WATER MANAGEMENT AND EXTERNALITIES IN TROPICAL ARID- AND SEMI-ARID AGRICULTURE: AN ECONOMIC APPROACH BASED ON EXAMPLES OF NITRATE LOSSES AND SALINIZATION

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ABSTRACT

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This paper summarizes the theory of welfare economics and natural resource allocation, and illustrates how application of this theory can lead to Pareto improvements in the design of irrigation systems in tropical arid- and semi-arid agricultural systems. Drawing on the two examples of nitrate losses and salinization, the paper concludes that economic analysis can help design policies to conserve natural resource stocks, but that such analysis is limited by insufficient data, methodological difficulties, and by a lack of integration between natural and social research.

A portion of the paper discusses the physical and biological processes associated with nitrate losses and salinization, and reviews research on their magnitudes in arid- and semi-arid zones. This review demonstrates that certain combinations of hydrological conditions and farming systems will likely result in nitrate losses and/or salinization. The design of irrigated systems in these cases might be improved by analyzing the externalities associated with these phenomena.

There are two policy approaches -- curative and preventive -- to mitigate losses from nitrate pollution and salinization. Preventive policies, which entail directive measures (e.g., standards) or the creation of economic incentives and disincentives, are generally more cost-effective and risk-reducing. Taxes, however, should be avoided

where natural resource demand is inelastic. Combining subsidies along with taxes allows greatest flexibility in cost distribution, but also reduces the net benefits from mitigation. Given the institutional weakness of most lower income countries, governments attempting improved management of natural resource stocks must rely primarily on conservation subsidies rather than on standards, taxes, or curative measures.

More generally, informed judgments on the magnitude of externalities associated with any one natural and social environment depend on information being available to economists. Yet comparative data on agro-ecological dynamics in the tropics are lacking. Detailed and generalizable models have been developed for water and soil salinity, but are less available for nitrate losses. Unlike salinity research, nutrient loss studies have not widely benefitted from coordinated agronomic and economic research. Regionally coordinated agro-ecological research should be designed to illuminate the policy implications of natural resource management. The effects of irrigation on soil fertility and on water quality are two high-priority topics.

Economists need to evaluate the contributions of natural resource quality to individual well-being. Policy makers meanwhile may choose to establish qualitative standards for important natural resources, and to distinguish among broad categories of natural resources based on their substitutability. Economists also must understand how current policies generate farm-level incentives for overuse of natural resources, since corrective policies may vary for different incentives. Finally, economists should closely examine the degree to which the use of high discount rates in economic analysis leads to environmental degradation by reducing the value of future benefits and costs.

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I. INTRODUCTION

A. Conceptual Background

Conservationists, economists, and policy makers largely agree that environmental degradation is among the more important development problems facing lower income nations (Southgate 1988, USAID 1988, Repetto 1988). This degradation reduces the economic productivity of natural resources such as soil, forests, water, and air. In many lower income countries researchers and policy analysts attribute declining trends in per capita agricultural production to a loss of soil productivity (Eckholm 1976, WCED 1988). (See Appendix A.)

Environmental degradation partly results from short-sighted agricultural policies and development projects that neglect the potential negative impact of agricultural production on adjacent ecosystems; cause migration to more fragile lands; subsidize urban populations thereby increasing the stress on rural resources; or ignore changes that occur to agricultural production systems over time (Repetto 1988, Spears 1988). Market "failure" and information deficiencies can also provide important incentives to overly exploit renewable natural resources (Upstill and Yapp 1987, Southgate 1988, Ruff 1977). Farmers are not commonly held responsible for the off-site costs their production decisions may induce. Often they may not even be aware of these or other costs stemming from their land-management decisions.

The allocation of natural resources has recently become an important topic for economic analysis (Barbier 1988, Southgate 1988, Dorfman and Dorfman 1977). There is growing recognition that certain uses of natural resources can result in real economic losses which undermine improvements in human welfare (Dixon et al. 1988). The agricultural policy agenda of many lower income countries, and especially of bilateral and multilateral organizations, includes a new emphasis on the conservation and management of natural resources (USAID 1988, Spears 1988, WCED 1988). Despite these foci, economic analysis and agricultural policy usually omit establishing values for environmental goods and services because the analytical framework for doing so remains underdeveloped, and the tradeoffs unclear. Hence, their contributions to social welfare remains insufficiently understood (Quiggin 1988, Southgate 1988, Hufschmidt et al. 1983).

Randall (1981) refers to economic externalities whose elimination results in greater gains than losses to the economy and leaves no one worse off, as Pareto-relevant externalities. Externalities consist of consequences of the activities of one enterprise that negatively or positively affect the production or utility function of others. For example inland industrial effluent which reduces revenue from coastal tourism may present a situation where overall economic performance may be enhance if resource use in one activity is reduced or in some way altered. Pareto-relevant externalities should be a priority focus of policy makers who wish to improve resource allocation.

Externalities associated with some environmental problems such as deforestation and the loss of genetic diversity are often absorbed (in unknown quantities) by future generations unable to represent their interests. These problems raise questions as to what criteria a society should use to measure improvements in economic performance over a longer time frame. Specifically, criteria are needed to guide decisions on consuming today versus conserving for tomorrow. Although identifying Pareto-relevant externalities may prove a useful guide, for an actual Pareto improvement to occur the winners must compensate the losers. Where winners and losers are separated by time rather than space, compensation is more difficult, and hence achieving the Pareto improvement becomes less likely. A second priority of policy makers, thus should be to identify resource allocations which achieve some equitable balance between the interests of present and future populations.

These priorities require an economic analysis of natural resource goods and services. The undertaking of these analyses raise problems which can be grouped into three categories. First, predicting the effects of agricultural production on the environment remains complex and uncertain (Upstill and Yapp 1987, Hufschmidt et al. 1983). Data and knowledge limitations greatly restrict one's ability to estimate the effect of human activities on the environment. This is especially true in lower income countries where limited knowledge and data exist on the linkages between natural resources, environmental quality, and economic growth (Barbier 1988, Simons 1989). Until scientific and economic research begins to establish empirical relationships among these three areas, it will be difficult to resolve conflicts between the competing needs of lower income countries for both production and resource conservation.

Moreover, many economists believe that environmental problems are fundamentally caused by a divergence between social and private costs

(Ruff 1977). Even if correct, such a position is simplistic in that the derivation of social versus private cost curves is difficult and controversial. This is particularly true when environmental and natural resource goods and services are involved for which prices are not readily obtained (Dorfman and Dorfman 1977). Establishing the value of these goods and services requires that analysts recognize and carefully examine interactions between the social and natural environments. This in turn requires a longer-term analytical framework than is conventional, and one which captures ecological dynamics. The more common static perspective with private (financial) rather than social (in this case physical) boundaries does not give adequate weight to the ecological and biological interactions that affect environmental quality.

Finally, determining economic tradeoffs presupposes a common standard by which to measure value so that policy incentives, public investments, and social institutions can seek to allocate resources in accordance with their relative social value. Yet it is difficult to compare impacts which are highly dissimilar, e.g., particulate runoff on the one hand and maize yield on the other. Often such comparisons rely on techniques to value natural resources in monetary terms which are neither well-developed nor widely accepted by economists and policy makers. Ruttan (1971, p. 715) concludes that "the formal analysis on which we can draw for environmental and resource planning and policy is seriously deficient."

B. Purpose and Scope of This Paper

This paper summarizes current theory on economic efficiency and natural resource allocation, and suggests how this theory might be applied to empirical resource management problems in lower income countries. The paper is based on the premise that some of the difficulties of incorporating natural resource use into economic analyses can be overcome by illuminating some of the key constraints to undertaking these analyses.

To illustrate the relationship between externalities, economic efficiency, and resource policy, in this paper I analyze water management in tropical arid- and semi-arid agricultural systems, and show how economic analysis can be applied to agricultural development issues associated with important externalities. Many lower income nations have significant land resources classified as arid- or semi-arid. In these areas irrigation may be an essential albeit costly option for increasing agricultural production. Water is usually the most productive input in arid environments (Vyas and Casley 1988, Johl 1980), and irrigation often significantly reduces farm-level risk (Carruthers and Clark 1983). Between 1974 and 1986, the World Bank issued one-third of its agricultural loans for irrigation investment (World Bank 1988).

Yet there are important externalities associated with water development (White 1978, UNESCO 1975). In the past, agricultural development projects have focused primarily on the engineering aspect and financial profitability of irrigation (Repetto 1988, Reisner 1986), and only secondarily on socio-economic impacts.[1] Even when taking socio-economic issues into account, project focus has usually been short-run. The biological and ecological changes which occur with irrigation have largely been neglected in spite of the fact that social losses are often traced to these changes. Irrigation has repeatedly led to unintended changes including soil degradation and agrochemical runoff, the creation of a breeding ground for diseases, and/or off-site impacts which have seriously damaged the productivity of agricultural land as well as the surrounding ecosystem (Johl 1980). In fact, secondary effects from irrigation can present the most serious arguments against irrigation development (Abrol et al. 1988). Several studies have demonstrated that such impacts are important and should be included in economic analyses of water policy (Repetto 1988).

This paper considers two externalities commonly associated with water management. One is the loss of nitrates to adjacent ecosystems. The impact of water management on gradual soil and water salinization illustrates some of the issues associated with a loss of productive capacity over time. An assessment of the specific difficulties associated with economic analyses of these two externalities illustrates the issues confronting similar externalities in other agricultural systems, and suggests an approach for evaluating them.

[1]A 1978 paper presented to the International Commission for Irrigation and Development stated that "[i]rrigation systems...in arid regions, where farming is impossible without irrigation...seldom require any particular economic examination before design" (cited in Carruthers and Clark 1983, p. 3). Quiggin (1988, p. 636) states that when early irrigation schemes in Australia proved commercially nonviable, "there was a general acceptance of the principle that irrigation schemes should be undertaken whenever it was technically feasible to establish intensive agriculture on the irrigated land."

In Section II I review the theory on contemporary welfare economics as it relates to environmental quality and natural resource management. This section also discusses and compares various policy mechanisms and intervention strategies that can be used to improve the performance of agricultural systems. Section III reviews literature on nitrate losses and on salinization in tropical arid- and semi-arid agricultural systems. The review illustrates how system design may result in measurable (but usually uncounted) economic costs both over time and across private boundaries.

Section IV develops a more detailed economic analysis of nitrate losses and salinization caused by water development projects. This discussion identifies the relevant questions, and highlights some important analytical and policy issues.

Finally, Section V summarizes the constraints facing economists and policy makers in formulating improved solutions to natural resource management. I offer several suggestions concerning some analytic and policy approaches useful for addressing problems of natural resource allocation in lower income countries.

II. ECONOMIC THEORY, ENVIRONMENTAL QUALITY, AND PUBLIC POLICY

Economists have developed and refined a theory of natural resource allocation within the broader scope of contemporary welfare economics. This theory and methodological developments which have emerged from it are reviewed below. This section also examines differences between various policy options to improve the performance of agricultural systems.

A. Economic Theory of Environmental Quality

1. Contemporary Welfare Economics and Externalities

Economic theory defines a Pareto-efficient resource allocation as one where no people can be made better off without others being made worse off, and therefore no opportunities remain for voluntary trade (Randall 1981). Where an allocation is not Pareto-efficient, a different allocation (called a Pareto improvement) exists such that some in society can be made better off without anyone being made worse off. Historically, economists have tried to identify situations where resource allocation is not Pareto-efficient, and to formulate policies which can lead to a Pareto improvement.

In terms of marginality principles, Pareto efficiency also means that the marginal social benefit from producing a good equals its marginal social cost. Economic theory also hypothesizes that under ideal market conditions, farmers produce an output level that equates marginal social benefits with marginal social costs -- i.e., is Pareto efficient. When ideal market conditions are not met, private decisions may not be Pareto efficient. That is, opportunities for voluntary trade (a Pareto improvement) may exist.

There are numerous instances where economic "distortions" restrict opportunities for trade that would otherwise lead to a Pareto improvement. One example is a system in which one person's production decisions affect the production or utility function of another.[2] This is generally referred to as a problem of "externalities" since some production costs are borne externally by individuals other than the decision maker.[3] Here market "failure" is said to result because producers undervalue the true social worth of resources used in production. This private undervaluation of production costs (reflected in a downward shift in the supply function) from a social viewpoint is associated with overproduction and therefore lower commodity prices (Randall 1981). It also implies that the possibility for a Pareto improvement exists.

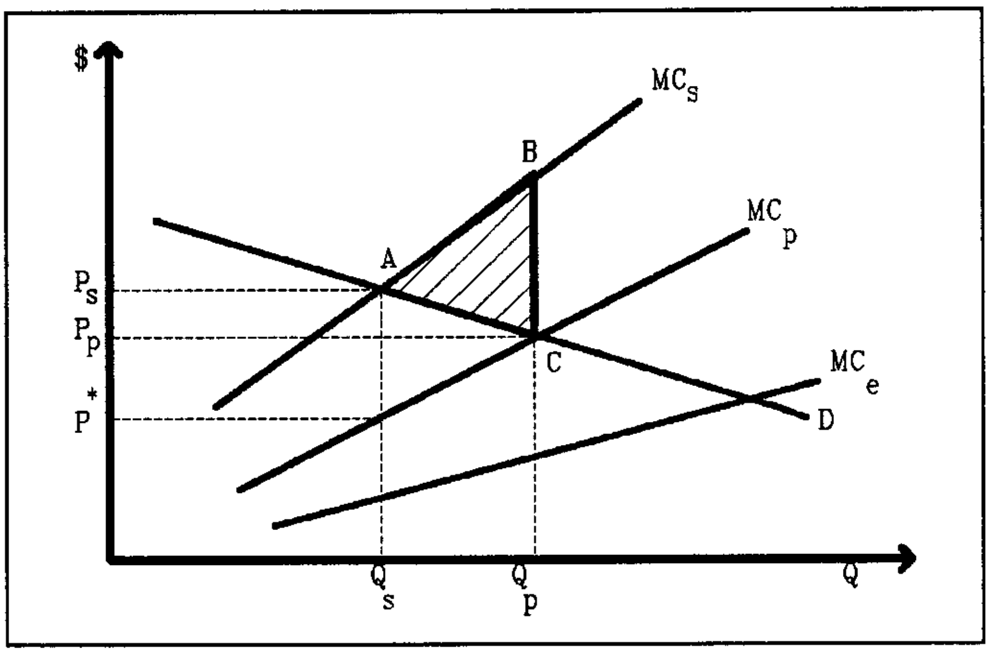
To illustrate the relationship between externalities and Pareto improvement, Figure 1 begins with the concept of market failure. The private marginal cost, or supply function, of farmers for commodity Q is given as MC_p. Private decision making is usually assumed to be Pareto efficient when external costs do not accumulate (Blyth and McCallum 1987). Assume, however, that some pollutant for which producers are not

[2]Economists use the concept of a utility function to describe the well-being or "utility" derived by individuals from increasing levels of consumption.

[3]Although this discussion is limited to external costs, external *benefits* may similarly be relevant.

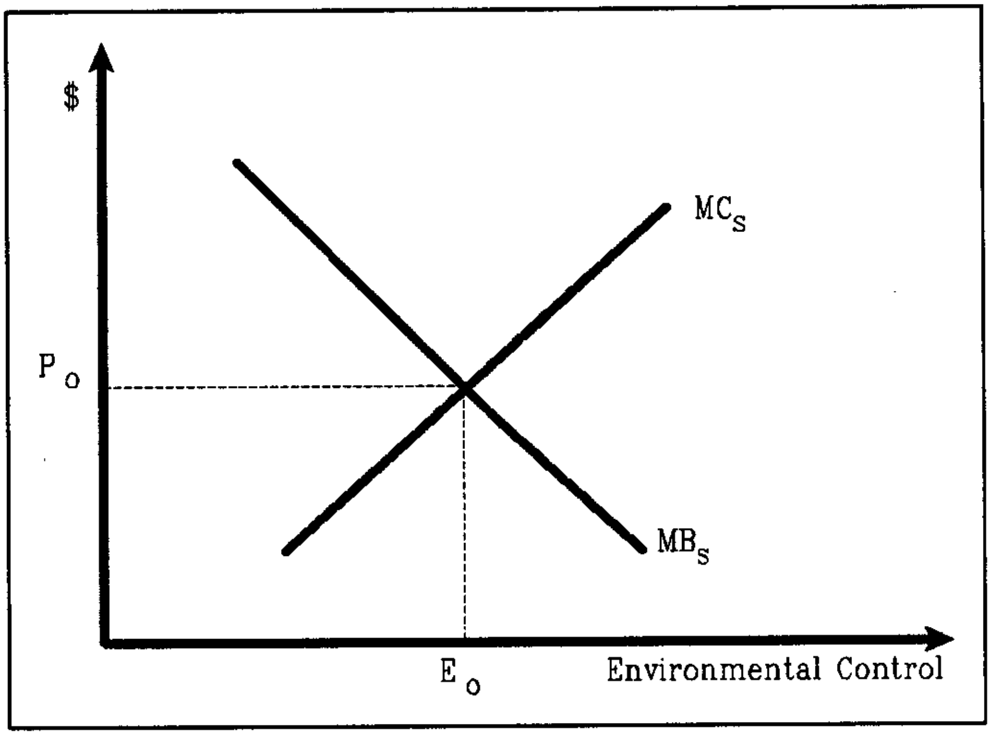
held responsible is associated with producing Q. The cost of this pollutant to society is represented by MC_e. Adding this to MC_p derives the marginal cost curve from commodity Q for all of society (MC_s). Supply curves interacting with demand curves (D) in perfectly competitive markets determine the quantity of Q produced and its price.

