left[\frac{1}{2}\phi\ M(s;T) - \delta \right] + \frac{1}{2}\phi(\phi-1)\sigma^2 ds \right. & , 0 \le t < T \\ E_0(D_T x_T)\ \exp\left(r - \sqrt{B^2 - 4AC_{(a)}} \right)(t-T) & , t \ge T \end{cases}$$

In the case where T = 0, since M(t.;T) = $\dot{M}^{I}$, equation [25] reduces to:

$$[26]\quad E_0\left[\, D_t x_t \right] = D_0 x_0 \exp\left(r - \sqrt{B^2 - 4AC_{(a)}} \right) t \qquad , t \ge 0$$

where $D_0 x_0 = (\alpha)^{1/(1-\alpha)} \theta_0^{\phi}$

On the other hand, if we let $T \to \infty$, [25] reduces to:

$$[27]\quad E_0\left[\, D_t x_t \right] = D_0 x_0 \exp\left(r - \sqrt{B^2 - 4A} \right) t \qquad , t \ge 0$$

where $D_0 x_0 = \theta_0^{\phi}$

Since, again , $\sqrt{B^2 - 4A} < \sqrt{B^2 - 4AC_{(a)}}$ the expected total damage under $T \to \infty$ will be higher than under T = 0. More generally, under our assumptions, the expected total damage will be positively correlated with T, as shown in fig.3.

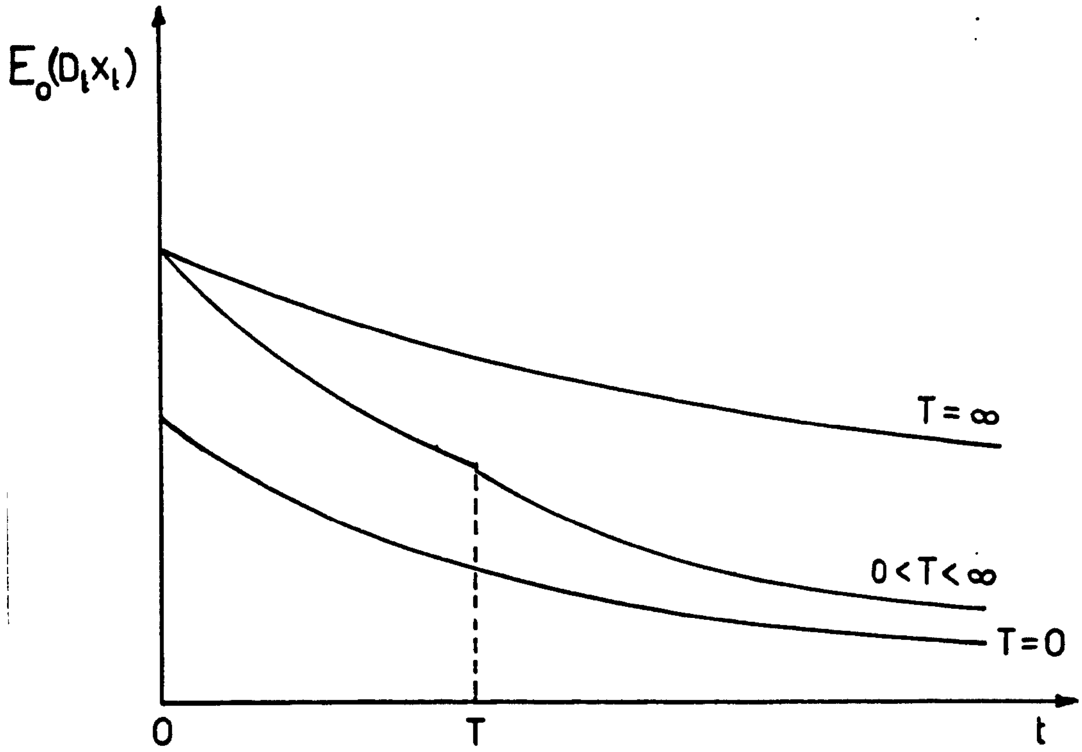

fig.3 Expected total damage pattern under alternative time profile and $(r < \sqrt{B^2 - 4A})$

3.2.3 Finally let us consider the effect of uncertainty about future realizations of the soil quality parameter θ on maintenance expenditure, the firm's market value and environmental damage.

According to [14] and [20], maintenance expenditure is an increasing function of M^{II} and M(t;T). This implies that the effect of uncertainty on m can be analyzed by looking at the effect of σ^2 on M^{II}, for $t \geq T$, and on M(t;T) for $0 \leq t < T$.

From [13] and [19], it follows that if $\Phi > 1$ ($0 < \Phi < 1$) an increase of σ^2 leads to an increase (decrease) in M^{II} and in M(t;T). In other words, higher volatility about future realizations of soil quality may either lead to an increase or a decrease in the maintenance expenditure pattern depending on the parameters α, β and ν which make up Φ.

Following the same line of reasoning, the same results apply to the firm's market value.

The effect of uncertainty just described arises from the fact that both the firm's first and second stage instantaneous cash flows, under optimal maintenance expenditure and x use, are convex (concave) functions of θ whenever $\Phi > 1$ ($0 < \Phi < 1$), As a result, increased uncertainty tends to increase (decrease) the value of future cash flows the firm expects to obtain from one unit of land. This, in turn, from the firm's point of view, is equivalent to a reduction (increase) in marginal cost associated with the decision of "improving" soil quality through maintenance expenditure, Or, taking a slightly different perspective, convexity (concavity) of the firm's profit function implies that the disadvantages of expected "bad news", i.e. low future realizations of θ, are more (less) than compensated for by the advantages of "good news", and, the marginal expected profitability of maintenance expenditure increases (decreases).

Let us now analyze the action of uncertainty with regard to the expected environmental damage per unit of land. From [25], [26] and [27], we obtain:

$$\frac{d\ E_0(D_t x_t)}{d\sigma^2} = \begin{cases} > 0 & \text{if} \quad \Phi > 1 \\ < 0 & \text{if} \quad 0 < \Phi < 1 \end{cases}$$

In other words, if, as in fig.3, we assume total damage decreases over time, increased uncertainty may either reduce or increase the expected rate of such decline, depending, again, on the value of the technical parameters which make up Φ.

4. THE AGENCY'S OPTIMAL MANAGEMENT RULES AND THE CHOICE OF OPTIMAL TIME PROFILE FOR ENVIRONMENTAL FEES

4.1 Before trying to characterize the choice of the optimal time profile, let us take T as exogenously given and identify the agency's optimal management rules, in terms of maintenance pattern and x use.

We assume the agency wishes to maximize the following objective function:

[28] $$\max_{m,x} W(\theta_0;T) = E_0\left\{ \int_0^T \left(Q(x_t,\theta_t) - m_t - D(\theta_t)x_t\right)e^{-rt}dt + \right.$$

$$\left. + \int_T^{\infty} \left(Q(x_t,\theta_t) - m_t - D(\theta_t)x_t + \rho D(\theta_t)x_t\right)e^{-rt}dt \right\}$$

In other words:

- the agency is assumed to take care of environmental damages over the entire planning horizon $[0,\infty)$;
- the agency's welfare function, which is assumed to be separable in its arguments, includes the firm's utility;
- the agency is assumed to receive a utility from collecting funds through environmental fees, and the parameter ρ ($0 < \rho < 1$) has to be interpreted as the net "social" benefit of such collection.

Adopting a procedure similar to that undertaken in section 3 when dealing with the firm's maximization problem, the agency's optimal variable input level can be obtained:

$$
[29] \quad \begin{cases} x^{*}_{(p)t} = (\alpha)^{1/(1-\alpha)}\, \theta_t^{(\nu+\beta)/(1-\alpha)} & \text{for} \quad 0 \le t < T \\ x^{**}_{(p)t} = \left(\dfrac{\alpha}{1-\rho}\right)^{1/(1-\alpha)} \theta_t^{(\nu+\beta)/(1-\alpha)} & \text{for} \quad t \ge T, \quad \rho < \bar{\rho} \\ x^{**}_{(p)t} = \bar{x} & \text{for} \quad t \ge T, \quad \rho \ge \bar{\rho} \end{cases}
$$

Moreover, since $0 < \rho < 1$, the following inequality holds: $x^{*}_{(p)t} \le x^{**}_{(p)t}$, for given θ_t. By substituting [29] by [28], and keeping [8], the agency's maximization problem reduces to:

[30]

$$
\max_{m} W(\theta_0;T) \equiv E_0\left\{\int_0^T \left[C_{(p)1}\theta_t^{\phi} - m_t\right] e^{-rt}dt + \int_T^{\infty} \left[C_{(p)2}\theta_t^{\phi} - m_t\right] e^{-rt}dt\right\}
$$

$$
\text{where} \quad C_{(p)1} \equiv C_{(a)} = (1-\alpha)(\alpha)^{\alpha/(1-\alpha)} < 1
$$

$$
C_{(p)2} = (1-\alpha)\left(\frac{\alpha}{1-\rho}\right)^{\alpha/(1-\alpha)} < 1
$$

$$
\phi = \frac{1}{1-\alpha}\nu + \frac{\alpha}{1-\alpha}\beta
$$

By adopting the same procedure as in section 3, when dealing with the firm's II-stage maximization, and assuming the same restrictions about the parameters ξ, γ and ϕ, the solution for the agency's welfare value at II-stage is:

[31] $$W^{II}(\theta_t) = N^{II}\theta_t^{\phi} \qquad \text{for } t \geq T$$

where $$N^{II} = \frac{B - \sqrt{B^2 - 4AC_{(p)2}}}{2A}$$

$$A = \frac{1}{4}\phi^2$$

$$B = r + \phi\delta - \frac{1}{2}\phi(\phi-1)\sigma^2$$

The agency's optimal maintenance expenditure rule then becomes:

[32] $$m_t = \frac{1}{4}\phi^2 \left[N^{II}\right]^2 \theta_t^{\phi} \qquad \text{for} \quad t \geq T$$

which implies that the stochastic differential equation [2] reduces to:

[33] $$d\theta_t = \left[\frac{1}{2}\phi\, N^{II} - \delta\right]\theta_t dt + \sigma\, \theta_t dz_t \qquad \text{for} \quad t \geq T$$

We can now go on to I-stage maximization, on condition that the agency's welfare value at time T coincides with the discounted scrape value given by [31]. Again following the same procedure adopted in section 3, we obtain:

[34] $$W(\theta_t, t;T) = N(t;T)\,\theta_t^{\phi} \qquad \text{for} \quad 0 \leq t < T$$

where:

$$N(t;T) = \frac{N_2^I - N_1^I \left(\dfrac{N^{II} - N_2^I}{N^{II} - N_1^I} \right) \exp\left(\sqrt{B^2 - 4AC_{(p)1}}\right)(T-t)}{1 - \left(\dfrac{N^{II} - N_2^I}{N^{II} - N_1^I} \right) \exp\left(\sqrt{B^2 - 4AC_{(p)1}}\right)(T-t)}$$

$$N_1^I = \frac{B - \sqrt{B^2 - 4AC_{(p)1}}}{2A}, \qquad N_2^I = \frac{B + \sqrt{B^2 - 4AC_{(p)1}}}{2A}$$

It is easy to show that, when t = T, $N(T;T) = N^{II}$. However, unlike what was seen with regards to the firm, in this case :

$$\frac{N^{II} - N_2^I}{N^{II} - N_1^I} < 0$$

In other words, if the agency were able to decide time profile T, it would choose T = 0, because, according to the above inequality its welfare is higher during the period when environmental fees are charged than during the period when firms are exempt from taxation. Obviously, since N^{II} increases with ρ, the higher the net marginal “social” benefit of collecting taxes, the higher is the agency’s welfare loss in moving away from T = 0.

Moreover it is easy to obtain from [34]:

$$\lim_{T \to \infty} N(t;T) = N_1^I$$

$$\lim_{T \to 0} N(t;T) = N^{II}$$

with $N_1^I \le N^{II}$, as shown in fig.4.

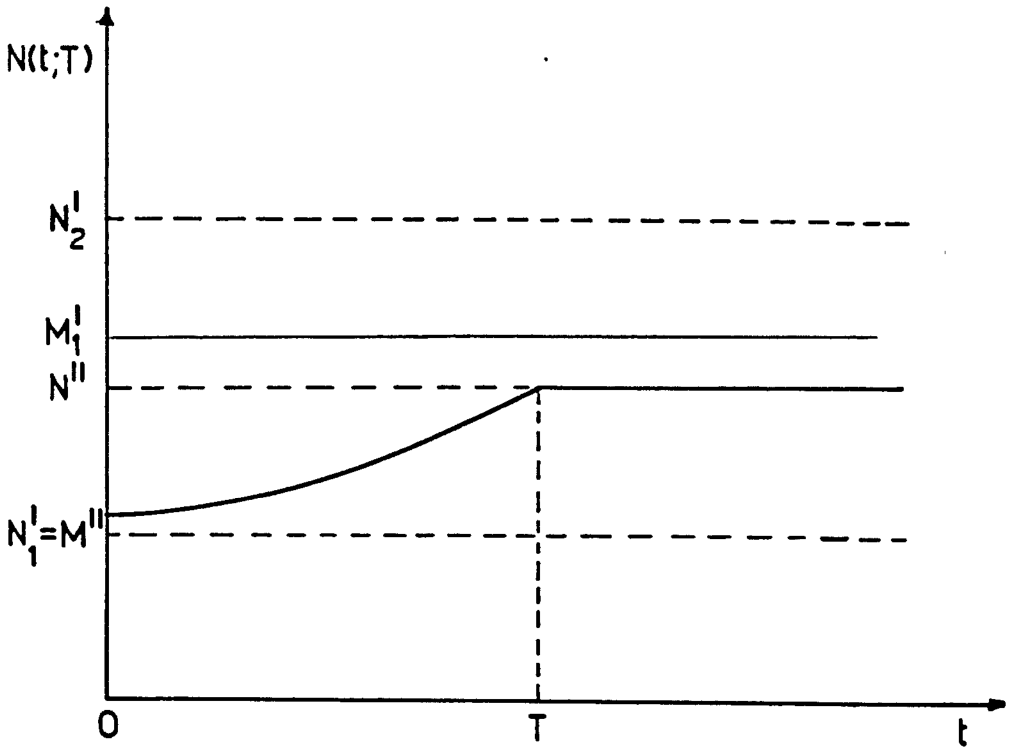

fig.4

The agency's optimal maintenance expenditure policy in the I-stage is described by:

[35] $$m_t = \frac{1}{4}\phi^2 \left[N(t;T) \right]^2 \theta_t{}^{\phi} \qquad \text{for } 0 \le t < T$$

whilst the stochastic differential equation [2] for θ reduces to:

[36] $$d\theta_t = \left[\frac{1}{2}\phi\, N(t;T) - \delta \right] \theta_t dt + \sigma\, \theta_t dz_t \quad , \text{ for } 0 \le t < T$$

Notice that, by replacing $M(t;T)$ by $N(t;T)$, the same results described in 3.2.1, 3.2.2, and 3.2.3 apply to the agency.

4.2 On the basis of the results proposed in the above sections, we may now consider the problem of optimal choice of T, assuming that this choice is undertaken by the same subject for whom in section 4.1 the optimality conditions for m and x were derived.

In the following discussion it will be assumed that, as far as management decisions are concerned, only two strategies are open to the firm: adoption of its own optimality rules for x and m (described in equations [6], [14] and [18]), or, alternatively, the agency's rules (equations [29], [32] and [35]).

Let us start by summarizing in fig.5 the results obtained in section 3 concerning the relationship between T and the firm's market value (evaluated at the beginning of the planning horizon) and those derived in section 4.1 concerning the relationship between T and the agency's welfare value (evaluated at the beginning of the planning horizon).

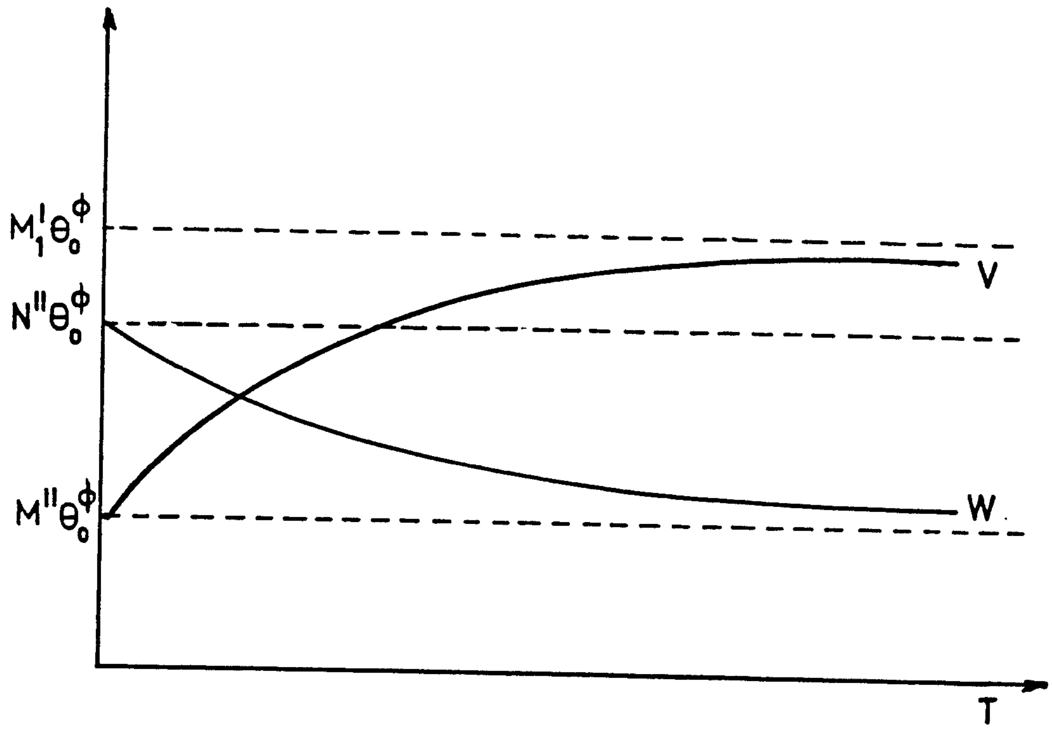

fig.5

From [3] and [28] it can be easily shown that the following identity holds:

$$[37] \qquad V(\theta_0, t=0;T) \equiv W(\theta_0, t=0;T) \; +$$

$$+ \; E_0 \left\{ \int_0^T \left[D(\theta_t)x_t \right] e^{-rt}dt \; - \; \int_T^{\infty} \left[\rho \; D(\theta_t)x_t \right] e^{-rt}dt \right\}$$

If the firm adopts the agency's optimal management rules, according to [37], its market value, hereafter $V^P(\theta_0;T)$, becomes equal to the agency's optimal welfare value, described in fig.4, plus the expected value of the difference between discounted social damages in [0,T) and the agency's utility derived from taxation in $[T,\infty)$, which are both evaluated under the agency's optimal rules. On the other hand, if the agency "accepts" the firm's management rules, according to [37] its welfare value, hereafter $W^a(\theta_0;T)$, becomes equal to the firm's market value, described in fig.1, minus the expected value of the above difference evaluated, now, under the firm's optimal rules.

Considering the agency's objective function, if it were able to monitor the firm's actions, and if it wanted the firm to adopt the optimal "social" rules, [29], [32] and [35], the best decision would be non-postponement of the introduction of environmental fees. However, if the principal is unable to carry out such monitoring, he has to define an incentive which would induce the agent to self-select the "socially" desired maintenance expenditure pattern and x use. In our framework, this means that the agency has to identify a set of T values which ensure that the firm's market value under the agency's management rules is higher than under its own rules: in this case the time lag granted to firms assumes the meaning of a "premium" they will receive in exchange for accepting the agency's desired management rules. The problem facing the agency consists of picking on, among the set of time profiles which satisfy such a property, the one providing the highest welfare value, T*.

Notice, however, that T* may not be "sustainable" or even optimal for the agency(5). In fact we can take a different perspective and imagine that the firm "offers" the agency, in exchange for acceptance of its own rules, the "opportunity" of setting a different time profile, T^{**}. If T^{**} is "sustainable" and implies a higher welfare value than the one associated with T^{*}, then the agency will find it profitable (6).

To clarify the above statements, let us start by spelling out the firm's reaction in terms of management decisions to the agency's announcement of T. Since we assume the firm will choose x and m after this announcement, its best reply function consists of comparing its market value under its own optimal rules, $V(\theta_0;T)$, with $V^p(\theta_0;T)$. That is:

$$[38] \qquad (x,m) = \max \left[V(\theta_0;T) \; , \; V^p(\theta_0;T) \right] \quad \text{for given } T$$

In other words, the firm will adopt its own optimal rules or the agency's ones depending on which, given T, brings the highest market value.

On the other hand, taking account of the "incentive constraint" [38], the agency will define the optimal time profile by looking at the value of T which makes the welfare value maximum. Formally:

$$[39] \qquad \begin{cases} \max\limits_{T} \left[\max \left(W(\theta_0;T) \; , \; W^a(\theta_0;T) \right) \right] \\ \text{s.t. } [38] \end{cases}$$

In other words, the backward-induction logic requires that the agency foresee that the firm will respond optimally to any time profile announced.

To provide, through a diagrammatic form, a solution for the problem [39], let us preliminary describe the form taken on by the firm's best reply as implied by [38]. In this respect it is possible to identify at least four situations.

<u>Case 1</u> If the following inequalities hold:

$$\sqrt{B^2 - 4AC_{(p)1}} - \sqrt{B^2 - AC_{(p)2}} > \rho\left(\frac{\alpha}{1-\rho}\right)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2 - 4AC_{(p)2}}}$$

$$\sqrt{B^2 - 4AC_{(p)1}} - \sqrt{B^2 - 4A} < (\alpha)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2 - 4AC_{(p)1}}}$$

the firm's market value under the agency's optimal rules, V^P, may take, relative to V, the forms depicted in fig.6 (see appendix B):

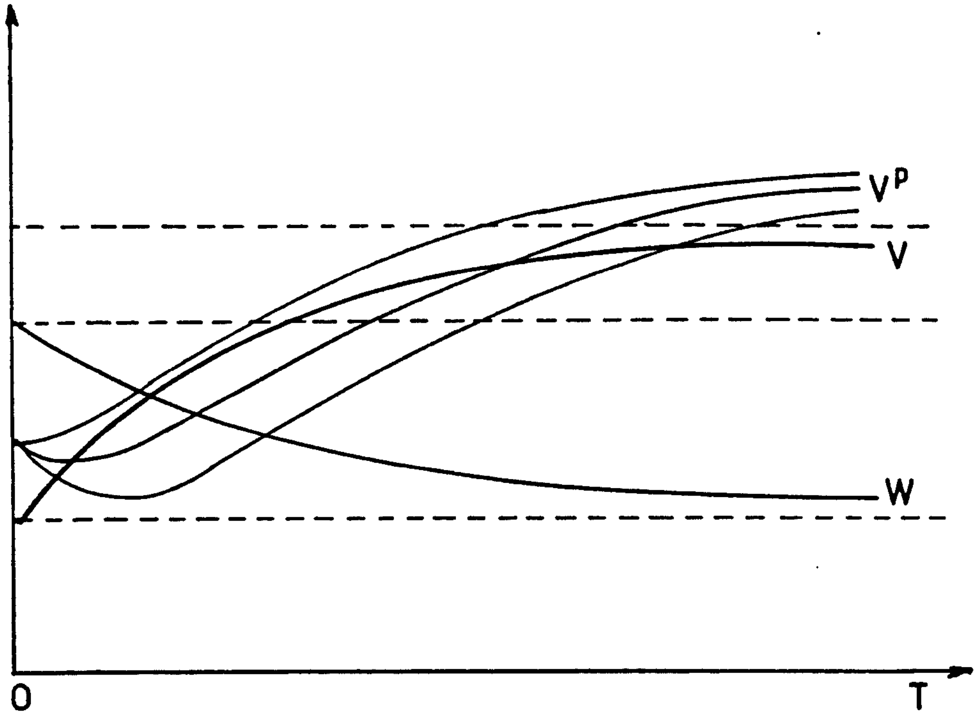

fig.6

It is evident, in this case, that we can have two possible solutions for [38] :

[40] $$V_{max} = V^{P} \quad , \quad \forall\, T \quad \Leftrightarrow \quad (x,m) = (x_{(p)},\ m_{(p)})$$

or,

[40′] $$V_{max} = \begin{cases} V^{P} \quad , & T < T_1 \quad \Leftrightarrow \quad (x,m) = (x_{(p)},\ m_{(p)}) \\ V \quad , & T_1 < T < T_2 \quad \Leftrightarrow \quad (x,m) = (x_{(a)},\ m_{(a)}) \\ V^{P} \quad , & T > T_2 \quad \Leftrightarrow \quad (x,m) = (x_{(p)},\ m_{(p)}) \end{cases}$$

Whether [40] or [40′] represent a solution for the incentive constraint [38] depends on the shape of V^P, which, in turn, depends on the technical parameters related to the production function, damage function and maintenance technology as well as on the net "social" benefit of tax collection ,ρ.

<u>Case 2</u> If the following inequalities hold:

$$\sqrt{B^2-4AC_{(p)1}}-\sqrt{B^2-\ AC_{(p)2}} > \rho\left(\frac{\alpha}{1-\rho}\right)^{1/(1-\alpha)}\frac{2A}{\sqrt{B^2-4AC_{(p)2}}}$$

$$\sqrt{B^2-\ 4AC_{(p)1}}-\sqrt{B^2-4A} > (\alpha)^{1/(1-\alpha)}\frac{2A}{\sqrt{B^2-4AC_{(p)1}}}$$

V^P may take on, relative to V, the forms depicted in fig.7:

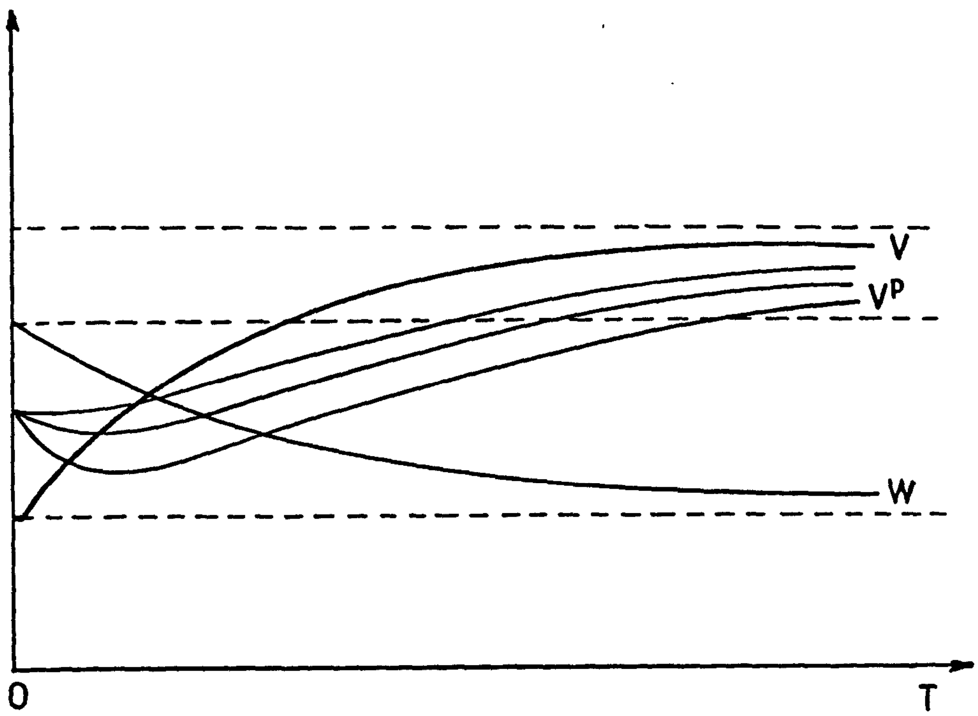

fig.7

In this case, the solution for [38] appears to be:

$$[41] \qquad V_{max} = \begin{cases} V^P \quad , & T < T_1 \iff (x,m) = (x_{(p)}, m_{(p)}) \\ V \quad , & T > T_1 \iff (x,m) = (x_{(a)}, m_{(a)}) \end{cases}$$

<u>Case 3</u> If the following inequalities hold:

$$\sqrt{B^2 - 4AC_{(p)1}} - \sqrt{B^2 - AC_{(p)2}} < \rho\left(\frac{\alpha}{1-\rho}\right)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2 - 4AC_{(p)2}}}$$

$$\sqrt{B^2 - 4AC_{(p)1}} - \sqrt{B^2 - 4A} < (\alpha)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2 - 4AC_{(p)1}}}$$

V^P may take, relatively to V, the forms depicted in fig.8:

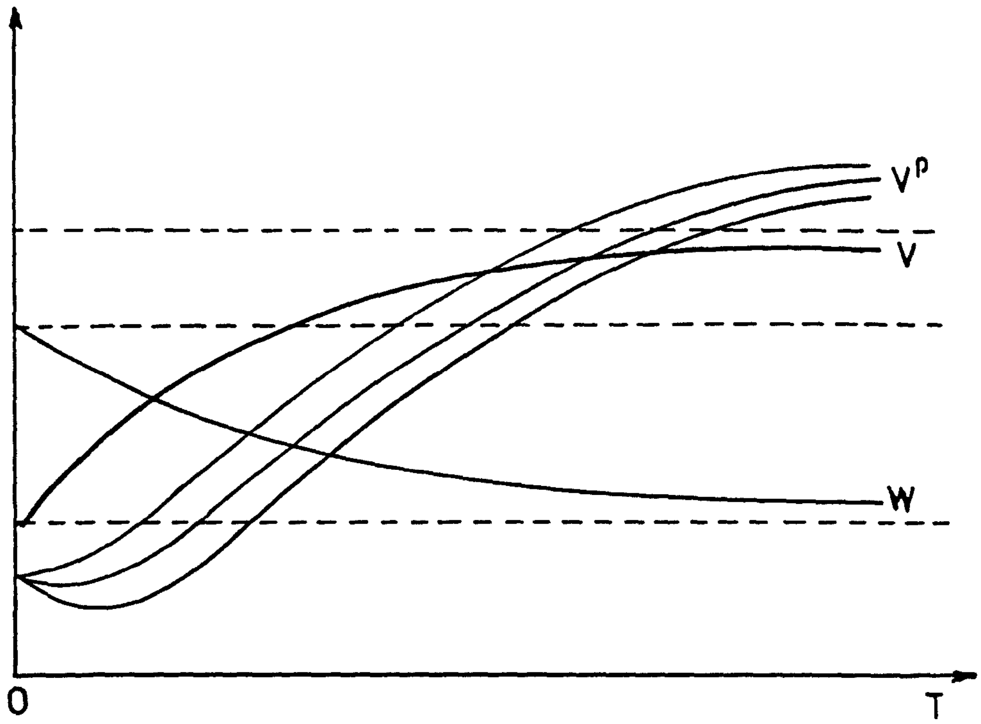

fig.8

In this case, the solution for [38] appears to be:

[42] $$V_{max} = \begin{cases} V , & T < T_1 \Leftrightarrow (x,m) = (x_{(a)}, m_{(a)}) \\ V^p , & T > T_1 \Leftrightarrow (x,m) = (x_{(p)}, m_{(p)}) \end{cases}$$

Case 4 Finally, if the following inequalities hold:

$$\sqrt{B^2 - 4AC_{(p)1}} - \sqrt{B^2 - AC_{(p)2}} < \rho\left(\frac{\alpha}{1-\rho}\right)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2 - 4AC_{(p)2}}}$$

$$\sqrt{B^2 - 4AC_{(p)1}} - \sqrt{B^2 - 4A} > (\alpha)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2 - 4AC_{(p)1}}}$$

V^p may take on, relative to V, the forms depicted in fig.9:

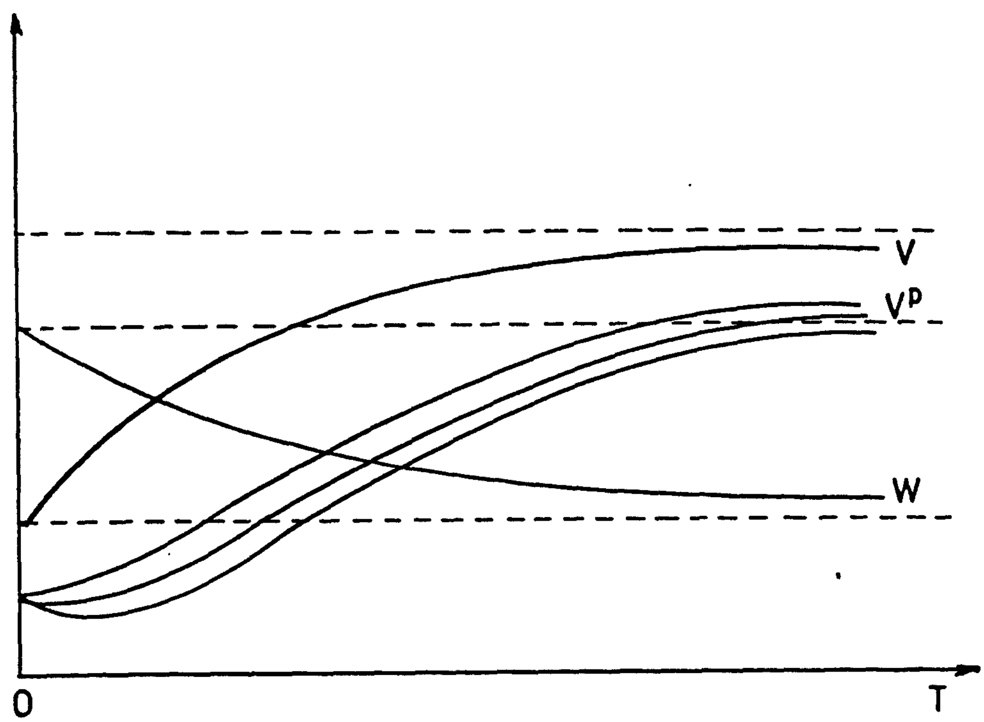

fig.9

In this case, the solution for [38] appears to be:

[43] $V_{max} = V \quad , \quad \forall\, T \quad \Leftrightarrow \quad (x,m) = (x_{(a)},\ m_{(a)})$

To summarize, the solution to incentive constraint [38] may give rise to a variety of situations, which range from the one ([43]) where, whatever time lag is granted, the firm will never find it profitable to give up its own optimal rules, to the one ([40]) where the firm will always find it profitable to "internalize" the agency's rules. There are intermediate situations where the choice of T may affect management decisions by switching the firm's choice from its own rules to the agency's ones, and vice versa ([40'], [41], [42]).

On the grounds of these results, a characterization of some representative solutions for [39] may be obtained by overlaying on fig.6-9 the corresponding agency's welfare value evaluated under its own rules (W) and the firm's ones (W^a). As shown above with reference to the firm's market" value under the agency's rules($x_{(p)}$, $m_{(p)}$), the latter's welfare value, evaluated under ($x_{[a)}$, $m_{(a)}$), may take different shapes, depending, once again, on the parameters of the technical relations considered and on ρ. As a result, a great variety of solutions may be identified. Hereafter, however, we shall merely consider those thought to be sufficiently representative.

Let us start by assuming that the inequalities considered in case 1 hold. Even if W^a may take different shapes, such inequalities imply that as T tends to zero or infinite, W^a becomes lower than W (see appendix B), as shown in figs.10a and 10b where alternative shapes of V^p , picked from fig.6, are also drawn.

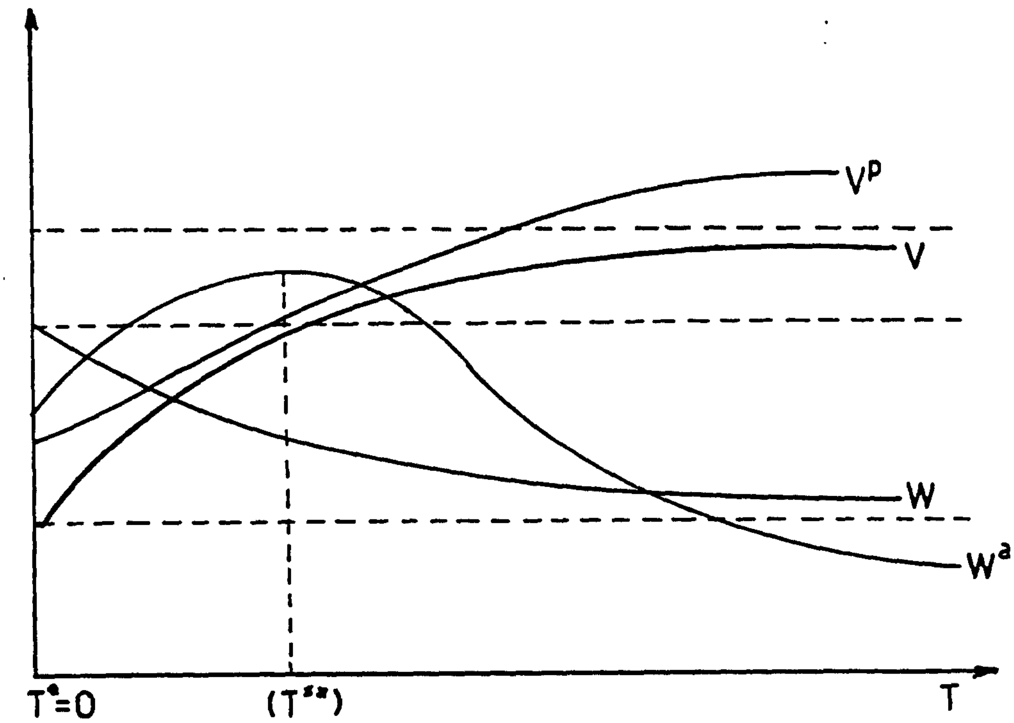

fig.10a

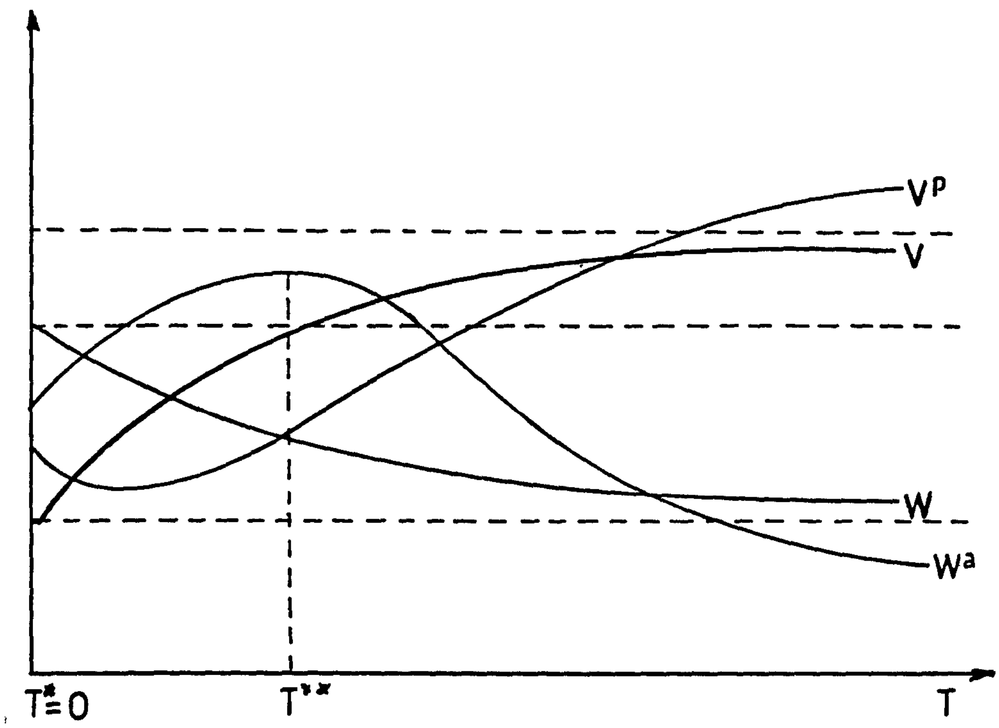

fig.10b

According to fig.10a the highest agency welfare value may be found at T**. However since the firm will always find it profitable to adopt $(x_{(p)}, m_{(p)})$, this time profile is not sustainable. Among the sustainable time profiles, the best choice appears to be non-postponement of the introduction of environmental fees, i.e. $T^* = 0$, On the other hand, if we find ourselves in the situation described by fig.10b, T^{**} appears to be both a sustainable and optimal incentive: therefore, by delaying the introduction of tax payment at date T^{**} and allowing the firm to adopt its own optimal rules, the agency reaches a higher welfare value.

If the inequalities considered in case 2 hold, W^a still becomes lower than W when T tends to zero, whilst it can be either higher or lower than W when T tends to infinite. In figs.11a and 11b two possible shapes of W^a are drawn together with a possible shape of V^p picked from fig.7.

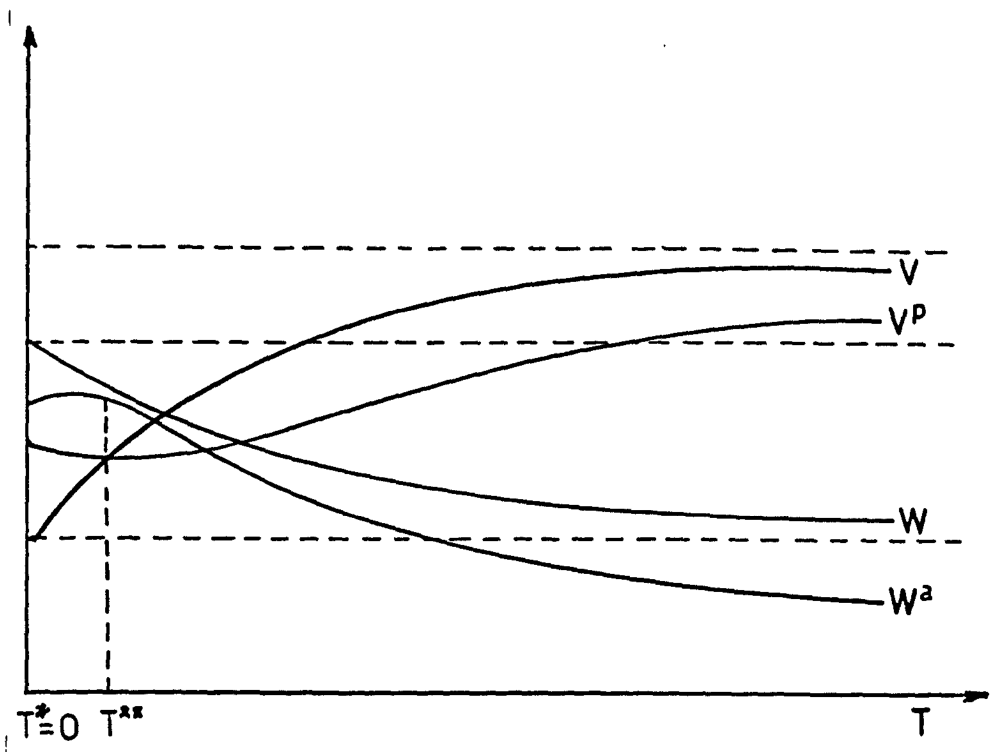

fig.11a

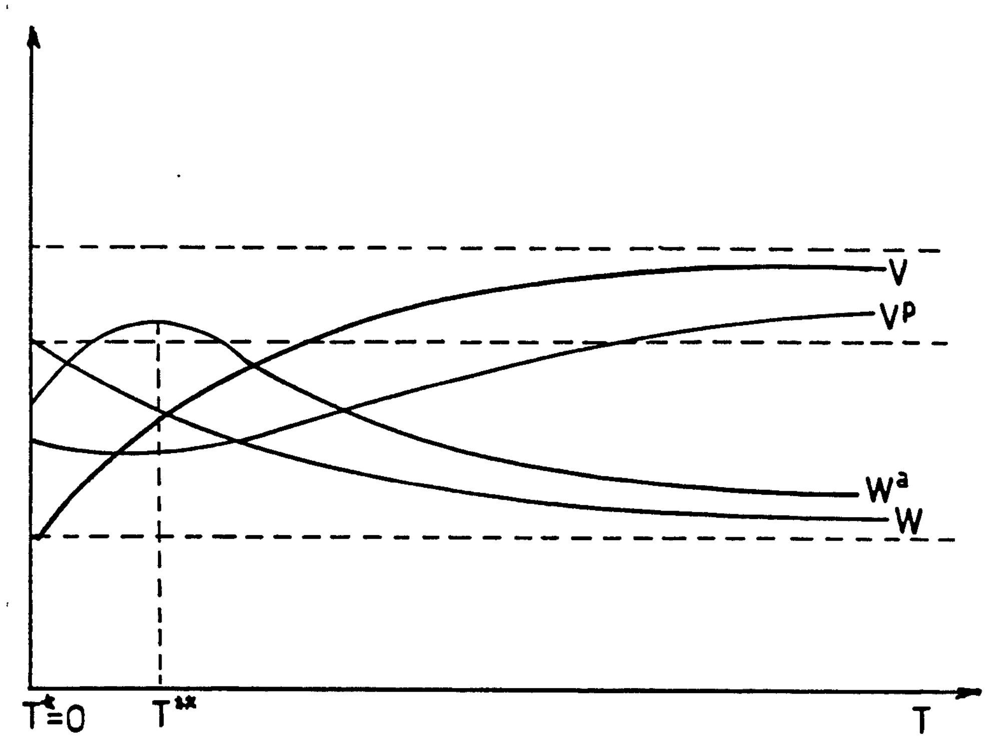

fig.11b

In both situations, figs.11a and 11b, the only sustainable time profile appears to be $T^* = 0$.

If the inequalities considered in case 3 hold, W^a becomes lower than W when T tends to infinite, whilst it can be either higher or lower than W when T tends to zero. In figs.12a, 12b and 12c three possible shapes of W^a are drawn together with a possible shape of V^p picked from fig.8.

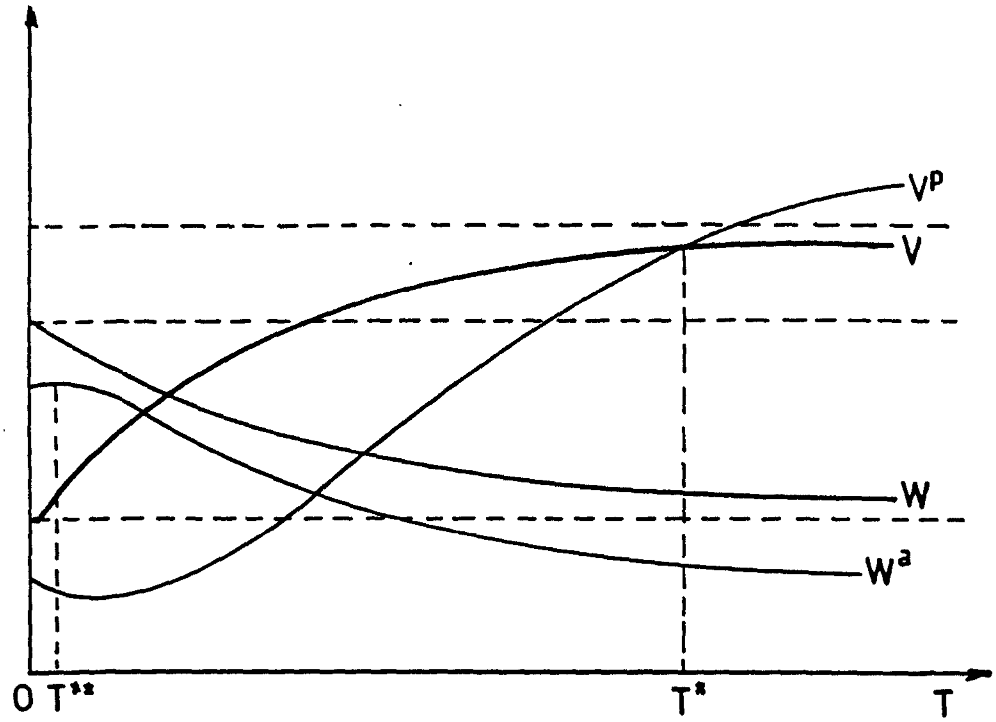

fig.12a

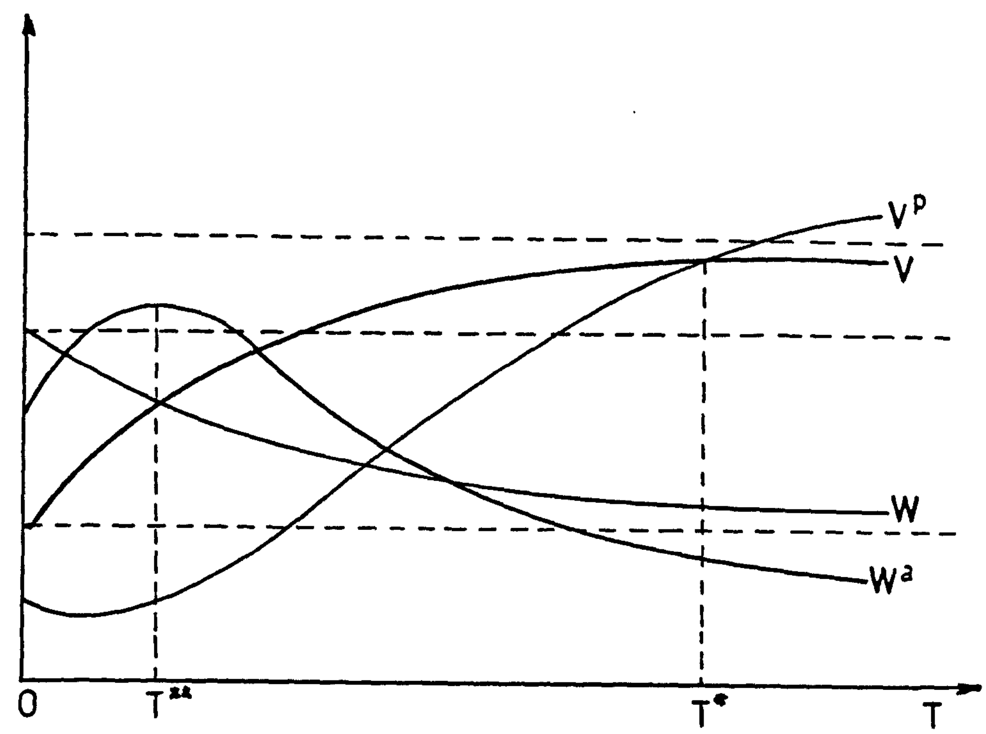

fig.12b

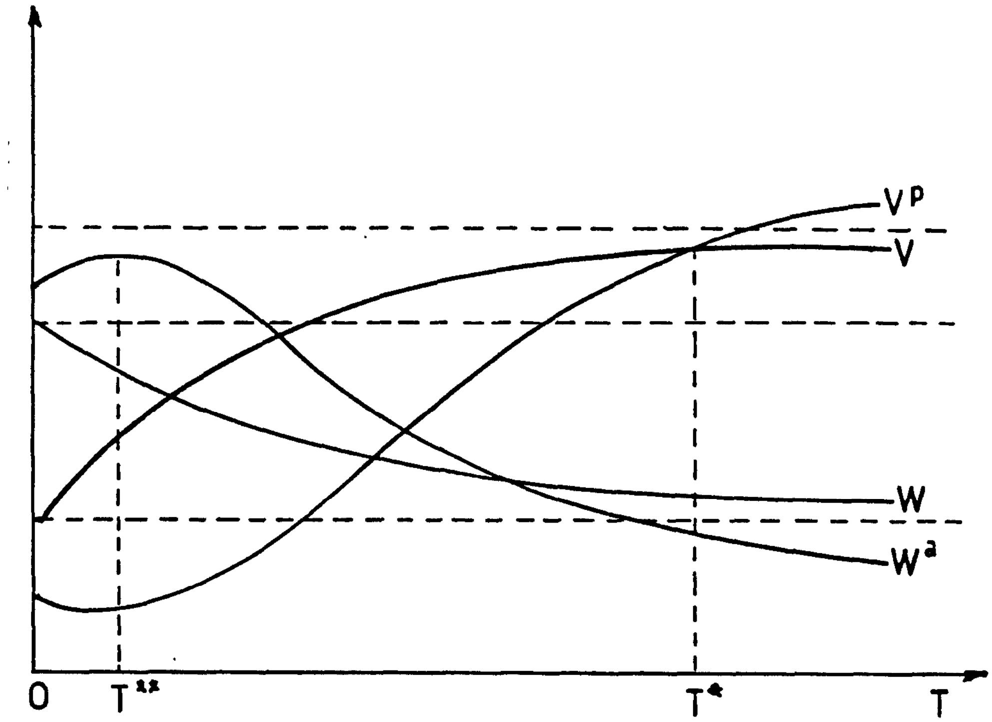

fig.12c

Considering fig. 12a, both T^{**} and T^* are sustainable; however the former, combined with firm's choice ($x_{(a)}$, $m_{(a)}$), provides the agency with the highest welfare value. On the other hand, in figs.12b and 12c the agency would reach the highest welfare value by non-postponement of the introduction of fees (i.e.(*) = 0), which however is not sustainable: in both situations T^{**} and T^* are sustainable but the former is the best choice for fig.12b, whilst the latter is best for fig.12c. Whilst in the situation depicted in fig.12b the agency finds it convenient to allow the firm to adoptits own rules in exchange for a “short” period of tax exemption, in fig.12c the agency finds it profitable to induce acceptance of the “socially” optimal management rules through a wider period of exemption from payments,

Finally, if the inequalities considered in case 4 hold, W^a can be either higher or lower than W when T tends to zero and infinite. Again, in fig.13a, fig.13b two possible shapes of W^a are drawn together with a possible shape of V^P picked from fig.9,

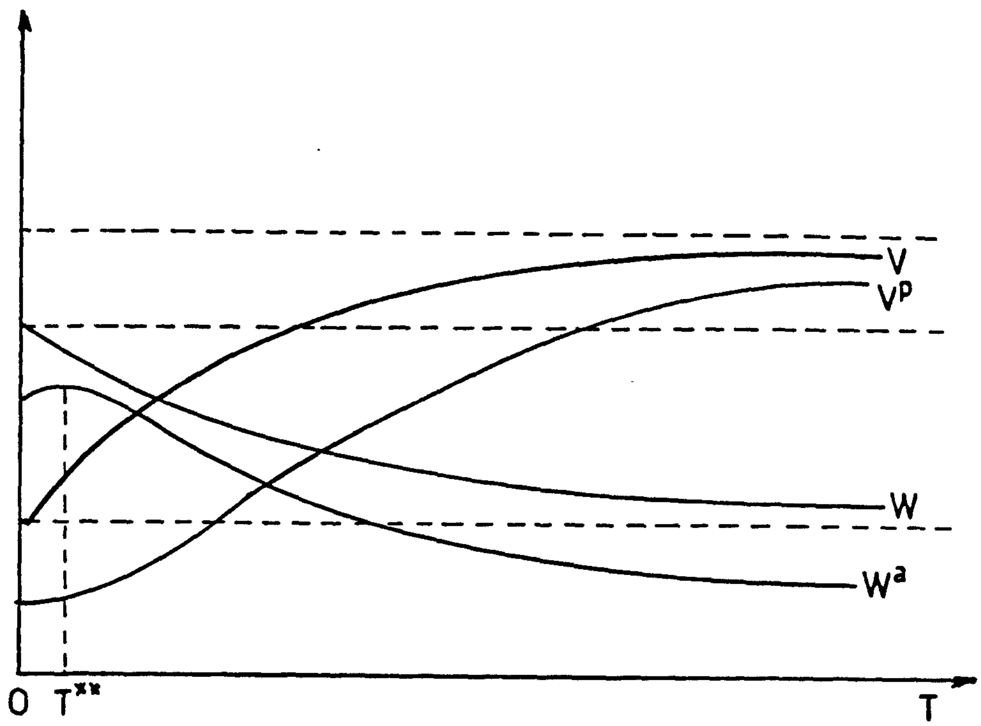

fig.13a

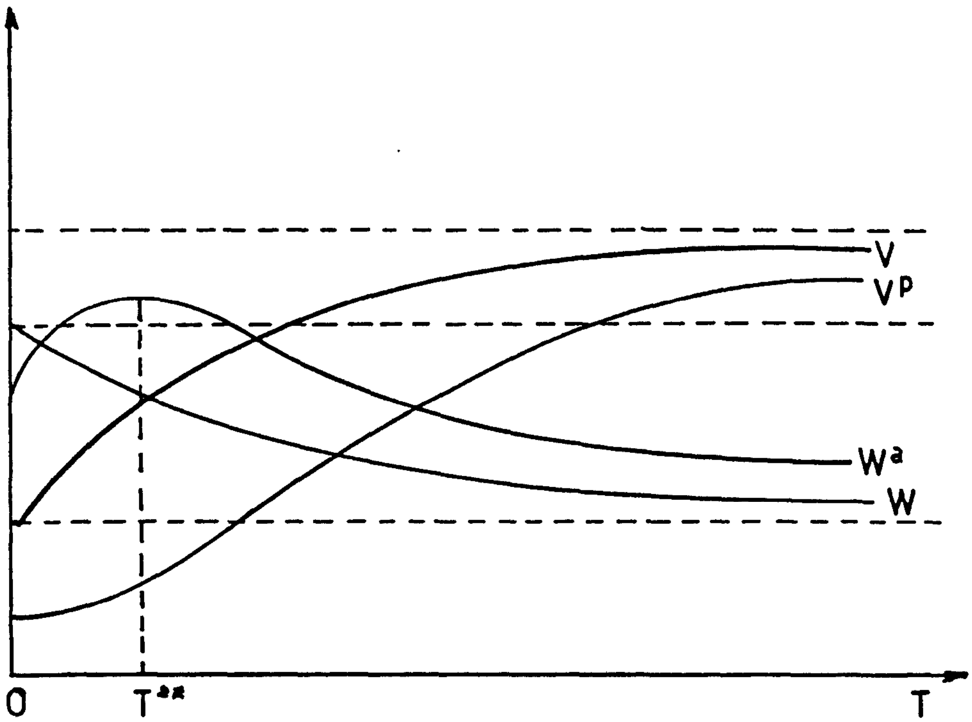

fig.13b

The situation described in fig. 12a may be regarded as symmetrical with respect to the one depicted in fig. 10a: the agency would achieve its highest welfare value at $T^* = 0$, i.e. when no time lag is allowed and the firm adopts as management rules $(x_{(p)}, m_{(p)})$. However, since the firm will never find it profitable to give up its optimal rules, this time profile is not sustainable, therefore, the agency has to look for a T^{**} which makes it better off under $(x_{(a)}, m_{(a)})$. Even in the situation depicted in fig.13b the firm will never find it profitable to adopt $(x_{(p)}, m_{(p)})$. Nevertheless , in this case, the agency's best choice would be to introduce environmental fees from the beginning of the planning period, i.e. $T^{**} = 0$, since, by doing so, it would achieve a higher welfare value.

5. FINAL REMARKS

The basic aim of the paper was to enhance the results provided in other contributions on NPP control with further insights concerning the role of policy instruments in influencing suspected polluters' productive decisions as well as the allocative properties of alternative regulatory schemes.

In particular we have concentrated on the application of what has been termed an "indirect approach", focusing on two issues which, as far as we know, have received little attention. Firstly, we have tried to deal formally with the possibility that the production site's physical characteristics (the firm's "typology") may vary over time because of non-monitorable actions taken by suspected polluters in conditions of uncertainty regarding the performance of the actions themselves. Secondly, we accounted for the possibility that the legislator might consider the opportunity of delaying the introduction of management practice incentives.

Non-monitorability of the firms' management practices provides the agency with the rationale for selecting the time profile at the beginning of the planning period. Moreover, according to our findings, the decision of delaying the introduction of "environmental fees" may, under certain conditions, constitute an optimal decision from the agency's point of view.

The analysis proposed in the above pages is undoubtedly conditional on a number of assumptions introduced in the paper. These assumptions concern the availability of information regarding maintenance technology, the "form" of uncertainty, the general structure of the technical relationships which make up the model, and the objective functions assigned to the hypothetical actors.

As far as the maintenance technology is concerned, we have assumed that the firm(s) and the agency share the same information as well as the same uncertainty about future realization of the soil quality index. The rationale behind this assumption is that, even if at some point in time the firms are unaware of the maintenance technology, the informational gap could be eliminated by the agency by transmitting all the technical information it possesses before setting the regulatory scheme. Furthermore, if the performance of maintenance decisions is believed to be affected by on-going exogenous shocks, the agency might also include the probability distribution of such shocks in the "informational package".

Turning to the form of uncertainty, it has been assumed that future realizations of the soil quality index will always be uncertain, with a variance which grows linearly with the time horizon. Obviously, this may not always be the case, and the plausibility of modeling the uncertainty along the lines of a Brownian motion process has to be assessed on a case by case basis.

Analogous considerations apply to the assumptions concerning the general structure of the technical relationships introduced in the paper. Moreover, since we deliberately tried to keep the analysis as more general as possible, rather than straightforward conclusions a menu of possible results has been proposed, each of which depends on the values taken by the parameters appearing in the model. It follows that to move a step forward with respect to a merely theoretical analysis would require not only careful assessment of the plausibility of the assumptions concerning the general structure of the technical relationships, but also, a more precise specification of the values taken on by all the relevant parameters.

The set of relevant parameters includes not only those characterizing the technical relationships, but also the net “social” benefit of raising funds through taxation and the rate of discount.

The former was introduced to take account of the possibility that the social planner might receive a utility from taxation as such. In this case, environmental charges not only play the role of instrument for reducing the pressure exerted upon the environment by private economic activities, but are also regarded as means for collecting additional tax revenues. This double role gives rise to a sort of trade-off between environmental quality improvements and increased tax revenues. Depending on the relative weight attached to these two conflicting objectives, different optimal time profiles may arise. Again, the plausibility of assuming the existence, from the agency’s point of view, of such a trade-off should be assessed on a case by case basis, but, in our view, it should not be discarded a *priori.*

As far as the second relevant “non technical” parameter, the discount rate, is concerned, it should be noted that we have assumed that the social planner and private agents share the same intertemporal preferences. Since this assumption may appear to be somewhat questionable, an interesting extension of the basic framework we developed consists of exploring the implications of different discount rates in terms of management practice decisions as well as in terms of optimal choice of the time profile for environmental charges.

Further extensions include analysis of the implications on policy design, of abandoning the hypothesis of identical availability of information concerning the initial status of the production site’s physical characteristics which are believed to affect the extent of pollutant emissions at field level. Whilst assuming the existence of uninformed agents should not significantly modify our basic framework, in that its main implication is that the firm’s reply function has to be defined in terms of expected value according to the probability distribution of θ, the case of an uninformed principal would make the structure of the game much more complex from an analytical point of view. In this case, the analytical framework will take on the form of a true Principal-Agent model, where incentives take on the sense of instruments to extract information from private agents about their initial “typology”.

FOOTNOTES

(1) For a discussion of (Itô's) diffusion processes and stochastic differential equations see, for example, Arnold (1974) and Karlin and Taylor (1981). For economic applications of stochastic calculus techniques used throughout the paper, see Malliaris and Brock (1982).

(2) Although w is set equal to zero for technical reasons, it is not so implausible to imagine situations where the price of (potentially) polluting inputs is, relatively speaking, very low or even zero. Examples are nitrogen fertilizers in EEC Countries or nutrients contained in slurry available for farms with mixed crop-livestock production.

(3) In formulating these restrictions we are indebted to the work of Vorst (1987) and Moretto (1991). Notice that $\xi = \frac{1}{2}$ is just in the middle of the domain of ξ and by this restriction we find that the Hamilton-Jacobi-Bellaman equation [11] has only quadratic or linear terms in V_{θ}^{II}. The second restriction is for technical reasons.

(4) Restriction $\gamma = \frac{1}{2}\phi$, introduced in the context of II-stage maximization, together with $\gamma = \frac{1}{2}\varphi$ implies $\phi = \varphi$. Thus, our assumptions imply that the shapes of the firm's cash flow function, gross of maintenance expenditure, at I-stage, and of the firm's cash flow function, gross of m, at II-stage, are the same and differ only by a constant.

(5) In the context of the present paper the term "sustainable" time profile refers to a $T \in [0,\infty)$ which, conditionally on

$$\begin{cases} V(\theta_0;T) > V(\theta_0;T') \\ \text{or} \\ V^P(\theta_0;T) > V^P(\theta_0;T') \end{cases}$$

makes the agency better off, that is:

$$\begin{cases} W^a(\theta_0;T) > W^a(\theta_0;T') \\ \text{or} \\ W(\theta_0;T) > W(\theta_0;T') \end{cases}$$

(6) T^* and T^{**} indicate the sustainable time profiles which make the agency better off under its own optimal rules and under the firm's optimal ones, respectively.

APPENDIX A

This appendix contains a general procedure to find a solution of the control problems presented in the text.

Let $F(\theta_t,t)$ be the maximum of the "value function" (market value for the firm, welfare value for the agency) at time t. If this function is differentiable, then $F(\theta_t,t)$ has to be a solution of the following dynamic programming equation:

$$\text{(A1)} \quad -F_t + rF = \max_m \left[\left(C\,\theta_t^{\phi} - m_t \right) + \left(m_t^{\xi}\,\theta^{-\gamma} - \delta \right)\theta_t\,F_\theta + \frac{1}{2}\sigma^2\,\theta_t^2\,F_{\theta\theta} \right]$$

where F_t, F_θ and Fee are partial derivatives of F with respect to the time and θ.

From the equation (A1) we are able to sum up both the firm optimization at the second stage when F = V^{II}, F_t = 0 and C= $C_{(a)}$, and at the first stage when F = V, C = 1 with the appropriate terminal condition at time T,respectively. Besides setting F = W^{II}, W_t = 0 and C = $C_{(p)2}$ we obtain the agency's optimization at the first stage, and, setting F = W, C = $C_{(p)1}$ and the terminal condition at time T, the agency's optimization at the second stage.

Equation (A1) is known as the Hamilton-Jacobi-Bellman equation of the stochastic version of the optimal control theory.

Differentiating the right-hand side of (A1) with respect to m, we get:

$$\text{(A2)} \qquad m_t = \left(\xi\,F_\theta\,\theta_t^{1-\gamma} \right)^{1/(1-\xi)}$$

Substituting (A2) into A(1) the latter becomes:

$$-F_t + rF = C\,\theta_t^{\nu} - \delta\,\theta_t F_\theta + \left(\frac{1-\xi}{\xi}\right) m_t + \frac{1}{2}\sigma^2\theta_t^2\,F_{\theta\theta} \tag{A3}$$

Equations (A3) together with (A2) can be expressed as a nonlinear second-order partial differential equation of parabolic type in F, which is solvable under some restrictions on the parameters of marginal productivity of soil quality and of maintenance technology.

Let us start with optimization at the first stage. Assuming $\xi = \frac{1}{2}$ and $\gamma = \frac{1}{2}\,\phi$ the Bellman equation (A3) reduces to:

$$F_t - rF + C\,\theta_t^{\phi} - \delta\,\theta_t F_\theta + \frac{1}{4}\theta_t^{2-\phi}\,F_\theta^2 + \frac{1}{2}\sigma^2\,\theta_t^2\,F_{\theta\theta} = 0 \tag{A4}$$

with the boundary conditions:

$$F(\theta_T;T) = \bar{S}\,\theta_T^{\phi} \quad , \bar{S} > 0 \text{ and constant}$$

$$F(0\,;t) = 0$$

where $\bar{S}$ stands for the scrape level of the value function at the terminal time T.

A functional form candidate for a solution of this partial differential equation is:

$$F(\theta_t,t;T) = S(t;T)\,\theta_t^{\phi} \tag{A5}$$

Taking the partial derivatives of (A5) with respect to t and θ yields:

$$F_t = S'(t;T)\,\theta_t^{\phi} \tag{A6.1}$$

$$F_\theta = \phi\,\theta_t^{-1}\,F \tag{A6.2}$$

$$F_{\theta\theta} = \phi(\phi-1)\,\theta_t^{-2}\,F \tag{A6.3}$$

Then the partial differential equation (A4) reduces to the ordinary differential equation:

$$\text{(A7)} \quad S'(t;T) = -\frac{1}{4}\phi^2 S^2(t;T) + (r + \delta\phi - \frac{1}{2}\phi(\phi-1)\sigma^2)S(t;T) - C$$

with boundary condition

$$S(T;T) = \bar{S}$$

Setting $A = \frac{1}{4}\phi^2$ and $B = (r+\delta\phi - \frac{1}{2}\phi(\phi-1)\sigma^2)$ the ordinary differential equation (A7) can be rewritten as:

$$\text{(A8)} \quad S' = -A S^2 + B S - C$$

(A8) is a Ricatti differential equation, which can be solved by separation of variables. The solution is:

$$\text{(A9)} \quad S(t) = \frac{S^{(2)} - S^{(1)}K \exp[A(S^{(1)} - S^{(2)})t]}{1 - S^{(1)}K \exp[A(S^{(1)} - S^{(2)})t]}$$

where the constant K is determined by the boundary condition (A7), and $S^{(1)}$ and $S^{(2)}$ are the solution of the second-order characteristic equation of the r.h.s of (A8), that is:

$$\text{(A10)} \quad S^{(1)} = \frac{1}{2A}\left(B - \sqrt{B^2 - 4AC}\right)$$

$$S^{(2)} = \frac{1}{2A}\left(B + \sqrt{B^2 - 4AC}\right)$$

In order for the value F to be positive, at least one of the two constants in (A10) must be positive. Under the hypothesis $B^2 - 4A > 0$, it follows, from the signs of (A10), that $0 < S^{(1)} < S^{(2)}$.

Now imposing the boundary condition (A7) to evaluate the constant K, we get:

$$\text{(A11)}\quad S(t;T) = \frac{S^{(2)} - S^{(1)}\left(\frac{\bar{S} - S^{(2)}}{\bar{S} - S^{(1)}}\right)\exp\left(\sqrt{B^2 - 4AC}\right)(T-t)}{1 - \left(\frac{\bar{S} - S^{(2)}}{\bar{S} - S^{(1)}}\right)\exp\left(\sqrt{B^2 - 4AC}\right)(T-t)}$$

It easy to check, from (A11), that $S^{(1)}$ is a locally asymptotically stable level of maximal expected discounted value if we let the horizon time T approach to infinite. In other words, letting T tend to infinite, the scrape value disappears and the root $S^{(1)}$ is necessary and sufficient for the value function F, i.e. the expected discounted flow of profit, to converge.

Finally the optimal expected value function can be written as (A5) with S(t;T) given by (A11).

Considering now the maximization at the second stage it is immediate to note that, since the horizon goes from T to infinite, it becomes time homogeneous? i.e. the scrape value is equal to zero and $F_t = 0$. The (A8) is no longer a differential equation but only a second-order characteristic equation in S, which gives two distinct roots as in (A10), Recalling that only $S^{(1)}$ **guarantees** the existence of F , the optimal expected value function will be as in (A5) with S(t;T) constant and equal to $S^{(1)}$.

Finally, it should be noted that with a stochastic differential equation such as [2] in the text, with $f(m,\theta)$ given by [81, there might be a positive probability that the process $\{\theta_s\}$ becomes zero (negative) or even infinite. On this matter Vorst (1987) and Moretto (1991) showed that under the optimal maintenance policy this probability is zero for the cases under analysis. In other words, the left boundary (zero) and the right boundary (infinite) are not attracting for the process $\{\theta_s\}$, at least in a finite expected time. In the rest of the paper we refer to this result in guaranteeing the necessary and sufficient conditions for the firm's value function [3], and the agency's welfare function [27] to exist (i.e. to be bounded).

APPENDIX B

From identity [37], if the r.h.s. is evaluated under the agency's management optimal rules, we get, at the beginning of the planning period:

$$\text{(B1)}\quad V^P(\theta_0;T) = W(\theta_0;T) + E_0\left\{ \int_0^T D_t x^*_{(p)t} e^{-rt} dt \ - \int_T^\infty \rho D_t x^{**}_{(p)t} e^{-rt} dt \right\}$$

where $\{\theta_t\}$ evolves according to [36] in $[0,T)$ and to [33] in $[T,\infty)$.

Since, as indicated in the text the following limits hold:

$$\lim_{T\to\infty} N(t;T) = N^I_1$$

$$\lim_{T\to 0} N(t;T) = N^{II}$$

it is possible to verify that, if T = 0:

$$\text{(B2)}\quad V^P(\theta_0;T{=}0) = N^{II}\ \theta_0^{\ \phi} \ - \ E_0\left\{ \int_0^\infty \rho D_t x^{**}_{(p)t} e^{-rt} dt \right\}$$

$$= \left[\frac{B-\sqrt{B^2-4AC_{(p)2}}}{2A} \ - \rho\left(\frac{\alpha}{1-\rho}\right)^{1/(1-\alpha)} \frac{1}{\sqrt{B^2-4AC_{(p)2}}} \right]\theta_0^{\ \phi}$$

whilst, if T = ∞ :

$$\text{(B3)}\qquad V^P(\theta_0;T=\infty) = {}_1N^I\ \theta_0^{\ \phi} + E_0\left\{\int_0^\infty D_t x^*_{(p)t} e^{-rt} dt\right\}$$

$$= \left[\frac{B-\sqrt{B^2-4AC_{(p)1}}}{2A} + (\alpha)^{1/(1-\alpha)}\frac{1}{\sqrt{B^2-4AC_{(p)1}}}\right]\theta_0^{\ \phi}$$

Equations (B2) and (B3) allow us to examine the trends of (B1) when T tends to zero or infinite. Recalling that $V(\theta_0;T) = M(0;T)\theta_0^{\ \phi}$, if $T = 0$:

$$V^P(\theta_0;T=0) \gtrless V(\theta_0;T=0)$$

$$\Leftrightarrow$$

$$\sqrt{B^2-4AC_{(a)}} - \sqrt{B^2-4AC_{(p)2}} \gtrless \rho\left(\frac{\alpha}{1-\rho}\right)^{1/(1-\alpha)}\frac{2A}{\sqrt{B^2-4AC_{(p)2}}}$$

whilst if $T = \infty$:

$$V^P(\theta_0;T=\infty) \gtrless V(\theta_0;T=\infty)$$

$$\Leftrightarrow$$

$$\sqrt{B^2-4AC_{(p)1}} - \sqrt{B^2-4AC} \lessgtr (\alpha)^{1/(1-\alpha)}\frac{2A}{\sqrt{B^2-4AC_{(p)1}}}$$

Since $C_{(a)} = C_{(p)1}$, by combining the above inequalities, we get the four situations described in the text.

To analyze the behavior of (B1) in the interval $[0,\infty)$ we can take the first and second derivative with respect to T. The first derivative yields:

$$\text{(B4)}\qquad \frac{dV^P}{dT} = \frac{dW}{dT} + e^{-rT}(1+\rho)\ E_0\left(D_T\ x^{**}_{(p)T}\right)$$

Since $\frac{dW}{dT} < 0$, and $E_0\left(D_T x^{**}_{(p)T}\right)$ is positive, the sign of (B4) is not determined a priori. Moreover, notice that (B4) describes the "trade-off" between the firm's marginal loss when T increases, evaluated according to the agency welfare function, and the expected marginal benefit the firm will receive, in terms of reduced tax payments, when the introduction of fees is delayed.

Taking the second derivative we obtain:

$$\text{(B5)}\quad \frac{d^2V^P}{dT^2} = \frac{d^2W}{dT^2} - re^{-rT}(1+\rho)E_0\left(D_T\, x^{**}_{(p)T}\right) + e^{-rT}(1+\rho)E_0\left(\frac{d(D_T x^{**}_{(p)T})}{dT}\right)$$

where:

$$\frac{d^2W}{dT^2} = \frac{1+Ke^{(\sqrt{B^2-4AC_{(p)1}})T}}{1-Ke^{(\sqrt{B^2-4AC_{(p)1}})T}}\sqrt{B^2-4AC_{(p)1}}\;\frac{dW}{dT} > 0$$

$$K = \frac{N^{II}-N^{I}_{2}}{N^{II}-N^{I}_{1}} = -\frac{\sqrt{B^2-4AC_{(p)1}}+\sqrt{B^2-4AC_{(p)2}}}{\sqrt{B^2-4AC_{(p)1}}-\sqrt{B^2-4AC_{(p)2}}} < -1$$

$$E_0\left(\frac{d(D_T x^{**}_{(p)T})}{dT}\right) = \left[r - \int_0^T \frac{1}{2}\phi^2\frac{dN(t;T)}{dT}dt - \sqrt{B^2-4AC_{(p)2}}\right]E_0\left(D_T\, x^{**}_{(p)T}\right)$$

The last expression is derived from [241 substituting N(t;T) instead of M(t;T). Considering that $\frac{dN(t;T)}{dT} < 0$, it iseasy to check that as T tends to zero the second derivative can be positive, whilst as T tends to infinite, it becomes negative. In other words, depending on the value assumed by the technical parameters V^P may be downward sloping and convex when T is close to zero and upward sloping and concave as T increases, with a minimum given by $\frac{dV^P}{dT} = 0$, as shown in the text.

In the same way, from [37] we can obtain the welfare function evaluated under the firm's rules:

$$\text{(B6)} \quad W^a(\theta_0;T) = V(\theta_0;T) - E_0\left\{ \int_0^T D_t x^*_{(a)t} e^{-rt}dt - \int_T^\infty \rho D_t x^{**}_{(a)t} e^{-rt}dt \right\}$$

where $\{\theta_t\}$ evolves according to [21] in [0,T) and to [15] in $[T,\infty)$.

Again taking account of the following limits:

$$\lim_{T\to\infty} M(t;T) = M_1^I$$

$$\lim_{T\to 0} M(t;T) = M^{II}$$

it is possible to verify that, if T = 0:

$$\text{B7)} \quad W^a(\theta_0;T=0) = M^{II}\ \theta_0^{\ \phi} + E_0\left\{ \int_0^\infty \rho D_t x^{**}_{(a)t} e^{-rt}dt \right\}$$

$$= \left[\frac{B-\sqrt{B^2-4AC_{(a)}}}{2A} + \rho(\ \alpha\ )^{1/(1-\alpha)} \frac{1}{\sqrt{B^2-4AC_{(a)}}} \right]\theta_0^{\ \phi}$$

hilst, if T = ∞ :

$$\text{B8)} \quad W^a(\theta_0;T=\infty) = M_1^I\ \theta_0^{\ \phi} - E_0\left\{ \int_0^\infty D_t x^*_{(a)t} e^{-rt}dt \right\}$$

$$= \left[\frac{B-\sqrt{B^2-4A}}{2A} - \frac{1}{\sqrt{B^2-4A}} \right]\theta_0^{\ \phi}$$

Equations (B7) and (B8) allow us to examine the trends of (B6) when T tends to zero or infinite. Recalling that $(\theta_0;T) = N(0;T)\theta_0^{\ \phi}$, if T = 0:

$$W^a(\theta_0;T=0) \gtrless W(\theta_0;T=0)$$

$$\Leftrightarrow$$

$$\sqrt{B^2-4AC_{(a)}} - \sqrt{B^2-4AC_{(p)2}} \lessgtr \rho(\alpha)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2-4AC_{(a)}}}$$

whilst if T = ∞:

$$W^a(\theta_0;T=\infty) \gtrless W(\theta_0;T=\infty)$$

$$\Leftrightarrow$$

$$\sqrt{B^2-4AC_{(a)}} - \sqrt{B^2-4A} \gtrless \frac{2A}{\sqrt{B^2-4A}}$$

Moreover, since:

$$\rho\left(\frac{\alpha}{1-\rho}\right)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2-4AC_{(p)2}}} > \rho(\alpha)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2-4AC_{(a)}}}$$

$$(\alpha)^{1/(1-\alpha)} \frac{2A}{\sqrt{B^2-4AC_{(a)}}} < \frac{2A}{\sqrt{B^2-4A}}$$

Confronting the above inequalities with those of V^P and V, we get the four cases shown in figs. 9-12.

Finally, to analyze the behavior of (B6) within the interval $[0,\infty)$ we take the first and second derivative with respect to T. The first yields:

$$\text{(B)} \qquad \frac{dW^a}{dT} = \frac{dV}{dT} - e^{-rT}(1+\rho)\, E_0\left(D_T\, x^{**}_{(a)T}\right)$$

Since $\frac{dV}{dT} > 0$, and $E_0\left(D_T x^{**}_{(a)T}\right)$ is positive, the sign of (B9) is not determined a priori. The first term on the r.h.s. represents the agency's marginal gain when T increases, evaluated according to the firm's value function. The second term, in turn, is the expected marginal loss the agency will incur, in terms of reduced tax payments, when the introduction of fees is delayed.

Taking the second derivative we obtain:

$$\text{(B10)}\quad \frac{d^2W^a}{dT^2} = \frac{d^2V}{dT^2} + re^{-rT}(1+\rho)E_0\left(D_T\, x^{**}_{(a)T}\right) - e^{-rT}(1+\rho)E_0\left(\frac{d(D_T x^{**}_{(a)T})}{dT}\right)$$

where:

$$\frac{d^2V}{dT^2} = \frac{1+K'e^{(\sqrt{B^2-4AC_{(a)}})T}}{1-K'e^{(\sqrt{B^2-4AC_{(a)}})T}}\sqrt{B^2-4AC_{(a)}}\;\frac{dV}{dT} < 0$$

$$K' = \frac{M^{II}-M^I_2}{M^{II}-M^I_1} = -\frac{\sqrt{B^2-4AC_{(a)}}+\sqrt{B^2-4A}}{\sqrt{B^2-4A}-\sqrt{B^2-4AC_{(a)}}} > 1$$

$$E_0\left(\frac{d(D_T x^{**}_{(a)T})}{dT}\right) = \left[r - \int_0^T \frac{1}{2}\phi^2\,\frac{dM(t;T)}{dT}\,dt - \sqrt{B^2-4AC_{(a)}}\right]E_0\left(D_T\, x^{**}_{(a)T}\right)$$

where the last expression is derived from [24]. Considering that $\frac{dM(t;T)}{dT} > 0$, it is easy to check that as T tends to zero the second derivative can be negative, whilst as T tends to infinite, it becomes positive. In other words, depending on the value assumed by the technical parameters, W^a may be upward sloping and concave when T is close to zero and downward sloping and convex as T increases, with a maximum given by $\frac{dW^a}{dT} = 0$, as shown in the text.

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INTERTEMPORAL INCENTIVES AND MORAL HAZARD IN NONPOINT-SOURCE POLLUTION

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INTERTEMPORAL INCENTIVES AND MORAL HAZARD IN NONPOINT-SOURCE POLLUTION

1. Introduction

The inability to measure with sufficient accuracy the emissions of individual dischargers in nonpoint-source pollution renders inadequate the standard instruments of environmental policy (Pigouvian taxes, controls, etc.) as a means of inducing dischargers to follow socially desirable policies. Measurement of ambient pollution concentration at some “receptor point” without any possibility of inferring individual emissions is a source of moral hazard. Dischargers will choose higher than socially desirable emission levels if by doing so they can increase their profits.

This type of moral hazard problem can be prevented by introducing incentive schemes that include penalties, rewards, or some combination of both that depend on deviations between desired and measured ambient concentration levels (Meran and Schwalbe 1987, Segerson 1988, Xepapadeas 1991). These schemes, however, are essentially static since they ignore the dynamic process of the pollutant accumulation and the effects it might have on individual behavior if it is imposed, through some incentive scheme, as a restriction on the dischargers’ intertemporal profit maximization problem.

The purpose of this paper is to explore the possibilities of designing intertemporal incentive schemes that would induce dischargers to follow a policy resulting in a socially-desirable long run equilibrium concentration level. In this pollutant context, differentiable incentive schemes depending on deviations between desired and observed ambient levels and the social dynamic shadow cost of pollutant concentration are examined under conditions of certainty and uncertainty. In the latter case, uncertainty is associated with random natural pollution decay rate and results in incentive dependence on the mean and variance of the pollutant concentration along the socially optimal path. The problem is analyzed in a differential game framework. Incentives corresponding to open-loop, feedback and perfect conjectural Nash equilibrium (Basar and Olsder 1982, Fershtman 1987, Fershtman and Kamien 1985) are formulated. The resulting schemes are not in general similar to the static ones, by which optimal behavior can be secured by penalizing each discharger with the full social cost of deviations from desired ambient levels (Segerson 1988, Xepapadeas 1991). As a result, adoption of static rules might lead to socially inefficient outcomes in the long run.

The paper is organized as follows. The second section analyzes the path of pollutant accumulation resulting from the objectives of maximizing either profit or some welfare indicator that accounts for damages due to environmental pollution, using deterministic and stochastic specifications. The long run equilibrium accumulation levels of the pollutant under the different objectives are determined and compared. In the third section, the efficient incentive schemes are constructed under different assumptions about the information structure that the discharging firms use to determine their strategies with respect to their emissions, and the stochastic properties of the model. Efficient schemes are determined as unit charges on deviations between observed and desired pollutant accumulation levels. Comparisons of the equilibrium pollutant accumulation 1evels under static and dynamic incentives are also performed. The last section provides some concluding remarks.

