METHODS DEVELOPMENT FOR ENVIRONMENTAL
CONTROL BENEFITS ASSESSMENT

Volume VII

METHODS DEVELOPMENT FOR ASSESSING ACID
DEPOSITION CONTROL BENEFITS

by

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Volume 6, The Value of Air Pollution Damages to Agricultural Activities in
   Southern California, EPA-230-12-85-024.

     This volume contains three papers that address the economic implications
of air pollution-induced output, input pricing, cropping, and location pat-
tern adjustments for Southern California agriculture. The first paper esti-
mates the economic losses to fourteen highly valued vegetable and field crops
due to pollution. The second estimates earnings losses to field workers ex-
posed to oxidants. The last uses an econometric model to measure the reduction
of econanic surpluses in Southern California due to oxidants.

Volume 8, The Benefits of Preserving Visibility in the National Parklands of the
   Southwest, EPA-230-12-85-026.

This volume examines the willingness-t-pay responses of individuals surveyed in
several U.S. cities for visibility improvements or preservation in several Nation-
al Parks. The respondents were asked to state their willlingness to pay in the
form of higher utility bills to prevent visibility deterioration. The sampled
responses were extrapolated to the entire U.S. to estimate the national benefits
of visibility preservation.

Volume 9, Evaluation of Decision Models for Environmental Management, EPA-230-
   12-85-027.

This volume discusses how EPA can use decision models to achieve the proper role
of the government in a market economy. The report recanmends three models useful
for environmental management with a focus on those that allow for a consideration
of all tradeoffs.

Volume 10, Executive Summary, EPA-230-12-85-028.

     This volume summarizes the methodological and empirical findings of the series.
The concensus of the empirical reports is the benefits of air pollution control ap-
pear to be sufficient to warrant current ambient air quality standards. The report
 indicates the greatest proportion of benefits fran control resides, not in health
benefits, but in aesthetic improvements, maintenance of the ecosystem for recreation,
and the reduction of danages to artifacts and materials.

PREFACE

. . . .

Many individuals have made useful contributions to the preparation of this report. Drs. Alan Carlin and Dennis Tirpak of the USEPA were particularly helpful in setting bounds on the nature of the problem during the early stages of the research. A meeting in October 1979, with the members of the Committee on Biological Effects of the National Atmospheric Deposition Program was also helpful in this respect. Michael Marcus of the Department of Zoology at the [University of Wyoming has provided written material which has been used as the basis for several lengthy passages in Chapters II and V. Intermittent conversations with and written commentaries from Dennis Knight, William Schulze, and Harold Bergman have been instrumental in shaping some of what appears in the following pages. None bears any responsibility for any errors or omissions. Reza Sepassi has contributed in numerous ways as the primary research assistant for the research effort. Finally, Carol Steadman has employed her multiple talents to provide worthy research, editing, bookkeeping, organizational, and typing services throughout the project.

ABSTRACT

     There has recently been increasing awareness that some environmental
pollutants, because of the broad geographical scope of their effects, impose
not only the direct affronts to human life and property of the traditional
urban pollutants, but also attack the pleasures and the life support services
that the earth's ecosystem scaffolding can provide. Acid precipitation might
be one of these pollutants. The basic purpose of this report is to suggest
those types of natural science research that would be most helpful to the
economist faced with the task of assessing the economic benefits of
controlling acid precipitation. However, while trying to formulate these
suggestions, inadequacies in the supporting material the ecologist could offer
the economist, and in what the economist could do with whatever the ecologist
offered him, became apparent. Therefore part of our effort has been devoted
to initial development of a resource allocation process framework for
explaining the behavior of ecosystems that can be integrated into a broadened
benefit-cost analysis which captures traditional ecological concerns about
ecosystem diversity and stability. Our intent has been to make a start at
providing a basis for the ecological and the economic disciplines to ask
better-defined questions of each other.

     Some reasonably well-defined questions have nevertheless been asked and
tentative answers have been provided for a few of them. In particular, most
of the existing techniques for assessing the benefits of pollution control
require knowledge of the <u>magnitude</u> of the response of the entity of interest
to variations in the quantity of pollution to which it is exposed. The entity
that is the object of interest in these estimates of response surfaces or
functions must itself have value to humans or it must contribute in some <u>known</u>
fashion to another entity having value to humans. Otherwise, the economist is
unable to perform his tasks. Additional properties that response surface
research must have to be most valuable for the empirical implementation of the
techniques of benefit-cost analysis are outlined in the text.

     The simplest of these available techniques is applied in a first exercise
at using known response surfaces to assess the benefits of controlling acid
precipitation in Minnesota and the states east of the Mississippi River.
Current annual benefits of control are estimated to be \$5 x 10^9 in 1978
dollars, with materials damages constituting the largest portion of these

benefits. The reader must not treat this estimate as definitive, although the ordering of current annual control benefits by sector is highly plausible.

The known response surfaces used to construct the above estimate sometimes displayed two properties that could impart "all-or-nothing" and "now-or-never" features to the acid precipitation control decision problem, These two features arise because the marginal benefits of reducing acid precipitation appear to be increasing over a substantial interval of increasing pH values, and because the effects of acid precipitation upon ecosystem buffering capacities are less than fully reversible, both technically and economically.

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I. INTRODUCTION

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It is now widely accepted that the average pH of the annual precipitation
in nearly the entire United States east of the Mississippi River was below 5.0
in 1972-73 [Glass (1978, pp. vii, 19)].   Only northern Wisconsin and southern
Florida were exempted.   Since 1972-73, no increase in rainfall pH is believed
to have occurred.   With the likely increased combustion of coal in Canada and
the United States, most commentators expect further reductions in pH levels
and a further spreading of the geographical areas subjected to acid
precipitation and acidifying depositions.   This expectation persists even
though doubts have been publicly expressed about whether some of the
instrumentation used to measure precipitation acidity is accurate [Galloway,
et al. (1.979)] , and whether current measures actually represent a decline from
historical pH levels [Perhac (1979)].

Substantial concern has been expressed in both scientific and lay circles
about the impacts of increasingly acidic precipitation upon the flows of
material resources and amenity and life support services provided by forest
and aquatic ecosystems.   Because of these potential impacts, policymakers in
the U.S. and Canada are now being asked to weigh the benefits provided by
these resources and services against the costs of controlling emissions of
acid precursors from fossil fuel combustion.   Allied with these concerns are
numerous proposals for more research on the biological and economic effects of
acid precipitation.   In this report we attempt to provide policymakers with
some of the information they need to choose intelligently from among these
proposals and to prepare adequately for the findings of whatever research
programs are ultimately adopted.   Although researchers have made considerable
progress in identifying those features of different ecosystems that render
their economically valuable components and processes more-or-less vulnerable
to disruption as a consequence on long-term acid precipitation, the goal of
providing consistently dependable guidance to policymakers has not yet been
reached.

Toward this end, we have, after this introduction, structured this report
in four chapters.   The next chapter provides an economist's review of the
existing literature on the biological and physical effects of acid
precipitation.   The overview content is combined with limited information on
the market values of the affected material resources and amenity and life

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support services to arrive at no better than order-of-ma.gnitude assessments of
the current annual economic losses to existing activities caused by acid
precipitation in the eastern United States (Minnesota and the states east of
the Mississippi River). The emphasis in this second chapter is on identifying
the economic sectors that appear to be suffering the greatest damages from
acid precipitation. Only the simplest of economic methods are used to perform
this first exercise in assessment. A third chapter raises two plausible
special features, nonconvexities and irreversibilities, of the ecosystem
effects of acid precipitation that are likely to cause special. difficulties
for control decisionmaking as well as difficulties for the application of both
the simple and more sophisticated methods of assessing the economic benefits
of control. In a fourth chapter, we present a somewhat broader framework for
assessing the economic benefits of control than the framework that underlies
traditional assessment methods: we provide a start in the development of a
framework which, in principle, allows one to assess the economic impact of
pollution or anv source of stress upon ecosystem yields and ecosystem
diversity. This framework has been developed because of the inattention given
by traditional economic assessment procedures to questions of fundamental
concern to ecologists, and because of our perceived lack of an
ecological-theoretical. framework which could guide the questions the economist
asks of the ecologist. Finally, while drawing upon the information generated
in the previous parts, we develop and try to defend a set of recommendations
for natural science research on the biological effects of acid precipitation.
Our recommendations assume that without exception all natural science research
into these effects is directed toward the provision of information for assess-
ing the economic benefits of acid precipitation control. This last chapter is
the culmination of our current efforts. The reader should therefore view the
report not as an assessment of the economic benefits of specific control
alternatives but rather as a prelude to that assessment.

The Tasks of the Economist

     We divide into six tasks the role of the economist in providing decision-
makers with information to assess the benefits of controlling acid precipita-
tion. Since attempts to treat these tasks, within the limits of research time
and resources, compose the bulk of this report, we offer only the briefest
treatment here: :
     1.) To enumerate a set of economic indicators capable of communicating
national. and regional. economic benefits of alternative types and degrees of
control of acid precipitation.
     2) To identify those features of acid precipitation that when altered
have direct implications for the aforementioned indicators. These features
may affect directlv the components of ecosystems and the economic activities
that depend upon them. Alternatively, they may alter the behavior of these
components, resulting in changes in ecosystem processes and the economic

activities which employ them.   An example of a direct effect is a reduction in the yield of a vegetable due to acid precipitation-induced inhibition of photosynthesis in the standing stock of vegetable plants.   An indirect effect might consist of the changes in successional Patterns of a forest due to the differential effects of acid precipitation upon particular tree, understory, and soil microbe species.

3) To identify and, where appropriate, develop a theoretical framework for assessing the potential national and regional economic benefits of alternative acid precipitation control strategies.   This framework should generate refutable hypotheses about the causal relationships between the features of various control strategies and the responses in economic terms of relevant ecosystem processes and components.   In short, the framework should make easier the appropriate specification and estimation of the economic and ecosystem parameters needed to explain and to make predictions of the magnitudes and the timing of the potential benefits of alternative control strategies.

4) To identify the data required to estimate the aforementioned parameters.   The data requirements should be as parsimonious as the theoretical framework will allow.

5) Given the current state-of-the-world, to estimate the current values of the relevant economic and ecosystem parameters, while employing properly constructed variables, applicable statistical and numerical tools, and an appropriate sample of ecosystems.

6) To incorporate the estimated parameters into a body of knowledge that will predict the values of the economic indicators resulting from adoption of alternative acid precipitation control strategies.

Generally speaking, each of these tasks is served by an analytical framework or model encompassing a greater range of phenomena than did previous models.   Of the six tasks, however, the third and the fourth are most likely to be of greatest relative interest to the professional researcher, while the other four tasks assume greatest relative importance for the decisionmaker. In those parts of economics relevant to the assessment of the benefits of air pollution control, there has frequently been inadequate attention by analytical investigators to possibilities for improved empirical implementation. Analytical investigators have on occasion indulged in illicit intercourse with beautiful models, as at least one economist has remarked. On the other hand, economists having some interest in empirical implementation have occasionally been too ready to indulge requests to generate estimates of the benefits of air pollution control.   From some perspectives, this report might accomplish the unusual act of being culpable on both counts.   The second chapter of the report   engages in an empirical exercise that is not solidly embedded in a theoretical framework.   The fourth chapter goes through a theoretical exercise which could be empirically implemented.   Nevertheless, in this report any beauty it has must be judged as an abstraction; it is provided

little empirical flesh. Only the third chapter makes a limited attempt to clothe the abstract in the empirical. We nevertheless feel that these rather disparate chapters do result in a set of natural science research recommendations, that when accomplished, are likely to be useful and inputs for assessing the economic benefits of alternative acid precipitation control strategies.

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A Dynamic Economic Sketch of the Ecosystem Effects of Acid Precipitation

In order to frame our discussion, we present in this section a model which outlines the economic nature of the problem of preventing ecosystem damages from acid precipitation. As will be near-universal throughout this report, knowledge of the dose-response function relating ecosystem effects to acid precipitation is central to any empirical application of the model.

Assume an industrial region, I, that generates a constant waste flow, $\bar{W}$, per time period. Some of these wastes are carried and transformed by atmospheric processes to a lake region, L. The waste that travels the distance, x^L, from I to L each period is given by:

$$W(x^L) = \frac{W}{Hu} - \int_0^{x^L} \mathcal{L}(x)dx, \tag{1}$$

where H is the mixing or scavenging height of the air column and u is the wind speed. H and u are assumed constant over $[o,x^L]$. $\bar{W}/Hu = \bar{W}/k$ is then the initial pollution concentration at I. $\mathcal{L}(x)$ is a pollution loss or trans-formation function which is assumed constant over distance. [1/] Thus

$$W(x^L) = \bar{W}/k - \bar{\mathcal{L}}x^L \tag{2}$$

is the waste concentration arriving each period at L as a result of $\bar{W}$ being generated in I.

Atmospheric processes cause the waste to be deposited and accumulated in L as a stock of pollution, P. This accumulation is:

$$\frac{dP}{dt} = g(W(x^L)) - \alpha P, \tag{3}$$

where $g(\cdot)$ measures the waste concentration in the lake region, and αP measures the abilities of the region's forest and aquatic ecosystems, R, to cleanse themselves of the pollutant. [2/] We assume that α is constant and independent of pollution. Forster (1975) discusses a model in which a is a decreasing function of P.

The dynamic evolution of R is governed by a pollution version of the Lotka (1925) biological growth function:

$$\frac{dR}{dt} = F(R,P), \tag{4}$$

where, for a given P, the F function has the usual Lotka shape. Increases in P will shift the entire curve downward in Figure 1. The environmental carrying capacity, $\bar{R}$, is thus an inverse function of the level of pollution. That is:

$$\bar{R} = \bar{R}(P); \quad \bar{R}' < 0 \tag{5}$$

Expression (5) is an example of what is commonly called a dose-response function. The loss in $\bar{R}$ may be thought of as the ecosystem damages caused by a change in the pollution level. Its critical importance to system behavior and thus human welfare can be illustrated by introducing a harvesting function relating man's harvest, H, from the system to his harvesting effort, a, and the size of the lake region's forest and aquatic ecosystem resources.

$$H = aR \tag{6}$$

For a given level of effort, the harvest will be larger if the resources are more plentiful. Using (6) and the previous expressions, the dynamic structure of the lake region ecosystem is governed by:

$$\frac{dP}{dt} = g\left(\bar{W}/k - \bar{L}x^{L}\right) - \alpha P. \tag{7}$$

$$\frac{dR}{dt} = F(R,P) - aR. \tag{8}$$

The limiting solution for pollution, PB, depends upon meteorological factors, the level of waste emissions in I, and the self-cleansing abilities of the lake region's ecosystems:

$$P\infty = \frac{1}{\alpha}\, g\left(\bar{W}/k - \bar{L}x^{L}\right). \tag{9}$$

This solution, which is globally stable, can be substituted into (9) to examine the limiting solution for R. The result of doing so is illustrated in Figure 2.