FIGURE 1:
Externalities and Market Failure



In Figure 1, Q_p and P_p represent the quantity produced and price of good Q, respectively, where producers only consider private costs. In contrast, the price P_s equates all of the marginal utility, or value, sacrificed to produce Q, and therefore represents a Pareto-efficient allocation of resources in the production of Q. The area ABC reflects the net welfare loss to society from producing Q_p; this also represents

FIGURE 2: Efficient Levels of Environmental Control



what society would be willing to pay to restrict production to Q_S.[4]

Achieving a Pareto improvement over the private decision-making environment represented in Figure 1 involves assessing and measuring external impacts (MC_e). The magnitude of these impacts also represents benefits from environmental control. In Figure 2, MB_S represents the marginal benefit to society of *increasing* levels of environmental control. Effectively, MB_S and MC_S (society's marginal cost for supplying this control) represent the demand and supply functions for environmental control; clearly gains from trade are possible since up to

[4]This loss simply follows from the fact that beyond Q_S society is giving up more welfare (MC_S) than it is receiving (D, which also represents the marginal benefit to society from producing Q).

E_0 amount of environmental control, society gains more than it loses. Therefore, engaging in any amount of environmental control up to E_0 would be a Pareto improvement over <u>no</u> environmental control, because theoretically some individuals can be made better off without anyone being made worse off.[5] The greatest possible Pareto improvement would be at E_0 in Figure 2 where the marginal gains are matched by the marginal cost of further control.

Identifying opportunities for a Pareto improvement becomes more difficult when externalities affect future utility or production functions. Where production today may lower tomorrow's productive capacity -- for example, if soil is eroded -- farmers theoretically take this into consideration so that they still maximize the present value of profit flows (Blyth and McCallum 1987). If this is indeed the case, under conditions of perfect competition economists consider resource use to be socially efficient.

For three reasons, however, farmers may undervalue the productivity of and therefore overutilize their natural resources so that undesirable costs accrue to future individuals. First, the productivity of natural resources may be undervalued where there is an incomplete distribution of information to decision makers. Farmers may lack information on the effect of their land-use decisions on the future productivity of their resources. For example, they may not know the exact relationship between erosion today and yields tomorrow. This may lead to "uninformed" decisions which overexploits soil resources,

[5]To be exact, this achieves a *potential Pareto improvement*. An *actual Pareto improvement* requires that the necessary compensation to losers occur. This paper addresses the former situation.

leading to unanticipated and avoidable future losses of productivity. Had more complete information been available, farmers might have altered their land-use decisions by investing in or conserving soil resources. Problems of uninformed decision-making are most likely to persist where weak institutional relationships between producers and government research and extension services exist.

A second reason why farmers' decisions may reflect an undervaluation of natural resource productivity is that price incentives rarely include the cost to society of natural resource depletion. Project and policy analysis often omit valuing these costs because the long-term physical changes and economic costs of this depletion are difficult to assess. Development and policy agendas most commonly seek short-term directly consumable outputs at the expense of longer-term capacity and less directly consumed goods. Lower income countries very commonly subsidize the irrigation, fertilizer, and pesticide costs of farmers to encourage their usage (Repetto 1985, Repetto 1986). Such incentives make diversified farming strategies traditionally relied on to conserve yields -- such as multi-cropping, relay planting, fallow farming, the use of manure, agroforestry, and mulching -- more expensive and hence less prevalent.

Finally, farmers' private time preferences may differ from that of society (Blyth and McCallum 1987). There is a growing body of literature which advocates either use of a social discount rate of zero for comparing intergenerational production and utility,[6] or more

[6]A discount rate (private or social) is the rate at which future earnings are deflated to arrive at their current value, and therefore represents the tradeoff between current and future consumption. A

specifically for sustaining a constant stock of natural capital such as water and soil quality. Proponents of the first position hold that future consumption is worth the same as present consumption. Where discount rates are higher than zero, resource allocation over time can encourage long-term environmental damage and declining future consumption, and in this sense may be socially undesirable (Pezzey 1988, Tietenberg 1984). Supporters of the second view hold that one generation should not pass on to the next a less productive stock of renewable natural capital because resources such as forests, soils, and water are not easily substituted for by other forms of capital. Such resources hence should neither be discounted nor permitted to degrade. It is less clear how such a constant-stock policy might be implemented, although, perhaps as a start, the World Bank's Environment Department has recently begun examining methods of incorporating natural resource depreciation into measures of national income (Peskin 1989, Magrath and Arens 1989).

To illustrate results from undervaluing the productive capacity of natural resources, Figure 3 presents a hypothetical production possibility frontier (PPF) with environmental quality (EQ) on the vertical axis, and production or net returns (P) on the horizontal. A PPF is commonly used to illustrate the combinations of outputs which an economy can produce from given resources. The economy in Figure 3 can produce any combination of the two "commodities" EQ and P, as long as it remains within the physical potential, or frontier F, of the economy.

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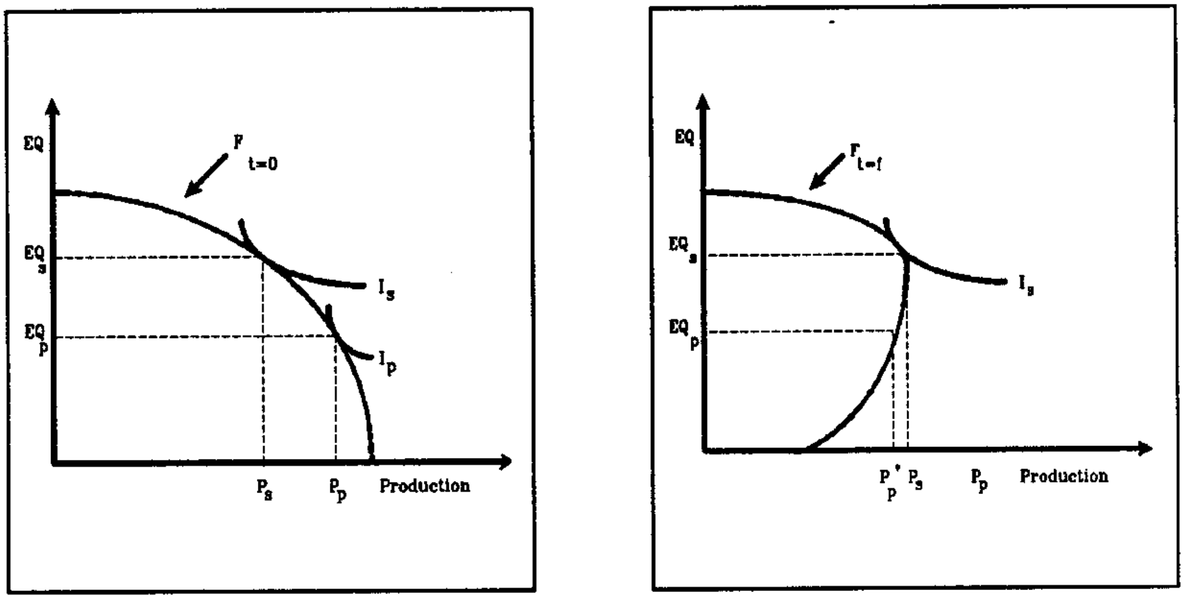
social discount rate of zero means that society is indifferent between gaining $1 today, $1 tomorrow, and $1 in one hundred years.

FIGURE 3: Production Possibility Frontiers
And Tradeoffs Between Environmental Quality and Production

3A: Time Zero   3B: Time Future



Ideally, an economy produces at the frontier since this represents the most it can efficiently produce of both goods given its resources. In the case of salinity, increasing the level of salt control (or water purity) requires resources and therefore reduces yields or profits (P). The rationale for "valuing" environmental quality -- i.e., conserving resources -- is that it ensures future welfare. Its exact value to a decision maker will thus depend on the weight given by the decision maker to future welfare reflected in the discount rate. If this person has a high discount rate, salinity control (future welfare) becomes relatively unimportant vis-à-vis production (immediate welfare). The private indifference curve may look like I_p in Figure 3A, whereby the economy produces P_p and engages in environmental control EQ_p.[7] If on the other hand, the *social* discount rate is zero, resource quality becomes more valuable since $1 of food tomorrow is worth $1 today. The indifference curve I_s in Figure 3A reflects a lower discount rate as production is traded for greater amounts of resource quality. Compared with the private scenario (I_p), this indifference curve results in society producing less (P_s) because it places higher value on resource quality (engaging in EQ_s).

For various reasons, production possibility frontiers change over time. When resources degrade in quality, an economy may be less capable of sustaining production. For those economies that have adopted higher levels of quality control, this change may be minimal. As shown in Figure 3B, choosing a lower level of control at time zero may compromise future production. In this figure, with a new PPF $F_{t=f}$, engaging in EQ_p environmental control at time zero results in P^*_p produced at time future which is below both P_s and P_p of Figure 3A. Under the alternative scenario of EQ_s, the economy achieves constant total consumption through time (P_s).[8] A Pareto improvement in the first scenario theoretically might be possible if individuals in "time future" with the capability of producing P^*_p could increase this capacity by

[7]An indifference curve represents all the different combinations of two goods which are equally valuable to an individual or group. It thus measures the welfare tradeoff between goods, and is usually assumed to be concave. Ideally levels of production are determined where indifference curves are tangent to the economy's PPF because if this is not the case, a Pareto improvement in resource allocation is possible.

[8]Although this social strategy is attractive for its intergenerational equity, it does not assure constant per capita consumption.

paying for environmental quality at "time zero".

2. Estimating and Valuing Externalities

Techniques for measuring welfare losses from externalities attempt to quantify welfare for one group resulting from the activities of another. These techniques fall into two categories (Upstill and Yapp 1987). One approach relies on market data to develop monetary values for nonmarket goods such as human health. If labor productivity is damaged or lost, theoretically there is a market-based proxy such as medical expenditures and earnings foregone from which to derive nonmarketed value, in this case the value of unimpaired health (Lave and Seskin 1977).[9]

The weakness of market approaches is that researchers frequently disagree over which market indicator to use, and whether it provides an adequate approximation of individual value. Market valuation of human health in particular raises wide debate over its application (Lave and Seskin 1977, Randall 1981). There appears to be wider acceptance for valuing non-health losses of utility. For example, the social costs associated with river pollution might be estimated by commercial fishing losses, losses in recreation or tourist revenue, and/or decreases in land or real estate values.[10]

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[9]For an example of market valuation of human health, see Richard Thaler and Sherwin Rosen (1975). "The Value of Saving a Life: Evidence from the Labor Market", in Household Production and Consumption, Nestor E. Terleckyj (ed). New York: National Bureau of Economic Research.

[10]In fact, the U.S. Department of the Interior has developed a computer model to calculate the economic costs of environmental damage from toxic spills. Animals such as geese and seals have been assigned values based on their market worth ($35.74 and $15.00 respectively) (Lancaster 1989).

The premise that market values reflect individual welfare is a restrictive one. Theoretically, it is true only in competitive economies (Kneese 1959); as mentioned, the economies of lower income countries are frequently distorted. Many critics also charge that market values invariably shortchange the true social value of environmental goods and services in that they fail to capture their long-term benefits and "existence values" (Lancaster 1989).[11]

The second category of techniques for valuing externalities consists of nonmarket data, generally derived from survey research. These surveys draw upon economic theory and methods to infer consumer preference for goods and services not traded in normal markets. They generally follow one of two approaches; (a) establishing individuals' "willingness-to-pay" to avoid some change such as increased water contamination; or (b) measuring their "willingness-to-sell" some qualitative state, determined by the amount of compensation they demand by a society before becoming indifferent to changes in this state.

The valuation of externalities becomes more difficult when the external effects of a current resource allocation decision accrues farther in the future. Measuring and valuing the total benefits (and costs) which accumulate over time, however, involves three distinct difficulties.

For one, exact relationships among activities and outcomes are rarely known, and become even harder to predict over time. For example there is wide controversy over the effect of deforestation on global

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[11]"Existence value" represents the utility people derive from simply knowing something exists (Bishop 1978).

warming. Similarly, the long-term effects of nitrate ingestion remain largely unknown. Second, it may be difficult or even impossible to identify the social group affected by overexploitation of natural resources. Contaminated groundwater and eroded topsoil may take centuries to regenerate, making it difficult to predict who suffers and how much during this time period. In addition, numerous technological, economic, and demographic changes could occur which might offset or magnify losses.

Finally, valuing future losses requires making some judgment about the tradeoff between future and present consumption. This raises two issues. First is the criteria used to establish this tradeoff. In its simplest form, economic theory maintains that peoples' aggregated time preferences along with an economy's investment opportunities establish the equilibrium rate of interest, an overall measure of this tradeoff. Theory also holds that this rate (called the private discount rate) leads to an economically efficient allocation of resources over time because it represents the opportunity cost of postponing benefits from public investment (Baumol 1968). If this rate is miscalculated, a serious misallocation of resources can result (Baumol 1968).

The argument for using private discount rates to approximate social discount rates is similar to the argument for using prices as a surrogate for social value. Private money markets may overstate the public's true time preference, however, leading to overuse and underinvestment in maintaining future productive capacity. Individual time preferences reflect private risks, inheritance taxes, uncertainties, and tax incentives which are not socially relevant.

"Life is short" from a private perspective, but it is less so from a social one. Such "distortions" in private resource allocation decisions cause individuals to heavily discount their future utility (Solow 1974).

A second issue raised in establishing the tradeoff between present and future consumption is whether or not discounting future utility is even legitimate. Many argue that public investment should seek goals other than economic efficiency. On moral grounds, it is indefensible to give inadequate regard to descendants: "the choice of a social discount rate is, in effect, a policy decision about...intergenerational distribution" (Solow 1974, p. 11). Some argue that high discount rates result in far too little invested in the future (Baumol 1968). Society must seek some "fairness of efficiency" since for any given discount rate, a different allocation of resources results, producing a different intergenerational distribution of utility (Ferejohn and Page 1978, p. 269).

Because measuring and valuing externalities poses so many difficulties, analysts are now incorporating resource constraints as a criterion in project analysis. Within this context, planners speak of "sustainable development", maintaining "genetic diversity", and avoiding "irreversible damage". Economists frequently propose abiding by "safe minimum standards" to avoid large unintended losses (Bishop 1978, Goodland and Ledec 1987). Sfeir-Younis similarly proposes that policies concerning the conservation of natural resources should fall within the objectives of "income maintenance" (1986, p. 77), rather than in the domain of income, employment, or foreign exchange objectives.

B. Policy Issues

On economic grounds, the public response to externalities is often to reduce the costs they impose on others as long as this can be achieved at a lesser cost -- or until a Pareto-relevant externality no longer exists. A variety of public policy measures can be used to achieve these reductions. The choice of which to use usually depends on various analyses of total social costs and benefits.

Environmental policy alternatives can be broadly categorized as either regulatory or involving economic incentives (Dorfman and Dorfman 1977). Regulatory policy consists of prohibitions, limitations, standards, and protective measures that are enforced by legal sanctions which specify, prohibit, or restrict the acts of polluters. For example, the government might stipulate that all producers of commodity Q (Figure 1, page 10) engage in environmental control at level E_0 (Figure 2, page 11).

Economic incentive policies consist of taxes, charges, subsidies, and marketable rights. These policies are designed to encourage or discourage certain uses of resources; individuals are generally free to respond to them as they prefer (Dorfman and Dorfman 1977). Taxes and charges penalize certain economic behavior, thereby inducing producers to alter this behavior. For example, the government may tax producers who contaminate water sources. Presumably a tax of P_0 on producers engaged in the polluting activity represented in Figure 2 would induce producers to practice E_0 amount of environmental control. Subsidies, which either defray the private costs of environmental control or reward pollution reductions below a designated level, are generally assumed to induce results similar to those of taxes (Alt and Miranowski 1979). Marketable rights allow for waste disposal after acquiring a purchased permit.

Some policies combine standards and incentives by achieving a standard through the use of economic rather than legal incentives. For example, the U.S. Department of Agriculture offers commodity program benefits to farmers who reduce the level of erosion to some target level (Barbarika and Dikes 1988).

Most economists maintain that economic incentives more efficiently lead to socially preferable economic behavior than do regulations and standards (Jacobs and Casler 1979). The latter become "mired in technical, legal, and administrative overburden" (Ruttan 1971, p. 715). Many advocate a policy of "the polluter pays", and compensating pollutees for their loss through transfer payments. Polluters have an economic incentive to limit their polluting activities, consequently the need for constant and effective public control diminishes.

Dissenters of the above view argue that an efficient incentive policy depends on accurately assessing the point at which the marginal social costs of control equal the marginal social benefit (P_0 in Figure 2). In practice this is difficult. Moreover, these policies require the need for a transfer mechanism to compensate the victims of pollution, while standards do not.

Judging the performance of incentives versus standards may require criteria other than strict efficiency, however. The use of incentives implies that while damages may not be avoided, in theory they will be compensated. If loss estimates are based on willingness-to-pay or market valuations of production lost, they may undervalue actual welfare losses if "losers" face greater risk or uncertainty. Standards on the other hand, avoid loss, and as such may result in less welfare loss.

III. EXTERNALITIES FROM WATER MANAGEMENT: EXAMPLES OF NITRATE LOSSES AND SALINIZATION IN TROPICAL ARID- AND SEMI-ARID AGRICULTURE

The previous section reviewed how externalities are treated in analyses of economic efficiency. This section reviews literature on the physical impact of nitrate losses and on salinization in tropical arid- and semi-arid agricultural systems. This review is intended to establish empirically the magnitude of these externalities. It also highlights key processes of agricultural systems which influence these externalities.