2. Long run pollutant accumulation

2.1 Model description

We consider a market consisting of i=1,...,n firms that produce a homogeneous product. output production generates pollution, that can be abated by using additional resources. The benefit of the ith firm at each instant of time can be written as a function of its discharges (Malik 1990)

$$B_i=B_i(e_i(t)),B_i'>0,B_i''<0 \quad for\ all\ t\in[0,\infty) \qquad (1)$$

where $e_i \in E_i \subset R_+$ denotes discharges of the ith firm and E_i is. assumed to be compact and convex.[1]

Let $x \in X \subset R_+$ with X compact and convex, denote the ambient concentration of the pollutant generated by the firms' productive activities. It is assumed that environmental uncertainty associated with factors like weather or topographical conditions result in random natural pollution decay rate (Plourde and Yeung 1989, Xepapadeas 1990). The evolution of the pollutant concentration can be described by the stochastic differential equation

$$dx(t)=[\Sigma_i e_i(t)-bx(t)]dt+\sigma x(t)dz(t), x(0)=x_0 \text{ non-random} \qquad (2)$$

where {z(t)} is a Wiener process. [2]

In (1), $\sum e_i - bx$ is the instantaneous expected change of x per unit of time, with (-bx) being the mean of the natural decay process, assuming exponential natural pollution decay rate, and $(\sigma x)^2$ is its instantaneous variance. Hence the accumulation of the pollutant follows a diffusion process. The effects of the pollutant accumulation are described by a damage function

$$D=D(x(t)),\ D'>0,\ D''>0, \text{ for all } t\in[0,\infty) \qquad (3)$$

The upper bound assumption on the pollutant accumulation means that $D(x)\rightarrow\infty$ as $x\rightarrow x_{max}$ (Kamien and Schwartz 1982). Relations (1) - (3) can be used to analyze the long run pollutant accumulation resulting from profit maximization or from the maximization of some welfare indicator.

2.2 Profit maximization

The ith firm will choose the discharge level that maximizes (1), resulting in first order conditions

$$B_i'(\hat{e}_i)\leq 0, \hat{e}_i\geq 0 \quad i=1,\dots,n \quad all\ t\in[0,\infty) \qquad (4)$$

The dynamics of environmental change, due to profit maximizing behavior, are obtained by substituting $\hat{e}_i$ into (2). To simplify things at a first stage, consider the certainty case $\sigma=0$. From (2) we obtain

$$\dot{x}=\Sigma_i\hat{e}_i-bx \qquad (5)$$

The long run pollutant accumulation corresponding to profit maximizing behavior is the equilibrium point $(x_\infty)\in X$ such that $\dot{x}=0$. From (5) we obtain

$$x_\infty=(\sum_i\hat{e}_i/b)$$

Since b>O, the equilibrium point is globally asymptotically stable.

When the uncertainty case is examined, the evolution of the pollutant concentration is determined by the solution of (2) with $e_i=\hat{e}_i$ all i and t. This stochastic differential equation has a unique solution as a diffusion process with drift coefficient $(\sum_i\hat{e}_i-bx)$ and diffusion coefficient $(\sigma x)^{2(3)}$. The diffusion property implies that given some pollutant accumulation x at time t, the probability that in some future time the pollutant accumulation will fall within the interval (a_1,a_2), $0<a_1<a_2<\infty$ can be determined.

Proposition 2.1 Let $(x_m)_\infty$ denote the expected long run pollutant concentration , then $(x_m)_\infty=x_\infty$.

Proof By taking expected values in the integral form of equation (2), using the properties of stochastic integrals (Karlin and Taylor 1981), and denoting $E(x)=x_m$ we obtain the following differential equation for x_m (Gard 1988, theorem 4.5)

$$\dot{x}_m=\Sigma\hat{e}_i-bx_m \text{ with } x_m(0)=x_o$$

which has equilibrium solution

$$(x_m)_\infty = \Sigma \hat{e}_i / b = x_\infty \qquad \blacksquare$$

Thus, in the long run the expected pollutant concentration is the same as in the deterministic case. [4]

2.3 Social optimum

A social planner will seek the discharge levels that maximize expected net benefits, that is

$$\max_{e_i} E_o \int_o^\infty \exp - \rho t \, [\Sigma_i B_i(e_i) - D(x)] \, dt$$

subject to (2) and $e_i \in E_i$ all $t \in [o, \infty)$ and i=1,...,n (P. 1)

where p>0 denotes the discount rate.

The cases of certainty and uncertainty are examined in the following.

(i) Certainty, $\sigma=0$

The current value Hamiltonian for problem (P.1) can be written as

$$H(x, e_1, \ldots, e_n, \lambda) = \Sigma_i B_i(e_i) - D(x) + \lambda(\Sigma_i e_i - bx) \qquad (6)$$

where $\lambda(t)$ can be interpreted as the dynamic social shadow cost of pollutant concentration.

The necessary and sufficient conditions for optimality, since H is concave in x, e_i (Seierstad and Sydsaeter 1987), are

$$\frac{\partial H}{\partial e_i} = B_i'(e_i^*) \leq -\lambda \ , \ e_i^* \geq 0 \quad i=1, \ldots, n \qquad (7.1)$$

$$\dot{\lambda} = (\rho + b)\lambda + D'(x) \qquad (7.2)$$

equation (2) with $\sigma=0$, and the Arrow type transversality conditions.

From (7.1) we obtain for interior solutions

$$e_i^* = e_i(\lambda) \tag{8}$$

with $e_i' = -1/B_i'' > 0$ by the implicit function theorem. Thus for $\lambda < 0$, a reduction in absolute value of the social shadow cost of the pollutant will increase discharges.

Substituting (8) into (2) and using (7,2), we obtain the Modified Hamiltonian Dynamic System (MHDS)

$$\dot{x} = \Sigma_i e_i(\lambda) - bx \tag{9.1}$$

$$\dot{\lambda} = (\rho + b)\lambda + D'(x) \tag{9.2}$$

The properties of the long run equilibrium for the pollutant accumulation that corresponds to the social optimum are summarized in the following proposition.

Proposition 2.2

(i) A unique optimal long run equilibrium (steady state) for the pollutant accumulation and its social cost defined as $(x^{*\infty}, \lambda^{*\infty}) : \dot{x} = 0, \dot{\lambda} = 0$, exists.

(ii) The steady state is a local saddle point.

(iii) For a small discount rate, the steady state is globally asypmtotically stable for bounded solutions of (9.1), (9,2).

Proof

(i) The isocines for the MHDS are defined for $\dot{x} = \dot{\lambda} = 0$. Solving (9.1) for x and substituting in (9.2), we obtain

$$g(\lambda)=(\rho+b)\lambda+D'(\frac{1}{b}\Sigma_i e_i(\lambda))$$

$g(\lambda)$ is continuous with g(0)>0. Also, since D'>0, $e'_i>0$ and $\lambda<0$, there exists a λ^+ sufficiently large in absolute value such that e_i and D' are sufficiently small, so that $g(\lambda^+)<0$. continuity of g implies a value λ^*_∞ : $\lambda^+<\lambda^*_\infty<0$ such that $g(\lambda^*_\infty)=0$. By the monotonicity of g, λ^*_∞ is unique. Then the unique equilibrium value for the pollutant accumulation is

$$x^*_\infty=\frac{1}{b}\Sigma_i e_i(\lambda^*_\infty)$$

(ii) The Jacobian of the MHDS around the equilibrium point is

$$J=\begin{vmatrix}-b & \Sigma_i e'_i\\ D'' & (\rho+b)\end{vmatrix}$$

since $|J|<0$ the equilibrium point is a local saddle point (e.g. , Beavis and Dobbs 1990). This means that there exists a one dimensional manifold M that contains the equilibrium point. For initial conditions $(x_0,\lambda_0)\in M$ in the neighborhood of the equilibrium point it holds for the optimal solution that $(x(t)),\lambda(t))$ tends to the equilibrium point as $t\to\infty$.

(iii) Let $K\subset X\times R$ be a compact set. A solution of the MHDS will be bounded if $(x(t),\lambda(t))\in K$ for all $t\in[0,\infty)$. Form the curvature matrix

$$Q=\begin{bmatrix}-H_{xx} & \frac{\rho}{2}\\ \frac{\rho}{2} & H_{\lambda\lambda}\end{bmatrix}=\begin{bmatrix}D'' & \frac{\rho}{2}\\ \frac{\rho}{2} & \Sigma_i e'_i\end{bmatrix},\quad H_{xx}=\frac{\partial^2 H}{\partial x^2},\quad H_{\lambda\lambda}=\frac{\partial^2 H}{\partial\lambda^2}$$

This matrix is positive definite for small ρ. It follows from Brock and Scheinkman (1976) that the steady state is globally asymptotically stable for all bounded solutions, That is, for any initial condition on the stable manifold, the solution will converge to the steady state as $t \to \infty$. ■

The solution is illustrated in Figure 1.

Proposition 2.3 The optimal discharge levels for a profit maximizing firm, $\hat{e}_i$, will exceed the socially desirable levels for the same firm, e_i^*, i=1,...,n. As a result the corresponding equilibrium accumulation of the pollutant x_∞ will exceed the socially desirable level x^*_∞.

Proof $\hat{e}_i$ is defined as the solution of $B_i'(\hat{e}_i)=\lambda$ with $\lambda=0$ while e_i^* is defined as the solution of $B_i'(e_i^*)=\lambda$ with $\lambda<0$. Since $e_i'(\lambda)>0$, it follows that $\hat{e}_i>e_i^*$ all t. This implies that for all t, $x(t)>x^*(t)$ and in equilibrium $x_\infty>x^*_\infty$. This can also be seen from Figure 1 where x_∞ corresponds to the equilibrium accumulation for $\lambda=0$. ■

This proposition implies that if individual discharges were observable, an effluent tax set as $\tau(t)=-\lambda(t)$ would secure socially optimal discharge and accumulation levels. If, however, perfect monitoring is not possible, moral hazard would appear. To demonstrate that, let $\xi_i \in [0,1]$ be the probability that firm i 's emissions will be observed. For $\xi_i=1$ there is perfect monitoring while for $\xi_i=0$ the firm's emissions cannot be observed at all. For $\xi_i \in (0,1)$ there is incomplete monitoring that can be associated, for example, with random measurement of firms' emissions. The profit maximization problem for firm i becomes

$$\max_{e_i} [(1-\xi_i)B_i(e_i)+\xi_i[B_i(e_i)+\lambda e_i]], \ i=1,\ldots,n, \quad t\in[0,\infty)$$

The first order conditions for interior solution are

$$B_i'(e_i^+)=-\xi_i\lambda$$

It is clear that only in the case of perfect monitoring, that is $\xi_i=1$ for all i , $e_i^+=e_i^*$ and the socially desirable pollutant accumulate on level can be obtained through an effluent tax. Under incomplete monitoring, $\xi_i\epsilon[0,1)$ and $e_i^+>e_i^*$ with $e_i^+=\hat{e}_i$ in the extreme case of $\xi_i=0$. Since $B_i(e_i^+)>B_i(e_i^*)$ the individual firms will not adopt the socially optimal discharge policy because by not doing so they can increase their expected profits. In terms of Figure 1, this implies that for $\xi\epsilon(0,1)$, the long run pollutant accumulation level will be between $x^{*\infty}$ and x^{∞}.

(ii) Uncertainty, $\sigma>0$

Applying the stochastic maximum principle (Malliaris and Brock 1982) , the generalized current value Hamiltonian is

$$H=\Sigma_i B_i(e_i)-D(x)+\lambda(\Sigma_i e_i-bx)+\frac{1}{2}(\sigma x)^2\lambda_x$$

where $\{\lambda(t)\}$ is a random process reflecting the social shadow cost of pollutant accumulation and

$$\lambda=\frac{\partial V(x)}{\partial x}=V_x \quad , \quad \lambda_x=\frac{\partial^2 V(x)}{\partial x^2}=V_{xx}$$

with V(x) being the optimal value function. This function solves the Hamilton-Bellman-Jacobi (H-B-J) equation

$$\rho V = \max_{e_i}\{\Sigma_i B_i(e_i) - D(x) + V_x(\Sigma_i e_i - bx) + \frac{1}{2}(\sigma x)^2 V_{xx}\} \qquad (11)$$

Since the Hamiltonian is concave in (x, e_i), the H-B-J is concave in x, thus $V_{xx}<0$, reflecting society's risk aversion. Assuming in the following a quadratic optimal value function, V_{xx} is independent of x.

The optimality conditions can be written as

$$B_i'(e_i^*) \le -\lambda \quad , \quad e_i^* \ge 0 \quad all \quad i \;, \quad t \qquad (12.1)$$

$$d\lambda = [(\rho+b)\lambda + D'(x) - \sigma^2 x\lambda_x]\,dt + (\sigma x)\lambda_x dz \qquad (12.2)$$

along with equation (2) , and a transversality condition (Brock and Magill 1979) that holds for the optimal random processes x-(t) and $\lambda^*(t)$ which solve problem (P.1).

$$\sup E_o[\exp(-\rho t) \quad \lambda(t)x^*(t)] \le 0 \quad as \quad t \to \infty \qquad (12.3)$$

From (12.1) the optimal discharge for interior solutions is determined as $e_i^* = e_i(\lambda)$. By substituting e_i^* into (12.2) and (2), a stochastic MHDS can be obtained. fissuring that a unique solution exists, this solution will be a diffusion process, with drift coefficients

$$[(\rho+b)\lambda + D'(x) - \sigma^2 x\lambda_x] \quad , \quad [\Sigma_i e_i(\lambda) - bx]$$

respectively and diffusion coefficients $(\sigma x\lambda_x)^2$ and $(\sigma x)^2$ respectively.

Some steady state properties. of the stochastic version of the MHDS defined above can also be presented. Assume the following:

(A.1) There exists a compact convex set $K \subset X \times R$ such that for all stochastic processes $x(t)$, $\lambda(t)$ with initial non-random conditions x_0, λ_0 that are solutions of the MHDS, it holds $(x(t), \lambda(t)) \in K$ for all $t \in [0, \infty)$.

(A.2) The optimal value function $V(x)$ is strictly concave and differentiable.

(A.3) The transversality condition (12.3) is satisfied.

Proposition 2.4 Under assumptions (A.1) - (A.3) and for small ρ it holds that:

(i) All bounded solutions $(x(t), \lambda(t)) \in K$ converge almost surely as $t \to \infty$ to the optimal processes $(x^*(t), a^*(t))$, that are solutions of the problem (P.1) for any non-random initial condition $(x_0, \lambda_0) \in K$.

(ii) There exists a distribution function $F(x)$ for the pollutant accumulation such that the distribution functions $F_t(x), t \in [0, \infty)$ converge to $F(x)$ as $t \to \infty$. $F(x)$ is the steady state distribution.

Proof

(i) Define the generalized curvature matrix

$$Q = \begin{bmatrix} -H_{xx} & \frac{\rho}{2} \\ \frac{\rho}{2} & H_{\lambda\lambda} \end{bmatrix} = \begin{bmatrix} D'' & \frac{\rho}{2} \\ \frac{\rho}{2} & \Sigma_i e_i' \end{bmatrix} \quad , \quad Q_{\lambda_x \lambda_x} = 0$$

Q is positive definite for small ρ, while the second derivative with respect to λ_x is non-negative. It follows, then from Brock and Magill (1979) that $x(t) \to x^*(t)$, $\lambda(t) \to \lambda^*(t)$ as $t \to \infty$ for any non-random initial condition.

(ii) Let $e_i^*(x) = e_i(x, V_x(x), V_{xx}(x))$ be the optimal policy function where V(x) solves the H-B-J equation and also let $f^*(x)=\sum e_i^*(x)-b(x)$, $\sigma^*(x)=\sigma x$. By proposition (i) above, $\|x(t)-x^*(t)\|\to 0$ in probability as $t\to\infty$, if also f*(x) satisfies a Lipshitz condition then following Brock and Magill (1979), there exists a stationary distribution for the pollutant accumulation. ■

Proposition 2.5 Let $(x_m^*)_\infty$ denote the expected long run equilibrium concentration of the pollutants then $(x_m^*)_\infty < x^*_\infty$.

Proof Take expected values in the integral form of the equations of the stochastic MHDS and denote $E(x)=x_m$ and $E(\lambda)=\lambda_m$. The following system of differential equations can be obtained for the expected values

$$\dot{\lambda}_m=(\rho+b)\lambda_m+E[D'(x)]-\sigma^2 x_m\lambda_x \qquad (13.1)$$

$$\dot{x}_m=\Sigma_i e_i(\lambda_m)-bx_m \qquad (13.2)$$

where $e_i(\lambda_m)$ is obtained by taking expected values in (12.1) and then solving for e_i^*.

In long run equilibrium, $\dot{x}_m=\dot{\lambda}_m=0$ [5], then from (13.1) we obtain

$$\lambda_m=\frac{-E[D'(x)]+\frac{1}{2}\sigma^2 x_m\lambda_x}{(\rho+b)}$$

By Jensen's inequality $E[D'(x)]\geq D'(x_m)$, also $\lambda_x<0$. Therefore the isocine corresponding to $\dot{\lambda}_m=0$, lies everywhere below the isocine corresponding to $\dot{\lambda}=0$ for the deterministic case, and is relatively steeper (Fig. 1). This implies that $(x_m^*)_\infty < x^*_\infty$. ■

Thus society's risk aversion as reflected by the risk premium $-\sigma^2 x\lambda_x>0$ causes expected socially optimal pollutant concentration in the long run to be lower than the corresponding level for the deterministic case. The stability properties of system (13) are the same as those of the deterministic case.

Proposition 2.6 The expected long run pollutant concentration $(x_m)_\infty$ under profit maximization exceeds the corresponding socially desirable pollutant concentration $(x_m^*)_\infty$.

Proof This follows directly from propositions (2.1), (2.3) and (2.5). By these propositions, the following relation holds

$$(x_m)_\infty = x_\infty > x^*_\infty > (x_m^*)_\infty$$

The result is also illustrated in Figure 1. ■

By the same proposition it follows that the deviation between profit maximizing and socially optimal concentration levels is greater under uncertainty. Thus although moral hazard can appear in a deterministic model in the absence of monitoring, the presence of uncertainty intensifies the problem in the sense that the gap between socially desirable and actual (profit maximizing) levels is greater relative to the deterministic case.

3. Intertemporal incentives

3.1 Efficient incentives and strategy spaces

In the absence of monitoring of individual discharges, the social planner can observe only deviations between desired and actual pollutant concentrations at some receptor point. The objective is to introduce an incentive scheme such that individual dischargers will be induced to follow a policy leading to a long run socially desirable level of pollutant accumulation. An incentive scheme with this property should depend on deviations between observed and desired levels at each instant of time. If deviations are observed, every potential discharger pays a penalty. If no deviations are observed, then no penalties are imposed. Deviation dependence is desirable since it can provide a basis for practical implementation. Furthermore, in the incentive scheme, deviations should be valued according to the social valuation of the pollutant accumulation. A scheme achieving the social planner's objective will be called efficient.

Condition for efficient incentive schemes Let $x(t)-x^*(t)$ be the deviation between observed and desirable pollutant concentration levels, with $x^*\infty$ being the socially optimal equilibrium level as $t\to\infty$. Let $\phi\equiv\phi(x(t)-x^*(t))$ be a function such as

$$\phi \gtreqless 0 \text{ as } x(t)-x^*(t) \gtreqless 0 \text{ with } \phi'>0$$

Let $x(\phi,t)$ be a pollutant accumulation path resulting when profit maximizer dischargers are subjected to the incentive scheme ϕ. The incentive scheme ϕ will be efficient if $x(\phi,t)\to x^*\infty$ as $t\to\infty$.

The requirement that the incentive scheme be such that it results in convergence to the equilibrium point seems reasonable, since immediate adjustment to the optimal path may require undesirable production cuts.

The analysis of incentive schemes is carried out in the context of an n-player non-cooperative dynamic game. The choice of the strategy space for these differential games depends on information structures (e.g., Basar and Olsder 1982, Fershtman and Kamien 1915). In the following, the analysis is limited to the often-employed open-loop and feedback structures.

Discharger i's information structure is said to be

(S1) Open-loop (OL) if $e_i=\theta_i(x_o,t)$ i=1,...,n

(S2) Feedback (FB) if $e_i=\theta_i(x,t)$ i=1,...,n

An (OL) or (FB) strategy is a time path $\{e_i(t)\}$ such that $e_i \in E_i$, all i and t.

Open-loop Nash equilibrium (OLNE) and feedback Nash equilibrium (FBNE) are defined for the strategy spaces (6) corresponding to the OL and FB information structures. OLNE solutions correspond to an infinite period of commitment. Players, that is dischargers, commit themselves to a particular path at the outset of the game and do not respond to observed variations of the pollutant concentrating. Discharge paths that constitute equilibrium for the game that starts at (x_o,t_o) do not constitute equilibrium for the game that starts at a different (x_o,t_o). Thus OLNE is not subgame perfect and this implies time inconsistency, a discharge policy that is optimal at the outset of the game is not optimal at a later period. Feedback strategies, on the other hand, depend on current ambient concentration levels. Firms do not commit themselves at the outset of the game and the FBNE is an equilibrium for any initial condition, thus it constitutes a subgame perfect (Selten 1975, Fershtman 1987,1988, Reiganum and Stokey 1985). The feedback equilibrium can be generalized to account for the conjectures of discharger i about the discharges of the rest of the firms. In this case, the strategy of the ith player is defined as

(S3) Feedback Complete Conjecture (FBC): $e_i=\theta_i\ (x,\underline{e}_{-i},t)$ where $\underline{e}_{-i}=(e_1,\ldots,e_{i-1},e_{i+1},\ldots,e_n)$. The FBCNE is a subgame perfect (Fershtman and Kamien 1985).

The payoff of firm i under the incentive scheme ϕ is defined as

$$J^i(e_1^*,\ldots,e_n^*)=E_0\int_0^\infty \exp-\rho t[B_i(e_i)-\phi(x-x^*)]\,dt \quad i=1,\ldots,n$$

each firm tries to maximize its payoff subject to (2) with $e_i \in E_i$. The OLNE (or FBNE or FBCNE) equilibrium is defined as an n-tuple of OL (or FB or FBC) strategies $(e_1^*,\ldots,e_n^*)$ where e_i is defined in (S1) - (S3) such that

$$J^i(e_1^*,\ldots,e_n^*)\geq J^i(e_1^*,\ldots,e_{i-1}^*,e_i,e_{i+1}^*,\ldots,e_n^*) \quad all\ i$$

In the following we examine the structure of the intertemporal efficient schemes that correspond to strategies (S1) - (S3).

3.1 Incentives under certainty

Efficient schemes corresponding to OLNE, FBNE and FBCNE are examined under the assumption that $\sigma=0$.

(i) OLNE

Proposition 3.1 Let $\lambda^*(t)<0$ be the social shadow cost of pollutant accumulation as defined by the solution of (P.1), then $\phi(x-x^*)=-\lambda^*(\rho+b)(x-x^*)$ is an efficient incentive scheme for OLNE.

Proof The current value Hamiltonian for the ith firm is defined as

$$H^i=B_i(e_i)+\lambda^*(\rho+b)(x-x^*)+\mu_i(e_i+\sum_{j\neq i}e_j^*-bx) \qquad (14)$$

where $\underline{e}^*_{-i}$ is the vector of the optimal responses of the rest of the firms. The necessary conditions for optimality are

$$B_i'(e_i^*) \leq -\mu_i, \quad e_i^* \geq 0 \quad all\ i \qquad (15.1)$$
$$\dot{\mu}_i = (\rho + b)(\mu_i - \lambda^*) \quad all\ i \qquad (15.2)$$
$$\dot{x} = \Sigma_i e_i(\mu_i) - bx \qquad (15.3)$$

Since ρ, b, λ are common for all i, it follows that $\mu_i = \mu, \forall i$. From (15.1) we obtain in equilibrium $(\dot{\mu}=0)$, that $\mu_\infty = \lambda^*_\infty$. Denote with x_∞^{OL} the equilibrium pollutant concentration under OLNE. From (15.3) we obtain

$$x(\phi, t) = x_\infty^{OL} \to \Sigma_i \frac{e_i(\lambda_\infty^*)}{b} = x_\infty^* \quad as\ t \to \infty \qquad \blacksquare$$

The result is illustrated in Figure 2.

The above incentive scheme is a type of effluent tax per unit of observable deviation between measured and desired accumulation levels. Under this scheme, once deviations are detected, every firm pays the same total amount(7), in contrast to the standard Pigouvian taxes where the total amount paid depends on individual discharges. It should also be noted that if past overdischarges caused deviation from the optimal path, then firms will pay the charge during the period of the adjustment to the optimal path, even if they currently follow optimal discharge policies.

The existence of a unique equilibrium point can be established by an argument similar to the one used in proposition (2.2.i). It is necessary, however, to examine the stability properties of the model in order to be sure that the proposed incentive scheme does not result in completely unstable equilibria.

<u>Proposition 3.2</u> The steady state solution $(x_\infty^{OL}, \mu_\infty)$ is

(i) A local saddle point.

(ii) Globally asymptotically stable for bounded, solutions.

<u>Proof</u>

(i) The Jacobian of the MHDS (15.3), (15.2) is defined as

$$J=\begin{vmatrix} -b & \sum_i e_i' \\ 0 & (\rho+b) \end{vmatrix}$$

since $|J|<0$, the equilibrium point has the saddle point property.

(ii) Following Sorger (1989), the curvature matrix can be written as (8)

$$K^{\gamma}=\begin{bmatrix} H_{xx}+\gamma\,[H_{x\mu}+H_{\mu x}-\rho]+\gamma^2 H_{\mu\mu} & -\frac{\rho}{2} \\ -\frac{\rho}{2} & -H_{\mu\mu} \end{bmatrix}=\begin{bmatrix} \gamma(-2b-\rho)+\gamma^2\Sigma_i e_i' & -\frac{\rho}{2} \\ -\frac{\rho}{2} & -\Sigma_i e_i' \end{bmatrix}$$

since $\mu(t)$ is bounded by the bounded solution assumption, if the e_i functions have bounded slope then there exists a finite $\gamma^+\epsilon$ $(0,(2b+\rho)/\sum e_i')$. For $\gamma=\gamma^+$ and sufficiently small ρ, the curvature matrix is negative definite. Thus the steady state is globally asymptotically stable for bounded solutions. ■

Solutions of the type described above suffer from the multiplicity of informationally non-unique Nash equilibria, A way of removing informational non-uniqueness is to restrict the equilibrium solution concept to a feedback Nash equilibrium (Basar and Olsder 1982). This type of restriction requires that players have access to the current value of the state. In the model described here, this is a plausible assumption since it can be assumed that the information that the social planner (environmental agency) has about the current accumulation of the pollutant becomes public knowledge without delay.

(ii) FBNE

In analyzing this type of equilibrium, the cross effects that describe conjectural variations make it very difficult if not impossible to study the problem in the general form. To obtain some insight into this type of equilibrium, a specific simple form for the conjecture function is assumed. In particular, for the conjecture function of firm i we assume

(AC1) $\quad e_j = c_j + ax \;,\; a<0 \;,\; j \neq i \;,\quad i,j=1,\ldots,n$

This conjecture function indicates that firm i expects other firms' discharge functions to contain an autonomous part and a part that depends linearly on current ambient concentration levels.

Proposition 3.2 Under (AC1) the efficient incentive scheme for FBNE takes the form

$$\phi(x-x^*) = -\lambda^*[(\rho+b)-(n-1)a](x-x^*) \quad (9)$$

Proof The current value Hamiltonian for this problem is

$$H^i = B_i(e_i) - \phi(x-x^*) + \mu_i\Big[e_i + \sum_{j\neq i}(c_j+ax) - bx\Big] \quad (16)$$

The optimality conditions are (15.1), (15.3) and

$$\dot{\mu}_i = [(\rho+b)-(n-1)a](\mu_i-\lambda^*) \quad i=1,\ldots,n \quad (17)$$

Thus $\mu_i = \mu$ all i. In equlilibrium $\mu_\infty = \lambda^*_\infty$, and

$$x(\phi, t) = x_\infty^{FB} - \sum_i \frac{e_i(\lambda_\infty^*)}{b} = x_\infty^* \quad as \ t \to \infty \qquad \blacksquare$$

The stability properties of the steady state can be analyzed in the same way as with OLNE. The equilibrium point is a local saddle point and globally asymptotically stable for bounded solutions.

(iii) FBCNE

To analyze this type of equilibrium, the following conjecture function for firm i is postulated

$$(AC2) \quad e_j = c_j + ax + \beta \sum_{k \neq j} e_k, \ j \neq i, \ i, j = 1, \ldots, n, \ k = 1, \ldots, j-1, j+1, \ldots, n \tag{10}$$

The third term of (AC2) reflects the fact that firm i expects firm j to adjust, its discharges by taking into account the discharges of all other firms. If we assume that all i=1,...,n firms are similar, it is not unreasonable to expect that the same weight be given to each of the rest if the firms' discharges, by every firm.

Proposition 3.4 Under (AC2) the efficient scheme for FBCNE takes the form

$$\phi(x-x^*) = -[\lambda^* - (n-1)\beta][(\rho+b) - (n-1)\alpha](x-x^*)$$

Proof The current value Hamiltonian for the problem is

$$H^i = B_i(e_i) - \phi(x-x^*) + \mu_i[e_i + \sum_{j \neq i}(e_j + ax + \beta \sum_{k \neq j} e_k) - bx] \tag{18}$$

The optimality conditions are

$$B_i'(e_i^*) \le -z_i \quad , \quad z_i = \mu_i + (n-1)\beta \quad , \quad e_i^* \ge 0 \qquad (19.1)$$
$$\dot{\mu}_i = [(\rho+b)-(n-1)a]\mu_i - [\lambda^* - (n-1)\beta][(\rho+b)-(n-1)a] \qquad (19.2)$$
$$\dot{x} = \Sigma_i e_i(z_i) - bx \qquad (19.3)$$

$\mu_i = \mu$ all i, thus in equilibrium we have $\mu_\infty = \lambda^*_\infty - (n-1)\beta$. Substituting into (19.3), we obtain

$$x(\phi, t) = x_\infty^{FBC} \to \frac{\Sigma_i e_i(\lambda_\infty^*)}{b} = x_\infty^* \quad as \quad t \to \infty \qquad \blacksquare$$

The stability properties of the steady state are similar to OL and FB solutions.

To compare the three incentive schemes derived above, let

$$\tau_1 = -\lambda^*(\rho+b)\,, \quad \tau_2 = \tau_1 + \lambda^*(n-1)a, \quad \tau_3 = \tau_2 + (n-1)\beta[(\rho+b)-(n-1)a]$$

denote the taxes per unit of deviation under OL, FB and FBC respectively. For a<0, $\tau_2 > \tau_1$. If firm i expects other firms to reduce their emissions when concentration increases, it has incentive to overdischarge, thus a relatively higher tax is required. A similar result applies if $\beta > 0 (a < 0)$. Then, $\tau_3 > \tau_2 > \tau_1$. If, however, $\beta < 0$, some inequalities might be reversed.

3.1.1 Comparison with static incentive schemes

In a static context, that is when the dynamics of the pollutant accumulation are ignored, the efficient differentiable incentive scheme charges the full social cost of deviations between observed and desired ambient concentration levels (Segerson 1988, Xepapadeas 1991).

This can be easily demonstrated along the lines of the model developed so far. The social planner solves the problem

$$\max_{e_i, x} \Sigma_i B_i(e_i) - D(x)$$
$$s.t. \quad x = \Sigma_i e_i$$

The optimality conditions imply

$$B_i'(e^*) = -\lambda^* = D'(x^*)$$

thus $\lambda < 0$ is the marginal social cost of pollutant concentration. Discharger i will follow a socially desirable policy if he faces the incentive scheme $\lambda^*(x - x^*)$. This is a standard non-balanced budgeting contract for preventing moral hazard in teams (Holmstrom 1982). Under this scheme, discharger i's problem is

$$\max_{e_i} B_i(e_i) + \lambda^* [(e_i + \sum_{j \neq i} e_j^*) - x^*]$$

where e^*_{-i} is the optimal response vector. Nash equilibrium implies $B_i'(e_i^*) = -\lambda^*$ which is the condition for the static social optimum. We are in a position now to analyze the implications. for the long run pollutant accumulation from the adoption of static incentive schemes.

Proposition 3.5 Under static differentiable incentive schemes of non-budget balancing type($\lambda^*(x - x^*)$), the long run pollutant concentration level compares as follows to the socially desirable level

$$(i)\ x_\infty^{OL} < x_\infty^*, \quad (ii)\ x_\infty^{FB} > x_\infty^{OL} \lesseqgtr x_\infty^*, \quad (iii)\ x_\infty^{FBC} \lesseqgtr x_\infty^*$$

Proof

(i) Under OLNE the following differential equation is satisfied for the shadow cost of the pollutant

$$\dot{\mu}=(\rho+b)\mu-\lambda^{*}$$
$$\textit{or at equilibrium } \mu_{\infty}^{OL}=\frac{\lambda_{\infty}^{*}}{\rho+b}<\lambda_{\infty}^{*}$$

under the plausible assumption that $0<\rho+b<1$, (where $\lambda,\mu<0$). This implies that

$$e_i(\mu_{\infty}^{OL})<e_i(\lambda_{\infty}^{*})\,\forall i \qquad , \qquad x_{\infty}^{OL}<x_{\infty}^{*}$$

The result is illustrated in Figure 3.

(ii) Under FBNE we have

$$\dot{\mu}=[\rho+b-(n-1)a]\mu-\lambda^{*}$$

or in equilibrium

$$\mu_{\infty}^{FB}=\frac{\lambda_{\infty}^{*}}{\rho+b-(n-1)a}>\mu_{\infty}^{OL}\lesseqgtr\lambda_{\infty}^{*} \quad as \quad [(\rho+b)-(n-1)a]\lesseqgtr 1 \ , \quad (a<0) \quad thus$$
$$e_i(\mu_{\infty}^{FB})>e_i(\mu_{\infty}^{OL})\lesseqgtr e_i(\lambda_{\infty}^{*}) \ , \quad x_{\infty}^{FB}>x_{\infty}^{OL}\lesseqgtr x_{\infty}^{*} \quad (Fig.\ 3)$$

(iii) Under FBC we obtain in the same way

$$e_i^{*}=e_i(z_{\infty}^{*}) \ \ where \ z_{\infty}^{*}=\mu_{\infty}^{FBC}+(n-1)\beta=\frac{\lambda_{\infty}^{*}}{(\rho+b)-(n-1)a}+(n-1)\beta\lesseqgtr\lambda_{\infty}^{*} \quad thus$$
$$x_{\infty}^{FBC}\lesseqgtr x_{\infty}^{*} \quad (Fig.\ 3) \qquad \blacksquare$$

The above results imply that "static" incentive schemes that charge the full cost of observed deviation between measured and desired pollutant concentration levels, when pollutant accumulation is a dynamic process, lead in general to supoptimal results. Under OL strategies, the "static" incentives lead to overabatement since their corresponding charge per unit deviation is greater than the charge required for long run convergence to the desired levels. Under more complicated strategies, however, the optimal dynamic charge per unit deviation could be higher than the static one, and in this case the static scheme would result in underabatement.

3.2 Incentives under uncertainty

We proceed to examine the structure of an efficient incentive scheme in the stochastic framework defined in (2) by analyzing the FBNE. (11) In the stochastic framework efficiency is defined in terms of equilibrium expected value of the pollutant concentration under the scheme. That is, the incentive scheme is efficient if

$$E[x(\phi,t)] \to E(x_\infty^*) = (x_m^*)_\infty \quad as \quad t \to \infty$$

with $(x_m^*)_\infty$ as defined in proposition 2.5.

Proposition 3.6 Under feedback strategies with conjecture functions as defined in (AC2), the efficient incentive scheme for the stochastic framework takes the form

$$\phi_i(x-x_m^*) = -\{[\lambda_m^*[(\rho+b)-(n-1)a] - \frac{1}{2}(\sigma^2 x)\mu_x^i](x-x^*) - \frac{1}{2}(\sigma^2 x)\mu_x^i x^*\}$$
$$where \ x_m^* = E[x^*(t)] \ , \ \lambda_m^* = E[\lambda^*(t)]$$

as defined in proposition 2.5 and $\mu_x^i = V_{xx}^i$ with V'(x) being the optimal value function that solves the H-B-J equation for the ith firm.

Proof The generalized current value Hamiltonian is defined as

$$H_i = B_i(e_i) - \phi^i(x - x^*) + \mu_i[e_i + (n-1)ax - bx] + \frac{1}{2}(\sigma x)^2 \mu_x^i$$

The optimality conditions can be written as

$$B_i'(e_i^*) \leq -\mu_i \qquad (20.1)$$

$$d\mu_i = [(\rho + b) - (n-1)a](\mu_i - \lambda_m^*)]dt + (\sigma x)\mu_x^i dz \qquad (20.2)$$

$$dx + [\Sigma_i e_i(\mu_i) - bx]dt + (\sigma x)dz \qquad (20.3)$$

Taking expected values for (20.2),(20.3) in integral form, we obtain the following system of differential equations for x_m, λ_m

$$\mu_m = [(\rho + b) - (n-1)](\mu_m - \lambda_m^*) \quad since \ \mu_m^i = \mu_m \ all \ i \qquad (21.1)$$

$$x_m = \Sigma_i e_i(\mu_m) - bx_m \qquad (21.2)$$

It can be easily seen that in equilibrium

$$E[x(\phi, t)] \rightarrow (x_m^*)_\infty \quad as \quad t \rightarrow \infty \qquad \blacksquare$$

The equilibrium point for system (21) is a local saddle point and globally asymptotically stable for bounded solutions, as can be easily shown by following the approach in proposition 2.2.

Some observations are in order with respect to the scheme of proposition 3.6. The deviations from the expected optimal path are valued according to their expected social cost and according to a risk premium that reflects discharger i's risk aversion. Thus the charge per unit deviation from the expected path is discriminatory. A uniform tax can be obtained under an assumption of "equal curvature" for optimal value functions $V^i_{xx} = V_{xx}$ all i.

The proposed incentive scheme might result in subsidies or penalties even if all firms follow desirable discharge policies because of random fluctuations that cause deviations from the expected path. One way of improving the scheme so that it does not result in a case where there is a continuous switch from subsidies to penalties and vice versa is to supplement it with a type of "confidence belt". Observed values outside this belt would not be regarded as resulting from random fluctuations and the charges would be imposed. One way of defining the belt is to use the diffusion property of the solutions of system (12.2), (2). By this property, a limit $x^+(t')$ can be defined such that the probability of the pollutant accumulate on exceeding this limit when all dischargers follow optimal policies is less than a predetermined probability, $Pr(x(t') \geq x^+(t')/e_i^*) \leq w$. If in time t', $x(t') > x^+(t')$, this can be regarded as establishing overemissions "beyond any reasonable doubt". Then the charge per unit deviation is triggered. Of course, if the excess accumulation of the pollutant is a result of events which are clearly beyond the control of the specified set of discharging firms (e.g., an environmental disaster caused by a third party), charges are not imposed. On the other hand, subsidies could never be paid. It is clear that they are the result of random factors since there is neither the incentive nor is it desirable to emit below the socially optimal levels.

4. Concluding remarks

Incentive schemes developed so far to deal with non point-source pollution problems are essentially static since they ignore the implication of the dynamics of pollutant accumulation, An attempt is made in this paper to develop incentive schemes which account for the dynamics of environmental change.

When individual emissions are not observable and consequently Pigouvian taxes are ineffective, profit maximizing firms emit more than is socially desirable. As a result, pollution accumulation levels exceed the respective socially optimal levels.

It has been demonstrated that incentive schemes can be constructed such that the path of pollutant accumulation under the scheme converges to the equilibrium socially desirable pollutant accumulation level. The schemes take the form of charges per unit of observed deviation between measured and desired levels. The charge depends on the pollutant's shadow cost , on the discount rate, on the natural pollution decay rate and on parameters associated with the information structure of the model. When discharging firms follow feedback strategies, the charge is higher as compared to the open-loop case. In general. , it is expected that firms will follow feedback strategies, since these types of strategies do not imply long period% of commitment and public information about the state of the pollutant accumulation makes their employment feasible. Similar incentive schemes can be constructed for a model with environmental uncertainty. The charge per unit deviation depends additionally on risk premiums under the appropriate risk aversion assumptions.

In general, application of static incentive schemes in dynamic situations results in suboptimalities, In particular, if dischargers adjust their emission policy according to current pollution accumulation (feedback strategies), application of static schemes may result in over accumulation of the pollutant in the long run.

Successful application of the incentive schemes requires the determination of the optimal path for pollutant accumulation. This could be a formidable task since it requires information on firms' production and abatement technologies, damages from pollutant accumulation, the natural characteristics of pollutant decay, and the information structure used by the discharging firms. On the other hand, the proposed scheme once approximated provides a flexible mechanism for dealing with dynamic non point-source pollution problems, since it can be treated as a simple Pigouvian tax on deviations from the optimal path. Incentives will, however, be ineffective if there are dischargers operating outside the incentive scheme and their contribution to the accumulating of the pollutant cannot be distinguished from the contribution of the dischargers that are subjected to the scheme.

FOOTNOTES

1. The benefit function is defined as

$$B_i(e_i) = \max_{q_i} \Pi(q_i, e_i) = \max_{q_i} [pq_i - C_i(q, e_i)] \quad all\ t$$

where q_i is the firm's output, p is the output price (reflecting marginal utility) and C_i (.,.) is a strictly convex cost function. Since Π (q_i, e_i) is concave in q_i, e_i, max $\Pi(q_i, e_i)$ is concave in e_i (Kamien and Schwartz 1981). We furthermore assume throughout the paper that private and social benefits and costs coincide.

2. The stochastic processes $\{x(t)\}$ and {z(t)} are defined as $x(t,\omega)$, $z(t,\omega)$ where $\omega \in \Omega$, $t \in [0,\infty)$ and $(\Omega, \mathcal{F}, P)$ is a complete probability space with $\mathcal{F}$ being a o-field on Ω and P a probability measure on $\mathcal{F}$. In the text, ω and in most cases t are suppressed.

3. A unique solution exists because the coefficient functions of (2) satisfy the Lipschitz and growth conditions and the initial condition is non-random (Gard 1988). Furthermore the solution is positive for the positive initial condition (Chang and Malliaris 1987).

4. A steady state probability distribution for the pollutant accumulation exists with the steady state density function satisfying (Malliaris and Brock 1982, Merton 1975):

$$\pi(x) = \frac{m}{(\sigma x)^2} \exp\left\{\int^x \frac{2(\Sigma_i \hat{e}_i - by)}{(\sigma y)^2} dy\right\}$$
$$with\ m: \int_0^\infty \pi(x)\, dx = 1$$

The existence of this distribution can be proven by showing that 0 and ∞ are repelling boundaries. This implies that the solution will neither explode or degenerate to zero (for this approach, see Gard 1986).

5. The existence of a unique steady-state can be shown in a similar way as is shown in proposition 2.2 by noting that by Jensen's inequality $E[D'(x)]\geq D'(x_m)>0$.

6. The strategy space is the set of all possible OL or FB strategies.

7. Since at this stage the model is deterministic and since firms have incentive to discharge more rather than less, it must hold that $x(t)\geq x^*(t)$, all t. Thus under certainty the scheme always works as a tax and not as a subsidy.

8. This approach is used because the maximized Hamiltonian is linear in x, that is $H_{xx}=0$.

9. The structure of the incentive scheme depends on the conjecture function. For example, if

$$e_j=c_j+\frac{1}{2}ax^2 \quad then$$
$$\phi=-\lambda^*[(\rho+b)(x-x^*)-(n-1)\frac{a}{2}(x-x^*)^2]$$

10. For $\beta=0$ we are in FB strategies while for $a=\beta=0$ we are in OL strategies.

11. Formulating the problem in the stochastic framework eliminates informational non-uniqueness. Furthermore it can be shown that if the players do not have access to the current state (as for example in OLNE), then general conditions for Nash equilibri-

um solutions under uncertainty cannot be obtained (Basar and Olsder 1982).

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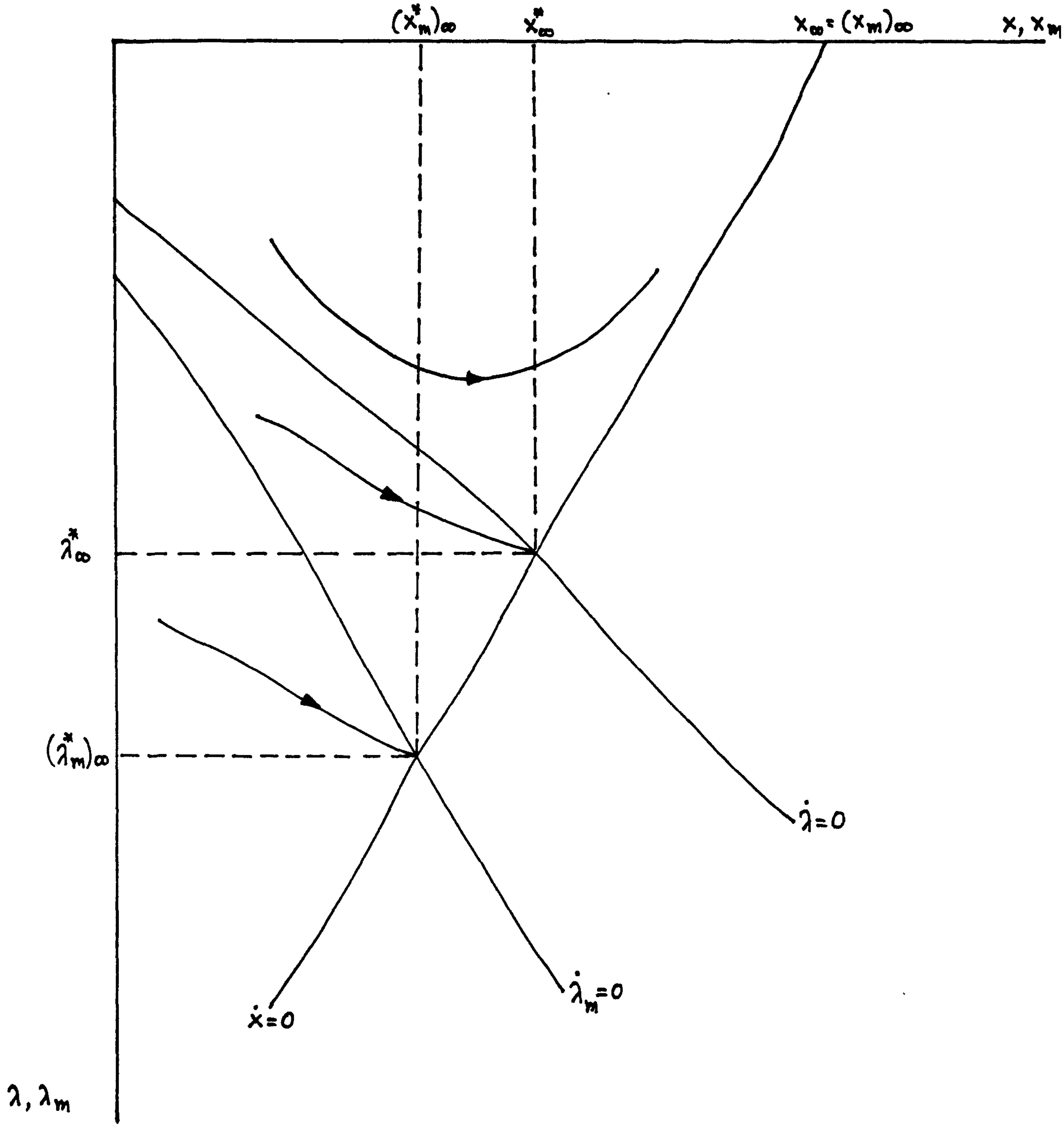

$(x_m^*)_\infty$
x_∞^*
$x_\infty = (x_m)_\infty$
x, x_m
λ_∞^*
$(\lambda_m^*)_\infty$
$\dot{\lambda}=0$
$\dot{\lambda}_m=0$
$\dot{x}=0$
λ, λ_m

FIGURE 1

May 28, 1991

Differences in the Transaction Costs of Strategies to Control Agricultural Offsite and Undersite Damages

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Paper for the 1991 AERE Workshop on the Management of Nonpoint-Source Pollution, held June 6 and 7 in Lexington, Kentucky and hosted by the University of Kentucky

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K William Easter*

Pollution of our water supplies by agricultural chemicals has been an area of growing concern since the second half of the 1980's when agricultural chemicals were found in many water samples from wells and springs across the United States. This was added to previous information that identified agricultural chemicals as an important source of surface water pollution. However, most efforts to alter agricultural chemical use have not been to prevent water pollution. Regulatory efforts have focused on preventing the hazardous health effects of pesticides during application and in keeping pesticide residues out of food.

Identifying and controlling the major sources of nonpoint agricultural chemical pollution are not easy. In most cases, farmers decide what, how much, and in what manner agricultural chemicals and animal waste" products will be applied to their lands. As a result they strongly influence how much may eventually reach surface or ground water supplies. Farmers' decisions are dictated by their own utility maximizing behavior and government policies and institutional arrangements that constrain or enhance their decision set (Figure 1). Soil type, topography, vegetation and climatic events all influence chemical movements towards various water sources as will farming practices. While farmers have little control over climatic events they can change farming practices and vegetative cover to alter the impacts of climatic events. Thus farmers' decisions and the policies and institutional arrangements that influence their decisions are critical in controlling agricultural chemical pollution from the use of fertilizers and pesticides.

When evaluating alternative strategies and policy instruments for controlling pollution, economists have focused on the efficient use of production resources and largely ignored transaction costs. They determine what tax or other policy instrument would be the least distorting in making producers internalize the externalities they create. However, the major costs involved in reducing water pollution in agriculture are likely to be the transaction costs of enacting and implementing alternative strategies and not distortions in production efficiency.

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This paper focuses on the differences in transaction costs of policies that change farmers' decisions concerning the use of chemicals and agricultural waste products (primarily manure). The effects of policy instruments and institutional arrangements are likely to be different on ground water than on surface water, suggesting that strategies for controlling agricultural chemical pollution must be carefully designed to account for differences in water sources as well as other physical and socioeconomic differences. One simple, nationwide strategy is not likely to be the most efficient in terms of either production or transaction costs.

Why Undersite and Offsite Damages?

Since many water sources polluted in rural areas are used by farmers, i.e., domestic wells, one may ask why farmers pollute their own water supply or that of their neighbors. There are, at least, five answers to this question. One is that farmers lack the knowledge or information concerning the adverse impacts that their farming practices and input uses have on water quality or more specifically, "their" water supply. A second explanation is that they are not concerned about water pollution costs imposed on their neighbors or those living downstream. This is the classic spatial and temporal, externality problem where upstream producers damage the water supply of downstream users, but not their own. Third, they may have decided that the use of chemicals or disposal of manure and the resulting increased income is more important than clean water. They may even be willing to buy bottled water instead of reducing chemical or manure applications. A fourth reason may involve imperfect information concerning the optimum use and application of inputs. For example, many livestock farmers in southeastern Minnesota apply 60 to 100 lbs. more nitrogen, in the form of manure, than is required for optimum crop production because of the lack of information concerning its nutrient value (Legg, 1991). The fifth reason is risk and uncertainty concerning economic and weather conditions that will affect crop production. Applying extra chemicals may help reduce weather related income losses. Thus, there is no one simple answer to the question. It is a combined problem of imperfect information externalities, risk, farmer income requirements and waste disposal.

An added reason for water pollution is the lack of clearly specified property rights concerning water quality for either surface water or ground water. Do consumers have the right to clean water, or do producers have the right to pollute the water? If the water is polluted, who has to pay to clean it up? In most cases, farmers are not prevented from polluting water supplies, and if a clean up is required, they generally do not pay any more than other consumers or taxpayers. Holding farmers financially liable for water pollution would clearly provide an incentive to stop water pollution and help internalize the externality.

Farmer Decisions

Farmers make long run capital decisions, such as the type of manure handling facility to install or farming system to use, that have important impacts on their chemical use and the transaction costs of changing chemical use. These decisions will depend on a number of uncertainties, including future commodity and chemical prices. Annual chemical use decisions are constrained by the capital assets that are in place. The farmer will decide on chemical use rates and timing based on crops selected, prices, weather, manure supplies, labor available, management capability and soil conditions. These decisions may change during the growing season in response to rainfall and temperature conditions. A heavy rainfall in areas with sandy soil may mean last week's fertilizer application has been lost and needs to be replaced. In contrast, dry conditions mean that less nitrogen is needed and different pest control practices may be required.

Management availability and risk play an important role in these short and long run decisions. Nitrogen in the U.S. is relatively cheap and pest control with herbicides and insecticides does not require as intense management as does mechanical and biological pest control. Furthermore, price and weather uncertainty along with the demands of part-time jobs encourage farmers to err on the high side of chemical use. A little extra nitrogen may increase crop yields in a good rainfall year by 10 to 20 percent. Also, if farmers do not control weeds early in the season with heavy use of herbicides, wet weather may prevent them from getting into their fields and applying the needed weed control until labor is scarce or the crop is too tall. Failure to control the weeds can result in as much as a 25 percent reduction in crop yields.

Differences between Surface and Ground Water

Externalities appear to be the most important explanation of surface water pollution since much of the damage occurs offsite or downstream. This explanation does not hold in all cases since local fish kills and lake pollution may directly impact the farmers that cause the pollution. Still, a major reason for surface water pollution is the external nature of the costs imposed by the pollution, while lack of information and income requirements are more important for ground water. Many externalities associated with ground water are localized while for surface water, they may occur in the next county or state.

The transaction costs of monitoring and enforcement would be quite different for surface and ground water. For surface water the problem is its mobility and the numerous sources of agricultural nonpoint pollution. Whose pesticides caused the fish kill? While it maybe difficult to identify the polluters of surface water, the most likely suspects are upstream farmers. Yet the mobility of surface water means that transaction costs of ex-post measure of contamination are likely to be high (frequent monitoring). For ground water pollution, monitoring and enforcement are also likely to be expensive because of the cost of monitoring sites. In many cases the monitoring of existing wells is not enough and special monitoring wells are necessary to locate contamination and polluters so that ground water quality standards can be enforced.[1]

[1] Pollution of ground water as well as surface water by pesticides appears to be highly related to improper use, storage or disposal of pesticides or extreme rainfall events following pesticide applications, with the exception of a few herbicides such as Atrazine. With extreme rainfall events, pesticide movement is generally accompanied by high levels of soil erosion, but not always. Ground water pollution appears to occur with normal application of nitrates and Atrazine, particularly on lighter soils. In the case of surface water, nitrate and Atrazine pollution is more closely related to high rainfall events.

Another important difference between surface and ground water pollution has to do with the values of water uses precluded by chemical pollution. There are, at least, two aspects to this difference. One is that surface water has a wider array of uses then does ground water. Irrigation, industrial, commercial and domestic water consumption are the main uses of ground water while surface water can also provide a long list of recreational opportunities. The second aspect is that the duration of pollution may be quite different between surface and ground water, particularly if the surface water is a stream or river. Many of the agricultural chemicals that contaminate water supplies are not as long lasting in the surface water as they are in the ground water. How these two aspects will influence the value of lost water uses will vary by location and water use. For example, when the ground water is, or might be, used for domestic consumption and no good alternative sources of water are available, the losses from pollution will be quite high. In contrast, if the ground water is used for irrigation and is not likely to be demanded for other uses, then the pollution losses are likely to be relatively small.

Benefits from Improved Water Quality

With both the amount of agricultural chemicals entering the water supplies and the demand for higher water quality increasing, the benefits from improving water quality are on the rise. The increased demand is due, in part, to the growth in U.S. incomes and population, as well as greater knowledge concerning the harmful nature of certain agricultural chemicals. The growth in demand for bottled water and water based recreation are both directly related to this increased demand for higher water quality. Of course, water for household uses requires a different level of water quality than does water for recreational uses. Yet agricultural chemicals have damaged water for both of these uses.

Recreational benefits are among the largest, if not the largest, class of potential benefits from surface water pollution control (Rogers, et. al., 1990). Currently, they exceed the health or other water treatment benefits from reduced surface water pollution. In contrast, the primary concern in ground water appears to be the potential health effects or the increased cost of water treatment. In a number of cases, chemical pollution of ground water has forced the closing of wells and caused shifts to alternative water sources.

On the supply side of pollution, there are certain geographic areas that are more susceptible to water pollution and, therefore, they offer higher returns from pollution control efforts. For ground water, these are likely to be areas with light soils and shallow aquifers, or karst aquifers. The susceptible areas are not as easy to identify for surface water. However, surface water sources surrounded by moderately or steeply sloping, intensively farmed lands are clearly susceptible to agricultural chemical pollution. Thus, the physical characteristics of land, climatic conditions, amounts and types of chemicals, and farming practices will all be important in determining the degree of chemical contamination and level of benefits from pollution control.

On the demand side of pollution abatement, growth in per capita income and in population, the availability of alternative water supplies and the cost of pollution cleanup will all be important. These factors help determine the value of protecting water quality for a range of water uses. Clearly, areas with large populations and low rainfall, such as Los Angeles, will have a high demand for good quality water and programs that prevent agricultural water pollution. However, given where Los Angeles must obtain much of its water supply, it has a very limited capacity to influence what agricultural chemicals get into "their" water supply. For example, water taken from the Colorado River to supply L.A. will contain agricultural chemicals that have come from farms as far away as Colorado and Wyoming. Thus the demand for public action or changes in property rights concerning water pollution from agricultural chemicals is growing in urban American and will continue to expand. In addition, because of water's mobility the demand for clean water may come from areas outside the source of supply as is the case for LA.

The cost of cleaning up polluted ground water is sufficiently high, in a number of aquifers, to preclude it as an efficient alternative. In contrast, we have been cleaning up polluted rivers for many years at a wide range of costs. The persistence and toxicity of the pollutants are both important in determining the cost of clean up. Finally, the benefits from preventing water pollution will be closely related to the cost of substitute water supplies and the intended uses to which they will be devoted. To illustrate, if water is used for irrigation, there will be little or no loss from nitrate contamination, but the losses could be substantial if the use shifts to human consumption. The demand for "cleaner" water will also depend on whether the pollutant causes cancer or just tastes bad during a few weeks in the spring. When the demand is for domestic water use and the clean up costs are high, with no good substitute supplies available, then the benefits from protecting the water source from agriculture pollution will be high, especially if the water source is susceptible to contamination (Figure 2).

Pollution control policies need to be directed at those areas and types of water uses where the highest net benefits to society can be achieved from protecting the water supplies. In addition, policies, programs and institutional arrangements need to be designed so that the cost of such protection is minimized. One of the critical costs that should be minimized is the transaction costs of alternative courses of action. These costs must be compared with the potential benefits to be achieved since different water sources and types of agricultural chemical pollution will have different control costs. For example, inducing farmers to reduce their excessive use of nitrogen is likely to be less costly than having them change weed control practices, i.e., reduce the use of herbicides.

Transaction Costs

When designing policies, programs and policy instruments to reduce the level of water pollution by agricultural chemicals, a clear understanding is required of the transaction costs involved in implementing each alternative including search and information costs, bargaining and decision making costs, monitoring and enforcement costs, as well as any litigation costs (Williamson 1985). The distribution of costs and benefits involved with each alternative approach will determine to a large extent, their political support and the level of transaction costs. Ways to reduce such transaction costs need to be explored across alternative policies and policy instruments.

The fundamental unit of analysis will be the transaction which Williamson (1985) defines as something that "occurs when . . . one stage of activity terminates and another begins." In the case of water pollution, transactions occur whenever water is treated, or has wastes dumped in it, or when new agricultural policies or institutional arrangements are developed. Transactions also include changes in farming enterprises or farming practices. The farm plan that SCS develops for farmers is a transaction that involves a contract with farmers that is difficult to enforce and costly to develop.

The transaction costs of principal interest in developing alternative policies, institutional arrangements, and policy instruments for reducing water pollution in agriculture include, 1) the costs of enacting policies and programs, and 2) the costs of their implementation with specific policy instruments and institutional arrangements. The latter involves government costs (monitoring and enforcement costs and administrative and information costs) and must consider compliance costs imposed on farmers and the chemical industry. There will be a feedback between the compliance costs imposed on farmers and the chemical industry and the transaction costs of enacting policies and programs. For example, the transaction cost of promulgating improved water quality (through less use of agricultural chemicals) as a specific objective in the farm bill would likely be high. Farm groups and the chemical industry would strongly, oppose the idea because of their expected loss in income. In contrast, it is likely. to be more difficult to build continued support among environmental groups to offset these increases in transaction costs because their gains are smaller per individual and less clear cut. However, environmental groups have used ideology as a means to reduce the transaction costs of organizing to promote such restrictions (Nabli and Nugent, 1989).

The size of these transaction costs will depend on a number of factors including: asset specificity, - information availability and use, – opportunism – frequency of transactions, — creditable commitments, - uncertainty, and — the characteristics of land and water resources involved.

Information and Opportunism

The assumptions of bounded rationality and opportunism will be particularly important since the benefits and costs of water pollution control will not be uniform across the landscape.[2] "Transaction cost economics pairs the assumption of bounded rationality with a self-interest seeking assumption that makes allowance for guile (opportunism). Specifically economic agents are permitted to disclose information in a selective and distorted manner. Calculated efforts to mislead, disguise, obfuscate and confuse are thus admitted." Transactions, therefore, must be organized "to economize on bounded rationality (limits on information and ability to process it) while simultaneously safeguarding transactions against the hazards of opportunism" (Williamson 1989, p. 12-13). Clearly, changes in the availability of information, the way it is presented and the ability of farmers and government agencies to process it will affect the transaction cost.

[2] Underline added by author.

Information is made more imperfect by the existence of opportunism and the incentives for farmers and/or chemical dealers not to cooperate. For example, what monitoring and enforcement costs of restrictions on chemical use would be required to assure that farmers do not under-report chemical use?

Asset Specificity

The differences in asset specificity across farm types mean that the transaction costs of responding to changes in policies or institutional arrangements will be quite different among farms, e.g. dairy farms as compared to wheat farms. "Asset specificity has reference to the degree to which an asset can be redeployed to alternative uses and by alternative users without sacrifice of productive value . . . It is asset specificity in conjunction with bounded rationality, opportunism and uncertainty that poses the contractional/organizational strains" (Williamson 1989, pp. 13-14). In the case of water quality, they will cause different levels of strain depending on which alternative control strategy is implemented.

Uncertainty and Frequency of Transaction

Along with asset specificity, Williamson (1985) identifies two additional dimensions which make transaction cost economics important in addressing the problems of agricultural pollution of water: 1) uncertainty and 2) frequency of transactions. Uncertainty is critical in both the farming operation and in the control of water pollution because of bounded rationality and opportunism. As uncertainty increases, more information must be processed in making decisions and in implementing decisions which adds to the transaction costs. In response to this uncertainty, investment may have to be made in information systems or in organizational changes at the farm or regulatory agency level (Galbraith, 1973).

The frequency of transactions is important because of the benefits from specialized governance structures or organizational arrangements. "Specialized governance structures are more sensitively attuned to the governance needs of nonstandard transactions than are unspecialized structures, *ceteris paribus*. But specialized structures come at a great cost, and the question is whether the costs "can be justified. . . . The cost of specialized governance structures will be easier to recover for large transactions of a recurring kind (Williamson 1985, p.61). For agricultural chemical pollution of water supplies the key question is whether or not it is possible to use existing agencies like Soil Conservation Service (SCS), Agricultural Stabilization and Conservation Service (ASCS) and the Extension Service to implement the necessary transactions to reduce water pollution. If they cannot or do not have the will to regulate pollution, and EPA or a new specialized agency must do the job, then the transaction cost of controlling water pollution will be substantially higher.

Another important aspect of governance structures or organizational arrangements is that they provide different levels of safeguards, incentives and adaptability. These differences would, therefore, occur across policy instruments and institutional arrangements since they require different types of governance structures. For example, taxes on agricultural chemicals depend on market governance while regulations are based on some form of direct government intervention. Taxes provide monetary incentives to reduce chemical use, but might not be very adaptable to changing conditions. In contrast, regulations do not provide monetary incentives but may be more adaptable to changes. Yet well enforced regulations offer better safeguards against exceeding specified levels of pollution than do taxes, although a combination of taxes and pollution standards offer both good safeguards and incentives.

Creditable Commitments

A final aspect of transaction costs that is likely to be important in the control of agricultural chemical pollution is the idea of creditable commitments or assurance concerning the action of others. For example, what assurance or commitment do we have that farmers will use pesticides according to the directions on the label? Williamson (1989) finds that legal sanctions are severely limited and that creditable commitments are needed because of those limitations. For agricultural chemicals this can be important in at least, three levels. First what creditable commitments need to be established between farmers and the public sector to implement an effective program to reduce agricultural chemical pollution? Second, other sectors of the economy have to make creditable commitments to reduce chemical water pollution so that farmers feel others are doing their fair share, i.e., urban residential and golf course users of chemicals. Third, creditable commitments have to exist among farmers so that they will abide by the rules and limit chemical use. If most other farmers are thought to be cheating, why should they follow the rules? Finally, the same types of creditable commitments need to be established with pesticide and fertilizer dealers. This is particularly important when they apply chemicals and/or are used as the point of regulation or taxation.

Policy Options.

To significantly reduce the level of water pollution by agricultural chemicals will require changes in the farming sector. Figure 1 indicates many of the important linkages in the farming sector, and shows where government policies and programs have an impact on the agricultural sector. These many linkages suggest that to significantly change chemical use in agriculture will require a broad-based approach, starting with trade and agricultural policies and working all the way down to technical assistance provided to farmers by SCS.

We need to be concerned with how trade and agricultural policies influence input use in agriculture. Do they encourage intensive farming and the substitution of agricultural chemicals for land and labor? If so, what changes can be made to reduce or eliminate such incentives? One starting point would be to make reduced agricultural chemical levels in water supplies a specific objective of agricultural policy, and include it in all legislation related to agricultural production.

The next step would be to develop specific policy instruments and institutional arrangements to help achieve this objective. An important aspect of selecting the policy instruments or institutional arrangements is that they are likely to have different degrees of effectiveness depending on whether they are used to reduce surface water or ground water pollution. Since surface water pollution is much more of an externality problem than is ground water pollution, the methods for improving surface water quality should be focused on internalizing the externalities. In contrast, ground water pollution appears to be more an information problem where educational and technical assistance programs should be more effective. Furthermore, there may be some important differences in the spatial variability of chemical pollutants that must be taken into account. For example, is Atrazine contamination more localized than that from nitrates?

Some of the alternative policy instruments and institutional arrangements that should be considered for managing water quality include the following (1) subsidies, technical assistance and education (the traditional approaches), (2) bans on chemical use, (3) taxes and user permits, (4) land retirement, restrictions on chemical use and direct payments and (5) pollution rights and liability. The transaction costs of these alternatives will vary widely because of the institutional and organizational arrangements that already exist in the agricultural sector. Differences in information uncertainty, and asset specificity across regions and farm types, along with the possibility of opportunistic behavior by farmers, creditable commitments and the frequency of transactions, will all have a major affect on the level of transaction costs.

Subsides, Technical Assistance and Education (traditional approaches)

A review of policy instruments suggests some wide differences in transaction costs, particularly in terms of support from the farming sector. Cost-sharing (subsidies), education and technical assistance, to encourage the adoption of best management practices, have been the traditional public sector approaches used in the U.S. to control soil erosion and to reduce nonpoint pollution of surface water (Easter and Cotner, 1982). This is not an accident. These approaches are the most acceptable to farmers because they are free to participate or not and the programs also reduce the farmer's costs of adapting conservation practices. The U.S. also has existing agencies that have experience in providing conservation and pollution control services, i.e., SCS, ASCS and the Extension Service. This combination of existing agencies, no enforcement costs, and farmer support lowers the transaction costs of this set of alternatives particularly in the case of surface water (Table 1). However, the same set of practices and cost-sharing arrangements are not as effective for protecting ground water quality as they have been in reducing soil erosion, although some would argue about their effectiveness in reducing soil erosion.

The subsidy for soil conserving practices is one that tries to reduce pollution by changing the technology used. Another more general type would be a subsidy for meeting a set level of water quality. Farmers could then meet the standard with the lowest cost method which may or may not involve a change in technology (practices). The subsidy based on meeting a given standard requires establishment of baseline water quality and monitoring of water quality which is usually very dependent on rainfall events. Both requirements would substantially raise the transaction cost of reducing agricultural water pollution. Again, this helps explain why the traditional approach is being tried.

If SCS continues to have a major role in helping to reduce water pollution, serious questions need to be asked concerning their basic approach. For example, is a whole farm plan a cost-effective way to control chemical pollution of surface or ground water? The dollars spent on developing farm plans might be better spent on developing new farming practices and promoting their use.[3] Since new approaches are needed, training programs for SCS, ASCS and county extension service personnel maybe critical for program effectiveness. Thus the transaction cost of using the traditional approaches may not be as low as it first appears.

It is likely that best management practices and farming systems to reduce agricultural water pollution will have to be region specific, which will raise the cost of their development. Research will be needed to determine the impact of alternative farming practices and systems on ground water supplies under different resource conditions. Currently the lack of such information limits the effectiveness of cost-sharing, educational and technical assistance efforts in the protection of ground water supplies.

The type of research and education effort that is needed is being conducted in the karst area of southeastern Minnesota. Nitrates were identified as the major agricultural chemical polluting the ground water in this porous soil with numerous sinkholes. Research conducted by Legg, et. al., (1989) showed that excessive applications of nitrogen were being applied mostly by livestock farmers that failed to give adequate credit for manure. Further research now suggests that even recommended rates of nitrogen fertilizer application are too high to optimize profits in corn production. The research also shows the nitrate levels in soil water below the root zone (five feet) increases rapidly as nitrogen applications increase (Figure 3). Educational material showing these relationships are now being used by the Minnesota Extension Service to moderate farmers' use of nitrogen fertilizer and manure.

Bans on Selected Chemicals

The U.S. experience with policy instruments includes bans on selected chemicals that have been identified as particularly damaging, such as DDT. Chemical bans have been quite effective, but it takes time to lower the transaction costs of this alternative by building up political support for enactment of a specific ban (Table 1). We are now at the point where bans on herbicides are being enacted because of herbicide pollution of ground water. Current discussions about bans are focused on Alachlor and Atrazine, both widely used herbicides in the U.S.

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[3] "In a dynamic setting where technology can change, there will be transaction costs involved in gaining access to that technology and inducing the relevant agents to adapt their routines so as to accommodate these changes. Hence in such a setting the distinction between production and transaction costs is likely to be blurred." (Nablo and Nugent, 1989, p. 69.)

An important transaction cost that must be considered when enacting bans or imposing chemical use restrictions is monitoring and enforcement costs. When bans or restrictions on chemical use are imposed, there is a trade-off between farmer compliance and the government's monitoring and enforcement costs. Farmers will tend to exceed chemical bans or use restrictions as long as their expected gains from illegal chemical use exceed their expected losses from government imposed penalties. These expected losses, OL, will be directly related to government monitoring and enforcement expenditures and the level of fines imposed (Figure 4). The marginal loss curve, OL is constructed based on a particular level of monitoring and evaluation expenditures. An increase in monitoring and enforcement expenditures will shift the farmers' marginal loss curve from using illegal chemicals to the right to OL" while a reduction will shift it to the left to OL'. Farmers will apply illegal chemicals up to the point the marginal gains, GO, equal the marginal losses from the expected government imposed penalties. If the farmers' marginal loss curve is OL then they will use OU chemicals (the point where the slope of OL is equal to the slope of GO. The optimum level of monitoring and enforcement is OQ at a cost of 01 given the pollution cost curve AFP (the minimum point on the total cost curve AFP and the point where the marginal cost of monitoring and enforcement equals the marginal pollution cost). The pollution cost curve is constructed from the locus of, equilibrium levels of chemical use given by OCDEN which is constructed from different OL curves.

The curve ART shows the total cost to society from pollution and its control. It is a combination of' monitoring and enforcement costs and pollution costs. Thus the higher the level of pollution costs, the greater the monitoring and enforcement costs that would be economical to use. More monitoring and enforcement would be justified if the pollution cost curve AFP shifts to the right and less if it shifts to the left. Improved monitoring and enforcement technology could also change the minimum cost level. This same relationship would exist between monitoring and enforcement and chemical sales if chemical and fertilizer dealers were regulated. In this case, both dealers and farmers would consider the potential gains and losses from selling and applying excessive chemicals.

If an individual state or nation bans selected herbicides, what might be the impacts on farmers, the input industry and rural communities? One likely possibility is that the impact of a ban on a few selected herbicides would be minor, particularly if there are good substitutes that are less likely to reach "the ground water, i.e., they are less water soluble or break down more quickly. Enforcement would also be less costly because the farmer's gain, GO, would be less from using the illegal chemical. The curve OCDEN would be lower as would the pollution cost curve AFP.

In the case of a ban on Atrazine, the impact on net returns to farmers and gains from noncompliance depends on the weather conditions for weed control (Cox and Easter, 1990). If the weather is good for weed control, substitutes for Atrazine provide satisfactory weed control with only a small decrease in net returns. When the weather is unfavorable for weed control, the decline in weed control and resulting drop in yields can be substantial. The drop in estimated farm net returns for southeastern Minnesota would be around $20 per acre with unfavorable weather (Table 2). Thus the impact of bans and enforcement costs will depend on weather conditions and how much risk farmers are willing to accept when selecting weed control methods.

Bans on Alachlor should have a smaller economic impact on farmers and probably involve lower enforcement costs than those for Atrazine, since there are a number of good substitute herbicides. However, when Alachlor was banned in Canada, the chemical firms exhibited opportunistic behavior and raised the price of the substitutes by over 15%, which significantly increased the cost of weed control. If both Atrazine and Alachlor are banned, the drop in net returns would be somewhat greater than for just Atrazine or Alachlor alone, because of limited substitutes. The loss in net returns to farmers would be even higher if cropping system changes are required to improve weed control, particularly when substantial new capital investments are required and existing capital assets have few alternative uses (high asset specificity and low salvage values). Farm asset fixity or specificity raises the transaction costs of making major changes in farming systems. Thus enforcement costs for a ban on both Atrazine and Alachlor could be high, particularly if it was a state or regional ban.

The ban could also have a differential impact regionally. For example, southeastern Minnesota generally has good rainfall and better weather conditions for a range of different herbicides to be used to control weeds. In contrast, western Minnesota is drier and weather is not as suitable for use of some herbicides. This means that a ban on selected herbicides could cause a greater increase in weed control costs for western Minnesota then it does for the southeast. Because of the dry conditions, farmers might have to shift mostly to mechanical weed control. Thus bans on selected herbicides may put certain regions, such as western Minnesota, at a competitive disadvantage and farmers would have greater incentives not to comply, which could raise enforcement costs.

Government bans on chemical use may take place at an even lower level than a state. Just as individual counties have raised their standards for domestic drinking water, they could also take direct action to ban farming practices that contribute to chemical water pollution. A county might ban certain manure handling practices, or the sale or use of Atrazine. In conjunction with such restrictions, the county could help farmers install manure storage facilities or develop markets for their excess manure. Subsidies for alternative, less polluting herbicides might also be used so the county's farmers are not at a competitive disadvantage to other regions. Such combined actions would help keep the negative financial impacts for farmers to a minimum and help reduce their opposition and the transactions cost of implementing such environmental restrictions. However, with outright herbicide bans, what is to prevent opportunistic farmers from taking their business across the border? This, of course, will not please local businesses and will raise the transaction costs of implementing an effective targeted ban. Thus the opportunistic behavior of farmers and input suppliers along with asset fixity could make the transaction costs high for a targeted herbicide ban particularly if it alters farming systems.

An additional problem arises if the ban is targeted just on areas susceptible to water pollution. The susceptible areas have to be identified, which will increase information costs and raise difficult questions concerning what farms to include in the targeted area. Should everyone with land over aquifer or near a stream be included, or should it be everyone in the county or watershed? Again, opportunistic behavior can be expected from farmers who do not want to be included in the targeted area. Combining this with the information costs suggests high transaction costs.

A final issue involves the impact on consumers of reduced agricultural chemicals. Likely, chemical bans will mean reduced U.S. agricultural production and more food imports. For the consumers' budgets, it would mean higher food prices. Since many agricultural commodities have price inelastic demands, producers will benefit and consumers will lose from higher prices. However, not all producers will benefit, and some will benefit more than the others. This will make the support for drastic restrictions on chemicals somewhat uncertain. Because of the uncertainty over who benefits and who loses, the agricultural sector will, in general, oppose the change, raising transaction costs. Those urban people with moderate to high incomes will probably support restrictions and will be willing to pay somewhat higher food prices for cleaner water. With low income people, the support is less clear cut because of the likely substantial impact of higher food prices on their limited incomes.

Taxes and Permits

The U.S. has had limited experience in using taxes or permits as a means for reducing chemical use. In contrast, Europe has had some success in reducing nitrogen applications through the use of taxes. The problem is that the demand for nitrogen fertilizer may be highly inelastic below certain levels, i.e., 50 to 150 lbs. per acre depending on the soil type, water availability and other factors. A similar situation may exist for certain pesticides. The advantage of taxes is that they can be implemented through fertilizer and pesticide dealers and provide farmers with market incentives to reduce chemical use. This means lower transaction costs in terms of tax collection as well as monitoring and enforcement costs. Dealing directly with each farmer, as would be required with application limits, would greatly increase these costs.

Permits could be used if we knew how much of a chemical is safe to use in a given area. Permits could then be sold or allocated up to the maximum acceptable level of use. One difficulty is that the permitted levels would have to be varied by area, depending on an area's physical characteristics (i.e., soil texture, vegetation and slope) and its location relative to water sources. This information requirement would substantially raise the transaction costs of a permit system. On the positive side, tradeable permits would put a value on the assimilative capacity of agricultural land, and encourage farmers to conserve it. They would also provide an incentive to limit the chemicals used because they could sell unused permits to other farmers. In fact, nonfarmers concerned about water quality could be allowed to buy up permits and reduce the quantity of chemicals applied in an area.

With taxes and tradeable permits farmers may have less incentives for opportunistic behavior and noncompliance than they would with an outright ban since they could legally obtain the chemicals but at a higher price. This should hold down the transaction cost involved with enforcement and those related to fixed assets since there would likely be fewer changes in farming systems than with outright bans.

Land Retirement, Restrictions on Chemical Use and Direct Payments

The U.S., through its farm commodity programs, has made extensive use of land retirement and direct payments to reduce agricultural production and support farm income. Land retirement could be used in the farm program, since it was part of a package which included commodity payments to participating farmers. Would it be possible to include chemical use restrictions as a requirement for participation in the farm programs and what would be the transaction costs of doing so? One major cost would be to get such a provision included as part of the farm bill. Clearly there is a precedent for restrictions on participation in the farm programs with the current requirements concerning soil conservation and wetlands. However, promulgating chemical restrictions, as part of the farm bill, is only one of the transaction costs involved.

The task of implementing a program to retire land or restrict chemical use would probably fall on either ASCS or EPA. In terms of being the most effective (highest will to regulate) in reducing pollution levels, EPA would be the clear choice. On the other hand, ASCS, with the help of SCS, may be the only agency in a position to implement the program since they have a presence in most U.S. counties. The problem is that implementation would require close monitoring and policing, particularly in areas where farmers apply their own chemicals. This, along with the idea that they will be more lenient and sympathetic to farmers, is why ASCS and SCS might be the first choice. Where chemicals are mostly applied by contractors, the control and monitoring could be done through them and transaction costs reduced.

The high transaction costs support the idea that use standards or direct control on the amounts of chemicals applied would work better in controlling agricultural chemical pollution than performance standards (Braden and Lovejoy, 1990). Since chemical use, particularly pesticides use, could be controlled mostly through dealers, the monitoring, enforcement and information costs would be relatively lower. In contrast, performance standards would require monitoring and enforcement at a more micro level which would substantially raise transaction costs. Thus, based just on transaction cost considerations, performance standards are not likely to be a desirable policy instrument (Table 1).

The level of penalties for not complying with chemical restrictions will also have an impact on compliance. If the penalty is a small fine, then compliance is likely to be low without intense monitoring. The farmer's loss, OL, would be low (Figure 4). In contrast, a loss of all farm program benefits because of the illegal use of agricultural chemicals would likely result in higher compliance levels, even without much monitoring. How neighbors respond to the program may also be quite important in determining what an individual farmer will do. If your neighbors support the idea of chemical restrictions and abide by them, then it will likely be more difficult for you to cheat (social or community pressures). There is also the fear and possibility that your neighbors will turn you in if they see you misusing agricultural chemicals or animal waste.[4]

A land retirement program and/or easements might be used to protect areas with highly valued water supplies (Figure 2). Programs such as Minnesota's RIM have been used to buy easements for restoring wildlife habitats. Similar programs could be designed to protect valuable ground and surface water supplies. The transaction costs could be high for such a program since the task of determining which water supplies to protect could be highly political. It will also require a lot more information concerning the susceptibility of ground water supplies to chemical pollution and their potential future use.

Land retirement and easements would also be costly in terms of direct payments to landowners. Yet easements would be lower in cost than land retirements if farmers could continue to use the lands as long as they did not apply herbicides. Still, someone would have to enforce such restrictions on herbicide use which, of course, raises the transaction costs. In addition, increasing mechanical weed control could increase soil erosion and augment surface water pollution particularly in steeply sloping areas.

Lovejoy (1990) suggests that SCS and ASCS buy the rights to certain types of erosive land use practices to control soil erosion in erosion prone areas. He further suggests an innovative method for reducing the transaction costs of enforcement, where the property rights are assigned to some group or organization interested in protecting water quality. Organizations interested in protecting the environment such as the Nature Conservancy or Izaak Walton League would be given the partial property rights and if these contractual obligations were violated by farmers, the environmental organization could take judicial action. This, however, is just a transfer of costs and not a reduction in transaction costs for society and could cause over protection.

Since the practices that might be prohibited for soil erosion would have a fairly visible impact on the landscape, monitoring should not be costly. However, if the same approach was tried for agricultural chemicals, more intensive monitoring would be necessary. For example, how do you know that a farmer applied two pounds of Atrazine per acre on a given field, rather than three or four pounds? (Two pounds per acre is the limit on Atrazine use in Wisconsin.)

Pollution Rights and Liability

Implementing changes in property rights regarding water quality is a much broader issue than just agricultural chemicals. It also must include point source water pollution as well as water pollution by soil particles since rights to clean water should

[4] The freedom and ability to organize and protest against unwanted externalities is one way to prevent excessive pollution. The lack of such freedom may explain why pollution got so bad in Eastern Europe. This same freedom and ability to organize locally in many areas of the U.S. could be used as a means to reduce the transaction costs of controlling agricultural pollution through bans or regulations.

involve all sources of contamination. The idea that the citizens of the U.S. have the right to clean water has been legislated in the U.S. clean water act. The problem is putting such objectives into practice. There are limits on the amounts of pollutants that point sources are allowed to deposit in lakes, rivers and streams, however, the rights to "clean" water do not exist de jure. This is particularly true in terms of nonpoint sources and a number of point sources. The transaction costs of implementing such a major change in water quality rights is high, particularly in the short run (Table 1). In fact, in the short run, it is almost impossible to implement because of the chemicals already in the soil or stream beds that will eventually enter our water supplies without any additional discharges or applications of chemicals.

One means of moving towards a policy of giving the U.S. citizens the property right to clean water would be to change the liability rules for water pollution. Polluters could be made liable for any damages or loss in uses caused by their pollution of water. For example, in Connecticut, liability has been imposed on individuals (including farmers) shown to have contaminated drinking water sources. This shifts enforcement to the court systems and, if strictly enforced, could produce some major changes in farming behavior. The major problem is being able to show or prove a farmer has polluted a particular water source while others have not.

The liability for water pollution could also be placed on agricultural chemical manufacturers or dealers. This would act as a tax on farmers because the manufacturers would have to charge a high enough price to cover the liability costs.[5] Manufacturer liability would work just as well as farmer liability except where nonpurchased inputs (manure) are used or where the methods and timing of application by farmers affect pollution rates (Braden and Lovejoy, 1990, p. 50-53). Thus, in areas where livestock production is important and/or farmers apply their own pesticides, a farmer-based liability may be necessary.

Thus, an important first step in making such a rule change effective would be to collect adequate information so that the polluters could be identified and the damages estimated. This would be a major monitoring cost for some types of pollutants because of the temporal and spatial nature of their damages. Another transaction cost is the litigation costs that would be imposed on an already overburdened court system which is not a small cost. For example, Kopp, et. al., (1990) point out that as much as "30 to 70 percent of all current expenditures related to Superfund take the form of legal fees, as opposed to expenditures for actual removal or stabilization of hazardous substances at waste disposal sites" (p. 13). Possibly court costs could be reduced or eliminated through bargaining to obtain out of court settlements which could benefit all participants. In fact, clearly established liabilities should encourage bargaining solutions if only a few parties are involved. However, the threat of court battles will not guarantee that the negotiated outcome is economically efficient (Porter, 1988.)

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[5] Negligence is like a regulatory standard where the firm has no incentive to do better than the safety standard. For the liability rule, there is always some incentive to do better since this will further reduce the chance of liability from pollution.

An alternative approach would be to take action at the county government level through county commissions and land use planning efforts. For example, in Fillmore County Minnesota, a farmer with excess soil erosion can be required to implement a soil erosion control program approved by SCS and enforced at the township level. Similar restrictions could be used by counties to protect their water supplies against chemical pollution. Enforcing restrictions or liabilities at the local level would reduce the transaction costs because farmers tend to know what their neighbors are or are not doing, which could reduce information costs. These local efforts would be most effective where the pollution affects a substantial number of county residents other than the individual farmer, i.e., there is a large negative externality. Highly visible erosion and pollution such as gully erosion, muddy streams and murky lakes are good targets for local action. People can see the damages and are willing to put pressure on county commissioners to take action.