In Figure 2, pollution reduces the growth rate of the lake region's resources and thereby reduces the region's environmental carrying capacity from $\bar{R}(0)$ to $\bar{R}(P\infty)$. With a given level of harvesting effort, the bioeconomic equilibrium stock size is reduced from R* to $R\infty$ and the equilibrium harvest suffers a decline from H* to $H\infty$. The equilibrium resource stock size is

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Figure 1.1

Evolution of the Resource Stock
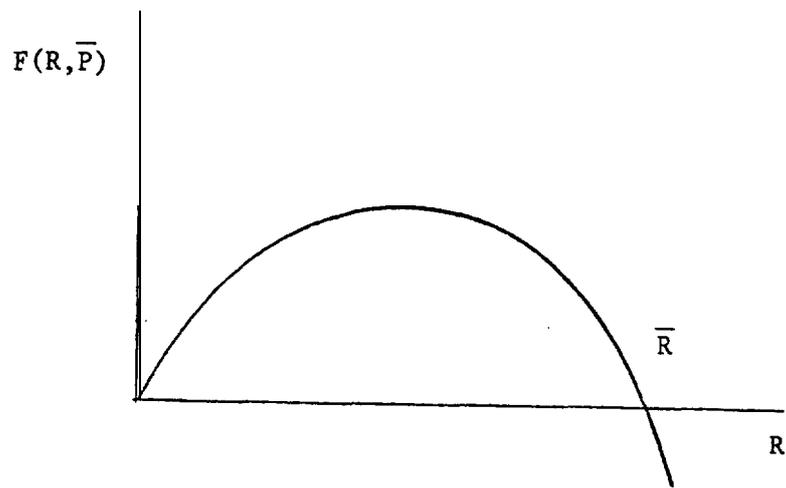
Figure 1.2

Equilibrium Pollution and Resource Stock
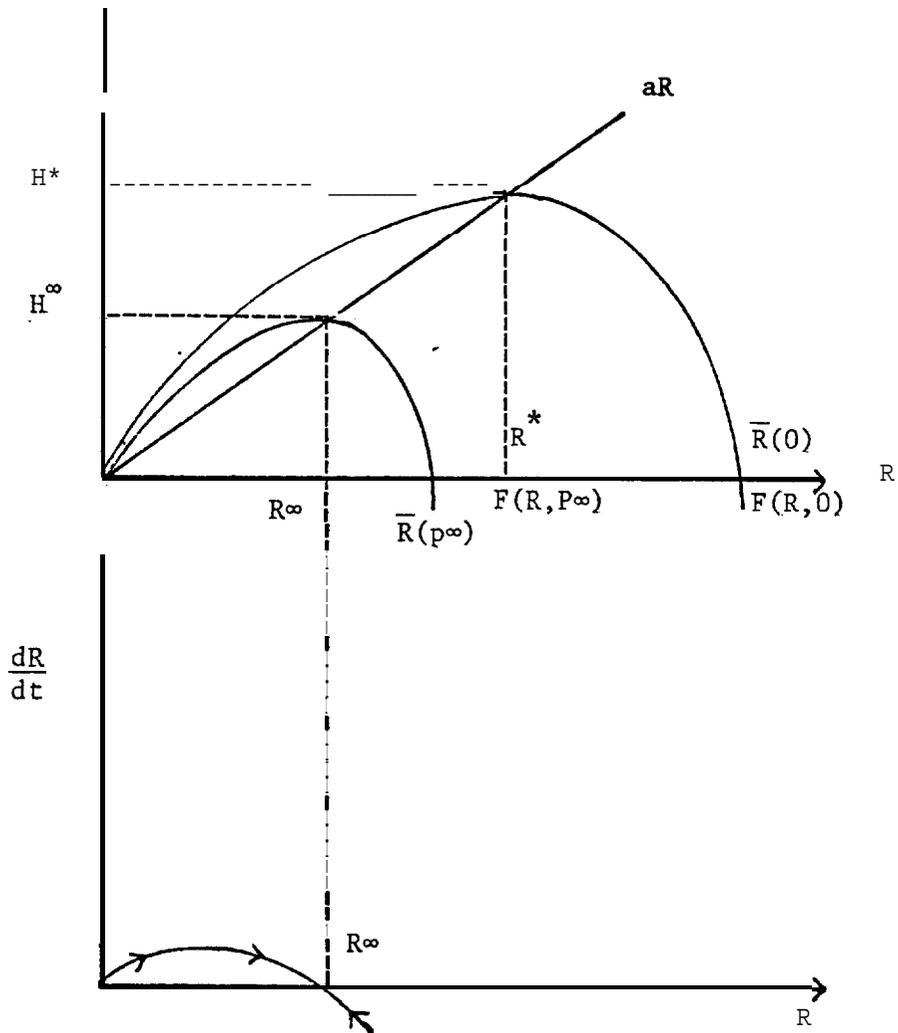
stable.   Efforts to enhance the resource base by restocking fish or fert-
ilizing forest soils may offer temporary respites by raising the resource
stock above $R\infty$.   However, with P continuing at $p\infty$, the stock must over time
decline again to $R\infty$.

The Meaning of Economic Benefits

Everything said in this report unequivocally assumes that man is the
measure of all things.   As Adams and Crocker (forthcoming) point out, whatever
a person does must be the best thing for him to do, given his knowledge of his
circumstances of the moment--otherwise, he would not do it: thus the person's
autonomous preferences are revealed by his behavior.   This is the perspective
of value that pervades economic analysis.   Contrary, however, to much common
usage, "economics" and "pecuniary" are not viewed as synonymous.   For example,
human behavior and the health, production, or aesthetic effects of a pollutant
on that behavior are directly "economic."   The effects of a pollutant on
vegetation are "economic" only insofar as that vegetation contributes to human
health and happiness.

The preceding perhaps conveys the stance of economics with respect to the
basis of values.   It fails, however, to state the units in which values are to
be measured or the context that bestows meaning on these units.   Assume, for
example, that a person derives satisfaction from an aesthetic phenomenon, such
as lush vegetation.   Tf there is a local decline in the lushness of
vegetation, the person will possibly feel he has been made worse off,
However, if there are other worldly things capable of providing him
satisfaction, then some additional provision of these other things may cause
him to feel as well off as he would without the decline in vegetation
lushness.   Finally, if these things can be secured by the expenditure of
income, or time that can be used to earn income, then there is some additional
income that in the face of the lushness decline, would make the person feel no
worse off.   The unit, therefore, in which economics would have us measure
value is money stated in terms of income.   Implicit in the acceptance of this
unit is the presumption that, even if the thing being valued cannot be secured
in the marketplace, there are in this marketplace collections of other things
from which the person receives equal satisfaction.   These other things, which
have market prices attached, can under a quite wide range of well-specified
conditions, serve as vehicles to infer the "values" of entities and services
for which no directly observable pecuniary prices exist.

In spite of the common sense approach to valuation sketched above, it
will often yield, depending on the conditions adopted for the analysis,
different values for the same quantity variation in the entity being valued.
For example, if one is interested in the control of a pollutant that is
damaging vegetation, the value that a person will attach to the reduction of

the pollutant can depend on whether one is measuring what the person is willing to pay for the reduction or what the person would have to be paid in order not to have the reduction. In the latter case, because the person is viewed as holding the legal right to stop the pollution, his revelation of preferences is not limited by his income. However, his income does limit what he can do when he must buy a cessation of pollution from someone else. As his money becomes scarce, he becomes reluctant to trade money for goods. Thus, the two measures would be identical only when variations in income play a trivial role in determining the quantity of the good that the person will choose to hold.

Other sources of variations in values of identical changes in the quantity of a particular good include whether, in an original and in a new state, the original is the most preferred or the least preferred quantity; whether the valuation in the new state is independent of adjustments in overall patterns of consumption in moving from the original quantity of the good to the new quantity; and whether the person can by his own actions adjust his consumption of the good in question or, as with many pollutants, must become resigned to an externally imposed fate. In short, to be meaningful and communicable, the exact context of a particular economic valuation measure must be explicitly and fully stated. The criteria for judging which of the several analytically correct valuation measures to apply to a particular real problem must often come from outside economics.

Benefits Assessment Methodologies

Schulze, et al. (forthcoming) provide an informative and succinct common theoretical basis for the alternative economic methodologies available to assess the benefits o   acid precipitation and other plausible environmental insults. ~/controlling They start their analysis with the recognition that all assessment methodologies presume that there exist marketplace collections of things other than the entity being valued from which the representative individual could receive equal satisfaction. These substitution possibilities are said to exist across alternative activities and locations, both of which are denoted $A_1, \ldots, A_i, \ldots, A_n$. Each of these activities and/or locations is associated with a particular level of environmental quality, $Q_1, \ldots, Q_i, \ldots, Q_n$. Increases in the Q_i represent environmental quality improvements.

The individual's weakly separable, quasi-concave utility function is written as:

$$U(A_i, Q_i, X), \qquad\qquad (10)$$

where X is a composite commodity the magnitude of which is unaffected by A_i and Q_i. Utility is assumed to be increasing in A_i, Q_i, and X. The

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individual's decision problem is then to maximize (1) subject to a budget constraint:[4/]

$$Y - \sum_{i=1}^{n} P_i A_i - X = 0,$$ (11)

where Y is current period income, P_i is the price of the ith activity, and X is assumed to have a price of unity. The necessary conditions for solution of the problem include

$$\frac{\partial U/\partial A_i}{\partial U/\partial X} \leq P_i, \text{ and } \frac{\partial U/\partial A_i}{\partial U/\partial X} - P_i = 0,$$ (12)

assuming that A_i is consumed in some positive quantity. This says that the individual will equate the marginal rate of substitution of the ith activity for X to the price, P_i, of that activity.

To determine the marginal willingness-to-pay for the environmental quality associated with a particular activity, i=1, Schulze et al. set (10) equal to a constant and then totally differentiate this expression as well as expression (11). When $dA_i = 0$ for $i \neq 1$, and by using (12), they obtain:

$$\frac{dY}{dQ_1} = \sum_{i=1}^{n} A_i \frac{dP_i}{dQ_1} - \frac{\partial U/\partial Q_1}{\partial U/\partial X}$$ (13)

This represents the additional income that in the face of an environmental quality change would make the individual feel no worse off. Considering only the total differential of (11), while continuing to assume that $dQ_i = 0$ for $i \neq 1$, they obtain another expression for dY/dQ_1:

$$\frac{dY}{dQ_1} = \sum_{i=1}^{n} A_i \frac{dP_i}{dQ_1} + \sum_{i=1}^{n} P_i \frac{dA_i}{dQ_1} + \frac{dX}{dQ_1}$$ (14)

When one equates (13) and (14), and cancels similar terms, the result is:

$$\sum_{i=1}^{n} P_i \frac{dA_1}{dQ_1} + \frac{dX}{dQ_1} = \frac{-\partial U/\partial Q_1}{\partial U/\partial X} < 0$$ (15)

In short, the last two terms in (14) are negative.

Schulze, et al. suggest that (14) and (15) provide a common and easily

grasped basis for interpreting the substantive analytical content and the data requirements of alternative economic methodologies for assessing marginal willingnesses-to-pay for changes in environmental quality. Consider, for example, an air pollutant which reduces the yield of an agricultural crop. One expedient method to assess the economic value of a quality improvement is simply to ask individual producers and consumers what their magnitudes of dY/dQ_1 are. This approach, probably because of the ready availability of price, yield, and location data, has not to our knowledge vet been used to assess agricultural damages from air pollution. Under the label of "bidding games" or "contingent valuations" it has been widely used to value environmental quality improvements where there is little or no historical experience with the potential improvement and where directly observable price and quantity data are unavailable to either the researcher or the individual producer and consumer. These circumstances aptly describe many aesthetic and health effects of air pollution. Schulze et al. thoroughly review and evaluate several of the existing contingent valuation studies, and provide a listing of many more. Brookshire and Crocker (forthcoming) provide further discussion of the real-world circumstances under which contingent valuation approaches are especially appropriate. Although the natural science informational requirements of these methods might appear to be minimal or nonexistent, all commentaries insist that great care must be taken in describing the state-of-the-world to which the interviewee is to be asked to respond. Otherwise, biases can be introduced that make interviewee responses uninterpretable. Thus , although natural science information is not an integral part of the analytical exercise involved in contingent valuation methods, it does play an important role in establishing the scenario that is to be valued.

If agricultural settings have seen but infrequent application of the contingent valuation methods that capture the right-hand-side of (14), they have experienced numerous applications of methods that focus on no more than the middle term, $\Sigma P_i (dA_i/dQ_1)$ on its left-hand-side. Examples are Benedict, et al. (1973) and Millecan (1976) . When the P_i are readily observable, the role the economist need play is minimal; the role of the natural scientist dominates because, by assumption, only the activities change in response to changes in environmental quality. Thus the natural scientist must translate alternative air pollution states into changes in plant growth, and changes in this growth into changes in useful yield. Given that crops and crop varieties display different tolerances to pollution, numerous dose-response functions similar to those established for alfalfa by Oshima and his colleagues (1976) may be required. Having obtained these dose-response functions for the list of activities in question, the determination of dY/dQ_1 is a simple matter of multiplying the changes in yields by the observed or inferred market prices.

If the scope of the analysis extends beyond yield effects upon existing

cropping and location patterns, the role of the economist for evaluating $P_i(dA_i/dQ_1)$ need not be quite so limited as the previous paragraph implies. In particular, a change in pollution may make alternative cropping and location patterns more appealing. Economic contributions are then useful in specifying those among the set of feasible grower alternatives that are worthy of detailed investigation. Nevertheless, the core of the exercise remains the estimation by natural scientists of the yield responses of individual crops to pollution under a variety of environmental conditions and in a variety of locations.

Two terms remain on the right-hand-side of (5) that we have not yet discussed: $\Sigma A_i dP_i/dQ_1$, the change in the price of the ith activity due to a change in the environmental quality parameter; and dX/dQi, the change in expenditures on the composite commodity due to a change in the quality parameter. Here the relative importance of the roles of the natural scientist and the economist is reversed from the earlier discussion. Cropping and location patterns are treated as being utterly unresponsive to changes in the quality parameter. All adjustments to variations in the quality parameter are reflected in inferred or market prices alone. Thus, for example, as Johnson and Hough (1970) and Crocker (1971) have done, one might estimate dY/dQ_1 by holding the levels of all agricultural activities constant, the prices of all other commodities except land constant, and the magnitude of expenditures on other goods constant, and then estimate the effect of variations in the quality parameter upon the market prices of agricultural sites. In this extreme case, the only role of the natural scientist would be to identify the sites that are subjected to a variety of levels of pollution. If the number of activities whose price responsivenesses to pollution was of interest were to be expanded for study purposes, the natural scientist's role would continue to be limited to specifying the existing levels of these activities. Just as with contingent valuation methods, the natural scientist's expertise on the behavior of organisms under stress has no role to play.

The importance of considering these dP_i/dQ_1 and dX/dQ_1 terms is readily perceived by considering a simple analytical model of price determination frequently used by agricultural economists. Specifically, the equilibrium price of agricultural commodities, in the aggregate or individually, may be derived from the intersection of the relevant supply and demand curves. The effects of air pollution may be viewed as a supply phenomenon, shifting the supply curve. Given the generally inelastic demand for agricultural commodities, the supply-demand model indicates that shifts in the supply curve will translate into rather large shifts in the equilibrium price of food. Thus, following from the nature of the demand-supply relationships, one may hypothesize changes in commodity prices if air pollution affects the position of the supply curve.

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The significance of these price movements is that agricultural prices cannot necessarily be assumed to be static or stable. Further, changes in agricultural prices do not occur in isolation but rather work their way through the system, affecting the welfare of consumers, producers, input suppliers, resource owners, and other parties. For example, given the generally inelastic demand for agricultural commodities, reductions in supply may actually increase farmers total net revenue, as the attendant price rise may be greater than the percentage reduction in quantity supplied or produced. Conversely, the increase in prices from a supply reduction will reduce consumers' welfare. Thus, if air pollution alters yield of a substantial proportion of a given crop or causes a reduction in planted acreage of that crop, then the overall change in supply may result in changes in the price at the farm level which will ultimately be felt at the consumer level. Alternatively, if farmers employ mitigative measures to adjust for the presence of air pollution, then any additional costs of such measures may also affect consumers through shifts in supply caused by changes in producers' cost functions.

Fortunately, the alternative methods available to assess the benefits of controlling pollution such as acid precipitation are not limited only to those which ask hypothetical questions of supposedly knowledgeable interviewees, consider the activity effects but not the price effects, or consider the price effects but not the activity effects, of a pollution change. Consider the following quadratic programming model, with which Adams, et al. (1979) have recently assessed the economic impact of air pollution upon southern California agriculture, as an example of the ability of many economic methodologies to capture both the price and the activity effects of pollution-induced damages. Again, however, the viability of the methodology is utterly dependent upon the availability of accurate dose-response functions.

Assume that the effect of acid precipitation upon a set of annual agricultural crops in a number of regions is of concern. The markets for each of the included crops in each region operate so as to solve the following problem:

$$\text{Max:} \quad \pi = C^T Q + 1/2 Q^T D Q - H^T Q \tag{16}$$

$$\text{Subject to:} \quad AQ \leq b$$

$$Q \geq 0$$

The symmetric matrix D in the objective function is negative definite, and the constraints are convex. The terms of (16) are defined as follows.

A is a m x n matrix of production coefficients indicating the

invariant amount of each of a variety of inputs required to produce
any single unit of a particular output.

Q is a n x 1 column vector of crop outputs.

D is a m x m matrix representing slope values of the linear demand
structure for the fourteen included crops.

H is a n. x 1 column vector of invariant unit costs of production
for the included crops.

C is a n x 1 column vector of constants.

b is a m x 1. column vector of inputs.