External effects from water management may result within a relatively short time. Water that is not recycled via evapotranspiration percolates either vertically and laterally through the soil profile or is lost via runoff. The amount of water typically applied in irrigation systems exceeds the rate of evapotranspiration by a wide margin (Kneese 1959). This excess water may enter an underground aquifer, or may reappear downstream as seepage into a river or natural sink (Pillsbury 1981). The impact of this lost water on adjacent ecosystems depends on the water's quality, quantity, and final destination. Water quality is determined by the amount and form of solutes and particulate matter carried with it. Dissolved lead, nitrate, and chloride particles can immediately contaminate drinking water supplies, whereas dissolved pesticide intermediaries and phosphates can reduce the economic productivity of downstream aquatic and coastal marine systems. Part A of this section reviews research on the process and extent to which agriculturally derived nitrates have entered water bodies in the arid- and semi-arid tropics.

Water management can also result in externalities which develop over a longer time frame. Undissolved particulate matter such as clay, silt, and organic substances can build up over time, clogging waterways and interfering with biological processes. Long-term ecological changes fostering human disease and supporting new plant pathogens are also a major concern with water management. Part B focuses on the process of salinization from water management, since it poses one of the main problems of irrigated agriculture in arid- and semi-arid regions (El-Swaify et al. 1983, Moore 1972).

A. Externalities of Water Management: Nitrate Losses

Nitrogen is one of the most important plant nutrients in agricultural production (Nwoboshi 1980, Mughogho et al. 1985), and significant nitrogen losses to adjacent ecosystems occur in many agricultural systems (Craswell and Vlek 1979). These losses commonly occur in the form of nitrate leaching (Lal 1980, Wetselaar 1962).

Nitrogen enters agricultural soil from organic matter decomposition, inorganic fertilizer application, precipitation, and biological nitrogen fixation. There are three principal ways in which nitrogen can be lost from the system prior to plant uptake: volatilization (the conversion of ammonium to ammonia), denitrification (the conversion of nitrate to dinitrogen gas), and nitrate runoff and leaching.

Nitrate (NO_3) loss from leaching and runoff is of particular concern for two reasons. First, nitrate leaching can be an important factor influencing the nitrogen efficiency of tropical soils, and hence acts as a yield determinant (Holding 1982, Wild 1972, Bartholomew 1977). Second, nitrate contamination of groundwater sources may present a public health hazard (Lal 1980, Craswell and Vlek 1979, Nwoboshi 1980). Water containing more than 10 mg-N/L of nitrates may cause methemoglobinemia in infants and livestock, a disease which inhibits the oxygen carrying capacity of the blood and which can lead to oxygen deprivation and even asphyxiation. Recent research also suggests a link between nitrate ingestion, the formation of nitrosamines in the intestines, gastric and stomach cancers, and miscarriages (Walker 1988, OECD 1986). Infants, the aged, and animals ingesting high amounts of nitrate appear especially susceptible to these disorders (Hartman 1982, Sampson 1986). Studies on the long-term affect of nitrate ingestion have not appeared in the literature, thus the severity of this risk remains uncertain (Hartman 1982).

In the United States, the legal limit for nitrate concentration in drinking water is 10 mg-N/L,[12] but many domestic water sources in rural areas do not meet these standards (The Conservation Foundation 1987, Singh and Sekhon 1979, Walker 1988).[13] Developed countries are experiencing dramatic upward trends in nitrate contamination of groundwater (IIED and WRI 1988, OECD 1986). However, little is known

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[12]Nitrate concentrations can be expressed as milligrams of nitrate ion per liter of water ($mg\text{-}NO_3/L$ or ppm) or, as here, milligrams of nitrogen as nitrate per litre (mg-N/L). Since 100 $mg\text{-}NO_3/L$ is equivalent to 22.6 mg-N/L, the U.S. standard can also be expressed as 44 ppm nitrate.

[13]Hartman (1982) reports that in 1969, three percent of all public water supplies serving one percent of the U.S. population exceeded the U.S. standard for nitrate concentration.

about groundwater quality in the tropics (IIED and WRI 1987, Singh and Sekhon 1979, WHO 1978). Nevertheless, anecdotal evidence as well as indications from affluent countries suggest that nitrate losses from agricultural production in the tropics may be significant.

1. The Role of Water Management in Nitrate Losses

Valdivia (1982), researching nitrogen cycling in irrigated sugarcane in non-saline semi-arid Peruvian soils, concluded that nitrate leaching losses could be controlled through more judicious water management practices. He estimated that under given water-management practices, about 33 kg NO_3-N/ha/18-months is leached with the application of the economically recommended 300 kg-N/ha of urea fertilizer (Valdivia 1982).

In Santiago, Chile, other researchers investigated the common practice of using untreated sewage effluent and industrial waste-water as fertilized irrigation water on lettuce, spinach, beets, and celery crops (Schalscha and Vergara 1982). In this semi-arid region, the capital city's sewage effluent provides the sole source of irrigation water and fertilizers. The authors estimated that sewage provided approximately 780 kg N/ha/year to the fields; and that of this, only about 21 to 37 percent was assimilated by the vegetable crops. They concluded that most of the remaining nitrogen was drained from the system in the form of nitrates. The nitrate level of subsurface drinking water wells in the study area was reported to contain between 11.5 to 16.5 mg/L of NO_3-N,[14] whereas a nearby subsurface stream channel

[14]The Conservation Foundation (1987) reports that values greater than 3 mg NO_3-N/L may indicate other than natural sources.

not receiving the effluent had a measured NO_3-N concentration of .9 mg/L.

Similarly, researchers in Piracicaba, Sao Paulo, Brazil traced the movement of nitrogen fertilizer applied to bean fields over three successive cropping periods in a rainfed system supplemented with irrigation (Libardi et al. 1982). After applying urea labelled with ^{15}N at a rate of 100 kg-N/hectare, researchers traced its movement over the next three cropping periods. Results indicated that over this period (621 days), about 20 to 25 percent of the first year's nitrogen was leached as nitrate, and only about 35 percent of the initial nitrogen still remained in the soil profile. Of the nitrogen initially applied, at most 40 to 45 percent had been converted into yield after three cropping periods.

Singh and Balasubramanian (1980) studied nitrogen cycling in the savannah region of Nigeria and found that traditional cropping practices relying on rainfall, limited organic fertilizers, and fallow periods resulted in minimal NO_3 losses. The research findings suggested that leaching may occur in years of average or above average rainfall, and is more likely if inorganic fertilizers are applied. Singh and Balasubramanian estimated that, while depending on the soil structure, perhaps 25 percent of applied nitrogen will be leached.

Ganry et al. (1978) measured nitrogen uptake and yields of millet crops under varying input combinations in a non-irrigated semi-arid region of Senegal. Their study examined nine combinations of incorporated straw and nitrogen fertilizers on both millet-cropped and bare plots. An average of ten percent of applied urea was leached as nitrate in cultivated soil, with this loss positively correlated with fertilization rates.[15]

Over a two-year period, Wild (1972) traced soil mineral nitrogen in the top 120 cm of unfertilized bare-fallowed soil at Samaru, Nigeria to determine the relationship between nitrate leaching, rainfall and drainage. He found only gradual nitrate leaching under unfertilized and normal rainfall conditions of about 1000 mm/yr. Wild attributed these low rates of leaching to soil structure (the existence of large macropores), a positively charged B horizon (nitrate is an anion), and the rapid infiltration and percolation of high intensity rainfall. Other studies have suggested that with broadcast application of nitrogen the nitrates move downward in the soil profile two to eight times faster per unit of rain than do the organically derived nitrates because they are more likely to be washed into drainage channels (Wild 1972).

Jones (1975) investigated the effect of planting dates on nitrate movement from both inorganic and soil sources in rainfed agricultural systems in northern Nigeria. He found that the leaching rate of soil nitrogen was very slow, while that of applied nitrogen was about twice as rapid: about 0.5 cm/cm-rain. Unlike soil nitrogen, applied nitrogen is more easily leached with later planting dates, probably due to the heavier rains characteristic at this time. Lal (1980) found the highest nitrate losses just after fertilizer application and when the soil is saturated. Santana and Cabala-Rosand (1982) found monthly nitrate losses to be proportional to rainfall.

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[15]The authors concluded that higher straw incorporation rates also led to less nitrogen efficiency, but this is probably due to denitrification and immobilization and not to nitrate leaching.

Wetselaar (1962) studied nitrate accumulation in cultivated subsoils, and concluded that nitrate movement can be traced to soil structure and texture, rainfall, and root growth and patterns. In sandy soils, he estimated that nitrates move about twice as fast per unit of rainfall then in clayier soils. Nitrate losses can be minimized through practices which immobilize nitrogen at the beginning of the wet season, such as through straw additions.

Bernhard-Reversat and Poupon (1980) studied the nitrogen cycle of 12 *Acacia senegal* trees in the Sahelian region of Africa for several years. Their research concluded that this tree-herb-soil system resulted in little if any nitrate losses. Although studies of the soil profile indicate a downward movement of nitrates, during a normal rainy season of 300 mm/year losses probably are limited by minimal water movement. During periods of heavy rainfall, especially where the herb cover is sparse, there may be sufficient water movement to leach nitrates.

Much of this research indicates the important role of water management in nitrate movement. Where excessive water is applied, as it typically is in irrigated systems, the potential for nitrate loss is high. The movement of water through the soil profile is partly determined by soil structure and texture. The existence of macropores can limit the contact between soil nitrates and drainage water, thereby reducing leaching potential. This is especially true of organically derived nitrates, produced through nitrification within aggregates (Wild 1972). However, in general, as nitrogen levels increase, so do leaching losses. This seems particularly true when inorganic fertilizers are used (Koyama and App 1979). Where no inorganic fertilizers are applied, nitrate losses seem minimal except under high rainfall conditions. Careful water and fertilization practices, such as amount and timing which approximate biological needs, especially under sandy soil conditions, will result in less nitrate loss.

2. The Role of Cropping Strategies in Nitrate Losses

The amount of nitrogen leached to below the root zone as nitrate is largely determined by the amount and distribution of rain and irrigation water, but it is also affected by the type and duration of vegetation. By absorbing nutrients and water, developed and diversified rooting systems minimize nitrate losses; losses appear least likely under perennial cropping systems because well-developed roots are in place whenever water is applied. Losses are less likely to occur where deep-rooted fallow species are established because nutrients that would otherwise be lost are taken up from the subsoil. Losses are most likely to occur under continuous cultivation, especially when cultivating shallow-rooted crops such as sorghum (Wetselaar 1962).

Ganry _et al_. (1978) concluded in their study on nitrate movement that greater root activity limits the concentration of nitrates in the soil and reduces the potential for leaching. Under all conditions, nitrate losses were highest in non-planted soils. Lal's review (1980) of research on nutrient losses from agricultural systems suggests that the nitrate leaching rate is from 6 to over 10 times higher in bare than in vegetated soil.

Wetselaar (1962) concluded that nitrate losses can be minimized by altering growth patterns (e.g., planting dates) and crop density. Wild

(1972) estimated that all soil nitrates could be retrieved if crop roots extended to 15 cm before 300 mm of rain has fallen. If the roots are 45-90 cm long by the mid- to late-rainy season the crop is able to exploit the high nitrate concentrations which diffuse and percolate to these depths. Thus, leaching rates will depend on the rooting pattern of the crops (Wild 1972, Jones 1975).

Substantial research documents the rapid accumulation of nitrogen in fallow vegetation. These accumulations occur in part from the deep rooting pattern of fallow species. Bartholomew (1977) investigated the movement of soil nitrates following the fallow period, and its influence on yield declines during the cropping period. He estimates that up to 50-400 kg of nitrate-N may be present in a 2 meter depth of soil after a two-year period of cultivation; much of which would be leached if the land were continuously cultivated. Bartholomew concluded that the high rate of yield declines were attributed in part to the downward movement of nitrates in the soil profile, and that rooting patterns and depth are important in nitrate uptake efficiency.

Two separate studies on the nitrogen cycle of cacao and coffee trees in Venezuela concluded that under traditional unfertilized conditions, there is minimal nitrate leaching from these plantations (Aranguren et al. 1982a, Aranguren et al. 1982b). However, another study of a cacao plantation in Bahia, Brazil, found a higher nitrate leaching rate in unfertilized than in fertilized stands (Santana and Cabala-Rosand 1982).[16] In both cases leaching losses were slight. The

[16] Both stands were, however, shaded by species of the leguminous tree genus *Erythrina*, which provides significant quantities of organic nitrogen.

authors concluded that the lower rates associated with fertilized stands may possibly be attributed to the more efficient rooting and rootlet system developed under fertilized conditions.

A study of an unfertilized mature teak plantation in Ibadan, Nigeria described the monthly nitrogen cycling pattern in this plantation (Nwoboshi 1980). Although he did not measure nitrate losses in this system, Nwoboshi inferred that nitrogen deficits are likely to occur, and that losses were virtually zero. He concluded that careful use of nitrogen fertilizers may be appropriate.

All else the same, nitrate leaching will be minimized in cropping systems where there are greater demands on the soil for nitrogen both over time and throughout the soil profile. As Ganry et al. (1978) conclude, losses will be higher at the beginning of the crop cycle when roots are small. In general, nitrogen losses will always be determined by the stage of the plant growth, and the associated fertilization timing and rate (Lal 1980).

B. Externalities of Water Management: Salinization

Irrigation water always contains variable quantities of dissolved salts, mainly products of weathered parent material. Ionic species most common in irrigation water include chloride, sulfate, calcium, sodium, and magnesium (Bresler 1981).

The salt balance of soil water is determined by the hydrological cycle. Without irrigation, a hydrological balance exists between rainfall and streamflow, groundwater levels, and evapotranspiration that maintains salt concentrations at levels that change only over geologic time intervals. Irrigation establishes a new salt balance. The residual portion of irrigation water not evaporated or transpired (broadly estimated by Kneese (1959) as about one-third of total water) contains most of the water's original dissolved salts. If irrigation water is very low in salt content, and is applied in sufficiently high quantities, it is possible that irrigation will lead to a lowering of soil salinity. However, water in arid regions is commonly enriched with salts due to high rates of evapotranspiration. Hence, irrigation in dry regions usually leads to salts accumulating in the soil profile. Moreover, if irrigation water is repeatedly recycled, water salinity can become very high thereby hastening the process of soil salinization.

Both the amount and kinds of salts present in water determine the suitability of the water for irrigation. Salinity problems occur when the total quantity of soil salts is sufficiently high to accumulate in the root zone; this occurrence is most frequently attributed to irrigation practices which relocate and concentrate salts.[17] Yield declines usually begin occurring at soil salt concentrations of 600 parts per million (ppm),[18] and salinity is considered a severe problem

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[17]The saltiness of water is related to the proportion that is evaporated or transpired into water vapor. While natural occurrences could lead to salt accumulation, as in the case of the U.S.'s Red River Basin (Brown 1984), it is generally associated with human activity, in particular irrigation and storage reservoir schemes (Pillsbury 1981, Hotes and Pearson 1977).

[18]Salinity is commonly expressed as parts per million (ppm) of dissolved solids. This paper consistently reports this measure, although the studies cited use a variety of measures. In practice, salinity is often measured as the electrical conductivity of the soil water in millimhos (mmhos/cm) or micromhos (umhos/cm) per centimeter. Approximately 640 ppm correspond to 1 mmhos/cm (Carruthers and Clark 1983), and one mmhos/cm is also approximately equal to one decisiemens per meter (dS/m) (Jensen 1984). These approximations, however, are

at levels of 2-3,000 ppm or more (Westcot 1980, El-Swaify et al. 1983).[19]

As mentioned, salt content can be assessed either in the irrigation water or in extracted soil solution. A high soil water concentration implies that water is less available to plants because of salt's high osmotic pressure. Saline soils therefore often produce uneven or irregular crop growth and reduced yields, and can support only a restricted choice of crops. In extreme cases plants become stunted or die, and land is lost to agricultural production.

Past as well as recent history provide numerous examples of soils lost to production after water diversion schemes have left soils toxic to plant growth (Reisner 1986, Pillsbury 1981, Szabolcs 1976). Perhaps most remarkable is the demise of the Mesopotamian empire four thousand years ago; salinity damage is commonly cited as a major cause of the empire's decline (Gelburd 1985, Dougrameji et al. 1980). Today, UN agencies and researchers estimate that over 50 percent of the world's irrigated land suffers from salt-related problems (Szabolcs 1976, El-Swaify et al. 1983),[20] and researchers frequently question the permanence of irrigated agriculture (Allison 1964, Kelley 1964, Moore 1972).

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highly dependant on the species present.

[19]Rain water usually contains between 10-50 ppm of salts (Kovda 1983), and ocean water averages about 35,000 (Pillsbury 1981). The EPA's acceptable limit of total dissolved solids in drinking water is 500 ppm (Brown 1984).

[20]Yaron (1981) provides a more conservative estimate of one-third of all irrigated land.

The magnitude and pervasiveness of soil salinization from irrigation is easily illustrated. Fifty percent of all soil in the Euphrates Valley in Syria reportedly is affected by high salt levels (El Gabaly 1977). In Iraq's Southern Mesopotamian plain, sixty percent of the agricultural land is affected to some degree by salinity (Johl 1980). After one decade of irrigating, eighteen percent of irrigated land in the Jordan Valley is affected by salts (El Gabaly 1977). Two million acres of Egyptian irrigated land beyond the Nile flood plain have also developed serious and increasing salinity problems; this area represents one-third of the nation's cultivated land (Zikri and El-Sawaby 1980a). Gelburd (1985) estimates that salinity affects over half of California's 8.6 million acres of irrigated land. Repetto (1988) assesses 30 million hectares of land in India and Pakistan as seriously damaged by salinization. Salinity problems in agricultural lands have also been reported in Australia (Quiggin 1988); the Soviet Union (Keller 1988); Tunisia, Iran, and China (Kovda 1983); Somalia and the Sudan (Gaddas 1977); Argentina (Musto 1977); and Senegal (Beye 1977). Appendix B provides a more complete listing of the worldwide extent of salinity problems.