Enforcement and monitoring costs may also be lower because of the social closeness of people in the rural community. When those causing the water pollution are well known in the community, social pressures, obligations and respect for neighbors will influence farming decisions (Robinson and Schmid, 1989). The greater this sense of social closeness the less likely a farmer is, knowingly, going to create a negative water pollution externality In fact, social closeness can be sufficient to maintain negative production externalities at social optimal levels. Such a level would be reached when the cost to the polluter of reducing the negative externality would equal the increased utility received by the pollutee. Community education concerning the impacts of farming on water quality could be an important policy instrument that would complement local attempts to reduce water pollution. As Braden (1990) suggests "a sense of obligation may be transferred with the knowledge that one's action substantially affect other people" (p. 27). This sense of obligation will even be stronger with social closeness.

Difficulties arise- with county-specific regulations because of the fear that they may put the county's farmers at a competitive disadvantage and also because chemical water pollution is not visible and crosses county and state boundaries. This is why state or national standards and pollution control efforts that focus on the watershed or aquifer are important. Such approaches can internalize many of "the externalities that cross political boundaries.

Strategy to Reduce Agricultural Chemicals in Water Supplies

Because of past levels of agricultural pollution, implementing an effective clean water policy for agriculture requires a long run point of view. It means cutting back on chemical use in agriculture, a much greater use of alternative farming practices, and

[6] The lack of social closeness had a lot to do with the soil erosion restriction imposed in Fillmore County. An increased number of absentee landowners who employed outside management to run their farms was a major concern of Fillmore County Officials. They felt that these "outsiders" were operating with very short time horizons and that excessive soil erosion was taking place. Since these people were outsiders, social pressures were not effective in inducing them to reduce their erosion externalities. Thus more formal means where found to limit the erosion.

reductions in lawn chemical use in cities and towns. Farmers are not going to cooperate if they feel others are not making "credible commitments" to the reduction in chemical use (Williamson 1985). In addition, the effect of such cutbacks may be limited at first because of the chemicals that already exist in our soil and water resources. A first step would be technical assistance and education, with demonstrations concerning what can be achieved with fewer chemicals applied more often, but in smaller amounts. Use of fewer chemicals in smaller quantities will require more labor and better management skills to maintain production levels. Moderate sized farms may have an advantage over large or corporate farms because of the importance of timing in areas dependent on rainfall. Irrigated areas not dependent on rainfall during the crop season may also have an advantage over nonirrigated areas because control over water reduces the uncertainty involved in weed control and fertilizer use.

The real question is what mix of policies and policy instruments has the best chance of reducing agricultural chemicals in our water supplies over the long run. The whole process of reducing chemical use in agriculture would be facilitated if it was a goal of U.S. farm policy. Such a national goal would lower the transaction costs of taking action at the state and county levels. As a start, local variation should be allowed because of the wide differences amongst regions in terms of physical and climatic conditions, crops grown and inputs used. Experimentation should be allowed, since we still have a great deal to learn about the effects of reduced chemical use and how the chemicals can best be kept out of water supplies. Experimentation is also needed with rule making for monitoring and enforcing agricultural pollution control. If rules are flexible, innovative ways can be developed that reduce transaction costs.

A strategy involving technical assistance, education and cost-sharing for best management practices is favored by the existence of agencies which provide these services. Currently this strategy is being tried on a limited scale for ground water protection and should be evaluated for its cost-effectiveness. Other alternatives should be tested, including bans, use permits and easements in sensitive areas. The education effort should not be limited just to farmers, but should involve the broader population so that they have a better understanding of the problem. This has at least two possible benefits: first, an informed population will be more willing to pay higher food prices resulting from reduced chemical use and second, the nonfarm population can apply community pressure on farmers to limit their chemical uses.

Conclusion

Although the extent of agricultural chemicals in U.S. water resources is still a matter of debate, most individuals would agree that it is a problem for many areas with intensive crop and/or livestock agriculture. The question is, what can and should be done about it? First we should quickly expand our research effort so that we have a better information and knowledge base from which to design our strategies for reducing agricultural chemical water pollution and reduce the transaction costs of implementing different strategies. Second, we need to improve the information available to farmers, and its transmission to farmers, concerning how "best" to use agricultural chemicals and animal wastes while minimizing their negative impacts on water supplies. As part of this, demonstrations of different low input agricultural strategies should be developed throughout the U.S. Cost-sharing arrangements should be tried for system changes that involve high asset fixity and, therefore, high transaction costs. Third, if education and technical assistance along with cost-sharing are not effective then more coercive instruments will have to be used. For example, the liability rule could be changed so that farmers are liable for their water pollution damages. User permits and taxes should also be tried.

Finally, a broad based educational program is needed for the general public so that they can make "better" informed decisions concerning water pollution. For example, what chemical levels pose real risks to humans? In addition, why is it alright to have different chemical levels in the water supply, depending how the water will be used in the future and the assimilative capacity of the water resources? Nitrates in drinking water can cause adverse effects on humans and livestock, but in irrigation water, it can increase crop yields and lower fertilizer costs.

Transaction costs play a major role in determining the U.S. strategy for managing agricultural chemical use. This is why the President's Water Quality Initiative emphasizes the traditional approaches, such as technical assistance, education and cost-sharing, which are implemented by existing agencies. If these efforts are not successful, there will be increased pressure to try more coercive control measures with correspondingly higher transaction costs. This is when farmer compliance with alternative pollution control instruments will become critical and determine the level of monitoring and enforcement costs that will be necessary to achieve water quality goals. A noncooperative rural community could mean that monitoring and enforcement costs are prohibitively high. In additional, a strongly opposed rural community could raise the transaction costs so high that passage of any effective legislation to reduce agricultural chemical use would be blocked.

As economists, we need to estimate the transaction costs for alternative approaches to reduce agricultural water pollution and help design institutional and organizational arrangements that will reduce transaction costs. For example, can arrangements be designed that channel community concerns towards effective local and state based efforts to reduce agricultural chemical pollution of ground water? Farmer response and the transaction costs of reducing chemical use will not be uniform across the United States, or even across an individual state. Consequently, community based approaches might be the most cost effective approach, particularly when water pollution impacts are mostly localized, i.e. ground water pollution. However, when the problem crosses state or county boundaries, these local efforts are not likely to be enough. In addition, when other concerns such as economic development dampen local interests in reducing water pollution, then the federal government may have to step in to prevent or reduce water pollution.

Of course, decisions will have to be made before all the information we would like is available. The Canadian ban on Alachlor is an example of one such decision. Hopefully, the U.S. can approach the problems of reduced agricultural chemical water pollution in a more systematic and targeted fashion.

TABLE 1. THE TRANSACTION COSTS OF ALTERNATIVE POLICY INSTRUMENTS

Policy Instruments	Transaction Costs					
	Search and Information	Bargaining and Decision Making	Monitoring and Enforcement	Litigation	Compliance Costs	Program Effectiveness in Controlling Pollution
Traditional Approach (cost-sharing, technical assistance & education	moderate	low	low	none	low	low
Chemical Bans						
1. National	moderate	high	low	low	high	high
2. Local	high	moderate	high	low	high	moderate
Taxes	moderate	high	low	low	low	low
Permits	high	high	moderate	low	moderate	high
Land Retirement	moderate	high	moderate	low	moderate	moderate
Easements	moderate	high	high	low	moderate	moderate
Standards						
1. Performance	high	high	high	moderate	high	high
2. Use or Practice	moderate	moderate	low	low	low	moderate
Property Rights and Liability	high	high	high	high	high	high

Table 2. CHANGES IN NET RETURNS DUE TO HERBICIDE BANS ON SOUTHEASTERN MINNESOTA FARMS USING CONVENTIONAL TILLAGE PRACTICES

TYPE OF BAN & DECISION RULE	TYPE OF WEATHER THAT OCCURS GOOD	BAD
	--------------(per acre)-------------	
BAN ATRAZINE		
Maximum Net Returns, Assuming Good Weather	-$0.51(0%)	-$20.50(10%)
Maximum Net Returns, Assuming Bad Weather	-$7.73(3%)	-$7.73(4%)
No Herbicide	-$11.62(4%)	-$71.76(35%)
BAN ALACHLOR		
Maximum Net Returns, Assuming Good Weather	-$0.10(0%)	-$20.15(10%)
Maximum Net Returns, Assuming Bad Weather	-$2.64(1%)	-$2.64(1%)
BAN ATRAZINE AND ALACHLOR		
Maximum Net Returns, Assuming Good Weather	-$0.51(0%)	-$20.56(10%)
Maximum Net Returns, Assuming Bad Weather	-$9.53(3%)	-$9.53(5%)

Source: Craig A. Cox and K. William Easter, 1990.

Figure 1. Agriculture Related Water Quality System

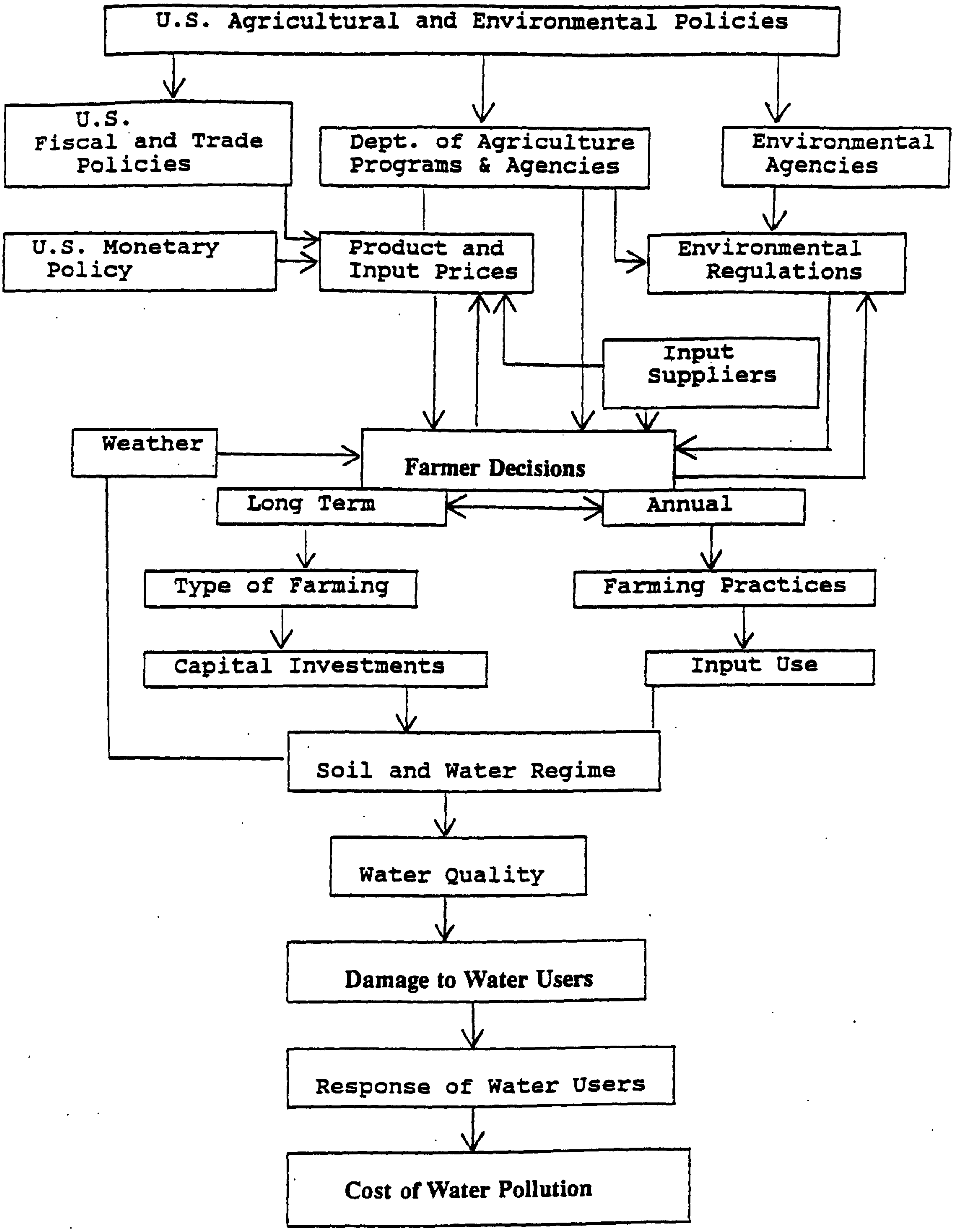


Figure 2. Relative Size of Benefits from Controlling Agricultural Chemical Pollution of Water Supplies

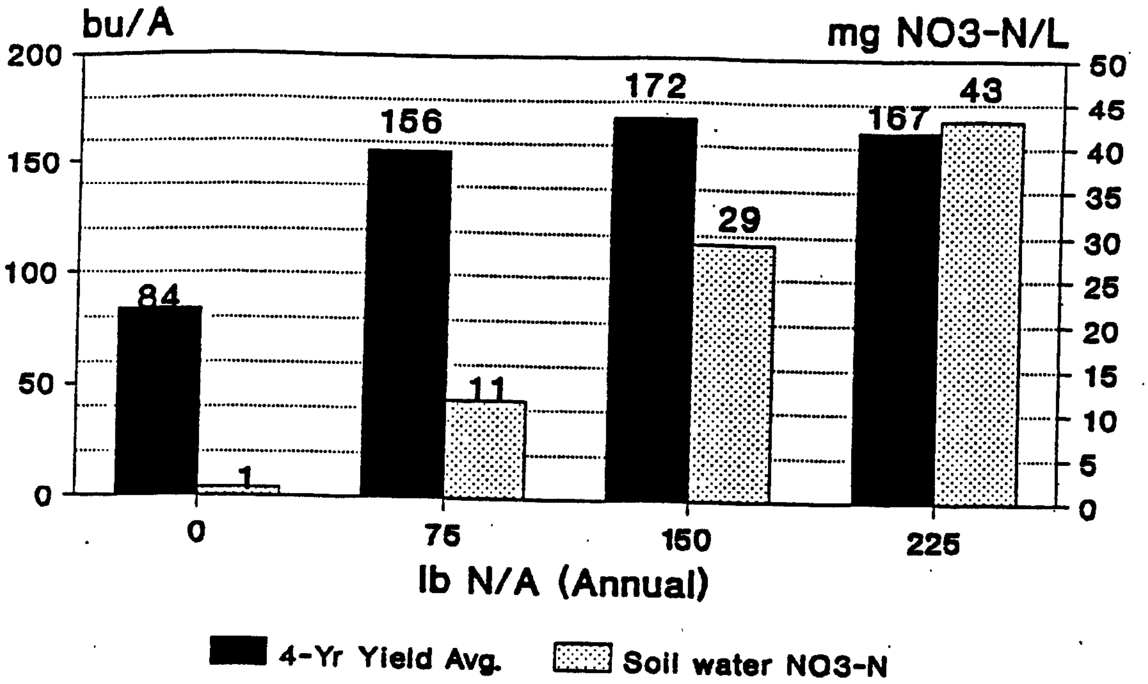
Non-point pollution
Susceptibility of Water Source

Value of Water Use[1]	Low	Medium	High	Cost of Alternative Water Supplies[2]
Low				Low
Medium				Medium
High			High Benefits	High

[1] This is a function of water use in the region and the growth in population, income and the water using sectors of the economy. Irrigation water uses would generally have a low value while industrial and domestic consumptive uses would have a high value.

[2] Includes cost of clean up as an alternative.

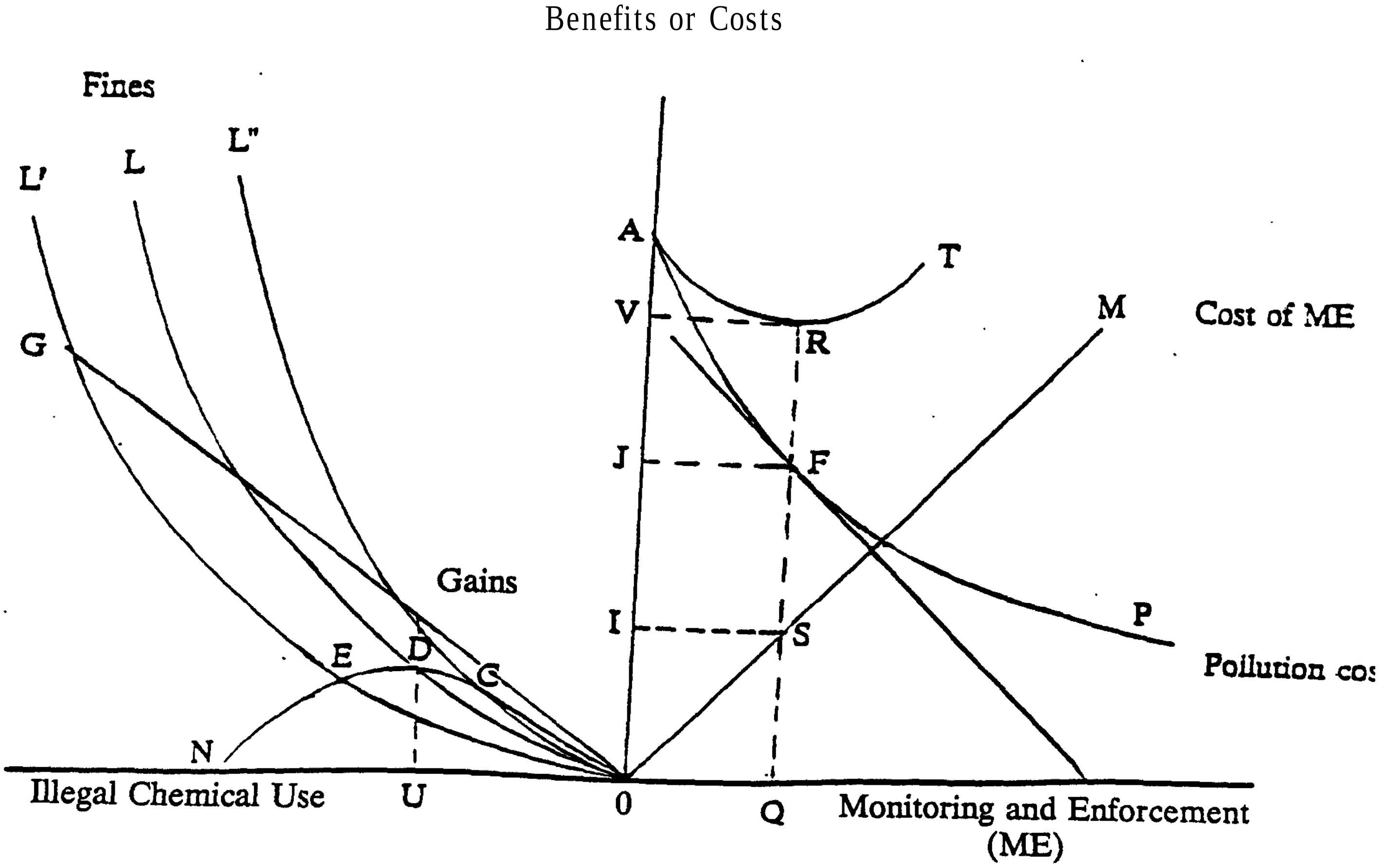
Figure 3 Relationship between Yield and Nitrate-N in soil water at 5' in September 1990.



Lawler Farm, Olmsted County, Minnesota.

Source: Randall, et. al., 1991.

Figure 4 Monitoring and Enforcement Cost of Restriction on Agricultural Chemicals Used.



Source: Adapted from Nabli and Nugent, 1989, p. 48.

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Optimal Policies for the Control of Non Point-Source Pollution in a Second Best Environment

by

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Optimal Policies for the Control of Non Point-Source Pollution in a Second Best Environment

Policy prescriptions for the control of non point-source pollution have been examined by a variety of authors (e.g. Griffen and Bromley, Shortle and Dunn, Segerson). Implicit in most models and analyses has been the assumption that the externality represented the only source of market failure in the sector being analyzed. However, in cases involving agricultural non point-source pollutants there are likely to be distortions other than those generated by the externality that effect the input or output markets for the primary product. Crop prices may be supported through commodity programs or marketing orders, input prices may be distorted or input use may be controlled through quantity restrictions and use regulations. Irrigation water supplied by the Bureau of Reclamation is an example of a case in which a primary input is provided at a subsidized price, but in limited quantities. Subsidized electric rates, nitrogen fertilizer taxes, cost sharing for soil conservation practices, and pesticide use regulations provide additional examples of input market distortions. The general theorem of the second best (Lipsey and Lancaster) suggests that the use of first-best policy tools to correct production externalities under these circumstances may actually move the economy away from a second-best optimum, rather than nearer to it.

Several authors have addressed the second-best conditions created by output market distortions (e.g. Lichtenberg and Zilberman, Buchanan), but relatively little attention has been paid to input market distortions. This paper develops a conceptual framework for examining non point-source control in a setting characterized by input market distortions that are particularly relevant to irrigated agriculture. Two conditions giving rise to a second-best problem are examined: input price distortions and institutionally set quantities of the input. These conditions describe surface-water supply institutions in much of the western United States. In addition to its importance as a primary input to crop production, irrigation water is also a primary input in the generation of agricultural externalities in many areas.

The implications of input market distortions on non point-source problems are examined first in a single firm context. The analysis is then extended to multiple firms to examine optimal policy tools when the input price and quantity distortions differ between firms. Finally, an empirical example is presented that explores policy options for addressing the agricultural drainage problem emanating from irrigation activities in California's San Joaquin Valley. Prices and quantities of water delivered to farmers in the study area are set by the Bureau of Reclamation and do not reflect market forces. Hence, non-point source emissions with distortions in the market for an input (water) characterizes this problem.

Conceptual Framework

In this analysis, profit maximizing farmers are assumed to jointly produce crops and a volume of effluent through the use of two inputs, a polluting input (x) and an abating one (z). For purposes of discussion, input x may be thought of as irrigation water while z may represent irrigation technology or management. Farms face a maximum constraint $(\bar{x})$ on the quantity of the polluting input used and receive that quantity at a distorted price (w^i). The optimization problem for a representative farm is to choose input levels to:

$$\begin{aligned} &Max\ \pi^i = pf^i(x_i, z_i) - w^i x_i - r z_i \\ &subject\ to\text{:}\ x_i \le \bar{x}_i, \end{aligned} \tag{1}$$

where π^i represents net returns to land and management for the i^{th} farm, p is the price received for crop output, $f^i(\cdot)$ describes crop production opportunities for the i^{th} farm and is assumed to be twice differentiable and concave, and r represents the marginal cost of the abating input (z). The social objective in this scenario is to maximize social welfare (S) defined as the sum of farm-level net returns less the social damages incurred as a result of the production externality:

$$\textit{Maximize } S = \sum_i \left(pf^i(x_i, z_i) - w^s x_i - rz_i\right) - p_d D\left(g^1(x_1, z_1), \ldots, g^I(x_I, z_I)\right) \quad (2)$$
$$\textit{subject to: } x_i \le \bar{x}_i \quad \forall i,$$

where w^s represents the "true," per unit cost of supplying the input x (the supply price), $D(\cdot)$ represents total effluent levels, p_d represents the marginal social cost of damages from emissions, and $g^i(\cdot)$ is a production function describing non-point source emissions from the i^{th} farm. The non-point source production function is assumed to be twice differentiable and convex, and to increase with input x and decrease with levels of z, i.e. $g_x > 0$, $g_z < 0$, $g_{xx} \ge 0$, , and $g_{xx} \ge 0$. Subscripts on functions denote partial derivatives. In addition, from here on it is assumed for simplicity that total damages are an additive function of farm-level emissions, i.e. that $D = \sum_i g^i(x^i, z^i)$.

Non point-source control studies often suggest the use of input taxes to motivate optimal behavior (Griffen and Bromley, Stevens, Pfeiffer and Whittlesey). Under the assumptions that the input constraint is strictly nonbinding and that all farms face the true cost of water (i.e. $w^i = w^s, \forall i$), it can easily be seen that a set of input taxes $t_x = p_d g_x$, $t_z = p_d g_z$ will motivate socially optimal behavior on the part of farms.

Modeling policy choice for markets with resource constraints is straightforward when it can be assumed that the nature of the restriction is invariant to the parameters of the problem, e.g. if a constraint is binding before a price change, then it will be binding for the regulated firm. However, the impact of a policy may be such that movement on to, or off of, the constraint is generated. A taxonomy of "regimes" can be defined to describe these alternatives, as shown in Table 1. The cases in which the constraint is always binding (regime 1), as is generally the case with fixed inputs, or is nonbinding (regime 2) lend themselves to traditional analytic methods. Two additional regimes are necessary to describe a switch in conditions: regime 3, in which the constraint is binding prior to government regulation of the externality, but not binding afterward, and the opposite case in which the constraint is initially nonbinding, but binds at the optimum as a result of a policy (regime 4). Comparative statics results cannot be used to analyze the case of inequality constraints or discrete changes in the values of policy instruments, as is necessary to address the conditions implied by the latter two regimes. Policy analysis should, however, be conducted with respect to all four regimes. A different approach is therefore required.

Table 1. Taxonomy of Policy Regimes

	Solution Incorporating Externality		
Pre-policy Solution	Input constraint:	Binding	Not Binding
	Binding	Regime 1	Regime 3
	Not Binding	Regime 4	Regime 2

Single Firm Analysis

Input price distortions can be analyzed by assuming that a representative firm faces a private input price of w^p $(w^p \neq w^s)$. In the irrigated agriculture example, w^s may represent the cost to the Bureau of Reclamation of producing and delivering an acre-foot of water, while w^p

represents the price paid by farmers to the Bureau. The definition of $\mathbf{w^s}$ does not incorporate the external costs associated with the use of the water (these are addressed explicitly in this analysis) though it may include external costs associated with development of water **supplies.**[1] Farmers often pay less than full cost for water deliveries due to the subsidies that are built into the Bureau's pricing structure, so that $w^p < w^s$. In addition, farmers are allotted a fixed quantity of water per acre per year.

The level of input prices plays an important role in determining whether the input constraint is binding in a given problem and, therefore, whether the constraint is relevant for analysis of non point-source problems. This relationship suggests an alternative approach in which a virtual price $w^v(p, w, r, \bar{x})$, defined as the price at which the constraint would be "just" binding, is specified. In this example, $\mathbf{w^v}$ represents the price at which the firm's demand for input x would equal $\bar{\mathbf{x}}$ and is implicitly defined by $x(p, w^v, r) = \bar{x}$. The work of Neary and Roberts illustrates the use of virtual prices to model consumption when some commodities are rationed. Analogously, $\mathbf{w^*}$ is the level of the input price that would induce the firm to choose the socially optimal level of the input and is implicitly defined by $x(p, w^*, r) = x^*(p, w^s, r, p_d)$.

The taxonomy of regimes and the response to policy instruments that can be expected from the firm under each regime can be depicted graphically. The firm's demand for input x is labeled $\mathbf{VMP}|_z$ (Figure 1). The social cost function is defined as the private marginal cost of the input plus marginal social damages, i.e. $MSC|_z = w^s + p_d g_x$. The inputs are assumed to be

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[1] External costs of developing water supplies include degradation of wildlife habitat, declines in fisheries, and loss of white water recreation. For a more complete description of these costs see Willey.

technically independent ($f_{xz} = g_{xz} = 0$) since without this assumption a family of curves is necessary to depict social marginal costs and private marginal benefits in a two dimensional figure. The marginal product in the input demand function and the marginal effluent product in the social damage function are both evaluated at z = z^*. The unconstrained (first-best) social optimum for x is denoted by x^* and is defined by the intersection of the value of marginal product and marginal social cost curves, i.e. where $pf_x = w^s + p_d g_x$, as is required by the social first order conditions (derived from equation (2)). An unconstrained firm would choose x = x^p in the absence of policy intervention. However, the input constraint is binding so the firm chooses x = $\bar{x}$.

The theory of the second-best suggests that attempts to apply first-best tax rules to correct an externality may be suboptimal when other distortions are present (Lipsey and Lancaster). A contrapositive is also suggested: if one can address all distortions, then optimal taxation rules *will* be effective. Therefore, it is equally important to consider the potential for policy tools to restore a first-best state as to examine the implications of first-best tools under second-best conditions. A distinction is made between first and second-best optima. In this paper, it is assumed that market distortions occur in two forms: (i) input market distortions including a price distortion and a quantity constraint, and (ii) an environmental externality. The first-best, or social, optimum is defined without regard to the market distortions while second-best optima result from maximization of social welfare subject to the condition defining the second-best nature of the problem.

Figure 1 depicts a constraint that is binding in both the social and private optimization problems (regime 1). Traditional prescriptions for attaining a social optimum suggest an input tax on x equal to $p_d g_x$, the value (cost) of the marginal effluent product. This is the distance $w^* - w^s$ in Figure 1. The second-best optimum is attained at $\bar{x}$ when the constraint is immutable. This is the point at which net social benefits are maximized subject to the input constraint.[2] The firm is already using the second-best optimal quantity of x in this case and, because the virtual price is greater than that implied by the true social cost (w^*) of using the input, it will not respond to a tax reflecting marginal social damages. Neither the constraint nor the externality are policy relevant in this case. The price distortion in the presence of a binding input constraint serves only as an income transfer and in the short run does not influence the firm's decision regarding the quantity of input used when w^p and w^s are both less than w^v, regardless of whether the private price reflects a tax or a subsidy.[3]

When the input constraint is not binding at the social optimum only policy instruments that address both the price distortion and the social damages associated with the use of the input will assure first-best optimality, regardless of whether the constraint is binding in the private problem (regimes 2 and 3). Figure 2 illustrates this result for the case that the constraint is binding at the private (pre-policy) optimum. The diagram is

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[2] For diagrammatic purposes, the optimal level of the abating input (z) is implicitly assumed to be invariant to the level of the polluting input (x), implying that $f_{xz} = g_{xz} = 0$. However, it is likely that the optimal level of z will vary with the constraint on input x, i.e. $z^*(p, w^s, r, p_d) \neq z^*(p, w^s, r, p_d; \bar{x})$, requiring a policy instrument to induce the firm to use the optimal quantity of the abating input. The results regarding the constrained input remain valid when $f_{xz} \neq 0$ and $g_{xz} \neq 0$, provided that the cross input effects are small enough that the constraint remains binding after the shift in the value marginal product and marginal social cost curves.

[3] Though taxes and subsidies have identical implications on the margin, firm entry and exit decisions may be different under a price subsidy than with a tax. The long run effects may therefore also differ (see Spulber).

similar to Figure 1 but depicts a constraint at an input level that is greater than socially optimal. The unregulated firm chooses input level $x = \bar{x}$. Policy intervention is required to assure social optimality in this case.

The firm chooses x^*, the optimal level for input x, only when faced with a price of w^* (Figure 2). The policy instrument must therefore increase the cost to the firm of using input x from the private marginal cost (w^p) to the true shadow price (w^*). This distance is composed of two parts: a) the price distortion, illustrated as the distance from w^p to w^s, denoted t_w, and b) the distance from w^s to w^*, which represents the marginal cost of the externality at the optimum and is denoted t_x^*. The optimal tax on input x in this case is a composite one that includes a correction for the price distortion and a term equal to the marginal social cost of input use to correct for the externality: $t_x = (w^s - w^p) + p_d g_x$. An input tax that does not account for both problems (setting $t_x = t_x^*$ for example) will motivate the firm to move towards but not to the optimum level for input x. Furthermore, if the true private cost is artificially high, i.e. $w^p > w^s$, then ignoring the price distortion and setting $t_x = t_x^*$ will cause the firm to move beyond the social optimum to a level $x < x^*$.

An alternative to input taxes to address non point-source problems is a tax on estimated effluent levels. When input prices are distorted, it is not possible to specify an effluent tax that will, by itself, assure a first-best optimum. Two instruments are necessary to correct both the price distortion and the externality. A tax on the input x must be introduced in addition to a tax on estimated effluent levels. The form of these will be $t_w = w^s - w^p$ and $t_d^* = p_d$, respectively.

When the supply price is below the virtual price, but the private price is not ($w^s < w^v < w^p$) and the true cost, w^*, is less than w^v, the input constraint is not binding in the private problem but is binding in the social optimization framework (regime 4) and a second-best approach is required. This case occurs when the input is taxed, rather than subsidized, and is illustrated in Figure 1 where the firm faces an input price of $w^{p\prime}$. The second-best optimum could be achieved with a partial correction of the price distortion, such as subsidy equal to $w^{p\prime} - w^v$.

Regime 4 can also be examined with Figure 2 by allowing the constraint, $\bar{x}$, to represent a minimum rather than a maximum level for the use of input x. This situation may apply to farmers with appropriative water rights who also have an alternative water source such as groundwater. Appropriative water rights are often assigned on a "use it or lose it" basis and these farmers will use a minimum volume of water each year to maintain their rights. In the general case, the unregulated firm will select $x = x^p$, as it does under regime 3. An input tax $t_x^* = p_d g_x$ (illustrated as the distance $w^* = w^s$) will induce the firm to reduce the use of the input only to $\bar{x}$, the point that the constraint becomes binding. A tax set equal to marginal social damages ($p_d g_x$) will be unnecessarily large. The input level $\bar{x}$ represents the second-best optimum because the constraint is binding at the social optimum and a tax of only $t_x^* = w^v - w^s$ is sufficient to motivate optimal behavior.

In sum, an input tax imposed on the constrained input has no impact when the input constraint is binding at the social optimum (regimes 1 and 4). However, an input tax is an effective means of inducing the firm to consider the social damages associated with the input when the constraint is not binding at the social optimum (regimes 2 and 3). This will be true whenever the true shadow price (including the social cost) for the polluting input is greater than the virtual price, i.e. $w^* > w^v$. Policy intervention is necessary to induce optimal behavior when the constraint is not binding in the private optimum but is binding at the social optimum. However, the level of the policy instrument may be different from that specified as optimal under first-best conditions and found to be appropriate for regimes 2 and 3. These results suggest that the existence and location of an input constraint can be critically important for the policy maker and may have a direct effect on optimal policy choice.

Multiple-firm Analysis

The potential danger of ignoring the conditions that create second-best policy environments when addressing externalities is illustrated in the following example. Two firms or regions are assumed to contribute to a water pollution problem with emissions that arise from use of input (x). The firms are identical except that they face different institutional parameters related to the polluting input. Firm 1 receives a relatively low input allocation and pays a relatively high price per unit, while firm 2 receives a larger quantity of the input and a large price subsidy, i.e. $\bar{x}^2 > \bar{x}^1$, and $w^2 < w^1$. This example of heterogeneous institutional parameters reflects Bureau of Reclamation water supply policies.

The implication of variation in institutional parameters for externality control is examined in Figure 3, where firm 1 and firm 2 are depicted on the right and left sides of a back-to-back diagram. The net private marginal benefits $(PMB_j = pf_x^j - w^j,\ j=1,2)$ and the social marginal benefits resulting from use of input x are illustrated for each firm. Social marginal benefits (SMB^j) equal net private marginal benefits minus the marginal social cost of using input x: $SMB_j = pf_x^j - w^j - p_d g_x^j,\ j=1,2$. Firm 1 does not receive a price subsidy in this example $(w^1 = w^s)$, firm 2 faces a non-binding resource constraint, there is a one-to-one relationship between input use and effluent production ($g_x = 1$), and the inputs x and z are independent ($f_{xz} = g_{xz} = 0$). These assumptions are made for diagrammatic purposes only and the results remain valid for less restrictive assumptions.

The socially optimal level of input use by each firm occurs where the marginal (private) benefits are equal to marginal (social) costs, or when net social marginal benefits are zero. In Figure 3 this occurs at input levels x_1^* and x_2^*. However, firm 2 will chose the input level that sets net private marginal benefits equal to zero in the absence of policy intervention ($x_2 = x_2^o$). Firm 1 would like to do the same but may not use more than $\bar{x}_1$ units of x so that $x_1^o = \bar{x}_1$. The sum of inputs used in the private solution ($X^o = \bar{x}_1 + x_2^o$) generates negative net social marginal benefits.

The marginal effluent products of input use are the same for firms 1 and 2 in Figure 3 and the marginal effluent contributions are additive. The socially optimal level of total input use, given optimal levels for other inputs, is $X^* = x_1^* + x_2^*$. The policy maker must either mandate optimal input levels for each firm or must devise a tool that will reduce input use from the pre-policy level (X^o) to the optimal level (X^*).

As described in the previous section, an input tax or Pigouvian tax on estimated effluent can be effective in the presence of a resource constraint that is not binding at the social optimum, but neither will be an optimal policy choice when a persistent price distortion exists. For example, an effluent discharge tax set at p_d will motivate firm 1 to select x_1^* but firm 2 will reduce input use only to x_2' because its input price is subsidized. Total input use and effluent levels are higher than optimal in this case. This result is a consequence of the price subsidy and not merely due to the difference in input prices. If w^2 represented the true value of the input used by firm 2, x_2^* would coincide with x_2' and the Pigouvian charge p_d would achieve the optimal solution. Under this condition, optimal effluent levels are higher than those implied by X^*. When the price paid by firm 2 is subsidized and does not represent the true value of the resource, the social value of input use is lower than that implied by PMB_2 and the higher input use and effluent production by firm 2 are inefficient.

To achieve the optimal level of input use, and thus of effluent production, it is necessary to develop policy tools that will motivate firms to consider the social costs of input use given that they observe a distorted input price. Under these conditions, there are several regulatory schemes that policy makers might consider.

One set of options includes a tax, either on estimated effluent levels or on input use, that incorporates the shadow value of an effluent (or input) constraint set at desired levels. Suppose that achieving optimal effluent levels requires a fifty percent reduction in use of the polluting input ($X^* = .5X^o$), as illustrated in Figure 3. The first order condition for firm optimization under appropriate tax options requires that input use and effluent production be allocated among firms so that the net private marginal benefits per unit of externality are equal:

$$\frac{(pf_x^1-w^1)}{g_x^1} = \frac{(pf_x^2-w^2)}{g_x^2} = t_d. \tag{3}$$

The PMB_j curves in Figure 3, defined as net private marginal benefits per unit of *input,* also represent the net private marginal benefits per unit of externality under an assumption that $g_x = 1$ for both firms. This assumption is made for diagrammatic simplicity only.[4] An effluent tax level of $\hat{t}_d$ might be expected to achieve the fifty percent reduction objective at least cost, as follows from equation (3) and is illustrated in Figure 3. This is the familiar result that efficiency is achieved when marginal abatement costs are equilibrated among polluters (Baumol and Oates). The input allocations that arise from a policy of charging an effluent tax of $\hat{t}_d$ are denoted $\hat{x}_1$ and $\hat{x}_2$ in Figure 3 ($\hat{x}_1 + \hat{x}_2 = X^*$). An effluent tax of $\hat{t}_d$ does represent an efficient solution to the fifty percent reduction objective when there is no price distortion, i.e. when w^2 represents a true price, but may move the firms further away from the optimal solution when a price distortion is present.

Another possible method for attaining optimal input use is to require uniform reductions in input use among firms. In this scenario, both firms are required to reduce input use by fifty percent. The activity levels resulting from this uniform reduction scheme are denoted x_1^u and x_2^u.

The uniform reduction scheme causes firm 1 to use too little of the input and firm 2 to use too much, relative to optimal levels. However, as seen from Figure 3, a policy such as an effluent tax that equates net private marginal benefits (marginal abatement costs), rather than increasing efficiency relative to the uniform reduction, actually requires further reduction in input use by firm 1 and less reduction by firm 2. Total welfare is thus reduced under this policy.

The welfare changes associated with input allocations implied by a policy of equating net marginal benefits relative to a uniform allocation are illustrated in Figure 3. Area (acdf) represents the loss in welfare experienced by firm 1 as a result of the reduced input use, while area (ghkl)

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[4] For example, a non-constant marginal effluent product can be incorporated in the diagram but will increase its complexity without changing the results.

is the true welfare gain to firm 2 from the greater input allocation. The net private welfare loss is area (acdf) - area (ghkl). The additional benefits that firm 2 would receive from the greater input allocation in the form of the price subsidy (area hijk) is an income transfer only, and is not included when measuring the efficiency gains or losses of a policy. The difference in net social welfare under the two allocation schemes is the sum of the areas between the social marginal benefit curves and the axis and between the input levels associated with each allocation, i.e. area (abef) + area (glmn). Areas (abef) and (glmn) represent negative values because social losses result from the reduced input use by firm 1 and from the increased input use by firm 2, Each input allocation represents the same level of total input use and effluent production in this example. As a result, the difference in (true) private net benefits (acdf-ghkl) is identical to the net social welfare loss (abef-glmn).[5]

Drainage Case Study

The presence and magnitude of welfare losses resulting from alternative policies introduced to address an externality in a second-best setting are examined with regard to an agricultural drainage problem in California. A brief description of the problem setting is presented next, followed by a summary of the model developed to simulate decision making in the area and results from simulations conducted under alternative drainage reduction policies.

Many of the West's most valuable agricultural lands are naturally arid and have been made productive only through large-infrastructure water delivery systems. Developed water is typically sold to water districts under contracts

5 Area (acdf) = (abef) + (bcde) and (ghkl) = (nhkm) - (glmn). In addition, (bcde) = (nhkm) by symmetry. It follows that (acdf) - (ghkl) = (abef) + (bcde) - (nhkm) + (glmn) = (abef) - (glmn).

that specify the quantity of water to be delivered and the price per acre-foot. The terms of these contracts vary by district so that one farmer may receive a generous allotment at a relatively low price while a farmer in a neighboring district may be more limited in the quantity of water received and pay a higher price.

Increasingly, many regions are facing salinity and drainage problems. In these regions, as in arid regions throughout the world, irrigation water is applied in excess of crop water requirements to leach accumulated salts out of the root zone and to provide the minimum amount of water required by plants in all portions of non-uniform fields. In areas with limited natural drainage, this excess applied water contributes to regional saline high water tables that can cause crop yields to decline on overlying lands through upward capillary motion of salts and, in extreme cases, saturation of root zones. Artificial drain systems may be installed to maintain sufficient depth to the high water table and sustain agricultural productivity in these areas.

Much of the water collected in subsurface drain systems installed on the westside of California’s San Joaquin Valley (Valley) is high in dissolved solids and contains naturally occurring selenium, molybdenum, boron, and other elements. The 1983 discovery of toxic concentrations of selenium in waterfowl at Kesterson Reservoir, a holding pond for agricultural drainage located in the Valley, underscored the complex pathways through which water collected in drain systems can concentrate in ecosystems both near the source and far away.

As a result of events at Kesterson, the State of California has established a water quality standard for selenium in the San Joaquin River and is considering standards for other elements and salts (California, 1988). It has been estimated that the river quality standard could be met with approximately thirty percent decreases in drain water volumes discharged from a 94,000 acre drainage study area on the westside of the Valley, and that these decreases are feasible with water conservation through improved management of irrigation applications (California, 1987). The means by which growers might be encouraged to adopt the changes necessary to achieve the recommended drainage reductions have not yet been determined.

The implications of alternative policies for addressing the drainage problem are examined with an agricultural production model designed to simulate farmer decision making in the drainage problem area. The model describes economic, agronomic and hydrologic characteristics pertaining to the area and predicts changes in agricultural production decisions and drainage volumes in response to policy alternatives.

Water can be conserved from agriculture in three ways: (i) water applications can be reduced, allowing crop yields to decline as a result of water stress; (ii) irrigation application efficiency and uniformity can be increased as water applications are reduced to maintain crop yields; or (iii) cropping patterns can be changed to replace crops that have relatively high water requirements with those with lower water needs. All three possibilities are incorporated in the simulation model.

Siphon tube furrow irrigation systems with half mile runs are typically used to irrigate cotton, tomatoes, sugarbeets, and melons (cantaloupes) in the area, while wheat fields are generally irrigated with border check systems. These crops represent 80 to 90 percent of irrigated acreage in the study area.

Changes in irrigation practices can conserve water and may help to reduce drainage production, but will necessarily increase costs. To incorporate this aspect of the problem, crop specific irrigation technology cost-efficiency functions are estimated and included in the model. Irrigation efficiency is defined as the ratio of the depth of water beneficially used (plant needs plus minimum leaching fractions) to the average depth of water applied to a field.

Production of the principle crops is modeled with crop-water production functions. Water applications (x) are multiplied by irrigation efficiency (z) in the production functions so that yield is a function of effective applied water, i.e. the amount of water available for plant growth.

The objective in this problem is to chose cropping patterns and crop-specific water applications and irrigation efficiency levels to maximize net returns to land and management from crop production, subject to water allotments, land availability and constraints on acreage allocations for selected crops, and given the technological relationships specified for crop production and irrigation technology costs. Collected drain water volumes are predicted with a mass balance equation adapted from the Westside Agricultural Drainage Economics model (Hatchett, et al.). The model is specified as a non-linear programming problem and solved with an appropriate algorithm (see Weinberg for a more complete model description).

Prices charged to farmers in the area range from $0 to $36 per acre-foot and allocations from approximately 2.3 to more than 4 acre-feet per acre. Official estimates of the irrigation subsidies for the area range from $15 per acre-foot to nearly $50 per acre-foot (United States).

Three “farms” representing different water districts are selected in order to incorporate heterogeneous institutional parameters in the analysis. One farm (Farm 1) represents a district with a token charge of $1 per acre-foot of delivered water and an allotment of 4 acre-feet per acre. The other two farms face a price of $60 per acre-foot, reflecting “full-cost” water, with farms 2 and 3 receiving allocations of 3.7 and 3.3 acre-feet per acre, respectively.

Base case results are consistent with expectations. Water applications are higher and irrigation efficiencies are lower in Farm 1 than in Farm 2. Similarly, more water is applied and efficiencies are lower in Farm 2 than in

Farm 3. The higher efficiencies required in Farms 2 and 3 to obtain optimum crop yields result in increased irrigation technology and management costs. Higher irrigation costs result in net returns to land and management that are highest in Farm 1 and lowest in Farm 3, although results for Farms 2 and 3 are quite similar. Larger water applications are expected to generate larger volumes of drain water. The model predicts that Farm 1 will generate 1.25 acre-feet per acre of drain water, while Farm 2 will generate 1.02 acre-feet per acre and Farm 3 will generate .96 acre-feet per acre.

As noted above, it has been suggested that water quality objectives could be met with roughly 30 percent reductions in drain water collected in the drainage problem area. A number of alternatives exist for allocating the reduction objective among farms and water districts in the area. Two methods are considered here: an equilibrating scheme and a uniform one. The equilibrating scheme, so called because it equilibrates private marginal abatement costs among farms, specifies a 30 percent regional reduction objective and solves for the least cost means of achieving that objective given observed water prices. The uniform reduction scheme requires that each farm reduce drain water volumes by thirty percent from base levels.

Figure 4 illustrates the income and efficiency effects of alternative drainage allocation schemes. Comparison of average private returns under alternative allocation schemes provides an indication of the efficiency of each. However, in a second-best world this comparison must be made net of the price subsidy to determine true welfare costs of choosing between allocation schemes, these are denoted net returns in Figure 4. The water supply constraints are binding in the base results for all three farms, but are not binding in any case in which the thirty percent drainage reduction is met, This problem can thus be classified in regime 3. The input constraint is not policy relevant in this case, as illustrated in the previous section.

Results indicate that the equilibrating allocation scheme appears to be socially optimal when the price subsidy is not considered. Average private returns are $350.20 per acre under the equilibrating scheme and $349.70 per acre the uniform scheme when comparing Farms 1 and 2 (Figure 4a). The value of the $59 per acre-foot price subsidy is $218 per acre to Farm 1. Deducting this payment prior to comparison of the policy alternatives reveals that average returns are $10 per acre higher with the uniform allocation than with the equilibrating one.

The results demonstrate that application of a first-best policy prescription in a second-best environment can be welfare reducing. This result is not universal, however, and the advantage of the uniform allocation scheme is reduced when comparing farms with increasingly larger differences in initial water allotments. For example, a comparison of Farms 1 and 3 reveals that average private returns are essentially the same under the two drainage allocation schemes and that the equilibrating scheme results in net returns that are $.75 less than with the uniform scheme (Figure 4b).

The impact of the tighter water constraint for Farm 3 is to increase the private value marginal product for the input, in effect increasing marginal abatement costs relative to those for farms with higher water allotments. The equilibrating scheme gains, relative to the uniform one, by incorporating these factors in the final drainage reduction allocations. Nevertheless, the uniform allocation performs as well or better than the equilibrating one in both cases considered here. In addition, uniform reduction policies may require less information and involve lower implementation costs than policies that are generally considered to be efficient for achieving environmental objectives, particularly for cases involving non point-sources of emissions.

Summary and Conclusions

This paper addresses the problem of the second-best in non-point source control and describes conditions under which an effluent tax or set of input taxes will not assure a social objective at least cost. The second-best conditions examined include price distortions for a polluting input and input allocations that are institutionally determined. These conditions characterize Bureau of Reclamation irrigation water distribution policies.

Results indicate that even if an input constraint is binding at the private optimum, it is not policy relevant if it is not binding at the social optimum. The principle result that both effluent taxes and a set of input taxes define optimal policy choices remains valid in this case. However, if the constraint is binding at the social optimum (and is immutable) then: (i) a first-best solution is not attainable, and (ii) the input constraint defines the socially second-best optimal level of the input. No policy action with respect to the constrained input is required, though it may be necessary to introduce a policy tool to motivate optimal changes in the levels of other inputs.

Results indicate that the policy maker can ignore the input constraint if it is not binding at the social optimum but must correct for the input price distortion to achieve a social optimum. A set of input taxes is optimal if the input tax on the polluting input is a composite one including both the marginal social cost associated with input use and the price differential between the “true” and actual price. An effluent tax alone is not capable of assuring that the social optimum is realized.

The price distortion is not policy relevant in the short run when the constraint is binding at the social optimum, although the input price distortion acts as an income transfer with distributional consequences that the policy maker may want to address. The long run implications may be different than the results expressed here.

An agricultural drainage problem is examined to illustrate the implications of policy alternatives in a second-best setting. Equating marginal benefits among firms reduces welfare in this example. The pervasiveness of government intervention in agricultural input and output markets motivates incorporation of these results when designing policies to address the environmental problems associated with irrigated agriculture. Policy makers that do not examine the implications of second-best conditions before making policy recommendations may reduce social welfare in the process of addressing externalities associated with irrigated agriculture.

This paper has re-iterated the warning of Lipsey and Lancaster that society can be made worse off by the attempt to apply “first-best” policy rules in a second-best setting. The results suggest that alternative sources of market failure may have important implications for environmental policy makers. The optimality of policy instruments for externality control requires that these implications are considered.

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Figure 1. Externality control in the case of input price and quantity distortions – Regime 1

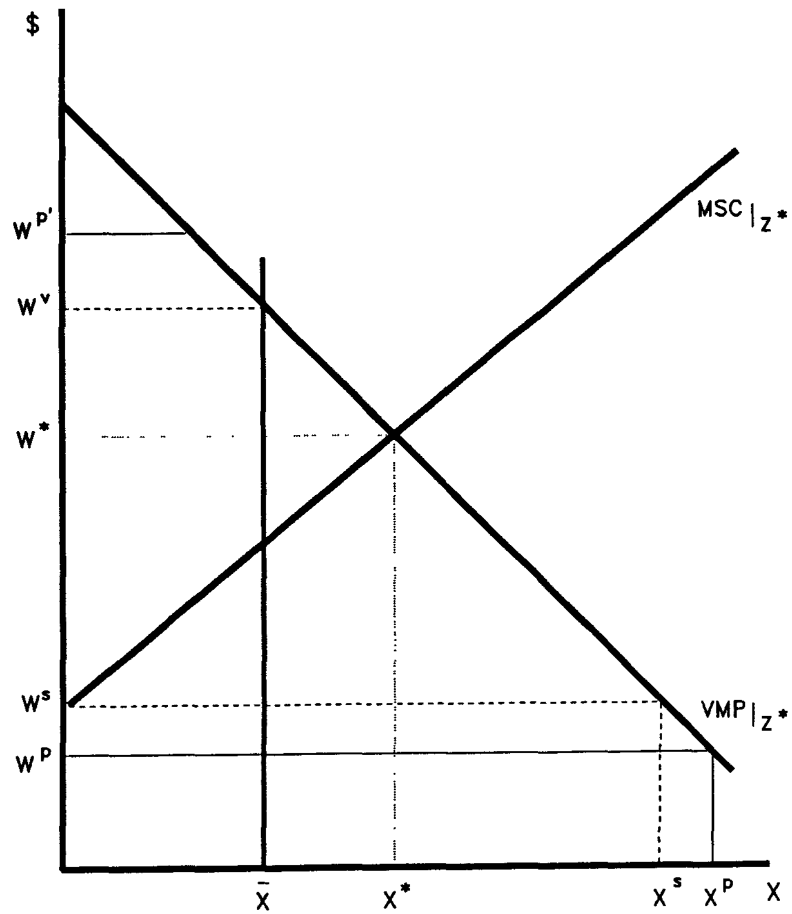


Figure 2. Externality control in the case of input price and quantity distortions – Regime 3

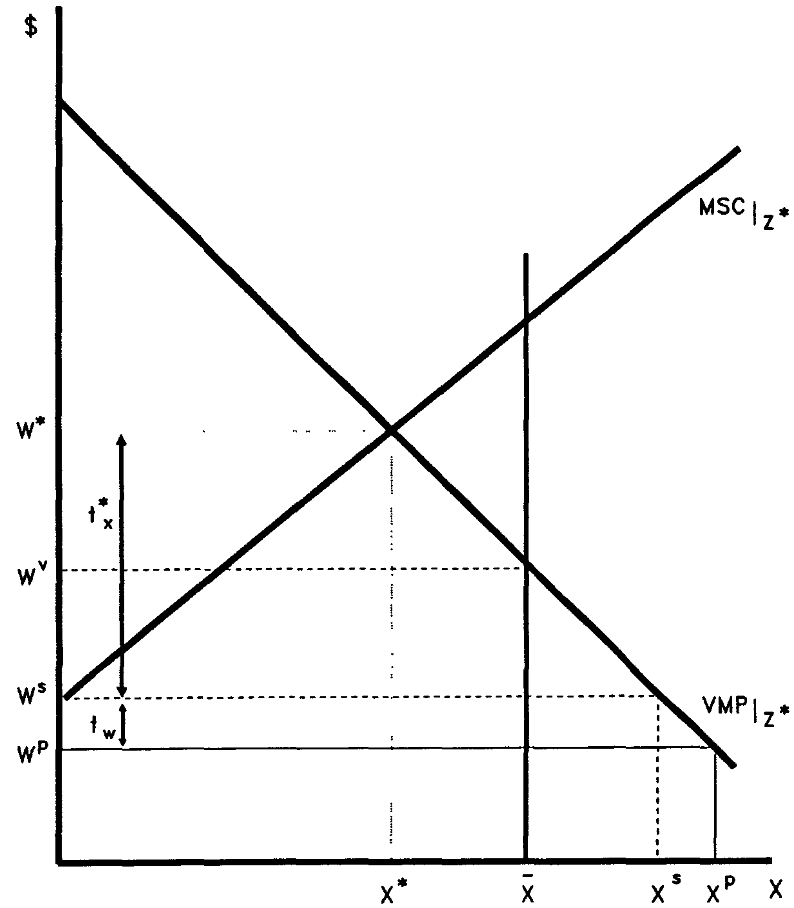


Figure 3. Welfare effects of policy alternatives under second best conditions

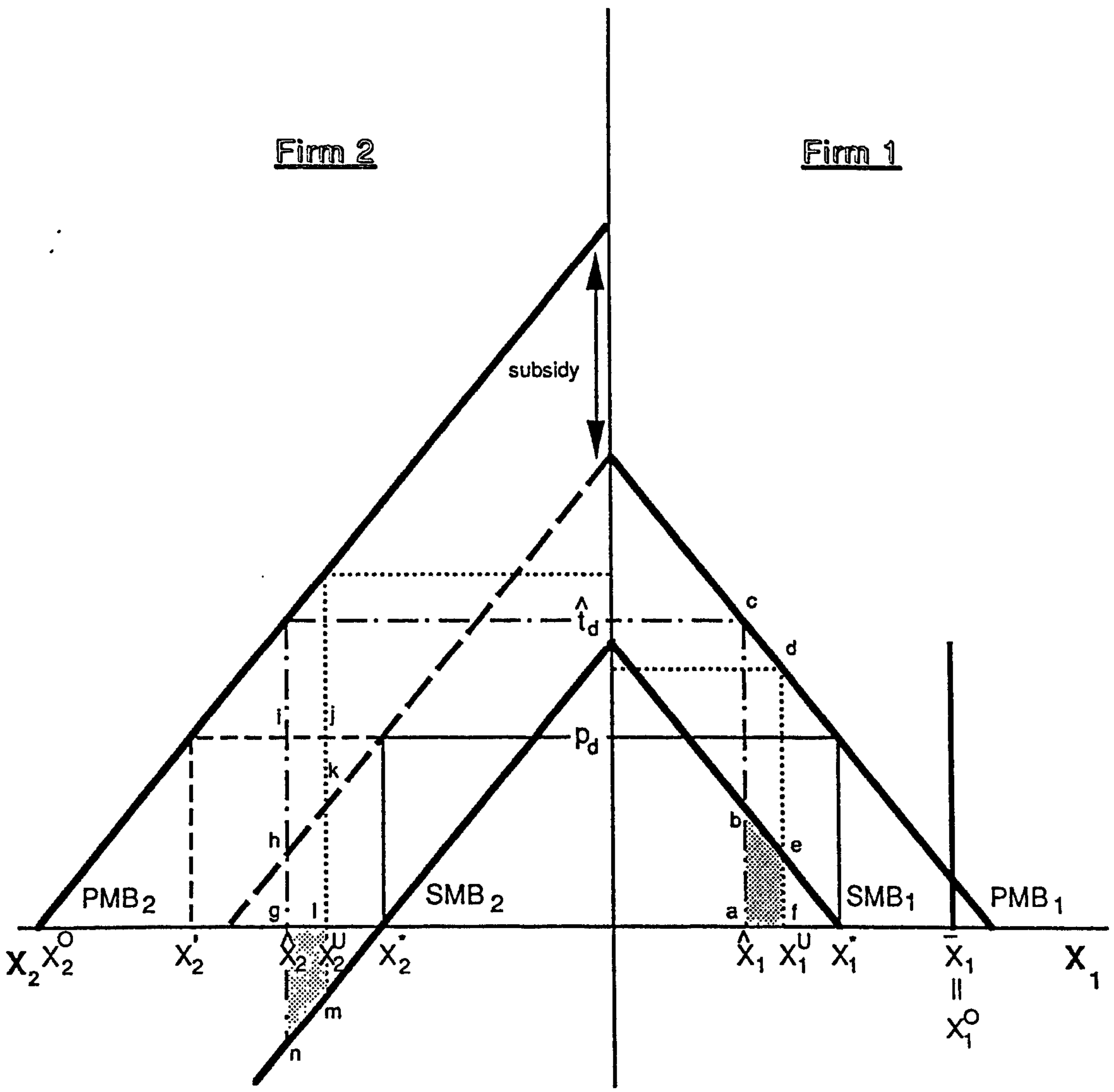
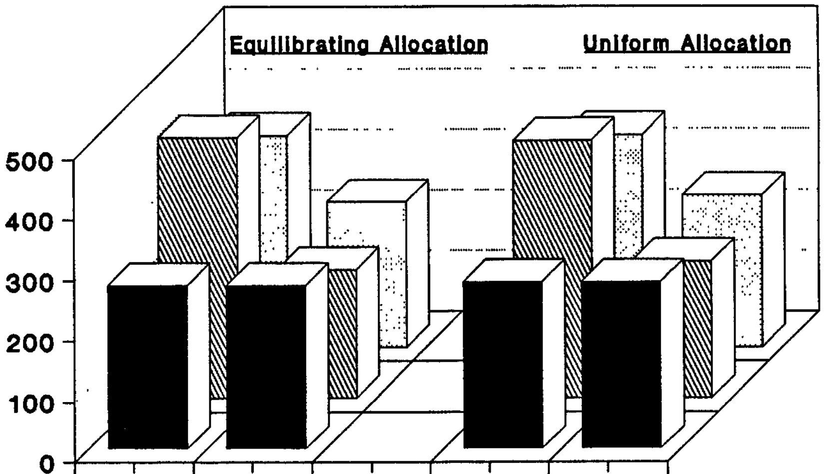
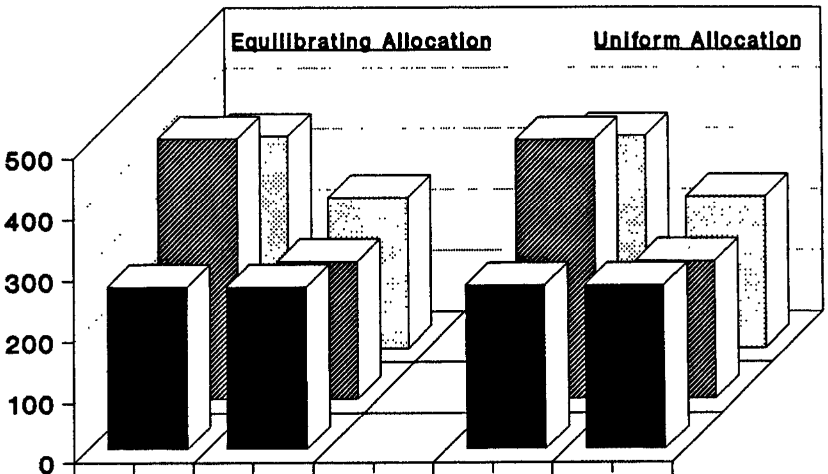


Figure **4. Income effects of policy alternatives**



	Equilibrating Allocation: Private Returns	Equilibrating Allocation: Net Returns		Uniform Allocation: Private Returns	Uniform Allocation: Net Returns
Average	350	241		350	251
Farm 2	430	212		424	226
Farm 1	270	270		275	275

a. Comparison of Farms 1 and 2



	Equilibrating Allocation: Private Returns	Equilibrating Allocation: Net Returns		Uniform Allocation: Private Returns	Uniform Allocation: Net Returns
Average	347	247		347	248
Farm 3	426	226		424	226
Farm 1	268	268		270	270

b. Comparisons of Farms 1 and 3

Point/Nonpoint Source Trading for Controlling Nutrient Loadings to Coastal Waters: A Feasibility Study

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ABSTRACT: This paper attempts to make an initial assessment of the feasibility of point/nonpoint source trading in coastal watersheds. A theoretical model finds that the relatively greater uncertainty and monitoring costs associated with nonpoint source loadings make the setting of a trading ratio difficult a priori. A set of simple screening rules reveals that ten percent of coastal watersheds have significant contributions of loadings from both point and nonpoint sources. These results suggest that point/nonpoint source trading is more likely to work in a small number of coastal watersheds than as a means from bringing nonpoint sources in coastal watersheds under control nationally.

I. Introduction

While the reduction of point source (PS) discharges since 1972 has yielded some improvements in the nation's water quality (e.g. in lower bacterial contamination and higher dissolved oxygen levels), discharges from nonpoint sources (NPSs) remain and have increased as a share of the water quality problem. Impairments from sedimentation, nutrient enrichment, runoff from farmlands, and toxic contamination of fish tissue and sediments have become more evident (USEPA 1990). Extending regulatory controls to include NPSs of water pollution may be necessary for the objectives of Federal water pollution control legislation to be met.

Federal authorities have stepped up efforts to control NPS pollution in recent years. The 1987 Water Quality Act (WQA) authorizes the expenditure of up to $400 million by the EPA to help control water pollution from NPSs States are also required to file management plans under Section 319 of the Act, identifying steps for reducing loadings from NPSs. In addition, the 1990 Amendments to the Coastal Zone Management Act (CZMA) empower EPA and NOAA to manage land use in coastal areas. The President's Water Quality Initiative, while aimed primarily at ground water protection, is also intended to support education and technical assistance efforts to promote voluntary adoption of farm management practices, which reduce agriculture-related impairments of surface water quality, in watersheds identified by 319 reports as having NPS pollution problems.

The principle behind these efforts, and the upcoming reauthorization of the Clean Water Act in 1992, is to treat ecosystems as a whole, rather than focusing on single sources of contamination or single pollutants. The recently enacted amendments to the CZMA serve as an example. While they stress the importance of managing land use in coastal areas to protect surface water quality, the amendments are coordinated with existing water quality management efforts. The most important provision, Section 6217 ("Protecting Coastal Waters"), requires each State to develop a new Coastal Nonpoint Pollution Control Program. The programs will seine as amendments to the State's existing Coastal Zone Management (CZMA) and Nonpoint Source Management (Section 319 of the WQA) Programs. The central purpose is to strengthen links between coastal zone management, water quality programs, and land use restrictions.

A holistic approach may be especially appropriate for bringing NPSs under control. NPSs have grown as a share of the water quality problem because they are harder to identify and control than PSs. NPS contributions depend upon localized factors such as land use, climate, and a host of geomorphological characteristics. Therefore, control of NPSs is more likely to be cost effective if approached in a way that allows for their site-specific nature.

One approach for dealing with these problems would be to allow PSs to sponsor implementation of NPS controls rather than install controls of their own. PS operators and local environmental officials may be better situated to identify and manage localized water quality problems than regional and national regulators, and a PS/NPS trading program may give them both the means and the motivation to do so. Moreover, if the NPS component of the overall water quality problem is significant, and the relative costs of NPS reductions are lower than the costs of additional PS controls, then water quality goals could be met at a lower cost by substituting NPS reductions for PS ones. Two PS/NPS trading programs presently exist in Colorado, and a third has recently been approved for the Tar-Pamlico Basin of North Carolina.

The purpose of this paper is to offer an initial assessment of the feasibility of the PS/NPS trading option for coastal water quality management. Our approach has two parts. First, we explore some of the conceptual and practical issues involved in designing and implementing PS/NPS trading programs. Second, we consider the number of coastal watersheds that satisfy simple but necessary conditions for implementation of PS/NPS trading. Our focus on coastal waters is motivated by the recent reauthorization of the CZMA, which calls for increased efforts by States to improve the condition of coastal waters. We emphasize agricultural nonpoint sources because they are recognized as the single largest (USEPA 1990), the means for their control are less capital intensive than urban runoff controls, and they are more controllable than runoff from forestland or barren lands.

Given this objective, we develop and analyze a simple model of PS/NPS trading in Section 11. This model shows how the random loadings and higher monitoring costs generally associated with NPSs pose difficulties for those who would attempt to set up a trading program. In Section III we discuss some the real world complexities omitted from the conceptual model and consider their possible effects on implementing trading programs. We find no shortage of potential obstacles, but many of the problems also would apply to any attempt to control NPS pollution. In Section IV we conduct a screening study as an initial assessment of the feasibility of trading Programs for managing coastal water quality. Section V offers our conclusion that PS/NPS trading does not have broad applicability for coastal water quality management but might work in a few locations.

II. The Economics of Point/Nonpoint Source Trading

1. Limitations of the Standard Model of Trading

The standard economic argument in favor of PS/NPS trading is a simple one: by allowing point sources with high abatement costs to trade pollution reductions with nonpoint sources that have lower abatement costs, the total costs of achieving a given level of water quality can be reduced. This argument has long been used to support proposals for establishing trading programs among point sources of pollution that have different abatement costs. However, the economics of trading are not as simple when nonpoint sources are involved.

To begin, the standard argument assumes that pollutant loadings are deterministic. This may be a reasonable assumption for point sources, but it certainly is not for nonpoint sources. A characteristic feature of nonpoint loadings is that they are stochastic: the loadings are influenced by a variety of climatic and gemorphological factors. As such, they are more difficult to control than point source loadings, and the effectiveness of nonpoint controls is more difficult to predict.

Nonpoint loadings are also more difficult to measure (Harrington *et al.*, 1985 and Segerson, 1988). This raises questions about how trades involving nonpoint sources would be monitored and enforced. Even if we assume that estimates of, say, average loadings are obtainable, the estimates are likely to be costly. As a result, enforcement costs for nonpoint sources are likely to be high.

Although researchers examining PS/NPS trading have recognized these problems (e.g. see Kashmanian *et al.*, 1986), little attention has been given to what they imply for the economics of trading.[1] Discussions of the problems typically conclude by recommending that uncertainty about the magnitude and effect of nonpoint loadings and the difficulty of measuring them should be accommodated by making the terms of trade less favorable for nonpoint sources (Kashmanian, 1986 and EPA, 1991). The usual recommendation is that the "trading ratio" should be set above one: a unit reduction in loadings by a nonpoint source should count for less than a unit reduction by a point source. Although this recommendation may be appropriate, by itself, it says little about how the trading ratio should actually be set. In this section we develop a simple model that allows us to examine the economics of PS/NPS trading more formally and to identify the factors that determine the magnitude of the "correct" trading ratio.

2. An Alternative Model

For simplicity, let us suppose that our hypothetical watershed contains just one point source and one non-point source (the model can be generalized to multiple sources quite easily). The differences between the two sources are: (i) the pollutant loadings from the point source are deterministic, while the loadings from the nonpoint source are stochastic; and (ii) it is cheaper to measure (or monitor) average loadings from the point source than

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[1] A notable exception is the work by Milon (1987), which explicitly takes into account the stochasticity of loadings and their effects on water quality.

the nonpoint source. These differences are obviously stylized. Point source loadings are likely to also be stochastic, albeit with less variability than nonpoint loadings. Furthermore, there are likely to be differences in the accuracy with which point and nonpoint loadings can be estimated, and not just in the costs of estimating them. (We assume that average loadings from the two sources are measured without error,)

Production, Technology, and Pollutant Loadings

For concreteness, suppose that the nonpoint source is a farm and the point source is an industrial plant and that both are risk-neutral profit-maximizers. The variable and capital inputs used by each are represented by the vectors $\mathbf{x}_i$ and $\mathbf{k}_i$; the vectors include both production inputs and abatement inputs. The capital input vector $\mathbf{k}_i$ characterizes each source's technology.

The inputs used by the point source (i = 1) determine its product output $q_1 = f_1(x_1,k_1)$, and its pollutant loadings $e_1 = g_1(x_1,k_1)$. The same is true for the nonpoint source (i = 2), except that its pollutant loadings also depend on a random variable ω: $e_2 = \tilde{g}_2(x_2,k_2;\omega)$, ω is intended to capture both the inherent stochasticity of nonpoint loadings (due to climatic and geomorphological factors), as well as uncertainty about the relationship between input use and the magnitude of loadings. We shall assume that larger values of ω imply higher loadings.

For our purposes, it is convenient to write nonpoint loadings as the sum of the average loading and a stochastic deviation term with mean zero:

(1) $$e_2 = g_2(x_2,k_2) + \epsilon_2(x_2,k_2;\omega),$$

where $g_2 = E[\tilde{g}_2]$ and $\epsilon_2 = \tilde{g}_2 - g_2$.[2] Note that the above specification does not, per se, impose any restrictions on the source's ability to control the distribution of its loadings. For instance, by choosing its inputs appropriately, the source could independently vary the mean and variance of its loadings.[3]

Environmental Damages

The pollutant loadings result in damages to the waterbody. The monetary value of these damages is given by $D(\theta_1 e_1 + \theta_2 e_2)$. We assume the damage function is "smooth" and that marginal damages are positive and non-decreasing: $D' > 0$ and $D'' \geq 0$. The constants θ_i allow for the possibility that loadings from the two sources have different effects on the waterbody (perhaps due to location or to the chemical composition of the loadings).[4]

Abatement Costs

To define abatement costs, we need to specify the loading parameter that sources would trade. An obvious choice is the average (or mean) loading over some period of time. We shall assume this is the parameter traded, not only because it is convenient analytically, but also because it is the quantity

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[2] Shortle (1990) uses a similar approach.

[3] Whether or not it could actually do this would depend on the form of the loading function $\tilde{g}(\cdot)$ If $\tilde{g}(\cdot) \equiv h(x,k) + \omega$, the source would only be able to influence mean loadings. However, if $\tilde{g}(\cdot)$ has the more general form $\tilde{g}(\cdot) \equiv h(x,k) + t(x,k;\omega)$, it would be able to influence both the mean and variance of loadings.

[4] Location would be relevant for what Tietenberg (1985, p. 22) terms nonuniformly mixed assimilative pollutants. Chemical composition would be important for pollutants such as phosphorus, whose effect on the environment, depends on its exact form, which can vary across point and nonpoint sources (see Krupnick, 1989, p. 16).

traded in existing programs. It should be noted, though, that depending on the pollutant and the characteristics of the watershed, it may be preferable to consider some other parameter, such as maximum loadings.[5]

Each source's abatement costs can now be defined in terms of its average loadings. Two types of abatement cost functions are defined: a restricted cost function $C_i(\bar{e}_i,k_i)$ and an unrestricted cost function $C_i(\bar{e}_i)$. The restricted cost function gives the source's costs, in terms of foregone profits, of achieving an average loading of $\bar{e}_i$ using a prescribed technology k_i. $C_i(\bar{e}_i,k_i)$ can be derived from the source's profit-maximization problem.[6]

The unrestricted cost function, $C_i(\bar{e}_i)$, gives a source's abatement costs when it is free to choose the technology it uses. We can define $C_i(\bar{e}_i)$ in terms of the restricted cost function:

$$(3) \qquad C_i(\bar{e}_i) \equiv \min_{k_i} C_i(\bar{e}_i,k_i).$$

For both the restricted and unrestricted cost functions, marginal abatement costs are positive over the relevant range: $-\partial C_i/\partial \bar{e}_i > 0$ and $dC_i/d\bar{e}_i > 0$. Furthermore, assuming the production and loadings functions have

[5] It may even be desirable to trade more than one parameter, for instance, average monthly loadings and maximum daily loadings. Milon (1987) discusses the shortcomings of using average loadings alone.

[6] Letting $f_i(x_i,k_i)$ represent a source's production function,

$$C_i(\bar{e}_i,k_i) \equiv \pi_i^* - \left[\max_{x_i} \{p_i f_i(x_i,k_i) - w_i'x_i - r_i'k_i\} \quad \text{s.t. } g_i(x_i,k_i) = \bar{e}_i\right].$$

The first term above (π_i^*) represents the source's profits when its actions are unrestricted; the second term represents its profits when it must restrict average loadings to $\bar{e}_i$ using the prescribed technology k_i. w_i and r_i are simply the prices of the variable and capital inputs; and p_i is the price source receives for its output.

the appropriate properties, the marginal abatement cost schedules will have the usual downward sloping shape: $-\partial^2 C_i/\partial \bar{e}_i^2 < 0$ and $-d^2C_i/d\bar{e}_i^2 < 0$.

Monitoring and Enforcement Costs

Given the importance of monitoring and enforcement costs in PS/NPS trading programs, we incorporate them in our model. However, to keep the model tractable, we restrict attention to enforcement policies that achieve full compliance, i.e., they ensure that neither source exceeds its allowed average loadings. The regulator accomplishes this by auditing the sources at random and fining them if they are found exceeding their allowed average loadings.

Let α_i denote the probability a source is audited, and $F(\bar{e}_i - \mu_i)$ the fine the source faces for exceeding its allowed average loading μ_i. For the source to be compliant, the marginal expected fine it faces when $\bar{e}_i = \mu_i$ must be no smaller than its marginal cost of abatement:

(4) $$\alpha_i F'(0) \geq -\partial C_i(\mu_i, k_i)/\partial \bar{e}_i,$$

Note that this condition is appropriate even when the source chooses its technology, because technology is presumably fixed when the source makes its day-to-day compliance decisions; these decisions would be based only on the source's variable costs.[7]

Taking the fine schedule as exogenous, condition (4) implies that by setting an audit probability of

(5) $$\alpha_i^c = \phi[-\partial C_i(\mu_i, k_i)/\partial \bar{e}_i],$$

[7] Thus we are modeling continuing compliance rather than initial compliance (Russell, *et al.,* 1986, p. 8).

where $\phi = [F'(0)]^{-1}$, the regulator could ensure a source's compliance, Enforcement costs in this situation would consist solely of audit costs.[8] Total expected enforcement costs would be $\alpha_1^c A_1 + \alpha_2^c A_2$, where A_i denotes the cost of an audit. Given our premise that it is more costly to monitor the loadings of nonpoint source than a point source, A_2 would be greater than A_1.

3. The Benchmark Optimum

We are now ready to specify a benchmark social cost minimization problem. The solution to this problem is intended to provide a realistic reference against which to compare trading programs; it does not represent the first-best solution. The benchmark problem is one where the regulator can dictate a source's average loadings (μ_i) and the technology it uses (k_i), but not its variable input use (x_i). (Enforcing variable input use is assumed to be prohibitively costly.)

The regulator ensures the source's compliance with μ_i, by auditing it with probability α_i^c. Ensuring the source adopts the prescribed technology is assumed to be costless at the margin -- it simply requires a one time check of the technology the source is using.

Formally, the benchmark problem is to find the technologies and average loadings that minimize the sum of abatement and enforcement costs, plus the expected damages from pollutant loadings:

$$(6) \quad \min_{\mu, k} \sum_{i=1}^{2} \left[C_i(\mu_i, k_i) + \alpha_i^c A_i \right] + E\left[D(\theta_1 \mu_1 + \theta_2(\mu_2 + \epsilon_2^*(\mu_2, k_2; \omega))) \right],$$

[8] There would be no fine-related costs, since fines would never be levied.

where $\epsilon_2^*(\mu_2, k_2; \omega) \equiv \epsilon_2(x_2^*(\mu_2, k_2), k_2; \omega)$ is the indirect loading deviation function.[9] In writing the expression for damages we have made use of the equalities $e_1 = \mu_1$ and $e_2 = \mu_2 + \epsilon_2^*$.[10]

Substituting for α_i^c from (5), and denoting the covariance operator by COV, the first-order conditions for the benchmark problem can be written as:11

$$(7) \qquad \frac{-\partial C_1}{\partial \mu_1} + A_1 \frac{\partial^2 C_1}{\partial \mu_1^2} \phi = \theta_1 E[D'(\cdot)],$$

$$(8) \qquad \frac{-\partial C_2}{\partial \mu_2} + A_2 \frac{\partial^2 C_2}{\partial \mu_2^2} \phi = \theta_2 E[D'(\cdot)] + \theta_2 \mathrm{COV}\left[D'(\cdot), \frac{\partial \epsilon_2^*}{\partial \mu_2}\right],$$

$$(9) \qquad \frac{\partial C_1}{\partial k_1} = A_1 \frac{\partial^2 C_1}{\partial k_1 \partial \mu_i} \phi, \qquad \forall\, k_1 \in \mathbf{k}_1,$$

$$(10) \qquad \frac{\partial C_2}{\partial k_2} = A_2 \frac{\partial^2 C_i}{\partial k_2 \partial \mu_2} \phi - \theta_2 \mathrm{COV}\left[D'(\cdot), \frac{\partial \epsilon_2^*}{\partial k_2}\right], \qquad \forall\, k_2 \in \mathbf{k}_2.$$

Optimal Allocation of Average Loadings

The first condition (7) calls for μ_1 to be set so that the sum of the marginal cost of abatement and the marginal cost of enforcement for the point

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9 The choice function $x_i^*(\mu_i, k_i)$ is the solution to the profit maximization problem in footnote 6.

10 The above specification assumes that the regulator is risk-neutral. Allowing for risk aversion yields qualitatively similar results to those obtained below for the case of a strictly convex damage function ($D'' > 0$).

12 We have made use of the relationship $E(ab) = E(a)E(b) + \mathrm{COV}(a,b)$. Here, $a = D'(\cdot)$ and $b = \partial \epsilon_2^*/\partial h$, where $h = \mu_2$ or k_2. Since ϵ_2^* has mean zero, $E[\partial \epsilon_2^*/\partial h] = \partial E[\epsilon_2^*]/\partial h = 0$.

source is equal to the expected marginal damage it causes. This is a natural extension of the usual rule that marginal costs should be equated to marginal damages.

The condition for the average loading from the nonpoint source (8) is somewhat more complicated: it contains an extra covariance term that reflects the uncertainty about nonpoint loadings. The sign and magnitude of the term depend, roughly, on the curvature of the damage function and on the relationship between the mean and variance of the source's loadings.

The term vanishes if damages are linear (since D'(.) is then a constant), o if the mean and variance of loadings are unrelated $(\partial\epsilon_2^*/\partial\mu_2 = 0)$. But if the damage function is convex (D" > 0) and larger average loadings imply a larger variability in loadings $(\partial\epsilon_2^*/\partial\mu_2\partial\omega > 0)$, the term is positive. In this case, the covariance term can be be thought of as representing the damage premium associated with the uncertainty about loadings.

Optimal Choice of Technology

Turning to the conditions for the capital inputs, (9) and (10), we can establish that the regulator's choice of technology is not the same as a source's. From (3), we can verify that a source would choose its technology so that $\partial C_i/\partial k_i = 0$, $\forall k_i$. Conditions (9) and (10) are more complicated than this: the regulator takes into account the effect of technology on enforcement costs and, in the case of the nonpoint source, on the damage premium.[12] The sources ignore these costs when choosing k_i, since they do not bear them,

[12] That technology may influence enforcement costs and/or the variability of loadings should not be surprising. Suppose, for example, that the farm can lower its nitrogen loadings by either building a retention pond or by reducing the amount of fertilizer it applies. Both enforcement costs and the variability of loadings are likely to differ for these two technologies.

Once again, the damage premium in (10) is positive only if damages are non-linear. If damages are linear, the variability of loadings is irrelevant, and the regulator will only worry about average loadings. However, if the damage function is convex, the regulator will exploit opportunities to reduce the variability of loadings by prescribing the appropriate technology.

The above results suggest the following two broad conclusions: (i) uncertainty about nonpoint source loadings is of concern only if the damage function is nonlinear; and (ii) allowing the regulator to prescribe the technology sources should adopt can reduce social costs, to the extent that the choice of technology influences the magnitude of enforcement costs and the damage premium.

4. Implications for the Design of a Trading Program

Let us examine the implications of the above analysis for the design of a PS/NPS trading program. We begin by considering the issue of the appropriate trading ratio. We shall assume, for the moment, that the regulator can dictate the technology a source adopts, and can thereby ensure that conditions (9) and (10) are satisfied. The question, then, is whether the regulator can set the trading ratio so that conditions (7) and (8) hold.

Rearranging (7) and (8), and dividing one by the other, we find

$$(11)\qquad \frac{\partial C_2/\partial \mu_2}{\partial C_1/\partial \mu_1} = \frac{-A_2\phi\cdot\partial^2 C_2/\partial\mu_2^2 + \theta_2 E[D'(\cdot)] + \theta_2 \mathrm{COV}[D', \partial\epsilon_2^*/\partial\mu_2]}{-A_1\phi\cdot\partial^2 C_1/\partial\mu_1^2 + \theta_1 E[D'(\cdot)]} .$$

The LHS of this equation represents the ratio of marginal abatement costs. Therefore, the RHS is the trading ratio required for the benchmark conditions in (7) and (8) to hold.

In the simple setting where loadings are deterministic and enforcement is costless, the RHS of (11) reduces to θ_2/θ_1; this is the familiar result that the optimal trading ratio should equal the relative environmental impacts of the loadings from the two sources.

The optimal trading ratio is considerably more complicated when loadings are uncertain and enforcement is costly. The ratio now depends, in part, on the magnitude of the damage premium. The argument that the trading ratio should be increased to compensate for the uncertainty about nonpoint loadings is corroborated by (11), provided damages are convex (or the regulator is risk averse) and higher average loadings imply greater variability. If either of these conditions does not hold, uncertainty about nonpoint loadings is irrelevant in determining the proper trading ratio. Thus, in recommending that trading ratios be set above one, it is being implicitly assumed that both these conditions do hold.

The relative magnitude of marginal enforcement costs also influences the optimal trading ratio. Enforcement costs are a function of the audit costs A_i and the slopes of the marginal abatement cost curves $\partial^2 C_i/\partial \mu_i^2$. If the cost curves have similar slopes, the relative magnitude of marginal enforcement costs is just a function of A_2 and A_1. In this case, the higher audit cost for the nonpoint source ($A_2 > A_1$) has the effect of lowering the optimal trading ratio. This is not surprising: if it is more expensive to ensure the nonpoint source's compliance, abatement burden should be shifted toward the point source, which requires lowering the trading ratio. This effect is reduced to the extent that the the marginal abatement cost curve for the point source is likely to be more steeply sloped than the curve for the nonpoint source. A priori, it is difficult to specify whether this difference would more than compensate for the difference in audit costs.

The above discussion reveals that calculating the appropriate trading ratio requires a substantial amount of information. Equation (11) shows us that the trading ratio must reflect all relevant social costs other than the direct costs of abatement. Since sources just bear the latter costs, the only means of forcing them to take into account the other social costs incurred is by adjusting the trading ratio appropriately.

We have assumed thus far that the regulator can dictate the technology that sources use. The form of equation (11) does not change drastically when sources are free to choose their technologies. Analytically, the primary difference is that the $\mathbf{k}_i$ must be replaced by each source's choice functions $\mathbf{k}_i^*(\mu_i)$ (see eq. (3)). This change does, however, have important implications for the regulator's ability to attain the benchmark optimum. The regulator now has only one policy instrument for each source, namely the allowed average loading, μ_i. Although the regulator will take into account the effect of the allowed loading on a source's choice of technology, it will not be able to costlessly influence the source's technology choice. As a result, the benchmark will no longer be attainable, and social costs will be higher.

Regardless of whether the regulator can prescribe technologies, the above analysis makes clear that setting the trading ratio is no simple matter. In particular, it shows that the two distinguishing features of nonpoint loadings -- their uncertainty and the higher costs of monitoring them -- may have opposing influences on the optimal ratio. Therefore it is questionable whether one can recommend *a priori* that trading ratios should favor point sources and thus be set above one. The analysis shows that one has to to consider the nature of the damage function and the relative costs of enforcement.