As advocated by Harberger (1971.), π is the sum of ordinary consumer
surpluses and producer quasi-rents. The supply functions for all producer
inputs purchased in the current period (seeds, labor, fertilizer, etc.) can be
assumed to be perfectly price-elastic. In addition, one can invoke Willig's
(1976) results and presume any differences between ordinary and compensated
consumer surpluses to be trivial. Since neither income elasticizes nor
ordinary consumer surpluses or expenditures as a percentage of incomes are
likely to be large for most crops or other entities affected by acid
precipitation, this invocation seems reasonable.

The left-hand-side of the objective function in (16) can be stated in
terms of observable by introducing a price forecasting expression:

$$P = C + 1/2 \ DQ, \tag{17}$$

where P is a n x 1 vector of farm level crop prices. In matrix form, the
objective function may then be expressed as:

$$P^T Q \ - \ H^T Q = C^T Q + 1/2 \ Q^T DQ \ - \ H^T Q \tag{18}$$

In order to capture the impact of acid precipitation upon crop yields, we
define a variable. Z^* ($0 < Z^* < 1$) for each included crop in each region. The Q
terms in (16), (17), and (18) can then be stated as:

$$Q^* {}^= (1 - Z) \ L^T Y \tag{19}$$

where:

Q^* is a n x 1 column vector of yields of the n crops in the presence
of acid precipitation.

Z* is a n x 1 column vector of indicies of yield reduction for the n crops.

I is a n x 1 column vector of unity.

L is a n x 1 column vector of the land acreage used for cultivating the n crops. The total land area available for all crops can be assumed to be fixed.

Y is a n x 1 column vector of yields per acre of the n crops in the absence of acid precipitation.

Given L and Y constant, the value of Q* varies inversely with the value of z*. Thus regions with higher acid precipitation will have higher values of Z* and consequently lower values for Q*. The yield price effects of these reductions in O* are then predicted by (17), the price forecasting expression. Impacts of these predicted price changes upon consumer surpluses, producer quasi-rents, and cropping patterns within and across regions can then be calculated by solving the quadratic programming problem.

The immediately preceding formulation is meant to be illustrative of what economic analysis can do in assessing the benefits of controlling acid precipitation. It by no means exhausts the techniques that might be applied to the various aspects of the acid precipitation issue, although it is representative of the most robust and economically meaningful of the available techniques. With the sole exception of contingent valuation techniques which employ stated answers to hypothetical questions as data, all these techniques use observed decisionmaker behavior as data. The economic interpretation of these data on observed behavior is generally unable to proceed unless believable and useful dose-response functions can be provided by the natural scientist.[2/] In the last chapter, we shall have a great deal to say about what a dose-response function must include if it is to be useful to the economist. For the next two chapters, we try to employ the knowledge the natural scientist has thus far accumulated on dose-response functions to gain some insights into the economic benefits of controlling acid precipitation.

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1/   This is adapted from **Hamlen** (1978), bearing in mind that emissions
      are generated in I only.

2/   Note that this formulation deals with deposition of sulfur or NO_x,
      as such, rather than. acidity. The United States – Canada **Research**
      Consultation Group (undated, **p.11**) states that this is common to all
      models in the area.

3/   See Freeman (1979) and **Maler (1974)** for additional treatments grounded
      upon an internally consistent theoretical framework.

4/   Further generality can be easily obtained **by** introducing a time con-
      straint. At the **level of** abstraction used in this section, no additional
      insights would be gained by doing so.

5/   It should be mentioned that, at least in principle, the duality between
      cost and production (dose-response) functions that the envelope theorem
      provides means that the economist, without any dose-response data what-
      soever, can use data on observed behavior to perform analyses of the
      benefits of controlling acid precipitation. For a clear treatment of the
      envelope theorem, see **Silberberg** (1978, pp. 309-312). Most interestingly
      perhaps, the theorem implies that one could estimate dose-response functions
      using only data on cost function parameters. This would permit the
      services of the natural scientist to be dispensed with entirely! However,
      given the somewhat disturbing findings of **Appelbaum** (1978) and others on
      the empirical reality of this dualism, we prefer to refrain, for now,
      from stating that the research of the natural sciences into dose-response
      functions is irrelevant. Nevertheless, a careful inventory of practical
      opportunities for empirical applications of duality principles to the
      valuation of pollution impacts would be **worthwhile.**

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II. "A"FIRST EXERCISE IN ASSESSING THE BENEFITS
OF CONTROLLING ACID PRECIPITATION

Introduction

In this chapter we undertake a first exercise in assessing the economic impacts of acid precipitation or acidifying deposition, given that it occurs at above-background levels. The main benefit of the construction, from the authors' perspective, has been to serve as a learning and organizing device about the state of natural science knowledge of acid precipitation effects upon life and property. We have concluded that the state of this knowledge is very incomplete, both in terms of empirically testable propositions derived from a broadly encompassing analytical structure as well as in quantitative bits of information that have been related to or associated with each other. Under these circumstances, it is tempting for the economist to plead the near-impossibility of his task, as if the natural scientist were responsible for any failings of the economist's attempts to value the effects. On some occasions, the plea is valid. On this occasion, 'it is, on balance, invalid. The reasons are two.

First, whatever the available research time and resources, economic analysis generally does not yet know how to assess quantitatively disruptions in ecosystem functions having major and broad economic impacts. [Building upon Scarf's (1973) work, Shoven and Whalley (1977), King (1980), and a few others show that the truth of this statement could be short-lived]. If the impacts of acid precipitation upon ecosystem nutrient storage, detrital decay, succession patterns, genetic pools, etc., are as dire as some natural scientists predict, and if alterations in these functions can legitimately be viewed as changes in natura 1 transformation processes (production technologies), then a full economic assessment may be analogous to comparing the welfare of the 18th century Jeffersonian yeoman farmer with his modern agricultural corporate clone: the worlds in which the two live(d) are so vastly different that neither the modern nor the Jeffersonian man could comprehend most of the opportunities and dangers familiar to the other. It is questionable whether the comparison would be economically meaningful.

The second reason why the blame for the economic limitations of the content that follows cannot be shifted to the natural scientist is because, in principle, it is both possible and practical to value many, perhaps most, of the effects of acid precipitation. The task is substantial but nevertheless accomplishable. Economists who read this chapter will recognize that only the most elementary economic analysis has been performed. In particular, we have generally resorted to an assumption throughout that acid precipitation affects only the yields from existing sets of economic activities. We, therefore, have, with only a few exceptions, disregarded any potential price effects, as well as changes in activity and location patterns. Finally, we have nothing to say for now about the economic implications of larger questions on changes in lifestyle, possibly unacceptable risks due to the destruction of life support systems, and the welfare of future generations. Effects on these larger questions, as well as those involving changes in activity and location patterns, are most likely to be generated by the buffering stock depletion impacts of acid precipitation. It is these stock depletion effects that, as was indicated in Chapter 1, economically distinguish acid precipitation effects from traditional analyses of pollution effects. In a later theoretical chapter, we will view these stock depletion effects as analogous to drawdowns in the biogeochemical energy available to a geographic location. In this chapter, so as to remind the reader that we do not view effects on the yields of current economic activities as the sole effect of acid precipitation worthy of economic attention, we present a brief treatment of the impact of acid precipitation on the buffering capacities of natural ecosystems.

Depletion of the Stock of Buffering Capacity

After deposition, the effects of acidifying components depend on sensitivities of the environments where the deposition occurs. This sensitivity is largely determined by the abilities of the depositional surface to buffer hydrogen ion additions. In turn, environmental buffering abilities initially depend on the bedrock and geological history of the region. Bedrock of volcanic or igneous origin tend to be low in most minerals important in buffering [Dillion and Kirchner, (1975)]. Over geologic time, the importance of the bedrock is moderated by glaciation, weathering and other soil building processes.

Buffering in soil systems is primarily accomplished through cation exchanges between soil solutions and colloidal clay and humus particles, also called micelles [Buckman and Brady, (1960)]. These micelles are negatively charged and, therefore, attract positively charged cations which enter soil solutions during soil mineralization, for example. Cations which are commonly adsorbed onto micelles in order of increasing affinity for micelle adsorption are: sodium, potassium, magnesium, calcium, aluminum and hydrogen. Increasing concentrations of hydrogen ions in soil solutions, as would occur with

incident acidifying depositions, shift the equilibrium between the soil solution and the micelles such that additional hydrogen ions become adsorbed onto micelles, thus displacing cations having less affinity. Adding lime to soils causes concentrations of hydrogen. ions in soil solutions to decrease as they react with the calcium ions. This shifts the equilibrium between soil solution and micelles: hydrogen ions desorb and calcium ions adsorb onto micelles.

The second major environmental buffering system, the primary buffering mechanism for aquatic environments, is the carbonate-bicarbonate system. While this system is also present in soil environments, it is generally of relatively minor importance [Buckman and Brady, (1960)]. In aquatic environments, the abilities of the carbonate-bicarbonate buffering system are normally measured by alkalinity determinations [Sawyer and McCarty, (1967)]. Buffering capabilities develop as carbon dioxide dissolves in water. The carbon dioxide reacts with water to form carbonic acid which dissociates to form bicarbonates and hydrogen ions. Bicarbonates can then further dissociate to carbonate and hydrogen ions. Additions of hydrogen ions to the aquatic environment will reverse this process and, with continued additions, carbon dioxide can eventually be released from the water.

Continued deposition of acidifying substances in ecosystems will cause buffering capacities to decrease as exchangeable ions, carbonates, bicarbonates and, eventually, pH values decrease. Simultaneously, titratable acidity, hydrogen ion concentrations and associated anions such as sulfates and nitrates increase. Also, mineralization rates of soil particles will increase with hydrogen ion additions and counter, at least for short periods, the effects of acidification [Maimer, (1.976)].

The frequencies and durations of acidifying depositional events will affect the severity and rate of ecosystem changes. However, as eposodic acid contributions continue, a system's buffering capacity is continually reduced and its hydrogen ion concentration increased. As the pH continues to decrease, the impacts on the ecosystem increase and the importance of the episodic frequency of the acidifying contributions in defining ecosystem impacts decreases.

Sufficient additions of acidic water to soil systems can cause the cation released through ion exchange buffering to leach from the system. The decreased pH can have additional effects on soil chemistry [Buchman and Brady, (1960), Maimer, (1976)]. As soil pH decreases below 8.0, the amounts of aluminum, iron, and manganese increase in soil solutions. At low pH levels, concentrations of these compounds can become toxic; and as concentrations of these dissolved compounds increase, they can react to fix phosphates as insoluble hydroxyl-phosphates. In such a complex this valuable nutrient is

not available for plant use.

Acidifying depositions enter aquatic ecosystems directly through surface deposition and indirectly through watershed runoff. Increases in the acidity of rivers and lakes occurs with acid precipitation events. But, depending on the buffering ability of the water, such changes may be relatively small and short in duration. More substantial acidity increases of longer duration can accompany spring snowmelt. These changes can be particularly dramatic when the snowpack melts rapidly [Likens et al., (1977)]. At such times, the pH of surface waters can decline to about 3.0 [Shaw, (1979)]. Gjessing et al. (1976) noted that when spring meltwaters are less dense than lake waters (water has a maximum density at $4^{o}C$), the meltwaters tend to flow over the surface waters of the lake and therefore do not mix with lake waters. During such times, waters having elevated acidities are discharged from lakes to produce maximum impacts on stream ecosystems. Runoff to lakes at other times of the vear generally mixes with lake waters, lessening the impacts on streams.

The influences of acidifying depositions on the chemistries of natural waters are similar to the effects produced on soils. Increasing the hydrogen ion concentration reduces the aquatic system's buffering abilities, increases solubilities of metals, complexes phosphates, etc. Continued addition of hydrogen ions causes carbonates and bicarbonates to be converted to carbon dioxide and water [Lewis and Grant, (1979)]. Carbon dioxide can then be lost to the atmosphere or dissolved concentrations can accumulate to levels which are directly lethal to aquatic organisms [EIFAC, (1969)]. Precipitation of normal suspended silt loads has also been noted for streams having increased hydrogen ion loading rates [Parsons, (1965)].

Studies of effects from acidifying depositions have shown the importance of buffering capacities in determining the constituents flushed out of water-sheds. Often the input of hydrogen ions is adsorbed by the watershed eco-system and no significant increase in hydrogen ion or other cation concentra-tions is observed in aquatic system outputs (e.g., Lewis and Grant, 1979). In other cases, the uptake of hydrogen ions by the watershed results in an increase in the output of other cations from the watershed. For example, Gjessing et al. (1976) report that outputs of calcium, magnesium, and aluminum ions from nine watersheds in Norway were proportional to inputs of hydrogen ions. In contrast, Lewis and Grant (1979) noted no significant change in outputs of calcium, magnesium, sodium, potassium, phosphate, or hydrogen ions accompanying increased hydrogen ion inputs to a Colorado, USA, watershed. Increased outputs were observed for sulfate, nitrate, ammonia and dissolved organic matter, while decreased outputs of bicarbonate were proportional to increased hydrogen ion inputs. Variations in output responses among the Norway and Colorado watersheds suggest that different buffering systems are

responding to the hydrogen ion inputs. The Norway watersheds appear to buffer primarily through cation exchange, primarily a terrestrial system, while the Colorado watershed appears to buffer primarily through reactions with bicarbonate. As both the Norway and Colorado watersheds are of granitic origin, variations in the buffering systems may reflect the relatively longer time which Norway has been exposed to acidifying depositions. With continued additions of acids, bicarbonate buffering systems become exhausted causing buffering to become increasingly dependent on cation exchange.

Knowledge of buffering capacities is helpful in sorting out habitat impacts of hydrogen ion inputs from impacts of other chemical constituents associated with acidifying depositions. Taken together, variabilities in depositional composition and mode as well as receptor buffering can be used to discriminate among these effects. In poorly buffered systems, depositional impacts tend to be more related to pH effects because of the low masses of SO_x and NO_x required to generate pH changes. Tn well buffered systems, the masses of SO_x 'and NO_x necessary to generate pH changes are relatively larger and, consequently, influences of other compounds tend to be enhanced. Thus, well buffered systems tend to respond more to components other than hydrogen ions in both wet and dry acidifying depositions; this relationship will tend to be maintained until the buffering capacity of the system is exhausted. With poorly buffered terrestrial systems, wet depositions will tend to have confounded responses to both hydrogen ion concentrations and other depositional contents until pH effects overwhelm other responses. Dry depositions to poorly buffered terrestrial systems will tend to cause responses primarily attributable to the depositional. compound (e.g., SO_x or NO_x); in some instances biochemical transformation and utilization of the compounds will generate accumulations of hydrogen ions causing pH effects to predominate eventually. Needless-to-say, poorly buffered aquatic systems will have similar responses to both wet and dry depositions as dry depositions essentially become wet deposition upon entrance into the aquatic system. These responses will be similar to wet depositions in poorly buffered terrestrial systems.

Agricultural Effects

Low-pH precipitation can affect crop yields in two ways. First, as it percolates through the soil column it accelerates the natural tendency of water to leach organic and mineral soil components from the root zone. At the same time, it reduces the soil pH level in this zone, thus making nutrients less available and toxic metals, such as soluble aluminum and iron, more available to plants. In addition, reduced soil pH levels can cause declines in microbe populations that break organic matter down into forms useful for plants. In the absence of this breaking down, the organic matter can accumulate and seal the upper layers of the surface while permitting various plant

toxins to be formed from the matter. The result for **all** these impacts is reduced growth and yields for plants located on the acidified soils [**Buchman** and Brady **(1960)**]. According to Schwartz and **Follett,** (1979, p. 2) the "preferred soil **pH** range for maximum growth" varies between 5.5 and 7.0 for most commercially important crops, e.g., clover, barley, corn, grasses, and soybeans. However.. this is by no means universal across crops, since the interval for alfalfa is 6.2 - 7.5; for asparagus and lettuce, it is 6.0 - 7.0; for blueberries and sweet potatoes, it is 5.0 - 5.7; for white potatoes, it is 5.0 - 5.4; and for cotton, it is 5.5 - 6.5. In the agricultural regions east of the Mississippi River, soil acidity levels frequently fall below these "maximum growth" intervals because of the region's high rainfall and because of grower soil additions of inorganic fertilizers. Consequently, it is a standard grower practice to mix calcitic or dolomitic ground limestone with the soil periodically.[1/] This practice returns the soil to something resembling its original state in terms of the availability of nutrients and toxic metals to plants. It also increases the sizes and the variety of microbe populations. We have found no evidence that the economics of liming causes farmers to fail to return soils to an approximation of the aforementioned state.