In spite of vast concern over and documentation of soil salinization problems, few countries, particularly those in the tropics, have collected quantitative data on the process of salt buildup and its concomitant economic importance. The balance of this section reviews literature on three different indicators of the extent to which irrigation can lead to important productivity changes in soil and water resources.

1. The Effect of Irrigation on Soil Salinity and Crop Productivity

The accumulation of salts in the soil water is directly correlated with the amount of salt in the irrigation water and the amount of water applied. In Israel, after only one irrigation season, the salt content of soil water increased from 128 to 1600 ppm with irrigation water containing 448 to 2,560 ppm (FAO-UNESCO 1973). Use of water with 290 ppm salts in Texas for 5 to 7 years increased the salinity content of soil water from 415 to 475 ppm (FAO-UNESCO 1973).

Theoretically, one can calculate the amount of salt which can accumulate in the soil. Massoud (1977) calculates that applications of water containing 200 ppm salts at a normal rate of 10,000 m^3/ha/year might result in an increased salt load of 2 tons/ha/year. Irrigation water containing 706 ppm salts contribute .96 tons of salt per acre-foot/year of water to the soil (Moore _et al_. 1974). Whether or not these concentrations are eventually leached from the root zone depends on water management practices.

Numerous researchers have attempted to establish the critical salinity values corresponding to expected plant yield decreases (Ayers and Westcot 1985, Massoud 1977, Abrol _et al_. 1988). Critical values are often disputed,[21] and are influenced by a number of factors including climate, plant nutrition, soil properties, kinds of salts, and water management practices. However, some generalizations emerge from these

[21]Many researchers maintain that the issue surrounding salinity is not how _good_ the water is, but rather how to achieve successful results with it (Allison 1964, Hardan 1977). From an economic standpoint, this difference is less important because in either case, "with water salinity exceeding a certain level, income losses are unavoidable" (Yaron and Ratner 1984, p. 441).

TABLE I

EXPERIMENTS AT THE U.S. SALINITY LABORATORY
RIVERSIDE, CALIFORNIA.
Soil Salinity in Parts Per Million[1] at 25 Degrees Centigrade
At Which Yield Decreased By:

CROP	10%	25%	50%
Barley, grain	7,680	10,240	11,520
Cotton	6,400	7,680	10,240
Wheat	4,480	6,400	8,960
Sorghum	3,840	5,760	7,680
Soybean	3,520	4,480	5,760
Sugarcane	1,920	3,200	5,440
Rice, paddy	3,200	3,840	5,120
Maize	3,200	3,840	4,480
Field Bean	960	1,280	2,240
Tomato	2,560	4,160	5,120
Potato	1,600	2,560	3,840
Onion	1,280	2,240	2,560

[1]Data are originally reported in mmhos/cm.

Source: Bernstein (1981).

studies (Schleiff 1980, Bernstein 1981). Scientists at the U.S. Salinity Laboratory in Riverside, California, conducted controlled studies of the salt tolerances of field, forage and vegetable crops (Bernstein 1981). They established the salt concentrations which reduced yields to 10, 25, and 50 percent of the nonsaline control plots. Table 1 shows the wide variability between crops in their response to saline water.

Five years of field research in India on the effect of different levels of water salinity on wheat yields established that with favorable rains, irrigation water containing 5,120 ppm salts may be safe for sandy loam soils, and water containing 10,240 ppm may be safe for dune sand soils (Mondal and Sharma 1979). Above these levels, yields declined significantly. Table 2 below summarizes the results of varying water quality on both wheat yields and salt accumulation for each of the soils investigated.

Tables 1 and 2 demonstrate the broad range of salt sensitivities in different crops under different conditions. Modest increases in salinity often lead to yield decreases of 10 percent in most vegetable crops, whereas salt tolerant species such as cotton and barley maintain high yields over a wide range of salinity levels. A more complete listing of other research estimating the salinity tolerance of various crops can be found in Appendix C. A series of experiments by the United Arab Republic's Ministry of Agriculture found that crop yields could be increased by 50 to 250 percent in one year through reclamation projects in which draining soils to leach salts played an important role; officials estimate yield improvements from salt leaching averaged 20 percent (Khatib 1971). Leaching and reclamation experiments in central Iraq also resulted in significant yield improvements -- overall, wheat yields increased from .9 to 1.7 tons/ha; rice from 1.2 to 2.1 tons/ha; and seed cotton from .7 to 2 tons/ha (Khatib 1971).

Khatib (1971) reported that the value of all Pakistan's agricultural products in 1960 was 25 percent less than it would have been had salinity not been a factor. Furthermore, he estimated that current salinity problems reduced the value of agricultural production by one additional percent per annum. Researchers in Syria estimated the loss of cotton from salinity to be about 70,000 tons per year, with a reported value of $17 million (Raslan and Fardawi 1971). FAO-UNESCO

TABLE 2

The Effect of Water Salinity on Soil Salinity and Yields.
A Five-Year Experiment on
Dune Sand and Sandy Loam Soils in India.[1]

		Treatment in Parts Per Millimeter (PPM) of Water Salinity					
		384	1280	2560	5120	7680	10240
DUNE SAND	PPM Soil Salinity						
	Year 1	270	500	450	900	1200	1590
	Year 5	510	608	800	1025	1408	1700
	Average Yield[2]	424	416	438	435	431	302
SANDY LOAM	PPM Soil Salinity						
	Year 1	400	490	540	950	1360	1640
	Year 5	480	700	830	1440	1860	3232
	Average Yield[2]	402	425	436	432	268	131

[1]Data originally reported in mmhos/cm.
[2]Average over 5 years, calculated as grams/7857 cm^2.

Source: Mondal and Sharma (1979).

(1973) reported that barley yields in the Mesopotamian plains of Iraq are now one-half to one-fourth what they were in ancient times. Hardan (1977) found a 23 percent decrease in potato yields in Iraq after three years of irrigating with groundwater containing 4,000 ppm dissolved salts.

Through use of a linear programming model, Oyarzabal-Tamargo and Young (1978) projected the effect of salinity in irrigation water in the Mexicali Valley in Mexico on net income. The average marginal damage over the range of 700 to 2,000 ppm salt in the Colorado River was 348,000 pesos per ppm in 1975 prices. Moore et al. (1974) reported that projected increases in Colorado River salinity from 960 to 1280 ppm by the year 2000, will cause a 14 percent decrease in Imperial Valley farmers' net returns to land and water. A further salinity increase to 1920 ppm would result in a total 26 percent decrease in net returns. In contrast, if river salt concentrations were reduced to half their current loads (or to 480 ppm), the net income of Valley farmers would increase by $7.4 million per year. Gardner and Young (1985) estimated that each ppm reduction of salts from 800 to 1100 in the Upper Colorado River Basin leads to an average cost of $51,400 in lost revenue in the Lower Basin.

2. The Effect of Irrigation on Levels of Downstream Salinity

To preserve the productivity of irrigated soil, salts must be leached through applications of excess water. However, this means that the resulting saline leachate may accumulate in groundwater and streams (Gardner and Young 1988). Very commonly, river salt loading results from saline return flows and seepage of excess irrigation water from fields, ditches, and canals. In the Grand Valley region of Colorado where the Colorado River irrigates some 55,000 acres of crops, Gardner and Young (1988) report that over one-half million tons of salts are discharged into the river annually from irrigation and natural sources.

In the Yemen Arab Republic, the Wadi Al Haima near Taiz is diverted for agriculture, and much of the drainage returns to the stream and is reused downstream. While the stream's upper reach has a salt concentration of 320 ppm, within a relatively short distance of 25 km the salinity increases sixteen fold. Cropping patterns along this gradient reflect the salt tolerance of crops (Ayers and Westcot 1985). Numerous other rivers and streams used for irrigation reflect a similar pattern of degradation. The Pisco River in Peru reports an upstream level of 430 ppm salts, while downstream it is reported to be 3,712 ppm (Ayers and Westcot 1985). At the confluence of the Tigris and Euphrates in Iraq, the river's salinity value is reported as over 600 ppm, whereas upstream near Baghdad and Falluja the salinity ranges from 300 to 500 ppm (Dougrameji et al. 1980). Since the construction of the Aswan Dam and the introduction of modern irrigation in Egypt, the salt concentration of the Nile River has risen from about 175 to 200 ppm, although some researchers such as Zikri and El-Sawaby (1980b) doubt that the dam will create a major salinity problem in this river.

The Rio Grande River offers one of the most documented examples of the effect of recycled irrigation water on downstream salinity. Outside Albuquerque, between 1934 and 1953, dissolved solids in the river averaged 221 ppm. Downstream 725 kilometers on the Texas-Mexico border during this time period river water averaged 1691 ppm (Hotes and Pearson 1977).

The headwaters of the Colorado River contain trace dissolved salts, while Hotes and Pearson (1977) report levels of 900 ppm in the lower basins, and even higher levels in Mexico. A large irrigation project in Arizona just north of the Mexican border was established in 1961, and resulted in the discharge of highly saline water into the Colorado River. Between 1960 and 1962, river water on the Mexican side of the border increased in salinity from 850 ppm to 1500 ppm (Hotes and Pearson 1977).

Salinity is similarly low (100 ppm) in the upper section of the Murray River in Australia. This level rises to about 250 ppm in the river's middle sector, and attains 1200 ppm in its downstream reaches (Pels and Stannard 1977). Studies estimate that increased salts from both surface and groundwater inflows can increase the Murray River's salt load by 850,000 tons per year (Pels and Stannard 1977).

Saline water not only affects the productivity of water on agricultural land, but also adds costs for capital replacement, additional detergents, water softening, and the cost of bottled water (Gardner and Young 1985). Reisner (1986) reports that each additional ppm of salts in the Colorado River costs the citizens of Los Angeles county an extra $300,000 per year in salt-induced damage.

3. Effect of Irrigation on Groundwater Salinity and Water Table Levels

Irrigation can also lead to rising groundwater levels and increased groundwater salinity. Rising groundwater may inhibit normal root development by limiting aeration through capillary moisture movement, and by the upward movement of salts into the rhizosphere.[22] Therefore, when saline water tables rise, there is a general rise in the

[22]According to Shalhevet (1973), a water table of a meter or more is usually sufficient for normal root growth for annuals, while a somewhat deeper water table of 1.5 to 1.8 meters may be required for perennial crops.

TABLE 3

The Number of Hectares with Different Water Table Levels: A Comparison Between 1892 and 1950 in the Chaj Doab District of Pakistan.

WATER TABLE (in meters)	1892		1950	
0 to 1.5	0	(0)	94,450	(7)
1.5 to 3	6,500	(.5)	693,700	(52)
3 to 4.5	118,400	(9)	335,800	(25)
> 4.5	1,221,500	(91)	221,700	(16)
TOTAL	1,346,400	(100)	1,345,650	(100)

Numbers in parentheses reflect percentages.

Source: Worthington (1977).

root uptake of saline groundwater.

For example, an irrigation scheme in eastern Jordan has resulted in decreased groundwater quality. The salinity level in one well increased from 275 ppm in 1971 (before irrigation) to 1615 ppm by 1977. Other wells in the area showed the same trend after only a few years (Ayers and Westcot 1985).

Evidence from a half century of irrigating in Chaj Doab, Pakistan, presented in Table 3, reveals the dramatic shifts in the depth of water tables which can result from irrigation (Worthington 1977). During this time, the water table of over half of the district's 1.3 million hectares rose above three meters. Quiggin (1988) reports that the irrigated Australian Murray River Basin has also experienced rising water tables.

4. Other Effects on Agricultural Performance

In addition to the three broad areas of concern discussed above, there are other costs associated with increased salinity. FAO-UNESCO (1973) report that salt problems decrease humus reserves in the soil and reduce fertilizer efficiency. For example, a study in the USSR found that yields of raw cotton increased by 90-330 kg/ha with fertilizer applications on soils with .7 to 3.3 percent salt. After draining and leaching salts from this land, fertilizers increased yields by 930 to 1660 kg/ha in the first year, and positively influenced yields in succeeding years (FAO-UNESCO 1973). Salinity can also affect the quality of crops. Cotton fibre grown under saline conditions may become shorter and more brittle; the sugar content of sugar beets may be reduced; the size and flavor of fruit from fruit trees is often poorer; and grain kernels are stunted (FAO-UNESCO 1973).

IV. ECONOMIC AND POLICY ANALYSIS OF EXTERNALITIES IN AN IRRIGATION PROJECT

This section integrates the theories and concepts discussed in the previous two sections to illustrate relevant economic analyses as they relate to the development of irrigation projects. Through this illustration, this section also identifies the more important empirical, analytical, and policy issues in improving irrigation project design where externalities are anticipated. A presentation of methods for quantifying the associated external welfare losses is followed by a discussion of the choices and corresponding costs of avoiding or mitigating these losses. Part C then examines policy options based on economic analyses of the benefits and costs of environmental control, as well as further assessments of distributional impacts, market conditions, and institutional capacity.

A. Measuring and Quantifying Individual Losses from Externalities

Water is the primary transporter and relocater of both nitrate ions and mineral salts. Investigating an area's hydrological cycle -- both as it functions naturally and under irrigated conditions -- is critical in estimating the importance of individual losses from irrigation.

1. Nitrate Losses

If secondary use of irrigation water is likely to include human or animal consumption, the potential for nitrate buildup in drinking water sources may be an important factor to consider in irrigation development. Several variable factors influence the concentration of nitrates in groundwater. These are affected by water quantity and management; followed by nitrogen sources, quantities, and management; and finally by the cropping mixture. These factors largely determine the extent to which, other things equal, irrigation leads to nitrate contamination of drinking water.

If hydrological conditions coupled with farm management indicate a probability of some water contamination, irrigation design might be improved by considering the loss associated with this contamination. Section II identifies two categories of methods for estimating the magnitude of these losses. One category, which employs market data, depends first on predicting the physical importance of contamination, second on linking this to welfare losses, and third on expressing this loss in terms of marketable commodities. Using this method to assess the losses associated with nitrates accumulating in drinking water involves predicting the degree of contamination; estimating the number of persons and animals affected by this contamination; and valuing the extent to which this impairment affects economic productivity.

The data required for these analyses are inexact even in higher income countries (Hartman 1982),[23] and probably less reliable in lower

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[23]Hartman (1982, p. 211) outlines gaps in research and current knowledge concerning the effect of nitrate ingestion on human health; these gaps "hinder critical evaluation of the nitrate-nitrite problem....[although m]any of these gaps can be readily filled by further research." Taylor and Frohberg (1977, p. 33) state that "until...information [on the complexity of the nitrogen cycle] becomes available, it will not be possible to compute the social value of damages abated by nitrogen restrictions." Batie (1988, p. 5) adds that "[o]ur current ability to detect pesticides and nitrates in groundwater far exceeds our understanding of their significance."

income ones. Even if these linkages were better understood, in lower income countries the relationships between economic productivity and market labor rates are controversial and at best difficult to assess. Consequently, techniques relying on market data for estimations of individual losses are not very useful in the case of nitrate contamination.

The second category of methods for evaluating losses, nonmarket techniques, consists of the "willingness-to-pay" (WTP) and "willingness-to-sell" (WTS) criteria. Although many economists argue that the two provide identical estimates of value (Gregory 1986), in fact, analyses considering the "pollutee" as supplier vis-à-vis demander of water quality may differ.[24]

If water quality rights rest with pollutees (pollutees are suppliers), then the WTS criterion is appropriate since pollutees effectively sell water quality. However, if irrigators own water quality rights, than the WTP criterion is more appropriate as pollutees effectively will buy rather than sell water quality. Analysts using nonmarket techniques should be aware of the biases which may be introduced by an inappropriate theoretical choice.

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[24]These analyses may differ for two reasons. First, the assumption that they are similar is based on the notion that people are indifferent between two states of existence: an inferior State X with money $, and a superior State Y without money $. Yet this argument fails to recognize that individuals more commonly evaluate gains and losses from some specific reference point (Brookshire et al. 1980), and that losses from this point are usually weighted more importantly than gains (Knetsch and Sinden 1987). Therefore, endowment effects (whether one is a buyer or seller) as well as wealth effects becomes important since WTP is limited by a person's income while WTS is not. Second, WTS will include a risk premium while the WTP will not. The relevance of this premium may be most important where essential sources of utility such as health and life are involved (Randall 1981, Gordon and Knetsch 1979).