III. Technical and Practical Aspects

1. Real World Complexities

Implementing a successful PS/NPS trading program is of course much more difficult than the above analysis suggests. Many simplifying assumptions of the model depart from reality in important ways. These departures represent technical and practical complications in implementing trading. Significant among them are the model's represention of regulators' objectives, their monitoring and enforcement capabilities, their knowledge of the costs and effectiveness of NPS control methods, and the fate and transport of target pollutants. Many of these problems would encumber any attempt to bring NPSs under control, but some are unique to the trading option. If NPS control in some form is deemed necessary for achievement of legislated water quality goals, then the latter set of problems is more germane to our discussion. To avoid the all too common mistake of assigning all of these problems to trading programs alone, we shall discuss each of them in turn for the purpose of classification.

2. Problems Unique to the PS/NPS Trading Option

The trading option differs from other approaches to NPS control because it relies on a market to coordinate the actions of relevant economic agents. A key aspect in successfully coordinating them is having them in the appropriate number. Failure can come from having too many or too few participants in a pollution rights market.

The first side of the coordination problem pertains to measurement: we cannot measure pollutant loadings from individual NPSs therefore we do not know the previous loadings of individual NPSs. Without historical records for NPS loadings, we cannot calculate the pollutant reductions to be traded. PSs are unlikely to enter into what would essentially be a purchase of an ambiguous property right. We are able to estimate gross NPS loading and even to classify them by origin: urban, forestland, and agricultural runoff, etc. Because of our ability to measure gross but not individual loadings it might be necessary to involve all or most of a watershed's farmers in a given trade.

The coordination problem also has its abstract side. While often described conceptually as a perfectly competitive market, PS/NPS trading more closely resembles a private subsidy scheme: PSs avoid costly abatement by paying farmers to alter their practices. Unfortunately lower marginal abatement costs for NPSs alone may not make trading a reality. Coasian transactions costs are likely to eliminate some trades that would lower total control costs. A "stick" provision might be necessary to encourage farmers to participate in what might otherwise seem to others a profitable trade but to them is costly to arrange.

These two problems could exacerbate one another. To get an accurate estimate of the potential pollutant reduction, a large number of farmers might need to participate. Bargaining costs, though, might prevent transactions with numerous participants. In any case, a sufficient number of NPSs will have to exist to create a loading reduction the PS can use. The fact that failure can come from having too many or too few participants for the pollution rights market is part of what makes this coordination problem difficult. Other problems affecting NPS control in general also encumber the PS/NPS trading coordinator.

3. Four Problems with NPS Controls in General

Modeling Regional Objectives

The conceptual model has a single objective while the region's water quality managers generally will have more than one. Cooperation among participating regulators is necessary for any type of NPS control to achieve its cost and environmental goals, so this problem would likely confront any NPS control plan. The single coastal water quality regulator of the model in reality is probably several cooperating watershed authorities. Within any portion of the watershed, trading is likely to be unworkable since the entire area of influence of trading must be included if its water quality is to be protected. The entire system is likely to fall into several political jurisdictions, and a single authority would have to be empowered to run a trading program:

The absence of such an authority would mean further institutional change is necessary and make implementation of trading more difficult. Regardless, the smaller jurisdictions will not be quick to relinquish their powers to a regional authority. Two of the existing examples of PS/NPS trading are suggestive here. For the PS/NPS program for Dillon Reservoir, a threatened growth moratorium provided the motivation behind the formation of the Northwest Colorado Council of Governments. On the other hand, the Tar-Pamlico program in North Carolina faces no such crisis and its Basin Association is having more difficulty setting rules for trading (Anderson 1991). In sum, wedding the objectives of individual jurisdictions with regional cost minimization can prove difficult.

Monitoring and Enforcement

Any approach to NPS control would encounter monitoring and enforcement problems to some degree, but a trading program may be more susceptible to them. If a market for pollution reductions is to be established, the regional authority must be able to enforce trades and detect violations. Two problems may exist. First, many states simply do not have standards for nutrients. These states could establish them or link nutrient discharges to dissolved oxygen standards, but either approach would require the use of water quality models and is not costless. For a number of reasons such models are far from simple. (a) Estuarine models must include the effects of tidal incursions, normal surface flows, groundwater inflows of nitrates, and benthic sediments. (b) Because of the presence of both saline and fresh water, more than one pollutant can be limiting. (c) Phosphorus and nitrogen must be in dissolved and in inorganic forms to be available to phytoplankton for growth. Chemical and biological activity can convert other P and N forms to these forms, and vice versa. Second, trades may be difficult to enforce and violations difficult to detect with the present monitoring capacity and more difficult still to attribute to individual sources (Segerson 1988). PSs do not present so much of a problem here, but NPS controls themselves would probably have to be monitored rather than the resulting loadings.

Uncertain Performance of NPS Controls

NPS controls, as mandatory measures undertaken for water quality improvements, are virtually untried compared to our history with PS controls. This uncertainty would affect any attempt at NPS control and not just trading. The cost and effectiveness of NPS control methods are not known to regulators or dischargers with certainty. The level and number of acceptable violations under PS/NPS trading in reality is a stochastic decision problem.[16] Pollutant loadings and the physio-chemical reactions to them are uncertain. The added risk could make PS/NPS trading difficult to defend, both politically and to potential participants. Explaining the parameters of a risk management problem might prove difficult since the public is used to the relative certainty of PS controls. Reduction of the likelihood of a violation may be politically desirable, but too high a required likelihood might have costs exceeding the possible benefits (Milon 1987). Also, to encourage program participation, farmers uncertain of the efficacy of NPS controls might need a “stick” provision to go along with the “carrot” (i.e. the subsidy from the PS) that trading would provide.

Fate and Transport of Pollutants

We do not know enough about the fate and transport of target pollutants. NPS controls that reduce pollutant loadings to surface water (e.g. grassed waterways and animal waste treatment lagoons for the Tar-Pamlico Basin

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[16] Point source control is also a stochastic decision problem, although, to a lesser extent.

program) may actually increase loadings of these or other pollutants to groundwater. While predictive models are available, large areas of uncertainty remain.[17] Obviously, the regulatory authority should consider the consequences of these problems for trading as well and would have to for any other type of NPS control.

4. The Appropriate Question

PS/NPS trading may pose higher administrative costs and greater environmental risks than technology standards for PSs. If some type of NPS control is soon in coming, however, the appropriate question relates to the additional problems the trading option creates relative to other approaches to NPS control. Arguably, our ignorance and uncertainty related to monitoring and enforcement, untried NPS controls, and the fate and transport of pollutants extends beyond the trading option to any attempt to bring NPSs under control. The additional problems unique to trading pertain to our lack of historical information that would enable calculation of actual loading reductions and to transactions costs. These latter problems appear considerable but are distinct from the more general difficulties with NPS control. Below, one of these issues (size of contribution of PSs and NPSs to loadings) is the basis for a screening study that serves as a conservative means for assessing the number of coastal areas nationally for which a PS/NPS trading program might be feasible.

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[17] USEPA (1989) surveys the types of models available for rivers and lakes.

IV. Feasibility of Trading for Managing Coastal Water Quality

1. Background

In this section, we focus on the question of whether PS/NPS trading of pollution reductions is feasible in America's coastal water systems. We examine coastal water systems for several reasons. First, it limits the scope of our analysis to a manageable level: instead of analyzing all pollutant sources and water quality conditions nationwide we can look at a subset of watersheds in coastal states. Second, coastal water quality issues are highly policy relevant, given the recent Coastal Zone Management Act amendments and the renewed interest in protecting coastal water quality. Finally, as we discuss below, there exist several detailed data sources on sources and types of pollutant flows into coastal waters that facilitate a screening analysis.

Below, we develop simple screening rules to identify water systems which may be potential candidates for PS/NPS trading. We apply these rules using the data on coastal water systems to get an initial assessment of how many water systems could potentially be managed with PS/NPS trading.

2. Data Sources

Data used in this study come from three basic sources: the National Coastal Pollutant Discharge Inventory, the National Resources Inventory, and the EPA's AGTRAK database.

- The National Coastal Pollutant Discharge Inventory (NCPDI) has been developed by the National Oceanic and Atmospheric Administration (NOAA). The NCPDI contains pollutant loading estimates for all major types of pollutant sources located within coastal counties in the continental US (excluding the Great Lakes). Data are calculated on a base line of 1985 conditions. The pollution estimates are drawn from a variety of sources and based on many different methodologies (See Basta et. al. for details).

  Data were obtained from the NCPDI on pollutant loadings for four types of pollutants: Nitrogen, phosphorus, suspended sediments, and 5-day biochemical oxygen demand (BOD5). Pollutant sources were broken down into eight categories: Wastewater treatment plants, powerplants, industrial sources, urban runoff, cropland runoff, pastureland runoff, runoff from barren land, and upstream sources (pollutant loadings from inland regions).

- The National Resources Inventory (NRI) is conducted every five years by the USDA's Soil Conservation Service. The NRI is designed to obtain natural resource data usable for analysis at substate (multi-county) level, such as watersheds. The NRI records a variety of land use and resource conditions, including agricultural uses, cropping history, soil condition, conservation need and practices, and estimated soil and wind erosion.

  Data were obtained from the NRI to augment the pollutant loading data from the NCPDI. The NRI data were obtained for sample points in coastal watersheds (USGS cataloging units). For each coastal cataloging unit, estimates were obtained of total soil erosion, soil erosion from cropland, average rates of soil loss (tons/acre/year), and the number of acres which were identified in the survey as in need of some form of soil conservation treatment.

- The EPA's AGTRAK database records citings of water quality impairments related to agricultural sources. The citings are taken from state inventories of nonpoint source pollution problems filed with EPA under Section 319(h) of the Water Quality Act of 1987 and Section 305(b) of the Federal Water Pollution Control Act. The AGTRAK database records the number of identified impairments related to pesticides, nutrients (nitrogen or phosphorus) and sediment in each county.

It should be emphasized that the AGTRAK data system only gives a qualitative assessment of water quality. Impairments are simply reported by the number of identified water quality problems in each county. There is no indication of the geographic extent of the reported problems (such as number of river miles or acres of lakes impaired. Also, "impairments" are rather loosely defined; impaired bodies are those the states have determined to be of insufficient quality to meet "designated uses." Standards as to what constitutes "meeting designated uses" vary from state to state. Accordingly, the AGTRAK data should simply be used as an indicator that somewhere within a given county there have been identified impairments of surface water quality related to agricultural sources of pollution.

Data from these three sources have been combined to give an overall characterization of the sources and types of coastal water pollution and related resource conditions. Given to the amount of data, complete descriptive statistics are not presented here; summary tables are presented in the Appendix. Complete details on data sources and estimation procedures are available from the authors on request.

3. Data Analysis

Characteristics of Coastal Pollutant Flows

Data from the NCPDI covered 350 USGS cataloging units and 415 counties. Table 1 summarizes on a regional basis the relative shares of pollutant flows provided by point and nonpoint sources.

Overall, agricultural sources supply about forty percent of all nitrogen loadings, about thirty percent of phosphorus loadings, about 45 percent of sediment loadings, and about 28 percent of BOD5 loadings.[18] Agricultural loadings of nitrogen and sediment generally exceed point source loadings of these pollutants. (The figures reported in Table 1 are, of course, regional averages. Substantial variation is found among individual watersheds — see Appendix A).

We single out agricultural nonpoint sources (as opposed to all nonpoint sources) for several reasons. First, the EPA has identified agricultural nonpoint sources as the largest single component of nonpoint source pollutant

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[18]Agricultural sources are defined here as pollutant loadings from harvested cropland, non-harvested cropland, pastureland, and rangeland. Other non-point sources in the NCPDI which are considered non-agricultural sources for our purposes include forestland, barren land, and urban non-point runoff.

loadings for the nation as a whole (USEPA 1990). Second, nonpoint source pollution controls on agricultural lands are less capital intensive (and thereby less expensive) than urban nonpoint controls (which involve installation of stormwater runoff control systems). Finally, other non-urban, non-agricultural sources of pollutant flows (such as runoff from forestland or barren land) are not readily controllable, at least to the extent that runoff from harvested cropland may be.

Table 1 also reports erosion conditions and soil conservation needs obtained from the NRI. Average erosion rates are highest in the East and Gulf regions. Also, the percentage of agricultural lands identified as needing some form of conservation treatment is highest in the Gulf and the East. Significantly, soil erosion is less severe in the coastal watersheds in the West: erosion rates in 1987 were less than 2 tons/acre/year: agricultural lands accounted for less than one-fourth of all erosion, and slightly less than 30 percent of agricultural lands were thought to need some form of conservation treatment.

Table 1

Sources of Coastal Pollution Loadings
And Related Erosion Data,
By Region

	Shares of Nitrogen Loadings		Shares of Phosphorus Loadings		Shares of Sediment Loadings		Shares of BOD5 Loadings		Average Cropland Erosion	Agland Share of all Erosion	Pct of Ag. Lands Needing Conservation Treatment
Region	Ag.	Point	Ag.	Point	Ag.	Point	Ag.	Point			
East	44.9	14.1	32.2	23.2	44.0	13.7	21.2	22.3	3.99	70.5	43.4
Gulf	38.7	19.3	10.2	42.5	57.8	3.5	33.5	8.1	2.90	63.7	65.7
west	44.1	13.6	33.4	25.3	43.3	6.6	28.5	23.4	1.99	24.7	28.9

Data based on cataloging unit-level estimates of pollutant loadings from NCPDI. Data on erosion and lands identified as needing conservation treatments from NRI. Agricultural loadings are loadings from cropland and pastureland. Point source loadings are loadings from wastewater treatment plants, powerplants, and industrial sources. Erosion estimates are tons/acre/year. Total number of coastal cataloging units assessed: 350.

Agricultural Impairments of Coastal Water Quality

Data from the AGTRAK database were examined to determine the extent of agricultural nonpoint source pollution in coastal counties[19]. State reports to the EPA indicate that half of the coastal counties contained at least one water body which did not meet designated uses due to pollution from agricultural nutrients (nitrogen or phosphorus). About a third of the coastal counties had at least one water body impaired by agricultural sources of

[19]The AGTRAK database is recorded on a county-by-county basis, rather than by USGS Cataloging Unit. The NCPDI data are available on either basis. Although it is technically feasible to construct county-level estimates of erosion and conservation needs from the NRI, the NRI was not designed to give statistically meaningful estimates at the county level. Accordingly, county-level data were not drawn from the NRI.

sediment. To get some idea of the relationship between agricultural pollutant loadings and impairments, the data from AGTRAK were merged with data from NCPDI, and screened for “significant” agricultural pollutant loadings. We define “significant” agricultural loadings in three ways: 20 percent, 25 percent, or 30 percent of total nutrient or sediment loadings. Between 114 and 132 counties out of 415 showed both identified agricultural impairments and significant agricultural sources of nutrients. Between 84 and 96 counties showed both significant agricultural sediment loadings and identified impairments from agricultural sediment.

Table 2
Agriculture’s Contribution Water Quality Impairments
in Coastal Counties

	Nutrients (No. of Counties)	Sediment (No. of Counties)
Counties with identified impairments from agricultural sources	225	155
Counties with identified impairments and agriculture contributes at least 20 percent of pollutant loadings	132	96
Counties with identified impairments and agriculture contributes at least 25 percent of pollutant loadings	122	87
Counties with identified impairments and agriculture contributes at least 30 percent of pollutant loadings	114	84

“Impairments” are defined as indication by state authorities that a county has a water body which does not meet designated uses in their 319 reports” to EPA. Data on impairments from AGTRAK database. Data on sediment and nutrient loadings from NCPDI, Total number of coastal counties assessed: 415.

Application of Screening Criteria for Potential Point-Nonpoint Trading

In order for point-nonpoint trading to contribute to overall water quality improvements in a watershed, several conditions have to be met. The data were examined to find how many watersheds satisfy some simple screening rules. The objective is to see how many coastal watersheds might pass a conservative test of their potential as sites for PS/NPS trading. One such test is to see in how many coastal watersheds both point and agricultural nonpoint loadings contribute "significantly" to total pollutant loadings. Simply put, if either the agricultural share of total pollutant flows or the point share of total loadings is small, then trading point and nonpoint reductions is unlikely to be feasible or to contribute much to water quality improvement.

Table 3 reports the number of coastal watersheds which satisfy some simple criteria of this sort. The data were examined to identify coastal watersheds where both point and agricultural nonpoint sources of pollutant loadings exceeded 20, 25, or 30 percent of total loadings each. We chose 20 percent of loadings by both point and nonpoint sources as a minimum criterion to ensure that there is enough potential for changes in loadings from point and nonpoint sources to affect overall water quality. Thirty percent of loadings from each class of pollutant source was the most conservative criterion; only a handful of water systems in the database had more than 35 percent of loadings coming from both pollutant sources.

Looking at the least strict criterion first, if we require that point and agricultural nonpoint sources both must account for al least 20 percent of total loadings, then out of 350 coastal watersheds 32 meet this requirement for nitrogen, 37 for phosphorus, 17 for sediment, and 32 for BOD5. If the requirement is that both point and nonpoint sources account for 30 percent each of total loadings, the numbers are considerably smaller: 16 watersheds meet this criterion for nitrogen and phosphorus, 13 for BOD5, and 8 for sediment. (See also Table 4 and Figures 1 - 3.)

Table 3

Coastal Watersheds Meeting Screening Criteria
For Potential Point-Nonpoint Trading
(Number of Watersheds)

	Nitrogen	Phosphorus	Sediment	BOD5
Cataloging Units with both point and agricultural sources supplying at least 20 percent of pollutant loadings	32	37	17	34
Cataloging Units with both point and agricultural sources supplying at least 25 percent of pollutant loadings	25	23	13	22
Cataloging Units with both point and agricultural sources supplying at least 30 percent of pollutant loadings	16	16	8	13

Data based on cataloging unit-level estimates of pollutant loadings from NCPDI. Agricultural loadings are loadings from cropland and pastureland. Point source loadings are loadings from wastewater treatment plants, powerplants, and industrial sources. Total number of coastal cataloging units assessed: 350.

Table 4
Distribution of Cataloging Units Meeting 30 Percent
Point, Nonpoint Pollutant Loading Shares
By Pollutant Category and Region

Region	Nitrogen Only	Phosphorus Only	Sediment Only	BOD5 Only	Multiple Pollutants
East (19)	4	6	3	3	BOD5, P:1 N,P:1 P, s: 1
Gulf Coast (13)	5	2	0	2	BOD5, N:4
west (11)	1	4	3	2	BOD5, N,P,S:1

Data based on cataloging unit-level estimates of pollutant loadings from NCPDI. Agricultural loadings are loadings from cropland and pastureland. Point source loadings are loadings from wastewater treatment plants, powerplants, and industrial sources. Total number of coastal cataloging units assessed: 350.

As revealed by Table 3 and 4 and Figures 1 - 3, it would appear that our initial screening does not show any particularly widespread potential for PS/NPS trading possibilities. Only at most 10 percent of the total number of coastal watersheds examined meet the most optimistic criteria we have established. Our analysis suggests PS/NPS trading might work in a few locations, but is unlikely to bring about NPS control in coastal regions nationally by itself.

Our results mirror those of a recent EPA study of all water bodies. The study examined information on waterbody impairments in 37 states, the District of Columbia, and two U.S. possessions for a count of the number of rivers, lakes, and estuary segments which a) do not meet designated uses from nutrient enrichment, and b) contain industrial point sources, municipal point sources, or both along with nonpoint sources (agriculture, silviculture, construction, resource extraction, land disposal, or hydro/habitat modification).

Their study showed that out of about 10,000 water bodies in their database not fully supporting designated uses due to nutrient loads, about 6 percent (618) impaired rivers, lakes, or estuaries could be considered for nutrient load trading.[20]

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[20]Personal communication and memorandum supplied to the authors by Chris Faulkner, US EPA, Office of Water, Assessments and Protection Division.

Figure 1:

Eastern Cataloging Units Meeting Criteria

For Potential Point-Nonpoint Trading

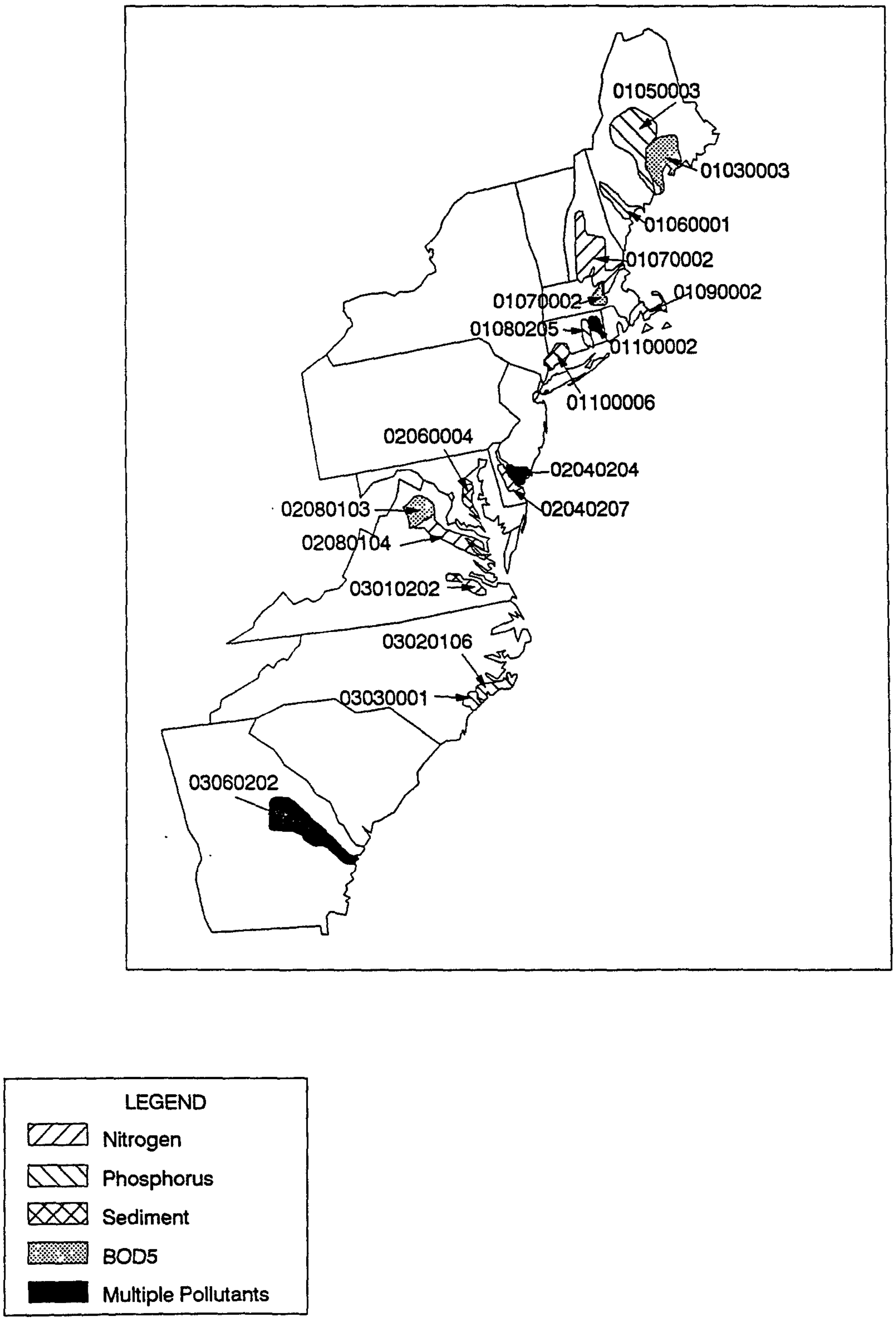


Figure 2:

Southern Cataloging Units Meeting Criteria
For Potential Point-Nonpoint Trading

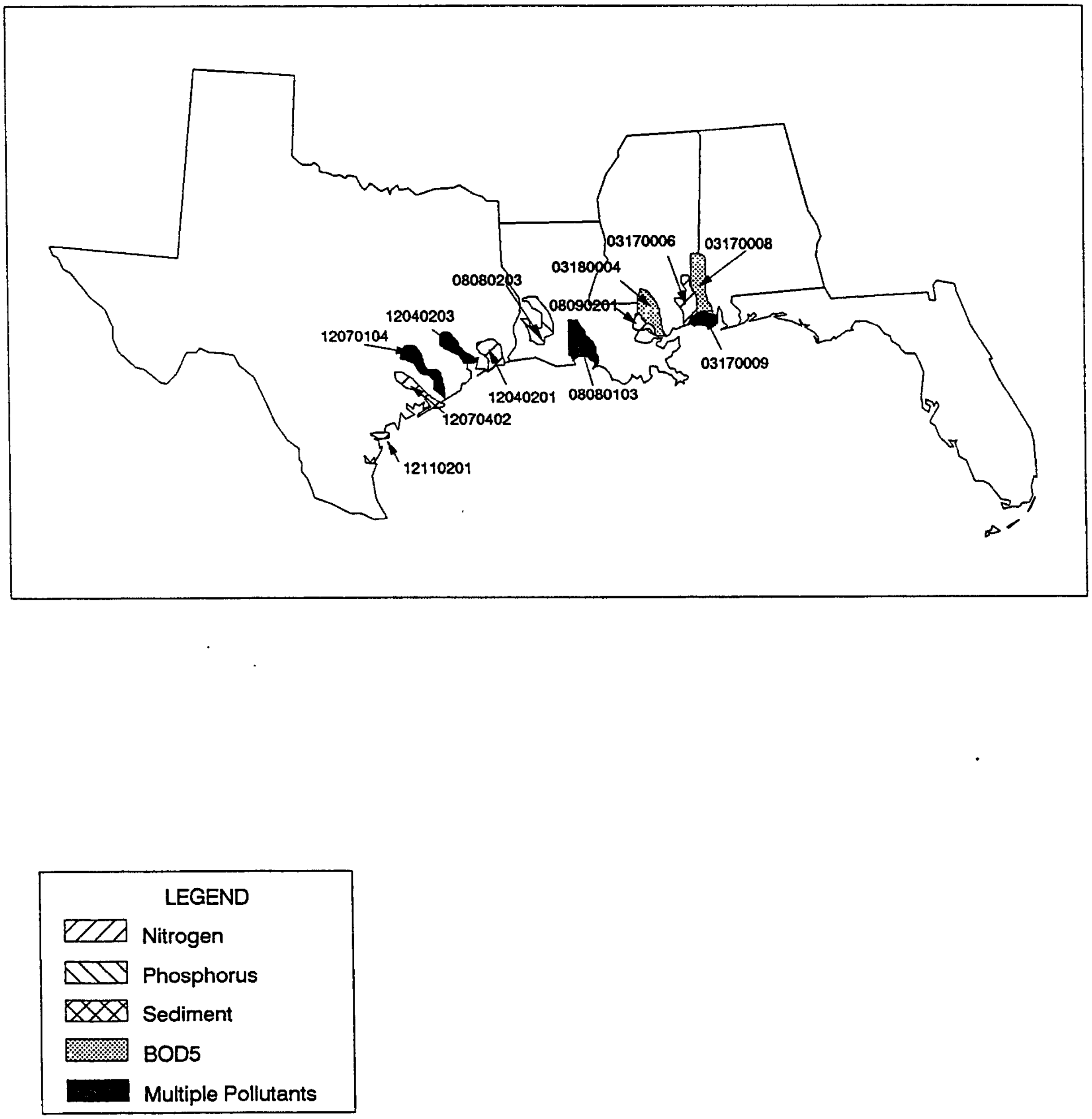
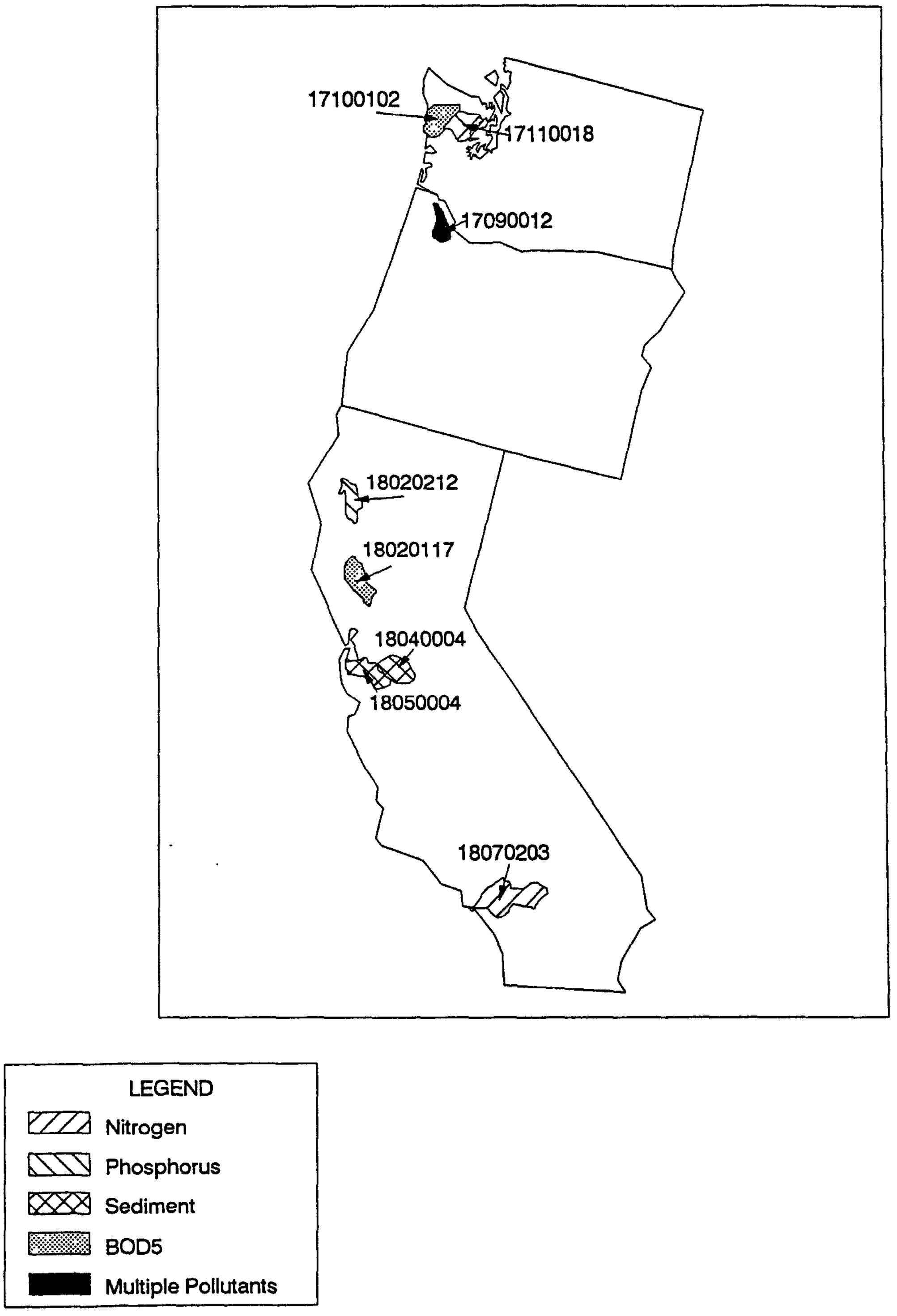


Figure 3:

Western Cataloging Units Meeting Criteria

For Potential Point-Nonpoint Trading



V. Conclusions

The purpose of this paper was to offer a national perspective on the feasibility of the PS/NPS trading option for coastal water quality management. We have used a conceptual model and a screening analysis to explore this question analytically and empirically. Our analysis indicates that it is unlikely that PS/NPS trading can form the basis of a national coastal water quality management program. It may, however, be a viable and attractive policy instrument in a few locations.

Our conceptual model showed that more attention needs to be paid to the issue of setting the appropriate trading ratio. Although it is often recommended that the ratio be set above one to allow for the randomness in nonpoint source loadings (as it has been for the Dillon Reservoir program), the greater difficulty in monitoring NPS loadings may call for a smaller ratio. Setting trading ratios above one may not result in cost effective outcomes.

PS/NPS trading does introduce some new problems because it approaches water quality management by attempting to create a market for pollution rights. A key element in properly coordinating such a market is having an appropriate number of participants. The number must be large enough so that the potential loadings reduction is of use to a PS and can be measured with some accuracy, yet small enough so that bargaining costs are not prohibitive. On the other hand, many of the obstacles to implementing PS/NPS trading programs apply to other approaches to NPS control also. The problems of implementing trading programs do not seem to be much greater than those associated with other approaches to controlling nonpoint source pollution.

We conducted a conservative screening analysis based on the consideration that the proportions of loadings contributed to a watershed by PSs and NPSs at least be consistent with the possibility of trading. Approximately ten percent of the coastal watersheds were reported by states as having “significant” contributions (twenty percent) of sediment, nutrient, or BOD5 loadings from both point and agricultural nonpoint sources. This nationwide screening analysis cannot locate “good” candidates for trading programs but does allow us to rule out many coastal watersheds, so researchers and planners can better focus their water quality efforts.

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APPENDIX A

Sources of Coastal Pollutant Loadings And Related Erosion Data

Unit	Shares of total --- NITROGEN --- Agric.	Point	Shares of total --PHOSPHORUS -- Agric.	Point	Shares of total -- SEDIMENT -- Agric.	Point	Shares of total --- BOD5 --- Agric.	Point	Cropland Erosion Rate	Cropland's Share of All Erosion	Percent Agric. Land Needing Cons. Treatment	Region
0003	94.4	0.2	78.9	12.9	7.8	83.4	72.2	12.9	0.00	0.0	0.0	East
0005	0.1	0.0	0.0	0.9	0.0	34.6	0.0	0.4	0.00	7.3	55.7	East
0003	57.6	0.1	62.7	2.6	25.0	50.0	38.7	38.1	2.18	56.3	51.4	East
0002	51.9	23.9	3.9	90.7	0.1	98.6	0.7	96.6	2.11	16.3	18.4	East
0001	95.4	0.0	92.2	2.8	26.4	60.4	88.8	3.0	0.00	19.4	0.0	East
0002	76.4	0.0	76.0	0.7	53.1	29.0	75.8	0.7	0.00	13.0	53.8	East
0003	37.2	0.2	31.0	2.3	35.2	40.6	21.6	4.5	0.00	14.5	38.1	East
0001	93.6	0.0	90.6	1.2	50.1	40.5	90.0	1.2	0.71	58.7	21.7	East
0002	15.6	43.9	2.7	91.1	0.3	98.5	0.1	95.5	0.00	0.9	14.7	East
0003	87.0	0.6	67.8	17.3	14.2	70.2	46.3	18.8	4.10	61.3	59.1	East
0002	96.6	0.1	57.5	34.3	2.3	94.1	24.2	68.8	8.22	32.4	23.7	East
0004	0.0	58.1	0.0	92.8	0.0	97.9	0.0	87.5	0.00	1.4	0.0	East
0005	86.0	1.1	72.4	23.6	18.7	78.1	49.7	38.8	0.29	41.2	8.7	East
0205	79.8	0.0	65.4	1.1	40.6	36.7	64.0	1.3	6.49	79.6	37.1	East
0207	91.7	0.3	72.1	17.1	5.9	86.8	70.0	13.1	2.89	68.8	6.1	East
0001	77.8	0.1	65.9	8.6	22.8	70.9	52.8	18.1	5.34	27.8	12.5	East
0002	69.8	0.3	31.7	42.7	2.1	91.3	38.8	10.0	2.39	2.5	65.0	East
0003	64.1	1.3	26.5	54.9	4.2	89.7	8.9	77.6	17.15	73.0	12.5	East
0004	94.5	0.0	80.8	9.2	19.8	68.8	70.8	12.1	8.70	95.4	56.9	East
0005	72.1	0.0	56.7	4.9	14.9	66.6	55.6	3.0	1.85	42.3	70.4	East
0001	84.6	0.0	80.4	13.0	14.8	80.8	79.3	9.9	15.21	74.9	33.3	East
0002	79.4	0.4	85.6	9.6	33.8	63.5	51.7	38.2	0.12	0.8	18.2	East
0003	6.9	8.4	2.9	1.8	1.6	4.1	0.4	2.4	5.10	19.7	33.3	East
0004	46.0	12.1	29.5	2.6	7.2	9.9	10.2	0.4	3.48	63.3	38.3	East
0005	40.9	0.8	19.2	9.8	4.4	7.2	9.0	2.8	7.01	5.5	21.1	East
0006	52.7	41.0	71.0	2.2	18.4	15.4	11.4	77.0	2.59	12.6	33.3	East
0007	88.0	0.3	80.5	5.3	67.2	17.2	0.2	19.3	0.00	0.0	0.0	East
0003	26.2	56.3	16.9	35.0	15.0	24.7	0.2	49.3	9.72	74.2	50.7	East
0004	33.2	63.2	71.6	25.5	60.1	36.1	2.9	81.1	6.35	86.2	64.5	East
0005	0.0	3.9	0.0	0.1	0.0	0.4	0.0	0.0	8.42	73.8	42.7	East
0006	19.8	0.1	27.1	0.7	17.1	1.8	1.3	1.1	11.63	50.0	49.6	East
0007	39.4	0.6	79.9	4.1	53.4	28.8	2.9	14.1	9.75	68.8	46.6	East
0008	33.1	0.0	38.1	0.0	100.0	0.0	18.8	0.0	13.81	31.3	30.0	East
0101	0.3	42.3	25.6	5.8	14.2	22.0	0.1	6.1	7.62	3.3	60.0	East
0102	13.1	1.4	19.0	0.1	11.0	0.7	1.3	1.3	0.00	0.0	0.0	East
0103	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0	0.00	2.4	2.4	East
0104	0.5	1.0	42.8	3.5	26.7	18.0	0.0	3.6	7.78	81.0	58.3	East
0105	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	7.70	92.1	48.1	East
0201	61.0	0.0	30.1	1.6	5.2	2.2	15.0	0.3	4.27	6.3	64.3	East
0202	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0	7.92	89.7	61.4	East
0102	0.0	29.9	0.0	30.2	0.0	53.2	0.0	35.1	0.00	0.0	0.0	East
0104	30.8	0.1	77.8	0.5	65.5	4.4	3.3	1.5	25.83	6.3	70.0	East
0105	99.8	0.0	100.0	0.0	100.0	0.0	100.0	0.0	7.98	69.6	49.7	East
0201	63.4	0.0	91.2	0.0	85.0	0.0	36.2	0.0	5.80	77.3	46.2	East
0202	28.6	4.5	94.2	0.0	90.6	0.1	5.8	29.7	7.30	86.3	48.6	East
0203	18.7	40.2	15.4	79.6	1.7	97.4	0.6	90.1	7.11	68.3	58.8	East

: Agricultural sources include harvested cropland, non-harvested cropland, pastureland, and rangeland. Point sources include
ewater treatment plants, powerplants, and industrial sources. Pollutant loadings used to estimate shares by point and nonpoir
ces from the NCPDI. Erosion rates, cropland's share of all erosion, and percent of agricultural lands needing conservation treat
from the NRI.

nit	Shares of total --- NITROGEN --- Agric.	Point	Shares of total --PHOSPHORUS -- Agric.	Point	Shares of total -- SEDIMENT -- Agric.	Point	Shares of total --- BOD5 --- Agric.	Point	Cropland Erosion Rate	Cropland's Share of All Erosion	Percent Agric. Land Needing Cons. Treatment	Region
4	18.5	13.4	30.4	63.1	6.3	91.9	0.9	85.4	0.00	0.0	0.0	East
5	97.8	0.0	98.9	0.0	99.7	0.0	98.2	0.0	7.22	67.6	36.6	East
6	88.6	0.0	92.4	0.0	87.5	0.0	60.8	0.0	7.99	97.7	59.5	East
7	39.1	0.6	70.6	6.2	41.3	35.0	6.8	12.7	2.52	96.0	30.8	East
1	0.2	65.3	12.9	55.0	17.4	10.0	0.0	96.5	6.20	76.6	39.0	East
2	3.7	0.5	98.2	0.2	95.6	1.8	0.3	6.6	4.48	26.8	18.9	East
6	4.6	0.7	64.7	0.9	46.2	10.5	5.1	4.8	9.52	16.6	68.8	East
1	4.0	0.7	88.8	1.7	72.9	14.0	0.3	10.1	0.00	0.0	0.0	East
2	23.1	0.1	74.7	1.0	46.0	10.1	40.1	1.5	3.85	95.6	56.7	East
3	67.7	1.7	91.9	3.0	73.8	19.3	10.1	35.3	4.66	66.0	46.1	East
4	42.6	40.8	13.7	83.7	1.2	98.5	0.5	96.1	16.18	97.6	83.7	East
5	61.6	0.2	82.2	1.2	63.9	9.5	30.4	8.2	2.11	99.7	31.4	East
6	1.4	69.0	66.2	0.8	52.3	4.2	0.3	5.6	14.46	83.4	74.9	East
7	6.6	2.7	30.5	17.9	11.5	52.5	2.2	22.3	2.23	98.7	44.2	East
8	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0	1.32	98.1	9.0	East
9	63.6	0.0	69.8	0.0	100.0	0.0	22.9	0.0	1.74	96.7	13.3	East
10	0.9	10.8	25.9	4.6	14.5	21.2	0.0	3.5	1.98	94.3	31.1	East
08	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.22	69.8	46.5	East
10	15.9	37.3	6.5	66.4	15.9	0.0	0.3	87.2	5.02	41.8	43.2	East
11	12.2	35.6	9.1	34.2	22.7	5.2	0.5	69.1	4.57	69.8	39.6	East
01	3.8	44.4	4.7	62.2	15.1	0.9	0.1	87.7	0.00	0.0	0.0	East
02	3.1	14.3	3.3	15.3	7.4	0.0	0.1	34.7	3.36	91.3	61.4	East
03	87.8	0.2	89.6	4.5	99.6	0.0	32.5	59.2	10.30	50.4	60.7	East
04	57.6	21.5	38.4	33.8	6.8	71.6	18.4	39.7	2.67	70.7	30.0	East
05	9.7	6.3	7.9	29.2	9.0	11.8	1.0	61.9	4.86	74.8	48.8	East
06	28.6	1.8	31.3	11.4	17.1	4.7	4.9	64.9	5.41	41.6	43.2	East
07	9.4	88.0	97.3	1.2	93.6	6.3	72.6	25.5	3.66	62.1	87.5	East
08	55.2	1.7	34.4	5.7	14.3	0.0	13.1	17.2	1.30	38.9	0.0	East
09	24.5	0.2	43.2	5.5	43.9	21.2	24.4	21.8	1.59	91.2	55.8	East
10	40.0	3.6	43.6	16.4	23.4	1.7	11.3	57.7	3.09	92.2	52.8	East
205	24.8	1.0	15.2	1.1	7.3	1.7	4.8	1.8	0.88	44.3	14.1	East
206	26.1	38.3	13.6	5.2	6.9	12.2	3.5	1.4	5.20	84.6	46.7	East
207	38.3	17.4	5.9	78.5	24.0	2.2	1.0	10.1	8.67	67.3	46.4	East
208	9.6	90.4	99.8	0.2	100.0	0.0	0.0	100.0	2.80	81.0	40.8	East
107	99.9	0.0	99.9	0.1	99.1	0.9	71.9	26.9	3.19	98.0	40.4	East
201	97.9	0.0	91.7	0.1	84.0	0.6	68.1	0.4	6.92	87.6	81.2	East
202	64.5	30.2	68.0	0.1	51.0	1.2	32.4	0.3	3.95	78.9	45.2	East
203	92.7	0.2	92.2	0.3	90.7	0.0	29.6	13.2	2.48	44.7	17.8	East
204	86.3	0.0	59.5	0.1	39.2	0.1	26.1	0.0	6.69	84.9	55.7	East
205	99.4	0.1	98.4	0.5	97.4	1.4	0.5	71.2	1.37	98.8	45.3	East
103	39.3	6.2	9.9	0.2	7.3	0.9	1.1	0.0	3.74	97.2	22.9	East
104	99.2	0.0	97.8	0.2	79.8	2.9	24.5	32.0	2.90	99.9	59.9	East
105	94.0	0.0	79.5	0.0	71.8	0.0	35.3	0.3	1.75	100.0	55.0	East
106	91.5	1.4	84.9	11.3	51.0	46.3	13.2	64.3	2.04	99.2	0.0	East
202	88.1	0.2	90.3	0.7	90.2	0.0	1.5	24.4	4.08	99.6	55.5	East
204	91.1	0.6	84.7	3.0	75.1	10.7	28.4	19.7	2.15	99.8	34.0	East
001	86.1	2.1	70.0	22.3	52.8	40.7	4.5	64.8	4.78	99.3	75.0	East
005	0.0	31.1	0.0	55.6	0.0	46.2	0.0	20.5	3.41	96.1	17.9	East
006	0.1	75.4	3.7	0.0	3.0	0.0	0.0	0.0	3.18	98.8	57.6	East
007	10.4	48.5	8.1	28.1	6.1	1.8	1.3	22.8	2.78	99.6	55.6	East
201	99.9	0.1	100.0	0.0	100.0	0.0	100.0	0.0	4.82	90.7	34.5	East
202	14.7	30.5	10.4	7.2	3.8	8.1	2.8	7.6	4.02	89.4	44.5	East

: Agricultural sources include harvested cropland, non-harvested cropland, pastureland, and rangeland. Point sources includ
water treatment plants, powerplants, and industrial sources. Pollutant loadings used to estimate shares by point and nonpoi
es from the NCPDI. Erosion rates, cropland's share of all erosion, and percent of agricultural lands needing conservation trea
from the NRI.

Unit	Shares of total --- NITROGEN --- Agric.	Point	Shares of total --PHOSPHORUS -- Agric.	Point	Shares of total -- SEDIMENT -- Agric.	Point	Shares of total --- BOD5 --- Agric.	Point	Cropland Erosion Rate	Cropland's Share of All Erosion	Percent Agric. Land Needing Cons. Treatment	Region
203	7.6	30.3	6.4	3.1	3.5	0.1	1.3	7.5	0.00	0.0	0.0	East
204	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.26	93.1	49.7	East
205	1.9	0.8	1.7	0.2	1.2	0.4	0.1	0.8	2.86	97.9	21.8	East
206	11.4	45.4	17.0	0.2	10.8	0.1	2.6	0.1	3.84	99.6	41.0	East
207	0.0	39.2	0.0	0.0	0.0	0.0	0.0	0.0	2.82	96.8	100.0	East
111	62.6	21.3	50.2	6.0	14.3	28.5	24.7	9.4	5.64	85.7	44.6	East
112	35.9	0.4	20.2	0.7	9.5	0.2	6.4	6.7	2.98	88.4	32.3	East
201	47.7	0.2	27.2	6.6	15.6	15.0	7.7	3.6	2.31	91.1	5.9	East
202	0.0	48.4	0.0	38.4	0.0	75.7	0.0	25.4	3.74	69.2	68.2	East
205	55.7	1.2	33.7	15.9	17.1	49.0	5.6	16.9	2.22	97.9	7.1	East
206	21.5	4.3	10.7	10.3	6.0	20.9	1.4	10.5	2.46	98.8	10.2	East
207	14.1	63.1	27.1	13.1	15.0	45.8	2.0	30.3	2.17	91.2	18.3	East
208	74.3	0.5	46.1	3.8	37.2	6.8	9.8	3.5	1.91	89.9	12.4	East
109	0.0	10.7	0.0	35.1	0.0	5.7	0.0	43.6	3.96	96.8	28.6	East
202	88.7	1.0	32.3	50.3	34.2	32.2	18.7	34.3	3.62	50.0	33.7	East
203	0.0	58.0	0.0	15.7	0.0	33.1	0.0	9.0	4.74	98.0	64.0	East
204	59.2	13.6	7.9	12.1	6.7	21.5	0.0	23.1	1.92	0.6	0.0	East
106	80.7	0.3	32.0	8.8	14.0	35.1	10.3	8.6	3.33	87.8	21.1	East
201	1.8	23.3	1.2	31.7	1.3	12.0	0.1	18.1	4.50	94.9	46.0	East
203	25.5	70.9	73.8	4.3	68.7	6.9	4.8	74.6	0.00	0.0	0.0	East
204	0.0	52.3	0.0	83.0	0.0	9.8	0.0	66.7	1.23	10.0	76.1	East
205	76.3	4.0	61.2	3.9	21.6	6.3	35.9	2.3	0.00	0.0	100.0	East
101	0.0	0.0	0.0	1.8	0.0	1.2	0.0	0.1	0.31	16.6	53.6	East
102	92.6	0.1	92.6	0.7	91.2	1.9	83.3	5.5	1.61	14.4	66.9	East
103	82.9	0.3	62.7	14.9	22.9	40.8	45.1	9.3	0.93	65.8	32.2	East
201	80.0	0.2	82.2	0.1	64.7	0.8	63.6	0.1	0.00	0.0	11.8	East
202	0.0	13.6	0.0	3.8	0.0	1.3	0.0	19.2	0.00	2.4	20.0	East
203	14.4	1.1	4.2	8.7	0.3	27.6	2.0	4.8	0.00	0.7	97.0	East
101	0.0	15.1	0.0	28.8	0.0	54.7	0.0	8.7	0.66	64.3	68.9	East
102	97.9	0.0	95.0	0.2	64.1	3.5	89.3	0.3	0.34	0.1	93.7	East
103	80.1	0.0	56.9	0.6	20.0	6.9	36.5	0.9	0.00	0.0	90.9	East
201	0.0	0.4	0.0	1.2	0.0	2.1	0.0	0.2	0.00	0.0	0.0	East
202	37.4	0.0	34.1	0.0	25.2	0.0	15.4	0.0	0.25	63.5	50.6	Gulf Coast
202	93.4	0.0	86.2	0.1	42.4	0.5	63.6	2.2	0.25	63.5	50.6	East
203	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.0	0.0	Gulf Coast
204	0.7	0.0	0.7	0.0	0.9	0.0	0.3	0.0	0.96	72.0	66.7	Gulf Coast
205	1.0	0.0	0.7	0.0	0.6	0.0	0.5	0.0	1.14	75.4	61.4	Gulf Coast
101	2.9	0.0	1.6	0.0	0.9	0.0	0.4	0.0	0.61	0.5	37.5	Gulf Coast
102	0.1	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.00	2.6	39.8	Gulf Coast
103	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.00	3.9	62.5	Gulf Coast
201	1.9	0.0	6.4	0.0	0.6	0.0	3.1	0.0	1.35	58.7	11.5	Gulf Coast
202	25.2	0.0	9.2	0.0	1.7	0.0	4.5	0.0	1.16	70.1	52.7	Gulf Coast
203	34.5	0.0	13.8	0.0	6.4	0.0	6.1	0.0	0.68	15.0	49.1	Gulf Coast
204	0.6	0.0	0.6	0.0	0.9	0.0	0.1	0.0	2.10	0.3	26.9	Gulf Coast
205	1.0	0.0	0.4	0.0	0.3	0.0	0.1	0.0	0.00	12.1	92.6	Gulf Coast
206	0.2	0.0	0.2	0.0	0.3	0.0	0.0	0.0	0.55	37.2	60.7	Gulf Coast
207	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	1.13	33.5	64.5	Gulf Coast
208	7.7	0.0	1.0	0.0	0.8	0.0	0.4	0.0	1.97	59.4	80.6	Gulf Coast
208	0.0	1.2	0.0	42.0	0.0	57.2	0.0	9.4	1.97	59.4	80.6	East
101	1.0	0.0	0.1	0.0	0.2	0.0	0.1	0.0	1.89	3.2	80.4	Gulf Coast
101	0.0	69.2	0.0	0.0	0.0	0.1	0.0	18.7	1.89	3.2	80.4	East
102	16.0	0.0	2.9	0.0	0.5	0.0	2.0	0.0	0.51	62.4	31.4	Gulf Coast

Agricultural sources include harvested cropland, non-harvested cropland, pastureland, and rangeland. Point sources includec water treatment plants, powerplants, and industrial sources. Pollutant loadings used to estimate shares by point and nonpoint es from the NCPDI. Erosion rates, cropland's share of all erosion, and percent of agricultural lands needing conservation treat- from the NRI.

Init	Shares of total --- NITROGEN --- Agric.	Point	Shares of total --PHOSPHORUS -- Agric.	Point	Shares of total -- SEDIMENT -- Agric.	Point	Shares of total --- BOD5 --- Agric.	Point	Cropland Erosion Rate	Cropland's Share of All Erosion	Percent Agric. Land Needing Cons. Treatment	Region
03	84.2	0.0	82.1	0.0	38.3	0.0	74.7	0.0	6.10	88.9	58.8	Gulf Coast
01	84.5	0.0	35.1	0.0	5.2	0.0	23.6	0.0	1.44	55.7	21.9	Gulf Coast
02	99.0	0.0	95.6	0.0	60.1	0.0	91.2	0.0	4.84	97.8	52.6	Gulf Coast
03	98.1	0.0	91.1	0.0	43.1	0.0	83.4	0.0	4.78	98.4	61.3	Gulf Coast
05	89.7	0.0	30.8	0.0	13.4	0.0	27.6	0.0	0.92	93.7	18.9	Gulf Coast
06	79.7	0.0	57.6	0.0	11.3	0.0	42.6	0.0	2.42	95.5	26.9	Gulf Coast
01	74.0	0.0	47.9	0.0	6.6	0.0	31.5	0.0	3.31	87.5	27.7	Gulf Coast
03	97.6	0.0	75.2	0.0	17.8	0.0	58.6	0.0	3.23	54.8	29.1	Gulf Coast
04	93.8	0.0	43.8	0.0	8.0	0.0	26.0	0.0	6.84	86.9	50.0	Gulf Coast
11	97.0	0.0	73.7	0.0	16.5	0.0	56.6	0.0	0.99	46.5	0.0	Gulf Coast
12	99.0	0.0	89.8	0.0	37.3	0.0	80.2	0.0	4.14	80.3	54.3	Gulf Coast
13	1.5	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.00	0.0	0.0	Gulf Coast
14	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.0	0.0	Gulf Coast
01	44.8	0.0	48.7	0.0	7.0	0.0	32.6	0.0	0.00	0.0	0.0	Gulf Coast
02	45.2	0.0	3.3	0.0	0.2	0.0	1.6	0.0	2.92	97.5	34.0	Gulf Coast
03	92.6	0.0	56.1	3.2	8.1	13.5	39.2	2.5	5.30	37.4	25.7	Gulf Coast
04	74.2	0.0	30.9	0.0	11.9	0.0	27.2	0.0	5.81	97.3	55.4	Gulf Coast
05	41.7	0.0	6.5	0.0	0.5	0.0	3.2	0.0	0.00	0.0	0.0	Gulf Coast
06	78.0	0.0	31.2	3.5	8.8	61.2	30.3	2.8	4.51	71.9	57.1	Gulf Coast
07	18.0	0.0	3.7	3.3	0.5	55.7	3.1	2.9	3.93	0.6	100.0	Gulf Coast
02	99.7	0.0	97.2	0.0	70.5	0.0	94.2	0.0	4.46	17.3	41.4	Gulf Coast
03	98.9	0.0	90.6	0.0	40.1	0.0	81.8	0.0	3.81	93.6	42.6	Gulf Coast
04	99.0	0.0	88.9	0.0	35.2	0.0	78.6	0.0	4.33	73.0	23.0	Gulf Coast
05	80.2	0.0	44.6	0.2	24.0	0.0	41.5	0.2	4.95	87.7	61.4	Gulf Coast
04	33.1	0.5	7.7	2.5	7.4	0.0	6.1	22.3	5.16	57.1	58.4	Gulf Coast
03	41.7	1.1	10.7	66.9	1.5	83.1	7.2	72.9	4.72	13.7	46.4	Gulf Coast
04	27.6	3.9	2.9	48.3	0.1	88.8	1.2	71.6	1.10	79.6	0.0	Gulf Coast
05	86.3	0.0	47.0	5.8	10.2	39.3	40.4	5.9	6.70	99.0	68.6	Gulf Coast
06	77.3	0.4	35.6	44.2	1.5	88.1	28.6	40.2	5.91	81.1	62.0	Gulf Coast
07	95.3	0.0	98.1	0.0	98.0	0.0	98.1	0.0	9.36	71.5	51.1	Gulf Coast
08	94.5	0.5	80.9	9.8	24.9	40.5	48.3	41.0	6.88	85.9	40.1	Gulf Coast
009	69.6	0.2	40.5	30.8	4.1	68.1	30.8	31.8	5.67	52.9	60.0	Gulf Coast
004	94.0	0.3	67.0	18.9	11.0	59.1	45.5	35.3	10.02	35.3	57.8	Gulf Coast
005	94.4	0.0	70.3	0.0	14.8	0.0	52.9	0.0	6.11	66.8	50.2	Gulf Coast
100	2.8	93.0	0.3	98.5	0.0	99.9	0.1	99.5	0.00	0.0	100.0	Gulf Coast
201	90.5	0.1	66.8	22.7	4.7	91.1	61.7	24.8	5.58	57.4	47.7	Gulf Coast
202	64.9	0.0	53.7	5.6	10.1	49.8	46.9	5.2	6.18	39.5	51.3	Gulf Coast
203	88.0	0.1	69.3	7.1	14.5	43.6	58.9	8.5	4.85	54.9	49.5	Gulf Coast
204	35.8	4.6	4.5	51.6	0.2	73.2	1.3	70.4	4.38	96.5	32.9	Gulf Coast
205	89.5	0.1	71.6	12.7	14.1	64.6	66.4	11.9	5.35	41.5	62.3	Gulf Coast
300	84.4	0.3	32.7	25.0	2.0	59.9	21.8	16.9	3.99	99.5	28.9	Gulf Coast
101	74.7	0.6	27.5	46.8	1.1	87.0	19.9	44.8	4.68	98.5	74.2	Gulf Coast
102	89.6	0.3	73.1	15.6	12.5	71.7	67.2	16.2	4.41	99.1	60.7	Gulf Coas
103	92.6	0.6	37.8	51.4	1.5	92.7	34.5	45.5	3.62	95.8	78.5	Gulf Coas
201	97.3	0.1	86.7	7.8	30.5	48.7	82.2	8.4	5.20	98.8	58.1	Gulf Coas
202	98.3	0.0	86.9	6.0	24.8	45.6	80.1	5.7	3.38	99.4	84.2	Gulf Coas
203	97.0	0.0	91.7	1.9	34.4	33.9	86.2	1.6	3.90	73.9	74.1	Gulf Coas
205	76.8	0.0	81.6	1.4	45.8	5.1	76.9	1.5	2.96	81.5	63.0	Gulf Coas
206	56.7	5.9	8.2	68.1	0.3	87.1	6.3	56.7	3.65	96.3	67.2	Gulf Coas
100	0.0	80.3	0.0	98.9	0.0	99.9	0.0	98.5	0.00	0.0	0.0	Gulf Coas
201	50.7	0.2	30.7	34.4	1.6	88.0	27.1	30.0	8.79	83.1	55.0	Gulf Coas
203	0.0	4.9	0.0	70.2	0.0	88.3	0.0	54.5	0.00	0.0	0.0	Gulf Coas

: Agricultural sources include harvested cropland, non-harvested cropland, pastureland, and rangeland. Point sources includ
ewater treatment plants, powerplants, and industrial sources. Pollutant loadings used to estimate shares by point and nonpoi
ces from the NCPDI. Erosion rates, cropland's share of all erosion, and percent of agricultural lands needing conservation trea
from the NRI.

Unit	Shares of total --- NITROGEN --- Agric.	Point	Shares of total --PHOSPHORUS -- Agric.	Point	Shares of total -- SEDIMENT -- Agric.	Point	Shares of total --- BOD5 --- Agric.	Point	Cropland Erosion Rate	Cropland's Share of All Erosion	Percent Agric. Land Needing Cons. Treatment	Region
301	6.7	20.6	0.8	65.2	0.0	97.3	0.2	77.0	2.69	98.1	59.0	Gulf Coast
302	20.9	36.7	1.5	79.7	0.0	94.0	0.5	87.7	4.18	99.7	24.4	Gulf Coast
005	9.2	0.5	8.4	17.2	1.6	60.5	4.0	55.0	0.67	4.3	50.7	Gulf Coast
003	35.0	7.6	0.9	77.8	0.0	92.7	0.6	70.6	0.00	0.0	2.5	Gulf Coast
007	93.6	0.0	71.1	6.8	8.6	51.3	58.7	2.0	2.31	76.3	54.5	Gulf Coast
202	13.9	0.0	38.3	1.9	10.0	30.6	33.5	1.8	20.82	0.9	57.0	Gulf Coast
203	76.6	0.0	76.4	6.9	11.4	60.0	69.3	2.4	0.92	61.0	18.2	Gulf Coast
101	36.8	1.0	2.0	65.7	0.0	89.1	2.2	21.1	0.00	0.7	73.8	Gulf Coast
102	89.3	0.5	9.5	74.5	0.2	94.8	13.1	37.8	1.97	66.7	45.3	Gulf Coast
103	13.0	0.5	1.2	50.7	0.0	82.0	0.9	24.9	0.00	0.0	49.0	Gulf Coast
104	78.0	4.0	3.6	86.4	0.1	97.0	5.1	63.6	1.31	92.1	53.5	Gulf Coast
201	91.0	0.4	37.6	44.1	1.6	86.7	26.9	44.6	0.99	92.3	17.2	Gulf Coast
202	97.6	0.0	79.7	8.4	10.5	65.9	70.7	6.4	2.02	99.8	17.6	Gulf Coast
203	97.4	0.8	57.6	37.1	1.6	96.1	52.1	37.5	0.77	99.5	10.5	Gulf Coast
204	66.5	1.1	22.6	63.8	0.6	94.1	25.5	42.8	0.51	73.9	15.9	Gulf Coast
205	70.4	0.2	53.8	29.8	3.6	85.1	51.5	20.5	0.97	87.7	5.4	Gulf Coast
104	88.0	1.6	45.8	46.1	1.5	94.8	42.7	41.9	1.62	23.9	50.4	Gulf Coast
302	98.2	0.0	88.6	5.8	16.9	68.2	85.6	3.1	1.29	78.4	61.2	Gulf Coast
401	89.2	0.0	86.3	6.2	15.7	69.2	81.4	5.4	1.41	34.3	35.5	Gulf Coast
402	92.1	0.3	56.7	32.6	3.4	87.4	56.9	20.3	0.76	55.3	75.8	Gulf Coast
101	96.8	0.0	75.0	11.7	7.7	73.0	67.9	6.6	1.96	60.5	54.4	Gulf Coast
102	98.0	0.0	92.9	3.0	28.8	56.8	90.6	1.9	2.49	91.5	23.2	Gulf Coast
204	69.7	1.2	19.1	55.7	0.5	90.1	14.7	44.4	4.54	11.4	26.5	Gulf Coast
303	3.3	0.0	2.0	0.0	0.0	0.0	1.3	0.0	4.87	61.6	51.7	Gulf Coast
401	93.0	1.0	88.5	4.3	23.1	56.2	84.7	2.6	1.81	72.4	56.3	Gulf Coast
402	95.7	0.1	83.3	7.8	12.2	69.1	73.5	10.0	2.02	89.5	42.4	Gulf Coast
403	2.1	0.1	7.1	1.7	2.0	24.7	3.5	45.1	0.00	0.0	0.0	Gulf Coast
404	94.0	0.0	70.7	4.1	10.5	36.7	55.8	1.7	0.00	0.0	0.0	Gulf Coast
405	69.2	0.3	21.4	26.3	0.9	67.5	14.0	12.9	0.44	3.2	86.8	Gulf Coast
406	67.9	0.0	51.5	8.5	4.8	47.3	38.3	1.1	2.57	33.4	68.6	Gulf Coast
407	88.4	0.0	82.3	6.0	13.9	61.3	76.5	2.0	2.30	90.7	91.5	Gulf Coast
111	92.1	0.0	72.3	13.6	6.7	76.7	66.0	8.4	3.80	61.5	60.3	Gulf Coast
201	78.2	0.2	38.8	36.9	1.5	87.6	32.5	28.8	1.06	76.1	88.9	Gulf Coast
202	72.9	0.4	18.7	65.1	0.6	92.9	21.5	40.0	1.22	67.0	93.0	Gulf Coast
203	6.6	0.0	8.3	0.0	0.9	0.0	4.6	0.0	1.04	16.5	100.0	Gulf Coast
204	13.5	0.0	12.0	7.1	1.9	65.9	11.2	3.1	2.72	36.4	74.6	Gulf Coast
205	71.1	0.0	83.6	0.7	46.0	24.0	81.2	0.3	2.67	34.9	93.5	Gulf Coast
206	0.8	0.0	0.2	0.3	0.2	6.2	0.2	0.0	2.67	13.3	93.5	Gulf Coast
207	0.2	0.0	0.1	0.0	0.1	0.0	0.1	0.0	2.67	18.7	67.8	Gulf Coast
208	50.1	0.3	23.1	44.6	0.8	90.9	20.3	33.9	1.81	74.1	87.5	Gulf Coast
002	0.0	100.0	0.0	100.0	0.0	100.0	0.0	100.0	1.56	99.3	81.4	Gulf Coast
001	50.7	8.5	20.6	21.4	3.1	44.8	11.7	9.7	0.11	6.1	38.3	West
002	82.8	0.0	80.1	1.0	56.1	6.4	68.2	0.9	0.28	0.6	31.8	West
003	27.2	35.3	14.4	5.1	2.0	7.6	6.5	8.2	0.27	0.1	47.1	West
005	96.0	1.8	96.3	1.1	23.5	75.5	69.0	15.9	0.00	0.0	18.8	West
006	99.5	0.4	96.2	3.8	87.5	12.5	77.0	22.9	0.00	0.0	100.0	West
012	37.1	61.7	60.7	33.2	33.0	60.4	40.4	36.3	3.67	24.4	85.7	West
101	100.0	0.0	99.8	0.2	98.0	2.0	97.6	2.4	0.00	0.0	0.0	West
102	92.3	2.0	92.0	5.8	87.3	4.9	54.6	41.3	0.00	0.0	0.0	West
103	96.5	0.3	95.7	0.4	93.6	1.0	45.3	25.8	0.44	1.1	32.4	West
104	85.7	3.2	51.5	27.7	29.2	48.7	12.2	57.1	0.00	0.1	23.3	West
105	100.0	0.0	99.6	0.4	97.2	2.8	96.4	3.5	0.00	0.0	0.0	West

Agricultural sources include harvested cropland, non-harvested cropland, pastureland, and rangeland. Point sources include water treatment plants, powerplants, and industrial sources. Pollutant loadings used to estimate shares by point and nonpoin es from the NCPDI. Erosion rates, cropland's share of all erosion, and percent of agricultural lands needing conservation treat from the NRI.

nit	Shares of total --- NITROGEN --- Agric.	Point	Shares of total --PHOSPHORUS -- Agric.	Point	Shares of total -- SEDIMENT -- Agric.	Point	Shares of total --- BOD5 --- Agric.	Point	Cropland Erosion Rate	Cropland's Share of All Erosion	Percent Agric. Land Needing Cons. Treatment	Region
6	8.6	81.5	10.1	88.1	1.7	98.3	0.3	98.5	0.00	0.0	92.9	West
1	99.9	0.0	99.3	0.7	92.2	7.8	97.0	3.0	0.00	0.0	0.0	West
2	98.3	0.1	76.0	21.4	39.7	58.3	55.9	21.2	0.00	0.0	100.0	West
3	99.6	0.4	98.4	1.5	99.0	1.0	78.1	21.8	0.00	0.0	64.7	West
4	99.6	0.0	99.7	0.1	99.3	0.7	99.1	0.3	0.00	0.0	70.6	West
5	6.3	2.0	47.5	2.5	37.6	11.0	0.3	24.0	0.00	0.0	70.0	West
6	89.0	0.1	53.9	8.6	21.8	38.0	24.7	8.8	0.00	0.0	62.5	West
7	99.7	0.2	92.9	7.1	56.2	43.8	82.1	17.9	0.00	0.0	0.0	West
01	96.8	0.0	98.1	0.0	93.1	0.6	94.0	0.7	0.00	0.0	91.7	West
02	99.9	0.0	99.6	0.2	98.1	1.7	98.5	0.7	0.31	0.1	88.9	West
03	100.0	0.0	99.9	0.1	98.8	1.2	99.5	0.5	0.82	17.5	46.5	West
04	100.0	0.0	99.9	0.1	99.5	0.5	98.3	1.7	0.00	0.0	0.0	West
05	99.2	0.0	99.2	0.5	83.8	14.2	98.8	0.6	0.00	0.0	30.5	West
06	3.7	0.9	2.4	7.9	0.6	70.5	2.0	19.8	0.00	0.0	69.4	West
10	25.8	1.9	10.2	43.2	0.5	97.0	7.6	53.3	0.06	0.0	83.3	West
11	40.5	0.0	59.7	1.0	37.1	33.4	58.4	2.4	0.00	0.0	92.9	West
12	5.2	0.0	5.9	0.3	5.2	9.7	5.9	0.9	0.00	0.0	100.0	West
01	6.8	0.0	8.5	1.4	6.8	36.1	6.9	5.3	0.00	100.0	1.9	West
02	4.0	0.0	5.0	0.4	4.2	13.5	4.9	0.9	0.83	11.9	40.0	West
03	1.6	0.5	2.1	3.7	1.1	54.8	1.6	20.3	0.00	0.0	5.9	West
04	38.4	0.9	15.7	36.1	0.8	91.4	14.7	23.0	0.24	5.4	42.9	West
05	1.8	0.0	4.7	1.1	3.2	32.0	4.7	2.1	0.00	0.0	0.0	West
06	13.6	64.2	2.8	64.6	2.2	11.2	1.1	64.2	0.00	0.0	0.0	West
007	0.0	10.1	0.0	6.4	0.0	0.2	0.0	19.3	0.66	12.3	45.7	West
008	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.5	0.01	0.0	48.1	West
009	0.2	1.2	3.9	13.3	4.6	0.2	0.0	0.4	0.00	0.0	100.0	West
010	88.7	0.0	71.0	0.4	34.8	0.0	49.3	0.0	0.07	0.5	63.6	West
011	3.2	2.0	4.6	18.6	0.3	5.6	0.1	46.9	0.24	15.6	32.7	West
012	42.9	1.0	41.0	1.0	31.3	5.7	9.1	11.8	0.00	0.0	88.2	West
013	49.4	0.9	21.9	34.2	8.4	62.9	3.3	19.0	0.00	0.0	54.2	West
014	42.2	25.5	16.7	40.4	12.5	47.0	2.4	48.8	0.15	0.1	78.6	West
015	26.3	16.4	7.4	73.0	2.0	91.9	0.6	76.6	0.00	0.0	88.4	West
016	95.5	2.1	94.1	0.7	84.3	6.9	65.1	3.2	0.00	0.0	100.0	West
017	82.8	1.2	62.5	2.9	39.8	20.2	18.6	4.3	0.00	0.0	100.0	West
018	52.2	31.9	44.6	15.7	26.8	38.4	1.0	94.6	0.00	0.0	0.0	West
019	91.7	0.1	53.4	9.8	14.2	46.7	27.3	9.4	0.26	7.0	58.6	West
020	98.8	0.0	99.7	0.2	98.6	1.4	95.5	3.2	0.00	0.0	66.7	West
021	97.9	0.2	96.8	0.5	94.7	2.2	70.8	5.5	0.00	0.0	57.1	West
101	97.2	2.7	82.1	17.9	76.1	23.9	25.8	74.2	0.00	0.0	66.7	West
102	0.0	0.1	0.0	2.5	0.0	9.4	0.0	1.2	0.00	0.0	0.0	West
103	99.7	0.1	96.4	3.5	79.3	20.7	81.4	18.4	0.00	0.0	0.0	West
0104	97.0	0.2	99.0	0.7	93.7	6.2	92.9	4.9	0.00	0.0	0.0	West
0105	0.0	67.5	0.0	85.4	0.0	3.7	0.0	41.6	0.15	0.0	57.1	West
0106	99.9	0.0	99.2	0.8	95.6	4.4	83.6	16.3	0.00	0.0	0.0	West
0107	21.6	0.1	13.2	3.2	6.4	13.4	1.7	1.9	0.00	0.0	0.0	West
0108	84.3	0.0	77.5	1.7	48.3	7.1	53.2	2.9	0.00	0.0	5.3	West
0109	12.2	0.3	15.3	2.5	13.3	7.1	0.5	1.4	0.00	0.0	0.0	West
0110	97.7	0.1	85.9	8.5	70.2	23.0	67.2	18.6	0.00	1.2	9.8	West
0111	16.3	0.0	15.6	0.0	8.5	0.0	9.6	0.0	0.00	63.1	100.0	West
0209	99.4	0.0	99.8	0.0	100.0	0.0	99.2	0.0	0.00	0.0	0.0	West
0212	17.8	0.1	51.1	4.7	60.7	34.7	19.3	16.0	0.00	0.0	100.0	West
0214	83.4	13.9	70.3	19.9	84.0	1.5	4.2	93.1	0.00	0.0	0.0	West

e: Agricultural sources include harvested cropland, non-harvested cropland, pastureland, and rangeland. Point sources incluc tewater treatment plants, powerplants, and industrial sources. Pollutant loadings used to estimate shares by point and nonpoi rces from the NCPDI. Erosion rates, cropland's share of all erosion, and percent of agricultural lands needing conservation tre t from the NRI.

o Unit	Shares of total --- NITROGEN --- Agric.	Point	Shares of total --PHOSPHORUS -- Agric.	Point	Shares of total -- SEDIMENT -- Agric.	Point	Shares of total --- BOD5 --- Agric.	Point	Cropland Erosion Rate	Cropland's Share of All Erosion	Percent Agric. Land Needing Cons. Treatment	Region
0109	86.9	8.7	90.4	9.4	89.3	10.7	1.3	98.6	0.78	76.4	22.5	West
0111	99.9	0.0	100.0	0.0	100.0	0.0	99.6	0.3	0.81	37.1	0.0	West
0117	96.8	0.3	77.5	21.5	12.3	87.7	51.1	47.8	0.00	0.6	92.9	West
0002	25.0	0.0	78.6	0.4	91.0	5.8	24.0	1.6	0.15	48.7	26.5	West
0003	93.2	6.6	93.7	6.3	84.9	15.1	4.9	95.1	0.08	1.0	31.0	West
0004	56.0	41.5	98.3	1.5	98.1	1.9	0.1	99.8	0.34	1.1	10.9	West
0005	53.9	0.0	43.8	3.9	25.7	33.0	22.6	4.4	0.37	32.9	11.2	West
0013	3.4	9.2	24.4	2.3	17.2	13.3	0.2	1.2	0.00	27.9	51.5	West
0001	98.3	1.6	98.8	1.2	94.2	5.8	13.6	86.4	1.27	7.1	15.8	West
0002	95.9	0.0	97.2	0.0	98.7	0.7	95.9	0.0	0.00	2.0	13.6	West
0003	96.5	0.0	74.9	0.2	51.8	2.5	50.5	0.2	0.74	0.1	28.6	West
0004	55.8	40.7	95.6	0.2	94.8	4.3	93.8	0.2	7.32	1.6	61.9	West
0005	95.3	4.2	92.4	7.6	90.7	9.3	3.8	96.2	0.00	0.0	0.0	West
0006	32.6	18.8	12.6	15.3	7.3	8.5	3.6	11.9	8.64	4.6	66.7	West
0001	0.0	0.1	0.0	0.0	0.0	0.0	0.0	1.4	10.80	2.3	80.0	West
0002	24.3	0.5	15.8	5.2	24.3	18.2	3.9	3.6	2.72	2.6	18.7	West
0003	0.0	0.1	0.0	0.1	0.0	0.2	0.0	0.0	15.10	100.0	31.3	West
0004	0.0	0.1	0.0	3.9	0.0	14.4	0.0	2.0	4.53	11.9	33.8	West
0005	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	6.16	20.5	57.1	West
0006	1.6	69.2	0.0	99.9	0.0	100.0	0.0	99.9	2.00	0.0	44.4	West
0007	0.0	0.1	0.0	0.4	0.0	0.0	0.0	0.2	0.17	0.1	0.0	West
0008	0.0	11.9	0.0	95.4	0.0	5.6	0.0	71.9	0.33	0.0	0.0	West
0009	16.8	0.3	2.5	0.6	0.2	2.4	1.2	0.3	0.66	93.5	46.7	West
0010	57.0	0.0	19.4	0.0	1.6	0.0	9.9	0.0	3.65	0.9	24.1	West
0011	0.0	5.1	0.0	1.7	0.0	6.5	0.0	0.9	1.86	52.6	58.2	West
0012	0.0	9.0	0.0	84.5	0.0	81.8	0.0	32.9	0.00	0.0	0.0	West
0013	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.00	16.7	53.8	West
0014	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.00	0.0	0.0	West
0101	0.0	4.9	0.0	3.6	0.0	12.9	0.0	1.7	52.69	64.9	65.5	West
0102	0.0	0.7	0.0	3.1	0.0	7.6	0.0	2.6	0.67	0.1	3.8	West
0103	88.2	0.0	66.4	1.0	75.3	4.0	23.9	1.5	0.94	34.6	32.5	West
0104	22.5	0.0	13.8	0.1	27.9	0.7	2.4	0.1	0.00	0.0	0.0	West
0105	0.0	3.4	0.0	89.4	0.0	0.0	0.0	24.6	0.00	0.0	0.0	West
0106	0.4	2.3	0.5	6.5	0.7	65.3	0.2	43.0	0.00	1.7	0.0	West
0107	34.8	3.2	13.4	21.8	3.2	79.5	5.3	57.3	0.00	0.0	0.0	West
0201	29.1	1.7	30.4	15.5	5.8	74.9	24.3	27.2	0.00	0.0	0.0	West
0202	0.5	2.4	0.9	5.4	1.1	54.4	0.3	39.0	0.61	15.1	12.1	West
0203	72.3	4.5	42.4	45.2	7.0	87.4	7.9	73.1	0.28	0.2	36.6	West
0204	0.1	0.1	0.1	1.1	0.1	24.7	0.1	5.8	0.00	17.7	100.0	West
0301	8.8	1.9	5.9	47.8	0.2	95.6	3.2	63.5	0.68	0.0	46.2	West
0302	1.2	5.3	0.3	40.7	0.0	80.7	0.1	48.3	0.51	36.5	48.0	West
0303	1.9	2.6	2.7	31.0	1.4	89.4	0.5	52.9	0.82	6.4	38.5	West
0304	5.3	0.2	4.4	1.8	3.6	43.6	3.8	7.6	1.78	0.1	11.8	West
0305	8.0	2.0	4.2	22.6	0.5	86.3	2.1	45.6	0.08	0.0	0.0	West

: Agricultural sources include harvested cropland, non-harvested cropland, pastureland, and rangeland. Point sources includ
ewater treatment plants, powerplants, and industrial sources. Pollutant loadings used to estimate shares by point and nonpoi
ces from the NCPDI. Erosion rates, cropland's share of all erosion, and percent of agricultural lands needing conservation trea
from the NRI.

DRAFT 3

Integrating Economic and Physical Models for Analyzing Environmental Effects of Nonpoint-Source Pollution

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1. Introduction

Public concern over environmental issues has increased dramatically over the last two decades and agriculture has not escaped this environmental scrutiny. The impact of agricultural practices on resource quality and, in particular, on ground and surface water quality has received both political attention and public research dollars. One of the critical issues to be faced by policy makers is how to design institutions that protect environmental quality and are compatible with productivity growth. Such policy design requires, as we argue in this paper, a synthesis of research from social and physical scientists to identify and quantify the magnitude of the social benefit and costs associated with current agricultural practices in relation to environmental quality.

There are at least two reasons why, in the past, analysts have tended not to include the environmental and health impacts in their analyses of returns to agricultural research or in their evaluation of specific policies or programs: deficiencies in methodology and data. On the methodology issue, a comprehensive analytical framework is needed which combines field-level relationships among management practices, environmental attributes of the farmland, and nonpoint pollution with impacts on human health and the ecosystem. The research from various disciplines (physical, biological, economic, and health sciences) needs to be integrated into an analytical framework that, to be useful for policy analysis, makes the link between the physical changes in environmental and resource quality attributable to agricultural practices, and the valuation attached to the changes in environmental quality and the subsequent impacts on human health. With respect to data deficiencies, the concerns are in two related areas: the information needed to quantify the environmental quality and agricultural production relationships has generally not been available; and the data on health effects of exposure to agricultural chemicals are far from complete.

This paper begins to address these deficiencies, first, by developing an approach to integrating disciplinary research to quantify and value the impacts of agricultural chemical use, and second, by highlighting the data requirements for this research. The approach is illustrated using the issue of chemical contamination of groundwater.

Pollution of groundwater by agricultural chemicals is often categorized as a nonpoint-source pollution issue. Tietenberg offers the following distinction between point and nonpoint sources of water contamination: "Point sources generally discharge into surface water at a specific location through a pipe, outfall or ditch, while nonpoint sources usually affect the water in a more indirect and diffuse way" (p. 406). The control of nonpoint-source pollution has, until recently, received relatively little theoretical or empirical attention. The recent presidential Water Quality Initiative and its focus on agricultural sources of water pollution has helped focus policy and research interest on this problem.

In effect, the approach taken in this paper is to transform the nonpoint-source problem into a more manageable point-source problem. This is done by using a well-defined distribution of characteristics for a large number of heterogeneous sources (or fields) to simulate how much of a given chemical will reach an environmental medium. The link between the characteristics of the sources and the quantity of pollution is made using the chemical fate and transport models. Thus, one unique feature of the framework we propose is the integration of the physical science models, which deal with what occurs at the specific points of chemical application, with the policy models that need to effectively deal with a collection of heterogeneous points. Similar approaches have been used for air pollution control models. However, an important modification for the groundwater pollution problem is the need to model the movement and changes in composition of the chemical from the point of discharge (application) to the point of entry into the groundwater aquifer.

Benefit cost analysis (BCA) provides the framework in which to organize a coherent approach to incorporating environmental and health costs into public policy analysis and for addressing the uncertainties inherent in this type of analysis. The BCA process for addressing the environmental and health impacts of pesticide use is presented in Figure 1. The first step is to determine the effect of the policy or the change in technology on the output and input decisions of the farmers; the second step is to quantify how a farmer's response affects the magnitude of the benefits and costs. In the case of a pesticide use reduction, changes in environmental contamination, food residues, and occupational exposure give rise to the benefits; the effects on production and resource use determine the costs.

The environmental impacts of changes in pesticide use depend on the physical processes of pesticide transport through soil and water mediums and subsequent contamination of secondary food sources. Analysis of the effects of changes in pesticide use on human health involves both human and environmental risk assessment.

The third step is to express the benefits and costs in a common unit that reflects their valuation by the affected individuals. The valuation of the costs of the pesticide use restrictions or changes in production technology can be measured as changes in producer and consumer surpluses or related welfare measures. The valuation of the benefits involves predicting the impacts on the environment and estimating nonmarket values.

The final step in the BCA process is the determination of the net impact on social welfare. This requires a criterion for determining what qualifies as an increase in welfare, and a means for aggregating the impacts which may occur at different points in time and impact different groups of individuals.

The remainder of the paper is organized as follows: section 2 presents an overview of the characteristics of the physical models that can be used to predict the movement of chemicals in soils and discusses a prototype model for assessing pesticide concentrations in the soil and groundwater. In section 3 attention is focused on modification of economic production models. Section 4 addresses the methodological issues that arise in integrating physical and economic models for use in the benefit-cost framework.

2. Physical models for quantifying contamination levels

Physical models for quantifying chemical pollution externalities need to address movement of chemicals to both surface water and groundwater. In the last three decades an extensive literature has been generated by research aiming to trace the movement of surface water contaminants. Climate, watershed and soil characteristics, and crop management practices have been found to affect the magnitudes of the impacts (see Jury et al., 1987).

Concern over groundwater contamination is a relatively recent development and, as a result, models that predict chemical leaching to groundwater are less developed than models that predict chemical runoff to surface water. To predict potential loadings to groundwater, a model is needed to trace the movement of the chemical from the application site down through the unsaturated zone and into the saturated zone. The saturated zone is the area in which all the void spaces are filled with water; in the unsaturated zone, the void spaces are filled with both air and water, the proportion of which is important in modeling transport rates.

The fate of a chemical applied to soil depends on the pesticide’s properties. Persistence is a measure of a chemical’s rate of degradation and is usually measured in terms of a chemical’s half-life. Solubility, sorption, and volatility determine how a compound partitions among water, soil, and air phases and affect whether the chemical is moved primarily with sediment or water. When a pesticide is applied, some of it will adhere to the organic carbon in the soil particles; this is called adsorption. Some of the pesticide will mix with soil water and move down with the soil water. An inverse relationship exists between the solubility of the pesticide and its sorption to soil. A partition coefficient value is used to describe the ratio of pesticide concentration in the adsorbed phase and the solution phase. The smaller the partition coefficient, the greater the concentration of pesticide in solution. Hydrologists have noted that the greatest threat to groundwater through leaching is associated with a pesticide with a small partition coefficient and a long half-life.

2.1 Chemical transtport models: An overview

Although the specific structure of the chemical fate and transport models vary, most models contain some standard components. These include:

(i) Surface runoff generation component: describes the transformation of precipitation into runoff. The soil surface and profile provide major controls on the response of the surface-water system. During interstorm periods, pesticides may be applied and undergo a variety of transformation and degradation processes affecting the total mass of each constituent available for entrainment and transport. Land-use practices such as tillage affect the infiltration, runoff, and erosion processes. The processes composing the surface-runoff system are hydrology, sediment, nutrients, and pesticides. (A detailed presentation of modeling surface runoff is provided by Beasley et al., 1989.) The USDA Soil Conservation Service Curve Number (SCSCN) model is commonly used to estimate runoff. This method relates direct runoff to daily rainfall as a function of a curve number representing soil type, soil drainage properties, crop type, and management practice.