In a verbal communication, **N.R.** Glass (1979) of the United States Environmental Protection Agency has stated that if all the sulfur dioxide emitted annually east of the Mississippi River were to fall as acid precipitation on the agricultural soils of the region, ". . . a five percent increase in liming would be required." According to a verbal report on January 17, 1980, from Dr. Ed Strobe, Professor of Agronomy at Ohio State **University,** the 1979 cost of liming, including spreading, in the Ohio **Valley** region is $6 to $8 per ton. Raising soil **pH** to 6.0 for most cropping systems (e.g., alfalfa, clover, corn) on average mineral soils requires the application of 2 to 3 tons of lime per acre every 4 to 5 years. There is substantial variation across soil types, however, as **Buchman** and Brady (1960, p. 419) show. In the 1970's according to the U.S. Department of Agriculture (various issues), the annual consumption of pulverized lime in the **United** States varied from a low of 26.7 x 10^6 tons in 1972, to a high of 39.8 x 10^6 tons in 1976. In real terms, its price, independent of the cost for spreading, was consistently around $3 per ton in 1978 dollars, with the 1972 price being $3.14 and the 1976 price being $3.08. It seems unlikely, therefore, that a five percent increase in biological liming requirements would have much of an effect on either the cost of liming or on the farmer's perceptions of the economically optimal amounts of lime to spread.

Table 1 gives the average acreages, yields, and values of eight major field crops for 1975-77. In addition to Minnesota and the states east of the Mississippi River, Iowa and Missouri are included. With the exception of potatoes, the eight crops listed are the major field crops produced in the

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TABLE 2.1

Average Acreage, Production and Gross Value for Selected Commodities, by Region and Total, 1975-77 Crop Year.

USDA Production Regions

	Appalachian [1]			Delta [2]			Corn Belt [3]			Northeast [4]			Southeast [5]			Total	
	Acres	Prod.	Value	Acres	Prod.	Value	Acres	Prod.	Value	Acres	Prod.	Value	Acres	Prod.	Value	Acres	Total
								(Millions)									
Wheat	1.09	35.50	95.50	0.72	25.20	63.20	6.41	254.80	713.70	0.71	24.80	17.11	0.34	9.50	26.90	9.26	973.40
Corn	4.35	284.50	715.50	0.26	12.30	27.20	36.10	3513.80	7740.30	2.59	218.90	500.00	3.30	172.80	390.70	46.60	9373.70
Barley	0.20	8.60	12.75				0.04	1.62	2.38	2.62	12.50	19.41	0.04	1.01	1.44	2.90	36.00
Sorghum	0.14	7.12	14.70	0.31	14.80	29.30	0.79	50.20	93.50	–			0.08	2.93	6.34	1.32	143.90
Cotton	0.40	0.31	78.30	2.60	2.60	696.80	0.24	0.20	51.50	–			0.73	0.58	160,70	4.00	987.30
Sugar Beets							0.03	0.62	15.50	0.01	0.02	0.34				0.03	15.80
Tobacco	0.80	1623.30	1792.10	6/	0.09	0.11	0.02	46.10	50.90	0.04	366.65	52.28	0.16	324.00	356.90	1.01	2252.30
Soybeans	4.83	116.70	683.60	10.20	230.40	1370.80	11.84	113.90	4978.70	0.74	18.80	107.20	3.10	87.80	508.20	30.20	7648.40
Alfalfa	0.47	1.22	67.50	0.09	0.23	12.80	4.02	12.50	692.50	2.08	5.46	305.70				6.66	78.50
Total	12.30	7/	3460.00	14.20	7/	2200.30	59.50	7/	14339.00	8.80	7/	1002.10	7.80	7/	1451.20	102.60	22452.60

Source: USDA Agricultural Statistics, 1978. Washington, D.C. 1979

[1] Includes states of Kentucky, North Carolina, Tennessee, Virginia and West Virginia.

[2] " " " Arkansas, Mississippi and Louisiana.

[3] " " " Illinois, Indiana, Iowa, Missouri and Ohio.

[4] " " " Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania , Rhode Island and Vermont.

[5] " " " Alabama, Florida, Georgia and South Carolina.

[6] Less than one thousand acres.

[7] Different units of measurement for production preclude aggregation. Specifically, wheat, corn, barley, sorghum and soybeans are measured in bushels, tobacco in pounds, sugar beets and alfalfa in tons and cotton in bales (500 pounds).

indicated subregions. The range of tolerances to acidified soils for these crops varies from highly sensitive (alfalfa) to moderately tolerant (corn). Potatoes are not included because of their relatively high tolerance for acidified soils (pH = 5.0 - 5.4, as previously noted). Table 1 indicates the substantial magnitude of agricultural activity in these five subregions, individually and in the aggregate. The eight field crops in the subregions account for 38 percent of the total harvested acreage of nearly all crops in the United States, with the Corn Belt subregion comprising the single largest area of crop land. Furthermore, these eight crops assume a dominant role in United States agricultural exports. In terms of gross "on-farm" value, the $24 x 10^9 in receipts represents 25 percent of the total value of all agricultural commodities, including livestock, produced in the United States. In terms of individual crops, corn represents the single largest component of gross value, followed by soybeans.

Assuming that the 117.35 x 10^6 acres of the field crops in Table 1 would annually require the application of 0.40 to 0.75 tons per acre of pulverized lime in the absence of acid precipitation, anywhere from 46.94 x 10^6 to 88.01 x 10^6 tons would be used each year. The lower bound of this interval exceeds the maximum amount (39.83 x 10^6 tons) ever used for agricultural purposes in the entire United States. We, therefore, assume that annual use in the region of interest in the absence of acid precipitation would be 35 x 10^6 tons. Thus, if acid precipitation were to add 5 percent to the quantities of lime farmers in the region choose to use, an additional 1.7 x 10^6 tons would be applied. At $6 to $8 per ton in 1978, the annual cost of purchasing and applying the lime would be $10.50 x 10^6 to $14.00 x 10^6. This estimate, which is probably exaggerated for several obvious reasons, should, however, be contrasted with the estimate of the Commission on Natural Resources of the National Academy of Sciences (197 , p. 178). The latter employed a 197 cost, including spreading, of $14 to $18 per ton for an additional 12 x 10^6 tons of lime to counter only the effects of acidifying atmospheric depositions. Implicitly, this presumes that from farmers' decisionmaking perspectives, one-third as much lime is required to counter the soil-acidifying effects of these depositions as is required to counter the soil-acidifying effects of inorganic fertilizers and reasonably pristine precipitation.

In addition =to its soil acidifying effects, acid precipitation can directly harm plants by causing foliar necrosis, reduction of leaf area, leaching of leaf surface minerals, and cuticular erosion as the plant foliage intercepts the precipitation [Cowling (1978, pp. 49-50)]. The seriousness of these effects in terms of yields is thought to differ widely across plant species and across different life stages of the same plant. However, unless one is willing to make rather tenuous analogies with the well-known effects of sulfur oxides [Committee on Sulfur Oxides (1978, pp. 80-129)], there appears to be only minimal knowledge about the effects of acid precipitation upon crop

yields. The two most commonly cited studies referring specifically to acid
rain effects on yields seem to be Ferenbaugh's (1976) research on pinto beans,
and a degradation in the exterior appearance of Yellow Delicious apples that
Cowling (1978, p.59) mentions. These results are hardly a sufficient basis
for any analysis, sophisticated or otherwise, of the economic impact of the
direct effects of acid precipitation upon agricultural yields. Dose response
functions on a greater variety of crops are expected to be available soon,
however. The United States-Canada Research Consultation Group on the
Long-Range Transport of Air Pollutants (undated, p. 19) reports that results
from studies of the sensitivity of field crops to simulated acid precipitation
were to be made public in the spring of 1980. In the meantime, the
Consultation Group (p. 19) states that "... there is every indication that
acid rainfall is deleterious to crops," and that there is "... the potential
for widespread economic damage to a number of field crops."

In spite of the current absence of dose-response data to do either an
unsophisticated or a sophisticated economic analysis of the direct effects of
acid precipitation upon crop yields, one might hazard a guess about the
magnitude of the "potential" for widespread economic damage by drawing anal-
ogies with other cropping systems that are known to have been exposed to
continuing high levels of air pollution. If the forms of representative plant
responses and representative farmers' decisions based on these responses are
roughly similar across types of air pollutants, combinations of crop types,
and geographical areas, the hazarded guess would have some basis in reality.

Adams, et al. (1979) have studied the economic effects of photochemical
oxidants in southern California upon twelve vegetable and two field crops.
Their study took into account differences in the tolerances of the yields of
the various included crops to oxidant exposures, changes in cropping patterns,
input substitutions, and locational changes. For many of the included vege-
table crops, southern California has a seasonal near-monopoly. Major adjust-
ments in cropping patterns and cropping locations were predicted and observed
within the region in response to increasing ambient oxidant levels. After all
these adjustments, a 3.01 percent decline in the sum of producer rents and
consumer surpluses occurred, with three-quarters of this percentage decline
being producer rents. The total on-farm value figure of $24 x 10^9 in Table 1
for eight field crops in five agricultural subregions of the eastern United
States might or might not be a reasonable approximation of the sum of producer
rents and consumer surpluses obtained from these crops.[2/] If the 3.01 percent
reduction is applied to the $24 x 10^9 gross on-farm. value, a loss of $720 x
10^6 results. In 1978 rather than 1976 dollars, this would be about $828 x
10^6, a figure that is to be taken no more seriously than the credence one is
willing to give the analogies and assumptions from which the figure is
derived. If one were to include various fruit crops such as apples, oranges,
and peaches, legumes and tubers such as peanuts and potatoes, and ornamental

in the calculations (in 1977, the U.S. on-farm value of the production of these crops was more than \$4.0 x 10^9), the figure might reach \$1.0 x 10^9.

Forestry Effects

Of all the potential effects of acid precipitation, the effects upon forest ecosystems "seem to be least understood. General qualitative descriptions of what might happen abound, however, e.g., Abrahamsen and Dollard (1978), Cowling (1978), Dochinger and Seliga (1976), and Tamm (1976). As with agricultural systems, the health of forest ecosystems can be affected directly and indirectly by acid precipitation. Moreover, the health states can be aided as well as hindered [Tveite (1980)].

Detrimental direct effects upon the physiological and metabolic processes of forests are likely to occur when foliage intercepts acid precipitation. Reductions in leaf areas, excessive leaching of organic materials, cuticular erosion, necrosis, and reductions in photosynthesis and the cycling of nut-rients to other system components are all regarded as likely events. However, Abrahamsen and Skeffington (1979, p.D.2.1) note that carefully documented field cases relating necrosis to acid precipitation do not exist. Our search of the literature has not turned up field demonstrations of the other effects, although they all have been found for one or more tree species in controlled experimental settings.[3] All of these studies of particular types of effects fail to make clear how the observed effect is related to tree growth. There nevertheless appears to be a consensus that the physiologically most active developing tissues are the most sensitive [Knabe (1976)].

Neither the field nor the experimental studies of the effects of acid precipitation on tree growth have yielded consistent findings. In fact, findings of no effects or positive effects of acid precipitation upon tree growth dominate the literature. The results reported in a recent paper by Lee and Weber (1979) are typical. These authors subjected the seeds of eleven tree species, including Douglas fir, eastern white pine, yellow birch, sugar maple, sumac and hickory, to an artificial rain containing enough dilute H_2SO_4 to lower rainfall pH levels to 4.0, 3.5, and 3.0. The plants were exposed to these rains for three hours for each of three days a week over 1.5 years. In general, acid precipitation had either no effect or a positive effect upon the proportion of seeds that germinated and upon the top dry weight and root dry weight of the germinated seeds. The soil solutions in which the seeds were placed had high buffering capacities, causing the authors to infer that whatever effects were observed were direct. In particular, they did find some negative effects upon foliage but they also frequently found enhanced rates of growth. The latter they attributed to fertilization of the soil solution by sulfur, nutrients leached from the plant surfaces, and increased plant uptake of soil nutrients. For Tee and Weber (1979), the "stimulator" effects of

acid precipitation over the 3.0 - 4.0 pH interval nearly always dominated or negated the "inhibitory" effects. The results of these authors thus support **Abrahamsen's** and **Dollard's** (1979, p.8) statement that: "Statistically significant effects [in laboratory settings] have been observed only when applying 'rain' with pH 3 or lower."

....

Elsewhere, Abrahamsen and **Skeffington** (1979, p.D.2.4) indicate that they and their discussants "... could think of no evidence to indicate yield reductions under treatment with acid rain in laboratory conditions at realistic levels of acidity (i.e., pH≅4). Later on, they state that "...no **artifical** acidification experiment has produced a growth reduction at pHs equivalent to those commonly observed in rain, and some have produced a stimulation." These stimulator effects of acid precipitation (or **acidifying** deposition) have recently been broadcast in several widely read periodicals, e.g., **Maugh** (1979). The implicit position would then seem to be that some limited amount of acid precipitation in excess of that which is natural has a positive impact upon plant growth, probably as an amendment to sulfur and nitrogen deficient soils. Given that the time interval over which these positive amendment effects would occur is typically unspecified, one is left to presume they would continue at least as long as the amendments continue.

Forest ecosystem acidification leads to reduced nutrient cycling rates. Not **only** can nutrients become **complexed** at low pH levels, as for phosphates, but decomposition of organic material is slowed. Many potential decomposes are less active or inactive at pH levels much below 5.0, and protozoa and earthworms are very rare in soils having pH levels below about 4.0 (Abrahamsen et al. 1976). Lohm (1980) observed that acidification decreased decomposition rates for both needles and litter as well. as decreasing fungal lengths, **bacterial** numbers and cell sizes. The results of Francis et al. (1980) also suggests that acidification of forest soils may cause significant reductions in leaf litter decomposition and reduce nutrient recycling rates in forest ecosystems by slowing ammonification, vitrification and denitrification. Tamm et al. (1976) found that even moderate additions of sulfuric acid to soils produce obvious effects on nitrogen turnover rates.

Besides affecting nutrients through reduced decomposition rates, depressed **environmental** pH levels depress nitrogen fixation. Denisen et al. (1976) found that lowered pH levels caused both the nitrogen fixing bacteria Azotobacter and nitrogen fixing blue-green algae to disappear from the soil.

The net consequence for terrestrial ecosystems of decreased nutrient cycling and increased nutrient leaching caused by acidification is reduced productivity [Abrahamsen et al., (1976); Glass and Loucks (1980); Leivestad et al., (1976)]. Such responses have been termed "self-accelerating **oligo-trophication**" by Grahn et al. (1974). Acidification could reduce or remove

nutrient pools, constraining the redevelopment of previous biomass levels [Glass, (1978)].

Using nitrogen as his example nutrient, Tamm (1976, p. 237) presents a flow diagram which places all of the preceding results in a most meaningful perspective. We reproduce with minor adaptations his diagram as Figure 1. Tamm (1976) states that the diagram is meant to be a hypothetical representation of the effects of increased strong acid in a system where much of the nitrogen supply comes "... from decomposition in an organic A_0 horizon with a high carbon/nitrogen ratio (>15)." This suggested structure is consistent with experimental. observations in which simulated acid precipitation contributes positively to growth, even though it is expected that, in the long-term, growth would decline once the buffering capacity of the soil is exhausted. Tamm (1976, p. 338) also notes that the structure could account for the frequently observed experimental failure of lime to enhance growth rates: by immobilizing the nitrogen in organic matter, the availability of nitrogen to the trees is reduced.

The only available estimate of the long-term effects of acidifying deposition upon pH levels and base saturation appears to be McFee, et al. (1976). These authors estimated that for a "typical" midwestern soil, precipitation with a strong and disassociated acidity of pH = 4.0 at 100cm annually for 100 years would reduce soil base saturation by 19 percent and lower soil pH by 0.6; at pH 3.7 (a doubling of acidity) and PH 3.0, 50 and 10 years would be respectively required to bring about the same changes. If the soil initially has a fairly high pH level, effects of this magnitude would probably not be noticeable in terms of plant growth results; however, if the soil already has low pH, it is generally thought [McFee (1978, p. 66)] that leaching would rapidly increase. Generally, therefore, it is thought that $d^2(\text{Leaching})/d(\text{pH})^2 > 0$, implying that the rate of loss of soil nutrients would get progressively worse as time passes. It thus seems that insofar as forest soils are concerned, acid precipitation has all the attributes of acquiring possible short-term gains in forest growth at the cost of probable long-term losses in forest soil fertility. For both biological and economic reasons, it seems unlikely that liming can counter the fertility decline. In addition to the previously mentioned biogeochemical argument of Tamm (1976), Tisdale and Nelson (1976, p. 428) note that "... particulate of limestone cannot move in the soil, and consequently they must be placed where they are needed." Tilling lime into extensive areas of forest soils would seem both a technical and an economic impossibility.[4/] In anything other than geological time, accelerated soil acidification, therefore, appears irreversible.