2. Salinization

As discussed in Section II, there are only a few instances where irrigation has not led to soil salinization. Therefore, it is a reasonable assumption that a given irrigation project will result in rising soil salinity if countermeasures are not taken (Shalhevet 1973). Moreover, unused irrigation water can be anticipated to negatively affect the economic productivity of groundwater and downstream water. Current knowledge of the movement of salts through soils and the establishment of salt balances is more advanced than knowledge on nitrate accumulation. Salinity changes associated with irrigation practices may be predicted with a reasonable degree of accuracy.[25] Likewise a well-developed although controversial body of knowledge

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[25]In general, river salt loads per acre of crop activity can be found by multiplying the deep percolation and seepage of water by the amount of salt found in subsurface water and displaced into rivers. Researchers have developed mathematical models to describe this relationship as well as others, such as the movement and concentration of salts in the soil with different quantities and quality of irrigation water. Tanji et al (1967) developed a computer program to predict the equilibrium concentration of salts in a system at any given moisture content. More complicated models have also included the physical and chemical characteristics of the soil (Shalhevet 1973). The Soil Conservation Service used a model to estimate the relationship between agricultural drainage water in the Colorado Basin and salt discharges into the Colorado River, and found this relationship to be about five tons of salt per acre-foot of drainage water (Gardner and Young 1988). Another model used by the U.S. Department of the Interior estimates that each 9,900 tons of salt discharge in the Upper Basin of the Colorado increases the river's salt load in the Lower Basin by one ppm (Gardner and Young 1985). The accuracy of such estimations, however, is debatable. These models depend on reliable chemical data, valid assumptions such as the amount of water discharged into rivers, and plausible theoretical considerations. According to Gardner and Young (1985), the federal government overestimated expected salinity increases in the Colorado River and have repeatedly lowered subsequent estimates. These errors were in part due to inaccurate assumptions (high river levels in 1983 and 1984 led to large drops in observed salinity), and in part to theoretical and modelling errors.

exists on crop responses to different levels of soil and water salinity.[26]

Evaluation of individuals' losses associated with changes in water salinity can be done using market or nonmarket data. The simplest market indicator might be differences in land values. If land values reflect social welfare, under conditions of perfect information the loss of welfare due to an irrigation project which raises water salinity might be estimated by variations in land value according to soil and water salinity. However, such variations must exist in the region where irrigation is being considered.

More commonly, economists predict future welfare losses by estimating regional net income lost or expenditure incurred from increased water salinity. Moore _et al_. (1974) estimated the effect of Colorado River salinity on net farm returns in California's Imperial Valley. Oyarzabal-Tamargo and Young (1978) calculated the impact of Colorado River salinity on farmers' income in the Mexicali Valley. Gardner and Young (1985) estimated agricultural damages from Colorado River salinity by comparing net farm income under two different water salinity levels. Feinerman and Yaron (1983) predicted the effect of declining water quality on net farm income in Israel. To estimate non-agricultural losses, Gardner and Young (1985) calculated the additional municipal expenditures for water softening, other water treatment costs,

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[26]As previously discussed, expert opinions vary on this point (see footnote 21). It is difficult to make generalizations on crop responses to salinity since results vary with location, soil, and management regime. The fact that many studies do not take place under field conditions may result in underestimating the effect of salinity on yields because for a given level of salinity, yields decrease when water is less available (Yaron and Ratner 1984).

and for the repair and replacement of equipment for each ppm increase in Colorado River salinity.

Such estimations of individuals' losses from salinity are complicated by the fact that there are ways of adapting to rising water salinity. Yaron and Ratner (1984) pointed out that with salinity increases, farmers can adapt their irrigation techniques and/or cropping mix to maintain levels of productivity higher than would be possible with unchanged cultural practices. Their study of alternative salinity levels predicted that farmers growing saline-sensitive fruit in Israel will suffer twice the losses of other farmers. To reduce losses in urban centers, galvanized steel water pipes can be replaced with the more salt-resistant copper ones (Gardner and Young 1985).

Water management and technology and drainage conditions also significantly influence yields where irrigation water is saline. Hardan (1977) argues that altering these components, along with cropping patterns, will achieve better results when water is highly saline. Sprinkler and flood irrigation techniques differ in their ability to leach soil salts.[27] Planting and ridging techniques can avoid some of the economic harm from saline water by preventing salts from accumulating where they will do most harm. Highly-controlled irrigation techniques (such as the use of trickler irrigation) can also help concentrate salts where they do less economic damage to crops (Shalhevet 1973). Where inadequate natural drainage conditions exist (e.g., in the presence of heavy soils, impermeable aquifers, or low water tables),

[27]According to Shalhevet (1973), slow leaching at rates below the infiltration capacity of the soil is more effective at removing accumulated salts.

artificial drainage will generally create more favorable salt conditions for irrigated crops.

Finally, water quality and quantity to some degree are economic substitutes in crop production (Yaron and Ratner 1984, Feinerman and Yaron 1983, Moore 1972). Few if any studies have estimated the marginal rate of substitution (MRS) between water salinity and water quantity (Gardner and Young 1985) -- rates that will be subject to wide variations based on locale, soils, climate, technology, and crops. Yaron and Ratner (1984) attempted to estimate this tradeoff under two different scenarios in Israel. They found the MRS between "good" water and "poor" water containing 750 ppm salts to be between 1.1 and 1.2.

Even accounting for technical, agronomic, and managerial adaptations to salinity, predicting the amount of salinity-induced damage individuals absorb is difficult without knowing the extent to which farmers are willing and able to alter cropping strategies. Damage estimates may be biased downwards if technological or cropping adaptations are overestimated.[28]

While economists commonly use market-based techniques to estimate the external costs of salinization, the rigor of such techniques can be questioned because of uncertainty over the physical changes which occur under irrigated conditions, and disagreement over the effect of soil salinity on agricultural production. In addition, agricultural markets

[28]Gardner and Young (1985) attribute the difference between their estimation of salinity-control benefits and that of the United States Bureau of Reclamation ($46,300 versus $15,600 per ppm salt reduction) to restrictions the authors placed on cropping pattern adjustments.

in lower income countries are commonly distorted.[29] Where these distortions are complex, arriving at the true social opportunity cost of water salinity through use of farmgate prices or other market data will be difficult. Use of WTP or WTS criteria may be a preferred technique. For example, rather than predicting lost revenue, as has been done, analysts might undertake surveys to determine the value of good water to Mexicali Valley farmers. Assumptions of entitlement rights would determine the appropriate technique to use. In this example, U.S. and Mexican negotiators have already agreed that Mexican farmers are entitled to a certain standard of water quality. Hence, the WTS criteria for establishing the value of good water to Valley farmers (or the magnitude of welfare losses if such a commodity were lost) would be an appropriate nonmarket technique.

In addition to water salinity, the on-farm process of soil salinization may or may not result in economic losses. Where economists or policy makers determine that private and social discount rates differ, or that farmers' management of their soil resources is not socially optimal for other reasons, these losses may be important. Their measurement would depend on which discount rate was determined as socially optimal.

[29]For various reasons, these countries intervene quite extensively in the agricultural sector. Repetto (1986) estimates that Lower Income nations on average recover only 10 to 20 percent of their costs of building and operating public irrigation works. In a study of nine countries, Repetto (1985) found the median subsidy for pesticides to be 44 percent.

B. Estimating the Social Cost of Controlling Externalities

1. Nitrate Losses

Walker (1988) outlines three possible strategies for controlling nitrate contamination of drinking water supplies: 1) seek out alternative water sources; 2) treat contaminated water so that it is restored to its uncontaminated state; and 3) control or eliminate the source of contamination.

The cost of finding alternative water sources is simply the cost of delivering water from an uncontaminated aquifer plus any additional transportation costs. These costs may also be expected to increase over time as uncontaminated aquifers become depleted.

Several U.S. and European municipalities have installed ion exchange systems to treat nitrate-contaminated water (Walker 1988, OECD 1986). Such systems use chemical, biological, or physical processes to remove nitrate ions from water. Nitrate-contaminated water can also be treated through storage until denitrification occurs with bacterial action, or through reverse osmosis processes (OECD 1986). The cost of such treatment strategies includes the costs of building, maintaining, and operating these systems. In general, such costs are high (OECD 1986). OECD (1986) examined the qualitative problems associated with nitrate-treated water, and concluded that these processes are not yet advisable in OECD countries.

Controlling or eliminating agricultural sources of groundwater nitrates requires some combination of reduced water usage, limited applications of nitrogen fertilizers, and altered cropping patterns. For example, many European countries now regulate the number of cattle allowed per hectare of land in order to control the amount of nitrogen applied in the form of animal manure. The social cost of these strategies can be calculated as the profit lost from altered or restricted farming practices. OECD (1986) suggests that these costs may be lower than treatment costs.

2. Salinization

Two of the three categories of strategic options listed above also apply to salinity control options. Options can be broadly categorized as either preventive or curative; the third option of providing alternative water sources would only be relevant where reused irrigation water is used solely for non-agricultural purposes.

There are numerous ways of "curing" salinity-related problems. Brown (1984) discussed a range of alternative plans considered by the federal government for improving water quality in the U.S.'s Red River Basin. These included diluting Basin water with imported fresh water; constructing desalinization plants; relocating especially saline waters via pipelines; and collecting and disposing saline flows in off-site containment reservoirs. The federal government selected the latter strategy at an estimated 1982 cost of $178 million in order to reduce the river's chloride concentration to 250 ppm. The Salinity Control Act of 1974 authorized construction of several salinity control works in the Upper Colorado Basin (Gardner and Young 1985). These works primarily extract highly saline water and divert effluent flows, disposing these waters elsewhere.[30]

[30] Such water diversion schemes, however, are not without their own external costs. In California's San Joaquin Valley for example, the U.S.B.R. initiated a project to drain subsurface saline irrigation water

The only known methods of effectively addressing problems of existing soil salinity are investing in improved irrigation and drainage systems and/or increasing the amount of water applied. While leaching salts from the soil reduces soil salinity, flushed salts may be relocated elsewhere -- for example in groundwater or downstream -- so that total economic harm is only redistributed. In this case, the economic cost of reclaiming saline soil may be much higher than the private cost, resulting in a socially inefficient solution.

Preventive measures can also prove successful in avoiding salinity buildup in water and soil resources. Such options usually involve adapting water management practices and investing in irrigation technology which are efficient in their water usage. For example, the U.S. Soil Conservation Service (S.C.S.) has implemented on-farm assistance programs to limit irrigation return flows, thereby reducing salinity levels in the Colorado River (Gardner and Young 1985). These programs provide investment incentives to improve irrigation and management practices such as drip irrigation, land leveling, and canal lining. Such improvements reduce both deep percolation and the river's salt load. The government's cost in investment subsidies represents the social cost of this strategy.[31]

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and dispose of it elsewhere. For several years, Kesterson Reservoir served as a temporary holding basin. Toxic concentrations of selenium accumulated which proved fatal to the Reservoir's wildlife. Since June 1986, further water storage has been prohibited (Reisner 1988).

[31]Although these investments often reduce farm labor requirements (Gardner and Young 1985), which would decrease the social cost of this policy.

C. Policy Options

Lower income countries have much fewer resources at their disposal with which to effectively formulate policy. Limited financial resources implies that there must be readily apparent benefits to investments in policy formulation. Limited professionals with training in economics, statistics, and natural sciences also makes it more difficult to undertake complex analyses. Moreover, necessary data often do not exist and cannot be gathered within any realistic time frame. Lower income countries are also disadvantaged in that they lack the institutional capability necessary for effective and efficient public intervention. These additional resource and institutional constraints make the formulation of natural resource policy in these countries especially difficult. With this in mind, a more general discussion of policy issues follows.

Economic assessment of individual welfare with and without environmental control will help guide policy makers to decide whether a given level of control can lead to a Pareto improvement in resource allocation. Frequently, intervention or control is assumed appropriate as long as Pareto-relevant externalities persist. Determining the exact level involves comparing the total costs of preventing or treating water contamination for example, with the benefits gained by individuals from each strategy (Brown 1984, Gardner and Young 1985). However, such assessments alone do not necessarily provide prescriptions for the best policy to pursue. Policy decisions can be further guided by analyses of costs, benefits, market conditions, and institutional capacity.

There are three categories of public policy response to the water-related externalities of irrigation discussed here. One is to construct centralized public works, such as ion exchange systems and desalinization plants, to treat identified problems. These are generally expensive (OECD 1986, Oyarzabal-Tamargo and Young 1978). The second is to undertake structural changes such as diverting contaminated streams and storing contaminated water, lining irrigation canals and ditches, and developing alternative water sources. These two categories are similar in that they rely on technological solutions to treat water contamination, and are generally curative in approach. Furthermore, the costs of these solutions are commonly and most conveniently paid for by the public at large.

The third category of public policy response involves directing institutional change to modify the incentives under which farmers make land-use decisions, or modify the features of an overused resource so that producers alter their use of it. Gardner and Young (1985) concluded that on the grounds of economic efficiency, preventive on-farm salinity measures proved more cost-effective than structural salinity projects. Institutional changes in addition avoid an irreversible commitment of funds.

Institutional changes can be pursued through a variety of mechanisms. Directive mechanisms, such as establishing the amount of salts or nitrate permitted in farm effluent, remain difficult to monitor. In lower income countries especially, this is difficult because of the number of polluting sources (each parcel of land) as well as the nonpoint source of contamination. Both nitrates and salts usually affect water quality through diffuse (nonpoint) rather than exact (point) channels. Nonpoint sources of pollution prohibit easy control (Gardner and Young 1988); even in the U.S., they receive much less legislative attention than do point sources (Tietenberg 1984). Establishing effective effluent standards remains a highly unlikely option in poorer countries. However, other forms of direct control on emitter behavior may be possible. Influent standards may provide one alternative. Controlling water or nitrogen fertilizer applications appears more realistic, although monitoring such restrictions will still continue to pose problems.

An alternative institutional change involves establishing incentives and disincentives for certain types of activities with the objective of reducing emissions. For example, the Soil Conservation Service (S.C.S.) and the United States Bureau of Reclamation (U.S.B.R.) have undertaken a variety of on-farm salinity control programs in the Colorado Basin (Gardner and Young 1988). Through irrigation investment subsidies, these programs reduce existing incentives for excessive water use. Alternatively, negative incentives can be provided through policies such as water taxes.

Two major criteria in policy selection include a policy's efficiency in meeting its objective, and the distribution of its costs across various members of society. With respect to the first, Alt and Miranowski (1979) demonstrate that in the face of uncertainty (defined as large confidence intervals in estimates of economically efficient standards), the efficiency of standards versus economic incentives depends on the elasticities of environmental control supply functions.

FIGURE 4: The Efficiency of Policy Mechanisms
Under Elastic and Inelastic Supply Conditions

4A: Standards 4B: Price Incentives

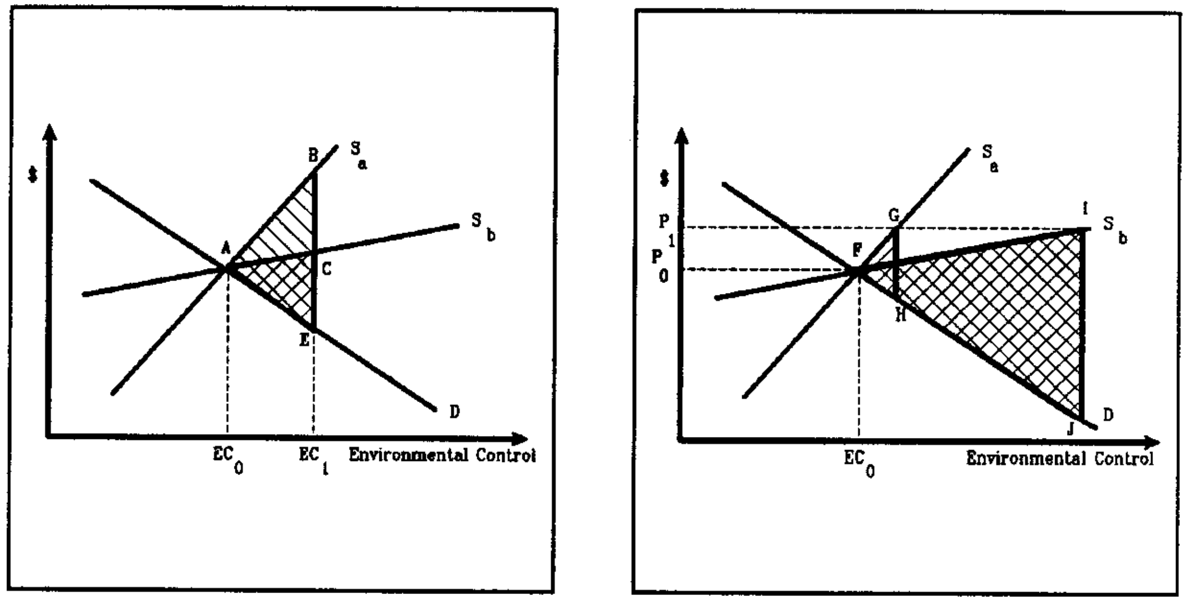

Figure 4A shows that where standards have been set at EC_1 beyond their efficient point of EC_0, losses associated with relatively inelastic supply curves (S_a) exceed those of elastic curves (S_b) (area ABE is larger than ACE). Contrary results are obtained when price incentives are established too high, for example at P_1 rather than P_0 in Figure 4B (area FGH is smaller than FIJ). If the supply of environmental control is relatively inelastic, and estimating efficient standards is highly uncertain, price incentives may be a preferable option. Alternatively, standards might be more efficient if supply functions are more elastic. Similarly (but not shown here), where demand for environmental control is relatively inelastic, either policy option in the face of uncertainty over the point at which marginal cost equals marginal benefit will likely be inefficient.

Where supply functions for environmental quality are not known with certainty, information on demand conditions alone can still be used to achieve efficient control levels. This can be obtained through use of variable price incentives. Declining price incentives corresponding with society's demand function for increasing levels of environmental control will allow suppliers to choose their own efficient level of control.

A useful linear programming model, developed by Gardner and Young (1988), simulates both the efficiency and distributional impacts of three on-farm salinity control policy alternatives in Colorado's Grand Valley. These policy alternatives comprise irrigation investment subsidies; salt discharge taxes (an effluent tax); and irrigation water taxes (an influent tax). The model allows for some cropping pattern adaptations and a choice from ten combinations of irrigation techniques and management. The model's solutions for different government policies are based on maximizing net annual returns.

Gardner and Young's findings, partially presented in Table 4, illustrate the differences between various preventive strategies in terms of the distribution of social costs. The authors found wide variations between policies both in their social costs and in the distribution of these costs between farmers and the government. While their initial analysis found effluent taxes to be the most cost-efficient in reducing salt-discharges, they are impractical to implement. Therefore, Gardner and Young limit their evaluation to comparing irrigation subsidies, water taxes, and combinations of the two

TABLE 4

The Social Costs and Distributional Impact
of Water Tax Policies and Irrigation Subsidies
to Reduce Salt Discharges in Grand Valley, Colorado.