(ii) Soil and groundwater component: describes chemical movement through the unsaturated soil zone and may also describe movement into the saturated zone. Not all models trace the movement of chemicals through the unsaturated zone to the saturated zone.

(iii) Erosion component: estimates soil loss due to erosion. This is important when determining potential for groundwater contamination because soil sediment is a medium of transport for adsorbed pesticides. A pesticide or nutrient that is transported off the field via eroded soil is not available for leaching to groundwater. The Universal Soil Loss Equation (USLE), or a modification of the USLE, is frequently used to model erosion. The USLE accounts for factors such as rainfall, crop management, slope conditions, and erosion control practices in calculating soil loss per acre.

(iv) Soil adsorption and desorption component: estimates the partitioning of a chemical between adsorbed particles and dissolved chemicals. This component estimates what portion of the chemical may be transported by soil sediment and what portion may be transported by soil water. It may also model volatilization and decay of the chemical.

Chemical transport models can be divided into three broad categories: research models, screening models, and management models (Wagenet and Rao, 1990, provide a detailed discussion of these models). Research models provide quantitative estimates of water and solute movement, but usually involve extensive data demands on the system to be simulated. Management models are less data intensive, and less quantitative in their ability to predict water and solute movement under various environmental conditions. Although most managerial and research models are field scale models, Wagenet and Rao indicate that there has been limited field testing of either the research or management models to date, and thus little attention has been focused on the so-called management models for the actual purpose of managing pesticide or fertilizer usage. The existing research models are useful for management purposes only if computer facilities and time are virtually unlimited.

Screening models are used to evaluate and compare pesticide fate and transport under alternative environmental conditions. The screening models have relatively low data demands, and are designed to be relatively easy and inexpensive to use. One useful output of these models is to categorize chemicals into broad behavioral classes. These models have relevance in the pesticide registration process, where the properties of a pesticide which has not been field-tested can be inferred from the class in which it is placed. Several simple indexes useful to screen and rank pesticides in terms of their potential to leach into groundwater have been developed by Rao et al., 1985. These ranking schemes are based on a screening model which determines the relative travel time needed for the pesticide to migrate through the unsaturated zone, and the relative mass emissions (loadings) from the unsaturated zone into the groundwater.

Jury et al., 1987, have also developed a screening model of the pesticide leaching process. This model relaxes the uniform first order decay assumption for pesticide degradation in the unsaturated zone which characterizes the Rao et al., 1985, model and replaces it with a biochemical decay relationship which decreases with soil depth. The results of both screening medals indicate a significant dependence on site-specific soil and environmental conditions, suggesting that these factors, as well as the pesticide properties, need to be taken into account when screening for groundwater pollution potential.

Wagenet and Rao caution against using existing screening models to predict environmental changes. They indicate that the recent interest in using models to predict the fate of pesticides in water and soils has provided an impetus to improve upon the accuracy of both screening and research models. One of the most promising avenues to proceed for developing policy models is condensing the comprehensive descriptions provided by research models. Examples of such an approach are the recent changes to the PRZM and LEACHM models (see Wagenet and Hutson, 1987) and the Jury et al. (1 987) model and the prototype model discussed in section 3.2.

2.2 A simple Desticide leaching model

One major disadvantage of the large scale research simulation models is their lack of attention to the movement of chemicals through the unsaturated zone, although groundwater components have recently been appended to some models. A second disadvantage of these models is simply the size and data requirements. Most utilize daily and often hourly climate data to simulate chemical movement.

As an alternative, researchers have been developing screening models to evaluate pesticide groundwater pollution potential (Jury et al., 1987; Rao et al., 1985). This approach is promising for use in regulatory BCA, and thus we illustrate the integration of such a model into the net benefit specification.

Two key variables in assessing the behavior of chemicals as they leach into groundwater are pesticide residence time and the fraction of the pesticide remaining as functions of depth in the unsaturated zone. Physical relationships can be used to estimate residence time, ti, and the time required for a pesticide particle to travel from land surface to the depth of interest, z_i, as a function of physical parameters such as: water flux per unit surface area; residual moisture content; dry bulk density; the organic-carbon partition coefficient of the pesticide and the percentage of organic carbon in the layer.

The fraction of the pesticide remaining at the depth of interest is calculated taking into account both the decay and root uptake processes. The fraction of the pesticide that remains after decay that occurs during its transport through each soil layer can be calculated by solving the equation for irreversible first-order reactions allowing for the known half-life of the pesticide:

$$r_i = e^{-0.693\,(t_i/h_i)} \qquad (1)$$

where r_i denotes the fraction of the pesticide remaining after transport in the ith layer; t_i denotes the time of travel (residence time) in the layer of interest, in days; and hi denotes the half-life of the pesticide in the layer, in days.

These latter values are assigned to each layer in the system based on empirically obtained figures from field and laboratory experiments. The percentage of the original pesticide applied to the land surface that remains after transport through more than one layer is the product of the values of r_i for each layer. The percentage of the pesticide remaining after transport and decay through all layers is then

$$r_z = \prod_i r_i \ . \tag{2}$$

The key parameters in determining the amount that remains generally are half-life of the chemical, porosity, partition coefficient (which is determined by the organic-carbon coefficient of the pesticide, and the percentage of organic carbon in each layer), water flux, and water content.

The root uptake process also must be estimated, and as a first-order approximation, can be assumed to be proportional to the root uptake of water, evapotranspiration. To obtain the fraction of the pesticide remaining after these two processes (root uptake and decay) have occurred, the amount of pesticide remaining after decay is multiplied by the ratio of the amount of water flux at the depth of interest to the amount of water entering the ground at land surface:

$$C(X_j) = (r_z)(q/w)(X_j) \tag{3}$$

where q denotes water flux per unit surface area, X_j denotes the amount of pesticide applied, and w denotes the rate at which water enters the ground. Equation (3) could be incorporated into a net benefit analysis as illustrated in section 4.

To utilize this kind of model, information would be required on soil (physical) and pesticide characteristics. The soil characteristics include the rate at which water enters the ground; the rate of deep percolation below roots; the thickness of the root zone; the depth to water table; and the density of solid matter in the unsaturated zone. Other layer-specific physical characteristics include the type of material; the residual moisture (water) content; the porosity; and the organic carbon content of the soil. Pesticide characteristics of importance to these models are organic-carbon partition coefficient; and the half-life in each layer. In addition, data on pesticide applications are also needed. Of the above information, only the pesticide application levels and the amount of water entering the ground at time of application would need to be collected each period.

2.3 Environmental Exposure Modeling

More general approaches to environmental quality modeling are also being developed. The standard approach to modeling environmental exposure is to assume that chemicals are distributed into various environmental compartments as functions of chemical properties, environmental factors, and chemical use according to equilibrium partitioning models (Mackay et al., 1985). For example, it may be assumed that a pesticide applied to a field will be partitioned among air, water, soil, flora, and fauna. Symbolically,

$$C_{ij} = C_i(X_j, K_{ij}, E_i)$$

where:

C_{ij} is the concentration of the jth chemical in the ith partition;

X_j is chemical use;

K_{ij} is the partition coefficient; and

E_i is a vector of environmental factors.

The environmental contamination in each partition can be translated into exposure of the kth species through the expression

$$e_{jk} = \Sigma_i C_{ij} A_{ijk} (\gamma)$$

where

e_{jk} is the exposure of the kth species to the jth chemical;

$A_{ijk}(\gamma)$ is the rate of uptake of the jth chemical in the ith partition by the kth species; and

γ is a vector of individual species characteristics.

Thus, in general total exposure of the kth species to the jth chemical is a function $e_{jk}(X,K,E,\gamma)$, where the arguments are vectors of chemicals used, partition coefficients, environmental characteristics, and species characteristics. These exposure measurements can in turn be valued and used in BCA.

3. Economic Production Models

The economic behavior of agricultural firms can be represented as a two-level decision process corresponding to the short-run and the long-run (Figure 2). In the short-run, firms make production decisions regarding outputs (types of crops and allocation of acreage among crops) and variable inputs (such as labor hours, fertilizer applications) taking as given the available technology and the existing stocks of physical capital and other resources used in production. These short-run decisions may be important in the analysis of externalities because they may include the use of agricultural chemicals which are a source of pollution. In the long-run, firms make investment decisions based on their expectations of future market conditions, technology, and resource availability. Their long-run decisions include the total acreage of the farm operation and the quantities of physical capital employed. The long-run decisions may also have important consequences for externality generation. For example, the choice of tillage method (conventional tillage versus reduced or no-till) may have an impact on soil erosion and herbicide use, and hence on pollution caused by chemical runoff.

3.1 Producer behavior in static models

The analysis using a static model focuses on the output and input decisions that are made in each production period, given technological, economic, and resource constraints. Farmers are assumed to be concerned with the private benefits and costs of their farm operations, and thus do not take into account the longer-term impacts of their production activities on the ecosystem or on human health caused by agricultural pollution that occur off their farms. For the measurement of externalities, the effects of the output and input decisions on physical resource stocks and living organisms in the ecosystem can be quantified. To measure the sequence of externalities generated over time, the biological system's changes can be incorporated into the economic model to define the resource constraints on production in the next period, and the analysis can be repeated.

The short-run economic behavior of an agricultural producer can be modeled in terms of profit maximization; more generally risk management and other objectives can be introduced, but as a first-order approximation, profit maximization is a useful starting point. Analysis of the profit-maximizing firm is based on the representation of the production process using the production function

$$Q_t = f(X_t, Z_t, \tau_t, R_t, S_t)$$

where Q_t is the maximum rate of output that can be produced in period t with variable inputs X_t (generally, a vector measuring labor, fertilizer, pesticides, etc.), fixed (capital) inputs Z_t (a vector measuring land, structures, machinery and tools, etc.), and parameter τ_t representing the state of the technology (traditional seed variety versus modern seed variety, for example).

The role of physical and biological resources in the production process is represented by the vectors $\mathbf{R}_t$ (physical resources) and $\mathbf{S}_t$ (living organisms) in the production function. The vector $\mathbf{R}_t$ could measure physical attributes of the resources used in production, such as soil and water quality, and the vector $\mathbf{S}_t$ could measure populations of pests and natural enemies to pests.

The profit maximization problem is represented as

$$\max \pi_t = P_t f(X_t, Z_t, \tau_t, R_t, S_t) - W_t X_t$$

where P_t is the price of output and W_t is a vector of prices corresponding to the elements of X_t.

By assuming that the production function is concave in the variable inputs X_t, the dual restricted profit function,

$$\pi_t = \pi\left[P_t, W_t, Z_t, R_t, S_t, \tau_t\right],$$

can be defined as the maximum profit the firm can earn, given Pt, W_t, Z_t, τ_t, R_t, and S_t, by choosing levels of output and variable inputs. A property of the profit function is that the firm's profit-maximizing output, Q*, and its profit-maximizing input vector, X*, satisfy the following relationship:

$$Q_t^* = \partial\pi\left[P_t, W_t, Z_t, R_t, S_t, \tau_t\right] / \partial P_t = Q^*\left[P_t, W_t, Z_t, R_t, S_t, \tau_t\right]$$

$$X_t^* = -\partial\pi\left[P_t, W_t, Z_t, R_t, S_t, \tau_t\right] / \partial W_t = X^*\left[P_t, W_t, Z_t, R_t, S_t, \tau_t\right]$$

The complete production model is represented by the system of the three previous equations. Since the first equation measures short-run profit, it can be interpreted as measuring the producer surplus (net returns) used in BCA. For example, if a new seed variety was introduced, but prices, physical capital, and resource stocks were constant, the profit function would indicate the resulting change in producer surplus attributable to the new seed variety. The equation system also shows that the introduction of the new seed variety would generally have an effect on supply of output and on the demand for inputs. The introduction of a new variety would affect the demand for agricultural chemicals. This change in the use of agricultural chemicals would provide the link from the economic behavior of the farmers to the physical and biological models used to quantify pollution externalities.

The production model also shows that, generally, the economic relationships in period t depend on the resource stocks and living organisms represented by R_t and S_t. The economic model does not determine these variables in the current production period, rather R_t and S_t play the role of constraints on the production process. The values of R_{t+1} and S_{t+1} in the next period are determined in part by the production decisions in period t. Thus the physical, biological, and economic sectors of the model interact dynamically according to the particular structure and parameterization of the systems of equations used to represent them. Given estimates of the parameters of these equations, initial values of the stocks R_t and S_t, and predictions of the "forcing variables" such as prices that are determined outside of the model, the system of equations can be used to generate predictions of the time paths of agricultural production (Q_t), input use (X_t), and the physical and biological stocks $(R_t$ and $S_t)$.

3.2 Long-run dynamic investment models.

In some cases it is not appropriate to use a short-run static production model to analyze externality generation. A long-run model may be needed for a variety of reasons: because the choice of capital stock is important in the amount of externality created; or because farmers do take externalities into account in their decision making; or for long-run regional analysis of externality creation where the effect of the externality feeds back into the production process. To illustrate, consider a modal in which physical capital evolves over time according to

$$Z_{t+1} = (1 - \delta)\, Z_t + V_t\,,$$

where δ is the rate of capital depreciation and V_t is the rate of gross investment each period. Similarly assume that the dynamics of the resources R_t and species S_t are given by

$$R_{t+1} = H(R_t, X_t, Z_t)$$

$$S_{t+1} = B(S_t, X_t, Z_t, R_t)\,.$$

The long-run maximization problem of the farmer is now defined as choosing the sequence of investments to maximize the present discounted value of profit from each period over the relevant planning horizon:

$$\max_{\{V_t\}} \sum_{t=1}^{T} \eta_t \left\{ \pi\left[P_t, W_t, Z_t, R_t, S_t, \tau_t\right] - U_t V_t \right\} + J\left[Z_{T+1}, R_{T+1}, S_{T+1}\right].$$

subject to:

$$Z_{t+1} = (1-\delta) Z_t + V_t$$

$$S_{t+1} = B(S_t, X_t, Z_t, R_t)$$

$$R_{t+1} = H(R_t, X_t, Z_t)$$

where η_t is a discount factor depending on the rate of interest, U_t is the price of investment goods, and J measures the terminal value of the physical capital and resource stocks.

The above problem can be solved using optimal control or dynamic programming techniques. For example, the solution can be obtained by maximizing the following Hamiltonian equation:

$$H_t = \eta_t \left\{ \pi\left[P_t, Q_t, Z_t, R_t, S_t, \tau_t\right] - U_t V_t \right\} + \lambda_t \left\{ (1-\delta) Z_t + V_t \right\}$$

$$+ \mu_t B(S_t, X_t, Z_t, R_t) + \rho_t H(R_t, X_t, Z_t)\,,$$

where λ_t, μ_t, and ρ_t are the multipliers for Z_t, S_t and R_t and represent the marginal capital values of these stocks. Maximizing the Hamiltonian and solving the resulting set of first-order conditions along with the constraints of the maximization problem gives an investment demand equation of the form

$$V_t = V\left[Z_t, S_t, R_t, P^t, W^t, r^t, U^t, \lambda_{T+1}, \mu_{T+1}, \rho_{T+1}\right],$$

where $P^t = (P_t, P_{t+1}, \ldots, P_T)$ and similar notation applies to other variables. Thus the optimal investment in each period is a function of the current stocks of capital and resources, current and future prices, and the terminal values of the capital and resource stocks.

Using the investment demand equation for V_t together with the equations of motion for R_t and S_t and the equation for output supply and input demand, one can solve for the long-run paths of all variables determined by the farmer. Note that the short-run and long-run models suggest a very different model of interaction between the economic, physical, and biological models. With the short-run economic model, economic decisions are made given the states of the physical and biological variables, and the physical and biological models are solved given the behavior of farmers. Time paths for the variables in each model are obtained by sequentially solving each model and using its results to condition the solution of the other model. In contrast, in the dynamic economic model, economic decisions are made taking into account the dynamics of the physical resource stocks and the population dynamics of species. Thus the time paths for the economic, physical, and biological variables are determined jointly in the solution of the dynamic economic model.

4. Mode Integration

4.1 Methodological Issues

Several methodological issues arise as the physical and economic model components are brought together into an integrated model. Successful integration requires compatible mathematical structures for numerical models and consistent statistical criteria need to be developed. In addition, several conceptual differences in model approaches exist across disciplines that need to be taken into consideration. The most important point to be emphasized in conducting this integration is the need for communication across disciplinary lines.

Physical versus Behavioral Modeling. First, there is a conceptual difference between the physical modeling, which relies upon physical constants, and behavioral models based on the assumed optimizing behavior of people. The structure of a physical model is invariant to changes in government policy, for example, but a model of farmer behavior may need to take into consideration the way farmers form expectations about policy. Consequently, the structure of a behavioral model may change over time as policy and other parameters change. The change in the structure of the behavioral model may in turn alter the linkages between the physical and economic models.

Experimental versus Nonexperimental Data. The physical and biological sciences rely primarily on data generated by controlled experiments. Economic analysis is generally based on nonexperimental data. Econometrics is devoted to the modification of classical statistical analysis so that valid inferences can be drawn from nonexperimental data. The differences in statistical methods need to be reconciled in the design of data surveys and research methodologies.

Modeling Approaches. Various disciplines find particular mathematical structures to be appropriate for their problems. For models to be integrated across disciplines, all disciplinary model components must be consistent with the ultimate goal of linking the models for policy analysis.

Selecting the Unit of Analysis: The Aggregation Problem. A basic methodological problem arises in any attempt to integrate the physical, health, and economic model components into a coherent whole; each component relates to a particular unit of analysis, each of which is generally different from the unit of analysis on which cost-benefit analysis should be based. The solution to this problem is to provide a statistical representation of the integrated model that can be defined over a common unit of analysis, and then to statistically aggregate to the unit of measurement meaningful to cost-benefit analysis.

4.2 A Statistical Approach to Model Integration

A key factor that needs to be taken into account in the modeling methodology is the heterogeneity of the physical environment and the related heterogeneity of agricultural production practices and associated environmental and health effects of those practices (Antle and Just, 1990). For example, an analysis of environmental fate of a pesticide based on a set of partition coefficients may be reasonable for a well-defined physical unit--say, 100 square meters of surface area--over which a specific set of parameters and input data are valid. But such a unit is generally much smaller than the economic or geophysical unit of analysis relevant to the assessment of social costs of chemical use. The relevant unit of analysis for social cost assessment may be as small as a farm or as large as an entire regional watershed.

To address the heterogeneity problem, an aggregate unit of analysis can be defined as a function of the problem context; e.g., for water quality problems the unit of analysis may be the land contained in a particular watershed. The land in the aggregate unit of analysis can, in turn, be disaggregate into sufficiently small units (plots) over which a valid set of physical and economic data and parameters can be defined. Associated with each plot is a vector of physical characteristics represented by $\boldsymbol{\omega}$. $\boldsymbol{\omega}$ may include physical characteristics such as depth to groundwater on the plot, the partition coefficients for the plot, the slope and elevation of the plot, and so forth. A stylized physical model can then be written C(X,w), where C is a vector of contaminant levels associated with the environmental partitions in the model (e.g., soil, air, water) and X is a vector of chemical applications.

As shown in section 3, a farmer's chemical-use decisions are functions X(P, ψ,τ,ω), where P represents prices of outputs and inputs, ψ represents policy parameters, τ is technology parameters, and w is as defined above. Let the environmental characteristics of each plot of land in the region be fixed at a point of time and distributed across plots according to a distribution defined by a parameter θ. This distribution of environmental attributes induces a joint distribution for input use X, crop production Q, and contamination levels. Define this joint distribution as $\phi(Q,X,C \mid P,\psi,\tau,\theta)$.

4.3 Statistical Aggregation

The joint distribution ϕ provides a basis for statistical aggregation across the plots into quantities that can be used to conduct policy analysis at the aggregate level. For example, by integrating X and Q out of ϕ, a marginal distribution of contamination can be defined: $\phi(C \mid P,\psi,\tau,\theta)$. Using this distribution, the tradeoffs between, say, mean chemical use and groundwater contamination can be estimated. This information can be combined with valuation data to estimate the value associated with groundwater contamination. In addition, an aggregate pollution function can be obtained by taking the expectation of C with respect to this marginal distribution, and that relationship can be used for analysis of pollution policy (see Antle and Just, 1990).

To illustrate the statistical aggregation procedures, let X and w follow a lognormal distribution such that

$$\begin{bmatrix} \ln X \\ \ln \omega \end{bmatrix} \sim N\left[\mu, \Sigma \mid P, \psi, \theta\right],$$

where μ is a (2 x 1) vector of means and Σ is a (2 x 2) covariance matrix. It follows that C is a random variable and its mean and variance are functions of μ and Σ, which are in turn functions of P, ψ, and θ. Thus, for example, the population mean contamination level may be expressed as a function of the population mean level of chemical use. This relationship can be employed in policy analysis. For example, if a dollar value could be attached to a specified reduction in environmental contamination, these data can be used in cost-benefit analyses of policies to reduce pesticide use.

4.4 A Simple Economic-Physical Groundwater Contamination Model for Policy Analysis

This section describes an integrated economic-physical groundwater contamination model for policy analysis. The model is defined for a given chemical at a given location, such as a plot or field, which is homogeneous with respect to both physical and economic characteristics. It is based upon the models presented in sections 2 and 3.

A Physical Model

Following earlier notation, let

X = quantity of chemical

C = concentration of chemical x in groundwater

z = depth to groundwater

m = time for transport from surface to groundwater

r = fraction of chemical remaining after transport to groundwater

t = time period t = 0,1,2,...

h = half-life of chemical in groundwater

$h^* = 0.693/h$

following the model presented in section 2.2, assume: the chemical does not move laterally in the soil or groundwater; it degrades according to first-order irreversible reactions; and the groundwater is uncontaminated at time t = O. Then:

$$C_t = \sum_{k=1}^{t} x_k R_{kt} \tag{4}$$

where

$$R_{kt} = r \exp\left\{h^* \left[t - (m + k)\right]\right\} \quad \text{if } t - (m + k) > 0$$
$$= 0 \qquad\qquad\qquad\qquad \text{if } t - (m + k) < 0$$

Note that R_{kt} is interpreted as the fraction remaining at time t > k from application at k, including the effects of transport to groundwater and decay in the groundwater. The equation (4) is quite general and compatible with any specification of the coefficients R_{kt}. For example, R_{kt} could be specified more generally to embody the effects of lateral movement of groundwater.

An "economic" interpretation of equation (4) is possible. Since $R_{k,(t+s)} = R_{kt} \exp(h^*s)$, and $R_{k,t+s} = 0$ fors < m, C_t can also be expressed as

$$C_t = \exp\left\{h^*(m+1)\right\} C_{t-1} + x_{t-m} R_{t-m,t} \,.$$

Thus C_t can be expressed in the form of an equation of motion of a capital stock, $K_t = (1 - \delta)K_{t-1} + I_t$, where K_t is the stock, δ is the depreciation rate of the stock, and I_t is gross investment. Under this interpretation, exp {h*(m + 1) } represents the depreciation of the "stock" of contamination due to the decay of the chemical that is already in the groundwater, and $X_{t-m}R_{t-m,t}$ represents the gross investment, which in this model is the additional chemical that was applied at time t - m and leaches to the groundwater at time t.

An Economic Model

To illustrate the basic economic relationships, assume the simplest possible conditions: production of a single crop Q with a single variable input, the chemical X, on the given unit of land. The farmer chooses X to maximize profit π subject to the production process

$$Q = X^{\alpha_1} .$$

Solving the profit maximization problem

$$\max_{x} \pi = p\,Q - w\,x$$

gives

(5)
$$X = \left(\frac{1}{\alpha_1} \frac{w}{p} \right)^{1/(\alpha_1 - 1)}$$

Impact of Policy Changes on Groundwater Quality

Consider now a policy that sets $p_t = p^*$ for all $t > t^*$. We have the following relationships:

$$\left. \begin{aligned} \partial C_t/\partial X_{t'} &= 0 \text{ for } t - t' < m \\ &= R_{t't} \text{ for } t - t' > m \end{aligned} \right\} \text{ and } t' > t^* .$$

Hence the elasticity of C_t with respect to $X_{t'}$ is

(6)
$$\epsilon_{tt'} = X_{t'}R_{t't} / C_t$$

The elasticity of X_t with respect to p_t is, according to the model in equation (5)

(7)
$$\eta_t = 1/(\alpha_1 - 1), \text{ for all t.}$$

It follows that the effect of raising p permanently at time t* by the amount $\Delta p^* = p^* - p_o$ is

$$\Delta C_t / \Delta p^* = \sum_{k=1}^{t} \left(\Delta X_k^* / \Delta p^*\right) R_{kt}$$

which in point elasticity form is, in general,

(8) $$\xi_t = \sum_{k=t^*}^{t} \epsilon_{tk}\, \eta_k$$

and using (6) and (7) becomes

(9) $$\xi_t = \sum_{k=t^*}^{t} X_k R_{kt} / C_t (\alpha_1 - 1) .$$

These relationships are illustrated in the Figure 3 under the assumption that before t^*, $p = p_o$, and input use occurs at fixed time intervals. Under the baseline scenario, input use generates a relatively slow increase in groundwater contamination levels; when policy raises 'the price of the crop, chemical use levels increase and the rate of growth in contamination increases. Observe that before t*, contamination levels increased by the amount ΔC_t each period, whereas after t* + m contamination levels increase by $\Delta C_t^* > \Delta C_t$ each period (note the delay of m between the time the policy change is implemented and it begins to have an effect on groundwater quality because of the transport time). The elasticity ξ_t measures the percentage increase in C_t for each time period. Note that ξ_t is zero fort* < t < t* + m and is an increasing value thereafter.

The analysis of a policy which reduced p once and for all would be similar and would show that a reduction in input use levels would reduce contamination levels over time. Note, however, that the effect of the policy on groundwater quality would occur with a delay of m.

This simple example illustrates several interesting points. First, equation (8) shows that, in general, the effect of policy on groundwater quality is a function of all of the physical and economic parameters required to obtain ϵ_{tk} and η_k, whether these values are estimated from simple or complex models.

Second, suppose that chemical input use was sufficiently low such that C = 0 for all t < t* because all of the chemical degrades in the soil during transport (r = 0). Then a policy that induced an increase in chemical use would not affect contamination until input use reached the critical level at which r becomes marginally positive. Hence it follows that a policy that increases input use does not necessarily decrease groundwater quality.

More generally, input use will not be at constant intervals and market prices will be changing over time in response to policy and market conditions, and the time path of contamination levels will be much more complicated.

Finally, note that this model applies to a specific site. As discussed in the previous section, it can be assumed that the physical and economic parameters follow well-defined distributions in the watershed. This distribution, in turn, defines a joint distribution in the watershed for C, Q, and X. This joint distribution can be used to represent the watershed statistically as a unit and to conduct policy analysis. For example, it would allow statements to be made about the effect of a policy change on the expected (average) contamination level, or about the probability that contamination at any site in the watershed is less than or equal to a critical value, such as a maximum contamination level set by a risk analysis.

5. Conclusion

Benefit-cost analysis provides the foundation for developing a framework for integrating the various strands of disciplinary research needed to assess the environmental impacts of agricultural chemical use. The ability to predict the likelihood that a chemical applied at a specific point will end up in the groundwater enhances the economist's ability to devise location-specific policies for efficiently meeting pollution standards. In essence, by utilizing appropriate economic and physical models, it may be possible to overcome some of the "nonpoint" characteristics of the problem.

The data needed to identify accurately the potential for environmental impacts of chemical use are location-specific and chemical-specific. These information needs include the characteristics of the chemical and the physical environment that provide a basis for estimation of the chemical's mobility and degradation in the environment, and farm-level and field-specific production data that allow the farmer's chemical-use decisions to be modeled.

The heterogeneity of the physical environment means that chemical transport must be modeled at a highly disaggregate level. Thus, farmers' chemical-use decisions must also be modeled at a disaggregate level. Policy issues must be addressed at a more aggregate level, however. The bridge between these two levels of analysis is a statistical representation of the physical environment and the producer population which provides the basis for statistical aggregation from the highly disaggregate level required for physical models to the more aggregate level of policy analysis. The integration of physical and economic models reveals that, in general, the effect of technological or policy changes on environmental quality will depend on key physical and economic parameters. Considering the demanding data requirements of the integrated physical and economic analysis, a critical issue facing researchers is to identify minimal information sets needed to accurately estimate physical and economic parameters.

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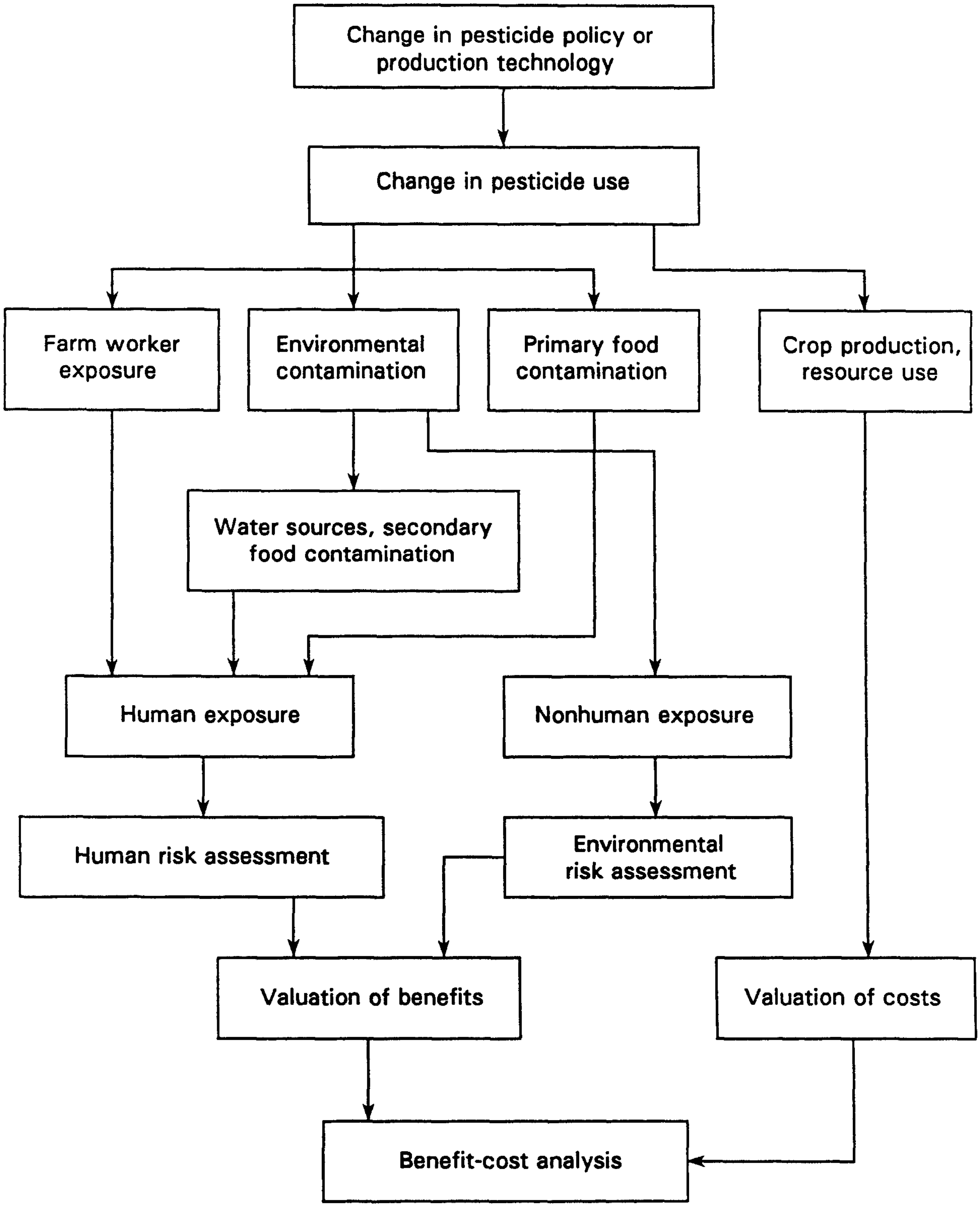

Figure 1. Major components of a benefit-cost analysis of a change in pesticide use.

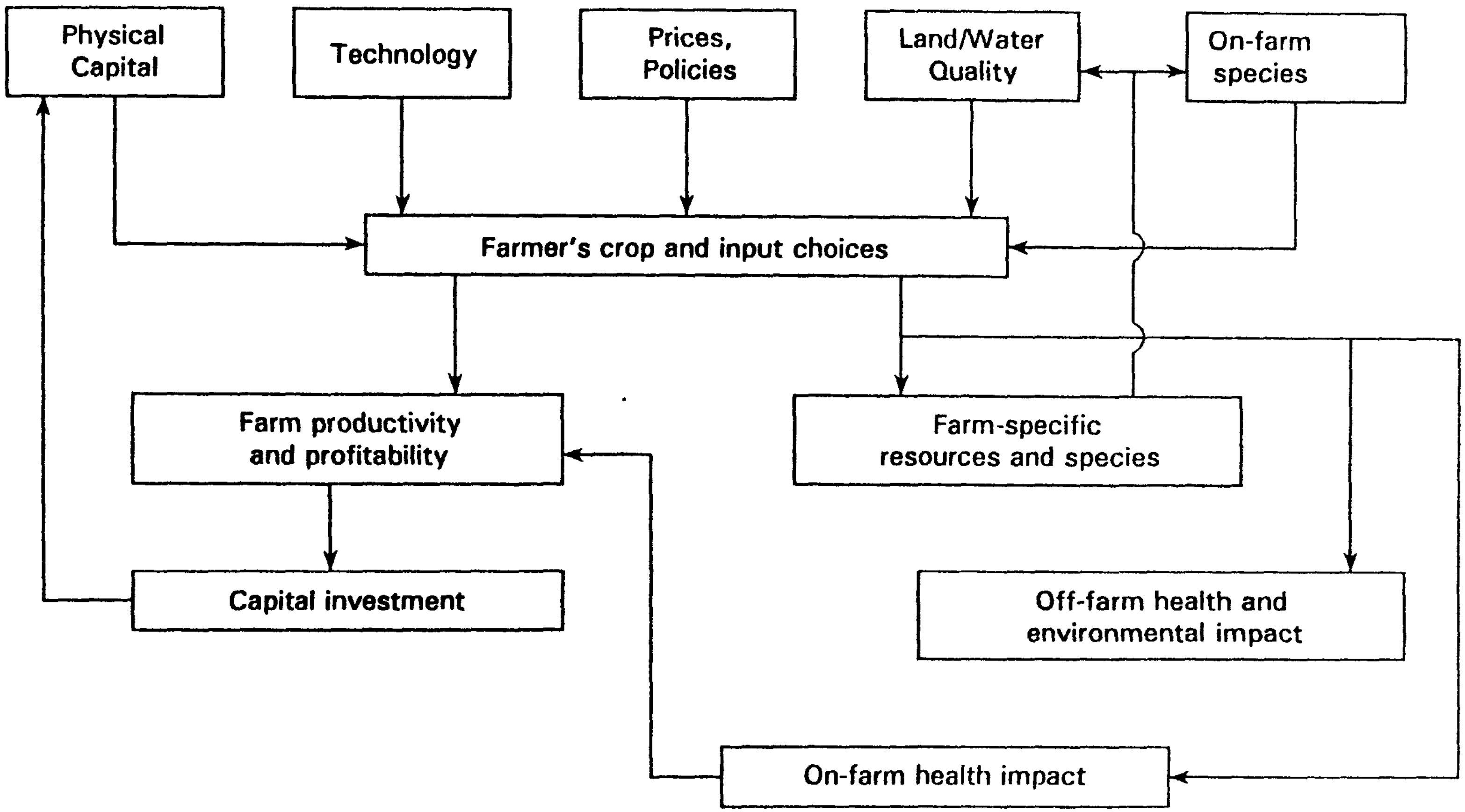

Figure 2. Production model linkages to on-farm resources and off-farm environmental and health impacts.

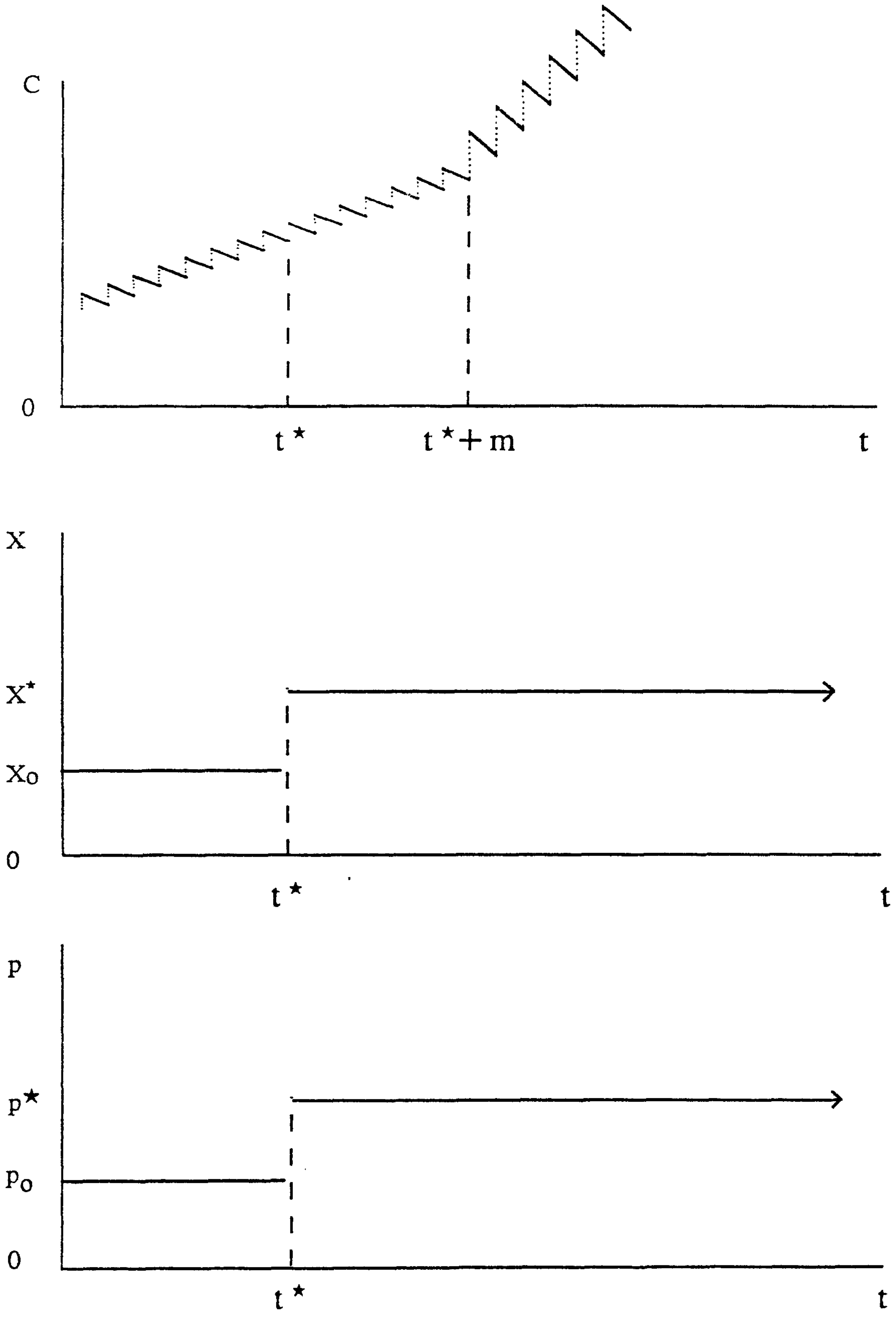

Figure 3. Time paths of output price (p), input use (X), and groundwater contamination (C) with a once-and-for-all change in price policy.

9

DATA REQUIREMENTS FOR MODELING AND EVALUATING NATIONAL POLICIES AIMED AT CONTROLLING AGRICULTURAL SOURCES OF NONPOINT WATER POLLUTION

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Since the inception of Federal water quality legislation (P.L.92-500, 1972), the aggregate control of nonpoint source pollution has been ineffective (GAO). Now several new national policy initiatives are in design or early implementation stages. States. are beginning to implement Section 319 programs under the 1987 Clean Water Act. The 1990 Coastal Zone Management Act mandates programs to reduce nonpoint pollution to coastal waters and authorizes regulatory approaches. Federal agencies are implementing the President's Water Quality Initiative to control largely nonpoint sources of agricultural chemicals leaching to groundwaters. And, just looming on the horizon is reauthorization of the Clean Water Act in 1992.

Despite incomplete theory and data, policy makers responsible for design and implementation of these national programs need analyses that cover the range of pollution conditions and potential economic effects. The challenge to economic researchers is to provide meaningful insights to the national policy process in the face of considerable scientific uncertainty. An aggregate evaluation of national policy alternatives should ideally possess several key features not always common to micro studies: endogenous prices, endogenous Federal program effects (e.g., agricultural commodity program participation), endogenous technology responses from the private and public sectors, regional tradeoffs in policy design, and complete government cost accounting. Ideally, these effects should be derived from a proper statistical aggregation as outlined by Antle and Capalbo, and by Opaluch and Segerson. But a comprehensive national data base necessary to perform such a sampling does not exist, and is unlikely to be built in the near future given budget constraints.

This paper examines the information requirements for modeling and evaluation of national nonpoint source pollution policies given scientific and data constraints. The planned economic evaluation of policies under the President's Water Quality Initiative1/ is used to illustrate the necessary analytical process. First, the basic policy-relevant questions guiding the data and modeling analyses are explored in some detail. Then, a preliminary modeling approach and data collection effort to address the static, short-run economic questions are described, including general model formulation. Possible empirical approaches and associated problems are presented. Future research priorities to enhance the policy relevance of economic analyses, such as induced technological change, are outlined at the close.

Focus of National Analysis

Three basic questions can be used to guide the economic investigations of national nonpoint source policies:

- What are the static and dynamic input and output changes from the policy initiatives?
- How do the input and output shifts map onto the natural resource base to produce positive or negative environmental effects?

1/ The Initiative is comprised of Federal programs of voluntary education, technical assistance and limited subsidies to achieve management practice changes that reduce potential agricultural chemical loadings, plus research and development programs to develop new technologies. Anticipated Federal expenditures are in the $400 - $500 million range over 1991-95. An evaluation of Initiative programs in comparison to alternative policies, such as regulation, is being directed the Economic Research Service.

- What are the economic costs to the private and public sectors, including program administrative expenses?

Input and Output Adjustments

Programs to control agricultural sources of nonpoint water pollution are designed to induce static shifts of inputs and outputs over space, and dynamic changes in production technology with positive environmental consequences. The desired end product is a series of static and dynamic input and output substitutions to reduce pollutant loadings to water resources. Ex ante modeling of the likely changes under alternative national policy approaches requires a clear delineation of many possible effects.

Perhaps the simplest starting point is short-run production behavior under profit maximization conditions (i.e., assuming a fixed total land base and technology). Potential substitutions of interest can be illustrated with a simple multiple input and output production relation (in notation consistent with Antle and Capalbo).

$$T(Q, X, Z, R, S, \tau) = 0 \tag{1}$$

Q is a vector of maximum rates of outputs with variable inputs X measuring labor, management, fertilizer, pesticides, etc., Z is a vector of fixed capital inputs including the total land base, structures, etc. , and parameter τ represents the state of technology. The production influences of physical resources, such as land and water qualities, are captured by R and biological organisms by S. The effects of input and output choices on R and S affect future production conditions and provide a dynamic production-environmental linkage. The production relation encompasses both intensive and extensive margin changes. How the inputs and outputs are jointly distributed over the environmental base then determines the nonpoint water quality consequences through time as Antle and Capalbo show.

Nonpoint water policies, such as the President's Water Quality Initiative, are mostly action program efforts including subsidized education and technical assistance (i.e, information) plus financial subsidies to shift the combinations of Q and X over space and time. Examples of these short-run (i.e., constant technology base) changes include reductions of leachable herbicides in favor of more management or mechanical tillage, and shifts in crop rotations to reduce nitrogen applications under highly leachable conditions. Estimating input and output substitutions thus becomes an important analytical focus.

The definition of appropriate input classes for estimating elasticities of substitution is a troublesome issue. For aggregate analysis, the input sets must be parsimonious. From an economic behavioral perspective, the classes should be substitutes in the decision-maker's mind. But to link the input changes meaningfully to the environmental relations, an economic class (e.g., corn herbicides) may need to be differentiated by leachability or half-life considerations. Obviously, a tractable aggregate analysis can not capture the full range of substitutions on all crops but should reflect the essential economic choices with basic environmental differences.

An important consideration to short-run output substitutions in agriculture is the role of Federal commodity programs. Through a system of program crop bases and differential deficiency (subsidy) rates plus acreage diversion requirements, the commodity programs bias the selection of crops relative to market conditions that might exist without commodity programs. Analyses of water quality policy effects on output choices must incorporate the roles of commodity programs and the potential competitive or complementary effects with nonpoint source control programs.

The discussion to this point has focused on the short-run economic effects, but nonpoint policies will occur in a dynamic, long–term context. Therefore, the values of Z and τ will vary from their fixed short run levels. And, the physical and biological variables, R and S, will change their temporal paths. In essence, the fixed factors, such as machinery types, and the production technology will likely change in response to private and public investment changes induced by alternative water quality policies. Antle and Capalbo explain the conceptual differences in time paths for the economic, physical and biological variables under short-run and longer-term, dynamic optimizations. For example, the President's Initiative will invest in excess of one hundred million dollars to develop new technologies by public research agencies. Special attention needs to be paid to changes in relative factor prices caused by regulation, subsidies, or taxes that induce technological innovation. Both private and public sector research and development will likely be affected. Because little information exists to characterize these longer-term economic processes, that analysis poses data and modeling challenges as explored at the end.

Environmental Effects

Estimating how the input and output changes occur over natural resource conditions is necessary to predict the potential water quality effects. This estimation process is tractable at the firm or even watershed level, but becomes very complex when considering regional or national aggregate responses.

Opaluch and Segerson outline a conceptual procedure to join microparameter models (Antle and Just; Just and Antle) with geographic information systems (GIS) to characterize the potential water quality effects induced by an aggregate policy action. In brief, the process involves three basic steps:

1. Determining the water quality pollution potential of a microunit (e.g., field or farm)
2. Applying the microparameter model to characterize the extensive and intensive margin changes on the microunit due to the policy.
3. Determine the spatial distribution of environmental responses to reflect aggregate impacts on water resource units of interest (e.g., regional aquifers).

The authors note three potential problems with application of the linked microparameter - GIS modeling system. First, the microunit of analysis for the microparameter model and GIS must be reconciled. In most cases the appropriate decision unit for the microparameter model is smaller than available GIS data. Second, the microparameter models predict the response of a representative farm with certain characteristics but not the particular farm in a GIS cell. This problem can be lessened by aggregating the microparameter model results to a level (e.g., county) consistent with the GIS cell. Finally, available GIS data technology may necessitate a larger microunit (e.g., collection of farms) for analysis, but at the cost of sacrificing natural resource diversity affecting the specific nature of nonpoint water quality conditions. Despite the potential problems, the general approach appears to be the only feasible method at present of aggregating environmental responses for regional and national analyses.

Even a successful implementation of the linked microparameter - GIS model approach leaves two possibly important deficiencies in the environmental effects assessment. The methodology described by Opaluch and Segerson is largely short-run, static for both economic and environmental effects. Where longer-term, dynamic processes are important to nonpoint water quality policy responses, the microparameter and pollution potential algorithms should be altered to capture those effects. Second, the spatial and possibly temporal environmental responses are expressed in physical units rather than a common money metric. Thus aggregation of potential environmental benefits to regions of the nation are not possible due to incomplete science and data on fate-transport relationships and willingness to pay information.

Economic Costs

National policy makers are keenly interested in the economic costs of alternative water quality policies, both private producer and consumer welfare changes, and net public government expense impacts. Indeed, the government cost component has received increasing weight of late due to the large and continuing budget deficit. So credible estimates of the short-run and longer-term paths of economic and government cost components are critical to a national policy evaluation.

A bottom-up statistical aggregation to a *national* level of microunit cost supply responses using the microparameter model is impossible given current databases. Therefore the aggregate analysis of economic costs must necessarily proceed with large national models without explicit natural resource linkages. Such an approach introduces the possibility of inconsistent microparameter and aggregate estimates due to different model formulations. One approach to reduce inconsistencies is to use results from the micro level analyses to condition the aggregate modeling procedure. An example is to use the range of estimated elasticities of input substitution from the micro analyses to bound the regional responses induced by agricultural nonpoint water quality policies.

A short-run economic cost analysis requires the incorporation of several important factors. First, the effects of cost and supply changes on crop and livestock prices must be estimated including international trade impacts (i.e., output price endogeneity). The second round price repercussions of a national policy may complement or offset first round effects on microunits. Second, the analyses must permit static input and output substitution between all relevant factors of production and commodities to capture intensive and extensive margin changes under existing technologies. Third, the influences of existing and anticipated Federal agricultural commodity and conservation programs on inputs and outputs should be incorporated. For example, the effects of land diversions under the commodity program acreage set asides and with the Conservation Reserve Program will likely increase land prices and cause farmers to substitute non-land inputs such as chemicals (Offutt and Shoemaker). Finally, the cost analysis should capture the expected changes in government expenses, including water quality policy administrative costs and commodity program savings from reduced supplies and increased market prices.

The more challenging task is to extend the economic cost analysis to the longer-term. Two factors are critical to developing estimates of long-run economic adjustments. The changes in the fixed capital base to accommodate water quality programs are relevant. An example is a switch to more efficient irrigation equipment to increase use efficiency and reduce excess runoff and percolation. Induced technology diffusion and change as a result of water quality policies and/or changes in relative factor prices may be the most critical long run component. Ex ante economic analyses of policy impacts often greatly exaggerate the ultimate industry and economy wide impacts due to the static capital and technology assumptions. Longer-term elasticities of substitution for inputs affecting water quality are necessary to estimate the ultimate economic cost path.

Aggregate Modeling Framework

A full articulation of the relevant questions is a necessary first step in the national analysis. Unfortunately our ability to ask policy relevant questions is not matched by our capacity to capture those effects with available data and empirical methods. Nonetheless, a specification of a general aggregate modeling system is necessary to gain insight about how the feasible analytical approach differs from the ideal conceptual methodology. The modeling framework to follow focuses primarily on the economic input and output adjustments conditioned by resource characteristics and leaves aggregate environmental effects to research challenges discussed at the conclusion. The role of special data collection efforts, termed “Area Studies”, to enable the aggregate modeling analyses is then discussed.

To summarize, the key challenge of our research is to examine the relationships between the natural resource base and production activities for national policies. That is, how do different resource characteristics affect production decisions, and given those resource characteristics, how do production choices affect environmental attributes associated with those resources?

To formalize these questions, we present a general model to provide a conceptual basis for analysis. In what follows we describe a static general producer optimization problem and the associated loadings of pollution conditioned on regionally specific resource characteristics. A microeconomic model is developed retaining the essential microparameter concepts where individual producers face parametric prices and endogenous commodity program participation. Firms are then aggregated based on regional distributions of resource characteristics to market level commodity supplies and factor demands. Factor supplies are assumed to be perfectly elastic but commodity supplies face market level demand curves thus endogenizing commodity prices.

We allow the firm to be characterized as a multiproduct firm employing several inputs and producing several outputs to keep the analysis as general as possible. To restate equation (1), assume production by the j^{th} firm is determined by a transformation function represented as,

(2) $$T^j(Q, X, Z, R, \tau) = 0$$

where Q is a vector of outputs, X is a vector of variable inputs, Z is a vector of fixed factors, R is a vector of resource characteristics that contribute to production and pollution and τ is an index representing a particular technology.2/ It is the elements of R and τ that define input, output and resource linkages that are critical for the analysis. For example, if the firm is located in a dry climate on a sandy soil, it is possible that the firm will use a technology involving irrigation. The potential environmental damages derived under these conditions is entirely different from what might occur in a more moist temperate climate on a clay soil.

The above arguments make clear that a pollution loading function is also a function of the same arguments. That is, potential loadings will be a function of the outputs produced, inputs used, and R and τ. The pollution loadings function is expressed as,

(3) $$h^j = h^j(Q, X, Z, R, \tau)$$

Firms are assumed to be profit maximizers, and assuming the transformation

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2/ For ease of exposition, the physical and biological resource characteristics are collapsed into one vector, R, for the general analysis. The resource vectors should be divided into classes to capture the essential economic and environmental dimensions of the problem under study. Opaluch and Segerson suggest a three way classification resource characteristics, i.e., those affecting production only, production and pollution, and pollution only.

function obeys the usual properties, a profit function (abstracting from government programs) can be defined as,

$$(4) \qquad \pi(p, w, Z, R, \tau) = \underset{Q,X}{Max}\{p{\cdot}Q - w{\cdot}X : T(Q, X, Z, R, \tau)\}$$

where p and w are the output and input prices. Maximal profits and the envelope conditions yield optimal input demands and output supplies as the respective gradient vectors,

$$(5) \qquad X_j^* = \nabla_w \pi(p, w, Z, R, \tau)$$

$$(6) \qquad Q_j^* = \nabla_p \pi(p, w, Z, R, \tau)$$

The pollution loading associated with the optimal inputs, X_j' and supplies, Q_j' for the j^{th} firm is,

$$(7) \qquad h^j = h^j(Q^*(p, w, Z, R, \tau), X^*(p, w, Z, R, \tau), Z, R, \tau)$$

Loadings are indexed to the j^{th} firm to emphasize the point that loadings are specific to firm activity levels and the firm's resource characteristics. 3/

Commodity Program Participation

Output decisions and factor demands are affected by participation in commodity programs. The production incentives derived from support prices and requirements for program participation affect relative factor demands at the

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3/ Indexing the H(.) functions by j is not meant to imply the functions differ over firms, rather it merely implies that there are multiple firms. This assumption could be relaxed if we treated τ as a random variable and then integrated over τ.

intensive margin and commodity supplies at the extensive margin. Producers choose to participate in programs based on the relative benefits and costs of program participation conditioned on their costs of production. That is, a high cost producer will more likely enter the program than low cost producers. The relative costs of production among producers are in part determined by the distribution of resource characteristics. Therefore we define a subset of the vector R to include variables that contribute directly to the productivity of firms and their ability to earn net returns. 4/ We define ω to be a variable that determines productivity which spans the range $[0,\bar{\omega}]$, where $\bar{\omega}$ is the upper value of ω. Given market prices and program parameters there is a critical value, denoted $\underline{\omega}$, associated with net returns where producers begin to participate. Therefore, for values between $\underline{\omega}$ and $\bar{\omega}$, net returns are sufficiently low that producers will participate (given program parameters). 5/

To keep things simple, we present a stylized version of programs. Program parameters are limited to a target or support price, $\bar{p}$, the set-aside rate, θ and the program yield rate, $\bar{q}$. Program benefits are determined as the product of the difference between the support and market price, $(\bar{p}-p)$, times land net of the set-aside and the program yield rate, $(1-\theta)A\bar{q}$. Producers choose to participate given the maximum profit of,

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4/ Here we are making the assumption that we can distinguish resource characteristics associated with program participation from other characteristics. While this is done for analytical convenience, it remains an empirical issue whether this distinction can be made.

5/ Program participation behavior could be estimated using a dichotomous choice model. Models of this sort have treated variables such as ω as unobserved. Within the current context, the variable may actually be observed.

$$\pi = \begin{cases} p \cdot Q - w \cdot X \ : \ T(Q, X, Z, R, \omega, \tau) & \text{out} \\ p \cdot Q + (\bar{p} - p) \cdot (1-\theta) \cdot A \cdot \bar{q} - w \cdot X \ : \ T(Q, X, (1-\theta)A, Z, R, \omega, \tau) & \text{in} \end{cases} \tag{8}$$

where land has been identified separately from the vector of fixed factors, Z and is denoted A. “Out” refers to producers out of the program and “in” refers to those that are in the program. The cost of participation is the opportunity cost of setting aside land. The resulting profit functions are,

$$\pi = \begin{cases} \pi(p, w; Z, R, \omega, \tau) & \text{out} \\ \pi(p, w; \bar{p}, \theta, \bar{q}, Z, R, \omega, \tau) & \text{in} \end{cases} \tag{9}$$

Aggregation

Aggregate or total industry supply and factor demands are found by integrating over the distribution of resource characteristics, R and ω.

$$Q = \int_0^{\underline{\omega}}\!\!\int Q^*(p, w; Z, R, \omega, \tau)\, dR\, d\omega + \int_{\underline{\omega}}^{\bar{\omega}}\!\!\int Q^*(p, w; \bar{p}, \theta, \bar{q}, Z, R, \omega, \tau)\, dR\, d\omega \tag{10}$$

$$X = \int_0^{\underline{\omega}}\!\!\int X^*(p, w; Z, R, \omega, \tau)\, dR\, d\omega + \int_{\underline{\omega}}^{\bar{\omega}}\!\!\int X^*(p, w; \bar{p}, \theta, \bar{q}, Z, R, \omega, \tau)\, dR\, d\omega \tag{11}$$

Total pollution loadings are found similarly as, 6/

(12) $$H = \iint h(Q^i(\cdot), X^i(\cdot), Z, R, \omega, \tau)\, dR\, d\omega$$

where i indexes participants and nonparticipant. By totally differentiating equation (12) we can determine the information requirements necessary for modeling changes in aggregate pollutant loadings. The change in total loadings is expressed as,

(13)

$$\begin{aligned} dH = \iint \Bigg\{ & \frac{\partial h}{\partial Q^i}\left[\frac{\partial Q^i}{\partial p}dp + \frac{\partial Q^i}{\partial w}dw + \frac{\partial Q^i}{\partial \psi}d\psi + \frac{\partial Q^i}{\partial Z}dZ + \frac{\partial Q^i}{\partial R}dR + \frac{\partial Q^i}{\partial \omega}d\omega + \frac{\partial Q^i}{\partial \tau}d\tau\right] \\ & + \frac{\partial h}{\partial X^i}\left[\frac{\partial X^i}{\partial p}dp + \frac{\partial X^i}{\partial w}dw + \frac{\partial X^i}{\partial \psi}d\psi + \frac{\partial X^i}{\partial Z}dZ + \frac{\partial X^i}{\partial R}dR + \frac{\partial X^i}{\partial \omega}d\omega + \frac{\partial X^i}{\partial \tau}d\tau\right] \\ & + \frac{\partial h}{\partial Z}dZ + \frac{\partial h}{\partial R}dR + \frac{\partial h}{\partial \omega}d\omega + \frac{\partial h}{\partial \tau}d\tau \Bigg\} dR\, d\omega \end{aligned}$$

where $\Psi = \{\bar{p}, \theta, \bar{q}\}$. Equation (13) suggests how elements such as resource characteristics and technology have direct and indirect influences on pollution loadings. If we express the above total differential in elasticity form, we can see that the parameters needed for evaluation are mostly standard

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6/ Of course, total pollution loading is an artificial construct in that pollution is defined by resource supply and demand. However, the aggregate pollution concept is useful to illustrate some aggregate production and environmental relationships of interest. Perhaps a reasonable example of this aggregate concept is the total leachable nitrates into groundwater aquifers.

producer behavioral parameters, i.e., supply and demand elasticities. We also assume that in the short run the resource and technology characteristics do not change, i.e., $dZ = dR = d\omega = d\tau = 0$.

The elasticity form is expressed as,

$$\hat{H} = \iint \{E_{hQ}[E_{Qp}\hat{p} + E_{Qw}\hat{w} + E_{Q\psi}\hat{\psi}] + E_{hX}[E_{Xp}\hat{p} + E_{Xw}\hat{w} + E_{X\psi}\hat{\psi}]\}dR\, d\omega \tag{14}$$

where E_{ij} is the elasticity of j with respect to i and the "^" denotes percent change. From equation (14) we see that in order to determine the change in pollution loadings, (in the certainty case) given assumed changes in input and output prices, there are three basic (hard-to-come-by) types of information required. The first type includes the elasticities of demand and supply with respect to input and output prices and policy parameters (participation rates). The second information requirement is the distribution of resource characteristics over the production space. Finally, knowledge of the fate and transport properties of various chemical inputs and soil profiles is needed to understand the relationships between inputs, outputs and loadings. The first information requirement represents an activity that economists have expertise in, and the second represents an important data collection exercise discussed below. The third requires knowledge of hydrology and geology, an area in which economists do not have a comparative advantage. Furthermore, the science that is developing in this area is generally limited to very small units of analysis, (e.g., fields or subfields) units that are below a relevant scope for policy analysis and also pose the problem that aggregation is impossible given present databases. Yet economists must work with physical scientist to insure that the fate and transport data are integrable with the economic analyses. For the present analysis, we limit our attention to agricultural production embodied in equations (10) and (11).

Equations (10) and (11) can be used to develop an aggregate economic model. Some of the key variables of interest are commodity supplies and prices and factor demands. Additional economic indicators are net income or rents and government outlays. An algebraic schematic of the model is presented in the appendix. The simulation model utilizes the aggregation methodology of Johansen (1972) and Hochman and Zilberman (1978). This aggregate model is presented to highlight the data needs for this kind of empirical analysis and to set the stage for a discussion of some data activities underway at the Economic Research Service.

The model could work as follows. We can think of the aggregation of firms as a grid, e.g. a collection of Major Land Resource Areas (MLRAs) or Area Studies (to be discussed below), or some other geographically defined regions. The acres of farmland in each region represent a percent or share weight of the total farmland. Within each region there is a distribution of resource characteristics. The distribution of resource characteristics condition commodity supplies, program participation and factor demands. Factor supplies are assumed to be perfectly elastic, therefore factor prices are treated as exogenous. Commodity supplies are aggregated according to their weights in each region and then are aggregated across regions according to the regional weights of the total acres of farmland. The model is closed with an aggregate commodity demand function which endogenizes prices. Commodity program participation could be modeled with a dichotomous choice model based on relative returns again conditioned on resource characteristics in the set ω. Given program participation, an accounting identity can be defined that determines government outlays.

Data Needs

The above model description highlights the data needs for this kind of analysis. The data can be categorized into two broad classes: (1) production data, e.g., input and output prices and quantities, and (2) resource characteristics. While there is certainly nothing unique about the first requirement for economic analysis, it is the scale of analysis combined with the second requirement that makes the data issues more demanding than usual.

Area Studies Project

USDA's Area Study project is, in part, a data collection effort designed to provide micro-level information on the relationship between agricultural production activities and characteristics of the resource base, information that is required by the model presented above. It is a practical matter that resources are not available to collect data on the full scope of agricultural production and natural resource conditions necessary to represent all categories of water quality problems related to agriculture. USDA is approaching the problem by selecting a set of "evaluation sites" such that the most important agricultural production and water quality combinations are covered. Not all contributions can be included because of limited data collection resources. Emphasis is placed on major field crops, such as corn, soybeans, and wheat, which rely heavily on chemical applications and cover broad geographical areas.

Specific objectives of the data collection component of the project are to:

1) Provide chemical use and farming practice information for selected National Water Quality Assessment (NAWQA) study sites to aid in understanding the relationship between farming activities and ground water quality for a variety of agroecological settings.

2) Sample a wide range of farming practices and resource characteristics using a consistent approach to provide for cross-comparisons and a comprehensive analysis of the national impacts of alternative policies.

A total of twelve Area Study sites will be investigated. Four areas have been selected for study in 1991—the Central Nebraska Basin, the White River (Indiana), the Lower Susquehanna Basin (Pennsylvania), and the Mid-Columbia Basin (Washington). Four new sites will be selected for study in 1992, and another four in 1993. Each of these areas corresponds to a USGS study site in the National Water Quality Assessment Study. This coincidence of study sites insures that analysis of the fate-transport aspects will be studied.

At each site, a chemical-use and farming-practice questionnaire will be administered to approximately 1000 farm operators. The location of the operator will coincide with a National Resource Inventory (NRI) sampling point. (The Soil Conservation Service conducts a National Resources Inventory every five years. The Inventory will be done again in 1992.) The NRI is based on a stratified random sampling design in which soil, water, and related natural resource data are collected at nearly a million sample sites. Choosing the sample so that it coincides with a NRI point insures that important information on soil properties will be available, and also provides a statistical basis for aggregation within the region.

The questionnaire will solicit information specific to the field associated with the NRI point and also for the whole-farm operation. Sufficient field-level data will be collected to describe in detail the cropping system used at the NRI sampling point (crop type, tillage practice, rotation scheme, chemical use, non-chemical pest control, etc.). More general whole-farm questions will be asked on acres planted by crop, chemical use by crop, general tillage practices used on the farm, and the size and type of livestock operation. Economic questions related to the whole-farm operation will also be asked to support development of economic models (such as the value of land, labor, and capital available to the operator and participation in government programs).

Possible Empirical Applications

The aggregate conceptual model described above requires bottom up statistical aggregation of the microparameter models. But the area studies data collection effort will fall short of the necessary coverage to perform that statistical aggregation for the nation as a whole. Two empirical approaches are possible recognizing the incompleteness of coverage.

Area study data can be used to estimate producer behavioral response functions (e.g., restricted profit functions, input demand, output supply) conditioned on the resource base. These area-specific supply and input demand functions would then describe an area-wide farm. A special challenge will be to estimate input and output substitution relationships with minimal cross-sectional input and output price variation. Given knowledge of the area study samples regarding input, output and resource relationships, the results could be extrapolated through application to other NRI points nationwide that match closest with the output and resource conditions studied. Such a procedure falls short of a proper statistical aggregation as outlined by Opaluch and Segerson on two counts. First, the area study models assume that firm-level behavior in relation to resource conditions can be approximated with one (or possibly two) field observations. Second, the extrapolation of estimated area study results to other areas based on output-resource matchings ignores possible technology variations across regions (e.g., fertilizer and pesticide practices).

The second approach is to capture essential aggregate and area-level production and environmental details in separate but linked analyses. The procedure would begin with the use of an aggregate (national) model of agricultural production and input use divided into major regions (e.g., collections of states). While the aggregate model is consistent with the above microeconomic-based analytical model, it does not include the explicit influences of the natural resource base. Important features of the aggregate model include price endogeneity, commodity program participation, output substitution and input substitution (relevant to water quality analyses). One candidate for the analyses is the US Agricultural Resources Model (USARM) (Konyar and McCormick). The USARM model does not have explicit natural resource detail since it uses aggregate regional production and cost responses. The area studies could be used to specify important input and output substitution relationships to provide some consistency between the aggregate and area study levels. In the second stage, the aggregate price shocks induced by policy shifts are entered into the area–level models along with other policy parameters (e.g., chemical restrictions) to simulate the net effects on output and input use in relation to the natural resource base. This second approach allows the area studies be separate investigations, but uses scientific insight from the survey analyses as both inputs to the aggregate model and as a mechanism to simulate aggregate level policy shocks. Extrapolation of the area study simulations to other regions based on common NRI output-resource pairings could proceed as in the first approach to estimate aggregate pollutant loadings and environmental shifts.

Future Research Priorities

The data and modeling approaches outlined are essential first steps mostly focused on the short-run economies, but do not cover longer-term or environmental issues. Areas for further investigation include induced technological change, fixed inputs, environmental effects, and government program expenses.

Technical Change

Economists recognize the critical and often complex roles of technology in resource and environmental management. Analyzing the impacts of environmental policy with a fixed technology set is rarely sufficient. The induced innovations literature has documented the role of relative factor prices in generating technology development and adoption (Hayami and Ruttan). Incorporating effective factor prices for non-market environmental services through public programs of subsidies, taxes, and/or regulation will likely induce technology shifts changing the longer term economic and environmental effects. Moreover, reform of commodity programs will likely change the technology stream. Two activities are planned to help incorporate the technical change influences. Studies of other environmental management programs will be consulted to determine if generalizations about technology response can be made for application to nonpoint water quality issues. Second, a Delphi technology assessment exercise will be conducted by interviewing public and private experts regarding emerging technologies relevant to nonpoint source control. Estimates of technical (input and output) performance, economics and environmental parameters will be obtained. Information from either source can be used to adjust input and output substitution relationships in the aggregate and area study models.

Fixed Inputs

Another dynamic process is the change in the short-run capital stock over time due to water quality policies. Examples include changes in pesticide or fertilizer application machinery and irrigation equipment. Antle and Capalbo present a long-run dynamic investment model wherein the farm chooses the sequence of investments to maximize present value of profit over the planning horizon. Conceptually, shifts in the fixed capital inputs change the parameter Z in the transformation (eq.2) and pollutant loadings (eq.3) functions which affects input demands, supply functions, economic costs, etc. Estimating Z endogenously requires knowledge of the investment demand structural equation and how that equation shifts in response to water quality policies.

Environmental Effects

Describing the impacts of national nonpoint policies on environmental resources may be the greatest challenge. As discussed, the area studies will be conducted in concert with USGS scientists to enrich the fate-transport analyses. It is unlikely that definitive information on the water quality effects of reduced chemical use will be available within the next decade.

Environmental process models can be used to describe changes in pollutant loadings at various points in the soil profile due to input-output shifts by water quality policies. Use of the NRI sampling points for the area studies provides critical physical resource information for the process models, including soils data, precipitation, and other variables. When these data are joined to estimated input and output changes from the area study behavioral models, then geographical summarization of the pollutant loadings can proceed along the lines advanced by Opaluch and Segerson. The estimation process would describe comparative static outcomes but not the dynamic path of pollutant change.

Valuation of the environmental effects is equally problematic. Given uncertain fate-transport knowledge and virtually no epidemiological data, objective exposure and health effects modeling is not feasible. Two approaches will be explored. First, for those water systems estimated to exceed maximum acceptable contaminant levels by survey data or process model extrapolations, the cost of obtaining alternative water supplies can be calculated as a minimum bound. The second approach is to elicit willingness to pay estimates through contingent valuation exercises.

<u>Government Program Costs</u>

With few exceptions, most studies of environmental policies ignore the roles and magnitudes of public expenditures. Though the costs are often transfers, their influence on decision making is important. For the President's Water Quality Initiative based on large scale eduction and technical assistance programs, government expenditures will total hundreds of millions of dollars. With a continuing Federal deficit problem, the minimizaiton of those expenses is an important objective. Estimates of the program costs will be assembled based on experience in demonstration and special water quality projects conducted under the Initiative. Estimates for other water quality policies in comparison to Initiative programs will be made based on Federal or State environmental policy experience or engineering projections.

Concluding Note

The evaluation of national water quality policies poses some very special data and modeling problems. Survey funds are not available to do comprehensive data collection consistent with theoretically-based microparameter models for a bottom-up aggregation to a national level. However, it appears possible and desirable to incorporate some micro-level detail, especially on production-resource economic and environmental linkages, into the aggregate framework. Longer-term issues of incorporating technological change, capital stock changes, and portraying aggregate environmental effects are important research agenda items.

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Appendix

The following is a skeleton representation of an aggregate model of agricultural production spatially distributed over various resource characteristics and commodity program participants.

(A1) $$Q^S = \iint [\Omega Q^{in}(\cdot) + (1-\Omega) \cdot Q^{out}]\, dR\, d\omega$$

(A2) $$\Omega = g(p, w, \overline{p}, \theta, \overline{q}, \omega)$$

(A3) $$Q^D = D(p)$$

(A4) $$X^D = \iint [\Omega X^{in}(\cdot) + (1-\Omega) X^{out}]\, dR\, d\omega$$

(A5) $$Q^S = Q^D$$

(A6) $$G = \iint [\overline{q}(\overline{p}-p)(1-\theta) A^{in}(\omega) + WQT]\, d\omega$$

Equation (A1) is total commodity supply integrated over all resource types and commodity program participants and nonparticiapnts. Equation (A2) is a dichotomous choice function which determines the commodity program participation rate, Ω. Aggregate commodity demand is represented by equation (A3). Total factor demands are represented by equation (A4), again weighted by participants and nonparticipants. Equation (A5) defines market equilibrium. Total government outlays, including cost-sharing or other water–quality transfers (WQT) is calculated in equation (A6).

Regional Modeling and Economic Incentives to Control Drainage Pollution

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Regional Modeling and Economic Incentives to Control Drainage Pollution

Introduction

Management of quantity and quality of irrigation water--both as an on-farm production input and as an off-farm agricultural drainage residual--is an increasing concern in many parts of the world, including the arid western United States. Agricultural drainage water often carries salts, pesticides, nitrates, selenium, and other trace elements that pollute soils, surface water resources, and aquifers. As a non-point source of pollution, agricultural drainage water directly and indirectly affects agricultural productivity, wildlife, public health, and amenity resources. In addition to the quality aspects, strong competition exist for water among urban, industrial, environmental, and agricultural users in western United States. Water conservation in irrigated agriculture may achieve the dual goal of extending fresh water supplies and improving environmental quality.

Identifying solutions to the irrigation water quantity/quality problem involves two challenges. One challenge concerns the complexity of modeling the relevant physical and biological systems and their relationships to economic decisionmaking. These systems include both spatial and dynamic dimensions. Economic decisions involve private decisions, such as investment in irrigation technologies and land use on the farm, and collective decisions, such as the optimal sizing, siting, and timing of joint treatment facility for drainage water at the regional level. The second challenge concerns the design of an economic incentive system that will, simultaneously, provide socially efficient solutions and be acceptable to all interested parties.

The economic literature on irrigation water quantity/quality problems has expanded tremendously in recent years. It includes feasibility studies of technologies to reduce pollution as well as modeling and policy analysis.

Various management strategies to limit irrigation-induced water quality problems are being evaluated at the field, farm and regional levels in the San Joaquin Valley (SJV) of California. Improved agricultural management to reduce the quantity of drainage for disposal may involve cropping pattern adjustments, changes in water application rates for a given crop-technology, adoption of water conserving technologies, and management practices, and adjustment of irrigated acreage base (SJVDP, 1990).

The use of evaporation ponds, which under certain conditions may reduce environmental damages through reduced drainage disposed directly to the environment has been evaluated by Ford (1988). Dilution of drainage water with freshwater prior to disposal was analyzed by Stroh (1991), and reuse of agricultural drainage water has been examined by Rhoades and Dinar (1991). Biological and chemical treatment of drainage water for selenium has also been considered by Stroh (1991).

Currently, treatment procedures involving evaporation ponds, dilution, and chemical/biological treatment have been technically evaluated in many places. Findings suggest the existence of economies of scale in the construction and operation of treatment facilities (see also Klemetson and Grenney, 1975; Ergas et al., 1990; CH2M HILL, 1986; Hanna and Kipps, 1990; Gerhardt and Oswald, 1990).

An understanding of the effects of irrigated agriculture on soil and water resources is essential to an appropriate economic analysis of alternative. However, modeling the relationships between agricultural activity and the physical environment in which agriculture occurs is very complex.

To demonstrate the complexity of the modeling task, Figure 1 provides a scheme of applied water and drainage relationships. These relationships involve multi-space and time dimensions, and third party effects. Figure 2 presents the area of drainage related problems in the SJV. Is one physical model appropriate to address this issue for an area of over almost 3 million acres? Of course not. Wide variability in physical conditions suggests that agricultural effects on the environment may vary substantially. Figure 3 highlights this variability for two locations in the SJV: the position of the geological formations, soil type, depth and thickness of the corcoran clay, land slope, depth to a confined aquifer, distance to a river for drainage disposal, and many other factors prevent us from relying on one general physical model to adequately address the problem. Even if one could model these relationships accurately, there are still concerns availability of substantial resources required for data development and processing of simulations.

Policymakers depend on the research community to provide them with models that areas relevant as possible, as well as tractable and manageable. This often requires a simplification of underlying physical relationships. The following models while less exhaustive in scope and detail then some earlier efforts, may serve as useful tools for policy analysis addressing drainage and related problems in a specific region of the SJV.

Modeling efforts have included both site-specific models based on detailed empirical relationships estimated from local information and more general models that can be calibrated for local conditions. The former approach often involves development of robust multidisciplinary models that include all possible components (agricultural-hydrological-economic). One example is the attempt by the **SJVDP**[1] to develop the Westside Agricultural Drainage Economic (WADE) model. The model was supposed to simulate policy effects on physical variables and economic behavior of farm operators in 181 "cells" in the SJV, but was heavily dependent on local data which therefore limited its application (Hatchett et al., 1991; Imhoff, 1991). Another example involves a simulation model developed by Gates and Grismer (1989) that must be run on a super computer.

A second approach is to develop relatively simple models that address important aspects of the problem to analyze limited policy scenarios. The success and usefulness of such models for policy analysis is dependent on the ability of their developers to identify and model the essential problem components. An example is the model in Caswell et al. (1990) that consists of a limited number of state variables and contains simple relationships.

Given the complexities of modeling a physical/economic system, and the data and software limitations, a third approach might involve the use of several models, each emphasizing a different aspect of the problem for a given location. Then, models can be combined for policy design and analysis purposes.

This paper provides an example of this combined approach. The paper proceeds as follows. First, physical modeling will be discussed in relation to alternative types of models. Then impact models and policy design models will be compared. In both cases, models are applied for conditions in the SJV. Technical and empirical data of agricultural activity and water pollution in the SJV are based on previous work (SJVDP, 1990; Swain, 1990; and sources cited in Dinar et al., 1991a,b,c) and so will not be extensively discussed here.

Modeling physical-economic relationships

Physical-economic models may have two purposes: impact analysis and policy design. In an impact analysis, a policy maker may wish to determine the effects of some policy such as tax on water quality, or quota on water quantity, or requiring that certain new technologies be adopted. Because of political considerations, effects of interest would include not only water

[1]**SJVDP** or San Joaquin Valley Drainage Program (1985-1990) was formed to address drainage and related contamination in the Kesterson Reservoir and other locations on the westside of the San Joaquin Valley

quantity and quality determinations, but also economic impacts such as costs and profits for producers who would pay taxes and/or adopt new technologies. In a policy design mode, a policy maker would want to know what would be the best type of tax (e.g., on water quantity or on water quality) or what type of technologies might be recommended. Obviously, for this second type of analysis, impacts would also be important to know, but they would be a part of an optimization problem.

In both cases, physical models are important. In the policy design case, mathematical equations describing the physical system become the constraints in an optimization model. In the impact case, the economic decisions of producers in response to potential policies determine physical effects.

Below, an example of each of these types is given. in the case of impact analysis, models are of two types: steady state and dynamic.

The impact models are used to assess profit maximization responses of farm operators under conditions of water scarcity, low input quality and externalities; and to evaluate incentive programs, taxes, and quantity-based restrictions as alternative methods of achieving policy objectives; The policy designmodel is used to design regional cooperation in water resource use, drainage reduction, and treatment to reduce pollution.

The steady state regional model of agricultural water use and drainage water quantity/quality is developed by integrating physical, biological and agronomic models for the region (Letey and Dinar, 1986) within an economic decisionmaking framework (Dinar et al., 1990: Dinar et al. 1991b). Efficiency of technical solutions is evaluated relative to urban and environmental constraints on water quality and quantity. The farm-level dynamic model is developed that considers the effect of present decisions on future outcomes. The model evaluates the effectiveness of water use technologies for irrigation, water quality mixing, drainage treatment, and other farming practices to meet water quantity and quality constraints, including the demand for water by competing sectors. The policy design framework for inducing regional cooperation uses physical-economic models to consider incentives and cost sharing schemes required for adoption of appropriate technologies, both at an individual producer and regional levels.

A steady state modeling frameword

Consider a region with a given number of farms, each having a limited homogeneous area of productive land and a limited amount of irrigation water supply (surface and ground water) with known salt concentrations. The farmers are served by a water district (from hereafter district) which has a long-term federal contract to receive a certain amount of surface water annually, for a given price. The farmers pay this base price plus a "district charge" to cover delivery, maintenance, and overhead costs, and costs of drainage treatment provided by the district.

A number of alternative crops can be grown on each farm and these can be irrigated with different combinations of water quantity and quality. Subsurface drain tiles have already been installed in farms where shallow ground water and drainage problems affect farming, Therefore, the installation of tiles will not be considered a decision variable. The district collects drain water from sumps on each farm for treatment and disposal. The disposal outlet is constrained in both total volume allowed and quality (salinity). The district may use part of its surface water allocation to dilute drainage in order to meet the quality constraint.

Several on-farm and district-wide management options to reduce the agricultural drain water quantity and/or quality will be evaluated here. Most of these options have been considered and described at field and farm levels (Knapp et al., 1986), and for regional planning purposes (SJVDP, 1990). They include reducing irrigation rates, changing cropping patterns, improving water application uniformity (management and equipment), and treating drainage water. Individual farmers and the district can select one or more of these options in response to policy measures.

The model presented here is a steady-state one. It is assumed that the optimal solution is found relatively early along the planning horizon and that once it is found, it will be followed by the farmers and the district for the entire time horizon. Therefore, it optimizes decision variables for only one year, including all long-term economic costs related to the agricultural production process.

The regional model is designed to maximize regional net income:

$$[1]\ \text{Max } R=\sum_i P_i^y \sum_j y_{ij}X_{ij}-\sum_j\sum_i M_{ij}X_{ij}-P^d\sum_j D_j\sum_j\sum_i K_{ij}$$
$$-\sum_j P_j^g\sum_i G_{ij}-\sum_j\sum_i P_{ij}^s S_{ij}-P^{ds}S^d-T_j\sum_j D_j$$

Here i is an index for crop and j is an index for farm. R is the regional net income, P^y_i is the crop price net of harvest and marketing cost, M_{ij} are the per unit area variable production costs net of water related cost, P^d is the cost of pumping the drainage water, P^g_j is the cost of pumping ground water, P^s_{ij} is the price of water to the farmer, and P^{ds} is the cost of diluting the drainage water to be discharged (assuming no additional dilution cost except flesh water price to the district). Assuming that the difference $\sum_i\sum_j(P^s ij - P^{ds})\cdot S_{ij}$ **equals** the overhead, them these two components should not be included in the objective function. All revenues raised by increasing prices or taxes are assumed to be rebated to the farmers in a way unrelated to surface water used or drainage produced.

The above objective function is maximized subject to several constraints. These constraints are presented and explained in detail in Dinar et al.(1990). In the following only several model equations will be explained.

Relative yield (f_{ij}), deep percolation volume (d_{ij}), and salt concentration in the deep percolation water (z_{ij}), are functions of the quantity (a_{ij}) and quality (c_{ij}) of applied water, the application uniformity (u_{ij}) measured by Christiansen Uniformity Coefficient (CUC-used as a measure for the irrigation technology), and climatic conditions expressed by pan evaporation during the growing season (e_{ij}).

$$y_{ij} = f_{ij}(a_{ij}, c_{ij}, u_{ij} \mid e_{ij})\cdot Y_{ij}$$

$$[2]\quad d_{ij} = g_{ij}(a_{ij}, c_{ij}, u_{ij} \mid e_{ij})$$

$$z_{ij} = h_{ij}(a_{ij}, c_{ij}, u_{ij} \mid e_{ij}).$$

The variable y_{ij} is used here to express absolute crop yields. Y_{ij} represents the maximum potential yield that a given farm can achieve under optimal conditions, and reflects differences in management other than those considered in the production function. The pan evaporation variable allows the model to be transferred to any location (Letey and Dinar, 1986).

Irrigation water that infiltrates the soil is used in evapotranspiration or lost to deep percolation. In some areas, an almost impermeable layer of clay impedes the percolation of water (see also Fig. 1). This water collects and must be drained away to maintain productivity. Part of the drainage may occur as subsurface lateral flow to adjacent fields or farms that presents externality problems. The total amount of drainage water produced on farm j is

$$[3]\quad D_j = q_j \sum_i d_{ij} X_{ij} - \sum_{n \neq j} \beta_{jn} q_j \sum_{k \neq j} \mu_{jk} D_k$$

where q_j represents the severity of the drainage problem on farm j, $(0 \leq q_j \leq 1)$; $q_j = 1$ means that all deep percolation results in drainage. It is assumed that each farm has homogeneous soil properties, so q_j represents on-farm drainage conditions. Parameter β_{jn} $(0 \leq \beta_{jn} \leq 1)$ is the fraction of drainage produced on farm j that arrives at farm n $(n \neq j; \sum \beta_{jn} = 1)$, and μ_{jk} $(0 \leq \mu_{jk} \leq 1)$ is the fraction of drainage from farm k that arrives at farm j $(k \neq j; \sum \mu_{jk} = 1)$. Subsurface lateral flow is one source of externality effect within the region. Where there are no lateral drainage flows, $\beta_{jn} = 0$ and $\mu_{jk} = 0$ for each n and k.

A quality (salinity) standard (C^d) may be imposed on discharged drainage. If the salinity exceeds that standard, the district must dilute the drainage with fresh water of a better quality. The quality constraint is:

$$[4] \quad \frac{\sum_j \sum_i z_{ij} D_j + S^d C^s}{\sum_j D_j + S^d} \leq C^d$$

where S^d is the amount of surface water with a given quality C^s (C^s is a higher quality than z_{ij}) used by the district for dilution. Both the quantity and the quality constraints (as a matter of fact the product $D^d C^d$) reflect the assimilative capacity value that society assigns to the water body. However, quality standards and regulations on drainage pollution were commonly associated with one of these components only.

Each farm has an annual quota (S^f_j) of fresh water (also of quality C^s) provided by the district. Farms can supplement their surface supply by pumping ground water. The amount of ground water that each farm can use is constrained by pumping equipment. The district has no control on the annual amount pumped by each farm. It is assumed that ground water is pumped from a confined aquifer and does not affect the shallow water table.