Unfortunately, it is quantitatively unclear how the above reductions in soil fertility will ultimately affect forest yields or the properties of other forest ecosystem components (water storage, game animal populations,

aesthetics, etc.) that humans directly value. We were unable to discover any information whatsoever that related acid precipitation to the latter components.⁵/ Jonsson and Sundberg (1972), in a frequently cited paper, suggest 2 to 7 percent declines due to acid precipitation in the annual growth rates of forests in southern Sweden during the 1950-1965 period relative to 1896-1949. However, Cogbill, .(.1976), in a simple time trend study of comparative tree ring growth in areas of eastern North America subject to and free of acid precipitation, could find no differences attributable to acid precipitation. The trends he observed started no later than 1890, and continued until 1.972-73. However, the credibility of Cogbill's (1976) results must be tempered by questions about the comparability of sites at which precipitation pH has been measured [Perhac (1979)] ; the errors inherent in much of the field instrument-ation traditionally employed to measure pH [Galloway, et al. (1979)]; and one study [Frinks and Voight (1976)] which gives cause to believe that, at least in one area of Connecticut, the pH of precipitation has been rather low and unchanged since the early 1900's.

Given the empirical. confusion that exists with respect to the ultimate impacts of acidifying deposition on rates of forest growth, we choose to adopt Jonsson's (1976, p. 842) position that there is "... no good reason for attributing the reduction in growth to any cause other than acidification." We adopt this position because it is consistent with existing knowledge of the biogeochemistry of forest ecosystems [e.g., Likens, et al. (1977)] as well as the economic law of variable proportions.

One's willingness to accept estimates, made using current price and production data, of substantial positive benefits from reducing forest expo-sures to acid precipitation or acidifying deposition must be tempered by evidence that the current elasticity of substitution between land and intensive forestry is very high. Clawson (1976), for example, argues that the following outputs of the national forest system can all be economically and simultaneously increased as follows: net annual growth can be twice as great; designated wilderness areas can be four times greater; outdoor recreation can be doubled; and wildlife stocks and water storage can be modestly increased. Miller (1978), while considering endangered animal species, also emphasizes that improved management techniques have great potential for maintaining existing wildlife stocks now suffering from environmental stresses. Hair, et al. (1980, pp. 514-519) note that if all opportunities offering at least a four percent net annual return on all forest land. in the South were to be exploited, the region's 1976 net annual timber growth would have increased by 86 percent. In the Northeast and North Central regions, the corresponding increase offering a similar minimum return would have been 25 percent. Berk' s (1979) recent finding that private forest owners act as if they faced a real before-tax interest rate of only five percent is consistent with management behavior which fails to exploit the opportunities that Hair, et al. (1980)

think to exist.

In Minnesota and the states east of the Mississippi River, there were, according the the U.S. Forest Service (1978, pp. 1, 97), 229.8×10^6 acres of commercial and productive reserved forest land that had a net annual 1977 growth of 6.12×10^9 cubic feet of softwoods and 7.51×10^9 cubic feet of hardwoods. Lerner (1978, p. 735) indicates that 1977 stumpage prices in 1978 dollars for the softwoods averaged about $1.30 per cubic foot, while for hardwoods they were about 50 cents per cubic foot. Jonnsson and Sundberg (1972) and Panel on Nitrates (1978, p. 577) judge that a reduction of five percent in net annual growth falls within the interval of reduced growth one might reasonably expect from the acid precipitation now falling upon Scandinavia and eastern North America. If the precipitation in Minnesota and the states east of the Mississippi River were pristine $(pH \cong 5.65)$, a five percent increase in 1977 net annual growth of softwoods would have amounted to 310×10^6 cubic feet. Hardwood net annual growth would have increased by 375×10^3 cubic feet. Assuming these increases would not have appreciably affected stumpage prices, the 1978 market value of the additional softwood would have been $\$404 \times 10^6$. The hardwood increase would have had a 1978 market value of $\$188 \times 10^6$. In 1978 dollars, the 1977 market value of the increase for both softwoods and hardwoods would, therefore, have been $\$592 \times 10^6$.

The valued outputs of lands devoted to forests are by no means limited to timber. These lands provide outdoor recreation, aesthetic satisfaction, water storage, wildlife habitats, and a variety of other services. Unfortunately, representative estimates of the value of the sum of these services are rare. The sole immediately and easily useful estimate we have been able to find is the study of Calish, et al. (1978). These authors, in a rather nonrigorous but nevertheless extremely clever and interesting paper, estimated that the 1978 annual non-timber value (in terms of harvestable game animals and fish, water flow, nongame wildlife diversity, visual aesthetics, and prevention of mass soil movement) of a representative Douglas fir forest in the Pacific Northwest was $87 per acre. Assume that this same value per acre can be applied to eastern forests; and further assume that current levels of acid precipitation reduce these non-timber values in the same proportions as was assumed for the timber values, i.e. , by about five percent. Calish, et al. (1978) attribute about 11 percent of the $87 per acre to the production of harvestable fish, implying that $77 per acre consists of non-timber values for which we have not otherwise accounted. Five percent of this $77 is $3.85. Thus , if pristine precipitation were to replace acid precipitation in Minnesota and east of the Mississippi River, these procedures imply that 1978 non-timber values in this area would have annually increased by ($3.85 per acre) $(299.8 \times 10^6) = \$1.15 \times 10^9$. Adding the timber and non-timber value increases yields an annual benefit of $\$1.75 \times 10^9$ in 1978 dollars. Assuming a 15 percent discount rate and that this increase could be sustained

indefinitely, a discounted value of 11.66×10^9 results.

Aquatic Ecosystem Effects

Contrary to the forest and forest soil effects of acidification, a substantial amount of economically useful quantitative information is available about the aquatic ecosystem effects of acidification. McFee (1976) conjectures that some soils in Scandinavia and eastern North America have been subjected to acid precipitation for approximately a century. Only within the last decade or two, however, are the soils thought to have become sufficiently acidified to influence the pH levels of inflows to fresh water bodies. Because of this relative immediacy, changes in the biota of these water bodies have been observed rather than being considered as historically preordained. In addition, there has for several decades been an historical record of the impact of acid mine drainage upon the biota in the streams of the Appalachian region. Barton, (1978, p. 314) states that in the United States 10,000 miles of streams and 29,000 surface acres of impoundments and reservoirs ".. . are seriously affected by mine drainage."

Declines to about 6.0 in the pH levels of fresh water bodies reduce primary production. Since primary production is reduced, detritus derived from the decay of plankton tends to disappear. As a consequence, water transparency increases, detrital material decreases, and there are increases in soluble alumina, irons, magnesia, and trace metals such as cadmium and mercury. The phytoplankton and zooplankton species which survive are obviously acid-tolerant, perhaps because they are resistant to heavy metals. However, they also concentrate these metals which, in turn, may make them toxic to many fish and bottom-dwelling (benthic) organisms. Although fish and insect kills occurring during heavy rains and spring snowmelts have been frequently observed [e.g., Gjessing, et al. (1976, p. 65)1, the major effects upon fish and insects are thought to stem from reproductive and recruitment failures caused by calcium metabolism difficulties, the accumulation of heavy metals in parents, and the exposure of those young that are produced to these metals [Fromm (1980)]. Schofield (1976, p. 229) indicates that increased salinity tends to make fish more acid-tolerant, apparently, according to Packer and Dunson (1970), because it enables fish to replace the body sodium losses that low pH causes. The reproduction failures can result in extinction of the species in acidified water bodies. Almer, et al. (1978, pp. 303-307) cite several cases where surviving individuals of some game fish species grew more rapidly and were of larger size than were similar individuals in less acidified water bodies. They attributed this to the lessened competition for the available food stock.

Both Almer, et al. (1978, pp. 308-309) and Gorham (1978, pp. 41-42) remark upon the possible implications of altered (generally reduced) diversity

and biological productivity of aquatic flora and fauna for organisms such as amphibians, birds, and mammals who spend important parts of their life cycles in and around aquatic environments or who are dependent upon these environments for some or all of their food supplies. Almer, et al. (1978), tell of fish- eating birds such as loons who have migrated from acidified lakes in Sweden to sites with more ample food supplies. Gorham (1978) expresses concern for the impact upon moose populations and distributions if production of the aquatic plants forming parts of their diets is inhibited. Birds of prey, such as eagles and osprey, and game birds, such as mallards and wood ducks, could be subjected to altered populations and distributions from reductions in their aquatic food sources. Little seems to be known, however, about the ease with which these animals could substitute nonaquatic food sources.

In the following pages several attempts will be made to develop rough economic values for the aforementioned aquatic ecosystem effects. Each attempt is made in order to exploit a particuar type of natural and/or economic information. The measures developed are not additive. Most important, as is the case throughout this report, all the measures are and very sensitive to minor perturbations in assumptions.

Although the criteria he uses for determining what is and is not "fishable" are unclear, Todd (1970, p. 303) shows 19.14×10^6 acres of fishable fresh-water streams and lakes in Minnesota and the states east of the Mississippi River. This excludes the Great Lakes which have an area of 38.00×10^6 acres. Adams, et al. (1973, p. 43) indicate that about 23 percent of the approximately 168×10^6 people twelve years or older then living in this area passed an average of 7 days fishing in the summer quarter of 1972. This represents 270×10^6 fishing activity days. No reliable information could be formed giving the seasonal distribution or the fresh-salt water distribution of fishing activity days. We, therefore, assume that 85 percent of this fresh-water fishing occurs during the summer quarter. Upon making the adjustments called for by these assumptions, we are left with 287×10^6 fishing activity days in Minnesota and the states east of the Mississippi in 1972. We assume that the number of activity days in 1978 was 300×10^6 more-or- less.

A number of studies of varying degrees of sophistication are available which purport to give the representative willingness-to-pay for an additional fresh-water fishing activity day. These estimates are distributed over a range from 10 to 30 mid-1970's dollars. The S20.72 uncompensated consumer surplus estimate obtained in 1972 by Gordon, et al. (1973) is adopted here. Assuming that increases in real income, in the value of time, and changes in relative fishing costs have not altered this figure, this amounts to $32.30 in 1978 dollars. If the marginal value of a fishing activity day is a constant regardless of the availability of opportunities to catch fish and if no

fishing occurs in the absence of fish, the extinction of fresh-water fish life
in Minnesota and the states east of the Mississippi River would have resulted
in economic losses of (300 x 10^6 activity days) ($32.30) = $9.69 x 10^9 in 1978.
Assuming the number of fishing days is linearly and inversely related to the
acres of fresh-water having fish populations, and if Tables 1 and 2 from
Chapter III are to be believed, well over half this loss would occur before
all fresh-water in the aforementioned area reached a uniform pH level of 6.0.
If large regions in the general area retained pH levels for fresh- water at
6.5 - 8.0, even though the average pH over the entire region was 6.0 or less,
the economic losses would be a great deal less. This is because fishermen
would readily be able to substitute away from an acidified to nonacidified
bodies of water.

The recreational value of the fresh-water resource in Minnesota and the
states east of the Mississippi is clearly rot limited to fishing activities.
One major additional use is for hunting, particularly waterfowl hunting.
According to Todd, (1970, p. 303), there exist in this area 48.83 x 10^6 acres
of natural wetlands "... of significant value to fish. and wildlife." In 1980,
the U.S. Water Resources Council (1968) projected that the number of
water-related hunting activity days in the area would be 220 x 10^6. This was
projected from the 162 x 10^6 days taking place in the same area in 1960. On
the other hand, the U.S. Fish and Wildlife Service (1972, p. 31) estimated the
number of 1970 waterfowl hunting activity days by people 12 or more years old
to be 17.58 x 10^6. Since, relative to fishing activity days, this seems more
plausible, we employ it here. Hammack and Brown (1974, p. 29), in their study
of the value of waterfowl, found that the average waterfowl hunter passed 9.7
days each season engaged in the activity, and acquired a consumer surplus of
$247 (= $462 in 1978 dollars) from the right to hunt during the 1968 season.

Even if fresh-water pH levels were universally to drop to 4.0 or less
throughout the region east of the Mississippi, it cannot be assumed that all
waterfowl in the region would disappear. Some invertebrates that constitute
part of the diet of waterfowl would survive. More important perhaps, there
are land-based food items (such as agricultural grains) that waterfowl might
employ as a substitute food source. Nevertheless, it should be noted that
several very important hunted species of waterfowl currently get 50 percent or
more of their food supplies from aquatic sources. Martin, et al. (1961)
state, for example, that 50 to 70 percent of the diets of the mallard, the
black, and the common goldeneye ducks consist of aquatic insects, crustaceans,
and mollusks. Moreover, the habitats of these birds, especially the mallard,
tend to be the ponds and shallow lakes thought to be more susceptible to acid
precipitation. Of the waterfowl hunters in the Hammack and Brown (1974, p.
39) sample, 47 percent stated that mallards were their first preference in
waterfowl hunting.

Given the almost total lack of information on the impact of fresh-water acidification upon the population and distributions of waterfowl, we choose to make some quite arbitrary assumptions. In particular, we assume that a representative member of a representative species currently obtains half its food supply from fresh-water aquatic environments. It, therefore, has some ability to substitute food from terrestrial or marine environments for the fresh-water source without having to alter its location in any given season. Without guidance from any source, we assume that the destruction by acidification of the acid-intolerant portion of the fresh-water food supply for waterfowl will result in a reduction of 20 percent in the waterfowl population during the hunting season in Minnesota and the states east of the Mississippi River. Further assume that the elasticity of waterfowl hunting activity days with respect to waterfowl populations is 0.5. If the number of waterfowl hunting activity days in the region of interest was approximately 30×10^6 in 1978, the assumed elasticity and reduction in waterfowl population implies a drop in 1978 activity days to 28.5×10^6. If the marginal value of an activity day is a constant, the Hammack and Brown (1974) estimate of the consumer surplus obtained from the right to hunt waterfowl implies an activity day valuation of $462.00/9.7 = $47.63. The ten million day reduction in waterfowl hunting activity days, therefore, yields a 1978 economic loss of 71.45×10^6. Given our long chain of arbitrary assumptions, we conclude that an estimate of 70 million dollars is reasonable. It is important to note, however, that Hammack and Brown (1974, p. 63) found that waterfowl hunters could annually shoot 76 percent of the waterfowl in the Pacific flyway without reducing the breeding population. It seems unlikely that fresh-water acidification would have an effect upon recruitment of this magnitude. One might, therefore, conclude that fresh-water acidification will cause no changes in waterfowl populations and distributions and thus no economic losses will be incurred. In cases where waterfowl and fish have been competing for the same aquatic insects, it may be that acidification by reducing fish populations will reduce competition for the aquatic insect food source and thereby enhance waterfowl populations.

If the food supplies of waterfowl are harmed. by fresh-water acidification, it follows that the food supplies of birds that are not legally hunted will also be harmed. The diet of the osprey consists entirely of fish, while bald eagles are mostly dependent upon fish for food. Cranes, including the sandhill crane, utilize a variety of aquatic vertebrates and invertebrates for food sources. In spite of the possibility that these and other birds will suffer population declines from fresh-water acidification, we have no basis whatsoever on which to formulate a guess at the economic losses a population decline might entail. Not only do we lack any quantitative information on population responses to acidification, we also lack any information about the values people place upon encounters with these birds.

A somewhat similar problem exists for mammals dependent upon aquatic ecosystems for part of their food or even for their everyday habitats. These would include, for example, racoon, mink, muskrat, marten, beaver, river otter, and even moose and black bear. No readily available information exists on the population responses of these animals to acidification-induced perturbations in their aquatic food sources or habitat. Nor does any economic information exist on the value of an encounter with these animals. Economic information does exist, however, on their market value when they are killed for their furs. The U.S. Water Resources Council reports according to Todd (1970, p. 270) that in 1966, 7.0 x 10^6 "fresh-water dependent" fur-bearing animals were captured in Minnesota and the states east of the Mississippi. These animals, in 1978 dollars, had a market value of $26.7 x 10^6, or approximately $3.80 per animal, assuming the resources employed in their capture had no valuable alternative uses. Measured solely then in terms of the worth of their fur when captured, the economic impact of a fresh-water acidification- induced population reduction will be minor. Again, however, this disregards the animals' value in terms of simple observations during encounters as well as the value they contribute to hunting recreational activities.