	Policy Characteristics			Net Ann	Annual	
	Base Water[1]	Added Water	Irrig. Subsidy	Social Cost	Cost to Farmers	Cost to Govern
	--$/Acre-Foot--		%	-$/Ton of Salt Removed-		
Scenario 1	15.80	15.80	0	7.16	44.25	(37.09)
Scenario 2	4.00	15.80	0	7.16	13.92	(6.76)
Scenario 3	4.00	14.00	40	7.62	5.11	2.51
Scenario 4	4.00	28.00	60	10.26	1.73	8.53

[1]Cost per acre-foot for first three acre-feet.

Source: Gardner and Young (1988).

policies.

Table 4 lists four of these comparisons. Under one policy assumption (Scenario 1) the authors price each acre-foot of water at $15.80, $4 of which covers delivery expenses and $11.80 of which represents a water tax. In Scenario 2, they price the first three acre-feet of water at the marginal cost of $4 each, and additional acre-feet at $15.80 a foot. Both of these scenarios lead to the same reduction in salt discharges (from 237,350 to 58,350 tons per year) at the same net annual social cost ($7.16 per ton of salt reduction). This finding implies that demand for the first three acre-feet of water is perfectly inelastic between $4.00 and $15.80; raising water prices on three acre-feet thus leads to no desired change in the emitters' behavior (water use and the resulting salt discharges). Therefore, the social cost of salinity prevention remains unchanged under the two scenarios. Yet the distribution of social costs changes dramatically. Where all water is priced at $15.80 (Scenario 1), Valley farmers pay $44.25 per ton of salt reduction; this reduces their collective net income from $2.5 million to a negative figure. In this scenario, the government collects $37.09 per ton of salt reduction from water tax revenue.[32] Downstream beneficiaries experience a gain in welfare of $30 per ton of avoided salt discharges avoided.

Under Scenario 2, which prices the first three acre-feet of water at $4.00 each, the social cost remains at $7.16 per ton of salt while the cost to Valley farmers is reduced significantly from $44.25 to $13.92 per ton. This also amounts to a 32 percent reduction in Valley farmers' net income from $2.5 to $1.7 million. The government's gain in tax revenue is likewise reduced from $37.09 to $6.76 per ton of salt reduction. Benefits to downstream water users remain unchanged at $30.00 per ton.

This simulation demonstrates that an efficient on-farm influent tax strategy should consider the elasticity of demand for the good being taxed. At least for water, it is likely that demand conditions are inelastic over some price range and critical quantity. Imposing a tax on an inelastic good will result in transferring resources from farmers to the government without inducing the desired behavior.

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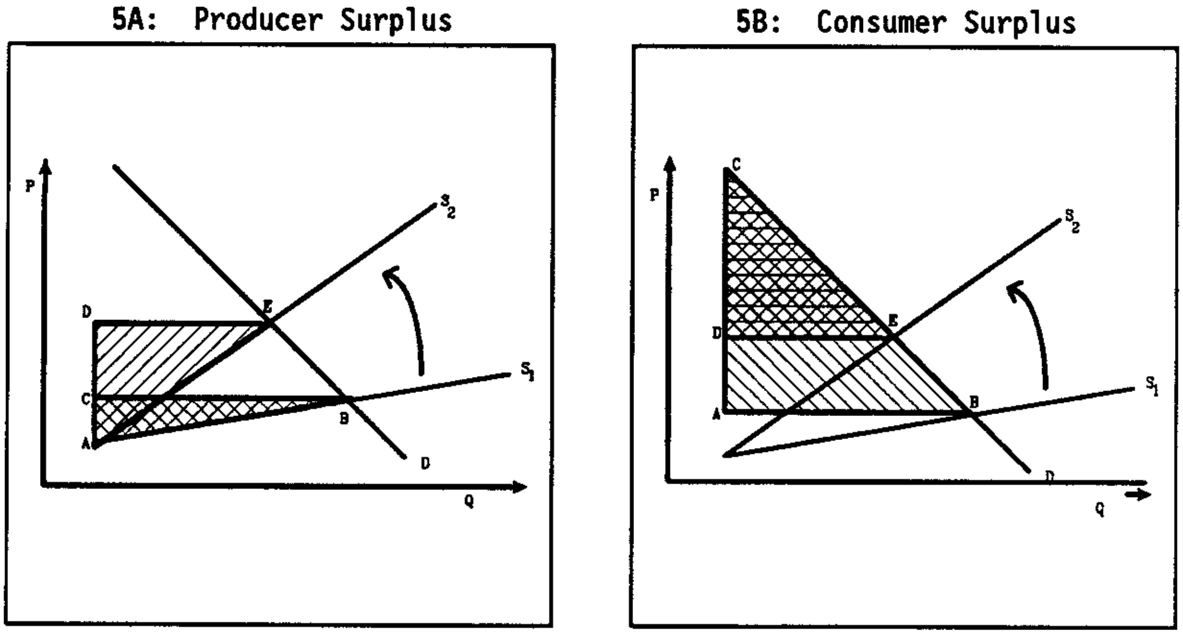
[32]Since farmers' tax payments to the government ($37.09) are a transfer rather than economic payment, $44.25 less $37.09 equals the social cost of control, or $7.16 per ton.

Gardner and Young also investigate the efficiency and distributional impact of government irrigation investment subsidies combined with water taxes. They found that providing these subsidies raised the social cost of salt discharge prevention, but significantly reduced the farmers' share of this cost. Under this dual strategy of an incentive and a penalty, any given water tax penalty achieves much higher levels of salt reduction as farmers more readily undertake investments in irrigation improvement which are partially paid for by the government (Table 4). Altering water taxes and/or altering the government's improvement subsidy can achieve any distribution of costs between the private and public sector. Scenarios 3 and 4 in Table 4 illustrate two possible combinations of these policies. In Scenario 3, farmers absorb two-thirds of the social cost of salt prevention ($5.11/$7.62) with a 40 percent subsidy, while a 60 percent subsidy (Scenario 4) reduces the farmers' share to 17 percent ($1.73/$10.26). A combined tax and subsidy policy is thus attractive in that it allows wide flexibility in cost distribution. As Table 4 indicates, however, shifting the share towards the government results in a loss of efficiency as social costs increase.

While Gardner and Young's analysis is instructive in that it evaluates the distributional as well as efficiency impact of economic incentives, the analysis is static and does not take into consideration the dynamic relationship between supply and demand. Under free market conditions, the imposition of a water tax shifts supply curves upward, resulting in a new competitive equilibrium solution. Changes in consumer and producer welfare from this tax are not obvious *a priori*.

FIGURE 5: Changes in Producer and Consumer Surplus
From a Tax Under Inelastic Demand Conditions



Producers or consumers could absorb all of the tax, or any distribution between the two could occur. Taylor and Frohberg (1977) demonstrate that under certain conditions, producers may even be better off with environmental control policies. Investigating the distributional impact of nitrogen fertilizer restrictions on consumers and producers, they found that 50 and 100 pound fertilizer restrictions reduced consumer surplus while <u>increasing</u> producer surplus.[33] This may result when there is an inelastic demand curve in the region where the upward shift in the

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[33]In their simulation of 100 and 50 pound per-acre nitrogen limitations, the authors found that consumer surplus decreased by $231 million and $3,325 million, while producer surplus increased by $21 million and $2,036 million, respectively.

supply curve occurs (see Figure 5). If a demand curve is inelastic, a price increase will yield greater revenue to suppliers because prices rise proportionally more than demand falls. This will yield greater producer surplus (profit) if this increased revenue is greater than the additional costs absorbed by producers from the environmental control measure. In Figure 5A, S_2 represents the new producer supply curve which includes the cost of providing environmental control. Under new market conditions, the producer's surplus is the area DEA, which is larger than the old surplus of ABC. Figure 5B illustrates that for the same change in market conditions, consumer surplus decreases from ABC to DEC.

V. CONCLUSIONS

This paper has illustrated how application of economic theory to water development projects in tropical arid- and semi-arid agricultural systems where relevant externalities are anticipated can complement policies to conserve natural resource stocks. Relevant externalities are broadly defined as externalities whose elimination can result in greater gains than losses to the economy. Drawing on the two examples of nitrate contamination and salinization in arid- and semi-arid irrigated systems, the paper concludes that application of economic analyses to such externalities can help achieve Pareto improvements in resource allocation. This can be done through careful assessments of losses, coupled with pragmatic evaluations of how these losses might be mitigated.

This paper finds that achieving improvements in natural resource allocation is hindered more by data limitations, methodological difficulties, and by a lack of integration between natural and social research than by economic theory. It has also identified several outstanding economic issues in natural resource allocation deserving of attention.

Identifying Pareto-relevant externalities in practice is complex because it requires understanding the physical processes associated with use of natural resources; quantifying the impact of these processes on welfare; and evaluating ways in which an economy can substitute for environmental quality. Moreover, to achieve Pareto improvements means that this information must then be incorporated into the process of policy formulation. A few of the numerous difficulties with undertaking this process are summarized below, along with suggestions on approaches economists and policy makers may take to address problems of resource degradation which may be Pareto-relevant.

The first issue confronting economists is the task of identifying the activities most likely to result in relevant externalities. With irrigation projects, this includes activities that significantly alter the hydrological cycle; that are associated with important secondary usage of irrigation water; and that include high amounts of chemical inputs. It may be clear in some cases that the risk of damage is minimal. For irrigation, this is especially true where hydrological or demographic conditions limit secondary usage, or where the complexity of ecological change introduced is matched by management sophistication.

Such informed judgments on the potential for agricultural systems to produce relevant externalities depend on information being available to economists. To date, however, comparative data on agro-ecological dynamics in the tropics are lacking. Detailed and generalizable models have been developed for water and soil salinity, but general models are less available for nitrate losses. There is strong empirical evidence that under irrigated conditions nitrates enter groundwater, but research remains generally indirect, scant, and site specific. Few if any studies develop more predictive models. Unlike problems of salinity, nutrient loss studies generally have not benefitted from coordinated agronomic and economic research. This points to a need for regionally coordinated agro-ecological research designed to illuminate the policy implications of natural resource management. The effects of irrigation on soil fertility and water quality are two high-priority topics. Undertaking such research, and making its findings accessible to economists, would almost certainly require some change in the interface between policy formulation and scientific research.

While it remains difficult to predict the exact degree of physical resource degradation associated with agricultural activities, this paper also suggests that, for some problems at least, there is sufficient understanding to incorporate estimated changes into policy analysis. Hydrological studies provide good estimates of the rate of salt build-up from irrigation, and scientists have well-documented the principal mechanisms which influence the nitrogen cycle. Economic analysis could rely on a range of estimated physical changes in resource quality. An alternative approach would be simply to recognize that qualitative changes will occur, to develop indicators of these changes for priority resources, and finally to monitor these indicators. Some priorities might be salinity levels in irrigation water, the percentage of organic matter in soils, and the availability and cost of fuelwood. These indicators could then be evaluated and fed back into policy formulation.

Obtaining and considering information on physical resource degradation still leaves a second issue concerning the relationship of these changes to individual welfare. In some situations, market data allow good indications of losses. Where externalities occur over an extended time period, losses become more difficult to estimate because the elasticity of substitution between natural and other resources may change. Loss estimates using market data become subject to three sorts of uncertainty: future developments in technology, the economics of technological adoption, and the rate at which farmers will adapt to changing economic conditions. Technological and economic change, as well as behavioral adaptation, could weaken the link between water salinity and social welfare for example. Species extinction and increased health risks, on the other hand, may represent irreversible losses of goods for which no substitute exists. With so many varied sources of uncertainty, a useful policy distinction might be made between categories of natural resources classified according to their substitutability -- e.g., the extent to which technological and other changes will affect their contributions to individual welfare. For relatively substitutable natural resources (e.g., water salinity, which has substitutes in water quantity and technology), economists should attempt to assess the likelihood that changes in relative factor prices or technology will alter these relationships. For relatively non-substitutable natural resources, policy makers may choose to implement a more conservationist, worst-case scenario.

In other situations, particularly where resource degradation presents increased health risks, nonmarket survey techniques for evaluating losses may be more appropriate. For a host of pragmatic reasons, however, this form of social research may be highly inappropriate. For one, nonmarket data from surveys are difficult and expensive to obtain. Moreover, the probability is high for biased survey results from poor survey techniques; the cost or time involved in surveying may be prohibitive; and inhabitants may be uninformed about the issues presented them. For these reasons, damage valuation might simply be represented by best estimates (or a range of estimates) of expected changes in resource quality, together with demographic and other social characteristics of the affected human and animal populations. This may provide the best indicator of risk and potential losses. Where data do not exist to establish these estimates, it may be preferable to establish acceptable levels of loss for particularly important resources -- such as rates of deforestation and erosion -- above which public action will be taken.

Third, natural resource degradation results from decisions made by farmers, and hence is a function of their decision-making environment. There are numerous reasons why incentives may exist for degradation. Incomplete information, subsidies which result in undervaluing resource conservation, and high discount rates are three examples. These types of economic "distortions" leading to temporal externalities present an especially difficult issue because they often lead to farmers favoring present over future consumption. It is difficult to argue that very poor farmers or countries should save for tomorrow. At the least, however, economists should recognize economic distortions which favor short time horizons so that appropriate corrective policies can be applied. Extension efforts introducing conservation technology may be more effective in instances where farmers face incomplete information, while subsidizing this technology might be more effective where other incentives discourage the adoption of this technology.

Because the time frame and discount rate used in project evaluation may undervalue natural resources, the effect of these choices on resource quality is a fourth issue that must be addressed by economists. Any discount rate above zero or any shorter time frame discourages valuing the productivity of environment-protecting goods and services. The appropriate rate will depend on a country's long range goals and its ability to undertake politically difficult short-term sacrifices. While on economic grounds a country may plan to encourage sustainable extraction of forest products, more immediate income demands may lead it to commit resources to rangeland development, resulting in deforestation. The decision of which discount rate to use is not one that can be arrived at through economic analysis alone since it raises issues such as intergenerational equity which economic theory does not adequately address. However, economists can help to illuminate the tradeoffs associated with different discount rate options.

A final issue is defining a government's role in addressing problems of natural resource degradation. Broadly, there are two policy approaches to mitigating environmental damage -- curative and preventive measures. The undertaking of curative solutions raises several concerns. One, they usually represent a large and irreversible commitment of funds. It is also difficult to distribute these costs among the responsible parties, should such a policy be preferred. The public at large rather than the polluting parties has paid for structural salinity and nitrate control projects in the U.S. In the case of salinity control, this arrangement has resulted in many economically inefficient programs and works.[34] Finally, diverting or

[34]For example, an economic assessment of salinity control programs in the Colorado River Basin by Gardner and Young (1985) found that economic benefits appeared to exceed costs in only five of the nineteen projects evaluated. One reason for this was that investment improvements were often decided on technical and political rather than

storing contaminated water may result in unaccounted ecological changes of economic importance.

From both efficiency and equity perspectives, preventive measures appear generally preferable to curative ones. To the extent that these involve institutional change, they involve their own difficulties. Monitoring water usage, instituting water taxes, disseminating and subsidizing new irrigation technology, and adapting farming systems to salt-control or salt-resistant practices involve institutional sophistication which may be beyond the current abilities of many lower income nations. Taxes can be used to drive up the price of certain resource-depleting activities such as excessive water and fertilization practices. While some governments may be able to tax fertilizer usage, most would have a difficult time widely taxing water usage. Subsidies, on the other hand, which can encourage the adoption of resource-conserving activities such as water-efficient irrigation systems, are generally easier for governments to implement via investment subsidies and extension services.

The appropriate government role will not only depend on who should pay (i.e., the public in general, pollutees, or polluters), but also on policy effectiveness. This implies that until or unless the institutional capacity exists to distribute costs and benefits in socially desired ways, conserving natural resources in lower income countries will mostly demand a commitment of public funds. Economists

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economic grounds. Another is that farmers tended to choose the most expensive package (providing them with the largest benefit), with little concern over the cost-effectiveness of their choice because the package was highly subsidized.

and conservationists will have the added responsibility of convincing policy makers that such use of public funds represents not only a Pareto improvement, but also an economic priority.