The model currently assumes drainage salinity equals deep percolation salinity. In reality, the existing shallow ground water acts as a buffer, so that changes in deep percolation quality are only partly matched by changes in drainage quality.

The amount of irrigation water used on each farm is

$$[5] \quad \sum_i a_{ij} X_{ij} = S_{ij} + G_{ij} + R_i X_{ij} \qquad , \forall j$$

where R_i is the seasonal effective rainfall for crop i, G_{ij} is ground water and S_{ij} is surface water applied.

The salt concentration in the irrigation water applied for crop i is

$$[6] \quad c_{ij} = [C^s S_{ij} + C^g{}_j G_{ij}] \,/\, [S_{ij} + G_{ij} + R_i X_{ij}] \quad \text{for each } j$$

where $C^g{}_j$ is the salt concentration of ground water in farm j.

One of several on-farm decisions is the type of irrigation technology used on crop i in farm j. In this model, uniformity of applied water is used as a surrogate for irrigation technology and irrigation management activities, with a more advanced technology being associated with a higher CUC value. Higher CUC values are associated with greater costs in irrigation hardware and/or management. The total irrigation cost (except for the cost of the water) is

$$[7] \quad K_{ij} = r_{ij}(u_{ij}) \cdot X_{ij}$$

where K_{ij} is the annual irrigation cost for crop i on farm j. It is assumed that $\partial r/\partial u > 0$ and $\partial^2 r/\partial u^2 \geq 0$. That is, the cost of achieving a better irrigation uniformity application is increasing. Also it is assumed that $\partial K/\partial X = 0$; that is, no economies of scale are assumed with regard to the size of the irrigated field.

The district's annual quota of fresh surface water is allocated to farms and used to dilute drain water. The district purchases surface water for a given price per unit volume (P^{ds}) and then provides it to the farmers for a given price of P^s_{ij} per unit volume. The model allows the district to discriminate among farms and crops. The district may try to control water consumption by increasing and decreasing this price. In addition, the model allows the district to charge either a flat rate or a tiered rate for water.

$$[8]\quad P^s_{ij} = \begin{cases} P & \text{for } S_{ij}/X_{ij} \leq H_{ij} \\ \\ v(S_{ij}/X_{ij}) & \text{for } S_{ij}/X_{ij} > H_{ij} \end{cases}$$

where P^s_{ij} is the price per unit volume of surface irrigation water applied on crop i in farm j; H_{ij} is a parameter determining the maximum amount of water per unit area of crop i in farm j that will be charged the basic rate. ($P \geq P^{ds}$ and includes only the overhead of the district). The function v has a positive first derivative with regard to the per unit area water volume.

The district can also impose (or relay) a tax (T_j) on volume of drainage created by each farm. This is done assuming that the district monitors each farm's outlet and that the monitoring costs are either zero or are already included in the district services charged to the farms.

Additional technical constraints include available land, idle land, quantity of disposed drainage water, annual surface water allotment, and ground water pumping capacity. Idle land can also be adjusted to represent land conversion to non-irrigated uses.

The model was applied to a particular water district on the west side of the SJV. The water district is comprised of 12 farms; for simplicity, this analysis is concerned with three farms. Data on cropping patterns, prices, costs yields and water quality (Dinar et al., 1990). While surface water is the primary source of irrigation water, ground-water pumping is used occasionally by the farms to augment irrigation supply. For the purpose of the analysis it is assumed that unlimited ground water is available. Water quality inputs were set at 450 ppm (EC[2]=.7) for surface water and 1280 ppm TDS (EC=2.0) for ground water for all farms. Quadratic functions for yield and deep percolation volume and quality (salinity) were estimated by crop using the model suggested by Letey and Dinar (1986). District farmers use primarily

[2] 1 EC (mmhos/cm) = 640 ppm Total Dissolved Salts.

surface irrigation--furrow and border strip--with a current CUC of about 75. Improvement in irrigation technology are represented by increases in CUC. Irrigation technology cost functions were estimated for each crop using the cost data from CH2M HILL (1989) the irrigation technology CUC values from University of California Committee of Consultants (1988). Potential yield levels for each farm were estimated using the procedure suggested by Knapp, Dinar and Letey (1986), based on yield data obtained from the district for 1987-1989. Estimated coefficients for crop yield, drainage quantity, and drainage quality functions, exponential irrigation cost functions by crops, variable and fixed production costs (excluding water), crop yield prices and weather data for the different crops can be found in Dinar et al. (1990).

The model was used to assess alternative strategies to restrict environmental pollution while maintaining agricultural production. While several policy instruments were evaluated (Dinar et al., 1991b), only two will be presented. The first involves a tax on discharged drainage. Values used varied from $0 to $40/ha cm,[3] where $0 represents the "no regulation" case. The second policy instrument involves a flat increase in surface water price. Values varied from $0 to $3/ha cm, and are based on actual water price increases charged in a neighboring district under a similar policy (Wichelns, 1991). For simplicity, administration costs associated with the programs are not considered. Also at this stage, environmental costs are not included.

The "no regulation" case is represented by policy values of $0 for both water price increase and drainage tax. The base situation was simulated using the value of CUC=75, representing the current technology level.

The farms differ in their cultivated land area, fraction of applied irrigation water resulting in drainage, and also potential levels of different crop yields (Table 1). Farm 2 produces the highest drainage fraction and farm 3 the lowest drainage fraction for all levels of applied water. Therefore, it is expected that farm 2 will be more sensitive to drainage tax relative to the other farms.

Table 2 presents regional level results. Regional net income is defined as regional income plus the amount of collected taxes. Water and drainage taxes collected in the district must be redistributed or re-invested locally, as districts are not allowed to accrue profits (this is further addressed in the last section). In the case of a drainage tax, net income drops linearly (Figure 4) from $1.15 million to $.7 million with increases in drainage tax of $0 to $40 ha cm. In the case of an irrigation water tax, net income drops exponentially (Figure 5) from $1.15 million to

[3] 1 acre foot = 12.35 ha cm

$.7 million with increases in water price of $0 to $3.5 per ha cm. The share of taxes in the regional income varies from .54 to .60 at the higher tax levels as compared to .10 to .20 at the lower tax levels.

In general, farmers respond to increased water prices and drainage taxes by reducing surface water application rates, reducing cotton acreage, and increasing the rate of ground to surface water use. Farmers are less likely to invest in improved irrigation technologies (Table 3), but instead to reduce applied water by either cutting back on irrigation water rates or reducing cropped acreage. The reduction in the average water application per unit land is not significant in the case of a surface water tax (116, 115, 108 ha cm for 0, 3 and 6 $/ha cm) but very substantial in the case of drainage a tax (116, 79, 92 ha cm for 0, 10, and 40 $/ha cm). Similarly, acreage reduction is more significant in the case of a drainage tax (36%) than in the case of a water tax (22%).

The effectiveness of the two policy tools to reduce irrigation drainage pollution can be evaluated using the information in Figures 6 and 7. In the "no regulation" drainage volume discharged was nearly 65 thousands ha cm with a salinity concentration of nearly 8.5 EC. Drainage volume is reduced with both policy instruments. However, concentration of pollutants in the drainage water increases (Letey and Dinar, 1986), and pollution load to the environment (the product of pollution volume and pollution concentration) decreases and then increases as taxes increase. This is due to two effects: (1) as farmers reduce water application pollutant concentrations rise exponentially, and (2) in the case of tax on irrigation water, farmers replace surface water with ground water of lower quality. How society measures pollution is, therefore, essential in evaluating the success of policies.

A dynamic modeling framework

While a steady state model may provide useful insights on the impacts of irrigation-induced pollution, the dynamic nature of drainage and salinity pollution are not addressed. Salinity and other toxic accumulation in soil and water bodies has a direct impact on the quality of the resource base over time. Effects of present production decisions on future opportunities may be significant and therefore, should be included in a full analysis. Moreover, many resource policies are time dimensional (e.g., phased reductions in water supply) and are better handled within a dynamic framework. Unfortunately, dynamic relationships are often relatively complex and empirical estimates may be lacking (Knapp et al., 1990). Although an optimal steady state solution may be reached after 3-5 years under certain boundary conditions (Dinar and Knapp, 1986; Yaron et al., 1982), a dynamic framework is preferable.

A dynamic model of a farm-level (or regional homogeneous) operation is described in this section. The objective (Eq. 9) is to choose over time horizon the level of acreage planted, selection of cropping patterns, application rates of irrigation water, mix of fresh and saline water, surface water sold to the market, and levels of land retired or idled:

$$[9]\quad \text{Max} \sum_t \left[\frac{1}{(1+r)^t}\right]\left\{\sum_i [(P_i-H_i)y_{ti}-V_i-K]x_{ti}-\sum_h (w_t^h)W_t^h-(D_t)G_t-(xI_t)K+(xR_t^{'})R+\sum_j (m_t^j)M_t\right\}$$

where t is year $(t=1,...,T)$; i is crop $(i=1,...,n)$, and r is real interest rate; x_{ti} is area of crop i in year t; P_i is market price for crop i; H_i is harvest cost per unit of yield; y_{ti} is yield; V_i is per acre non-water variable cost of production; w^h_t is total water use by supply source h at price W^h_t. D_t is drainage volume, and G_t is per unit cost of drainage disposal. K is per acre annual irrigation capital cost, xR'_t is acreage retired for salinity control and R_t is the per acre compensation. Variable m^j_t is surface water of supply type j sold in the water market at a price of M_t per acre-foot.

The intertemporal problem in [9] is maximized subject to production function relationships, land and water resource constraints, and initial conditions of certain variables. Several features distinguish this model from the steady state model discussed earlier. First, soil salinity is a state variable in the model, and is included in the production function relationships:

$$[10]\quad \begin{aligned} y_{ti}&=y_{ti}\Big(\sum_j a^j_{ti},\ C_{ti},\ sL_{ti}\Big) && t=0,...,T; && i=1,...,n,\\ s_{ti}&=s_{ti}\Big(\sum_j a^j_{ti},\ C_{ti},\ sL_{ti}\Big) && t=0,...,T; && i=1,...,n,\\ d_{ti}&=d_{ti}\Big(\sum_j a^j_{ti},\ C_{ti},\ sL_{ti}\Big) && t=0,...,T; && i=1,...,n. \end{aligned}$$

where y_{ti}, s_{ti}, and d_{ti} are per acre yield soil salinity at the end of the irrigation season, and per acre drainage volume. Each of these variables is dependent upon total applied irrigation water per acre $(\sum a^j_{ti})$, weighted salt concentration of the irrigation water (C_{ti}), and soil salinity following preseason leaching (sL_{ti}). These crop-water production functions were estimated from a multi-year lysimeter experiment conducted under conditions prevailing in the SJV (Dinar et al., 1991c). Technology effects on yield, salinity and drainage are reflected in factor adjustment to base function intercept (Rhoades, 1990)

Water supplies available to the farm include surface (base and supplement) and ground water. Surface water can be used for irrigation during the growing season, for pre-season leaching of soil salts, and for sale in a water market (where permitted). Ground water can also be used for irrigation and leaching. Surface water maybe blended with ground water, and the mix may vary for crop and preseason applications..

Use of saline water may cause accumulation of salts in the soil. The dynamic nature of the problem is driven by equation [11], the equation of motion for soil salinity, based on initial salinity aggregated over acreage base (SI_t), ending soil salinity (s_{ti}), idled acreage (xI_t), and acre base adjusted for land retirement (X_t).

$$[11]\ SI_{t+1}=SI_t\frac{xI_t}{X_t}+\sum_i\left[\frac{s_{ti}x_{ti}}{x_{ti}}\cdot\frac{x_{ti}}{X_t}\right]\qquad t=0,\ldots,T.$$

Equation [12] defines the leaching application function, based on initial salinity, soil salinity after leaching (sL_{ti}), weighted salt concentration of leaching water (CL_{ti}), and leaching factor (L).

$$[12]\ \sum_j aL^j_{ti}\geq\frac{L(SI_t-sL_{ti})}{sL_{ti}-CL_{ti}}\qquad t=0,\ldots,T;\ i=1,\ldots,n.$$

Total drainage produced on farm is the summation over the fields of the drainage produced by the leaching and irrigation applications. Drainage volume produced from crop water applications is computed from Equation [10] above. Drainage volume produced by the leaching activity is calculated as the difference between the amount applied water $(\Sigma_j aL^j_{ti})$ and the root zone water holding capacity (RC-WP), where RC is field capacity and WP is wilting point.

Additional technical and balance equations, upper and lower bounds on certain variables, and initial conditions are included in (Dinar et al., 1991a).

The model is applied to a representative region in the westside SJV over a planning horizon of 15 years. Representative cropping patterns include wheat, sorghum, and wheatgrass, based on availability of production functions from field lysimeter tests. Efforts are underway to estimate yield, soil salinity and drainage (quantity and quality) for a set of major crops on the west side of the SJV. Representative salinity concentrations for surface and ground water are .7 and 2.0 EC, respectively, with an initial soil salinity of 1.5 EC. The application assumes a gravity irrigation system (1/4 mile run), using variable and capital irrigation cost data from CH2M HILL (1989). At this time, irrigation technology choice is exogenous.[4] Prices, technical coefficients and assumptions used for this model can be found in Dinar et al., 1991a.

Several policies and scenarios have been simulated. We include in this presentation only two policy tools (1) water quotas, and (2) drainage disposal permits. The base case is represented by a full water quota of 1500 AF and unrestricted drainage quantity. Impacts of reductions of surface water quota and drainage permits on net present value of income, resulted

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[4] Development is underway on extending the model to incorporate endogenously the adoption of alternative irrigation technologies.

annual optimal drainage volumes, and initial soil salinity are presented in Figure 8 to 11.

Net regional income is plotted against surface water quota and drainage permit levels (Figures 8 and 9). Effects on regional income of drainage permits are modest (reduction of 2% to 33% when permits decrease from 78% to 22% of the base case value) relatively to the surface water quota (reduction of 1% to 50% when quotas decrease from 17% to 40% of the base case value).

Soil salinity and drainage volumes values over time as affected by levels of drainage permits and surface water quotas are ploted in figures 10 and 11. Use of water quota results in relatively lower levels of of soil salinity at the steady state value (around 3 EC), compared to the use of drainage permits (3.05-3.11 EC). However, with drainage permits the steady state value is reached quicker (2-3 years) than with surface water quota (5-7 years). Drainage volumes are reduced by both policy instruments. The optimal path of drainage volumes converges quicker to the steady state values in the case of drainage permits (2 years) compared to surface water quota (2-7 years). The drainage permit becomes an effective constraint after the second year in all levels of drainage permit use.

Annual optimal values for land use and rate of ground water use are provided in Tables 4 and 5. The overall result is under the conditions analyzed here, a steady state solution is achieved relatively early, between one to five years, depending on the variable and the policy instrument used. With quotas on surface water (Table 4), farmers tend to reduce the land used, utilize the surface quota and amend it by ground water pumping. As surface quota decreases, cultivated land is decreased, and ratio of ground water in the applied water mix is increased. Over time, this ratio decreases until reaches the steady state value. For the case of drainage permits (Table 5), the cultivated land does not change, total applied water is reduced as permit levels decrease. Over time there is a slight increase in water application rates, with an increase in the share of ground water. There is also a shift over time to rotations with more wheat. This is more significant as permit level decrease (not presented).

Framework for regional cooperation with irrigation externality problems

The physical-economic models discussed in the previous section provide economically feasible solutions to irrigation externality and non-point pollution problems. However, the suggested policies and solutions may not be acceptable by the parties involved because (1) not all parties were included in the modeling framework, and (2) considerations other than profit maximization are included in the objective functions of some parties.

Here we consider the problem of regional cooperation in water management from a game theory perspective. In contrast to market situations with large number of participants, the game situation in the SJV involves a relatively small number of producers. Producers are organized into water districts, with a board and a water district manager. The district manager has the power to set water rates and water use practices for the district with the acquiescence of the board. An enforcement body exists as well, namely the California Water Resource Control Board.

The current setting has the nature of a noncooperative game. In the noncooperative case, given market prices, participants need to obtain information about preferences of others and need only to choose their own actions based on their own preferences given the actions of others.

Traditional economic solutions for externality problems include use of Pigouvian taxes and Coasian bargaining. Pigouvian tax (Baumol and Oates, 1989) sets the level of the externality at the Pareto optimal level with respect to a noncooperative, rather than a cooperative solution. Coasian bargaining solutions to achieve Pareto optimality, preceded by a required definition of property rights, will also fail to achieve efficient agreement (Samuelson, 1985).

Game theory has previously been applied (e.g., Rinaldi et al., 1979) to externalities and public goods separately, whereas here we apply game theory for a combination of externalities and public goods. A regional cooperative system for water quality/quantity control and improvement has the nature of a public good in that it would provide benefits jointly to producers and consumers (Figure 12) who would value such improvements differently, and require to determine the method for its finance.

Improving water quality for recreation and other instream values would impose costs on agricultural producers, not voluntarily accepted unless offset by benefits of economies of scale and cost sharing schemes. The literature on cost allocation has viewed such situations as cooperative games (Young, 1985; Loehman and Winston, 1971). Recent research in public goods also concerns the free rider problem associated with obtaining demand information by using a demand revealing mechanism to induce truthful behavior as the best strategy. Such schemes generally do not satisfy Pareto optimality in that there will be a budget surplus resulting from the “truth tax”. To avoid this, we assume that a regional manager has access to information on preferences of the players.

We will follow here the framework in Loehman and Dinar (1991) that suggests the use of game theory concepts and a mechanism design approach to the regional externality problem caused by irrigated agriculture. Under this approach, a desired social outcome represents a Pareto optimal cooperative solution which is acceptable to all players and therefore can be sustained as an equilibrium outcome. In this application game players include upslope and downslope producers, consumers of recreation, and a regional manager whose role is to propose and enforce rules of the game. Acceptability of cooperative solutions can be determined for sets of political weights assigned to the parties involved. The mechanism for such game is displayed in Figure 13.

There are three situations in the regional externality problem: (1) the status quo, (2) the noncooperative, and (3) the cooperative solutions (Figure 14). Technical efficiency is represented by a production frontier for agricultural production (F) and environmental amenities (Q). Private technology is applied and operated by the individual producers on their own fields (irrigation systems). Each production technology has an associated frontier, and the noncooperative frontier represents the envelope of the intersections of the frontiers for each private technology. Due to externalities, this frontier maybe convex in the noncooperative case. Cooperative technologies are implemented at the regional level. They may include a regional water treatment facility, regional storage, reuse,drainage systems, and extension and information systems. If cooperative technologies are added to the existing private technologies, the resulting frontier can lie outside the existing frontier for certain combinations of efficiency-quality. The status quo point corresponds to a maximization of agricultural profit with private water use technologies.

Given a set of political weights, with known payoff functions and production relationships, the regional manager than computes taxes and cost shares for both the noncooperative and cooperative solutions by solving the joint maximum problems described below. The noncooperative solution will be achieved through applying Pigouvian taxes determined by the manager. This same set of weights will also be used by the manager to compute the consumers' share of the joint regional costs.

Thus, a higher weight implies more consideration in joint cost. By making cost shares in the cooperative solution equal to political weights used to compute noncooperative and cooperative solutions, consumers will evaluate benefits of environmental quality improvements to the costs of implementing them through the process.

If externalities are severe enough, consumers damaged by externalities may lobby for improved quality standards corresponding to production along the noncooperative frontier. The extent of consumers' success to achieve their objective is affected also by the political power of the producers.

The producers then choose between the pair of noncooperative solutions according to the highest payoff values. Producers can also reacquire that the game continue and political weights be revised if neither the noncooperative nor the cooperative solutions is attractive relative to the status quo.

If such a process stops at a cooperative solution, the equilibrium must be an acceptable cooperative solution such as (1) each player is better of than at the status quo, (2) each player is better off than in the noncooperative solution with the same political weights; and (3) joint costs of cooperation are covered by players.

Game players

Producers payoffs are naturally defined in terms of profits. Following recent environmental literature, the concept of Equivalent Variation (EV) is used to represent consumer preferences for environmental quality in monetary terms reflecting expenditure changes due to changes in environmental quality (health, recreation). Thus, payoffs for consumers and producers will be comparable.

The Producers:

Upslope and downslope producers (u and d, respectively) are characterized with a separate (per acre) production function of water (W) and technology (τ). Each producer has a limited area of land (A). Profits are calculated as revenue from agricultural production less charges for water use (v), taxes on water and land used (t^u_ω, t^u_ℓ, respectively), and annual fixed costs for water technologies $c(\tau^u)$.

The upslope producer's yield (Y^u) is related directly to his choices of water and technology. The downslope producer's yield is related to his choices of water and technology, as well as drainage caused by water use of the upslope producer. Both producers maximize profits by choosing optimal levels of cultivated acres, applied water per acre, and the water use technology.

The individual optimization problems for upslope [13] and downslope [14] producers under the status quo case are:

$$[13] \qquad \max_{\tau^u,\, W^u,\, A^u} \; [p_f Y^u - (v + t^u_\omega) W^u - c(\tau^u) - t^u_\ell] A^u$$

s.t. $A^u \leq \overline{A}^u$

$Y^u = Y^u(W^u; \tau u)$

and:

[14] $\max\limits_{\tau^d, W^d, A^d} \quad [p_f Y^d - (v + t^d_\omega) W^d - c(\tau^u) - t^d_\ell] A^d$

s.t. $A^d \leq \overline{A}^d$

$Y^d = Y^d(W^d, kW^u A^u; \tau d).$

Levels of profit given by the optimal solution in the status quo is π^i_0, for producer i.

Both producers generate pollution represented by the concentration of pollutants and the volume of drainage water (This may be a vector were several pollutants are considered). For the case of the up slope producer, pollution depends on land and water use decisions and technologies employed

[15] $S^u = \delta^u(\tau^u) W^u A^u.$

In the case of the downslope producer, pollution discharge depends on drainage from the upslope producer

[16] $S^d = \delta^d(\tau^d)(W^d A^d + kW^u A^d).$

The total pollution from the region (S) is the sum of the upslope and downslope discharges.

[17] $S = S^u + S^d$

The Consumers:

Preferences of consumers are represented by a utility function. As utility is not defined in dollar units, however, it is not directly comparable to producer profits. Using the Equivalent Variation measure provides a dollar measure of welfare which results in a ranking of outcomes similar to that of a utility-based criteria.

The expenditure function is defined from the indirect utility function:

[18] $\overline{U} = \overline{U}(M, S, p_f, p_h, p_r, p_z)$

where M is initial income, S is the regional drainage discharge, and p. denotes respectively the price of food, health, recreation and other goods. Improvement of drainage quality may improve consumer welfare. The amount of money which is equivalent to a change in the pollution level from the status quo S^o to S' $(S^o > S')$, satisfies the following relationship:

[19] $\overline{U}(M+EV, S^o, p_f, p_h, p_r, p_z) = \overline{U}(M, S', p_f, p_h, p_r, p_z).$

The EV is a function of drainage water quality in terms of the change in expenditures required to purchase food, health and recreation, relative to the base condition S^o. For simplicity we assume that food prices are not affected by level of production.

In the analysis below, an equivalent variation function, $EV(S;S^o)$, will be used to denote consumer welfare as a function of improved drainage quality. (Note that $\partial EV/\partial S<0$, i.e., as the pollution decreases, the equivalent variation increases.)

The noncooperative Nash Equilibrium and Pigouvian Taxes

The noncooperative Nash Equilibrium (NC) is a game solution in which each player chooses the strategy which maximizes that player's payoffs, given that the strategies of other players are fixed corresponding the noncooperative solution. This solution lies along the production frontier corresponding to choices made by producers among private technologies. A noncooperative solution achieves a tradeoff between drainage water quality and agricultural production and is indicated by the slope of the production frontier. Each solution can be related to a given set of political weights of the players.

The frontier is found by maximizing a weighted sum of payoff functions for game players with varying weights summing to one (Takayama, 1974). The joint welfare optimization problem is:

$$[20]\quad JW(\alpha;NC)= \max_{\substack{\tau^u,\, W^u, A^u,\\ \tau^d,\, W^d, A^d}} \alpha_c EV(S;S^o)+\alpha_u[p_f Y^u - vW^u - c(\tau^u)]A^u + \alpha_d[p_f Y^d - vW^d - c(\tau^d)]A^d$$

$$\text{s.t.}\quad \begin{aligned} & A^u \le \overline{A}^u \\ & A^d \le \overline{A}^d \\ & A^u W^u + A^d W^d \le \overline{\overline{W}} \\ & Y^u = Y^u(W^u;\tau^u) \\ & Y^d = Y^d(W^d, kW^u A^u;\tau^d) \\ & S^u = \delta^u(\tau^u) W^u A^u \\ & S^d = \delta^d(\tau^d)(W^d A^d + kW^u A^d) \\ & S = S^u + S^d \end{aligned}$$

Here, the weighted sum of producer and consumer payoffs is maximized over private technologies, irrigated acres, and water use. Constraints are the same as for the individual maximization problems in [13] and [14], except that there is a regional water constraint $\overline{\overline{W}}$. For the noncooperative solution corresponding to political weights α, the optimal pollution level is denoted by $S(\alpha;NC)$. (For the status quo, the weight on the consumer is zero.)

The noncooperative solution is achieved as a Nash equilibrium by producer profit maximization in response to appropriate taxes (e.g. Pigouvian tax) set by a regional authority. Pigouvian taxes to achieve a given noncooperative equilibrium are derived from first order conditions for the noncooperative joint maximum problem in [20]. Since the pollution is a non-point problem and pollution is determined by land and water use, taxes on pollution are equivalent to taxes on land and water (assuming knowledge of the physical relationships) which are preferred due to reduced information and enforcement costs.

Solving the frost order condition for marginal profit, the optimal taxes on water use and land for the upslope and downslope producers, respectively, are represented by the right handside of the following expressions:

$$[21]\ \frac{\partial \pi^u}{\partial W^u} = \frac{\mu}{\alpha_u}A^d - \frac{\alpha_c}{\alpha_u}\frac{\partial EV}{\partial S}(\delta^u + \delta^d k)A^u - \frac{\alpha_d}{\alpha_u}\frac{\partial \pi^d}{\partial \pi^u}$$

$$[22]\ \frac{\partial \pi^d}{\partial W^d} = \frac{\mu}{\alpha_d}A^d - \frac{\alpha_c}{\alpha_d}\frac{\partial EV}{\partial S}\delta^d A^d$$

$$[23]\ \frac{\partial \pi^u}{\partial A^u} = \frac{\lambda_u}{\alpha_u} - \frac{\alpha_c}{\alpha_u}\frac{\partial EV}{\partial S}(\delta^u + \delta^d k)W^u - \frac{\alpha_d}{\alpha_u}\frac{\partial \pi^d}{\partial A^u}$$

$$[24]\ \frac{\partial \pi^d}{\partial A^d} = \frac{\lambda_d}{\alpha_d} - \frac{\alpha_c}{\alpha_d}\frac{\partial EV}{\partial S}\delta^d W^d.$$

The shadow price for the regional water constraint is denoted by μ, and λ_u λ_d represent land opportunity values for the upslope and downslope producers, respectively.

Note that the taxes $t^i_\omega(\alpha;NC)$; $t^i_\ell(\alpha;NC)$ for each producer i are related to political weights α. Optimal taxes on water and land use for the upslope producer should be higher than for the downslope producer for equal weights and area planted, because upslope producers cause external costs for both down slope producers and consumers. In the optimal solution, water use is reduced relative to the status quo case where no taxes are imposed and the marginal profit equals zero (Figure 15). By the same token, less land will be irrigated when taxed, relative to the status quo.

Producers' profit for the noncooperative solution are obtained by subtracting taxes from profits in the joint maximum

$$[25]\ \pi^i(\alpha;NC) = [p_f Y^i - vW^i - c(\tau^i)]A^i - t^i_\omega(\alpha;NC)W^i A^i - t^i_\ell(\alpha;NC)A^i \quad ,\forall i=1,2.$$

Cooperative solution

In the cooperative case, regional technologies for drainage reduction and treatment are available in addition to private technologies. In the cooperative case, reduced pollution levels may be achieved at lower cost under regional facilities due to economies of scale.

The joint welfare problem solved for the cooperative and noncooperative cases are similar except that for an acceptable solution, the joint costs in the cooperative case should not exceed the difference between the weighted sum of payoffs in the cooperative and noncooperative cases. The cooperative solution is found without reference to the method of cost allocation. The optimization problem for the cooperative production frontier is:

$$[26]\quad JW(\alpha;CS)=\max_{\substack{\tau^R,\\ \tau^u,\, W^u, A^u,\\ \tau^d,\, W^d, A^d}} \alpha_c EV(S;S^o)+\alpha_u[p_f Y^u - v'(\tau^R)W^u - c(\tau^u)]A^u + \alpha_d[p_f Y^d - v'(\tau^R)W^d - c(\tau^d)]A^d - JC(S,\, W^R;\tau^R)$$

s.t.

$$A^u \le \overline{A}^u$$
$$A^d \le \overline{A}^d$$
$$A^u W^u + A^d W^d = W^R \le \overline{W}\, r(\tau^R)$$
$$Y^u = Y^u(W^u;\tau u)$$
$$Y^d = Y^d(W^d,\, kW^u A^u;\tau d)$$
$$S^u = \delta^u(\tau^u)W^u A^u$$
$$S^d = \delta^d(\tau^d)(W^d A^d + kW^u A^d)$$
$$S = S^u + S^d.$$

Regional technologies are denoted by τ^R, regional cost of treating and reusing drainage water is $JC(S\,,W^R;\tau^R)$, where W^R is total volume of water used in the region. Pollution S is related to the weights in the cooperative solution and is denoted by $S(\alpha;CS)$. Charges for water use (v') may be smaller than the charge for water in the noncooperative case (v), because of reuse.

Comparison of the cooperative (CS) and the NC joint maximum problems shows that a cooperative solution will result in a higher value for the objective function JW since private technologies are feasible for both the cooperative and noncooperative problems.

As in the noncooperative solution, imposing Pigouvian taxes on land and water use makes private water and land use decisions consistent with the joint welfare maximum. In addition to externality effects, the tax now includes marginal (variable) cost for the joint facility. Because of economies of scale, revenue from these taxes will not cover costs of the joint facility. Therefore, agreement to participate in the cooperative solution requires that producers pay a share of the joint cost (including fixed cost) of the regional facility.

Producer's profit in the cooperative solution, after taxes and cost shares are:

$$[27]\ \pi^i(\alpha;CS) = [p_f Y^i - v'(\tau^i)W^i - c(\tau^i)]A^i - t^i_\omega(\alpha;CS)W^iA^i - t^i_\ell(\alpha;CS)A^i - \alpha_i[JC(S,\ W^R;\tau^R)-TR] \quad ,\forall\ i=1,2$$

where TR denotes total tax revenues collected in the region.

Acceptable solutions

For a solution of [26] to be "acceptable", requires that all parties prefer a set of payoffs to both the noncooperative solution and the status quo, so that such a solution could be achieved voluntarily. For producers, two conditions must apply. First, the payoff in the cooperative case must be greater then in the noncooperative case

$$[28]\ \pi^i(\alpha;CS)-\pi^i(\alpha;NC)\geq 0,$$

and payoff in the cooperative case should exceed profits in the status quo case

$$[29]\ \pi^i(\alpha;CS)-\pi^i_o\geq 0.$$

If [29] holds, than enforcement cost can be minimized.

As mentioned before, the technology choice set for the noncooperative problem is contained in that for the cooperative problem. Therefore, profits will be greater in the cooperative solution than in the noncooperative solution, if (1) the joint cost share is less than the tax cost in the noncooperative solution, (2) private technologies are less expensive in the cooperative case, and (3) output is not reduced in the cooperative case.

Consumers are better off in the noncooperative case compared to the status quo since pollution is reduced. Pollution is at least the same in the cooperative case compared to the noncooperative case. However, since consumers do not have to pay in the noncooperative case, consumers are only better off in cooperative solution when:

$$[30]\ EV[S(\alpha;CS);S^o]-EV[S(\alpha;NC);S^o]\geq\alpha_c[JC-TR].$$

That is, water quality in the cooperative solution must be sufficiently higher than in the noncooperative solution to offset the cost share paid by consumers in the cooperative case.

Whether equations [28]-[30] hold will depend on the political weights, nature of the physical relationships and available technologies.

Application

The approach described above was applied to conditions in the SJV using a simplified example. Upslope and downslope producers grow the same crop with two irrigation technology options. Upslope drainage affects downslope water quality, and total drainage produced in the agricultural process pollutes a receiving water body which serves as a recreation source. The drainage water can be treated in a regional plant before discharged to the water body. Physical relationships, agricultural production costs, treatment cost, and consumer benefits are estimated and incorporated in the application.

Results are presented in Table 6 and Figure 16. Table 6 gives payoff values for the status quo, the noncooperative and cooperative solutions, for various consumers' weights. Figure 16 shows the corresponding production frontiers for noncooperative and cooperative cases. The nonconvexity of the noncooperative frontier reflects both externalities as well as indivisibilities of private technology choices.

For the cooperative solution with political weight of .33 for the consumers, area farmed and water applied is reduced, less drainage is produced because of increased irrigation efficiency, and the amount of drainage treated increases. As political weight assigned to consumers is increased, pollution is reduced although the consumer bears a larger share of cost obtained but also the consumer has a larger cost share. Producers reduce cultivated area to meat the quality constraint. For low pollution levels, drainage is reduced and producers pay a smaller share of the joint facility cost.
The consumer wieght of .40 produces an acceptable cooperative solution for the cost sharing method of shares equal to political weights.

Discussion, and future research needs

This paper deals with several problems. First, it demonstrates the complexity of physical relationships that are associated with agricultural irrigation pollution. Second, it suggests ways to overcome these complexities and still provide meaningful information to policy makers. Third, it argues that, given the case of nonpoint source pollution and externalities, cooperation between the parties involved and voluntary solutions may provide under certain conditions an easier way to achieve socially preferred policies.

During the course of our research on irrigated agriculture and environmental pollution in the SJV, we made several compromises,however we gained much experience and passed several junctions. We feel that we can now make several generalizations based on our research results, and would be happy to open it for discussion.

Appropriate modeling of the interacting system is essential for providing relevant impact and policy analysis. How do we do that? The instant answer is an interdisciplinary work including scientists that are familiar with the technical aspects of physical relationships. This means that we, the economists, need to collaborate with hydrologists, soil and plant science experts and environmentalists. We must suggest them what are the important variables that we need, and urge them to provide us with information and data that can be implemented by us in our models. Having appropriate data set is important for our analysis because although theoretical relationships may exist, their implementation for policy analysis may not be relevant.

Another important feature of our analysis is aggregation. Since our capacity to analyze properly real world economic and/or physical relationships is limited, we are facing a problem. Aggregation, if done properly, may reduce the burden. Aggregation may take place either over the parties involved in the problems to be solved, or over variables affecting the system.

Dynamic versus steady state approaches to model physical relationships as well as economic behavior have much been discussed in the past. Unfortunately, our data did not allow us to model the exact same problem under both dynamic and steady state approaches. It is clear the the dynamic approach provides a better and probably a more realistic description of the behavior of key variables. However, it also introduces an addition burden to the modeler. In the example introduced in this paper, for the initial condition used (soils salinity and water qualities), the additional information gained using the dynamic approach was very marginal since a convergence to steady state was reached very early. Application of the model under more extreme conditions will result in a different optimal behavior compared to a steady state model.

A non-relevant set of policy variables and instruments chosen by the policy maker, may misguide the analysis. For example, in the specific case of drainage water, there is a reciprocal relationship between discharged volume and the concentration of pollutants. Dealing with one only may mislead the policy maker.

Finally, the question whether acceptable cooperative versus the status quo or noncooperative solutions in a real world. For implementation of the mechanism suggested here, it is assumed that institutions already exist for data collection, computation of taxes, and dissemination of information. Even with potential gains, actual acceptance of cooperative solution is a remaining question. Further behavioral work should be undertaken to determine whether a cooperative game process such as that proposed here would result in an actually acceptable cooperative solution.

Table 1:

Acres, drainage conditions, and potential yield levels by farm (steady state model).

Farm No.	Potential Cultivated land (ha)	Irrigation to Drainage Ratio (fraction)	Potential yield level (ton/ha)			
			Alfalfa	Cotton	Tomato	Wheat
1	1875	.5	21.25	1.6	78	3.5
2	1775	.8	17.50	1.6	72	3.0
3	400	.2	21.25	1.6	85	3.5

Table 2:

Regional income, acres farmed, applied surface and ground water, and drainage quantity and quality, and collected taxes, as affected by policy measures (steady state model)

Policy and policy var. value	Regional income[a] ($$10^6$)	Acres farmed Alf. (ha)	Cot.	Tom.	Applied water Surface (10^6 ha cm)	Ground	Drainage water Quantity	Quality (EC)	Collected taxes ($$10^6$)
No regulation	1148.4	200	2174	1150	355.5	53.7	63.0	8.3	0
Water price (flat)									
$3/ha cm	970.7	200	2174	1150	355.5	53.2	62.8	8.6	177.7
$6/ha cm	300.4	200	1425	1150	101.8	198.9	33.2	14.7	356.2
Drainage fee (flat)									
$10/ha cm	763.1	200	1425	1150	255.3	23.1	24.6	21.5	246.8
$40/ha cm	277.5	200	1106	937	204.7	2.5	10.4	56.7	417.1

a Not including redistribution of taxes

Table 3:

Irrigation technology selected, area farmed, Applied surface and ground water by farm and policy measure (steady state model).

Policy	Farm 1	Farm 2	Farm 3
	CUC of irrigation technology used to Irrigate the main crop (cotton)		
No regulation	75	75	75
Water price-Flat $3/ha cm	75	75	75
Water price-Flat $6/ha cm	75	75	75
Drainage tax-flat $10/ ha cm	75	75	75
Drainage tax-flat $40/ha cm	80	87	75
	Area farmed (ha)		
No regulation	1875	1250	400
Water price-Flat $3/ha cm	1875	1250	400
Water price-Flat $6/ha cm	1875	500	400
Drainage tax-flat $10/ ha cm	1875	500	400
Drainage tax-flat $40/ha cm	1844	0	400
	Applied surface water (ha cm)		
No regulation	168750	150750	36000
Water price-Flat $3/ha cm	168750	150750	36000
Water price-Flat $6/ha cm	101771	0	0
Drainage tax-flat $10/ha cm	168750	49177	36000
Drainage tax-flat $40/ha cm	168750	0	36000
	Applied ground water (ha cm)		
No regulation	44137	0	9594
Water price-Flat $3/ha cm	43722	0	9459
Water price-Flat $6/ha cm	99999	51086	46434
Drainage tax-flat $10/ ha cm	16000	0	7084
Drainage tax-flat $40/ha cm	0	0	2497

Table 4:

Optimal annual values for land and water use in the case of reduced surface water quota

	Cropland use				Total water applied			
Year	------Surface water quota------				------Surface water quota ------			
	1500	1250	750	500	1500	1250	750	500
1	500	500	427	281	1816	1816	1443	957
2	500	500	322	234	2108	2039	1258	872
3	500	500	313	220	2117	2037	1243	847
4	500	500	310	213	2118	2037	1238	836
5	500	500	309	210	2118	2037	1236	831
6	500	500	309	208	2118	2037	1236	827
7	500	500	309	207	2118	2037	1236	826
8	500	500	309	207	2118	2037	1236	825
9	500	500	309	207	2118	2037	1236	825
10	500	500	309	207	2118	2037	1236	825
11	500	500	309	207	2118	2037	1236	825
12	500	500	309	207	2118	2037	1236	825
13	500	500	309	207	2118	2037	1236	825
14	500	500	309	**207**	2118	2037	1236	825
15	500	500	309	207	2118	2037	1236	825

Table 5:

Optimal annual values for land and water use in the case of drainage permits

	Cropland use				Total water applied			
Year	------Drainage permit ------				------Drainage permit ------			
	450	350	250	100	450	350	250	100
1	500	500	500	500	1816	1816	1816	1791
2	500	500	500	500	2108	2025	1924	1757
3	500	500	500	500	2117	2026	1925	1753
4	500	500	500	500	2118	2027	1925	1752
5	500	500	500	500	2118	2027	1925	1752
6	500	500	500	500	2118	2027	1925	1752
7	500	500	500	500	2118	2027	1925	1752
8	500	500	500	500	2118	2027	1925	1752
9	500	500	500	500	2118	2027	1925	1752
10	500	500	500	500	2118	2027	1925	1752
11	500	500	500	500	2118	2027	1925	1752
12	500	500	500	500	2118	2027	1925	1752
13	500	500	500	500	2118	2027	1925	1752
14	500	500	500	500	2118	2027	1925	1752
15	500	500	500	500	2118	2027	1925	1752

Table 6:
Payoff results ($000)

Status quo

Consumers benefits	Producer 1 payoff	Producer 2 payoff	Pollution ppb Se
248	825	516	31.91

Noncooperative solutions

Weight on consumers	Consumers benefits	Producer 1 payoff	Producer 2 payoff	Pollution ppb Se
.60	284	657	422	22.14
.50	279	712	436	23.54
.42	271	713	440	25.64
.40	271	711	440	25.64
.33	268	707	443	26.44

Cooperative solution

Weight on consumers	Consumers benefits	Producer 1 payoff	Producer 2 payoff	Pollution ppb Se
.60	265	799	495	14.43
.50	286	804	533	15.43
.42	271	837	542	15.64
.40	272	835	541	15.64
.33	252	834	541	22.55

Benefit/Loss of cooperative Solutions as related to weights

Weight on consumer	Consumers	Producer 1	Producer 2
	Compared to NC	Compared to SQ	
.60	-19	-26	-21
.50	+7	-21	+17
.42	0	+12	+26
.40	+1	+10	+25
.33	-16	+9	+25

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Figure legend:

Figure 1:
A schematic display of the crop-water-soil-drainage system.

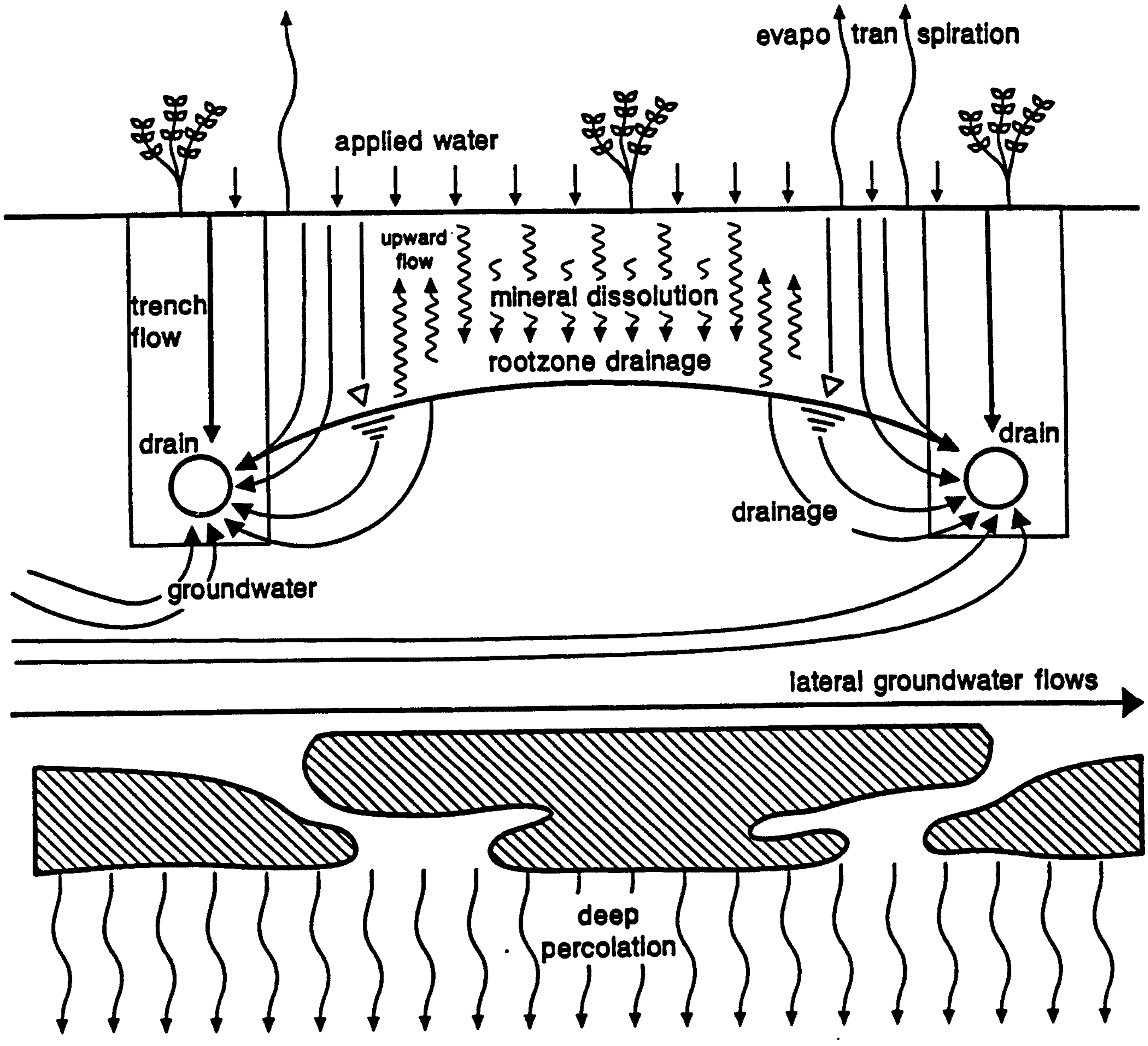


Figure 2:
Areas with drainage problems in the San Joaquin Valley, California.

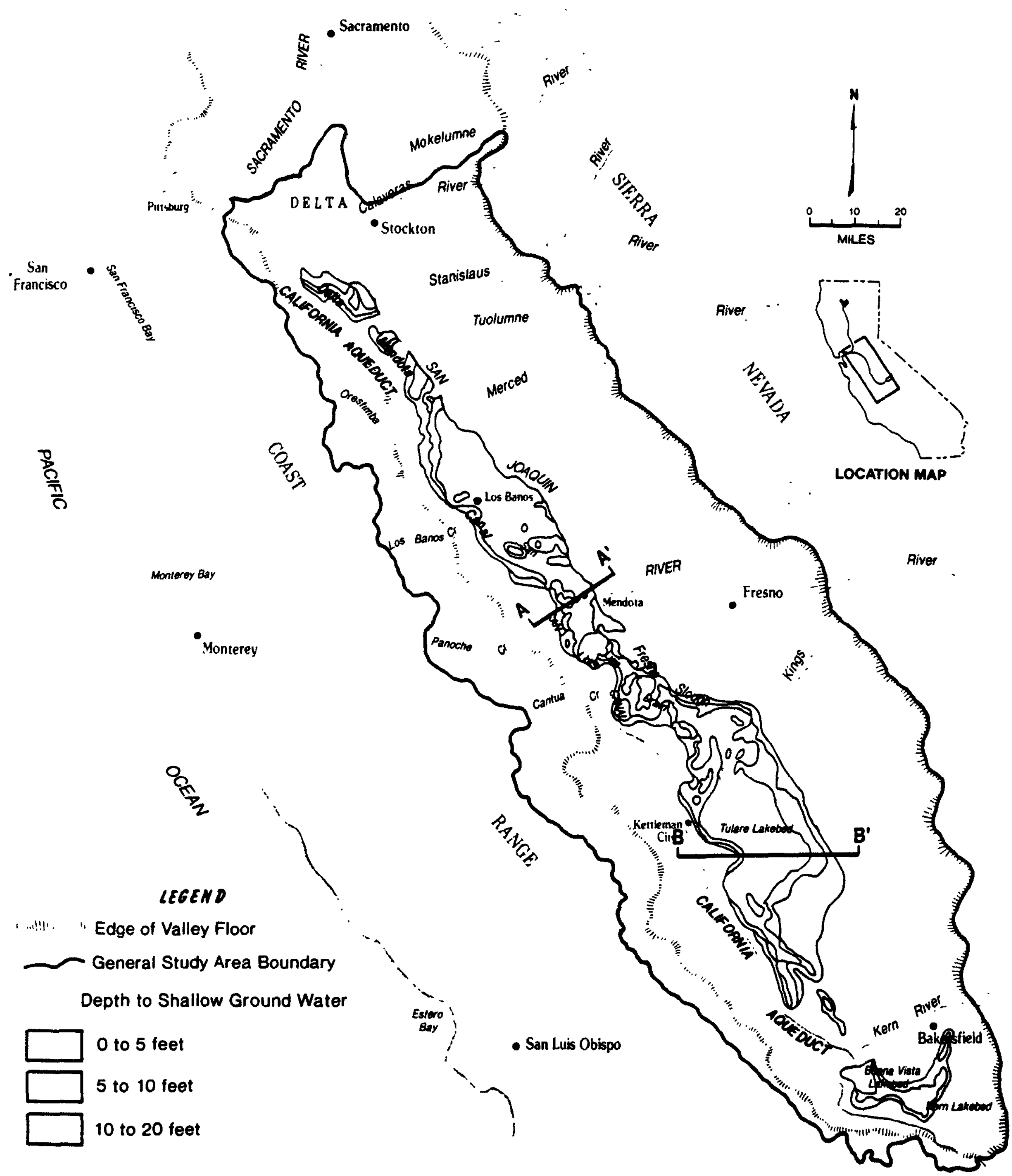


Figure 3:
Generalized Geohydrological cross-sections in the San Joaquin and Tulare basins (locations shown in Figure 2).

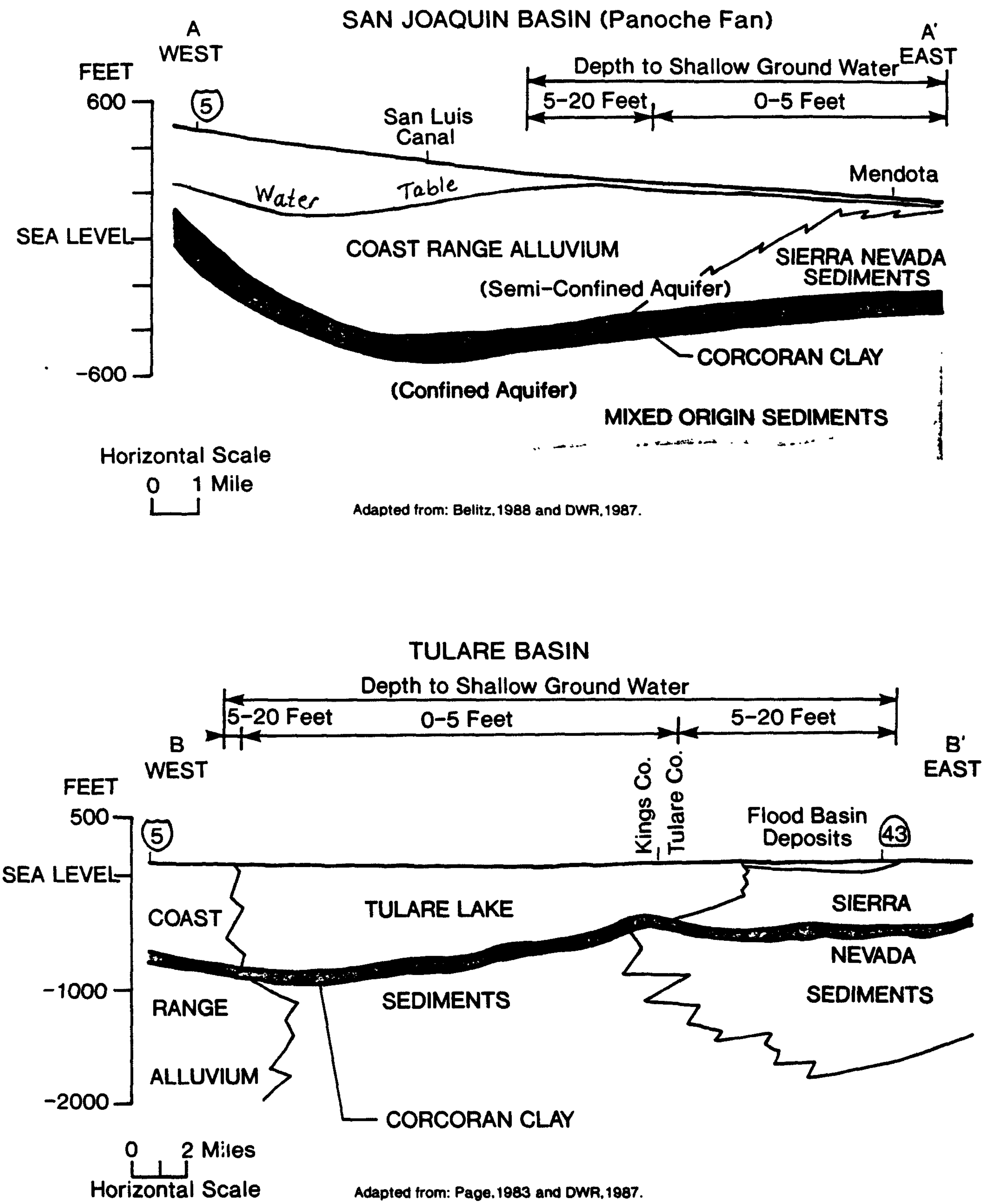


Figure 4:
Effect on regional net incom of different levels of drainage tax (steady state model)

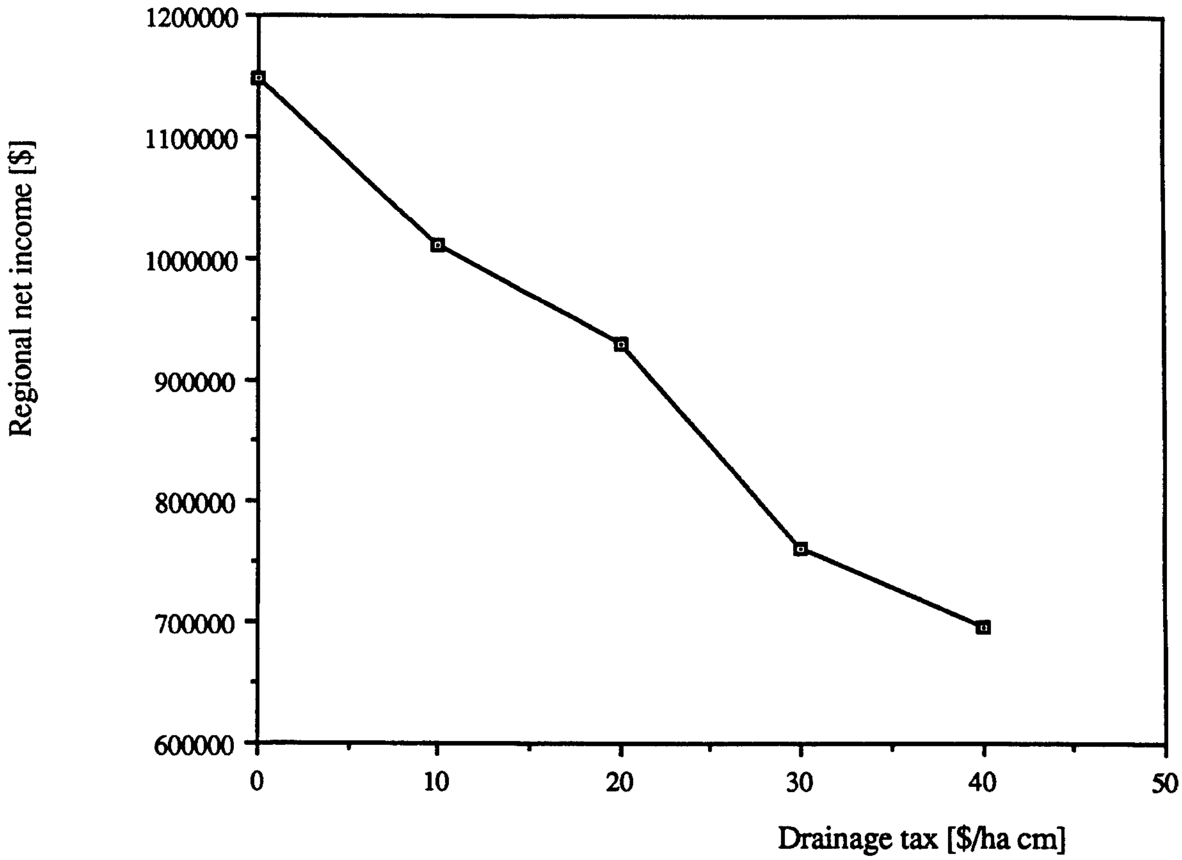


Figure 5:
Effect on regional net income of different levels of water prices (steady state model)

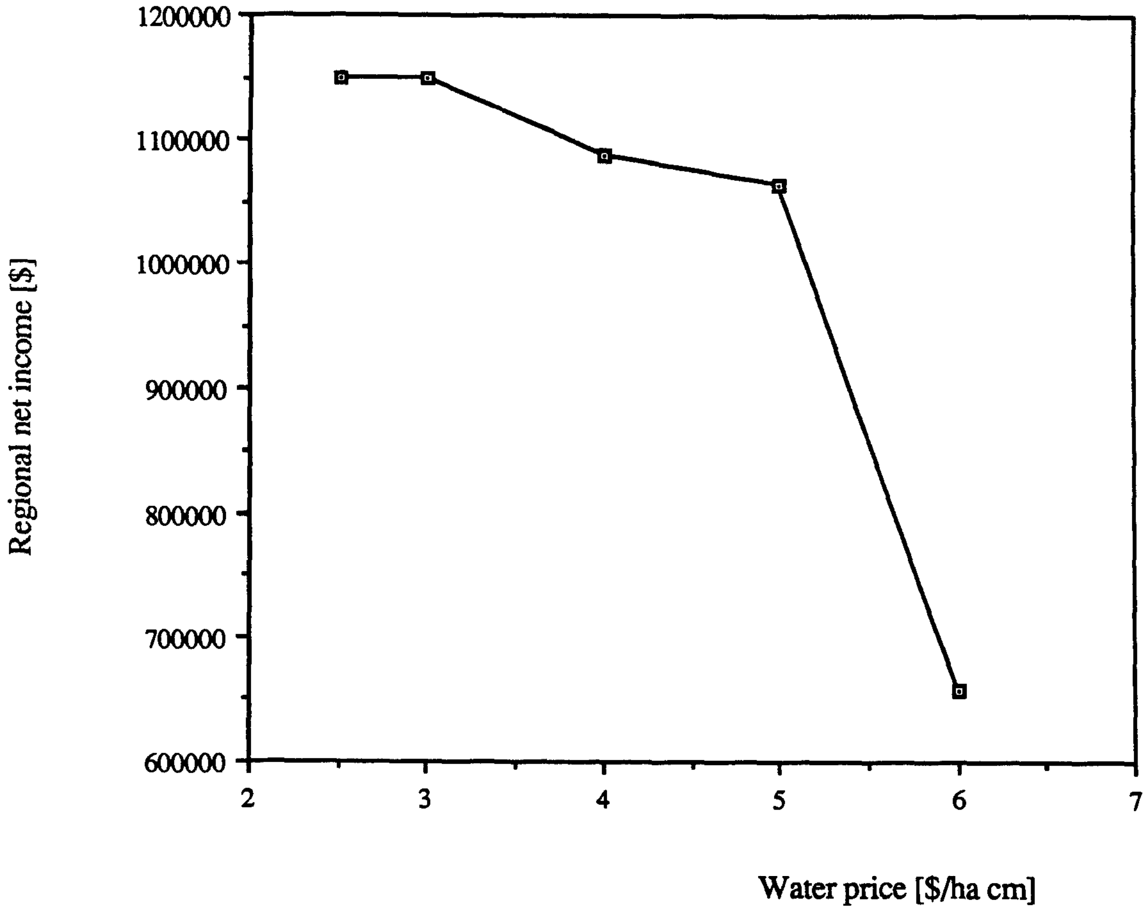


Figure 6:
Effect on drainage quolity and volume of different levels of drainage tax (steady state model)

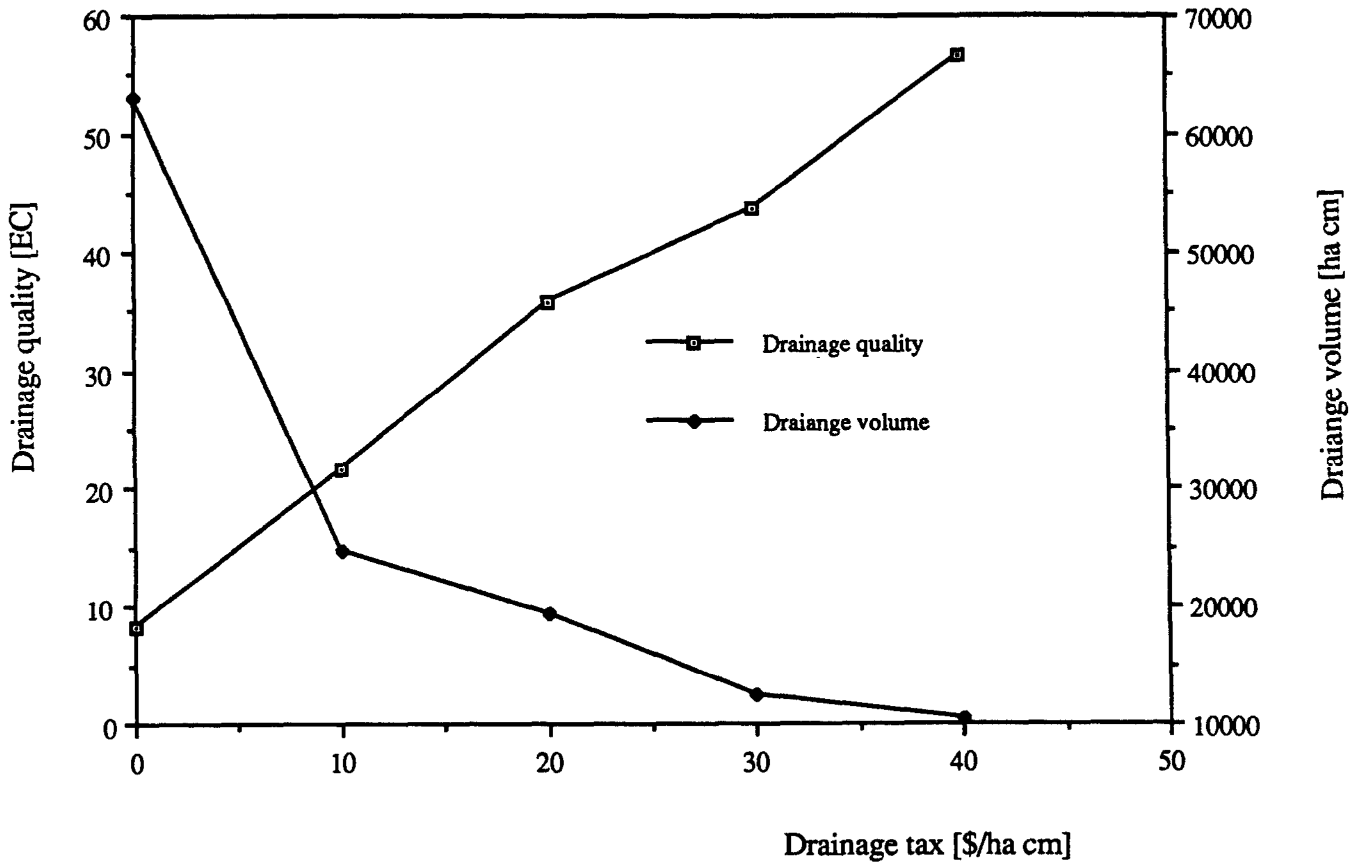


Figure 7:
Effect on drainage quality and volume of different levels of water prices (steady state model)

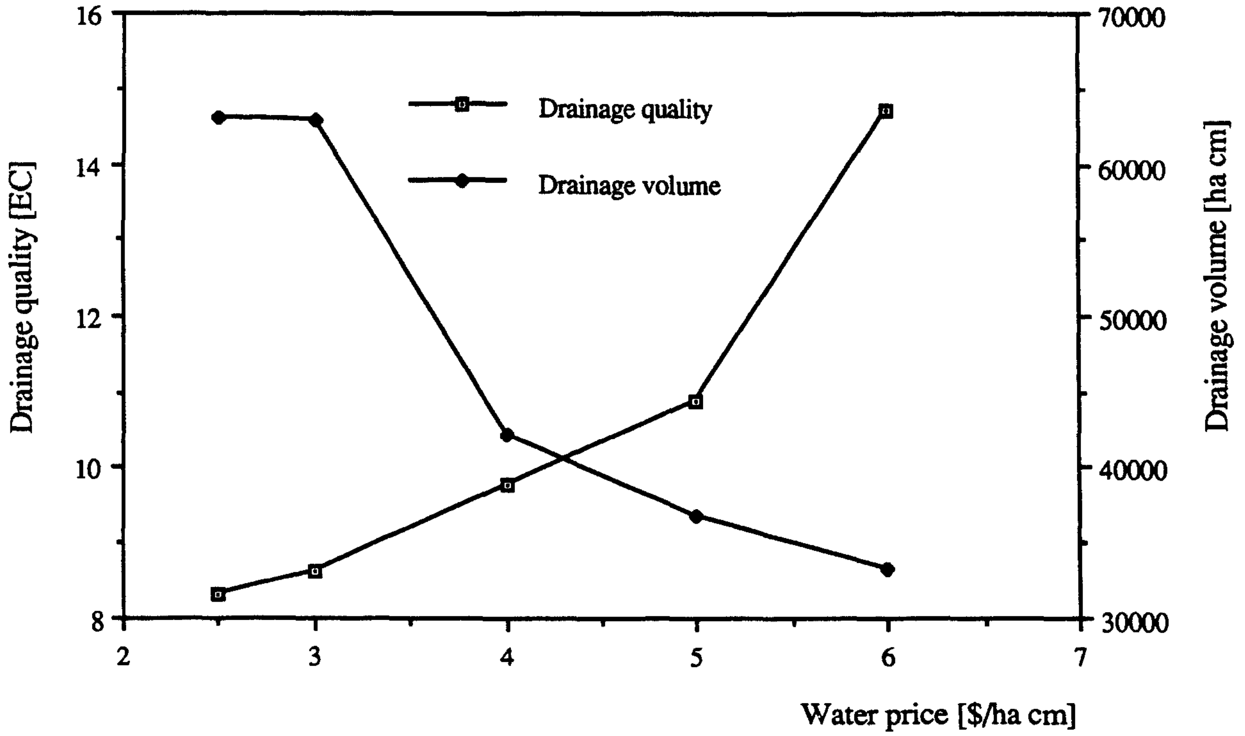


Figure 8:
Effect on net regional income of different surface water quota (dynamic model)

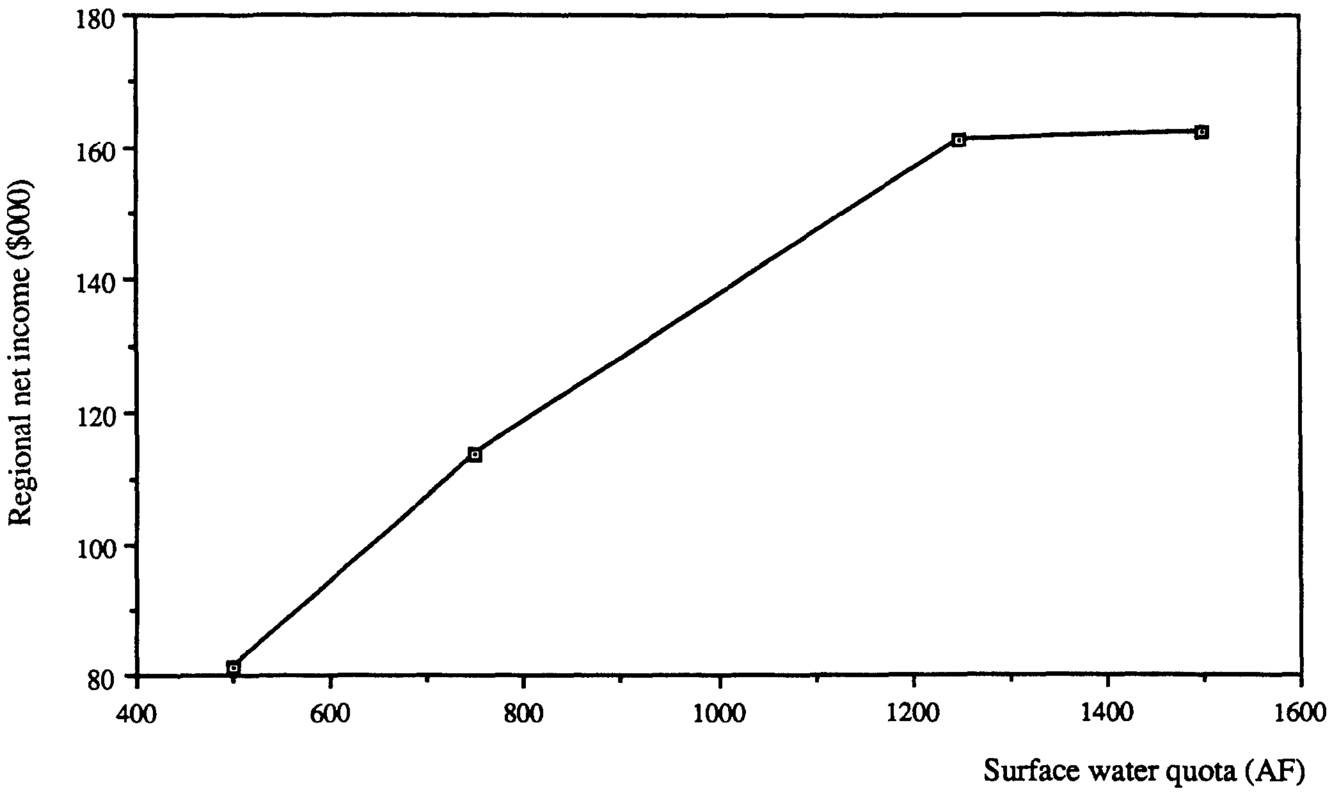


Figure 9:
Effect on regional net income of different
levels of drainage permits (dynamic model)

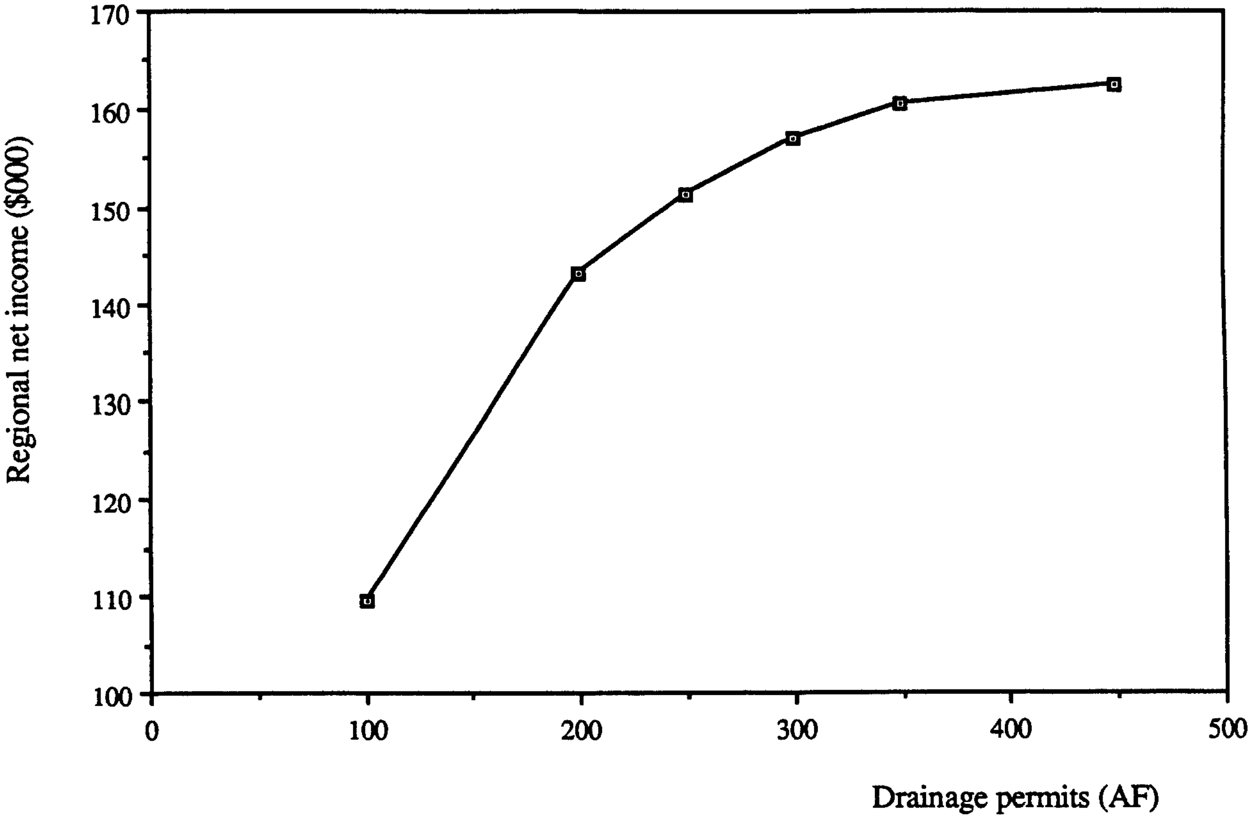


Figure 10:
Changes over time of discharged drainage
as affected by water quota and drainage
permits (dynamic model)

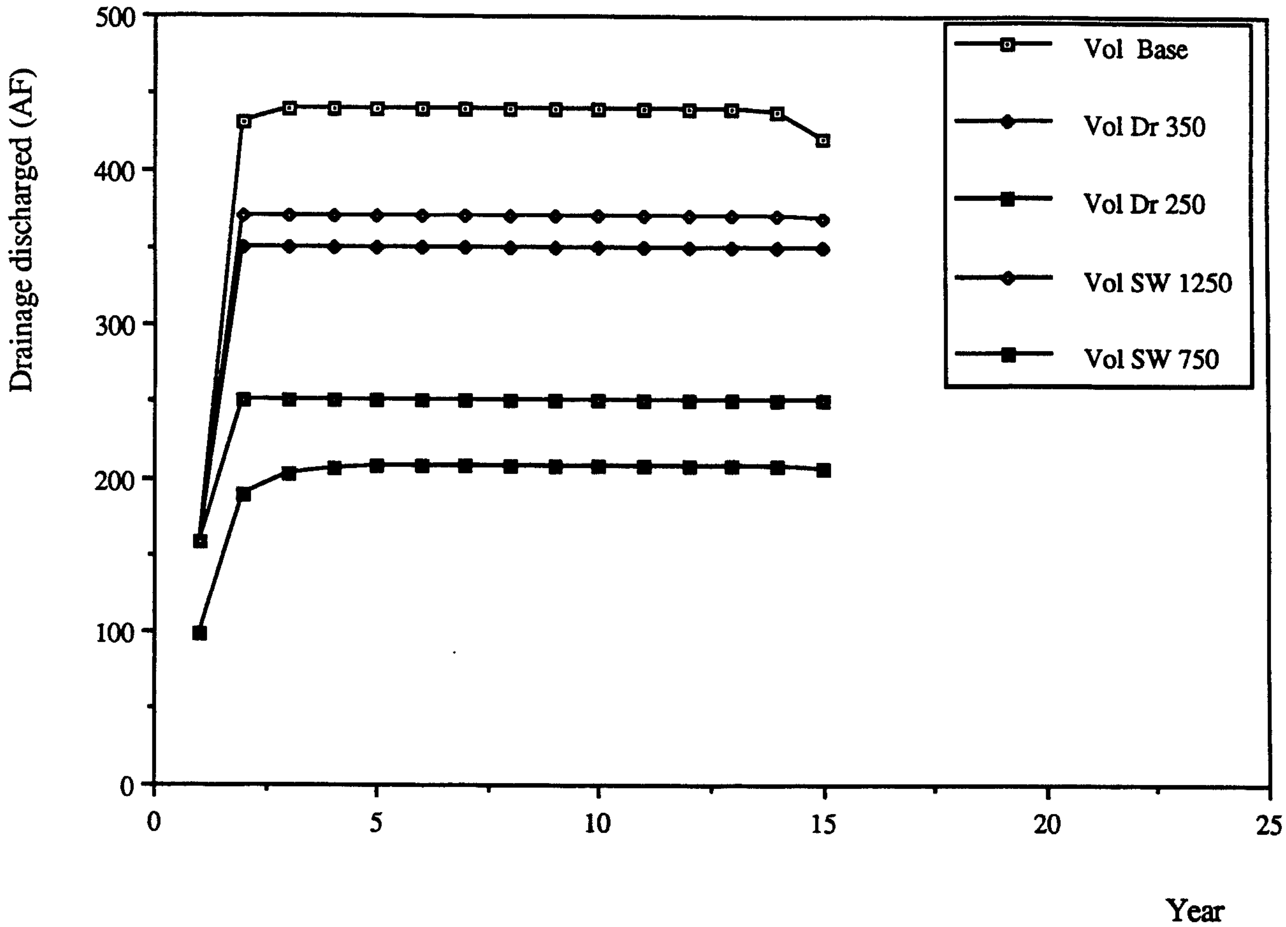


Figure 11:
Changes over time of initial soil salinity as
affected by waer quota and drainage permits

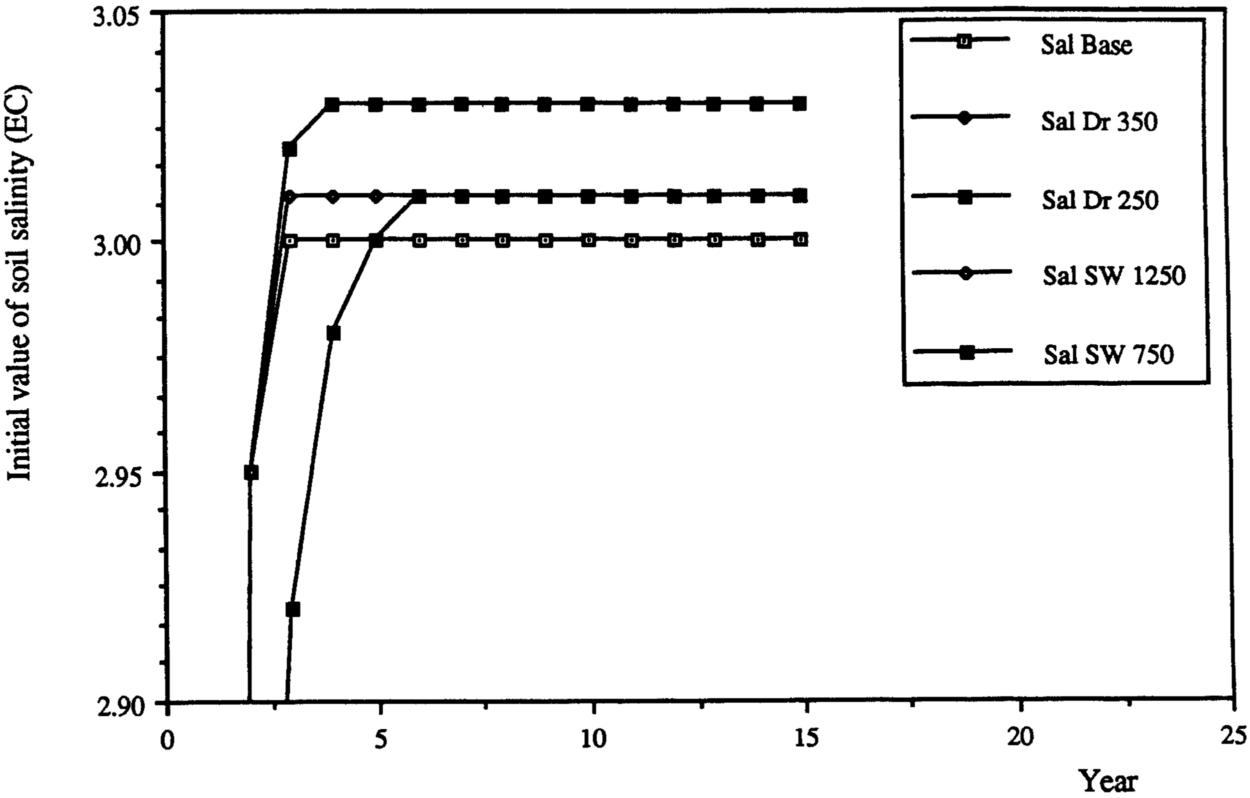
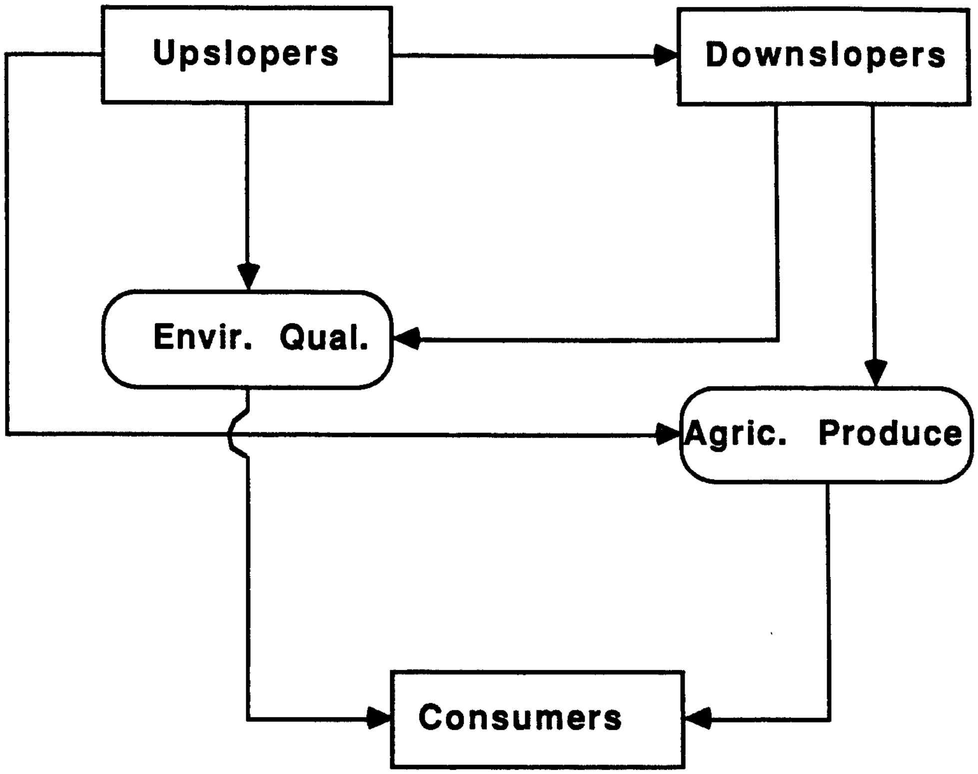


Figure 12:
The Framework for the Analysis-
The Game Parties and the System



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FLORIDA'S EXPERIENCE WITH MANAGING NONPOINT SOURCE PHOSPHORUS RUNOFF INTO LAKE OKEECHOBEE*

by

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FLORIDA'S EXPERIENCE WITH MANAGING NONPOINT SOURCE PHOSPHORUS RUNOFF INTO LAKE OKEECHOBEE

W.G. Boggess, E.G. Flaig and C.M. **Fonyo**[1]

Lake Okeechobee is the second largest freshwater lake contained in the contiguous United States with a surface area of 730 square miles and a drainage area of more than 4600 square miles (SWIM, 1989). Located in south central Florida (Figure 1), the Lake is the direct water supply for five municipalities, provides backup supply for the lower east coast of Florida, and provides ecological, recreational and irrigation benefits to many Users.z Lake Okeechobee is a shallow (i.e. average depth of 9 feet), highly productive, eutrophic lake which is in danger of becoming hypereutrophic due to excessive nutrient inputs, primarily phosphorus from agricultural activities.

The threat posed by phosphorus runoff to the Lake was first documented in a series of studies in the 1970s (Joyner, 1971, Davis and Marshall, 1975, and Federico, et. al., 1981). The latter study examined the trophic status of the Lake using a modified Vollenweider model which identified phosphorus as the limiting nutrient. The studies also determined that the Taylor Creek/Nubbin Slough (TC/NS) and Lower Kissimmee River (LKR) drainage basins (Figure 1) contributed 30% and 20% respectively of the phosphorus loads to Okeechobee, and 5% and 31% respectively, of the water inflows. Direct rainfall accounted for 39% of the water and 17% of the phosphorus.

Concurrent with Joyner's early study, the Governor called together a Conference on Water Management in South Florida in September, 1971. One of the conclusions of the conference was that the condition of Lake

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[2] In 1985-86 the combined recreational and commercial fishing industries generated $28.4 million in expenditures and sales (Bell, 1987). Bell also estimated that the lake's recreational user value was $8.3 million annually, or converting this to an asset value, the Lake Okeechobee fishery resource was valued at nearly $100 million. The lake also provides irrigation water for the sugarcane industry which is estimated to provide 18,000 jobs and to generate $1.3 billion annually of economic activity in the state (Mulkey and Clouser, 1988).

Okeechobee, the heart of both water quantity and quality in south Florida, should be improved (Special Project, 1976). The Governor's Conference was followed by a public hearing in 1972 sponsored by the Central and Southern Florida Flood Control District (renamed and rechartered as the South Florida Water Management District (SFWMD) in 1972)[3]. The results of this hearing, coupled with widespread public and governmental agency concern over the condition of Lake Okeechobee, prompted the Florida Legislature to establish and fund the Special Project to Prevent the Eutrophication of Lake Okeechobee in 1973. The final report published in 1976 identified the primary sources of phosphorus as high density dairy pastures and faulty dairy waste control systems. The report prioritized the TC/NS and LKR basins for implementation of phosphorus management plans.