Boating and swimming are additional recreational activities that conceivably could be affected by reductions in the pH of fresh-water. However, in a study of the acid mine drainage problem in the Appalachian region, Robert R. Nathan Associates (1969) was unable to find any variation in boating and/or swimming activity days with respect to different levels of pH. This is not too surprising since it is generally recognized that boaters and swimmers respond negatively to increased turbidity. Kramer (1978, p. 354) notes that there is an approximate doubling in transparency, measured as Secchi depth, for each unit decrease in pH over the 6.5 - 4.5 pH interval. He attributes this "... to the dissolution of iron and manganese colloids and the decrease in organic detritus with decreasing pH due to decreasing photosynthesis" (p. 354) .

Added to all the above results must be the commercial value of captured fresh-water fish. In 1965, Todd (1979, p. 270), using data from the U.S. Water Resources Council, estimated that 2801.6 x 10^6 pounds of "fresh-water-dependent" fish were caught in our region of interest. The market value of these fish was $196.2 x 10^6 (=$404.2 x 10^6 in 1978 dollars). Sixteen percent of this catch, was either fresh-water or anadromous species. Assuming the resources employed to catch this sixteen percent had no valuable alternative uses, if these fresh-water or andromous species were to be extinguished, the annual loss would amount to 65 x 10^6 1978 dollars. This includes the extinction of the catch from the Great Lakes.

When all the preceding effects of acid precipitation upon fresh-water

ecosystems in Minnesota and the states east of the Mississippi River are summed, one obtains 1978 annual benefits for preventing pH levels falling below 4.5 - 5.0 for all fishable fresh-water bodies in this area of $10 to $11 billion. Nearly all the benefits are attributed to the maintenance of recreational fishing opportunities. It must nevertheless be emphasized that the economic. and biological analyses on which these estimates are based have weak. analytical foundations. Given those limits, somewhat more creditability might be achieved by adopting a different approach to assessing the benefits of acid precipitation control.

If the acidification of fresh-water does negatively influence ecosystem attributes that human beings value, one would expect these negative influences to be capitalized into land that offers ready access to these valued attributes. In particular, economic theory predicts that land prices will be consistent with the values people attach to the differences in these advantages of access. Freeman (1979) outlines the circumstances in which these prices will reflect the variations in consumer surplus generated by the differences in access advantages.

Adams, et al. (1973, p. 111-110) present estimates by farming region of differences in the 1972 per-acre value of recreation land with and without water. For the states east of the Mississippi River, the median difference by region appears to be about $1,250 per acre. Recreation land without water is generally about 33 to 50 percent the value of land with water. If the values of both types of recreational land behaved as did farm real estate values between 1972 and 1978, their values approximately doubled [Economic Research Service (1978)], implying that the $1,250 per acre difference in 1972 had increased to a $2,500 per acre difference by 1978.

Excluding the Great Lakes, Todd (1-970, p. 301) finds that there exist 21.86×10^6 acres of inland water available for recreation in Minnesota and the states east of the Mississippi. This is somewhat greater than the 19.14×1.0^6 acres that he considers to be "fishable" (p. 303)$_6$ Of this "fishable" acreage, 3.83×10^6 is in streams, leaving 15.31×10^6 acres in natural and man-made lakes, exclusive of the Great Lakes. Assume that each surface acre of fishable freshwater lakes, excepting the Great Lakes, is associated with one riparian acre. This one acre figure is a judgment formed by using a table in Todd (1970, pp. 126-127) of the surface acreages and shoreline lengths of the largest lakes in each state of the United States. Man-made lakes tend to have shoreline mileages about 10 times as great as the square mileages of their surface areas, while the shoreline mileages of natural lakes tend to be about half as great as the square mileages of their surface areas. We estimate then, that of the nearly one billion acres of the surface of the globe contained within the boundaries of Minnesota and the states east of the Mississippi, 15.31×10^6 (or 1.6 percent) of them are riparian to lakes and

ponds.

Added to the lake and pond riparian acreage must be the acreage that is riparian to rivers and streams. Minnesota and the states east of the Mississippi contain about 36 percent of the 260,000 stream miles in the 50 U.S. states, meaning that they have about 94,000 stream miles. Assuming 50.66 riparian acres per stream mile, these 94,000 stream miles yield 4.76×10^6 riparian stream acres. By way of contrast, this means that we estimate 1.24 of riparian acres per acre of stream surface water, as opposed to the one riparian acre estimated for each acre of lake and pond surface water. Upon summing the estimated stream and lake and pond riparian acreages for Minnesota and the states east of the Mississippi River, we obtain 16.55×10^6 acres.

Now let us consider under some extremely strong assumptions what the extinction of fish life on these 16.55×10^6 acres might mean for their values. According to Unger, et al. (1976, pp. B.6 - B.8), there were 1.70×10^9 fresh-water related recreation activity days in 1970 in the United States, of which 0.63×10^9 days, or 37 percent, were devoted in some part to fresh-water fishing. Of the estimated $2,500 per acre difference in 1978 between the values of recreational land with and without ready access to water, we presume that exactly 37 percent, or a $925 premium is attributable to the fishing access the former offers. If the game fish in all of the waters to which the above mentioned acreages are riparian were simultaneously to disappear forever, the total value of these acreages would then decline by ($925) ($16.55 \times 10^6$) 15.32×10^9.

Earlier, using recreational data on fishing activity days and the consumer surpluses associated with them, we estimated that the disappearance of all game fish would have generated annual losses of 9.69×10^9 in 1.978. If these losses were to continue in perpetuity, and if they were all to be capitalized into the values of riparian acreages, a discount rate in excess of 200 percent would have to be applied in order to yield a present value of only 15.32×10^9. This hardly seems realistic. When a more reasonable discount rate of 15 percent is applied to this presumed present value for riparian property losses of 15.32×10^9, one obtains annual losses of 2.0×10^9. Lower discount rates imply still lower annual losses. For example, a discount rate of 5 percent. implies annual losses of only 0.73×10^9. On the other hand, if one applies a 15 percent discount rate to the earlier estimate of 9.69×10^9 in annual losses, one obtains a present value for this stream of losses of 74.29×10^9, a considerable amount of wealth indeed.

Before rejecting the previous 9.69×10^9 annual loss estimate in favor of an annual loss estimate derived from an equally limited treatment of the annual losses implied by reductions in riparian property values, it is important to keep in mind that these latter losses are related only to the

losses in consumer surplus suffered by the riparian property owners plus the
fees they might collect from present and future fisherman who wish to gain
access to fresh-water bodies over present owners' riparian lands. In the
great majority of cases, these fees are not collectable because of the costs
of policing access. One can thus make a strong argument that the impact of
acidification upon riparian property values, even if the structural conditions
outlined by Freeman (1979) are fulfilled , must be a substantial underestimate
because of the failure to register the surpluses accruing to fisherman who
have free access over the land.

The preceding accounting, in addition to its rather severe limitations in
terms of economic analysis, fails to consider the actions private and public
bodies might undertake to ameliorate the effects of acid precipitation. In
particular, one might lime and/or restock acidified fresh-water bodies. The
ecological evidence for the likely restorative successes that can be achieved
by liming is mixed. As for restocking, if one is willing to devote the
requisite resources, it is perhaps a feasible technical alternative. Whatever
the technical and financial feasibility of either or both of these restorative
procedures, it must be recognized that some set of decisionmaking bodies must
be formed and maintained to implement the restorative procedures. Given the
common property attributes of that which is to be restored and the public good
nature of the restorative and restocking actions, one might reasonably have
serious doubts about whether effective massive restoration and restocking
programs can be formulated and implemented.

In principle, the liming of fresh-water bodies will simultaneously serve
to raise the pH values of the water and to make the heavy metals abundant at
low pH substantially less available biologically. Hagerhall (1979) estimated
that in 1973 it would cost $45-70 x 10^6 to acquire and apply one million tons
of $CaCO_3$ in Sweden. Assuming similar costs in the United States, this amounts
to $66-103 per ton per year in 1978. He also states that apparently a one-
time application of 30-50 x 10^6 tons of $CaCO_3$ (p. 10) would suffice to return
22.24-27 x 10^6 acres of Swedish lakes to their pH states at the beginning of
the 20th century, if acid precipitation were to cease. Simply to counter the
current yearly increment in acidification over this lake area would, according
to Hagerhall (197,9, p. 10), require the annual application of one million
tons of $CaCO_3$. Thus restoration by liming in 1978 to pH levels of the early
20th century would require a one-time outlay from (30 x 10^6 tons/27 x 10^6
acres)($66) = $73.33, to (50 x 10^6 tons/22 x 10^6 acres) ($103) = $234.09 per
acre. The mid-range of this interval is $150 per acre. Thus, always remem-
bering the limits of the analysis, if all the 19.14 x 10^6 acres of fishable
fresh-water bodies in Minnesota and the states east of the Mississippi River
were to become acidified in a fashion similar to the aforementioned Swedish
lakes, a one-time 1978 outlay for liming of $2.87 x 10 would be required to
return them to pH levels in excess of 6.0. However, if the number of

acidified fresh-water acres is extended to the natural. wetlands ".. . of significant value to fish and wildlife" [Todd (1970, p 3031, this estimate increases to !(48.83 x 10^6 acres) + (19.14 x 10^6 acres)] $150 = $10.20 x 10^9. On an annualized basis, using a 15 percent rate of discount, these one-time outlays are respectively equivalent to $430 x 10^6 and $1.53 x 10^9.

Assuming the $150 per ton cost figure for the purchase and application of $CaCO_3$ to be reasonable, a rough check on the above estimates can be obtained by exploiting a statement of Holden's (1979, p. 11).

> "An alkalinity equivalent to 2.5 ugl^{-1} calcium carbonate requires the solution of 2.5 g. metre^{-3}. A lake of 10 ha with a mean depth of 2 m (a small lake), requires 500 kg of dissolved calcium carbonate, and probably ten times this amount of the solid to obtain sufficient in solution. A stream flowing at 1 m^{3}s^{-1} would require about 80 tonnes year^{-1} in solution. These quantities must be maintained each year. .."

Assume that the lake Holden (1979) describes is representative of most lakes in Minnesota and the states east of the Mississippi River. Further, assume that, because of the greater ability of streams to dilute materials, that representative streams annually require the injection of twice as much $CaCO_3$ as do the representative lakes. Given that "sufficient" solution of $CaCO_3$ requires the injection from all sources of ten times as much $CaCO_3$ in solid form, this is stating that each acre of our representative lake annually requires the introduction of about 450 pounds of $CaCO_3$. Streams, therefore, by our assumption, require 900 pounds of $CaCO_3$ for each acre of their surface areas. The 15.31 x 10^6 acres of fishable freshwater lakes and the 3.83 x 10^6 acres of fishable fresh-water streams in Minnesota and the states east of the Mississippi River, therefore, require the annual introduction of (15.31 x 10^6 acres) (450 pounds) + (3.83 x 10^6 acres) (900 pounds) = 5.17 x 10^6 tons of $CaCO_3$ or equivalent. If all this $CaCO_3$ had to be introduced by man, and if each ton of $CaCO_3$ cost $150 to acquire and inject, the total annual cost in 1978 would be $7?76 x 10^9. This figure is much greater than even the one-time cost earlier obtained using the data of Hagerhall (1979). Note, however, that the estimate from Holden (1979) assumes that all $CaCO_3$ entering these fresh-water bodies is somehow to be supplied by man. If, for example, half the $CaCO_3$ necessary to raise the pH level of a fresh-water body is naturally supplied from the catchment area, the $7.76 x 10^9 cost estimate would have to be reduced accordingly. However, if the 48.83 acres of wetlands "... of significant value to fish and wildlife" are taken into account, and if one assumes that the residence time of water in these wetlands is twice as long as in Holden's (1979) representative lake (implying that only half as much $CaCO_3$ is required as in the representative lake), then an additional (48.83 x 10^6 acres)(225 pounds) = 5.49 x 10^6 tons at an additional total cost of $8.24 x 10^9 would be required annually. Thus, at least in terms of the sets of

assumptions employed here, the use of Holden's figures on liming requirements leads to an estimate of the annual cost of liming more-or-less similar to the value of the fishery that would be lost if fresh-water acidification were allowed to proceed unhindered.

The literature occasionally mentions two other alternatives to ameliorating the effects of extant pH. For example, Swarts, et al. (1978) and Leivestad, et al. (1976) find differences within fish of the same age and species with respect to their ability to tolerate low pH. This raises hopes for natural selection processes serving as a means to maintain natural populations, given, of course, that the rate of toxic acidification does not outstrip the rate of natural genetic improvement in tolerance.

Alternatively, the awareness of differences in acid tolerance among species raises the prospect, [e.g., Schofield (1976, p. 230)] for the selective breeding or genetic engineering of acid-tolerant individuals. These individuals would then be used to restock acidified waters. However, if acidification also has a major impact upon the populations of other fresh-water organisms, it is unclear what the restocked fish would eat other then their peers. A similar comment applies to all restocking programs, whether or not with acid-tolerant individuals. Almer (1978, p. 307) nevertheless mentions several species in acidified waters which have substituted surviving invertebrates for their usual food which consists of other fish. It thus seems worthwhile to make some estimates, even if exceedingly rough, of the cost of restocking fish populations.

Bennett (1971, p. 89) presents a table showing average standing crops cf various species of fish per acre of fresh-water. The species range from trout and channel catfish to suckers and carp. The table was constructed to show the relative masses of the 19 species listed, given that some species are usually seen in combination with other species. The average of the mean pounds per acre for the 19 species is approximately 36. If there are 19.14×10^6 acres of fresh-water lakes and streams in Minnesota and the states east of the Mississippi River, this implies that there are 689×10^6 pounds of fish in these waters. According to Lerner (1974, Table 332) the Federal government spent 298×10^6 in 1973 for the maintenance of fish and wildlife populations. Part of this money was spent to propogate and distrribut,, according to the U.S. Fish and Wildlife Service (1974, p. 27), 303×10^6 fish eggs. If, on average, 5 percent of these eggs survive to one pound adults, and if all the above 298×10^6 1973 Federal expenditures were for fish, then each pound of adult fish cost the Federal government $19.67 (= $28.91 in 1978 dollars). For obvious reasons, this is an overestimate. Nevertheless, even if the survival rate is increased and/or the cost per adult fish is reduced, a substantial annual outlay remains. Given the extreme crudeness of the calculation, the difference from the $10-11 \times 10^9$ estimated annual value of a loss of the

entire game fishery in the region of interest does not seem suffcient to conclude, tentatively or otherwise, that the annual value of the impact of acid precipitation upon aquatic ecosystems is the cost of restocking game fish populations.

Health Effects

The Safe Drinking Water Committee (1977, p. 439) defines "hard" water as that containing 75 mg/liter or more of calcium carbonate or the equivalent. Increased hardness is indirectly associated with elevated pH. Although the Committee does not adopt an unequivocal position, it does state that the body of evidence for "soft" water being a causal agent in cardiovascular disease "... is sufficiently compelling so that 'the water story'is plausible . .." (p. 447). Soft water has a relative lack of inorganic solute health-support-ing agents such as calcium, magnesium, and manganese, and a relative abundance of health-degrading metal agents such as cadmium, lead, copper, and zinc. In addition, the effectiveness of several standard methods (chlorination, filtration and sorption, etc.) for reducing concentrations of bacteria, viruses, and protozoa in water intended for human internal consumption some-times varies directly with pH.

Our brief review of the available evidence makes us reluctant to map disease incidence into the pH levels of drinking water. Nevertheless, it is worth noting that quite small incidence due to the inorganic and organic solutes and the microbiological agents whose human health-degrading potentials are activated by low pH levels can result in large economic losses. For example, about half of the two million annual deaths in the United States are attributed to the various cardiovascular diseases. If only one percent of these one million deaths were indirectly attributed to low pH drinking water supplies, a toll of 10,000 deaths would result. In this context, it is worth noting that the Safe Drinking Water Committee (1977, p. 447) states: "On the assumption that water factors are causally implicated, it is estimated that optimal conditioning of drinking water could reduce this annual cardiovascular disease mortality rate by as much as 15% in the United States." Recent economic research [e.g., Thaler and Rosen (1975)] indicates the value of safety from death. to be about $500,000 - $1,000,000 in 1978 dollars. Using the lowest point in this range along with the one percent mortality assumption, would then result in annual economic benefits of 5×10^9. This figure would be increased substantially if one were to account for the losses in life- cycle earnings (and implicitly in labor productivity) due to cardiovascular diseases that do not now and perhaps never will result in death. Bartel and Taubman (1978), while working with a panel of 40-50 year old male twins, found that those with cardiovascular diseases had their annual earnings reduced by 20 - 30 percent relative to their healthy peers. According to the National Center for Health Statistics, 15.7 percent of the

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1976 U.S. population suffered from cardiovascular diseases [Lerner (1978, p. 120)] .