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APPENDICES

APPENDIX A

Indexes of Total and Per Capita Agricultural Production in Africa, Central America, South America, and Asia

	Index of Agricultural Production (1979–81 = 100)				Index of Food Production (1979–81 = 100)				Crop Yields					
									Cereals			Roots and Tubers		
	Total		Per Capita		Total		Per Capita		Kilograms per Hectare	Percentage Change Over		Kilograms per Hectare	Percentage Change Over	
	1964–66	1984–86	1964–66	1984–86	1964–66	1984–86	1964–66	1984–86	1984–86	1964–66	1974–76	1984–86	1964–66	1974–76
WORLD	**71**	**113**	**95**	**104**	**70**	**113**	**94**	**104**	**2,552**	**65**	**31**	**12,768**	**24**	**12**
AFRICA	**72**	**110**	**109**	**95**	**71**	**110**	**108**	**95**	**1,077**	**28**	**7**	**7,735**	**26**	**17**
Algeria	67	119	104	102	67	118	105	101	867	65	27	7,525	3	3
Angola	118	102	176	89	85	102	127	90	461	-47	-39	14,088	22	7
Benin	66	137	94	118	66	133	94	114	825	54	14	8,241	34	8
Botswana	82	96	135	79	81	96	134	79	178	-52	-70	5,385	35	16
Burkina Faso	83	128	110	114	85	127	113	113	690	33	30	6,568	96	40
Burundi	75	109	94	95	79	109	100	95	1,101	12	-3	7,538	-3	8
Cameroon	63	108	88	94	63	108	89	94	935	18	-2	2,455	10	3
Cape Verde	121	X	162	X	122	X	163	X	551	-11	23	3,007	-27	-29
Central African Rep	72	104	95	93	71	103	94	92	513	-31	-1	3,882	10	20
Chad	94	112	126	100	92	112	124	100	531	-13	-7	5,182	14	28
Comoros	70	X	114	X	70	X	114	X	1,116	-15	1	3,259	-4	-6
Congo	77	106	110	93	78	106	110	93	622	-43	-6	6,457	31	14
Cote d'Ivoire	47	113	85	94	40	122	73	101	981	23	21	6,282	70	38
Djibouti	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Egypt	73	113	102	100	70	117	98	104	4,471	26	14	18,572	7	4
Equatorial Guinea	X	X	X	X	X	X	X	X	X	X	X	2,395	-33	-12
Ethiopia	78	98	111	87	78	97	111	86	1,081	39	12	2,827	-7	-13
Gabon	93	106	110	98	93	106	110	98	1,481	-6	1	6,393	0	9
Gambia	107	127	150	115	109	127	152	116	1,207	15	29	3,000	-32	-8
Ghana	81	130	119	110	82	131	120	111	969	7	11	8,641	5	41
Guinea	78	104	105	93	78	104	106	93	728	-10	-13	7,089	-5	0
Guinea-Bissau	91	138	140	125	91	138	140	125	848	19	16	6,154	0	26
Kenya	60	115	108	93	67	105	119	85	1,611	31	3	8,929	21	14
Lesotho	91	95	127	84	86	94	120	83	683	-12	-12	15,000	3	16
Liberia	61	115	96	98	60	116	95	99	1,302	107	7	4,014	-3	3
Libya	43	162	78	133	43	162	78	133	616	107	37	6,777	48	34
Madagascar	73	112	106	98	72	113	105	98	1,731	1	-4	5,926	-7	-3
Malawi	55	110	83	94	58	105	87	90	1,162	24	7	4,231	-13	-7
Mali	70	115	97	100	73	115	100	100	807	3	4	9,240	12	2
Mauritania	97	101	143	87	97	101	143	87	431	20	4	1,903	-24	70
Mauritius	86	113	106	103	90	110	111	100	3,200	59	24	25,939	108	65
Morocco	73	120	110	106	73	120	110	105	1,145	58	9	5,420	-48	-53
Mozambique	83	98	138	85	80	99	132	85	660	-29	-8	5,783	20	18
Niger	74	98	105	85	74	98	105	85	366	-30	-7	8,877	10	34
Nigeria	76	121	126	102	76	121	125	103	1,121	67	69	11,260	43	12
Rwanda	47	104	76	88	48	102	78	86	1,289	2	24	7,780	42	-6
Senegal	94	116	155	102	95	115	156	101	709	24	-1	4,232	2	39
Sierra Leone	78	104	97	95	79	106	99	97	1,431	8	0	3,425	-6	-20
Somalia	78	105	144	90	78	105	144	90	725	47	14	10,792	8	0
South Africa	61	95	85	84	58	94	82	84	1,398	48	0	13,531	63	12
Sudan	62	113	93	98	59	110	89	96	508	-27	-22	3,408	-1	-5
Swaziland	44	112	65	96	46	112	68	96	1,528	225	3	1,815	-53	-52
Tanzania, United Rep	59	108	96	91	54	111	87	93	1,109	41	13	11,075	109	58
Togo	77	105	119	91	77	104	118	90	865	83	-8	10,498	-12	-19
Tunisia	69	119	96	107	70	119	96	107	808	17	-2	11,262	37	19
Uganda	73	132	119	112	67	131	110	111	949	5	-22	6,432	64	46
Zaire	76	116	110	100	75	115	110	100	851	24	14	7,016	4	2
Zambia	64	113	100	96	63	112	98	95	1,747	106	44	3,687	13	9
Zimbabwe	63	117	105	98	58	112	96	94	1,460	63	3	4,907	22	23
NORTH & CENTRAL AMERICA	**73**	**105**	**92**	**98**	**71**	**106**	**90**	**99**	**3,837**	**57**	**33**	**19,720**	**26**	**9**
Barbados	98	88	106	87	98	88	106	87	2,500	30	-4	8,013	-9	-25
Canada	82	115	100	109	82	115	100	109	2,299	32	13	24,917	35	16
Costa Rica	52	111	80	97	50	105	77	92	2,395	74	38	7,056	-4	-16
Cuba	58	113	72	110	56	113	70	109	2,691	127	32	6,377	18	17
Dominican Rep	65	111	97	99	63	113	94	100	3,229	55	20	6,307	0	4
El Salvador	65	91	104	78	55	102	88	88	1,747	48	13	15,090	95	29
Guatemala	56	103	88	89	53	112	84	97	1,672	81	15	4,528	18	18
Haiti	80	108	112	95	77	108	108	95	1,221	14	4	4,110	-3	-5
Honduras	57	103	91	87	59	102	94	86	1,412	21	29	7,262	36	109
Jamaica	75	110	92	102	74	110	91	102	2,033	73	15	12,102	13	19
Mexico	60	110	96	97	56	111	90	98	2,368	81	39	14,382	68	19
Nicaragua	70	89	111	76	63	90	99	76	1,862	81	78	5,764	39	42
Panama	63	111	93	99	63	108	93	97	1,567	63	31	8,149	-1	-1
Trinidad and Tobago	90	98	99	90	89	99	97	91	2,460	1	-14	12,069	35	2
United States	74	103	86	98	72	105	84	100	4,618	59	38	31,215	45	15
SOUTH AMERICA	**66**	**112**	**94**	**100**	**64**	**113**	**91**	**101**	**2,038**	**44**	**24**	**11,452**	**2**	**4**
Argentina	71	107	90	98	70	107	89	99	2,508	61	27	16,202	61	22
Bolivia	59	105	86	91	60	106	87	93	1,272	34	11	5,100	-7	-27
Brazil	61	117	88	105	56	119	81	107	1,719	28	21	12,072	-8	1
Chile	77	109	100	101	76	109	100	101	3,003	71	79	13,963	52	43
Colombia	57	104	82	94	56	107	79	96	2,608	96	10	11,404	49	23
Ecuador	73	115	113	99	73	114	114	98	1,783	96	32	11,026	34	3
Guyana	81	89	109	81	81	89	109	81	3,471	71	62	7,131	18	8
Paraguay	54	120	85	103	59	115	92	99	1,555	23	11	14,330	6	1
Peru	89	111	135	98	89	114	134	100	2,457	58	34	8,093	21	15
Suriname	83	122	88	116	81	122	86	116	4,007	31	10	6,234	-13	6
Uruguay	91	107	99	104	87	104	94	101	1,951	101	52	5,727	19	9
Venezuela	56	107	95	93	54	106	92	92	2,135	69	38	8,577	-1	12

	Index of Agricultural Production (1979–81 = 100)				Index of Food Production (1979–81 = 100)				Crop Yields					
									Cereals			Roots and Tubers		
	Total		Per Capita		Total		Per Capita		Kilograms per Hectare	Percentage Change Over		Kilograms per Hectare	Percentage Change Over	
	1964–66	1984–86	1964–66	1984–86	1964–66	1984–86	1964–66	1984–86	1984–86	1964–66	1974–76	1984–86	1964–66	1974–76
ASIA	**66**	**123**	**92**	**113**	**66**	**122**	**92**	**112**	**2,523**	**80**	**36**	**13,950**	**57**	**18**
Afghanistan	80	101	105	97	79	102	104	98	1,314	23	-1	13,648	36	0
Bahrain	X	X	X	X	X	X	X	X	X	X	X	22,407	12	3
Bangladesh	80	114	121	99	79	114	119	99	2,227	35	26	10,566	29	6
Bhutan	70	112	94	102	70	113	94	102	1,407	-1	-1	6,873	6	4
Burma	66	135	95	123	66	136	95	124	2,925	91	68	9,805	189	74
China	59	133	81	125	59	130	80	122	3,891	122	58	15,614	81	21
Cyprus	67	98	72	92	66	98	71	93	1,610	51	15	21,659	40	9
India	67	122	94	111	66	123	93	111	1,590	76	35	14,268	61	21
Indonesia	57	128	81	117	57	130	80	118	3,458	126	48	10,304	48	23
Iran	54	114	87	98	52	113	83	98	1,185	35	12	14,243	-21	-18
Iraq	64	130	106	108	63	129	104	108	1,020	31	19	16,474	65	83
Israel	53	114	80	103	57	117	87	106	1,679	-1	-21	38,518	66	26
Japan	96	108	113	105	94	110	111	107	5,901	34	5	24,495	33	15
Jordan	128	122	191	102	129	122	193	102	542	-47	-16	19,082	115	62
Kampuchea, Dem	191	162	201	144	182	161	191	143	1,209	8	-5	7,709	-24	-8
Korea, Dem People's Rep	49	120	73	106	49	120	72	106	4,388	53	22	12,954	23	6
Korea, Rep	60	112	80	103	60	113	80	104	5,625	86	36	21,456	27	27
Kuwait	X	X	X	X	X	X	X	X	X	X	X	15,000	X	17
Lao People's Dem Rep	63	138	93	124	63	139	93	125	2,118	159	61	9,901	33	0
Lebanon	74	119	93	118	73	120	90	120	1,225	29	0	20,951	78	200
Malaysia	47	117	68	104	43	123	63	109	2,772	33	2	9,653	3	-8
Mongolia	88	108	136	94	88	110	136	96	1,248	60	38	11,968	51	56
Nepal	82	113	116	101	82	114	116	102	1,651	-10	-6	5,676	-1	0
Oman	X	X	X	X	X	X	X	X	1,787	60	40	4,032	X	X
Pakistan	60	121	91	104	59	118	90	101	1,668	91	20	10,199	10	-5
Philippines	53	106	78	94	53	106	79	94	1,852	77	39	5,895	5	11
Qatar	X	X	X	X	X	X	X	X	2	X	-100	13,430	X	35
Saudi Arabia	41	220	80	176	41	221	80	177	3,356	150	356	18,900	115	274
Singapore	43	101	55	96	42	102	54	96	X	X	X	11,144	10	-4
Sri Lanka	63	97	84	89	51	95	68	87	2,850	60	56	11,141	81	151
Syrian Arab Rep	47	113	78	94	41	111	69	93	899	13	-5	16,840	68	29
Thailand	56	120	83	109	55	120	82	108	2,075	14	10	13,731	5	-4
Turkey	65	110	92	99	64	111	91	100	1,961	67	24	18,715	61	35
United Arab Emirates	X	X	X	X	X	X	X	X	2,190	X	X	10,257	X	-36
Viet Nam	66	128	93	116	66	128	93	116	2,698	41	27	5,603	-4	-13
Yemen	78	124	102	108	78	125	101	109	543	-27	-34	21,312	245	94
Yemen, Dem	74	101	101	88	71	99	98	87	1,655	36	0	13,558	171	19

Source: IIED and WRI (1988).

APPENDIX B

Global Distribution of Salt-Affected Areas

Continent	Country	Area, 1000 ha Saline/ Solonchaks	Sodic/ Solonetz	Total
North America	Canada	264	6 974	7 238
	USA	5 927	2 590	8 517
Mexico and Central America	Cuba	316	-	316
	Mexico	1 649	-	1 649
South America	Argentina	32 473	53 139	85 612
	Bolivia	5 233	716	5 949
	Brazil	4 141	362	4 503
	Chile	5 000	3 642	8 642
	Colombia	907	-	907
	Ecuador	387	-	387
	Paraguay	20 008	1 894	21 902
	Peru	21	-	21
	Venezuela	1 240	-	1 240
Africa	Afars and Issas	1 741	-	1 741
	Algeria	3 021	129	3 150
	Angola	440	86	526
	Botswana	5 009	670	5 679
	Chad	2 417	5 850	8 267
	Egypt	7 360	-	7 360
	Ethiopia	10 608	425	11 033
	Gambia	150	-	150
	Ghana	200	118	318
	Guinea	525	-	525
	Guinea-Bissau	194	-	194
	Kenya	4 410	448	4 858
	Liberia	362	44	406
	Libyan Arab Jamahiriya	2 457	-	2 457
	Madagascar	37	1 287	1 324
	Mali	2 770	-	2 770
	Mauritania	640	-	640
	Morocco	1 148	-	1 148
	Namibia	562	1 751	2 313
	Niger	-	1 389	1 389
	Nigeria	665	5 837	6 502
	Rhodesia	-	26	26
	Senegal	765	-	765
	Sierra Leone	307	-	307
	Somalia	1 569	4 033	5 602
	Sudan	2 138	2 736	4 874
	Tunisia	990	-	990
	United Rep. of Cameroon	-	671	671
	United Rep. of Tanzania	2 954	583	3 537
	Zaire	53	-	53
	Zambia	-	863	863

Continent	Country	Area, 1000 ha Saline/ Solonchaks	Sodic/ Solonetz	Total
South Asia	Afghanistan	3 103	-	3 101
	Bangladesh	2 479	538	3 017
	Burma	634	-	634
	India	23 222	574	23 796
	Iran	26 399	686	27 085
	Iraq	6 726	-	6 726
	Israel	28	-	28
	Jordan	180	-	180
	Kuwait	209	-	209
	Muscat and Oman	290	-	290
	Pakistan	10 456	-	10 456
	Qatar	225	-	225
	Sarawak	1 538	-	1 538
	Saudi Arabia	6 002	-	6 002
	Sri Lanka	200	-	200
	Syrian Arab Rep.	532	-	532
	United Arab Emirates	1 089	-	1 089
North and Central Asia	China	36 221	437	36 658
	Mongolia	4 070	-	4 070
	USSR	51 092	119 628	170 720
South-East Asia	Democratic Kampuchea	1 291	-	1 291
	Indonesia	13 213	-	13 213
	Malaysia	3 040	-	3 040
	Socialist Rep. of Vietnam	983	-	983
	Thailand	1 456	-	1 456
Australasia	Australia	17 269	339 971	357 240
	Fiji	90	-	90
	Solomon Islands	238	-	238

Continent	Country	Area, 1000 ha Saline/ Solonchaks	Sodic/ Solonetz	Potential Salt affected Soils	Total
Europe	Czechoslovakia	6.2	14.5	85.0	105.7
	France	175.0	75.0	-	250.0
	Hungary	1.6	384.5	885.5	1 271.6
	Italy	50.0	-	400.0	450.0
	Rumania	40.0	210.0	-	250.0
	Spain	/	/	/	840.0
	USSR	7 546.0	21 998.0	17 781.0	47 325.0
	Yugoslavia	20.0	235.0	-	255.0

Source: Abrol et al. (1988).

APPENDIX C

Yield Potential and Crop Tolerance of Selected Crops As Influenced by Irrigation Water Salinity (EC_w) or Soil Salinity (EC_e)

Yield Potential

FIELD CROPS	100%		90%		75%		50%		0% "maximum"[3]	
	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w
Barley *(Hordeum vulgare)*[4]	8.0	5.3	10	6.7	13	8.7	18	12	28	19
Cotton *(Gossypium hirsutum)*	7.7	5.1	9.6	6.4	13	8.4	17	12	27	18
Sugarbeet *(Beta vulgaris)*[5]	7.0	4.7	8.7	5.8	11	7.5	15	10	24	16
Sorghum *(Sorghum bicolor)*	6.8	4.5	7.4	5.0	8.4	5.6	9.9	6.7	13	8.7
Wheat *(Triticum aestivum)*[4,6]	6.0	4.0	7.4	4.9	9.5	6.3	13	8.7	20	13
Wheat, durum *(Triticum turgidum)*	5.7	3.8	7.6	5.0	10	6.9	15	10	24	16
Soybean *(Glycine max)*	5.0	3.3	5.5	3.7	6.3	4.2	7.5	5.0	10	6.7
Cowpea *(Vigna unguiculata)*	4.9	3.3	5.7	3.8	7.0	4.7	9.1	6.0	13	8.8
Groundnut (Peanut) *(Arachis hypogaea)*	3.2	2.1	3.5	2.4	4.1	2.7	4.9	3.3	6.6	4.4
Rice (paddy) *(Oriza sativa)*	3.0	2.0	3.8	2.6	5.1	3.4	7.2	4.8	11	7.6
Sugarcane *(Saccharum officinarum)*	1.7	1.1	3.4	2.3	5.9	4.0	10	6.8	19	12
Corn (maize) *(Zea mays)*	1.7	1.1	2.5	1.7	3.8	2.5	5.9	3.9	10	6.7
Flax *(Linum usitatissimum)*	1.7	1.1	2.5	1.7	3.8	2.5	5.9	3.9	10	6.7
Broadbean *(Vicia faba)*	1.5	1.1	2.6	1.8	4.2	2.0	6.8	4.5	12	8.0
Bean *(Phaseolus vulgaris)*	1.0	0.7	1.5	1.0	2.3	1.5	3.6	2.4	6.3	4.2
VEGETABLE CROPS										
Squash, zucchini (courgette) *(Cucurbita pepo melopepo)*	4.7	3.1	5.8	3.8	7.4	4.9	10	6.7	15	10
Beet, red *(Beta vulgaris)*[5]	4.0	2.7	5.1	3.4	6.8	4.5	9.6	6.4	15	10
Squash, scallop *(Cucurbita pepo melopepo)*	3.2	2.1	3.8	2.6	4.8	3.2	6.3	4.2	9.4	6.3
Broccoli *(Brassica oleracea botrytis)*	2.8	1.9	3.9	2.6	5.5	3.7	8.2	5.5	14	9.1
Tomato *(Lycopersicon esculentum)*	2.5	1.7	3.5	2.3	5.0	3.4	7.6	5.0	13	8.4
Cucumber *(Cucumis sativus)*	2.5	1.7	3.3	2.2	4.4	2.9	6.3	4.2	10	6.8
Spinach *(Spinacia oleracea)*	2.0	1.3	3.3	2.2	5.3	3.5	8.6	5.7	15	10
Celery *(Apium graveolens)*	1.8	1.2	3.4	2.3	5.8	3.9	9.9	6.6	18	12
Cabbage *(Brassica oleracea capitata)*	1.8	1.2	2.8	1.9	4.4	2.9	7.0	4.6	12	8.1
Potato *(Solanum tuberosum)*	1.7	1.1	2.5	1.7	3.8	2.5	5.9	3.9	10	6.7
Corn, sweet (maize) *(Zea mays)*	1.7	1.1	2.5	1.7	3.8	2.5	5.9	3.9	10	6.7
Sweet potato *(Impomoea batatas)*	1.5	1.0	2.4	1.6	3.8	2.5	6.0	4.0	11	7.1
Pepper *(Capsicum annuum)*	1.5	1.0	2.2	1.5	3.3	2.2	5.1	3.4	8.6	5.8
Lettuce *(Lactuca sativa)*	1.3	0.9	2.1	1.4	3.2	2.1	5.1	3.4	9.0	6.0
Radish *(Raphanus sativus)*	1.2	0.8	2.0	1.3	3.1	2.1	5.0	3.4	8.9	5.9
Onion *(Allium cepa)*	1.2	0.8	1.8	1.2	2.8	1.8	4.3	2.9	7.4	5.0
Carrot *(Daucus carota)*	1.0	0.7	1.7	1.1	2.8	1.9	4.6	3.0	8.1	5.4
Bean *(Phaseolus vulgaris)*	1.0	0.7	1.5	1.0	2.3	1.5	3.6	2.4	6.3	4.2
Turnip *(Brassica rapa)*	0.9	0.6	2.0	1.3	3.7	2.5	6.5	4.3	12	8.0