More recent figures for the entire Lake Okeechobee Watershed confirm that agriculture is the dominant source of phosphorus entering the watershed (Fonyo, et. al., 1991). The largest sources of net phosphorus imports to the basin are improved dairy and beef cattle pastures (45.9% of the total), followed by sugar mills (14.9%), dairy barns (14.3%), sugarcane fields (13.5%), and truck crops (6.9%) (Table 1). Table 2 summarizes phosphorus imports into the Lake Okeechobee Watershed by material. Fertilizer constitutes 73.2% of the total, and dairy feeds account for 15.9%. Together, fertilizers and feed account for 93.5% of the annual imports of phosphorus and agricultural production is responsible for 98 % of the net phosphorus imports to the watershed.

The purpose of this paper is to describe and then examine what can be learned from Florida's 15 years of experience with trying to control phosphorus runoff from agricultural lands into Lake Okeechobee. Specific objectives are: (1) to provide a brief description of the natural system, (2) provide an overview and chronology of phosphorus management/control programs; (3) outline and describe the evolution of monitoring programs and

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[3] The 1972 Florida Water Resources Act (Florida Statutes, Chapter 373) assigned the management of water rights to the State, and created a system of five water management districts in the state based on hydrologic boundaries. The Act was based on the Model Water Code developed by Maloney et. al. (1972). The districts are governed by a board of directors appointed by the Governor and have their own property taxing authority. The districts are charged with managing and protecting water resources. Although the districts have a great deal of autonomy in dealing with water resource issues, they are subject to legislative mandates (e.g. SWIM Act) and to various state agencies with ultimate responsibility for water resource issues such as Florida Department of Environmental Regulation.

analysis; (4) outline the evolution of phosphorus control technologies and incentives for adoption, (5) examine the costs and impacts of the various programs; and (6) derive lessons and implications for other similar problems.

Background

One hundred years ago, south Florida fresh water circulated in a slow, rain-driven cycle (40-65 inches per year) of meandering rivers and streams, shallow lakes, and wetlands including unique saw grass marshes. Starting at a chain of lakes south of Orlando, water flowed into the Kissimmee River. The Kissimmee meandered 110 miles south into Lake Okeechobee. During wet seasons, water spilled over the lake's low southern rim, and flowed south across the Everglades saw grass in a 50-mile wide sheet moving at a rate of approximately one hundred feet per day toward Florida Bay.

Modification of the natural freshwater system in south Florida began in the late 1800s as investors began developing the area. Over the next 100 years, a series of development, drainage, flood protection, and water supply programs resulted in the construction of 1400 miles of canals and levees. The most important project was the federally funded, massive flood-control and water supply project known as The Central and Southern Florida Flood Control Project which was authorized by Congress in 1948. Major modifications included: (1) the channelization of the Kissimmee river into a 52 mile-long, 300 foot-wide, 60-foot deep canal known as C-38; (2) construction of the 25 foot-high, Herbert Hoover Dike encircling Lake Okeechobee and providing control over all inflows to and outflows from the Lake; and (3) creation of three water conservation areas south of Lake Okeechobee to store excess flood waters and to provide supplemental water supply. A series of canals, control structures and pumping stations are currently used to control freshwater movement south of Lake Okeechobee.

Agriculture first began to develop around Lake Okeechobee in the 1920s. Originally agriculture was limited by poor drainage and poor soils. Identification of micronutrient deficiencies in the Everglades Agricultural Area (EAA) led to a significant increase in production in the 1930s. Establishment of the sugar program in the 1960s led to a dramatic increase in sugarcane and winter vegetable acreage. It was during this period that water quality problems first began to develop south of the Lake.

Agriculture north of the Lake consists primarily of dairy and beef cow/calf operations with limited acreage of citrus and vegetable production. Dairying, the most important agricultural industry, first began to develop in Okeechobee County in the early 1950s. Originally the south Florida dairy industry had been concentrated around Miami, but urban development after World War II forced them to move. The south Florida dairy industry is now centered in Okeechobee County just north of Lake Okeechobee.

As a result of Central and South Florida Flood Control Project, the major components of the natural drainage system can be controlled somewhat independently. Given this degree of independence and the differential nature of the water quality problems, current concerns overwater quality in central and south Florida have manifested themselves as three separate efforts: (1) the Kissimmee River Restoration Project which aims to "restore" the natural meandering flow of the river through oxbows and wetlands (Loftin, et. al., 1990); (2) the Lake Okeechobee SWIM* plan which is designed to control nutrient loads in order to protect the lake's vital water supply, recreational, and ecological benefits; and (3) the Everglades SWIM plan designed to address concerns about the quantity, temporal distribution and quality of water released from the Everglades Agricultural Area (EAA) south through the Water Conservation Areas (WCAs) into the Everglades National Park (Everglades SWIM, 1990).

The latter concern has been the subject of litigation. In October 1988, the U.S. Attorney's office in Miami sued the SFWMD and the Florida Department of Environmental Regulation (FDER), charging that state and federal water-quality regulations had been violated by allowing agricultural runoff from the EAA to damage Loxahatchee Refuge and the Everglades National Park. The lawsuit has been contested with considerable fervor. District and United States scientists have met on several occasions to resolve issues of fact. In January 1991, recently elected Governor Chiles negotiated a 60 day stay in order to conduct an Everglades Summit to resolve the lawsuit. Numerous sessions have been conducted and the state legislature recently passed additional

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[4] In 1987 the legislature passed the Surface Water Improvement and Management (SWIM) Act (Chapter 373.451-373.4595 Florida Statutes). The act dictates that the five water management districts in Florida design and implement SWIM plans for priority water bodies. The act also established a trust fund to provide financial support through FDER.

legislation to help resolve the issues. The case is scheduled for trial in September, 1991. In the interim, the District has begun implementation of the Everglades SWIM plan.

The remainder of the paper focuses on efforts aimed primarily at controlling nutrient loads to Lake Okeechobee which have culminated in the Lake Okeechobee SWIM Plan.

Phosphorus Management/Control Programs

Based on the 1976 recommendations of the Special Project to Prevent the Eutrophication of Lake Okeechobee, initial nutrient control efforts focused on reducing phosphorus runoff from dairies in the TC/NS basins (Albers, et. al., 1991). The first program was a state funded project called the Taylor Creek Headwaters Program (TCHP) which began in 1978 with the objective to fence cows from waterways and determine the impact on stream water quality. The project was limited in scope, confined to the headwaters of Taylor Creek.

In 1981, federal funds were obtained under the Rural Clean Waters Program (RCWP) to address water quality concerns in the entire TC/NS basin. The goal of the TC/NS RCWP was to reduce phosphorus concentrations in water flowing into Lake Okeechobee from the basin by 50% by 1992 (NWQEP, 1989). The objectives were to implement BMPs and evaluate the impact on basin water quality. The state-funded TCHP project was combined with the RCWP program to provide additional funds for the implementation of BMPs. The SFWMD was given the responsibility for monitoring water quality to determine the efficacy of BMPs for phosphorus reduction beginning in 1978 and continuing to date (Flaig and Ritter, 1989).

The Lower Kissimmee River RCWP was initiated in 1987 to reduce agricultural nonpoint source pollution in the LKR basin. The objective was to implement BMPs for each dairy to reduce loadings from animal waste and fertilizer. The goal was to reduce phosphorus loads to the Lake by 43%. The specific phosphorus load reduction goal and the design of the BMPs for the dairies were ultimately based on the provisions of the Dairy Rule and SWIM Act.

In August, 1985, the Governor directed the secretary of FDER to direct a study of the conditions affecting Lake Okeechobee and to make recommendations for its protection and improvement. FDER formed the Lake Okeechobee Technical Advisory Committee (LOTAC I) which concluded that the phosphorus concentrations in the lake doubled between 1973 and 1984 and that the lake was losing its ability to assimilate phosphorus (LOTAC, 1986). LOTAC I produced a number of recommendations including that detailed agricultural BMPs should be planned and implemented in the TC/NS and LKR basins which would prohibit discharge of barn wash water and retain the runoff from high cow density areas for the 25-year, 24-hour storm. LOTAC I also recommended that a set of research and demonstration projects totaling approximately $8 million be conducted to examine fertilization practices, dairy ration formulation, chemical and biological treatment of barn wash water, and basic biogeochemical behavior of phosphorus in soil and water.

In August, 1986 the Governor issued executive order 86-150 directing the secretary of FDER to implement the recommendations of LOTAC I with regulations to be in place by May, 1987. The Florida Department of Agriculture and Consumer Services (DACS) was directed to complete a cost share program patterned after the TCHP by October, 1986. FDER, working with SFWMD, SCS, and dairy representatives drafted the "Dairy Rule" (F.A.C. 17-6.330 through 17-6.337) which became effective June, 1987. The rule specified that the dairies in the TC/NS and LKR basins had to implement specified technologies to prevent the discharge of barn wash water and to retain the runoff from high intensity areas for the 25-year, 24-hour storm. A total of 49 dairies (approximately 45,000 cows) came under the jurisdiction of the Dairy Rule. DACS secured funds from the legislature to cost share the construction.

The dairy industry requested, and were granted, a buyout program for dairies that chose not to comply with the dairy rule. Dairymen were offered a payment of $602 per cow (approximately half of the money was provided by SFWMD and half by the State) in return for a deed restriction prohibiting the property from being used for a dairy or any other concentrated animal feeding operation. The dairymen retained ownership of the cows and the property. A total of 17 dairies signed contracts for the buyout which will eliminate 12,721 cows from the basin.

The 1987 SWIM Act directed the SFWMD to protect the water quality of Lake Okeechobee and specified that the long term annual phosphorus load should be reduced to 397 tons (Chapter 373.451-373.4595, Florida Statutes). The SFWMD was required to develop a plan to meet this reduction by July, 1992. The

SFWMD developed an interim plan (SWIM, 1989) consisting of research and regulatory initiatives. The regulatory component of the SWIM plan is to be accomplished primarily by the implementation of phosphorus performance standards A performance standard of 0.18 mg per liter average annual, total phosphorus concentration was adopted for inflows to the lake. The standard was calculated by dividing the 397 ton target loading by the long-term water inflow to the lake. The 0.18 performance standard is applied to tributary discharges but not to runoff from individual properties. For dairies, the allowable discharge concentration for total phosphorus was set at 1.2 mg per liter based on calculations that the assimilative capacity of streams and wetlands would result in the 0.18 standard being met at the lake inflow structure. However, the dairies were exempted from permitting and enforcement under the SWIM plan since they were currently under the jurisdiction of the FDER Dairy Rule. For improved pasture land uses, which include dairy heifer and beef cow-calf operations, the standard is 0.35 mg per liter. Other land uses are required to remain at their historical levels, with the exception that land uses currently below the 0.18 standard are permitted to come up to the standard. All land uses other than dairy are currently subject to permitting and enforcement under the SWIM plan (Rule 40E-61, F. A.C.).

Monitoring Activities

The monitoring program in the Okeechobee basin has three basic purposes: (1) to determine the effects of alternative land management practices on downstream water quality; (2) to evaluate the trends in water quality over time; and (3) to monitor compliance with runoff concentration standards. The water quality monitoring program for the TC/NS and LKR basins has evolved over time due to changes in land management and agency requirements. Over the last 17 years the program has expanded from simple collection of water samples at major structures to a complex network that includes automated water samplers and *in situ* water quality monitoring devices.

Water quality monitoring first began in TC/NS in 1973 as part of an Agricultural Research Service (ARS) study to identify the impacts of drainage on open charnel water quality (Allen et. al., 1976). In 1978, the SFWMD took responsibility for evaluating the water quality impacts of BMPs employed under the TCHP and expanded the ARS monitoring network to include surface water sampling sites throughout the TC/NS basin (Flaig and Ritter, 1989). The objective was to collect data for identifying trends and quantifying, where possible, changes in surface water quality that occurred due to changes in land use and/or implementation of BMPs.

In 1986, the network was modified to monitor discharge water quality for each dairy to provide a higher degree of resolution for identifying trouble spots, and to monitor specific site performance of BMPs under the Dairy Rule (Flaig and Ritter, 1989). The monitoring sites included automated sampling stations on the dairies and tributaries to provide information for estimating loads, verifying and calibrating water quality models, and developing a more complete water quality record. The additional information provided state and federal agencies responsible for administering cost sharing under the TCHP, RCWP, and Dairy Rule programs with a means of determining the effectiveness of the cost share funds. Expanded networks were essentially complete within LKR by fall, 1987 and within the TC/NS by summer, 1988 (Flaig and Ritter, 1989).

The monitoring program was modified again in 1989 to support the regulatory aspects of the “Works of the District” rule formulated under the interim Lake Okeechobee SWIM plan (Flaig and Ritter, 1989). The objectives of the program are to evaluate the efficacy of BMPs, to provide background information for a surveillance monitoring program, and to provide on-going checks on compliance with the runoff concentration standards. The District also provides runoff water quality data for each dairy to FDER to assist their evaluation of the Dairy Rule.

Under the “Works of the District” rule a total phosphorus concentration standard was selected over a phosphorus load standard due to ease of implementation and greater correlation to changing land use management (Flaig and Ritter, 1989). A load standard requires precise field measurements for calculation of discharge at each site. Accurate flow measurements are difficult to obtain for streams with a low gradient, poor access, and poor stream measurement sections. In addition, nutrient loads from storm runoff are very sensitive to hydrologic variation and long term monthly or annual phosphorus loads would depend upon rainfall patterns and seasonal influences which would complicate enforcement.

The concentration standard has been converted into a regulatory criteria to provide a workable, attainable standard requiring minimal data collection (Flaig and Ritter, 1989). The components of the off-site performance standard are: (1) a total phosphorus concentration standard not to be exceeded on an average annual basis; and (2) a maximum total phosphorus concentration not to be exceeded when fewer than six samples have been collected. These values are baaed on the 50% probability that the annual off-site phosphorus concentration limitation will be exceeded. The first criteria defines an average annual standard by which to evaluate long term behavior. The second criteria provides a means to identify a serious problem with a limited record of water quality samples. These criteria have been formulated into an administrative rule for permitting and enforcement (Rule 40E-61, F.A.C.).

Monitoring Data Collected

Monitoring activities in the TC/NS and LKR basins consist of surface water sampling, rainfall measurement, stream stage and ground water stage measurement (Flaig and Ritter, 1988). Surface water grab samples are collected weekly at all dairies and tributaries in both TC/NS and LKR. Samples are analyzed for nitrogen and phosphorus species and physical parameters: pH, dissolved oxygen, conductivity, turbidity, and color. Similar samples are collected and analyzed for quality assurance and quality control. In addition, the dairies are required under the Dairy Rule to sample phosphorus concentrations in groundwater on a quarterly basis.

The costs of the monitoring program area major concern in the implementation of the program. Water sample collection and analysis for total phosphorus range from $50 to $95 per sample. The cost increases where sample sites are difficult to reach, which is common with dairy discharge locations. Assuming two discharge locations, weekly sampling, and a cost of $50 per sample, monitoring costs would exceed $5000 per year for a dairy. The SFWMD is responsible for monitoring surface water discharges. The dairies are required by FDER monitor ground water quality on a quarterly basis.

Phosphorus Management Technologies and Incentives

To be technically effective, phosphorus control practices have to physically change phosphorus flows through the system (Figure 2). Phosphorus flows can be impacted in four general zones in the system: (1) phosphorus material imports (source reduction); (2) onsite treatment/storage; (3) phosphorus product exports (export enhancement); and (4) offsite treatment/storage. Control practices that operate in zones (1) and (3) may be classified as phosphorus use management practices, whereas those operating in zones (2) and (4) are phosphorus waste management practices. Practices operating in all four zones have been proposed and studied as options for controlling phosphorus runoff into Lake Okeechobee. However, to date, only source reduction and onsite treatment/storage technologies have been implemented.

Dairies in the Lake Okeechobee basin are currently implementing the third generation of phosphorus management BMPs with a possible fourth generation on the horizon. The various phases of BMP implementation tended to overlap and dovetail together making it difficult to quantify precisely the efficacy of the various stages of BMP implementation. A brief, chronological discussion of the four generations of technologies and incentives follows.

In the early 1970s the State and SCS encouraged the development of lagoon systems to capture milking barn wash water and to direct the effluent into seepage fields. The second generation of BMPs was associated with the TCHP program and consisted of pasture improvement and waterway protection to eliminate the direct loading of wastes (i.e. onsite storage). The TCHP program, initiated in 1978, was a small scale trial program limited to the headwaters of Taylor Creek which accounted for only 1 % of the water, but 12% of the phosphorus entering Lake Okeechobee via S191 (Albers et. al., 1991). The program was voluntary, with the state providing 100% cost sharing.

The TC/NS RCWP program was approved and funded in 1981. The primary goal of the TC/NS RCWP was to extend the scope of the TCHP by contracting with all twenty-four of the dairies in the drainage basin to implement pasture and waste management BMPs to reduce nutrient runoff (beef cow/calf farms that had been extensively drained and lands within a quarter mile of waterways were also targeted). Specific BMPs implemented included: fencing cattle from waterways, establishing vegetative filter strips along waterways, providing cattle crossings over streams and ditches, providing shade structures for cattle away from streams and waterways, and recycling barn wash water (RCWP, 1990). The program was voluntary, with 75% federal cost sharing. The TCHP program was combined with the TC/NS RCWP in 1981 and the state funds were used to leverage the federal cost sharing.

The LKR RCWP began in 1987. Originally it was envisioned as an extension of the TC/NS RCWP with the primary focus being to improve pasture and nutrient management on dairy and beef cow/calf farms via voluntary participation with federal cost sharing. However, in 1987 the state passed both the Dairy Rule and the SWIM Act which mandated implementation of technology standards by 1991 and performance standards by 1992. Faced with these new regulations, dairymen shifted their focus from low cost, pasture and nutrient management BMPs (second generation) towards more mechanical capture and removal methods (third generation) that would satisfy the technology standard specified in the dairy rule (RCWP, 1990). Thus, the incentive structure under the LKR RCWP has evolved from voluntary, with federal cost sharing into a technology based standard, with primarily state cost sharing.

The dairy rule represents the third generation of BMPs. Passed in June, 1987, the dairy rule specifies that all dairies were required to submit construction permit applications along with BMP designs by June, 1989. Within 18 months of construction permit issuance, the BMP construction must be completed and an operating permit obtained from FDER. In order to satisfy the technology standard, the dairy rule designs were required to: (1) collect all wastewater and runoff from barns and high intensity areas for a 25-year, 24-hour storm; (2) dispose of nutrients by approved methods, particularly land application by irrigation (3) fence cattle from waterways; and (4) monitor water quality discharges to insure system adequacy. The dairy rule technologies formalize the earlier focus on onsite storage enhancement and expand the focus to include nutrient recycling and source reduction. In addition, pilot onsite treatment options (chemical and biological) have been evaluated as have been options for exporting dairy wastes as a soil amendment.

Typical dairy rule designs call for constructing perimeter ditches around the hams and high intensity **areas**[5] to collect all of the runoff from a 25-year, 24-hour storm. The runoff is processed through a two-stage lagoon system and then applied, via center pivot irrigation systems, to forage production sprayfields. The sprayfields have to be sized to insure that annual application rates of phosphorus don't exceed the forage crop's uptake, generally 60 pounds per acre per year. In addition, the dairies must have sufficient land available for land spreading of solids collected.

The State initially planned to provide 75% cost sharing of construction costs under the dairy rule. However, escalating construction costs from an initial cost share estimate of $250,000 to over $1,000,000 per dairy, resulted in a revised sliding scale for cost sharing. DACS currently provides cost sharing ranging from $233 to $433 per cow depending upon the size of the dairy, with the smaller dairies receiving the higher rate (Conner, 1989). This sliding scale reflects the significant construction cost economies of scale enjoyed by large dairies (i.e. 1500 cows) relative to small dairies (i.e. 350 cows) (Giesy, 1987). The net result is that cost sharing under the dairy rule ranges from 30% to 75%.

The companion dairy buyout program provides an alternative economic incentive-based option to the technology baaed standard. A fixed payment of $602 per cow is offered (based on political "fairness" or equity **concerns).**[6] Thus, there is no "market" or competitive bidding for the easements. However, one would hypothesize that smaller, less efficient and/or dairies with particular location or drainage problems would be more apt to accept the buyout option. One-third (17) of the dairies have opted for the buyout. Fifteen of the seventeen dairies were relatively small with an average herd size of just over 650 cows versus an 1025 cow average for the thirty-two dairies that chose to comply. In addition, one operator decided to close two large

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[5] High intensity areas are defined as areas of concentrated animal density generally associated with milking barns, feedlots, holding pens travel lanes and contiguous milk herd pasture where the permanent vegetative cover is equal to or less than 80 percent.

[6] The $602 figure reflects an approximate 75% cost share of an estimated cost of $800 per cow to move a dairy 500 miles.

dairies (2900 cows) because he had inadequate land available for spreading of wastes. Dairymen had knowledge of the specific phosphorus concentrations in runoff from their lands prior to the final buyout deadline.

A fourth generation of BMP implementation looms on the horizon. Under the interim Lake Okeechobee SWIM plan, phosphorus concentration performance standards have been specified for the dairies and other land uses in the basin. The performance standards are not currently being permitted or enforced on the dairies, but the option exists. A few dairies, perhaps in anticipation of enforcement of the performance standards have chosen to go to total confinement or semi-confinement dairy systems which reflects the fourth generation of technologies.

The evolution of phosphorus control practices reflects a trend toward increasing collection and treatment of dairy wastes. The percentage of the dairy wastes being collected steadily increases from approximately 25% under the first generation of BMPs, to 65% under low-tech dairy rule designs, to 85% under the high-tech dairy rule designs, to essentially 100% under total confinement. In addition, the level and type of treatment of the wastes also increases from first generation simple lagoon/drain field to two-stage lagoon with controlled land application to potentially fourth generation chemical or biological treatment. The net effect has been a steady conversion of a primarily nonpoint source to a point source.

Uncertainty due to lack of information about the extent and mechanics of the phosphorus runoff problem and about the efficacy of alternative control technologies led to a cautious, evolutionary application of control technologies. The evolution of incentives for participation reflected the same uncertainties. Economic and "fairness" (equity) concerns dominated early programs. Whereas, efficacy and the certainty of effect have dominated more recent programs. As a result, incentives have slowly evolved from purely voluntary with 100% cost sharing, through voluntary with steadily decreasing cost sharing, to regulatory technology based standards with partial cost sharing, to the potential threat of performance based standards.

With the exception of the dairy buyout and cost sharing, economic incentives have not been employed. In the early stages of the problem, concerns over equity and in finding a "fair" solution limited the use of economic incentives to cost sharing. In the later stages, economic incentives were generally considered to be too uncertain in their effectiveness and strict technology and performance standards were imposed instead.

Three additional types of economic incentives would appear to be feasible options. One would be to convert the dairy buyout or easement program from a fixed amount to a market or bid system that would reflect the differential costs of compliance and values of the dairying property right across dairies of different sizes, locations, and management capabilities. This approach would combine the desirable efficiency aspects of economic incentives with the high certainty of efficacy sought by environmentalists.

Secondly, since over 90% of the phosphorus entering the basin is accounted for in fertilizers and feeds (Table 2), an inputs tax would be relatively easy to implement and administer. However, an inputs tax provides only indirect incentives to control emissions and thus as a sole approach would probably not be an effective means of achieving the rather stringent water quality goals dictated in the SWIM plan. It does provide a relatively cheap program to implement and administer, and it would provide a source of funds for companion cost sharing or abatement programs.

An emissions tax would provide more direct incentives for dairymen to control runoff. However, as discussed in the water quality monitoring section, runoff loads are very difficult to quantify and thus concentration standards and monitoring protocols have been developed for implementing the performance standards. Emissions taxes or tradeable emission permits could conceptually be based on the same concentration measurements (Segerson, 1988).

Summary of Costs and Impacts

Formal cost effectiveness calculations for the various programs or for the implementation of specific BMPs are complicated by several factors. First, the various programs and expenditures have been intertwined making it difficult to separate overall expenditures by program. Second, it is difficult to quantify the impact of specific changes in land use due to lags in effects, variations in rainfall, and overlapping practices. Third, many of the programs for which expenditures have been made are still in process - many of the dairies that accepted the buyout have yet to close and construction is still underway for many of the remaining dairies. It will take another year or two for all of the practices to be implemented and several years before the impacts can be measured. However, it is possible to trace out the history, source and magnitude of expenditures to date and to examine overall changes in water quality trends.

Program Costs

Nearly $33 million has been spent over the past ten years on programs to control phosphorus runoff from agricultural lands north of Lake Okeechobee (Table 3). Various government sources have provided approximately three-quarters of the total with farmers providing the balance. Expenditures for research, permitting, monitoring and enforcement are not included in the government total. Likewise expenditures for roofed structures and for operation and maintenance of the BMPs are not included in the farmers' total.

The State provided $15.55 million (63%) of the $24.6 million government total, the SFWMD provided $5.95 million (25%), and the federal government $3.14 million (12%). However, the federal government provided 82% of the government funding for the RCWP.

The breakdown of expenditures between the RCWP and the Dairy Rule are rather arbitrary since the two programs overlapped beginning in 1987. A total of $2.13 million was spent by the government and $435,277 by farmers under the TC/NS RCWP prior to the Dairy Rule. An additional $4.55 million has been expended under the auspices of the TC/NS-LKR RCWP since 1986. Much of this was spent in the LKR on practices required by the Dairy Rule which go far beyond the original RCWP goals pasture and nutrient management. This shift in emphasis is reflected in the difference in the average cost of BMPs installed in the two basins. In TC/NS, 27,897 acres were served by BMPs at a total cost of $1.72 million or $61 per acre. In the LKR, 6,926 acres were served by BMPs at a total cost of $3.16 million or $456 per acre (RCWP, 1990).

The dairy rule and dairy buyout programs have been funded without federal support. The state government provided the majority of the finding, although the SFWMD provided nearly half (49%) of the dairy buyout cost share (Table 3). Construction costs for the Dairy Rule plans range from $418 to $1086 per cow with an average cost of $659 per cow. Two dairies elected to construct total confinement barns at an approximate cost of $1200 per cow. Total government cost share has averaged $401 per cow under the dairy rule and $602 per cow under the dairy buyout.

Inpacts - Monitoring Data Analysis

The ecological health of Lake Okeechobee has been related to the total phosphorus (TP) concentration in the pelagic zone of the Lake (Federico, et. al. 1981). Where the concentration is below 50 mg/m^3 the Lake is considered healthy. Since the early 1970s the concentration appears to have risen steadily (Figure 3). In recent years the concentration has fluctuated dramatically from year to year. The in-lake phosphorus concentration shows little correlation with phosphorus loading (Figure 3). The poor correlation is due in part to fluctuating Lake stage and resuspension of bottom sediments which is common in shallow lakes. Although the long term health of the Lake is linked to the load, there is little year-to-year correlation between load and in-lake concentration.

There is also no clear pattern in the time series of annual loads for the tributaries TC/NS (S191) and LKR (S65E, S154) (Figure 4)[7]. The calculated loads at the basin scale are very sensitive to runoff volume. In particular, storm events following long antecedent dry periods tend to produce large TP flushes. In this region where tropical storms and long dry seasons are typical, there is rarely an average year. Consequently it is difficult to relate changes in phosphorus load to changes in land management. Experience has shown that the TP concentration in runoff is a function of cow density and proximity to open water runoff concentrations from lagoons range from 20 to 40 mg/l, while runoff from intensive pastures range from 2 to 5 mg/1 and unimproved pasture runs less than 1 mg/l.

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[7] T tests were performed comparing the mean loadings during the period 1973-1979 with the mean loadings during the period 1980-1989. The means loadings from TC/NS were 43 tons lower during the 80s (113.5 tons) than during the 70s (156.9 tons). However, the coefficient of variation was 0.44 and the difference was significant at only a 15% confidence level. Changes in loadings from S154 and S65E were not significant at levels less than 25%. Interestingly, average total loadings from all basins other than S191, S154 and S65E were essentially unchanged from the 1970s (304.7 tons) to the 1980s (298.8 tons) if the impact of the IAP is ignored. The IAP was initiated in 1979 and has been credited with reducing average TP loadings to the Lake by 10 tons per year (SWIM, 1989).

The long term trend in total phosphorus concentration in tributary runoff is a useful metric for evaluating land use change. The time series of TP concentration for runoff from TC/NS (S191) for the period 1973-1991 is presented in Figure 5. There are three distinct periods in the data. During the mid-1970s, cow numbers were increasing and water quality was steadily decreasing. This period corresponded with the Special Project Report in 1976 documenting that phosphorus was the limiting factor in the Lake and identifying the dairies as the primary source. During the late 1970s and early 1980s the “dairy phosphorus problem” began to receive a lot of attention resulting in the TCHP in 1978 and the TC/NS RCWP in 1981. Trend runoff concentrations of TP at S191 were essentially unchanged during this period. Under the RCWP, BMPs began to be implemented in the TC/NS basin beginning in 1983 and the result has been a significant downward trend during the 1980s. A similar trend is evident in the runoff concentration data from the Taylor Creek Headwaters area (Figure 6).

Median TP concentrations in runoff from TC/NS peaked at approximately 1.1 mg/l around 1980, since then they have declined by about 50% to between 0.5 and 0.6 mg/l. A similar decline in absolute terms is needed to reach the 0.18 mg/1 standard that has been established by the SFWMD. Most of the decline to date can be attributed to second generation BMPs installed under the TC/NS RCWP. It is too early to assess whether the combined effects of the Dairy Rule and the buyout will be sufficient to reach the target concentration at S191. At present, only six of the sixteen dairies in the basin that chose to comply with the Dairy Rule have fully implemented the Dairy Rule technologies and several of the ten dairies that accepted the buyout have not yet closed.

The long term trends in TP concentration from the LKR (S65E, S154) tell a different story (Figures 7 and 8). Runoff TP concentrations have been increasing steadily since 1975. In S154 (i.e. a small subbasin in the LKR) the increase has been significant and TP concentrations now are similar to those observed at S191. Concentrations at S65E, which represents the majority of the drainage from the LKR, are much lower with seasonal medians of less than 0.1 mg/l although peak concentrations have reached 0.5 mg/l. These concentrations reflect the greater volume of runoff passing through S65E and the lower density of dairy cows in the basin.

The increasing trends in TP concentration in runoff from the LKR can be attributed to several factors. First, cow numbers have continued to increase in the basin particularly in S154, during the 1980s. Second, it wasn't until 1987 that BMPs began to be installed under the LKR RCWP. By that time the Dairy Rule had been passed and the majority of the dairymen waited until their dairy rule designs were approved before they began to implement BMPs. The result is that a higher percentage of the dairies in the LKR have completed construction of their dairy rule designs (9 out of 12) compared to only 6 out of 16 in the TC/NS. Seven dairies in the LKR accepted the buyout.

Monitoring data from the individual dairies indicate that phosphorus concentrations in runoff increase during and shortly after construction of the dairy rule facilities. Thus, the short run impact of the dairy rule has been to increase phosphorus concentrations slightly. These effects are reflected in the overall runoff concentrations at stations TCHW 18, S65E, and S154 from late 1989 to 1991. Careful examination of Figures 6, 7, and 8 reveals that peak concentrations have tended to persist for longer periods of time during the construction phase and that the apparent trend is increasing slightly.

Overall, the results from the monitoring program indicate that the BMPs have improved water quality, particularly in the TC/NS basin. It is clear that water quality can be improved by practices that enhance soil storage, reduce P imports, and reduce availability of P to surface water discharge. However, runoff concentrations at many of the tributaries still exceed the 0.18 standard. More time is needed before the impact of the dairy rule and dairy buyout programs can be assessed.

Implications

One of the most obvious implications of the Lake Okeechobee experience is that programs designed to solve complex, nonpoint pollution problems are going to be evolutionary in terms of their complexity, rather than revolutionary. The political process of dealing with the uncertainty and lack of information about the problem and alternative solutions, equity concerns (including property right / takings issues), and administrative inflexibility once programs are put in place, all but guarantee a cautious, step-by-step approach. In the case of Lake Okeechobee, key components of the nonpoint programs have evolved in complexity over time including technologies, monitoring programs, and incentive mechanisms.

The evolution of technologies is in effect converting a primarily nonpoint source into a point source. Likewise, monitoring programs have evolved in purpose and design from an initial focus on problem assessment, to measuring efficacy of practices, and finally to providing a basis for implementing and determining compliance with performance standards. Finally, incentive mechanisms have evolved from purely voluntary with full cost sharing, to voluntary with partial cost sharing, to implied regulatory threats, to a technology based standard with cost sharing, to finally a performance based standard with no cost sharing. However, the threat of potential regulation throughout the process stimulated high levels of "voluntary" participation.

The second major implication is that communication and cooperation are essential if complex nonpoint problems are to be solved. Participation in the program by the dairies was greatly assisted by clear documentation that phosphorus loads affected the health of the Lake, and that dairies were the primary source of the problem. Likewise, although the SFWMD is often perceived as the "bad guys", the presence of monitoring and regulator staff in Okeechobee County greatly improves communication and understanding, particularly since the requirements of the landowners have continued to evolve over time. Finally, the TC/NS - LKR RCWP has experienced an unusual degree of cooperation between federal, state, district, and county governments as well as with the dairymen, which has been critical to the success of the program.

The third major implication is that traditional textbook economic incentives (emission taxes) have not been utilized in the Lake Okeechobee programs and may not be viable alternatives for many nonpoint source problems due to the uncertainty of effect, political aversion, administrative inflexibility, and monitoring (measurement) problems (Anderson, et. a1., 1990, Baumol and Oates, 1988). A broader concept of economic efficiency that accounts for the reality of deferential political and administrative costs associated with alternative incentive mechanisms needs to be encouraged. Economic incentives have and can continue to play a role in the Lake Okeechobee situation, however. Input taxes can be used to raise revenues to offset the costs of abatement, cost sharing can be provided, property right easements can be purchased, and marketable permit systems may be feasible in some circumstances.

The fourth major implication is that before emissions can be taxed or emission permits traded, emissions must be measurable. For many nonpoint source pollutants this is not technically or economically feasible. Measuring nonpoint source loadings is particularly difficult and expensive. However, the SFWMD has developed procedures to economically monitor nonpoint concentrations which are being used as the basis for assessing compliance with performance standards. Concentration measurements may also provide a feasible basis for implementing a marketable permit system or, if the political constraints can be resolved, an emissions tax.

The fifth major implication is that the combination of incentives, timely research and demonstration projects, and flexibility to respond has resulted in cost effective results. The formulation of performance standards in the Lake Okeechobee SWIM plan and the potential threat of enforcement is a critical factor in stimulating the development of a market for composting dairy wastes as a soil amendment and in the reduction in phosphorus content of dairy feed rations. Unfortunately, the performance standards are coming on the heels of a technology standard which has already limited the flexibility of the dairies to respond.

The power of the market was also exhibited indirectly in the Lake Okeechobee dairy buyout as the higher cost and/or dairies with higher discharge concentrations were selectively attracted to the program. The efficiency of the program would have been enhanced if a competitive bidding system had been employed.

The final implication is that nonpoint source problems are generally going to be addressed in a frost effectiveness context (Baumol and Oates, 1975) due to the difficulty of measuring benefits, uncertainty about key parameters of the problem, and the political preference for specifying specific targets (e.g. the SMIM Act's 397 ton target for Lake Okeechobee). However, cost effectiveness calculations are extremely complex in the case of most nonpoint source problems due to the evolutionary aspect of technologies and incentives, and the dynamics of the system including lags in effect and stochastic effects.

An accurate cost effectiveness assessment of the Lake Okeechobee nonpoint source programs is impossible at this point in time. However, preliminary results are consistent with two common characteristics of pollution control programs. First, the marginal cost of reducing emissions increases exponentially. Preliminary results from the TC/NS RCWP indicate that roughly a 45% reduction (0.5 mg/l) in seasonal median trend phosphorus concentrations was achieved at a cost of approximately $100 per cow. On the other hand, the Dairy Rule and buyout programs cost $400 and $602 per cow respectively, and the hope is that the median trend concentrations will fall another 0.4 mg/l. Second, increasing the reliability of nonpoint source regulations (i.e. a concentration standard which must be met ninety percent of the time rather than on average) would drive up costs dramatically as evidenced by the peak concentrations in the monitoring data.

Future Directions

Recently, the Chesapeake Bay Nonpoint Source Evaluation Report (1991) recommended that efforts to clean up the Bay: (1) take a mass balance approach, (2) employ a systematic planning framework, (3) target problem areas, (4) utilize a mix of regulatory and nonregulatory mechanisms, and (5) shift from using the term BMPs to Best Management Systems (BMSs) to reflect a more comprehensive, systems approach. The University of Florida is currently working with the SFWMD to assist them in developing a final SWIM plan for Lake Okeechobee that is consistent with the majority of the Chesapeake Bay Report recommendations.

A geographic information system (GIS) based, decision support system is being developed to assist District managers in evaluating alternative nonpoint source control plans. The system, dubbed LOADSS, takes a mass balance approach and provides a systematic planning framework for evaluating both pollution reduction and abatement practices. The GIS structure allows for spatial evaluation and targeting of phosphorus control practices. The purpose of LOADSS is to provide information on the cost effectiveness of alternative plans for achieving the 397 ton target. This information will be utilized along with evaluations of alternative incentive mechanisms to formulate the final Lake Okeechobee SWIM plan. The final plan will likely incorporate a combination of pollution reduction and abatement practices and a mix of incentive mechanisms.

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Table 1. Sources of annual net phosphorus imports to the Lake Okeechobee watershed.

Basin Activity	Annual Net P Import (tons/yr)	% Total Net P (%)
Nonpoint Source Activities		
Improved pasture	2736	45.8
Sugarcane	807	13.5
Truck Crops	412	6.9
Other agricultural	106	1.8
Urban	75	1.3
Total Nonpoint Sources	4136	69.3
Point Source Activities		
Dairy	850	14.2
Sugar mills/refineries	907	15.2
Sewage treatment plants	74	1.2
Total Point Sources	1831	30.7
Total All sources	5967	100.0

Source: Fonyo et. al. 1991.

Table 2. Summary of imports of phosphorus-containing materials to the Lake Okeechobee watershed.

Material	P Import (tons/yr)	% Total P
Fertilizer (P_2O_5)	5379	73.2
Feed supplements-beef	326	4.4
Feed-dairy	1168	15.9
Replacement heifers-dairy	16	0.2
Detergent-dairy	6	0.1
Sugarcane	304	3.1
Food and detergent-human consumption	145	2.0
Total Annual P Import	7344	100.0

Source: Fonyo et. al. 1991

Table 3. Costs of programs for controlling phosphorus runoff from agricultural lands north of Lake Okeechobee.

	Source of Funds ($)					
Programs	Federal	state	SFWMD	Total Government	Farmer	Total All Sources
RCWP No. 14	3,143,042	310,119	400,000	3,817,161	448,920	**4,302,081**[h]
Dairy Rule	-	**11,339,448**[i]	1,800,000[j]	13,139,448	**5,088,067**[k]	18,227,515
Dairy Buyout	-	3,904,368	3,751,456	7,655,824	**2,518,758**[l]	10,174,582
Total	3,143,042	15,553,935	5,951,456	24,612,433	8,055,745	32,704,178

Sources Rural Clean Water Project No. 14, Annual Progress Reports 1988, 1989 and 1990. Florida Department of Agriculture and Consumer Services.

[h]**$2,567,598** ($2, 132,321 government cost share and $435,277 farmer cost share can be apportioned to the TC/NS RCWP prior to the Dairy Rule.) The remaining $1,734,483 can be apportioned to the LKR RCWP - which has been implemented in conjunction with the Dairy Rule. (Figures are based on 1988, 1989 and 1990 RCWP No. 14 annual progress reports.)

[i]**Includes** $2,259,881 that was administered through the **RCWP.**

[j]**Does** not include research or monitoring costs.

[k]**Based** on estimated total construction costs for eligible items. Cost of ineligible items such as roofed structures are excluded as are operation and maintenance costs. Includes $553,002 of farmer cost share under the RCWP.

[l]**Estimate** based on 12,721 cows at $198 per cow (i.e. $800-602).

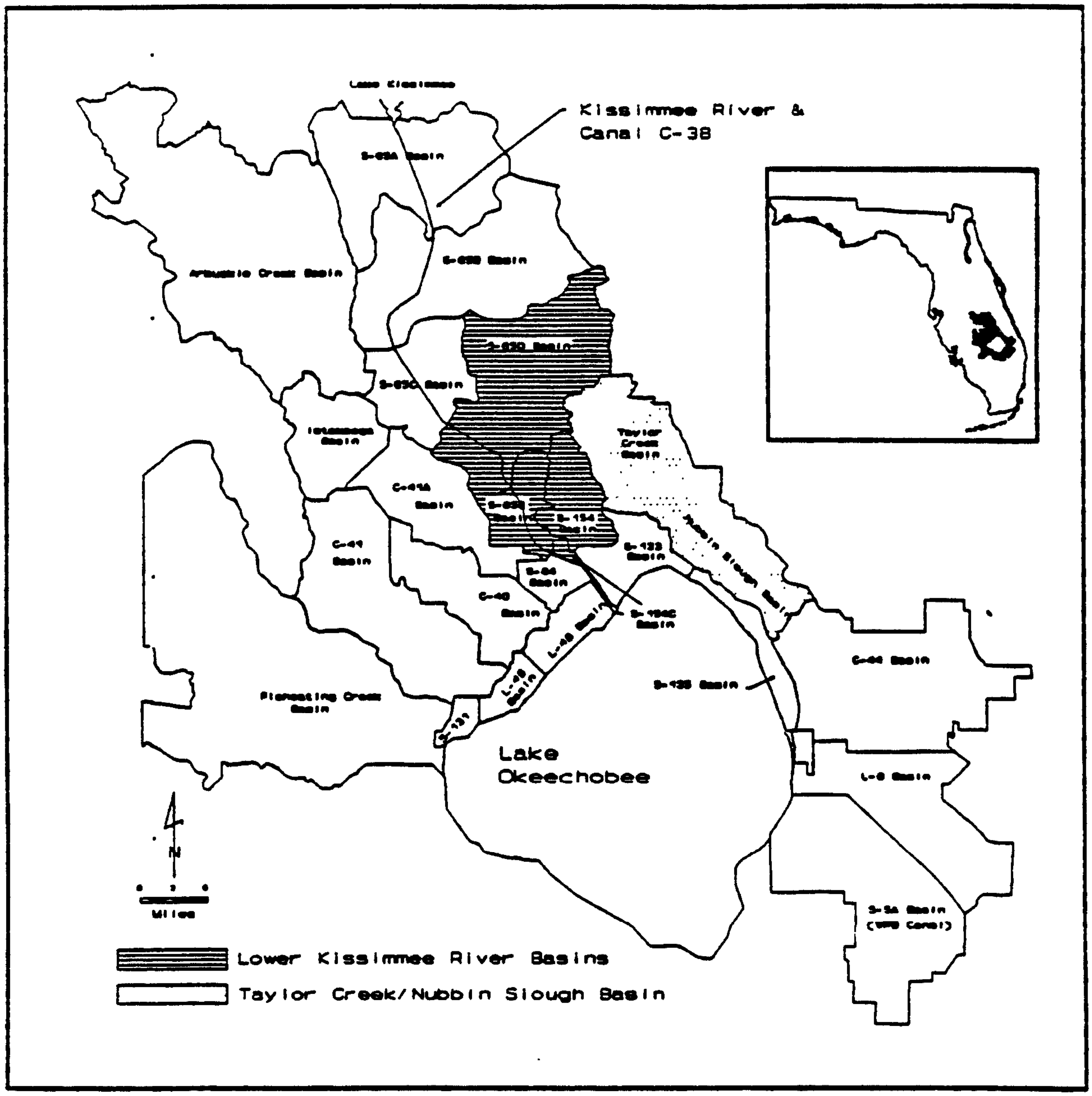

Figure 1. Location of Lake Okeechobee and the major drainage basins.

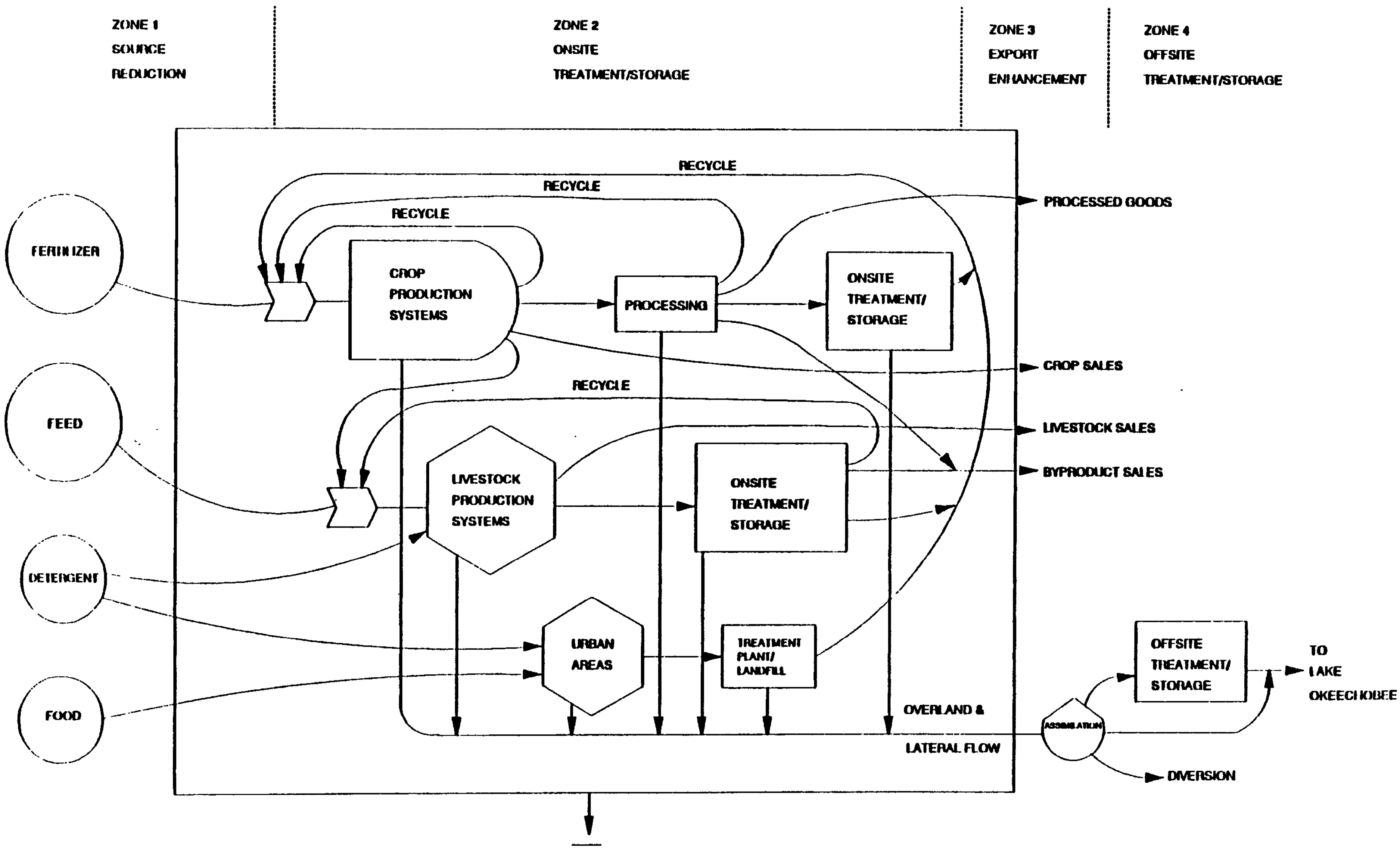

Figure 2. Phosphorus flow diagram for the Lake Okeechobee watershed.

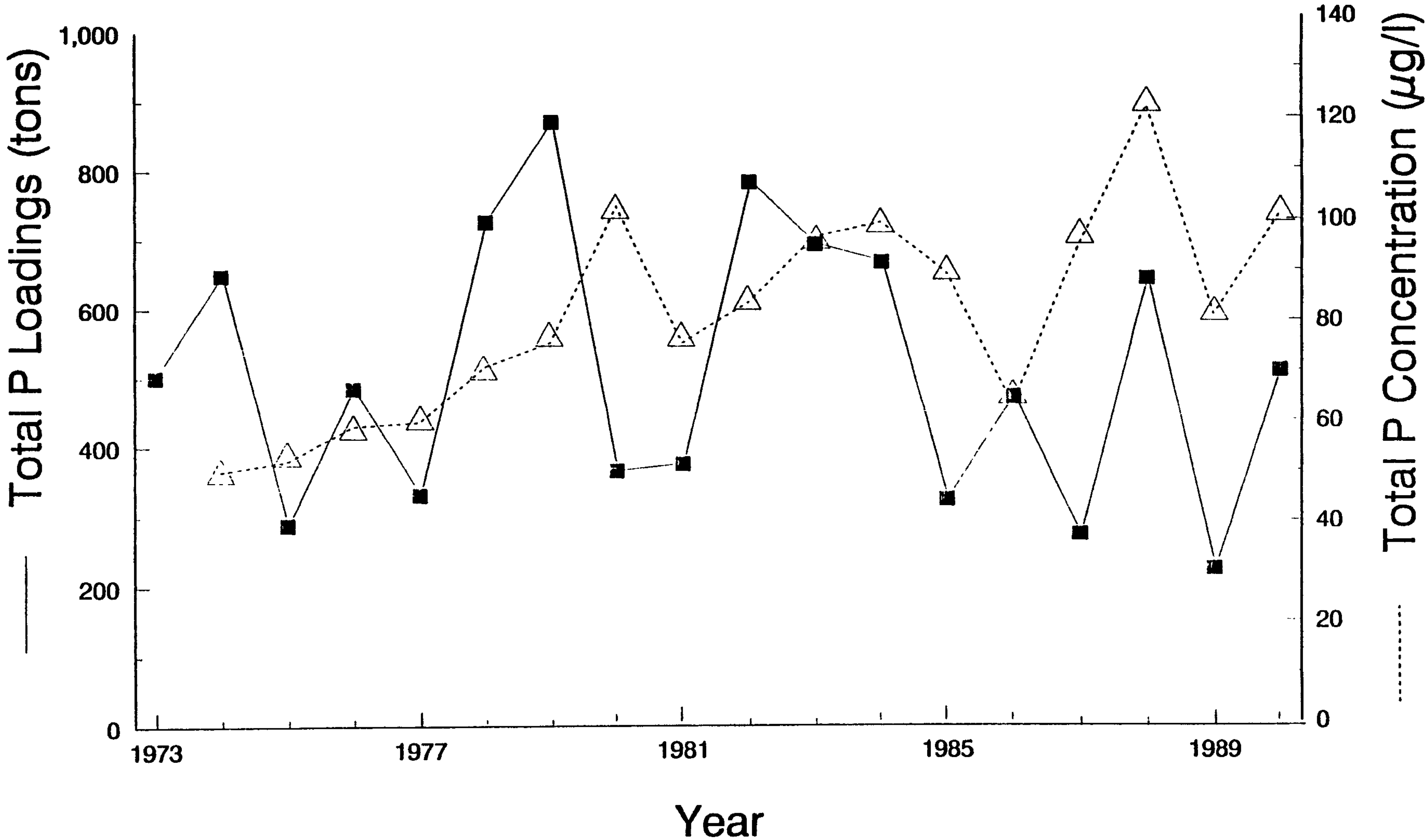

Figure 3. Total Phosphorus loads and in-lake TP concentrations for Lake Okeechobee (1973 - 1990).

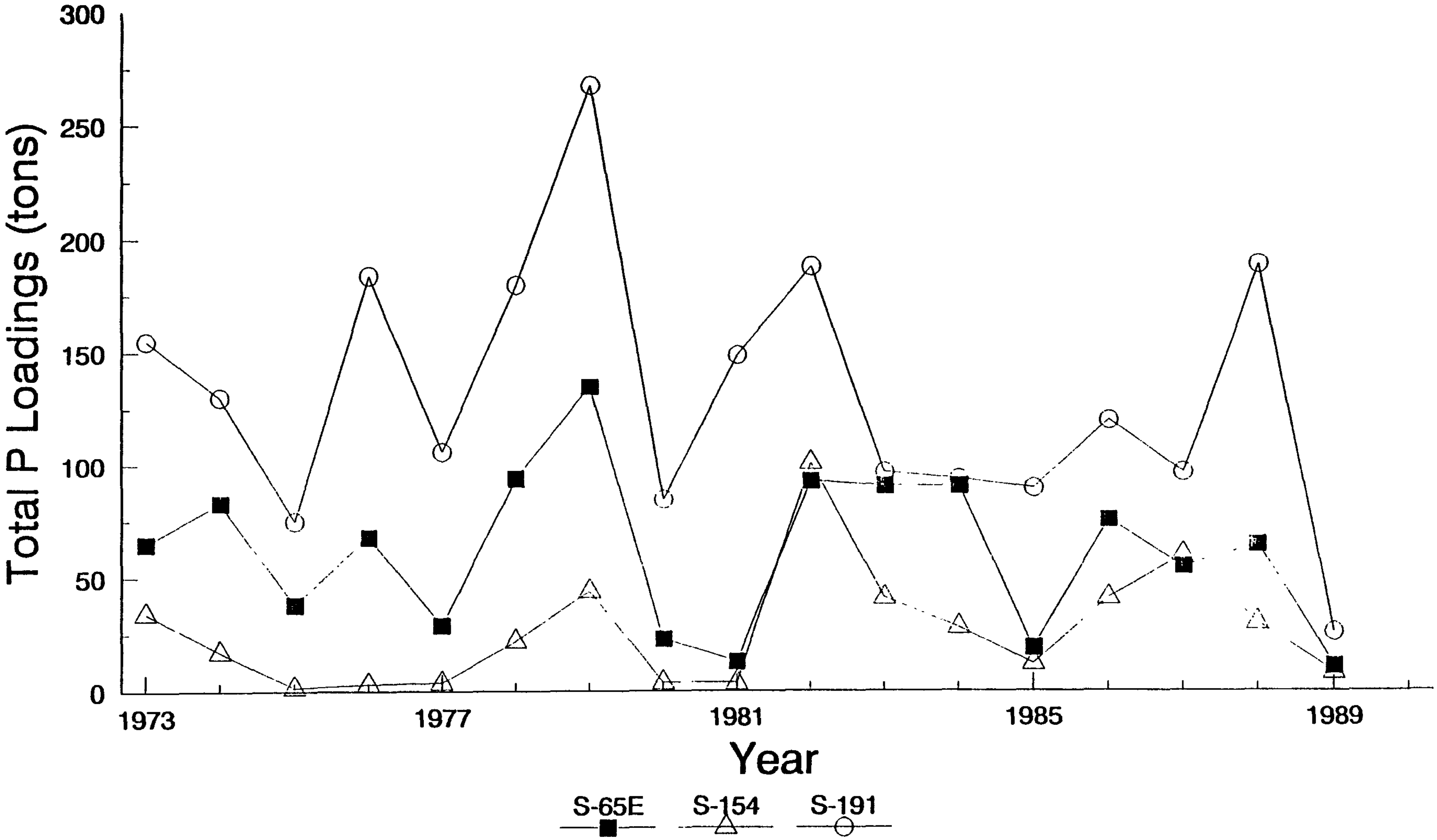

Figure 4. Total phosphorus loads to Lake Okeechobee from S-65E, S-154 and S-191 tributaries (1973 - 1989).

Analysis of Policy Options for the Control of Agricultural Pollution in California's San Joaquin River Basin

by

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ANALYSIS OF POLICY OPTIONS FOR THE CONTROL OF AGRICULTURAL POLLUTION IN CALIFORNIA'S SAN JOAQUIN RIVER BASIN

by

Marca Weinberg
Catherine Kling
James Wilen

California's San Joaquin Valley contains one of the nation's richest areas of agricultural production. The Valley is situated in the southern part of the State, between California's Coastal Range to the west and the Sierra Nevada Range to the east. The San Joaquin River drains the area, flowing northward and emptying into the San Francisco Bay. Parent materials for the region's westside soils are deep layers of marine sediments and hence these soils contain a significant amount of soluable salts and trace elements, including selenium, molybdenum, boron, arsenic, chromium and others. On the east side of the valley, relatively coarse alluvial soils have been deposited from the uplifted Sierra Nevada range. These soils are relatively free of the salts and trace elements that characterize the west slope soils.

The San Joaquin Valley would be an area of considerably lower agricultural productivity were it not for the large irrigation infrastructure that supports it. Water is supplied both through deep water wells and surface supplies delivered through large aquaducts that transport water throughout California. A patchwork of irrigation districts exists that facilitates allocation to members under contracts with the United States Bureau of Reclamation. Most of these contracts are 40 year contracts that specify fixed quantities to be delivered under fixed prices. Water prices are less than prices that would recover full costs of the delivery system andmay be less than variable costs.

While the soils of the Valley are rich, poor natural drainage hampers production in some areas. This problem is made acute by the presence shallow clay layers or lenses that are impervious to water. These clay lenses are particularly a problem in the valley trough where high water tables concentrate saline and trace elements in the root zones. To mitigate the harmful effects of salinity, farmers need to leach the salts through the soil profile by applying water in excess of plant needs to flush the soils. In upslope areas, leaching generates laterally moving groundwater with high concentrations of toxic elements, which then flows into the water tables of lower lying areas. In downslope areas over perched water tables, farmers have installed subsurface drainage systems to control water depth. These drain systems collect toxic drain waters which have historically been disposed in canals that empty into the San Joaquin River.

In 1983, the discovery of toxic levels of selenium in waterfowl in Kesterson Reservoir focused public attention on the San Joaquin Valley and the role of irrigated agriculture as the source of elements such as selenium, molybdenum, boron, and salts. As a result of the problems experienced at Kesterson, the State Water Resources Control Board adopted water quality standards for selenium and other elements in the San Joaquin River. These policies generated considerable research devoted to bio-physical modeling of agronomic and hydrological relationships as well as investigations of technological and engineering solutions. Unfortunately,insufficient attention has been given to the question of how to motivate changes in farming practices necessary to reduce drainage pollution and meet the standards.

This paper reports some investigations of several policy options available to address the agricultural pollution problem in the San Joaquin Valley. The study area is an interesting laboratory for investigating both point and non-point source pollution generated from agriculture. Leaching by upslope farmers generates polluted drain waters which flow subsurface into the perched water tables of lower lying farmlands. These interactions between upslope and downslope farmers, as well as lateral interactions between farmers in the same strata can be considered non-point source externalities. Mitigating activities undertaken by installing drain tiles creates a second-stage point source problem since pollutants at sump outfalls are, in principle, measurable. Thus conventional instruments such as effluent taxes as well as input taxes, subsidies, and technological requirements are all candidate policies.

The region modeled is a 68,000 acre area of diverse irrigated agriculture operating within a hydrological system of considerable complexity. This area includes lands with varied soil, elevation, and water table characteristics, nested within 9 water districts, each with its own water supply allocations and pricing policies. In the next section, we briefly describe the model and its principle features and assumptions. The following section describes some of the modelling results and the final section summarizes and offers some concluding thoughts.

II. MODEL STRUCTURE

In order to simulate regional response to various policy options, we developed an integrated economic/hydrological model and calibrated it to conditions representative of the San Joaquin Valley. The economic model predicts farmer decision making regarding crop choice, applied water, and irrigation technology/water management practices. The drainage area is divided up into physically homogeneous cells, each of which is similar with respect to soil type, drainage conditions, depth to impervious layer, and elevation above sea level. These cells are in turn divided into subcells corresponding to water district jurisdictions which vary in the characteristics of water contracts held. The model can be run as an integrated system encompassing the larger drainage area or as smaller subsystems to compare results under different economic, hydrologic, or institutional configurations.

The agricultural system simulated contains a variety of crops and agricultural practices. About half of the irrigated acreage is planted to cotton each year. Other primary crops include processing tomatoes, sugarbeets, melons, and wheat. Alfalfa hay and rice are important crops in some districts and a variety of vegetables and other specialty crops are also grown in the area. Cropping patterns vary by water district and are influenced by relative market conditions, rotational practice, drainage and soil conditions, etc.

Irrigation efficiency and the volume of drainage water generated vary by crop and with irrigation technology and management. Irrigation of salt-sensitive shallow rooted crops such as vegetables, melons, and small grains tends to be less efficient and hence generates more drain water than irrigation of long-season relatively salt tolerant crops. Irrigation system performance is an important factor in drainage generation and is included explicitly in the model. Irrigation efficiency enters the crop production functions and an irrigation technology cost function describes costs as a function of system performance.

The model describes joint production of two outputs, the primary crop yield and collected drain water. Water applied in excess of crop needs enters a drainage production function. The optimization component of the model selects crop acreage allocations, applied water, and irrigation efficiency subject to the technological relationships defining production, drain water generation, and irrigation technology costs. Resource and acreage limitations constrain the choices and policy instruments enter either as parameters that modify prices and costs or as constraints.

A. Crop Production Functions

Crop production functions in this analysis are developed (following Letey and Dinar) by combining von Liebig (plateau) functions with plant growth model results that predict relative yields as a function of root zone salinity. The procedure is as follows. First it is assumed that under non-saline conditions, yield achieves a maximum value (Y_{max}) for all values of applied water greater than ET_{max}, the minimum plant water requirement necessary to achieve Y_{max}:

$$(1) \qquad Y_{ns} = S(AW - AW_t) \qquad AW_t < AW < ET_{max}$$

$$= Y_{max} \qquad AW \geq ET_{max}$$

Where:

Y_{ns} is yield under nonsaline conditions

S represents the slope of the nonsaline production function

AW is applied water (acre-feet/acre)

AW_t is the minimum water application sufficient to generate positive yields (acre-feet/acre).

Under saline conditions, it is necessary to determine the yield decrement (YD) associated with various levels of water applications and salinity (EC). Since empirical data do not exist over a range of salinity and water applications, we generated data utilizing the physical plant growth model in Letey and Dinar. The model has performed well in comparisons with experimental data and consists of the equations:

for $AW_t < AW < ET_{max}$:

$$(2) \qquad \frac{100(YD)^2}{B \cdot S(AW - AW_t)} + YD \cdot C' - \frac{EC_i \cdot S \cdot AW}{2}$$

$$- .1EC_i \cdot S \cdot AW \cdot \ln\left[\frac{YD}{AW \cdot S} + \left(1 - \frac{YD}{AW \cdot S}\right)e^{-5}\right] = 0$$

for AW $\geq ET_{max}$:

$$(3) \qquad C' + \left[\frac{100YD}{B \cdot Y_{max}}\right] - \left[1 - \frac{ET_{max}}{AW} + \frac{YD}{AW \cdot S}\right]^{-1}$$

$$\cdot \left[.5EC_i - .1EC_i \cdot ln\left\{1 - \left[\frac{ET_{max}}{AW} - \frac{YD}{AW \cdot S}\right](1 - e^{-5})\right\}\right] = 0$$

where terms are as defined above, EC_i is an electrical conductivity measure of water salinity and C' is the value of salinity above which yield decrements begin to occur.

Equations (2) and (3) are in implicit form and describe yield response that would result from steady applications of water with a constant salinity level EC over time. Given values for maximum yields (Y_{max}), maximum and minimum crop water requirements $(ET_{max}$ and $AW_t)$, non-saline production function slopes (S), and Maas-Hoffman yield/salinity slopes (B), these equations can be solved for the yield decrement for a range of applied water and water salinity values. Input values were obtained from Letey and Dinar for cotton, wheat, tomatoes, sugarbeets, and alfalfa. Applied water was scaled by seasonal pan evaporation (E_p) calculated for the study area. The data generated were then used to fit crop production functions quadratic in applied water and salinity:

$$(4) \qquad RY_{s,a,c} = \alpha_{0,c} + \alpha_{1,c}(AW_{s,a,c}/E_{p,c}) + \alpha_{2,c}(AW_{s,a,c}/E_{p,c})^2 + \alpha_{3,c} \cdot EC_i + \alpha_{4,c} \cdot EC_i^2 + \alpha_{5,c}(AW_{s,a,c}/E_{p,c}) \cdot EC_i$$

Where:

s = 1,..,14 $\equiv$ cell index
a = 1,..,4 $\equiv$ subarea index
c = {alfalfa hay, cotton, melon, sugarbeets, tomatoes, wheat} $\equiv$ crop index
$RY_{s,a,c} \equiv$ relative yield (percent of maximum yield)
$AW_{s,a,c} \equiv$ water applied to crop c, in area a (af)
$EC_i \equiv$ soil salinity measure
$E_{p,c} \equiv$ seasonal pan evaporation (af/acre)
$\alpha_{i,c} \equiv$ estimated production coefficients, i = 0,...,5.

Fitted values and t-statistics for the production coefficients are presented in Table 1. A production function for melons is derived from observed data. Salinity variables are not included in the production function for melons because data describing electrical conductivity of applied water are not available.

Table 1. Fitted Crop Production Function Coefficients[a]

Crop:	α_0	α_1	α_2	α_3	α_4	α_5	R^2
Alfalfa Hay	-0.03	1.47	-0.13	0.02	0.00	-0.17	.9990
	(-5.11)	(159.00)	(-14.65)	(1.71)	(-0.40)	(-21.66)	
Cotton	-0.62	5.95	-5.46	-0.02	-0.01	0.03	.9999
	(-186.13)	(457.01)	(-293.63)	(-2.48)	(-2.96)	(4.26)	
Melons[b]	--	1.40	-0.49	..	--	--	--
Sugarbeets	-0.29	1.94	-0.11	0.02	0.00	-0.11	.9996
	(-71.63)	(159.87)	(-8.66)	(3.52)	(0.81)	(-18.38)	
Tomatoes	-1.19	3.26	-0.52	0.16	0.01	-0.43	.9982
	(-76.39)	(73.44)	(-13.92)	(7.97)	(0.50)	(-23.15)	
Wheat	-0.26	1.94	0.02	0.05	-0.01	-0.11	.9996
	(25.47)	(51.98)	(0.42)	(5.55)	(-1.38)	(-10.63)	

[a] t values are presented in parentheses, but no statistical properties are claimed because the data were generated using a simulation model of crop yields.

[b] Melon function parameters were derived rather than estimated so t values and R^2 can not be determined. Salinity (EC_i) coefficients were not derived for melons due to a lack of data.

For simulation purposes, these crop production functions were modified to allow for irrigation inefficiencies by assuming that the water available to the plant is applied water scaled by an irrigation efficiency (IE) parameter. Since the main means of reducing subsurface runoff is to improve irrigation efficiency, we model IE as a choice variable. Increasing irrigation efficiency imposes costs, and these are modeled by estimating an irrigation cost function described next, using available engineering and technical data.

B. Irrigation Cost Function

As discussed above, the key to reducing subsurface drain water is to improve irrigation efficiency and infiltration uniformity by adopting more efficient irrigation technologies or improving irrigation management. Infiltration uniformity is a function of irrigation technology, management, and the variation of soils throughout a field.

About 80% of the agricultural lands in the drainage problem area are currently irrigated with furrow or border strip systems that are operated at relatively low irrigation efficiencies. Variations in soil characteristics, the length of furrows, water delivery rates, and cultural practices influence the degree of infiltration uniformity observed in surface irrigated fields. Irrigators can improve irrigation efficiency and infiltration uniformity by reducing furrow lengths, compacting the furrows, and establishing a uniform grade throughout the field. Pressurized irrigation systems including sprinkler, surge, and low energy precision application systems will achieve greater efficiency and uniformity when field conditions are suitable and the systems are managed properly.

Water conservation and drainage reduction can be achieved through changes in irrigation practices but these changes will increase production costs. We compiled data from Davids and Gohring for eleven irrigation technologies and three management levels. These data include annualized capital, maintenance, and labor costs for selected technologies. These data were used to fit quadratic irrigation technology cost functions:

$$(5) \qquad ITC_{s,a,c} = \beta_{o,c} + \beta_{1,c} \cdot IE_{s,a,c} + \beta_{2,c} \cdot IE_{s,a,c}^{2}$$

where:

$ITC_{s,a,c} \equiv$ annualized irrigation technology and application cost ($/Acre)
$\beta_{i,c} \equiv$ estimated irrigation cost coefficients, i = 0, 1, 2.

using a full frontier quadratic programming approach (Aigner and Chu) to estimate the parameters. Crop-specific cost functions were estimated for alfalfa hay, melons, and wheat. A single function was estimated for row crops including cotton, sugarbeets, and tomatoes because these crops are irrigated similarly. Cost function coefficients are displayed in Table 2.

Table 2: Irrigation Cost Function Coefficients

Crop:	β_o	β_1	β_2
Alfalfa Hay	56.66	-110.92	227.69
Row Crops	118.77	-413.57	514.10
Melons	74.56	-287.20	403.67
Wheat	14.62	-73.60	208.80

C. Drainage Function

The exact relationship between applied water and collected drain water is not well understood, and is likely to be field specific and depend on soil properties, water quality, crop water requirements and root structure, seasonal timing of water applications, and the drain system design and spacing. Irrigation system choice and performance are also important in drainage production. Only water applied in excess of plant needs (on any portion of a field) contributes to drainage, and irrigation system parameters influence water application decisions (Feineman, Letey, and Vaux). A mass balance approach is used in this study to approximate water movement through the root zone.

The volume of collected drain water that is expected to result from irrigation and cropping pattern decisions is determined as a function of water applications and irrigation efficiency on overlying fields, soil properties, and water table conditions. This formulation is adapted form a similar one in the Westside Agricultural Drainage Economics Model (San Joaquin Valley Drainage Program 1989):

$$(6) \qquad CDW_{s,a} = [\{[\Sigma_c (AW_{s,a,c} \cdot (1-RO_{s,a,c}-EL_{s,a,c}-IE_{s,a,c}) \cdot ACRES_{s,a,c}/L_{s,a}] \cdot [2(POROS_s - SPRET_s)]^{-1}\} + ELUNS_s - (ELGR_s - DRNDPTH_s)] \cdot DA_{s,a} \cdot KD_{s,a}$$

where:

$CDW_{s,a} \equiv$ collected drain water (acre-feet)
$RO_{s,a,c} \equiv$ surface runoff (% of AW)
$EL_{s,a,c} \equiv$ evaporation losses (% of AW)
$L_{s,a} \equiv$ total irrigable land in subarea a (acres)
$POROS_{s,a} \equiv$ soil porosity
$SPRETUN_{s,a} \equiv$ specific retention of the soil
$ELUNS_{s,a} \equiv$ elevation of (bottom of') unsaturated zone in soil profile (feet)
$ELGR_{s,a} \equiv$ elevation of ground surface (feet)
$DRNDPTH_{s,a} \equiv$ depth of drains (below ground surface elevation) (feet)
$KD_{s,a} \equiv$ drain efficiency (%)

Surface runoff and evaporation losses are calculated as seven percent of water applications. The first term in (6) is expected deep percolation per acre. This is divided by specific yield to convert the volume of expected deep percolation to an equivalent depth that is added to existing ground water table heights. The difference between average water table depth and drain depth is multiplied by area to calculate the volume of water that is available to enter a drainage system. This volume is scaled by drain efficiency to obtain an estimate of expected drain water volumes.

D. The Programming Model

The optimization problem is to choose crop land allocations ($ACRES_{s,a,c}$), irrigation efficiency ($IE_{s,a,c}$), water applications ($AW_{s,a,c}$), and water sales ($SW_{s,a}$) to maximize net returns to land and management (equation (7)) subject to the production, drainage, and irrigation cost functions (8) through (13). Upper or lower bounds on crop land allocations are imposed on some crops. Total water and land constraints reflect the limited availability of these resources.

Equations (8) through (13) define technological relationships for relative yield, actual yield, collected drain water, and irrigation technology in the complete description of the simulation model, presented below. The total use of land and water resources is constrained to the amounts of these resources available in each subarea (equations (15) and (16)). Upper bounds are placed on crop land allocated to sugarbeets and tomatoes to reflect the limited number of [Bcontracts available for these crops and the small number of processing facilities in the area (equation (17a)). Maximum levels are also specified for melon acreage. A lower bound on cropland allocated to wheat reflects the typical use of this crop in rotation with other crops in the area (equation (17b)).

Simulation Model

(7) Maximize $$NRLM_{s,a} = \Sigma_c([(P_c - HC_c)\cdot Y_{s,a,c} - PC_c]\cdot ACRES_{s,a,c} - [P_{w,s,a}\cdot AW_{s,a,c} - ITC_{s,a,c}]\cdot ACRES_{s,a,c}) - DC\cdot DA_{s,a} + PM_w\cdot SW_{s,a} - t_d CDW_{s,a}$$

subject to:

(8) $$RY_{s,a,c} = \alpha_{0,c} + \alpha_{1,c}(AW_{s,a,c}\cdot IE_{s,a,c}/E_{p,c}) + \alpha_{2,c}(AW_{s,a,c}\cdot IE_{s,a,c}/E_{p,c})^2 + \alpha_{3,c}\cdot EC_i + \alpha_{4,c}\cdot EC_i^2 + \alpha_{5,c}(AW_{s,a,c}\cdot IE_{s,a,c}/E_{p,c})\cdot EC_i$$

(9) $$RY_{s,a,c} \leq 1.0$$

(10) $$Y_{s,a,c} = YMAX_{s,a,c}\cdot RY_{s,a,c}$$

(11) $$CDW_{s,a} = [\{[\Sigma_c(AW_{s,a,c}\cdot(1-RO_{s,a,c}-EL_{s,a,c}-IE_{s,a,c}) \cdot ACRES_{s,a,c}/L_{s,a}]\cdot[2(POROS_s - SPRET_s)]^{-1}\} + ELUNS_s - (ELGR_s - DRNDPTH_s)]\cdot DA_{s,a}\cdot KD_{s,a}$$

(12) $$ITC_{s,a,c} = \beta_{o,c} + \beta_{1,c}\cdot IE_{s,a,c} + \beta_{2,c}\cdot IE_{s,a,c}{}^2$$

(13) $$\Sigma_c\, ACRES_{s,a,c} \leq L_{s,a}$$

(14) $$(\Sigma_c\, AW_{s,a,c}\cdot ACRES_{s,a,c}) + SW_{s,a} \leq W_{s,a}$$

(15) a) $ACRES_{s,a,c} \leq A_{s,a,c}\cdot L_{s,a}$ (c= sugarbeets, tomatoes, melons)
b) $ACRES_{s,a,c} \geq A_{s,a,c}\cdot L_{s,a}$ (c = wheat)

(16) $$CDW_{s,a} \leq DRNLIM_{s,a}$$

(17) $$\Sigma_{s,a}\, CDW_{s,a} \leq DRNLIM$$

This specification pertains to any given cell (s) and subarea (a) combination. All variables in the model are described in this section. A complete description of notation is provided in Table 3.

Table 3: Alphabetical Guide to Simulation Model Notation

$s = 1,..,12 \equiv$ cell index

$a = 1,..,4 \equiv$ subarea index

c ={alfalfa, cotton, melons, sugarbeets, tomatoes, wheat} $\equiv$ crop index

$ACRES_{s,a,c} \equiv$ acres of crop c planted in area a (acres)

$AW_{s,a,c} \equiv$ water applied to crop c, in area a (af)

$CDW_{s,a} \equiv$ collected drain water (af)

$DA_{s,a} \equiv$ drained acres (acres)

$DC \equiv$ drain system costs ($/acre)

$DRNDPTH_{s,a} \equiv$ depth of drains (below ground surface elevation) (feet)

$DRNLIM \equiv$ maximum volume of drain water allowed (af/acre)

$E_{p,c} \equiv$ seasonal pan evaporation (af/acre)

ECi $\equiv$ soil salinity measure

$EL_{s,a,c} \equiv$ evaporation losses (% of AW)

$ELUNS_{s,a} \equiv$ elevation of (bottom of) unsaturated zone in soil profile (feet)

$ELGR_{s,a} \equiv$ elevation of ground surface (feet)

$HC_c \equiv$ harvest costs ($/ton)

$IE_{s,a,c} \equiv$ irrigation application efficiency

$ITC_{s,a,c} \equiv$ annualized irrigation technology and application cost ($/acre)

$KD_{s,a} \equiv$ drain efficiency (%)

$L_{s,a} \equiv$ total irrigable land in subarea a (acres)

P~&price of water in subarea a ($/af)

$P_c \equiv$ crop output price ($/ton)

$PC_c \equiv$ preharvest costs ($/acre)

$PM_w \equiv$ market price of water ($/af)

$POROS_{s,a} \equiv$ soil porosity

$RO_{s,a,c} \equiv$ surface runoff (% of AW)

$RY_{s,a,c} \equiv$ relative yield (percent of maximum yield)

$SPRETUN_{s,a} \equiv$ specific retention of the soil

$SW_{s,a} \equiv$ volume of water sold in water market (af)

$t_d \equiv$ drain water tax ($/af)

$W_a \equiv$ total volume of water available in area a (af)

$Y_{s,a,c} \equiv$ yield of crop c attained in area a (tons/acre)

$YMAX_{s,a,c} \equiv$ maximum yield attainable (tons/acre)

$\alpha_{i,c} \equiv$ estimated production coefficients, $i = 0,...,5$

$\beta_{i,c} \equiv$ estimated irrigation cost coefficients, $i = 0, 1, 2$

Policy parameters are included in the objective function of the model to represent incentive-based policy tools. Taxes and subsidies on selected inputs and outputs are examined as policy tools to *motivate* improvements in irrigation practices and drainage reduction strategies. Resource constraints are imposed in the model to examine policies that *require* changes in irrigation and drainage practices. Examples include water supply restrictions, drainage discharge standards, irrigation technology requirements, and restrictions on crop land allocations.

The volume of drain water generated in a given cell and subarea can be constrained with equation (16) while a regional drainage constraint is modeled by limiting the volume generated by all cells and subareas (equation (17)). A district-level constraint on collected drain water can be imposed by selecting the pertinent cells and subareas.

The drainage discharge constraints are set to nonbinding levels and all of the incentive-based policy parameters, including the water market price, are set equal to zero for the base case analysis. The policy parameters and resource constraints are allowed to vary when examining policy alternatives in the following section.

III. SIMULATION RESULTS

This section examines the environmental and economic implications of policy selection. The analysis provides an aggregated summary of policy response for the drainage area as a whole. Comparisons of relative costs and benefits of several policies are made by holding the level of drainage constant across all policies. Drainage reduction goals of 10%, 20% and 30% are specified to compare policies. Policies examined include crop-specific water taxes, uniform water taxes, effluent and irrigation efficiency standards, and combinations of uniform water taxes and irrigation efficiency subsidies, in addition to effluent taxes and water markets.

A. Base Case Analysis

The area encompassed in this analysis approximates the drainage study area as defined by the State Water Resources Control Board (California, 1987). The model cells included represent 68,000 irrigable acres, 44,000 of which are drained. Cells that overlap the drainage study area designation but do not contain drained acres are not included. A total of sixteen subareas are modeled, and nine water districts are represented in this analysis. Results are averaged for all subareas and thus represent the regional average value predicted for most variables. Cropping patterns are presented as the percent of total irrigable acres in the region that are predicted to be devoted to each crop. The results of the base case analysis are presented in Table 4.

In the absence of any policy intervention the model predicts that 66 % of the acreage will be planted in cotton, 9% in tomatoes, 7% each in sugarbeets and wheat and 6% in melons. It is predicted that 5% of irrigable acreage will be left fallow. These results are reasonable approximations of historic practices in the region with the exception of the large amount of cotton acreage predicted by the model. Averages of actual values for crop acreage allocations, water applications and yields reported in the region are presented in Table 5. The large cotton acreage allocations predicted arise as a consequence of the omission of many crops typically planted in relatively small acreages. These crops combine to makeup approximately 14% of the cropped acreage in the region, acres that are devoted by the model to cotton instead.

Table 4. Results of Base Case Simulation

	Cotton	Melons	Sugar-beets	Tom.	Wheat
Acres (% total)	66%	6%	7%	9%	7%
Applied Water (feet)	3.33	1.90	4.67	3.27	2.38
Irrigation Efficiency (%)	73%	67%	74%	78%	68%
Irrig. tech & mngmt costs ($/acre)	91.94	62.84	97.27	108.28	62.73
Yield (tons/acre)	0.62	8.83	29.92	32.89	3.02
Marginal DW Product (aw) (cdw/ac)	0.15	0.19	0.15	0.11	0.20
Marginal DW Product (ie) (cdw/ac)	-2.45	-1.36	-3.62	-2.31	-1.79

Fallow Acres	5%
Collected drain water (af/acre)	0.45
Collected drain water (af/drained acre)	0.69
Net Revenues ($/acre)	339.40

Table 5. Average reported values for cropping patterns, water applications, and yields

	Acres*	Applied Water"*	Yields"
	(% irrigable)	(af/a)	(tons/a)
Cotton	48%	3.24	0.67
Melons	8%	2.07	8.90
Sugarbeets	6%	4.57	29.70
Tomatoes	8%	3.22	31.74
Wheat	9%	2.30	2.79
Model Crops	86%	na	na
Fallow	7%	na	na

Notes: Weighted average (1984-1988) from primary DSA districts.. Source: BOR crop reports
** Average (1986-1988) for Broadview WD only. Source: Wicheins (1989)

Predicted irrigation efficiency levels range from 67% on melons to 78% on tomatoes, with levels of 68%, 73% and 74% for wheat, cotton and sugarbeets, respectively. The environmental consequences of these agricultural production activities derive from .69 acre feet of collected drain water produced per drained acre. This figure matches closely with the estimate of.7 acre feet per acre in the tile drained areas of the drainage study area arrived at by the San Joaquin River Basin Technical Committee (California, 1987). Net returns to land and management from crop production are estimated to be $339 per acre.

B. Comparison of Policy Options

The importance of characterizing the regional implications of policy alternatives for achieving drainage reduction goals was noted at the outset of this section. This issue is addressed in this section by examining the cost of achieving a specified objective with a first-best policy as well as through comparison with results associated with policies that are less efficient. Drainage reduction objectives often, twenty and thirty percent are considered.

Ten Percent Drainage Reduction

Results indicate that a ten percent reduction in collected drain water could be achieved with small adjustments in agricultural production activities and with minimal consequences for the region. A drain tax of $100/af would motivate the necessary changes and assure that the drainage reduction objective is met at least cost, *given existing water supply institutions*. A drain water discharge standard imposed regionally and allocated among cells and subareas in an efficient manner would accomplish the same objective, as would an appropriately specified set of input taxes. The efficiency implications of these policies, as reflected in average crop returns, are identical, though the fiscal implications are not. Simulation results for the drain tax scenarios are presented in Table 6. Base case results are included in the table to facilitate comparisons. Results indicate a cost of less than $3/acre for meeting the drain water objective. A drain tax would cost farmers an additional $40/acre in tax payments, on average. The fiscal costs of the instrument swamp the costs of meeting the environmental objective, in this case.

The ten percent drainage reduction objective could also be achieved as a result of a water market in which water is sold at a price of $72.50/af. Predicted responses to this policy are included in the third column of Table 7. Crop returns under this instrument are essentially the same as those achieved as a result of a first-best policy choice; the efficiency cost of a water market is only $.07/acre. Net returns increase with the positive revenues received from water sales and are $22/acre higher than base levels.

Twenty Percent Drainage Reduction

A twenty percent reduction in collected drain water in the region involves more significant changes than the ten percent reduction, though the least cost solution to this problem involves a reduction in crop returns of only $6.50/acre. The first-best policies considered that yield this result include a drain tax of $132 per acre foot of collected drain water and an efficiently allocated drainage reduction standard. Simulation results are presented in Table 7.