The preceding makes it appear that the health impacts and consequent economic effects of reduced pH in water used for internal human consumption could readily be very considerable. This assumes, however, that no ameliorative measures are available to either the consumer or the supplier of the water. If at a cost less than the value of the health effects, these ameliorative measures are able to raise pH to the levels that would exist in raw water supples in the absence of above-background acid deposition levels, then the proper health effects benefits to ascribe to the control of acidifying deposition are the ameliorative measure costs avoided.

Less than one percent of the water used by a community is consumed internally by humans [Safe Drinking Water Committee (1977, p. 104)]. Assuming that water consumption and fluid consumption are at most trivially different, this amounts to 1,000 - 2,400 ml/day for adults under "normal" conditions, and 1,000 - 1,670 ml/day for children aged 5 - 14 years [Panel on Low Molecular Weight Halogenated Hydrocarbons (1978, p. 164)]. A representative figure for the entire U.S. population might be 1,900 ml/day or 2 U.S. gallons per capita/day. In 1970, 29.42×10^9 gallons were withdrawn in the U.S. daily for rural domestic (2.39×10^9 gallons) and municipal (27.03×10^9 gallons) uses [Economic Research Service (1974, p. 37)]. Of these withdrawals, it is probably safe to assume that all were treated as if they were to be used for internal human consumption. Assuming no differences in per capita withdrawals for rural domestic and municipal uses across the United States, 1970 withdrawals in Minnesota and in the states east of the Mississippi River were 19.12×10^9 gallons/day. Now make the strong assumption that all ground and surface waters east of the Mississippi had pH levels (and associated inorganic and organic solutes and microbiological agents) requiring the addition of 21.6 kilograms of lime per million liters (per 264,175 gallons). This figure, which is due to Randall, et al. (1978, p. 66), refers to a thinly populated region in eastern Kentucky whose water supplies suffer from acid mine drainage. It therefore possibly has a low pH problem resembling that which would occur elsewhere if acid deposition became severe and widespread. Moreover, because of its small human population, the area is unable to fully exploit economies-of scale in its water treatment activities. Its unit costs of treatment are thus likely higher than in larger urban areas.

Randall, et al. (1978, p. 66) estimated the average market price of the "lime" used for water treatment in their study region to be $0.095 per kilogram, presumedly in 1976 dollars. In assigning a cost to the liming treatment, they did not attribute any additional costs to the maintenance and operation of treatment facilities since the facilities would be required even if there were no low-pH problems. Assuming constant unit and, therefore,

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marginal costs, the total cost of providing lime treatment for 264,175 gallons
of raw water would be $2.05. Given our previous assumptions about the
distribution of the U.S. population and water withdrawals for rural domestic
and municipal uses, this implies that the daily cost of lime treatments for
raw water supplies east of the Mississippi River would be about $148,000.
This amounts to an annual cost of $54 x 10^6 1976 dollars or $62.1 x 10^6 for
1978. Of course, given the existence of acid mine drainage in important
watercourses of the region as well as natural acidification due to soil
leaching, much of this cost burden would have to be borne independently of any
acid precipitation problem. Thus, the external costs (the environmental costs
of increased mining activity for $CaCO_3$ and its equivalents and the negative
health effects) of increased liming of raw water supplies would have to be
extremely large (at least as much as the value of the negative health impacts
of increased acidity) to justify a refusal to ameliorate the health impacts of
acid deposition by the liming of raw water intended for internal human
consumption.

Household, Commercial, and Industrial Water Supply System Effects

Reductions in the pH levels of water supplies may cause corrosion in
household, commercial, and industrial water conveyance systems and water-
using appliances, thereby shortening their useful lifes and reducing the flow
of their services while in use. In the absence of ameliorative measures, the
potential economic losses from this corrosion could be severe. On the other
hand excessively high pH levels can have similar effects due mainly to mineral
deposits forming on the interior surfaces of the systems and appliances.

Several studies are available that assess the impact of increased levels
of total-dissolved-solids (TDS) and/or water hardness upon the economic life-
times of household and commercial water supply and use systems. Using an
eight percent discount rate, d'Arge and Eubanks (1976) estimate 1975 economic
loses for a typical Los Angeles household to range from $620 to $1,010 in
present value terms for an increase in total dissolved solids from 200 to 700
mg/l. This estimate is three to four times higher than estimates developed by
Tihansky (1973) for a similar TDS range throughout the United States. In an
appendix to their study, d'Arge and Eubanks (1976, pp. 274-275) used data from
Black and Veatch (1967) to explore the extent to which the ratio of TDS to
total hardness was important to the useful lifetimes of household conveyance
systems and water-using appliances. They found that increases in the ratio
made a statistically significant positive contribution to the lifetime of
garbage grinders and a statistically significant negative contribution to the
lifetime of wastewater pipes. Total hardness, when entered as a separate
explanatory variable in a multiple regression, was negatively associated with
the useful lifetime of faucets but the association was not statistically
significant. It thus appears that the acidification of water supplies, to the

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extent that it "softens" water in areas with "hard" raw water, could have economically beneficial effects upon water conveyance systems and water-using appliances. Nevertheless, if the effects of excessive "softness," as induced by low pH, upon useful life are more-or-less symmetrical to those of excessive hardness, the economic impacts upon households, and commercial establishments could be considerable.. Since the ferro-alloys, copper alloys, and brasses used in household and commercial water supply systems and water-using appliances are also found in industrial systems and equipment, substantial economic impacts could also be expected in these sectors.

In spite of the potentially large economic impacts of low pH water upon household, commercial, and industrial. water supply systems and water-using appliances and equipment, it does not appear useful to try to calculate the magnitude of these impacts. The reason is that inexpensive neutralization techniques using hydrated or calcined limes are readily available.

In the health effects section, we have calculated the cost of liming rural domestic and municipal water supplies. According to Todd (1970, p. 312), the "optimal" pH levels for domestic water supplies are about neutral (pH = 7.0), although (p.320) the median for the 100 largest U.S. cities in 1962 was pH = 7.5. Thus, given that all rural domestic and municipal water supplies are treated as if intended for internal human consumption, acidification of raw water will have no extraordinary effects upon household and commercial conveyance systems and appliances.

Many industries supply and treat their own process water. The USDA Economic Research Service (1974, p.37) estimates that self-supplied industrial water withdrawals (excluding steam-electric power) from fresh surface and ground sources in the U.S. in 1.970 were 45.87 X 10^9 gal./day. According to Todd (1970, pp. 329330), most industrial processes require or are indifferent to water with pH in the 6.0 to 9.0 range. In 1968 [Todd (1970, p. 221)], about 72 percent or 33.03 X 10^9 gals./day of these withdrawals occurred in the states east of the Mississippi River and in Minnesota.

Assume that all of the self-supplied industrial fresh water east of the Mississippi and Minnesota has been acidified to the 4.0-4.5 pH range prior to withdrawal. Further assume that in order for it to be used as a process water its pH must be raised to an average 7.5-8.0 across industries, and that it would otherwise require no treatment prior to use. Thus , excluding the possibility of tying into municipal systems, each plant will have to construct and operate its own treatment facility. Our presumed necessary increase in pH happens to correspond to the pH increases experienced with several acid mine water treatment plants. For example, Bituminous Coal Research, Inc. (1971, pp. 133-134) reports total 1970 capital and operating costs of 97.3, 35.3, and 26.5 cents/1,000 gals. with a plant respectively processing 0.1, 1.0, and 7.0

million gallons/day. They also report on another plant which experienced 1970 total capital and operating cost of 13.6 cents/1,000 gallons for treating an average of 4.0 million gallons/day. In each case, the pH of the acid mine water was raised from about 4.5 to more than 7.0. Barton (1978, pp. 351-354) summarizes the experiences of one mine where water were raised from 4.0 or less to more than 7.5. The commercial operation with 12,800 tons of lime, processed 3.4×10^9 gallons of water with an original pH of 4.0 and a finished pH of 7.9 at 20 cents/1,000 gals.

Using a very fine and therefore more costly limestone slurry, the pilot plant raised a mine discharge of 2.8 pH to 7.4 pH at estimated 1970 capital costs of $55,000 to $766,000 for 100×10^3 to 600×10^6 gals/day operating capacities. Estimated 1970 operating costs, including amortization, were respectively 44 cents to 2 cents/1,000 gals. In the words of Barton (1978, p. 352) :

> "Limestone could be the preferable choice for treating nearly all
> but the most severly loaded discharges. It has the advantages of avail-
> ability, lower cost, reduced hazards, ease of application, simplicity of
> plant design, impossibility of water overtreatment, ease of storage, and
> higher solids concentration of the precipitated sludge."

A review of Todd (1970, pp. 246-274) makes it appear that the median water-using industrial establishment withdraws about 1.0 million gals./day. We therefore estimate, on the basis of the material presented in the preceding paragraph, that, including amortization of capital facilities, a representative 1970 total cost of raising 4.0 pH fresh water to 7.5 pH would be 50 cents/1,000 gals./day. On a yearly basis, therefore, the 1970 total annual cost of treating the 33.03×10^9 gals./day of self-supplied industrial water withdrawn east of the Mississippi could be 5.66×10^9 (=9.51×10^9 for 1978). Even though it is fair to presume that the unit cost of lime and treatment plants might increase with an increase in demand of the magnitude posited here, it should also be recognized that the posited increase in demand is also probably vastly exaggerated. Not all fresh water east of the Mississippi is likely to suffer a reduction in pH to 4.0 or even 5.0. Many industries are fairly indifferent to quite low PH. For example, in a survey of the impact of acid mine water upon patterns of industrial water use in the Appalachian region, Whitman, et al. (1969) found that the most impacted industry, primary metals, saw fit to raise the pH level of its cooling water only to 5.0. This was accomplished at minor cost by integrating lime treatments with otherwise existing water treatment facilities. Todd (1970, p. 345) makes it appear that possibly half the industries in the United States in 1959 had their own treatment facilities. Finally, we have not considered possible substitutions from fresh to saline water sources. [7/] The stated estimate might therefore readily be exaggerated by more than an order of

magnitude. One can be quite sure that it is not biased downward. Given the industrial treatment facilities already in place, we prefer to treat as trivial the additional costs of treatment attributable to acid precipitation.

Materials Effects

Several studies of the economic impact of air pollution upon the useful lifes of materials have been published, e.g., Gillette (1975), Commission on Natural Resources (197 , pp. 616-619, 695-699), Kucera (1976), and Salmon (1972). Nriagu (1978) presents an extremely thorough review of the physical science literature on the effects of sulfur pollution on materials. A brief review of the material effects of nitrate aerosols is available in Panel on Nitrates (1978, pp. 417-418).

Acid precipitation (or acidifying deposition) accelerates the decay rates of a wide variety of materials mainly because the presence of acids upon the material surfaces increases the flow across the surfaces of the electric currents that cause corrosion, discoloration, and embrittlement. Among the metals, ferro-alloys, copper, and some galvanized metals are known to be particularly susceptible. In some cases (e.g. zinc), the dissolution of the metal surface by acid precipitation is thought to increase the pH level of the product, thus resulting in an even more corrosive surface film. Carbonaceous building materials, such as limestone and cement, are more rapidly weathered, roughened, eroded, and stained. Paints are bleached and crystallized, and their drying and hardening times are increased. The tensile strength of textiles is degraded and textile dyes can suffer from fading. Losses of tensile strength also occur in paper, as does discoloration. Other cellulose products, such as wood, suffer similarly. Leather products deteriorate because the acids break down their fibrous structure. As Nraigu (1978) emphasizes, these processes are further intensified for those materials, such as cement, concrete, and some metals, often used in subaqueous and/or high temperature environments.

The recent economic impact studies of air pollution upon materials have yielded estimates for the entire United States of losses ranging from the $904 x 10^6 Gillette (1975) attributed to sulfur oxides in 1968, to the $3.8 x 10^9 Salmon (1972) attributed to all air pollution in 1970. None of these studies provides any substantial basis for attributing a portion of their estimated losses to acid precipitation, although the decline in sulfur dioxide levels throughout most of the eastern United States during the 1970's implies that an increased portion of whatever materials damages are occurring is attributable to acid precipitation. Most important, since all these studies basically do little more than inventory some existing materials, attach a market price to them, and then use physical science estimates of increases in replacement frequency to obtain a total loss estimate, they are susceptible to all the

criticisms that can be directed toward most of the estimates In this chapter. As Glass (1978, p. 34) correctly points out, many extremely resistant materials, such as aluminum clad steels, have been widely adopted in the last decade; most estimates relate to uncoated rather than coated galvanized steels; the economic lives of many materials are so short (e.g., paper) that air pollution does not have time to affect them in a noticeable fashion; and, that when a substitute material. is adopted, the cost differential often cannot be assigned entirely to pollution since the substitute may have features that reduce cost dimensions other than useful life. In addition to these factors, the available studies sometimes have failed to discount the stream of costs properly. Moreover, all the studies have failed to consider that individuals may choose simply to bear a reduced stream of services from a material rather than purchasing a replacement, may alter behavior patterns so as to compensate for the stress that acid precipitation imposes upon the material, and may adopt materials more resistant to the ravages of acid precipitation. Finally, entire categories of useful materials such as limestone and concrete structures, including dams and pipes, and automobiles have had no economic attention devoted to them. Given the lack of economically useful physical science information and the lack of sound economic information, it is tempting to plead an absence of any basis whatsoever to make a judgment about either the fact or the potential for the economic impact of materials damages from acid precipitation and acidifying deposition. This is particularly so because many of the factors for which information is lacking can have either a positive or a negative economic impact. Of those factors that are most likely positive, or most likely negative, it is impossible to tell which will dominate. One is thus unable to state whether any estimate of the total (or marginal) impact represents an upper or a lower bound.

In spite of the preceding, it should be recognized that the costs of acid precipitation-induced materials decay could indeed be very substantial. The Commission on Natural Resources (1975, p. 696) refers to studies which estimated 1970 damages in Sweden to painted steels from all corrosion of $25.00 per capita (= $41.98 in 1978 dollars). From the same source, the Commission (197) quotes $23.00 per capita (= $38.64 in 1978 dollars) as being the total annual cost of deterioration of painted woodwork associated with all sources of deterioration. The studies from which these figures come appear to have at least as complete a physical science basis as any available, and no worse an economic basis than any of the extant studies.

Other than the direct and indirect products of chemical weathering, soiling is the major source of reductions in the usefulness of materials. Some of what passes for soiling (e.g., staining of the exterior stone surfaces of buildings) may, in fact, be chemically-induced discoloration. We thus presume that, in economic terms, soiling is relatively minor as opposed to chemical weathering. Some enhanced chemical weathering occurs to materials

located near marine environments. This, however, is probably not an important source except for those materials frequently exposed to sea breezes and/or salt sprays. Some chemical weathering would naturally occur in humid areas since pristine precipitation is somewhat acidic ($pH = 5.65$). All these factors suggest that the aforementioned annual per capita costs in Sweden of the weathering from all sources of painted steels and painted woodwork are exaggerations of the 'losses caused by the impacts of acid precipitation upon these materials. However, as we previously emphasized, these materials constitute only a portion (though not a small portion) of the economically significant materials susceptible to acid precipitation-induced decay.

To generate a number for the materials damages caused by acid precipitation, we assume that the current per capita exposures of the great bulk of the Swedish population is very similar to the per capita exposures of the population in the eastern part of the United States. We further assume that the per capita mixes and magnitudes of painted steels and woodwork used by United States residents residing in Minnesota and east of the Mississippi River are similar to those of the Swedes. A simple multiplication of the sum of $41.98, for painted steels, and $38.64, for the painted woodwork, by the approximately 170×10^6 people residing in Minnesota and the states east of the Mississippi River in 1978, yields a calculated annual loss from materials damages of 13.71×10^9. It should be noted that this figure is an order of magnitude higher than previous estimates of all air pollution-induced materials damages over the entire United States. However, given both the physical science, economic, and inventory accounting limitations of the previous estimates (and this estimate), it seems as likely to be an underestimate and an overestimate. Nevertheless, given the difficulty and trivial gains to us in trying to justify the discrepancy between the above weak estimate for materials damages and those obtained by previous investigators, we do not deem this exercise to be a good forum for a display of intellectual stubbornness. We, therefore, state that materials damages are likely to be the largest category of the types of acid precipitation-induced damages we have surveyed, but we have no wish to assign to acid precipitation all materials damages that previous investigators have attributed to air pollution. We, therefore, set the 1978 materials damages caused by acid precipitation at 2×10^9, while recognizing that the figure could plausibly be much larger.