FORAGE CROPS	100%		90%		75%		50%		0% "maximum"[3]	
	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w
Wheatgrass, tall *(Agropyron elongatum)*	7.5	5.0	9.9	6.6	13	9.0	19	13	31	21
Wheatgrass, fairway crested *(Agropyron cristatum)*	7.5	5.0	9.0	6.0	11	7.4	15	9.8	22	15
Bermuda grass *(Cynodon dactylon)*[7]	6.9	4.6	8.5	5.6	11	7.2	15	9.8	23	15
Barley (forage) *(Hordeum vulgare)*[4]	6.0	4.0	7.4	4.9	9.5	6.4	13	8.7	20	13
Ryegrass, perennial *(Lolium perenne)*	5.6	3.7	6.9	4.6	8.9	5.9	12	8.1	19	13
Trefoil, narrowleaf birdsfoot[8] *(Lotus corniculatus tenuifolium)*	5.0	3.3	6.0	4.0	7.5	5.0	10	6.7	15	10
Harding grass *(Phalaris tuberosa)*	4.6	3.1	5.9	3.9	7.9	5.3	11	7.4	18	12
Fescue, tall *(Festuca elatior)*	3.9	2.6	5.5	3.6	7.8	5.2	12	7.8	20	13
Wheatgrass, standard crested *(Agropyron sibiricum)*	3.5	2.3	6.0	4.0	9.8	6.5	16	11	28	19
Vetch, common *(Vicia angustifolia)*	3.0	2.0	3.9	2.6	5.3	3.5	7.6	5.0	12	8.1
Sudan grass *(Sorghum sudanense)*	2.8	1.9	5.1	3.4	8.6	5.7	14	9.6	26	17
Wildrye, beardless *(Elymus triticoides)*	2.7	1.8	4.4	2.9	6.9	4.6	11	7.4	19	13
Cowpea (forage) *(Vigna unguiculata)*	2.5	1.7	3.4	2.3	4.8	3.2	7.1	4.8	12	7.8
Trefoil, big *(Lotus uliginosus)*	2.3	1.5	2.8	1.9	3.6	2.4	4.9	3.3	7.6	5.0
Sesbania *(Sesbania exaltata)*	2.3	1.5	3.7	2.5	5.9	3.9	9.4	6.3	17	11
Sphaerophysa *(Sphaerophysa salsula)*	2.2	1.5	3.6	2.4	5.8	3.8	9.3	6.2	16	11
Alfalfa *(Medicago sativa)*	2.0	1.3	3.4	2.2	5.4	3.6	8.8	5.9	16	10
Lovegrass *(Eragrostis sp.)*[9]	2.0	1.3	3.2	2.1	5.0	3.3	8.0	5.3	14	9.3
Corn (forage) (maize) *(Zea mays)*	1.8	1.2	3.2	2.1	5.2	3.5	8.6	5.7	15	10
Clover, berseem *(Trifolium alexandrinum)*	1.5	1.0	3.2	2.2	5.9	3.9	10	6.8	19	13
Orchard grass *(Dactylis glomerata)*	1.5	1.0	3.1	2.1	5.5	3.7	9.6	6.4	18	12
Foxtail, meadow *(Alopecurus pratensis)*	1.5	1.0	2.5	1.7	4.1	2.7	6.7	4.5	12	7.9
Clover, red *(Trifolium pratense)*	1.5	1.0	2.3	1.6	3.6	2.4	5.7	3.8	9.8	6.6
Clover, alsike *(Trifolium hybridum)*	1.5	1.0	2.3	1.6	3.6	2.4	5.7	3.8	9.8	6.6
Clover, ladino *(Trifolium repens)*	1.5	1.0	2.3	1.6	3.6	2.4	5.7	3.8	9.8	6.6
Clover, strawberry *(Trifolium fragiferum)*	1.5	1.0	2.3	1.6	3.6	2.4	5.7	3.8	9.8	6.6
FRUIT CROPS[10]										
Date palm *(Phoenix dactylifera)*	4.0	2.7	6.8	4.5	11	7.3	18	12	32	21
Grapefruit *(Citrus paradisi)*[11]	1.8	1.2	2.4	1.6	3.4	2.2	4.9	3.3	8.0	5.4
Orange *(Citrus sinensis)*	1.7	1.1	2.3	1.6	3.3	2.2	4.8	3.2	8.0	5.3
Peach *(Prunus persica)*	1.7	1.1	2.2	1.5	2.9	1.9	4.1	2.7	6.5	4.3
Apricot *(Prunus armeniaca)*[11]	1.6	1.1	2.0	1.3	2.6	1.8	3.7	2.5	5.8	3.8
Grape *(Vitus sp.)*[11]	1.5	1.0	2.5	1.7	4.1	2.7	6.7	4.5	12	7.9
Almond *(Prunus dulcis)*[11]	1.5	1.0	2.0	1.4	2.8	1.9	4.1	2.8	6.8	4.5

FRUIT CROPS [10]	100%		90%		75%		50%		0% "maximum" [3]	
	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w	EC_e	EC_w
Plum, prune *(Prunus domestica)* [11]	1.5	1.0	2.1	1.4	2.9	1.9	4.3	2.9	7.1	4.7
Blackberry *(Rubus sp.)*	1.5	1.0	2.0	1.3	2.6	1.8	3.8	2.5	6.0	4.0
Boysenberry *(Rubus ursinus)*	1.5	1.0	2.0	1.3	2.6	1.8	3.8	2.5	6.0	4.0
Strawberry *(Fragaria sp.)*	1.0	0.7	1.3	0.9	1.8	1.2	2.5	1.7	4	2.7

1 Adapted from Maas and Hoffman (1977) and Maas (1984). These data should only serve as a guide to relative tolerances among crops. Absolute tolerances vary depending upon climate, soil conditions and cultural practices. In gypsiferous soils, plants will tolerate about 2 dS/m higher soil salinity (ECe) than indicated but the water salinity (ECw) will remain the same as shown in this table.

2 ECe means average root zone salinity as measured by electrical conductivity of the saturation extract of the soil, reported in deciSiemens per metre (dS/m) at 25°C. ECw means electrical conductivity of the irrigation water in deciSiemens per metre (dS/m). The relationship between soil salinity and water salinity (ECe = 1.5 ECw) assumes a 15-20 percent leaching fraction and a 40-30-20-10 percent water use pattern for the upper to lower quarters of the root zone. These assumptions were used in developing the guidelines in Table 1.

3 The zero yield potential or maximum ECe indicates the theoretical soil salinity (ECe) at which crop growth ceases.

4 Barley and wheat are less tolerant during germination and seedling stage; ECe should not exceed 4-5 dS/m in the upper soil during this period.

5 Beets are more sensitive during germination; ECe should not exceed 3 dS/m in the seeding area for garden beets and sugar beets.

6 Semi-dwarf, short cultivars may be less tolerant.

7 Tolerance given is an average of several varieties; Suwannee and Coastal Bermuda grass are about 20 percent more tolerant, while Common and Greenfield Bermuda grass are about 20 percent less tolerant.

8 Broadleaf Birdsfoot Trefoil seems less tolerant than Narrowleaf Birdsfoot Trefoil.

9 Tolerance given is an average for Boer, Wilman, Sand and Weeping Lovegrass; Lehman Lovegrass seems about 50 percent more tolerant.

10 These data are applicable when rootstocks are used that do not accumulate Na^+ and Cl^- rapidly or when these ions do not predominate in the soil. If either ions do, refer to the toxicity discussion in Section 4.

11 Tolerance evaluation is based on tree growth and not on yield.

Crop Tolerance

TOLERANT[3]

Fibre, Seed and Sugar Crops

Barley	*Hordeum vulgare*
Cotton	*Gossypium hirsutum*
Jojoba	*Simmondsia chinensis*
Sugarbeet	*Beta vulgaris*

Grasses and Forage Crops

Alkali grass, Nuttall	*Puccinellia airoides*
Alkali sacaton	*Sporobolus airoides*
Bermuda grass	*Cynodon dactylon*
Kallar grass	*Diplachne fusca*
Saltgrass, desert	*Distichlis stricta*
Wheatgrass, fairway crested	*Agropyron cristatum*
Wheatgrass, tall	*Agropyron elongatum*
Wildrye, Altai	*Elymus angustus*
Wildrye, Russian	*Elymus junceus*

Vegetable Crops

Asparagus	*Asparagus officinalis*

Fruit and Nut Crops

Date palm	*Phoenix dactylifera*

MODERATELY TOLERANT[3]

Fibre, Seed and Sugar Crops

Cowpea	*Vigna unguiculata*
Oats	*Avena sativa*
Rye	*Secale cereale*
Safflower	*Carthamus tinctorius*
Sorghum	*Sorghum bicolor*
Soybean	*Glycine max*
Triticale	*X Triticosecale*
Wheat	*Triticum aestivum*
Wheat, Durum	*Triticum turgidum*

Grasses and Forage Crops

Barley (forage)	*Hordeum vulgare*
Brome, mountain	*Bromus marginatus*
Canary grass, reed	*Phalaris, arundinacea*
Clover, Hubam	*Melilotus alba*
Clover, sweet	*Melilotus*
Fescue, meadow	*Festuca pratensis*
Fescue, tall	*Festuca elatior*
Harding grass	*Phalaris tuberosa*
Panic grass, blue	*Panicum antidotale*
Rape	*Brassica napus*
Rescue grass	*Bromus unioloides*
Rhodes grass	*Chloris gayana*
Ryegrass, Italian	*Lolium italicum multiflorum*
Ryegrass, perennial	*Lolium perenne*
Sudan grass	*Sorghum sudanense*
Trefoil, narrowleaf birdsfoot	*Lotus corniculatus tenuifolium*
Trefoil, broadleaf birdsfoot	*Lotus corniculatus arvenis*
Wheat (forage)	*Triticum aestivum*
Wheatgrass, standard crested	*Agropyron sibiricum*

MODERATELY TOLERANT

Grasses and Forage Crops

Wheatgrass, intermediate	*Agropyron intermedium*
Wheatgrass, slender	*Agropyron trachycaulum*
Wheatgrass, western	*Agropyron smithii*
Wildrye, beardless	*Elymus triticoides*
Wildrye, Canadian	*Elymus canadensis*

Vegetable Crops

Artichoke	*Helianthus tuberosus*
Beet, red	*Beta vulgaris*
Squash, zucchini	*Cucurbita pepo melopepo*

Fruit and Nut Crops

Fig	*Ficus carica*
Jujube	*Ziziphus jujuba*
Olive	*Olea europaea*
Papaya	*Carica papaya*
Pineapple	*Ananas comosus*
Pomegranate	*Punica granatum*

MODERATELY SENSITIVE[3]

Fibre, Seed and Sugar Crops

Broadbean	*Vicia faba*
Castorbean	*Ricinus communis*
Maize	*Zea mays*
Flax	*Linum usitatissimum*
Millet, foxtail	*Setaria italica*
Groundnut/Peanut	*Arachis hypogaea*
Rice, paddy	*Oryza sativa*
Sugarcane	*Saccarum officinarum*
Sunflower	*Helianthus annuus*

Grasses and Forage Crops

Alfalfa	*Medicago sativa*
Bentgrass	*Agrostis stolonifera palustris*
Bluestem, Angleton	*Dichanthium aristatum*
Brome, smooth	*Bromus inermis*
Buffelgrass	*Cenchrus ciliaris*
Burnet	*Poterium sanguisorba*
Clover, alsike	*Trifolium hydridum*
Clover, Berseem	*Trifolium alexandrinum*
Clover, ladino	*Trifolium repens*
Clover, red	*Trifolium pratense*
Clover, strawberry	*Trifolium fragiferum*
Clover, white Dutch	*Trifolium repens*
Corn (forage) (maize)	*Zea mays*
Cowpea (forage)	*Vigna unguiculata*
Dallis grass	*Paspalum dilatatum*
Foxtail, meadow	*Alopecurus pratensis*
Grama, blue	*Bouteloua gracilis*
Lovegrass	*Eragrostis sp.*
Milkvetch, Cicer	*Astragalus cicer*
Oatgrass, tall	*Arrhenatherum, Danthonia*
Oats (forage)	*Avena sativa*

MODERATELY SENSITIVE	
Grasses and Forage Crops	
Orchard grass	*Dactylis glomerata*
Rye (forage)	*Secale cereale*
Sesbania	*Sesbania exaltata*
Siratro	*Macroptilium atropurpureum*
Sphaerophysa	*Sphaerophysa salsula*
Timothy	*Phleum pratense*
Trefoil, big	*Lotus uliginosus*
Vetch, common	*Vicia angustifolia*
Vegetable Crops	
Broccoli	*Brassica oleracea botrytis*
Brussels sprouts	*B. oleracea gemmifera*
Cabbage	*B. oleracea capitata*
Cauliflower	*B. oleracea botrytis*
Celery	*Apium graveolens*
Corn, sweet	*Zea mays*
Cucumber	*Cucumis sativus*
Eggplant	*Solanum melongena esculentum*
Kale	*Brassica oleracea acephala*
Kohlrabi	*B. oleracea gongylode*
Lettuce	*Latuca sativa*
Muskmelon	*Cucumis melo*
Pepper	*Capsicum annuum*
Potato	*Solanum tuberosum*
Pumpkin	*Cucurbita peop pepo*
Radish	*Raphanus sativus*
Spinach	*Spinacia oleracea*
Squash, scallop	*Cucurbita pepo melopepo*
Sweet potato	*Ipomoea batatas*
Tomato	*Lycopersicon lycopersicum*
Turnip	*Brassica rapa*
Watermelon	*Citrullus lanatus*
Fruit and Nut Crops	
Grape	*Vitis sp.*

SENSITIVE[3]	
Fibre, Seed and Sugar Crops	
Bean	*Phaseolus vulgaris*
Guayule	*Parthenium argentatum*
Sesame	*Sesamum indicum*
Vegetable Crops	
Bean	*Phaseolus vulgaris*
Carrot	*Daucus carota*
Okra	*Abelmoschus esculentus*
Onion	*Allium cepa*
Parsnip	*Pastinaca sativa*
Fruit and Nut Crops	
Almond	*Prunus dulcis*
Apple	*Malus sylvestris*
Apricot	*Prunus armeniaca*
Avocado	*Persea americana*
Blackberry	*Rubus sp.*
Boysenberry	*Rubus ursinus*
Cherimoya	*Annona cherimola*
Cherry, sweet	*Prunus avium*
Cherry, sand	*Prunus besseyi*
Currant	*Ribes sp.*
Gooseberry	*Ribes sp.*
Grapefruit	*Citrus paradisi*
Lemon	*Citrus limon*
Lime	*Citrus aurantiifolia*
Loquat	*Eriobotrya japonica*
Mango	*Mangifera indica*
Orange	*Citrus sinensis*
Passion fruit	*Passiflora edulis*
Peach	*Prunus persica*
Pear	*Pyrus communis*
Persimmon	*Diospyros virginiana*
Plum: Prume	*Prunus domestica*
Pummelo	*Citrus maxima*
Raspberry	*Rubus idaeus*
Rose apple	*Syzgium jambos*
Sapote, white	*Casimiroa edulis*
Strawberry	*Fragaria sp.*
Tangerine	*Citrus reticulata*

[1] Data taken from Maas (1984).

[2] These data serve only as a guide to the relative tolerances among crops. Absolute tolerances vary with climate, soil conditions and cultural practices.

[3] The relative tolerance ratings are defined by the boundaries in Figure 10. Detailed tolerances can be found in Table 4 and Maas (1984).

Source: Ayers and Westcot (1985).