Three policies that do not motivate a least cost solution are considered a water market in which water maybe sold between districts as well as outside the region, a uniform water tax and a crop-specific water tax. The latter is a policy that would fall between a first-best set of input taxes, i.e. a set that includes crop, cell and subarea-specific

Table 6. Predicted response to policies for a 10% reduction in regional drainage

		Base Case	Drain Tax	Water Market
Cotton				
	Acres (% total)	66%	69%	67%
	Applied Water (feet)	3.33	3.23	3.18
	Irrigation Efficiency (%)	73%	77%	75%
	Irrig. tech & mngmt costs ($/acre)	91.94	107.96	98.18
	Yield (tons/acre)	0.62	0.62	0.64
Melons				
	Acres (% total)	6%	4%	6%
	Applied Water (feet)	1.90	1.83	1.87
	Irrigation Efficiency (%)	67%	70%	68%
	Irrig. tech & mngmt costs ($/acre)	62.84	72.59	64.58
	Yield (tons/acre)	8.83	9.01	8.85
Sugarbeets				
	Acres (% total)	3%	7%	3%
	Applied Water (feet)	4.24	4.60	4.07
	Irrigation Efficiency (%0)	80%	75%	84%
	Irrig. tech & mngmt costs ($/acre)	118.76	103.84	133.15
	Yield (tons/acre)	30.82	29.92	30.91
Tomatoes				
	Acres (% total)	9%	9%	9%
	Applied Water (feet)	3.27	3.10	3.25
	Irrigation Efficiency (%)	78%	82%	77%
	Irrig. tech & mngmt costs ($/acre)	108.28	128.14	106.61
	Yield (tons/acre)	32.69	32.89	32.70
Wheat				
	Acres (% total)	7%	7%	7%
	Applied Water (feet)	2.38	2.29	2.14
	Irrigation Efficiency (%)	68%	72%	74%
	Irrig. tech & mngmt costs ($/acre)	62.73	72.75	75.03
	Yield (tons/acre)	3.02	3.02	3.03
Fallow Acres	(% total)	5%	3%	8%
Collected drain water	(af/acre)	0.45	0.41	0.41
Collected drain water	(af/drained acre)	0.89	0.62	0.62
Water Sales	(af/acre)	na	na	0.31
Crop Returns	($/acre)	339.40	336.92	336.85
Net Returns	($/acre)	339.40	297.30	359.02

Table 7. Predicted response to policies for a 20% reduction in regional drainage

		Drain Tax	Water Market	Crop-specific Water Tax	Uniform Water Tax
Policy Instrument:		Pd 131.96	Pm 88.00	Tw 92(1:2:.5)	Tw 87.45
Cotton					
Acres	(% total)	70%	61%	56%	60%
Applied Water	(feet)	3.18	3.01	2.96	2.99
Irrigation Efficiency	(%)	78%	78%	79%	78%
Irrig. tech & mngmt costs	($/acre)	112.98	108.48	111.67	109.66
Yield	(tons/acre)	0.63	0.64	0.64	0.64
Melons					
Acres	(% total)	5%	6%	4%	8%
Applied Water	(feet)	1.83	1.78	1.40	1.79
Irrigation Efficiency	(%)	70%	70%	80%	69%
Irrig. tech & mngmt costs	($/acre)	74.02	70.73	103.64	69.44
Yield	(tons/acre)	9.03	8.82	8.70	8.65
Sugarbeets					
Acres	(% total)	7%	2%	8%	2%
Applied Water	(feet)	4.40	3.91	4.63	3.87
Irrigation Efficiency	(%)	78%	87%	74%	88%
Irrig. tech & mngmt costs	($/acre)	113.12	148.79	93.47	153.65
Yield	(tons/acre)	30.10	32.46	29.88	32.80
Tomatoes					
Acres	(% total)	8%	9%	9%	8%
Applied Water	(feet)	3.06	3.13	3.10	3.11
Irrigation Efficiency	(%)	83%	80%	81%	81%
Irrig. tech & mngmt costs	($/acre)	132.03	118.84	122.08	120.95
Yield	(tons/acre)	32.71	32.70	32.69	32.69
Wheat					
Acres	(% total)	7%	7%	7%	7%
Applied Water	(feet)	2.18	2.03	2.00	2.03
Irrigation Efficiency	(%)	74%	78%	79%	78%
Irrig. tech & mngmt costs	($/acre)	77.75	84.59	86.79	85.09
Yield	(tons/acre)	3.02	3.04	3.03	3.05
Fallow Acres	(% total)	3%	15%	16%	16%
Collected Drain Water	(af/drained acre)	0.55	0.55	0.55	0.55
Water Sales	(af/ac)	na	0.65	na	na
Crop Returns	($/acre)	332.64	308.23	307.42	302.84
Net Returns	($/acre)	284.87	365.73	89.36	93.81

Note: Values for crop-specific water tax are: $92/af for cotton, tomatoes and wheat, $184/af for melons, and $46/af for sugarbeets taxes for each input, and a uniform input tax in terms of marginal information costs and efficiency benefits. A water market price of $88/af is found to result in the desired reduction in collected drain water. A value of $87/af accomplishes the same objective with a uniform water tax.

The difference in the values of the water market and water tax parameters is somewhat surprising. These instruments provide the same incentive regarding water conservation and therefore one would expect that the same value would be required to achieve the drainage reduction objective. The difference is that the initial allocative inefficiency of existing water supply institutions is corrected in the case of the water market but not with the water tax. More drainage is created in some areas as a result of the initial reallocation of water resources, so a slightly higher level for the instrument is necessary to motivate the 20% drainage reduction with a water market than with a uniform water tax.

In contrast to the drain tax, the water market and a uniform water tax motivate significant changes in cropping patterns. The reason for this is clear; water markets create a general incentive to reduce water use while drain taxes act as an incentive to conserve only that quantity of water applied in excess of crop needs. Thus, a crop such as melons that has a relatively high marginal value product of water is favored under a water market despite the fact that it tends to be irrigated less efficiently with a relatively high marginal drain water product. Sugarbeets, a high water using crop, is phased out under a market, but not in response to a drain tax.

A crop-specific water tax may also be specified to account for variation in drainage production that arises when water is applied to different crops. The tax examined here varies in proportion to the marginal drain water product of water for each crop, evaluated at optimal levels. The marginal drain water product for water used on melons is predicted to be twice that of water used on cotton. This value is approximately the same for tomatoes and wheat as for cotton, but is roughly two times as great as that for water used to produce sugarbeets. The tax examined is thus specified as $92/af for water used to produce cotton, tomatoes or wheat, $184/af for water used on melons and $46/af for water used on sugarbeets. Table 7 includes results from these scenarios.

As predicted, water market and uniform water taxes create an incentive to increase melon acreage and reduce acres allocated to sugarbeets, relative to the drain tax scenario. One advantage of the crop-specific water taxis that it reduces distortionary crop allocation incentives inherent in the uniform tax. Melons are predicted to occupy 5% of total acres under a drain tax, 4% with crop-specific water taxes, 6% under a water market and 870 with a uniform water tax. In contrast, sugarbeets represent 7% and 8% of the acreage in response to a drain tax and crop-specific water taxes, respectively, but only 2% of the acreage under a water market or uniform water tax. All three of the less efficient policies create incentives to reduce cotton acreage, which declines by 13% to 20%, and to increase the quantity of fallow land.

Predicted irrigation efficiencies are generally constant across policies. There are two exceptions: melons are irrigated at higher efficiencies with the crop-specific water tax than with other policy instruments; and, the water market and uniform water taxes result in irrigation efficiencies on sugarbeets that are 14% to 15% higher than the level implied by the optimal solution.

Crop returns (efficiency benefits) and fiscal implications associated with the policies considered are summarized in Figure 1. The efficiency costs of the water market and crop-specific water tax policies are $25 and $26 per acre, respectively. A uniform water tax is the least efficient policy considered with an efficiency cost of $30/acre.

It is interesting to note that the water market, which in effect represents a uniform water charge, results in crop returns that are slightly higher than under a water tax that incorporates variation in drain water production of crop-specific water use. The explanation for this lies in the inefficiencies created by current water supply institutions which are eliminated through the inter-district water market but which remain in place with a uniform water tax. These results provide empirical support for conclusions regarding the second-best implications of the institutional setting in the drainage problem area discussed in a companion paper.

Thirty Percent Drainage Reduction

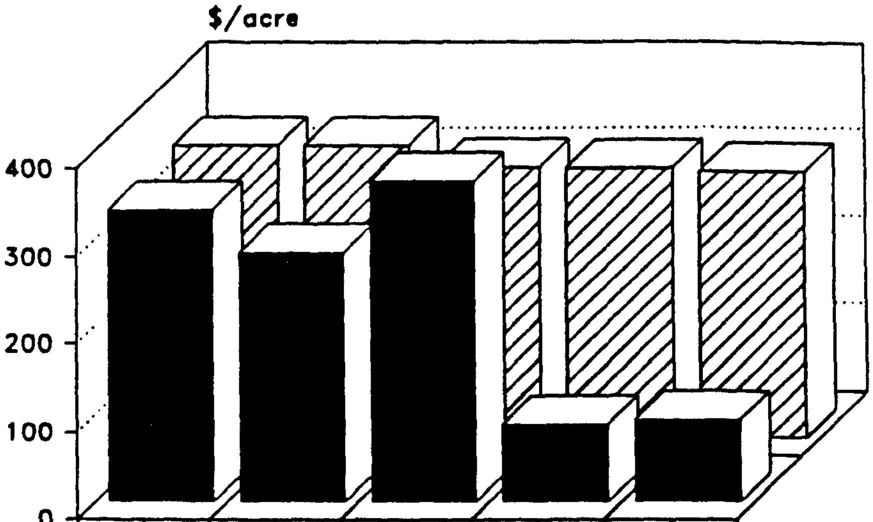
The thirty percent drainage reduction objective is significant in that it is the value suggested to be sufficient to achieve San Joaquin River water quality standards, as previously discussed. Simulation results for policies designed to achieve this objective are presented in Table 8. Results indicate a minimum cost of meeting the thirty percent drainage reduction objective of $14/acre. A drain tax of $190 per acre foot of collected drain water generates this result.

The San Joaquin River Basin Technical Committee that proposed the thirty percent reduction objective suggested that this objective could be achieved by increasing irrigation efficiencies in the study area to 80% (California, 1987). A policy of mandating irrigation efficiency levels was therefore included in this analysis. An irrigation efficiency standard of 83% was found to generate a thirty percent reduction in drain water volumes. Results from this analysis are included in the second column of Table 8. The efficiency cost of this policy is $8/acre.

Policies that combine uniform water taxes with subsidies for improving irrigation efficiency were also considered. There are many combinations of values for these instruments that will yield the desired drainage reduction, though none achieves the objective at least cost. Two combinations are shown here: a $45/af water tax combined with a 45% irrigation system cost subsidy and a $75/af water tax and 25% subsidy combination. Predicted responses to these policies are presented in the last two columns of Table 8.

The irrigation efficiency standard generates results that are very similar to the optimal solution in all respects except for those results describing irrigation management on melons. This is an important exception, however. Melons represent a production process with high marginal abatement costs in this analysis. Difficulties associated with improving irrigation efficiencies on melons are reflected in irrigation system costs for this crop that are $37/acre higher under the 83% efficiency standard than is optimal. According to economic theory, it is not optimal to require identical abatement levels from sources with different costs. Rather, optimality requires relatively more abatement from sources lower costs and less from the higher cost processes so that Bmarginal abatement costs are equated across sources. This principle is reflected in the results of the drain tax scenario in which melons are irrigated less efficiently than other crops. The inefficiencies created by standards on irrigation efficiency levels arise because there is no flexibility in the instrument to account for these factors.

Figure 1. Crop returns and fiscal effects of alternative policies for achieving a 20% reduction in drain water



	Standard	Tax	Water Market	Water Tax by Crop	Uniform Water Tax
Crop Returns	333	333	308	307	303
Fiscal Cost	0	-48	58	-218	-209
Net Returns	333	285	366	89	94

Table 8. Predicted response to policies for a 30% reduction in regional drainage

		Drain Tax	Irrigation Efficiency Standard	Water tax/ Irrigation Efficiency Subsidy	
Policy Instrument:		Pd: $191.38/af cdw	IE: 83%	Tw: $45/af	$75/af
				S: 45%	25%
Cotton					
Acres	(% total)	69%	69%	72%	67%
Applied Water	(feet)	3.09	2.93	2.94	2.90
Irrigation Efficiency	(%)	81%	83%	84%	82%
Irrig. tech & mngmt costs	($/acre)	123.60	129.67	134.37	126.81
Yield	(tons/acre)	0.63	0.62	0.62	0.63
Melons					
Acres	(% total)	5%	4%	5%	6%
Applied Water	(feet)	1.83	1.72	1.68	1.71
Irrigation Efficiency	(%)	71%	83%	77%	74%
Irrig. tech & mngmt costs	($/acre)	76.93	114.27	94.50	82.98
Yield	(tons/acre)	9.04	9.13	9.05	8.84
Sugarbeets					
Acres	(% total)	7%	8%	7%	3%
Applied Water	(feet)	4.19	4.06	3.98	3.71
Irrigation Efficiency	(%)	82%	84%	86%	92%
Irrig. tech & mngmt costs	($/acre)	126.43	133.49	144.09	172.22
Yield	(tons/acre)	29.92	29.90	30.08	30.82
Tomatoes					
Acres	(% total)	9%	9%	9%	9%
Applied Water	(feet)	2.99	3.03	2.92	2.95
Irrigation Efficiency	(%)	85%	83%	86%	85%
Irrig. tech & mngmt costs	($/acre)	139.12	129.67	145.80	140.07
Yield	(tons/acre)	32.70	32.69	32.70	32.69
Wheat					
Acres	(% total)	7%	7%	7%	7%
Applied Water	(feet)	2.03	1.91	1.97	1.92
Irrigation Efficiency	(%)	79%	83%	81%	83%
Irrig. tech & mngmt costs	($/acre)	88.29	97.37	93.27	96.30
Yield	(tons/acre)	3.02	3.02	3.02	3.03
Fallow Acres	(% total)	3%	2%	0.43%	8%
Collected Drain Water	(af/dr. acre)	0.48	0.48	0.48	0.48
Crop Returns	($/acre)	325.65	317.59	320.15	307.97
Net Returns	($/acre)	265.37	317.59	249.28	145.28

Melons make up a small portion of the crop mix regionally so the average efficiency cost of a policy of mandating irrigation efficiency levels is not large. Melons are not produced uniformly throughout the region however, so that distributional consequences may be significant with this policy. Farmers that devote relatively large portions of their operations to the production of melons and other shallow rooted salt-sensitive crops will bear a disproportionate sham of the cost of meeting regional drainage reduction goals.

Significant differences are apparent in the results of the tax and subsidy scenarios. In general, the 45% subsidy and $45/af tax combination favors production of all crops relative to the scenario with a lower subsidy and higher tax. Fallow land makes up 8% of total acreage in the latter case and less than 1% in the former. Cotton acreage is reduced by 7% and sugarbeets by 619% with 25% subsidies and a $75/af tax relative to the case that subsidies are 45% and the tax is $45/af, though melon acreage is 39% higher.

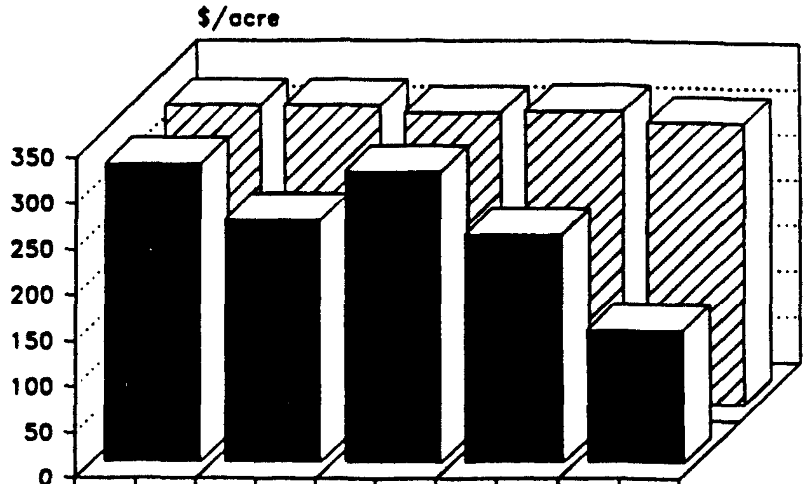
The tax and subsidy instruments result in irrigation efficiencies that are higher than optimal for all crops. The instruments effect crop-specific irrigation efficiencies differently, however. The combination with the higher subsidy rate results in higher efficiency levels, and lower water applications on cotton and melons, while the combination of a lower subsidy and higher water tax creates an incentive to increase efficiencies on sugarbeets beyond those implied by the $45/af tax and 45% subsidy combination.

Neither tax/subsidy combination is expected to be efficient because uniform rates are specified for each component of the instrument. The efficiency cost of this instrument varies with the exact combination considered. The instrument with greater emphasis on an irrigation subsidy is found to be more efficient than the instrument with a heavier weight on the water tax. Crop returns and returns net of the fiscal impacts of the policy instruments are illustrated in Figure 2. Crop returns are highest with the discharge standard and drain tax, as expected. The instrument with 45% subsidies and a $45/af water tax results in crop returns that are $5.50/acre lower than implied by the optimal solution. Efficiency costs associated with a policy of mandating irrigation efficiency levels are $8/acre, and are $18/acre for the 25% subsidy and $75/af tax combination.

Effluent (drain water) and input (irrigation efficiency) standards have no additional costs imposed at the farm-level and as a result are the instruments with the highest net returns. The net cost of the incentive instruments range from $60/acre with the drain tax to $163/acre for a policy of subsidizing 25% of irrigation efficiency costs but charging $75/af for applied water.

Results of this analysis consistently indicate low costs for meeting drainage reduction goals. These costs range from $3 per acre for achieving a ten percent reduction in drainage to $14 per acre for thirty percent reductions. One of the reasons that these costs are low is that they are averaged over the entire drainage study area, although drain systems have been installed in only two thirds of the area modeled. The average cost of meeting drainage reduction goals would increase somewhat if these costs were borne solely by farms in drained areas. The difference would not significantly effect the general results, however.

Figure 2. Crop returns and fiscal effects of alternative policies for achieving a 30% reduction In drain water



	Standard	Tax	IE Standard	Taxes (-45,.45)	Taxes (-75,.25)
Crop Returns	326	326	318	320	308
Fiscal Cost	0	-61	0	-71	-163
Net Returns	326	265	318	249	145

IV. Summary and Conclusions

This paper summarizes an analysis of incentive- and control-based policies for regulating agricultural pollution in California's San Joaquin Valley. The problem arises in the irrigated element-rich soils of the west slopes of the Valley. As these soils are irrigated, salts and naturally occuring trace elements are leached out and travel laterally through substrata until they empty into canals, the San Joaquin River, or in other low lying collecting basins. Salts and other elements can concentrate and bioaccumulate and cause deformities in wildlife and waterfowl. As a result of the discovery of deformities in waterfowl, California's Water Resources Control Board established water quality standards in the San Joaquin River. This study analyzes the impacts of various means of meeting these standards.

The model utilized here is a combined economic/hydrological model designed to simulate farmer decision making under various regulatory scenarios. The principle behavioral choices are assumed to be cropping patterns, applied water, and irrigation/water management technology. These are modeled under a diversity of conditioning factors calibrated to various subregions in the drainage area including: soil characteristics, weather, depth to water table, soil salinity, district water allocations and prices, plant yield characteristics, etc. A range of policy options is considered, including: effluent taxes, irrigation efficiency standards, water markets, and input tax/subsidy schemes.

Ordinarily, empirical analysis of non-point source pollution is difficult because there are multiple input and output points which are (by definition) impossible or difficult to measure. In the case examined here there are two fortunate differences. First, the hydro-physical system has been intensively modeled and hence there is information about input/output relationships. Second, the mitigating activities of installing drain systems have effectively converted a first stage non-point source problem into a second stage point source systems at the sumps. Thus unlike a pure non-point source system, it is possible, in principle, to tax effluents at the outfall in any given area. It is still difficult at this stage to accurately trace subsurface flows and correct for inter-cell externalities.

Our approach has been to consider each of 16 heterogeneous cells a decision making unit. Policies are examined in terms of their effects on the cell-specific generation of drainage and the economic efficiency and equity consequences. Of particular interest is the comparison between information-intensive, high transactions cost efficient policies (such as effluent taxes and cell-specific standards) and more broad brush and less efficient second best policies (such as input taxes/subsidies and water markets).

Results indicate that the range of pollution reduction targets currently under consideration is likely to be feasible using several policy options. The recommended 30% aggregate drain flow reduction can be achieved with irrigation efficiency standards, a uniform or non-uniform water tax, a water market, drainage standards, or effluent taxes. Different policies have different efficiency and equity implications, of course. The least cost solution involves a cost of about $14 per acre over the base case on average, achieved primarily by improving irrigation efficiency by 8-10%. This could be induced with an effluent fee of about $190 per acre foot of collected drain water or a cell-specific standard. Both policies would be costly to initiate, monitor, and administer. A second best policy easier to implement and manage would be a uniform water (input) tax. This policy is less efficient costing $30 per acre in efficiency losses over the effluent standard and it also would be fiscally onerous. To achieve a 30% drainage reduction with a uniform water tax, a tax of about $90 per acre foot of water would have to be levied. It is thus likely that some sort of offsets are necessary to make these viable, such as tiered water pricing, water tax/irrigation efficiency subsidies, or lump sum rebates. Wichelns (1991) has examined tiered water pricing and we analyze a combined water tax/irrigation efficiency subsidy. For the case where water is taxed at $45 per acre foot and irrigation costs are subsidized at a 45% rate, drainage reduction of 30% can be achieved. The efficiency costs are about $17/acre over the effluent tax case but net returns are significantly below the base case (about $90/acre). Thus further investigation needs to be devoted to analyzing schemes that improve efficiency at acceptable fiscal costs.

We thus also examine a water market as an instrument that could generally improve efficiency of water use, reduce drainage, and perhaps prove distributionally superior to tax schemes. Our model suggests that a water price of about $60 per acre foot would achieve an efficient initial redistribution within the drainage problem area and begin to free up water that could be sold outside. At this price average crop returns increase by about $18 per acre, the apparent social cost of current inefficient pricing and allocation. Equally important, net returns are higher than the base. At a water price of $90 per acre foot, for example, net returns are $28/acre higher than the base case, achieved at the targeted drainage reduction of 30%.

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SUBSIDIZING AGRICULTURAL NONPOINT-SOURCE POLLUTION CONTROL: TARGETING COST SHARING AND TECHNICAL ASSISTANCE

by

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SUBSIDIZING AGRICULTURAL NONPOINT-SOURCE POLLUTION CONTROL: TARGETING COST SHARING AND TECHNICAL ASSISTANCE

Introduction

As concern over nonpoint-sources of water pollution has risen, agricultural sources have increasingly become a focus of policy. One reason is that agricultural sources account for a large and growing share of pollutants such as nitrogen, phosphorus, pesticides and (in cases like the San Joaquin Valley, California) heavy metals. Recent estimates suggest that surface water damages from soil erosion and associated runoff of agricultural chemicals in the United States were on the order of $9 billion annually (Ribaudo). Groundwater contamination by leaching of agricultural chemicals has also become a serious concern nationwide (Patrick, Ford and Quarles).

Pollution from agricultural runoff is of special concern in the Northeast and Mid-Atlantic regions, where surface and ground waters are heavily used due to high population, so that damages from nutrient and pesticide pollution from agriculture tend to be very high. Ribaudo estimated that these regions incurred 23 percent of total offsite damage from soil erosion nationwide. Estimated damage per ton of soil eroded was $5.12, almost twice as high as damage per ton of soil eroded in the second highest region.

The traditional approach to soil erosion and agricultural runoff problems in agriculture has been to promote so-called “best management practices” (BMPs), defined as cultural practices that reduce soil and nutrient losses at reasonable cost. The U.S. Department of Agriculture (USDA) and agricultural experiment stations across the country have expended considerable effort developing, testing and adapting BMPs to local conditions. Moreover, a substantial share of the technical assistance provided to farmers by state cooperative extension services has been spent demonstrating the uses of BMPs and helping farmers incorporate BMPs into their production operations.

Until recently, adoption of BMPs has been strictly voluntary. Government policy has concentrated on developing BMPs, persuading farmers to adopt them and providing technical assistance to farmers wishing to adopt. Growing concern over agricultural nonpoint-source pollution, however, has led to some changes, notably the introduction of the "conservation compliance" provision of the 1985 farm bill, which requires farmers to use farming practices in accordance with conservation plans approved by the Soil Conservation Service (SCS). Failure to comply results in ineligibility for all agricultural benefits. Full compliance was required by 1991. To ease the burden of compliance, the bill created a program that reimburses farmers for a portion of the cost of installing approved BMPs. Under this Agricultural Cost Sharing (ACS) program, the federal government reimburses farmers for 50 to 75 percent of the cost of installing BMPs whose plans have been approved by the local SCS office. States may add funds to increase the cost share rate.

Economists have long argued that subsidies are a poor policy instrument for pollution control. Baumol and Oates noted that because subsidies increase the rate of return in the polluting industry, they eventually lead to expansion of the industry. If the subsidies attract enough new investment, total pollution may increase even though each firm is polluting less than previously. The corresponding case in agriculture is that subsidizing soil conservation and runoff control measures may make it profitable to cultivate land so highly erodible that it would have otherwise been left as pasture. Erosion and runoff will increase on this land and, if a sufficient quantity is brought under cultivation, total agricultural nonpoint-source pollution may actually increase. Theoretical considerations, then, suggest that measures such as fertilizer taxes or regulations mandating the use of animal waste storage facilities and other runoff control measures would be more efficient ways of controlling agricultural nonpoint-source pollution.

Why, then, are subsidies such as cost sharing and technical assistance for installation used in agriculture? The principal problem appears to be that of financial hardship imposed on farmers, especially small farmers, who may lack the collateral or the cash flow to finance or support investment in the runoff control structures favored by SCS. In such cases, conservation compliance might force them out of business. Alternatively, runoff control practices may exhibit economies of scale that would make them profitable for large operations but not on small ones. Such would appear to be the case for storage facilities for livestock wastes, for example (Holik and Lessley).

The literature on behavioral factors influencing adoption of new agricultural technologies in general and soil conservation technologies in particular also suggests a need for policies targeted at small farmers. It has been widely observed that small farmers are less likely to adopt new agricultural technologies, at least until their profitability is firmly established (see Feder, Just and Zilberman). One reason may be credit constraints. Another may be risk aversion: Large farmers are more likely to adopt new, riskier technologies because they can diversify more against risk (Just and Zilberman). In the U. S., several studies investigating the adoption of conservation tillage and other soil conservation measures have noted that adoption rates were higher for large farmers than small ones (Ervin and Ervin; Gould, Saupe and Klemme; Lee and Stewart; Norris and Batie; Rahm and Huffman).

This paper uses data from a 1986 survey of Maryland farmers to explore the relationship between farm size and (1) participation in the ACS program and (2) access to technical assistance in Maryland. Overall, the data indicate that both programs were used more heavily by larger farmers. This finding is disturbing. The most defensible rationale for these programs is as a means of helping small farmers maintain their competitive position. But if both programs are geared mainly toward large farmers, they may have the perverse effect of increasing the competitive advantages of large farmers and thus have negative repercussions on the structure of agriculture.

Because 1985 was the initial year of the cost sharing program, the information cannot be considered definitive and more complete study will be needed to understand fully the operation of the cost sharing and technical assistance programs in subsequent years. Nevertheless, the findings of this study point to a real need for a complete analysis of these issues.

Agricultural Nonpoint-source Pollution in the Chesapeake Bay Region

Agriculture has been a major focus of policies aimed at improving water quality in the Chesapeake Bay region for some time. Relatively high precipitation, hilly terrain, vulnerable aquifers and estuaries and heavy human use of water resources due to extensive urban areas has made water pollution problems associated with agriculture especially acute (Strand and Bockstael). It has been estimated that agricultural sources account for 57 percent of total nitrogen and phosphorus entering the Chesapeake Bay, including 60 percent of total nitrogen and 27 percent of total phosphorus (Krupnick). Geologic conditions suggest that groundwater in most areas is moderately to highly vulnerable to leaching (Nielsen and Lee) and several studies indicate strong links between agricultural activity and nitrate in drinking water wells (Bachman; Lichtenberg and Shapiro).

One of the major efforts on the part of both the Environmental Protection Agency's (EPA's) Chesapeake Bay Program and the USDA to reduce nutrient enrichment has been the provision of technical information about and cost sharing for BMPs. The State of Maryland, for example, augments federal cost sharing to provide 87.5 percent reimbursement on all eligible practices, Between 1984 and 1988, the federal-state Chesapeake Bay Program spent over $34 million on cost sharing and almost $10 million on technical assistance for BMP adoption. Together, these represented almost three-quarters of the Program's total expenditures during the period.

Small farms play a prominent role in the Chesapeake Bay region. In Maryland, for example, over one-third of all farm acreage in 1987 belonged to enterprises receiving less than $25,000 in annual farm sales, and 45 percent belonged to enterprises receiving less than $50,000 in annual farm sales. Farmers grossing less than $25,000 annually accounted for about 28 percent of total crop land and 23 percent of all cattle in the state. Farmers grossing less than $50,000 annually accounted for 39 percent of total crop land and 30 percent of all cattle. The economics of farming are clearly different for these operations than for full-time commercial farms. The average net cash return per farm from agricultural sales was negative for farms with less than $25,000 in annual sales and under $500 for farms with $40,000 to 49,999 in annual sales. (In fact, the average net cash return per farm from agricultural sales was only about $11,000 for farms with annual sales of $50,000 to 99,999 [U.S. Department of Commerce]). This suggests that programs that focus on small and part-time farmers, as cost sharing and technical assistance are presumed to be, will play a critical role in meeting targets for reductions in nutrient emissions into the Chesapeake Bay.

Data

The data used to examine the use of cost sharing and technical assistance by Maryland farmers came from a 1986 survey of 280 farmers containing information about 23 different runoff control practices. The sample was representative of the state farm population in terms of age and tenure but was weighted toward full-time commercial farmers, especially crop farmers.

The survey contained information on usage of three broad groups of BMPs. The first distinction usually made is between structural and managerial BMPs, former the referring to investments requiring significant capital outlays, the latter to changes in variable input use. Managerial BMPs are often subdivided into two groups, one consisting of practices related to soil management, the other, practices related to nutrient management. Most of the BMPs considered were eligible for cost sharing. Those that were not included minimum and no tillage, fertilizer and manure incorporation, split application of fertilizer and some cases of cover crops (e.g., double cropping with winter wheat). Structural practices included in the survey were gross- and rock-lined waterways, grade stabilization, sediment basins, ponds, troughs, spring development, waste storage structures and lagoons, terraces and diversions. Soil management practices included contour farming, stripcropping, critical area seeding, filter strips, permanent vegetative cover, wildlife habitat, minimum and no tillage and cover crops. Nutrient management practices included split applications of fertilizer and incorporation of chemical fertilizer and manure.

Information on participation in cost sharing and technical assistance programs was obtained as follows. For cost sharing, farmers were asked whether they had received cost sharing money during 1985 and, if so, for which BMPs. Twenty-nine farmers reported receiving cost sharing in 1985. Twenty had received funds for installing rock- or grass-lined waterways, the remainder for ponds. Regarding technical assistance, farmers were asked to report the number of times they had received information about soil conservation during the previous year from a variety of sources, including USDA sources (notably the Agricultural Stabilization and Conservation Service and SCS), the Maryland Cooperative Extension Service and other University of Maryland sources (abbreviated hereafter as MCES), word of mouth (friends and neighbors), print sources and other sources. The reported number of contacts was transformed into a dichotomous measure for each information source.

The survey contained information on several indicators of farm size. Acreage farmed and livestock numbers indicate technical scale of operation and wealth. In this survey, acreage included all land operated, both rented and owned, and thus reflected the scale of operation. However, since 82 percent of the respondents used in the analysis were full- or part-owners, acreage also reflects wealth to some extent. The percent of family income from farming indicates the importance of farming to the family. It may also reflect the opportunity cost of time. In particular, one would expect full-time farmers to have a lower opportunity cost of time, since there are ample periods when little labor is required on the farm. Part-time farmers, in contrast, usually have tighter time constraints and a higher opportunity cost of time in terms of forgone wages. Finally, farm sales reflect volume, cash flow and the economic activity generated by the farm in the community. [1,2]

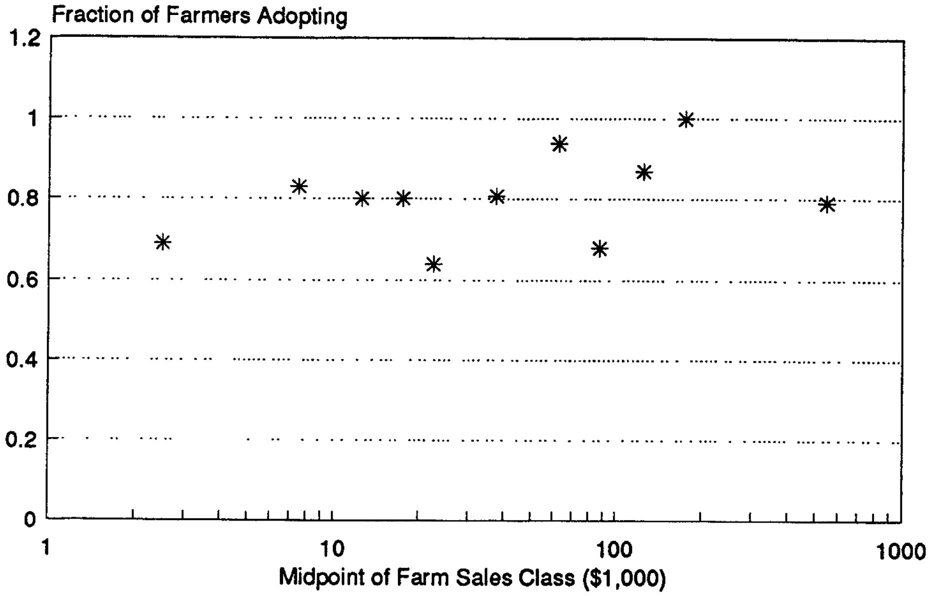
In addition to these variables, the survey also contained information on human capital (age, education, years of experience and attitudes toward environmental quality), topography (shares of land with slopes of 2-7 percent and 8 percent and up), and farm operating characteristics (the percentage of farm income derived from crops, tenure status, shares of crop acreage in corn, tobacco and soybeans).

BMP Adoption, Cost Sharing and Technical Assistance by Maryland Farmers

Figures 1 through 4 summarize some qualitative information about BMP adoption patterns and the use of cost sharing and publicly financed technical assistance as they relate to farm size. For this purpose, gross farm sales was used to measure size of operation, since it should capture much of the information from all of the other variables.

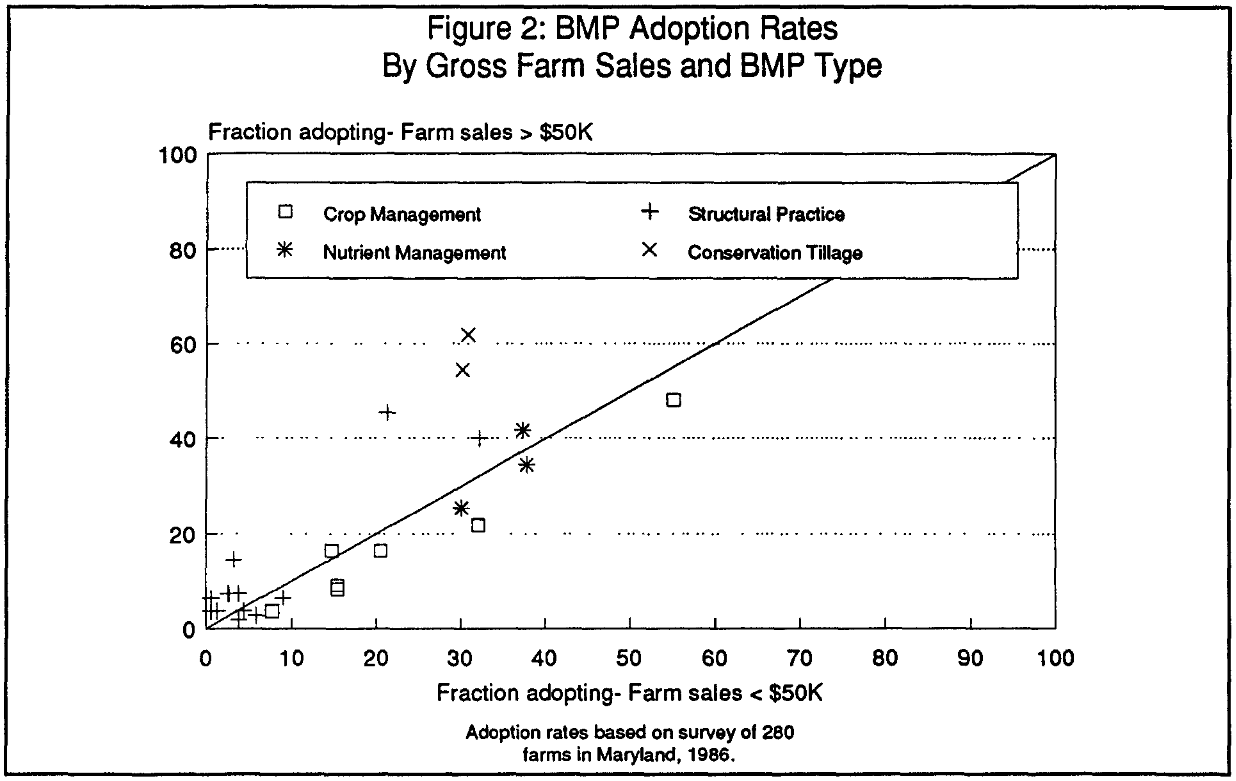
As noted above, existing empirical evidence indicates that larger farmers are more likely to adopt BMPs in the absence of cost sharing. Respondents of this survey were questioned regarding whether they had adopted BMPs without cost sharing. This information can be used to examine, in a very gross sense, the effect of farm size on relative profitability of BMP adoption, in that non-subsidized adoption rates should reflect the extent to which BMPs are believed to be profitable in and of themselves. As Figure 1 shows, a large majority reported having adopted at least one BMP without government aid. Moreover, there were no significant differences in these adoption rates as farm sales varied.[3] This suggests that, if farm size affects BMP adoption, it affects the types and numbers of practices adopted rather than whether a farmer adopts at least one BMP.

Figure 1: Fraction of Farms Adopting BMPs Without Cost Sharing, 1985



Percentages based on a survey of 280 Maryland farmers in 1986

Figure 2 plots adoption rates for large operations, classified as those having more than $50,000 in annual sales, against those for small operations (those with sales of less than $50,000 annually). The diagonal line from the origin represents all points where adoption rates for the two groups are identical. Adoption rates for structural BMPs were on or above this line, indicating that large operations had higher adoption rates. The difference in adoption rates was especially great for grassed waterways and for waste storage structures, both of which tend to have high investment costs. As noted above, budget information suggests that waste storage structures, at least, also exhibit economies of scale. Limited and no tillage were also used much more frequently by large farmers than smaller ones: interestingly, neither is eligible for cost sharing. Adoption rates for soil management practices lay on or below the line, indicating that small operations had higher adoption rates. Nutrient management practice lay quite close to the line, indicating no difference in adoption rates.

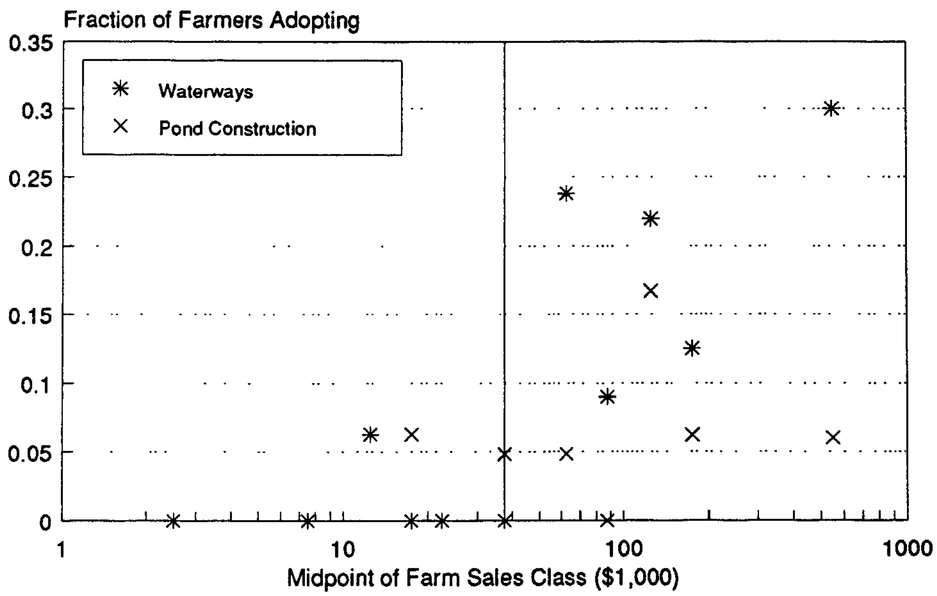


A multivariate analysis of these data performed by Lichtenberg, Strand, Lantin and Lessley confirms the patterns evident in Figure 2. They estimated a reduced form model of farmers' choices among 11 groups of BMPs using a maximum likelihood probit procedure. The results they obtained indicated that full-time farmers were more likely to use all structural practices. The use of grass- and rock-lined waterways and of ponds was not affected by acreage, indicating a lack of economies of scale.

Interestingly, their results indicated that human capital characteristics influence adoption of managerial BMPs but not structural ones. Older farmers were significantly less likely to adopt almost all managerial practices, while farmers with more experience and education were significantly more likely to use them. In contrast, human capital measures exerted no statistically significant influences on adoption of structural BMPs.

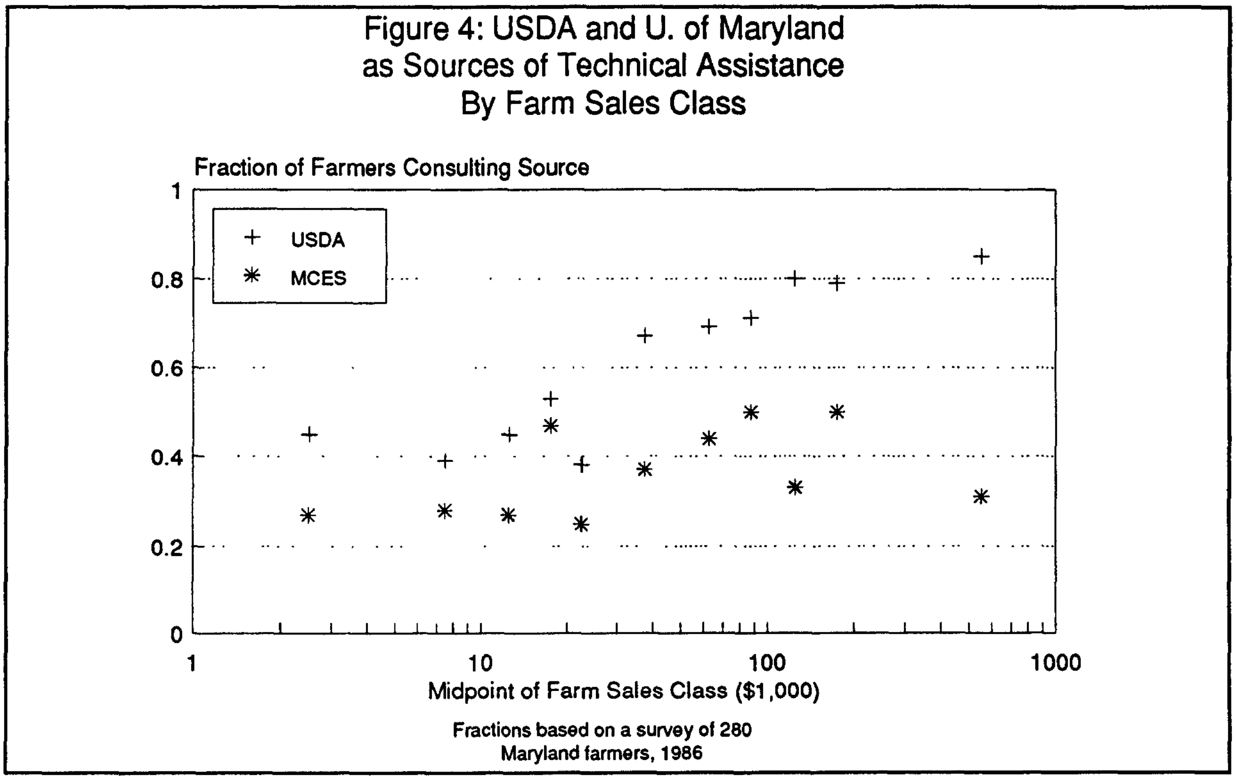
Figure 3 plots the fraction of farmers adopted BMPs with cost sharing against the log of farm sales. There is an apparent strong positive relationship between adopting with cost sharing and sales class. Moreover, this relationship appears to exhibit a threshold. Waterways, the BMP most often receiving cost sharing, had no adoption by farmers with 1985 sales less than $9,999, and almost no adoption by farmers earning less than $50,000 annually, despite the fact that they exhibit no economies of scale. On the other hand, nearly 30 percent of the farmers earning sales in excess of $200,000 received cost sharing for waterway construction.

Figure 3: Fraction of Farms Adopting BMPs with Cost Sharing



Fractions based on a survey of 280 Maryland farmers in 1986

The fraction of farmers having had some contact with the USDA and MCES are plotted by farm sales class in Figure 4. It can be seen that interactions with USDA sources about soil and nutrient conservation occurred twice as frequently among farmers in the largest sales class as among farmers in the lowest sales class. Interaction with MCES sources also increases with farm sales class but at a substantially lower rate, with the maximum fraction occurring in the $75,000-$200,000 range,



Modeling Participation in Cost Sharing and Technical Assistance

These patterns are suggestive, but need confirmation from formal statistical modeling in a multivariate framework. The farmer's decision process about whether to participate in cost sharing or obtain technical assistance was modeled as follows. It was assumed that farmers make simultaneous choices about which farming practices to adopt, whether to participate in cost sharing and whether to obtain technical assistance from federal or state agencies. Let $\mathbf{y_{ij}^*}$ be farmer j's expected gain from adopting practice i (or participating in cost sharing or seeking technical assistance from agency i). Assume that farmer j adopts each practice (engages in cost sharing, obtains technical assistance from agency i) for which yij* > 0. Let I_i be an indicator variable taking on a value of 1 if $y_{ij}^* > 0$ and a value of zero otherwise. Assume further that the expected gains from adoption and participation are a linear function of a set of K explanatory factors $X_i = (x_{i1}, \ldots\ x_{iK})$ plus a vector random components $U_i = (u_{i1}, \ldots\ u_{iM})$, so that the expected utility the i^{th} farmer derives from selecting the m^{th} practice or participating in the m^{th} program can be written:

$$\sum_{m=1}^{M} y_{im}^{*}\gamma_{mi} = \sum_{k=1}^{K} x_{ik}\beta_{km} + u_{im}, \tag{1}$$

or, in matrix form,

$$Y_i^{*}\Gamma = X_i B + U_i, \tag{2}$$

where T and B are respectively MxM and KxM matrices of parameters.

This system of equations can be solved to obtain a system of reduced form relationships

$$Y_i^{*} = X_i \Pi + W_i, \tag{3}$$

where $\Pi = B\Gamma^{-1}$ and $W_i = U_i\Gamma^{-1}$. ..If the random errors in the reduced form system are distributed normally, then the reduced form coefficients Π can be estimated consistently using a maximum likelihood probit procedure (Lee). The probit procedure in SHAZAM was used to obtain these parameter estimates (white).

These reduced form coefficients contain the combined direct and indirect effects of behavioral factors on the likelihood of participation in the cost sharing program and on the use of technical assistance, and thus cannot be used to examine interactions between cost sharing, technical assistance and BMP adoption in a definitive way. Moreover, they will reflect the effects of active government outreach, which will alter the transaction costs of acquiring technical assistance differentially according to the characteristics of the farm and farm operator. For example, farmers may decide to adopt a particular BMP and use the cost sharing program after being approached by county extension, ASCS or SCS agents. The reduced form coefficients will include the effects of targeting by these agencies as well as the effects of farmers' decisions.

These coefficients will, however, indicate the net effects of behavioral factors on cost sharing and technical assistance decisions. They are thus of interest for purposes of prediction and targeting, which is the focus of the present study. What matters in this context is not the outreach patterns intended by MCES or USDA or the group targeted for receiving cost sharing, but the net effect of those programs. In other words, what matters is which groups actually received cost sharing and technical assistance. It is precisely this information that the reduced form coefficients convey.

Reduced form equations of this kind were estimated for cost sharing and for technical assistance from the U.S. Department of Agriculture (USDA) and MCES. Farm size was measured in the four ways discussed previously: Gross sales was used as the major summary measure of size; percentage of household income derived from farming was used to measure the importance of farm income (and, possibly, the opportunity cost of labor); livestock numbers (dairy, beef and poultry) indicated scale of operation and wealth; and acreage cultivated indicated scale of crop operation and, to a lesser extent, wealth. To capture the nonlinearities apparent in Figures 1-4, quadratic terms were included for all four measures of farm size. Because the linear and quadratic terms were highly collinear for acreage, percentage of income derived from farming and livestock numbers, only one was included in the final regressions. The quadratic term fit best for acreage; the linear terms fit best for percentage of income derived from farming and livestock numbers. Also included in the estimated models were human capital indicators (age, education measured by years of schooling, experience measured by years farming and reported concern over environmental quality), tenure status (a dummy having a value of one for full- or part-owner operators and zero for tenants or landlords), topography (percentages of land with slopes of 2 to 7 percent and 8 percent or greater) and cropping patterns (shares of acreage in corn, tobacco and soybeans).

The estimated coefficients for these equations are shown in Table 1.

Farm Size and Cost Sharing

It is readily seen from Table 1 that cost sharing is more heavily used by farmers with larger operations no matter which way size is measured. The probability that a farmer received cost sharing funds in 1985 increased as farm sales rose for all size classes except the largest. (The marginal effect of farm sales is negative for sales of $306,000 or greater). Farmers with larger dairy herds and farmers with greater cultivated acreage were also more likely to have received cost sharing money, as were full-time farmers.

Human capital, type of operation and topography also influenced participation in cost sharing significantly. Participation was greater among older and more educated farmers. Farmers specializing in corn were also more likely to use cost sharing. Interestingly, full- or part-owner operators appeared to be less likely to use cost sharing than tenants. There is some indication that farmers operating more highly sloped land tended to use cost sharing more as well. This may reflect the use of cost sharing for grass- and rock-lined waterways, which, according to the results obtained by Lichtenberg et al., are used more prevalently in more highly sloped areas.

Farm Size and Technical Assistance

Table 1 also shows that farmers with greater sales are more likely to obtain technical assistance from both federal and state sources. In both cases, the probability that a farmer obtained technical assistance increased as sales increased for all except the very largest. (The marginal effect of farm sales on interaction with the MCES was positive for farms with sales under $312,000; the marginal effect of sales on interaction with USDA was positive for farms with sales under $323,000.) Full-time farmers were less likely to interact with University of Maryland sources; percentage of income derived from farming had no significant effect on interaction with USDA sources. Acreage and the size of the dairy or beef herd had no effect on interaction with MCES sources. Large poultry operations, on the other hand, appeared to use MCES sources less. Livestock numbers had no discernible effect on interaction with USDA sources, but there is some indication that USDA sources had greater contact with larger crop farmers.

Human capital considerations affected the likelihood of getting technical assistance from both sources. MCES sources were consulted more often by more highly educated farmers and those reporting greater concern over local environmental quality. USDA sources were more frequently consulted by farmers with more experience.

Policy Implications

Economists have long agreed that subsidies are a poor mechanism for pollution control because they create an incentive for industry expansion and may thus even result in an increase in total pollution (see for example Baumol and Oates). In the case at hand, cost sharing might make it profitable to cultivate land that would otherwise remain in pasture or forest. If runoff from this land were sufficiently large, total nutrient and sediment loadings into waterways like the Chesapeake Bay could increase, even with reduced runoff from existing agricultural land.

In the case of agriculture, subsidies like cost sharing and publicly provided technical assistance have been justified on the grounds of assisting small family farmers who may be forced out of business by strict pollution control requirements because of inability to finance needed runoff control practices or because these practices exhibit economies of scale that make them unprofitable for small farms. Yet according to the data presented here, the provision of cost sharing and subsidized technical assistance in practice at least appear to be incongruous with that goal.

The regression results presented in Table 1 suggest that cost sharing and subsidized technical assistance were used much more by larger farmers than smaller ones. Participation in cost sharing and use of subsidized technical assistance were increasing in sales for all except the very largest operations. Full-time farmers with greater sales, more crop acreage and larger diary herds were more likely to make use of cost sharing. Farmers with greater sales were more likely to have obtained information on runoff control from MCES and USDA sources as well. USDA sources appeared to be geared especially toward crop farmers. MCES appeared to be reaching part-time farmers more successfully.

Why does this occur? With respect to cost sharing, it is possible that the cost share rate is too low to alleviate credit constraints or to make investment in runoff control profitable for small farmers. Since the current cost share rate is 87.5 percent, this reasoning would imply that cost sharing is poorly suited to small farmers and that alternative approaches need to be found.

The emphasis of the cost sharing program may also be misplaced in terms of small farmer participation. The cost sharing program is geared toward investment in runoff control structures. By contrast, Lichtenberg et al.'s results indicate that small farmers are more likely to use management practices than structural ones. Management practices place a higher premium on managerial skill and own labor than on investment funding and are thus better suited to part-time farmers with smaller sales volume. This logic also suggests that other approaches, specifically training, may be more effective in reaching small farmers than cost sharing.

Time and effort may also be significant deterrents to small farmers. Participation in the cost sharing program has high transaction costs (i.e., "red tape"). These transaction costs tend to be especially great for part-time farmers, because their opportunity cost of time is likely to be higher and because they tend to be less familiar with the operations of agricultural subsidy programs. Full-time farmers usually have a lower opportunity cost of time because of slack time at various times of the year. Larger farmers, especially crop farmers, are more likely to enroll in other agricultural programs as well, and may thus find it easier to negotiate the USDA bureaucracy. The fact that larger farmers are more likely both to participate in cost sharing and to consult with USDA sources makes this rationale quite plausible.

Another possible factor is that of cost. The cost of BMPs is also typically higher under cost sharing, because all practices must conform to SCS specifications. Even with a high cost sharing rate, it may remain cheaper to install practices that do not conform to these specifications.

Thus, smaller operators may avoid the cost sharing program even when they plan on investing in structural measures.

With respect to technical assistance programs, it is possible that USDA and MCES have concentrated on larger farmers because environmental returns appear greater: A large farm will presumably have greater production activity and more potential for pollution and thus pollution reduction. Larger farmers are often perceived as community leaders, and other farmers may follow their lead in adopting new production practices. Larger farmers may also serve as demonstrators of risky new technologies because of their greater ability to diversify against risk. It is also likely that many of the small farmers are "hobby" farmers perceived to be unlikely either to pollute or respond to BMP promotion. Alternatively, USDA and MCES outreach may focus on popularizing runoff control structures instead of management practices that smaller farmers are more likely to find attractive. Finally, the analysis may be an artifact of the data. Although it is thorough in its scope, it dates from the initial year of the cost sharing program. Since that time, these programs may have become broader in scope.

Nevertheless, these findings raise some fundamental questions about the desirability of publicly provided financial and technical assistance for runoff control when this assistance is, in practice, geared toward larger farmers. From an economist's point of view, the soundest rationale for cost sharing and technical assistance for runoff control is a concern for maintaining a desired structure of agriculture, one in which small family farms remain viable. Cost sharing and technical assistance geared toward larger farmers may even undermine such a goal by increasing large farmers' competitive advantage and thus hastening exit of small farmers from the industry.

Moreover, in regions like the Northeast it is becoming increasingly important to have policies that reach small farmers strictly from the point of view of pollution control. Small farmers account for a large share of land operated and agriculture output produced and thus, presumably, for a large share of nonpoint-source pollution as well. In Maryland, for example, 45 percent of total farm land, 30 percent of total cattle and 24 percent of corn production are accounted for by farms that gross less than $50,000 annually. The ability to reduce nonpoint-source pollution from agriculture will clearly depend increasingly on the ability to reduce runoff from small farms. Thus, policies that reach small farmers will be increasingly needed.

If in fact current policies are poorly suited for reaching small farmers, as our results suggest, then a great deal of the current approach to agricultural nonpoint-source pollution control needs to be reconsidered, Research and development effort should be geared toward runoff control measures that will be effective and acceptable on small farms. Our empirical results suggest that management practices requiring low investment and low labor input will be used more widely on small farms, especially those operated by part-time farmers. Technical assistance should be geared toward augmenting management skill on small farms. Outreach should be tailored to reach small, part-time farmers.

Getting small farmers to adopt runoff control measures may require substantial innovations in policy design. Cross-compliance generally has no effect on small farmers, because it's uneconomical for them to participate in farm programs. Small farmers are less likely to be in contact with the traditional forms of technical assistance offered by USDA and MCES. Reaching them may require these agencies to devise forms of outreach that are radically different.

As with any true innovations, the costs of creating and implementing new policies may be large. Moreover, it is not clear that the potential gains in pollution reduction would be worth the costs. However, in that case, it is also not clear that there is any real basis to continue to subsidize investment in runoff control through cost sharing and publicly provided technical assistance. Giving up on small farmers leaves no solid economic rationale for pollution control subsidies in agriculture.

In sum, the current emphases in runoff control in the agricultural research and development system may be misplaced. Development of new runoff control technologies may not be the major problem; devising policies leading to the adoption of runoff control methods, especially by small farmers, may be. In other words, perhaps at this time it would be most productive to think carefull about the real objectives of nonpoint-source pollution control in agriculture.

Footnotes

[1] Farmers were asked to classify their 1985 farm sales into one of the following groups: 1) 0-$4,999; 2) $5,000-9,999; 3) $10,000-14,999; 4) $15,000-19,999; 5) $20,000-24,999; 6) $25,000-49,999; 7) $50,000-74,999; 8) $75,000-99,999; 9) $100,000-149,999; 10) $150,000-199,999 and 11) over $200,000.

[2] Net farm income, which is difficult to measure and may not reflect fully the size of the farm enterprise in terms of volume of product or sales, was not included.

[3] The fraction of farmers adopting at least one BMP without cost sharing was regressed against the mid-point of sales in each of the eleven sales classes (with average sales for farmers grossing $250,000 and up, as reported in the 1987 Census of Agriculture for Maryland, used as the midpoint for the eleventh class) using a double-log form. The coefficient of the log of sales was 0.033 with a t-statistic of 1.23 and R^2 of 0.14.

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Table 1
Estimated Coefficients from Reduced Form Probit Models

Variable	Cost Sharing	University of Maryland	USDA
Constant	-8.244 (3.182)	-3.044 (2.186)	-2.141 (1.504)
Age	0.045 (2.036)	-0.003 (0.238)	-0.012 (0.988)
Education	0.174 (2.320)	0.071 (1.528)	0.030 (0.639)
Concern About Environmental Quality	-0.064 (0.168)	0.498 (1.776)	0.285 (1.008)
Years Farming	-0.013 (0.967)	0.004 (0.506)	0.023 (2.636)
Gross Sales	0.009 (1.724)	0.009 (2.454)	0.010 (2.304)
Sales Squared	-0.148×10^{-4} (1.789)	-0.148×10^{-4} (2.46)	-0.159×10^{-4} (2.164)
Percent of Income from Farming	0.025 (3.191)	-0.005 (1.572)	0.293×10^{-3} (0.085)
Cultivated Acreage Squared	0.513×10^{-6} (3.182)	-0.189×10^{-7} (0.083)	0.106×10^{-5} (1.403)
Full or Part Owner	-0.687 (1.649)	0.233 (0.840)	0.092 (0.327)
Size of Dairy Herd	0.005 (1.723)	0.455×10^{-3} (0.228)	0.001 (0.352)
Size of Beef Herd	0.006 (0.973)	0.003 (0.904)	-0.689×10^{-3} (0.217)
Size of Broiler Flock	-0.118×10^{-4} (0.800)	-0.106×10^{-4} (1.417)	-0.670×10^{-5} (0.954)
Share of Acreage in Corn	1.405 (2.235)	0.051 (0.123)	-0.128 (0.298)
Share of Acreage in Tobacco	1.496 (1.161)	-0.824 (0.794)	-0.608 (0.642)

Share of Acreage in Soybeans	-0.825 (0.881)	0.126 (0.291)	-0.083 (0.197)
Percent of Land with 2-7% Slope	0.004 (0.534)	0.002 (0.533)	0.005 (1.290)
Percent of Land with Slope 8% or Greater	0.014 (1.515)	-0.004 (0.845)	0.006 (1.199)
N	220	167	167
McFadden R^2	0.448	0.102	0.160
Absolute values of asymptotic t-statistics shown in parentheses.			

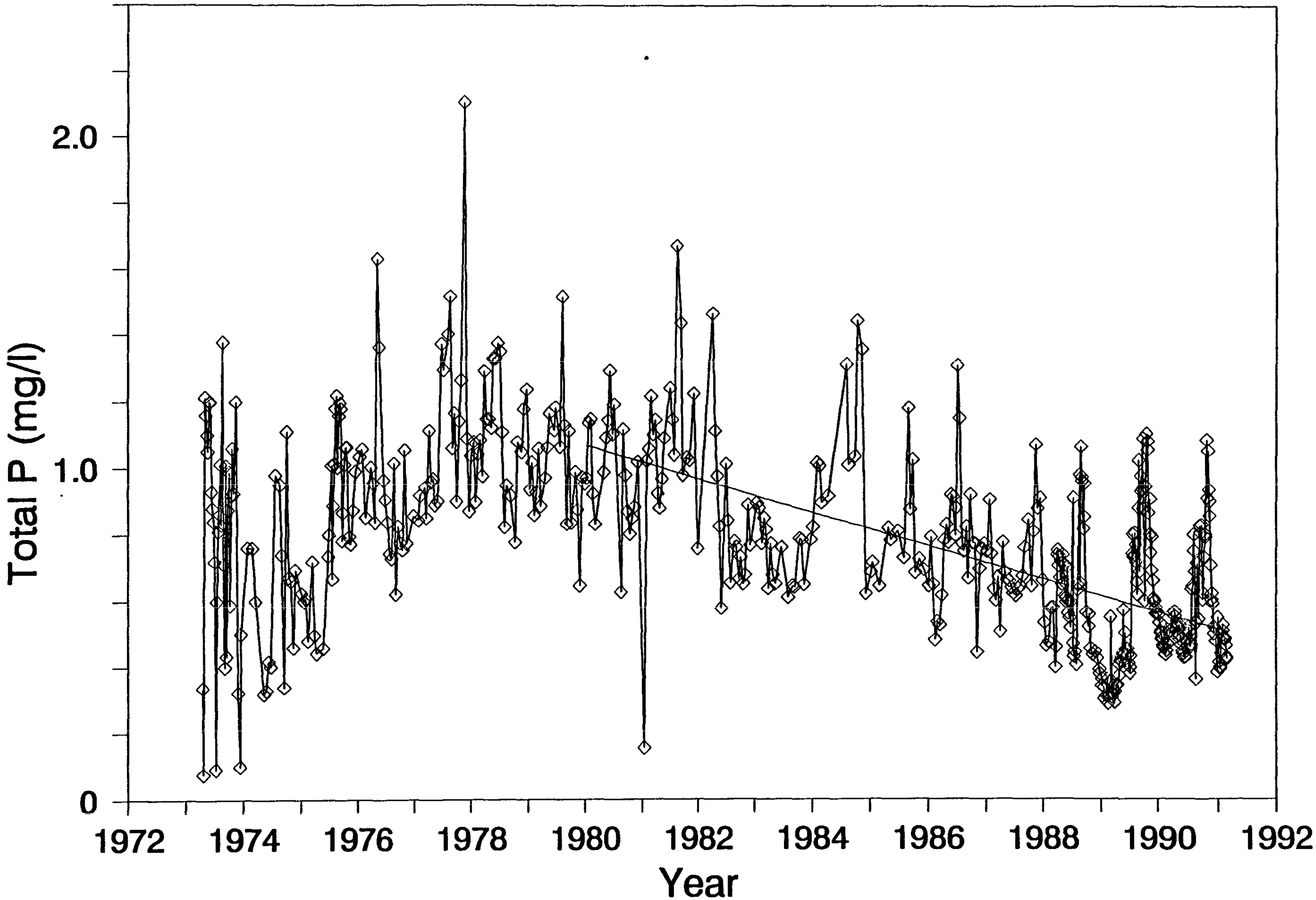


Figure 5. Total phosphorus concentration measurements for Taylor Creek-Nubbin Slough, Structure S-191, for the period 1973-1991 (trend based on seasonal medians).

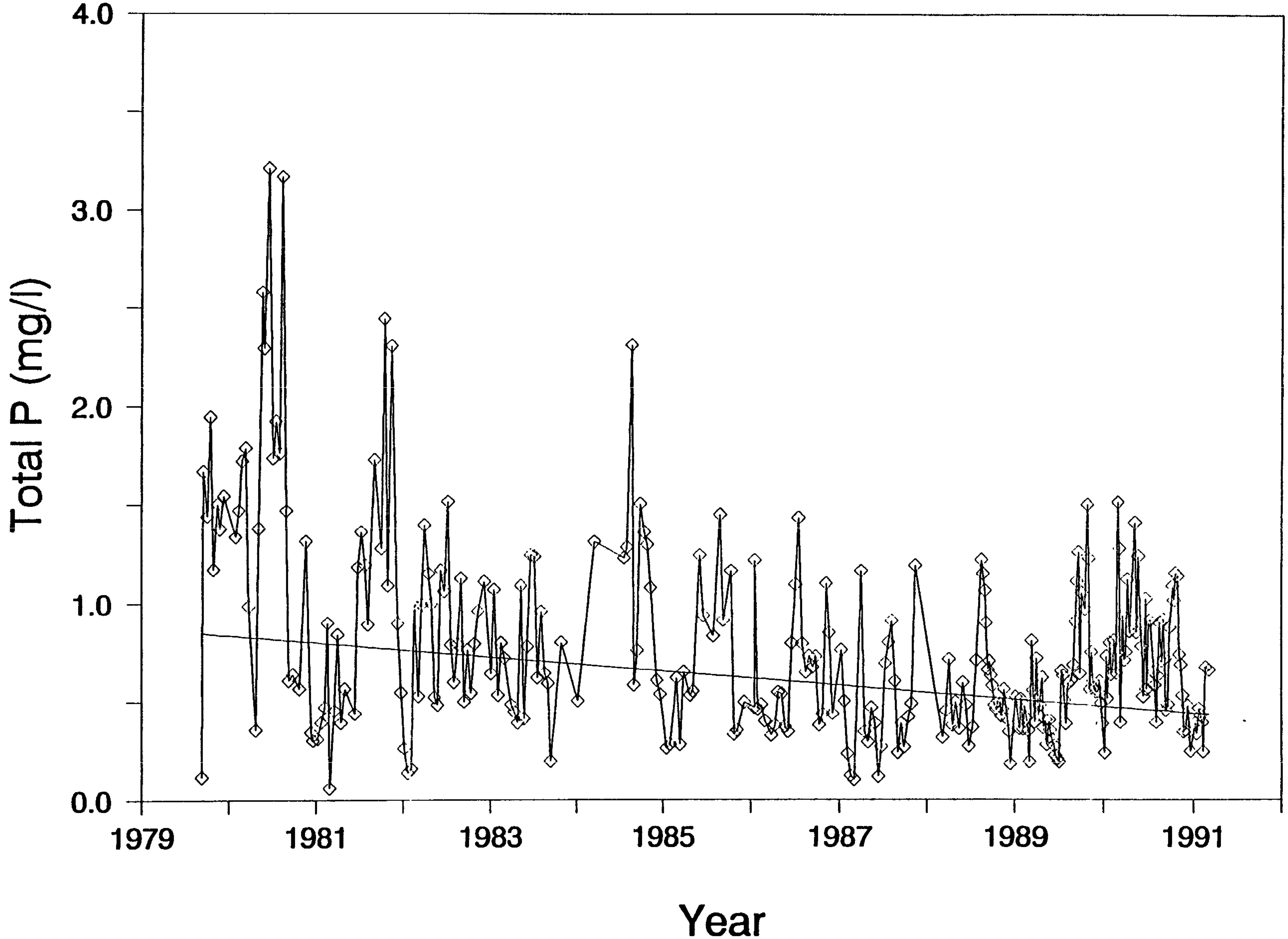


Figure 6. Total phosphorus concentration measurements for the Taylor Creek Headwaters, Structure TCHW 18, for the period 1979-1991 (trend based on seasonal medians).

Figure 13:
Cooperative weight determination game.

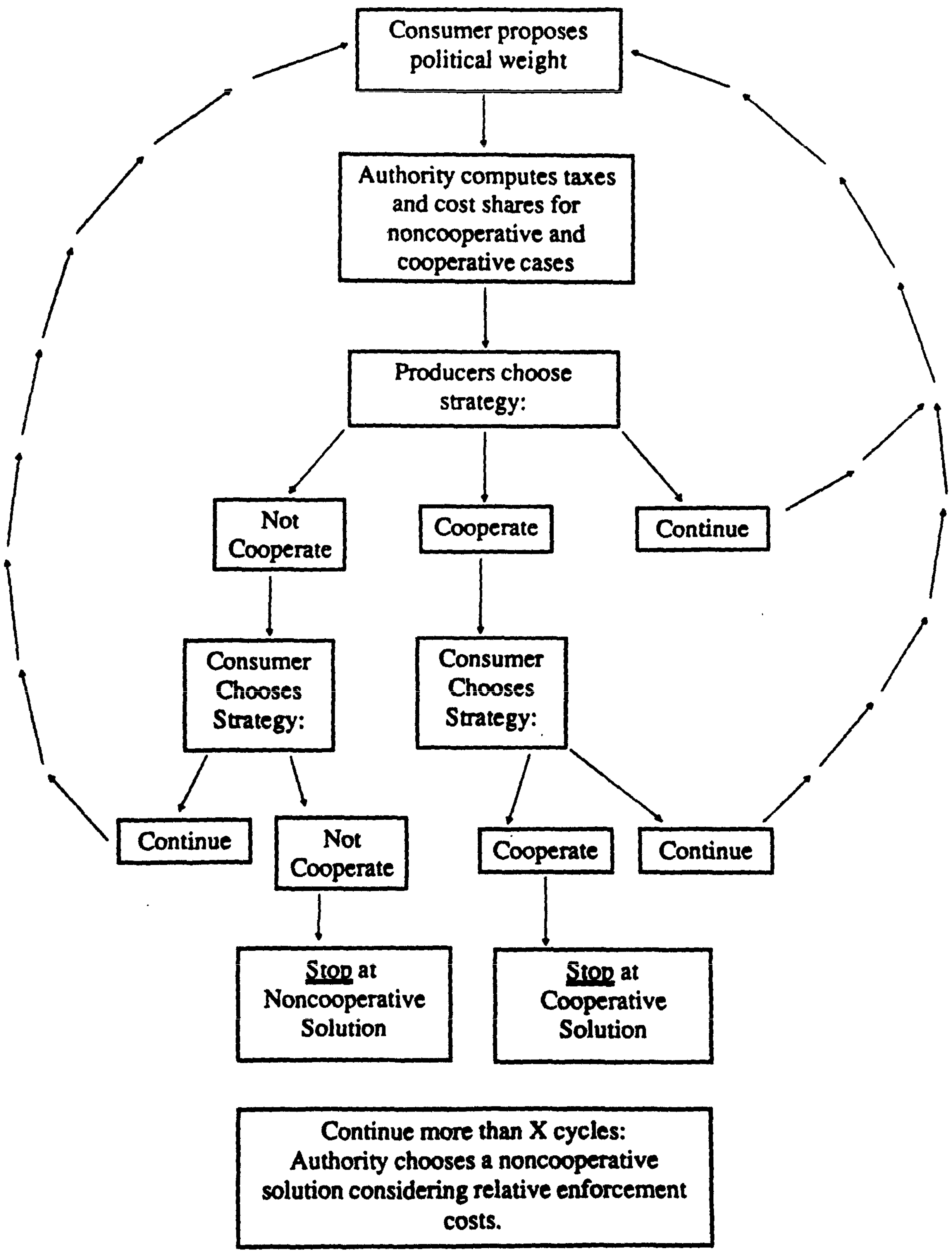
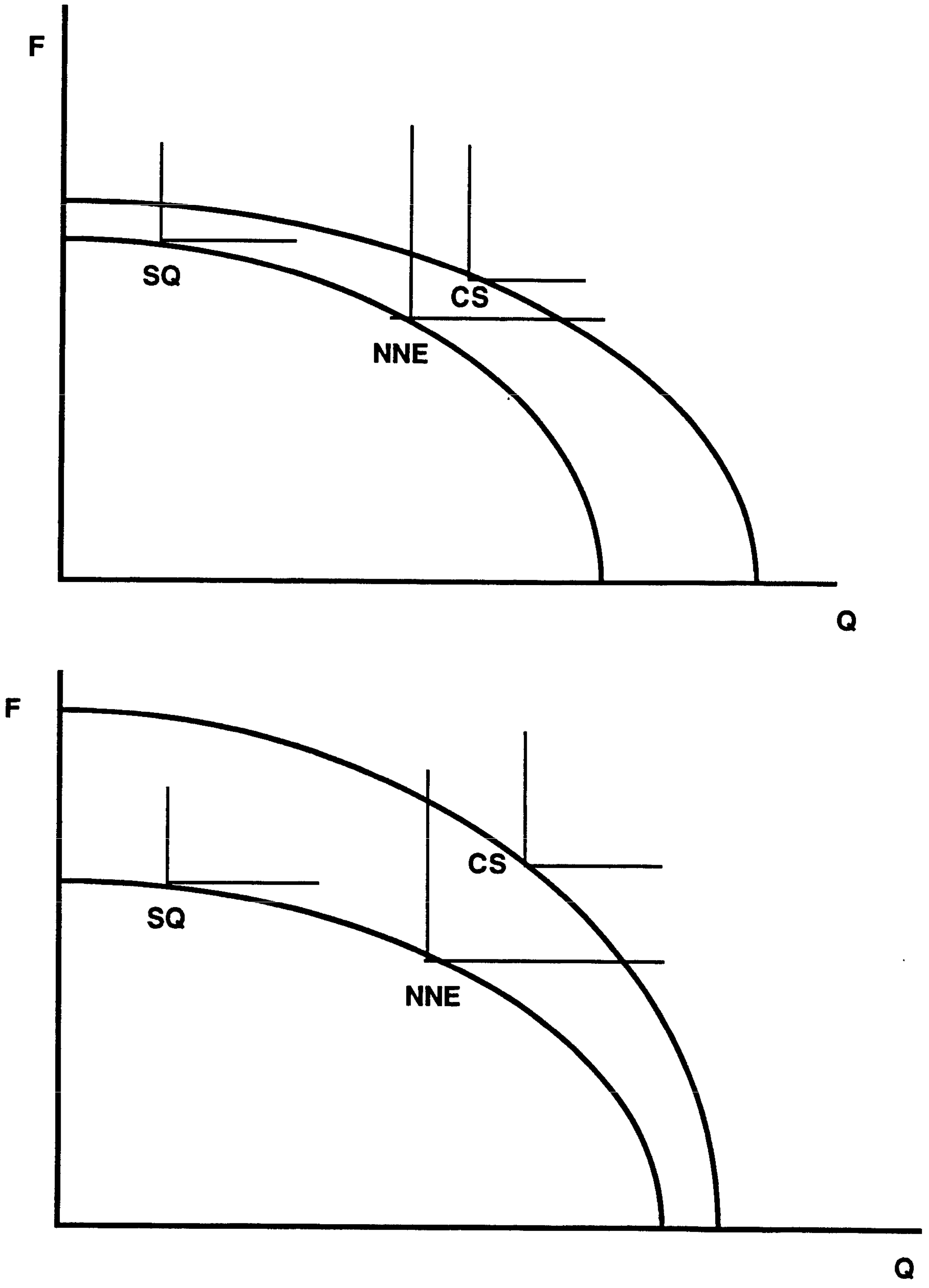


Figure 14:
A Cooperative Solution and the Corresponding NNE "Threat Point"



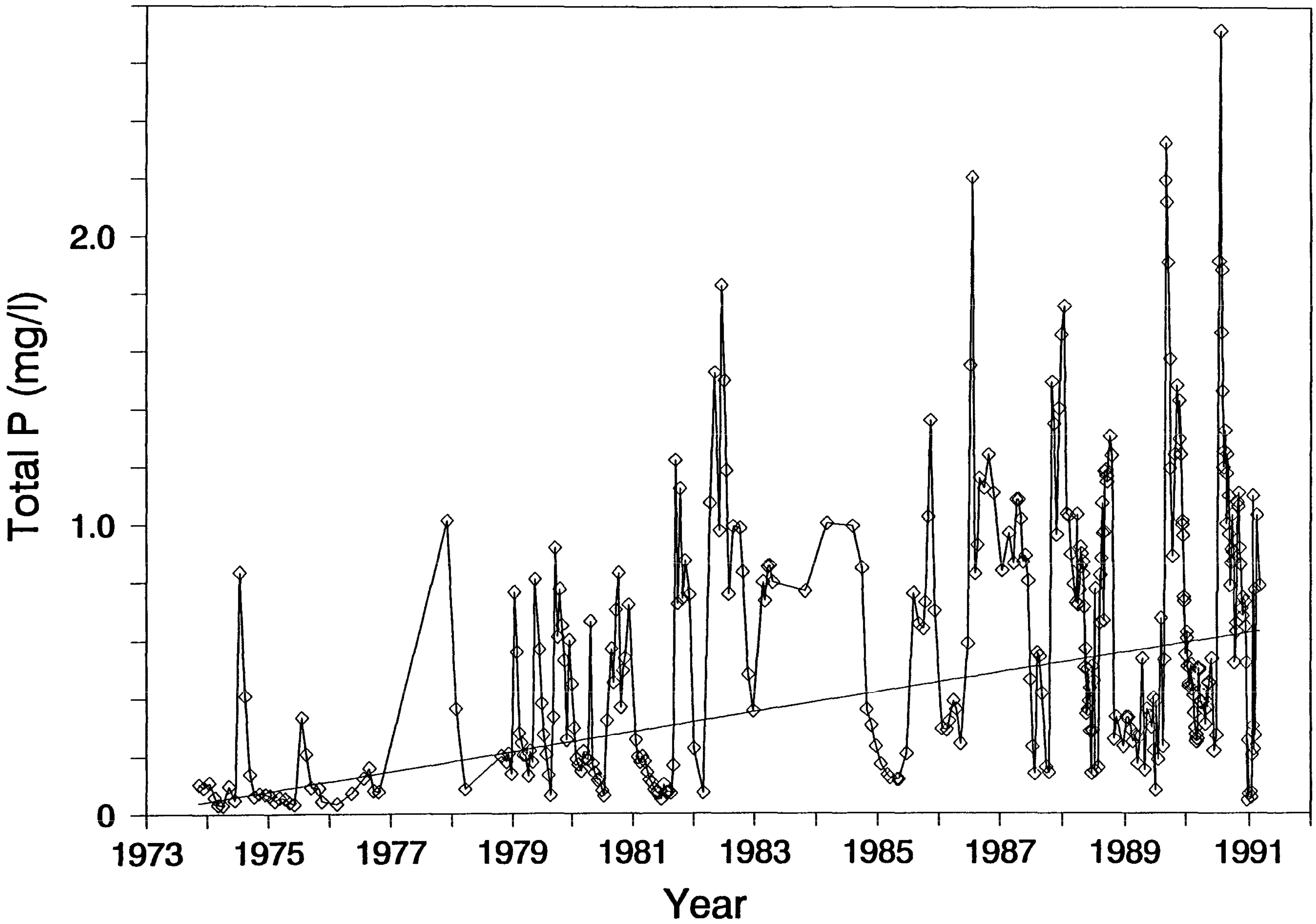


Figure 7. Total phosphorus concentration measurements for Structure S-154 in the Lower Kissimmee River Basin for the period 1973-1991 (trend based on seasonal medians).

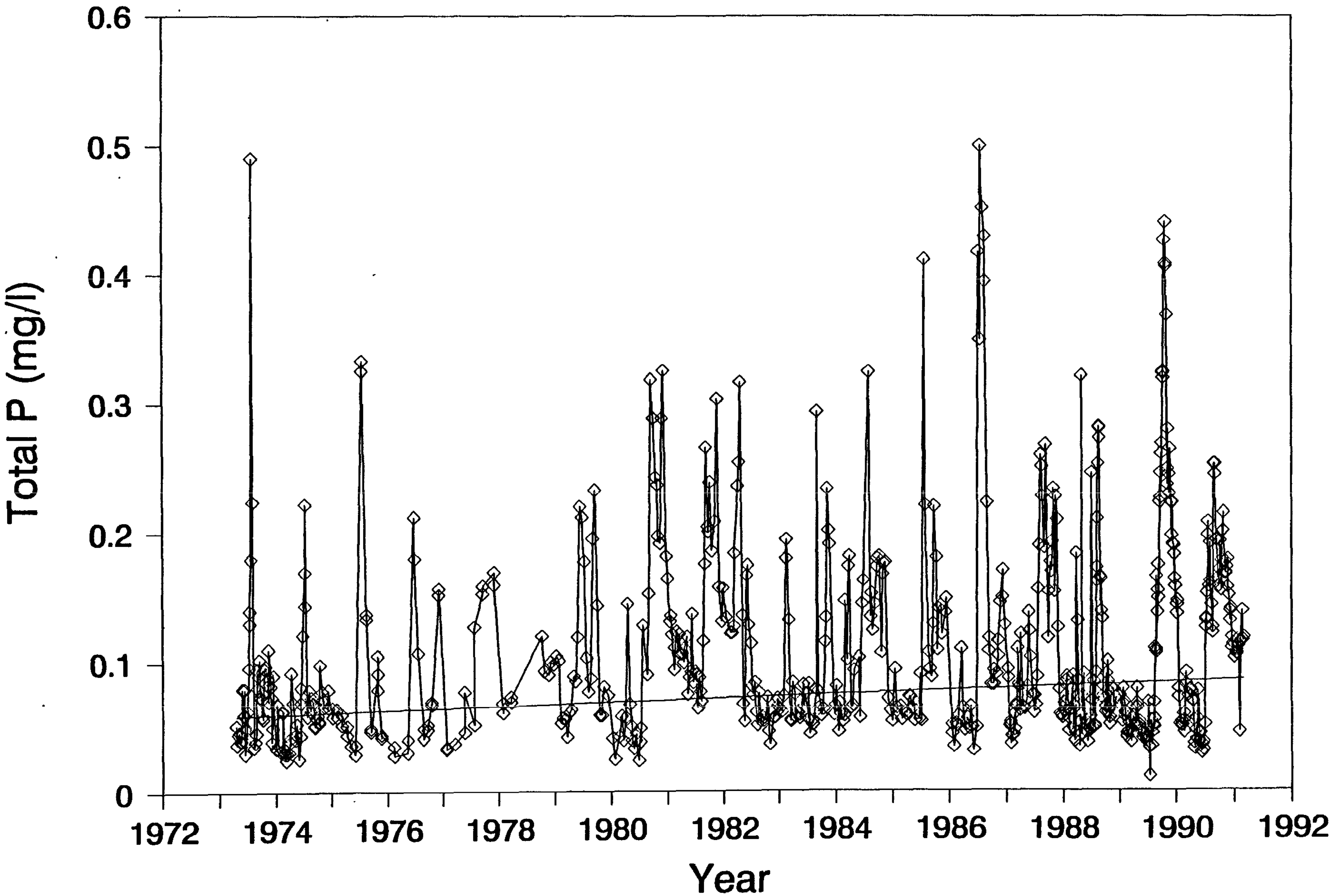


Figure 8. Total phosphorus concentration measurements for Structure S-65E in the Lower Kissimmee River Basin for the period 1973-1991 (trend based on seasonal medians).

Figure 15:
Tax on water use and Noncooperative Nash Equilibrium

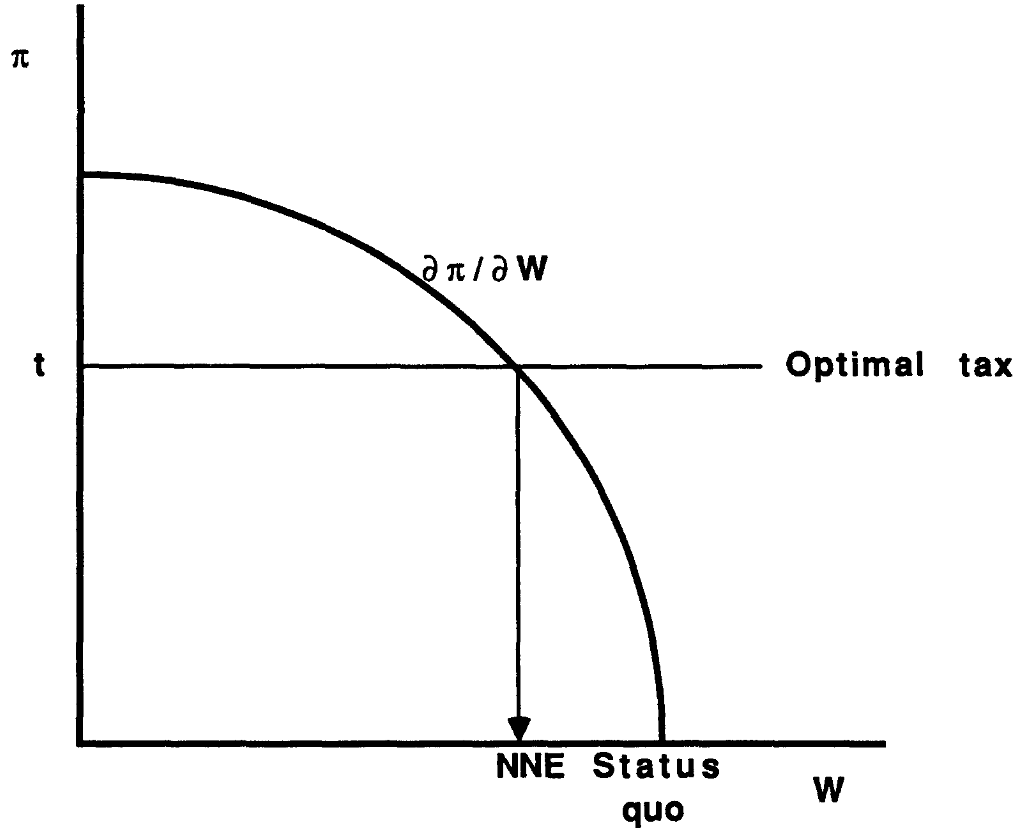


Figure 16:
Substitution between agricultural production and environmental quality in a regional setup with and without cooperation for various weights