Summary and Conclusions

Although most of the analysis has been rather primitive, the economic benefits likely to accrue to a variety of life and property forms from the control of acid precipitation have been surveyed. It must be recognized, given the robust techniques available for doing economic assessments of the effects of acid precipitation, that the estimated magnitudes presented in this chapter cannot be justified indefinitely.

On the basis of our survey and synthesis of a fairly large volume of biological literature and stock, price, and output information, we conclude that if sufferers are viewed as either having to accept it or to take actions at their own expense to negate its effects, it is very unlikely that the current annual benefits of controlling acid precipitation for existing economic activities exceed 5×10^9 in 1978 dollars in Minnesota and the states east of the Mississippi River. [8/] Our best estimates are that 2×10^9 is in materials benefits, 1.75×10^9 is in forest ecosystem benefits, 1×10^9 is in direct agricultural benefits, 0.25×10^9 is in aquatic ecosystem benefits, and 0.10×1.0 is in other benefits, including health and water supply systems. The rationales supporting each of these sector estimates are presented in the chapter text. With the exception of a few instances where analogies could be drawn with the results of other studies using more robust estimation techniques, all these estimates disregard acid precipitation-induced price, activity, and location changes. We therefore have substantially more confidence in the rank-ordering by sector of the current annual benefits than we do in our estimates of the absolute magnitudes of these benefits.

If acid precipitation events continue to worsen, certain sectors could readily climb in the above ranking. For example, aquatic ecosystems currently have a relatively low position only because the geographical scope and severity of the aquatic acidification problem does not yet seem to be large enough to reduce substitution possibilities greatly across fresh-water fishing and hunting sites. Because of the water and soil treatment facilities already in place that can readily be adapted to handle liming procedures, the acid precipitation control benefits accruing due to the prevention of human health effects, indirect agricultural effects, and household, commercial and industrial water supply system effects are now and are likely to continue to be insignificant compared to the other classes of effects. Large-scale liming of aquatic and forest ecosystems appears to be neither technically or economically feasible.

The preceding conclusions are not the major conclusions we wish to draw from our survey and synthesis of the acid precipitation literature. We are unconvinced that either the above ordering or the above absolute magnitude estimates of the current annual benefits of control (even if correct) constitute the really important issues to consider when evaluating the acid precipitation problem. Indeed, we are unable to reject the discomforting notion that the effects for which one may feel secure using these simple or the much more sophisticated but still conventional methods of economic analysis reviewed in Chapter I are those having the least long-term economic significance. Instead, we suspect that these important issues relate to the impact of acid precipitation upon the stock and the assortment of natural resources. The next two chapters consider the implications of some of these

issues regarding resource stocks and assortments for assessments of the benefits of controlling acid precipitation.

REFERENCES

1/ It should be noted that there may be some exceptions to acid precipitation acting as a sort of negative fertilizer. For example, Maugh (1979) reports on a TVA-sponsored study which found that if the sulfur emitted by coal- burning power plants in the Tennessee Valley region were removed, and not replaced by another sulfur source, crop production, especially cotton, would decline by at least 10 percent. Tisdale and Nelson (1976, p. 411) point out that raising soil pH in the Deep South to more than 6.0 may actually be harmful to yields.

2/ See Freeman (1.979) for a presentation of the conditions under which it would be a good approximation. For the Adams, et al. (1979) study, the on-farm value of the 14 crops was 16 percent less than the estimated sum of producer rents and consumer surpluses. When cotton was excluded the non- farm value of the 13 remaining crops was 20 percent less than the estimated sum of the producer and consumer surpluses.

3/ In a news item, Rich (1979) reports that field studies in southern Poland have attributed drops of 13 to 18 percent in the photosynthetic activity of pine needles subjected to wet and dry sulfur deposition. Dennis Knight of the Department of Botany at the University of Wyoming informs us that the Polish investigators believe that this reduction is due to SO_2 entering the leaf through the stomata and then being converted to H_2SO_4 within the leaf. This perspective may be contrasted with the bulk of the published literature which emphasizes the growth reducing properties of cuticular erosion and nutrient leaching from leaves and soil. Apparently, the Polish studies have not yet been widely distributed.

4/ In principle, the spreading of sufficient lime on top of forest soils might raise pH before precipitation moves down the soil column. Other than a vague statement by Rich (1979) on aerial lime spraying in Poland, we have found no commentaries on either the technical or the economic feasibility of this practice.

5/ If the forest growth effects of acid precipitation are viewed as analogous to a selective cutting policy, one could draw upon the technical forestry literature relating the effects of this type of cutting upon these components. We have not exploited the analogy here because of the likely wide

variations in responses of the components to tree species mixes, topographical attributes, and other factors.

6/ Let V be the present value ($15.32 x 10^9$) of the stream of losses, let A be the annual losses ($9.69 x 10^9$), and let r be the rate of discount. Then:

$$V = A \left(\frac{1+r}{r}\right)[1 - (1+r)^{-\infty}].$$

The term in brackets can obviously be disregarded when one is dealing with an infinite future.

7/ According to the Economic Research Service (1.974, p. 37), 1970 self-supplied industrial water from saline sources in the United States was 10.07 x 10^9 gallons/day. It is unclear, however, how this use is distributed over ocean, estuary, and saline groundwater sources. The pH of any saline source could obviously differ greatly according to the extent to which the acidic fresh-water had been diluted by the saline water.

8/ In Hick's (1973) terms, the qualifying "if..." phrase indicates that an equivalent, as opposed to a compensating, measure of value is being employed. In effect, it is assumed that those who cause acid precipitation, rather than those who suffer from it, have the de facto property rights to the air resource. Moreover, since all our crude assessments are in willingness-to-pay terms, they will be less than if the assessment had been made in willingness-to-accept compensation terms [Randall and Stoll, (1980)].

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III. DECISION PROBLEMS IN THE CONTROL OF ACID PRECIPITATION:
NONCONVEXITIES AND IRREVERSIBILITIES

Acid Precipitation Dose-Response Functions

The sole recurring theme of the previous two chapters is that empirical application of economic methods for assessing the benefits of acid precipitation control generally requires prior knowledge of the response of biological and material entities to variations in acid precipitation exposures, Increasing pollution has been treated as leading to progressive deterioration in the size of the resource stock and the flow of material and life support services issuing from it. Moreover, this deterioration could be reversed and, by reducing the level of pollution, recovery could occur along the same path as did deterioration. This behavior is a standard representation in the environmental economics literature. It leads to results in which some immediate environmental damages are borne in order to obtain some of the immediate benefits that a productive but polluting activity confers. Assuming the pollutant to be acid precipitation, Figure 1 introduces the costs of controlling the acid precursors in a comparative static version of the standard representation. Unit prices of the elements of the resource stock and of pollution control equipment are assumed constant.

The economically efficient pH level in Figure 1 will. be at A, where the marginal benefits of reduced acidity are equated to the marginal costs of controlling acidity. The marginal benefit of reduced acidity is the marginal damage that is avoided by having less acidity. That a point such as A is optimal can be seen from the following simple argument. Suppose that the benefits are measured by the size of the fish population denoted Pop. The Pop is an increasing function of the pH level as is the cost of control, C. The net benefit of a given pH level is

$$\pi = Pop(pH) - C(pH) \qquad (1)$$

This expression is maximized when its first derivative is set equal to zero, that is:

$$\frac{d(PoP)}{d(pH)} = \frac{dC}{d(pH).} \qquad (2)$$

65

Figure 3.1
The Standard Representation

d(Control Costs) / d(pH)   d(Damages) / d(pH)

$

pH   Greater Acidity ⟶   A   0
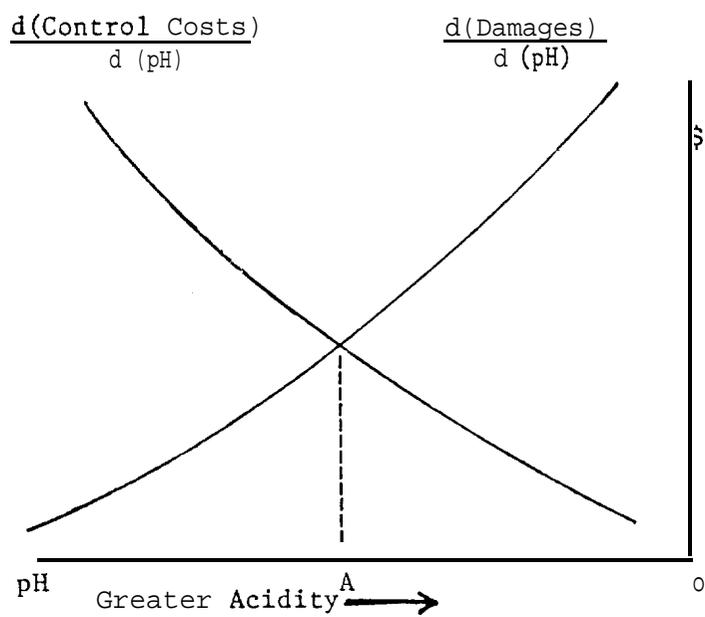
The left hand side is the marginal benefit (or marginal damage avoided) and the right hand side is the marginal control cost. In terms of Figure 1, for states to the left of A, the additional costs of control exceed the additional benefits of reduced damage; to the right of A, the opposite is true. A decisionmaker who wishes to maximize net economic benefits will, therefore, be striving for a point such as A. If instead the vertical axis represents the present value of a stream of expected damages and control costs, he will also strive for A, given independent damages and control costs across periods. In short, whether observed or inferred by benefit-cost analysis, the "prices" of additional damages or additional controls given to the decisionmaker in the neighborhood of an initial acidity state will always be a signal to move toward that state maximizing the net benefits of control.

There exist at least two reasons why the form of the underlying ecosystem dose-response function in the preceding figure may be inaccurate insofar as acid precipitation is concerned. The nature of the inaccuracies implies that the rationales usually offered for compromising between the benefits of pollution-generating activities and the prevention of ecosystem damages may not always be applicable to acid precipitation issues.

The All.-or-Nothing Feature: Nonconvexities

The preceding analysis has presumed that, within any one period, the increments to ecosystem damages are monotonj.tally increasing with respect to ecosystem acidity. At least insofar as fish and some other aquatic organisms are concerned, this presumption is contrary to some published evidence [Raddum (1978)]. Consider, for example, the following two tables constructed from data appearing in the study of Butler, et al. (1973) on the impact of acid mine drainage upon fish and other organisms in Pennsylvania streams.[1] For varying sustained pH levels, Table 1 shows the number of stream sections that had fish populations out of 25 sampled sections in different streams; while Table 2 shows, of the 116 fish species known to exist in Pennsylvania as of 1957, the variation with respect of pH of the number of species in these stream sections. Table 2 also indicates the pH levels at which assorted game and food fish disappeared due to lethal effects and/or recruitment failures. Both tables exhibit very rapid declines in fish populations once pH drops below 6.5. However, this decline itself rapidly decreases $[(\partial^2 \text{Population}/\partial PH^2)>0]$ as PH levels reach and drop below 6.4. Assuming that the implicit unit price of remaining fish and species is a constant, Figure 2 is a sketch of Tables 1 and 2.

Returning temporarily to Figure 1, in order that A be a maximum' rather than a minimum, it is necessary that,

Table 3.1-Sections with Fish at Various PH Levels for a Sample of Pennslyvania Streams Suffering from Acid Mine Drainage

pH	Stream Sections with Fish
6.4	24
6.3	12
5.9	6
5.3	3.
4.6	1
4.5	0

From: Butler, et al., (1973, P.112)

Table 3.2-Variation of Number of Fish Species with Respect to pH Levels for a Sample of Pennslyvania Streams Suffering from Acid Mine Drainage

pH	Number of Species Present	No Longer Present
≥ 6.5	116	
$6.4 \leq pH < 6.5$	48	Catfish, smelt
$6.2 \leq pH < 6.4$	41	Redfin pickerel
$6.1 \leq pH < 6.2$	36	
$6.0 \leq pH < 6.1$	34	Smallmouth Bass
$5.9 \leq pH < 6.0$	18	Brown Trout
$5.6 \leq pH < 5.9$	12	
$5.5 \leq pH < 5.6$	10	Yellow perch
$5.2 \leq ph < 5.5$	9	
$5.0 \leq pH < 5.2$	8	Brook Trout
$4.7 \leq pH < 5.0$	7	Largemouth Bass
$4.6 \leq pH < 4.7$	5	Chain pickerel
< 4.6	0	

From: Butler, et al., (1973, Pp. 96-99, 114)

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Figure 3.2
The Nonconvexity Problem
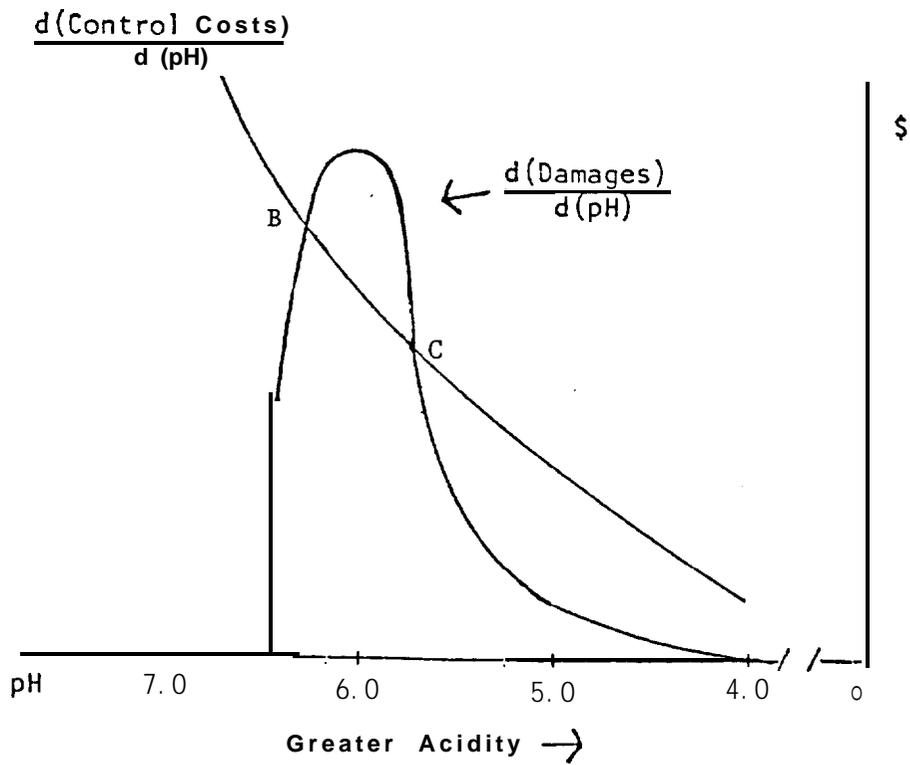
$$\frac{d^2(Pop)}{d(pH)^2} \quad \frac{d^2C}{d(pH)^2} \leq 0 \qquad\qquad (3)$$

and sufficient that

$$\frac{d^2(Pop)}{d(pH)^2} \quad \frac{d^2C}{d(pH)^2} < 0 \qquad\qquad (4)$$

A further sufficient condition is

$$\frac{d^2(Pop)}{d(pH)^2} < 0, \text{ and } \frac{d^2C}{d(pH)^2} > 0, \qquad\qquad (5)$$

as Figure 1 presumes.

The data in Table 2 shows that $d^2(Pop)/d(pH)^2 > 0$. Hence the sufficient conditions may not be satisfied at a point where marginal benefits are equated to marginal costs.[2/] The second order necessary condition could be violated turning such a point into a local minimum rather than a maximum. This is what happens at point C in Figure 2. The observed or inferred current prices existing at and to the right of C provide unreliable signals about whether the decisionmaker is at a maximum. or minimum and the direction in which he must move in order to obtain an increase in net benefits.

It is also evident in Figure 2 that if the environment were already highly acidified, a large cost burden with relatively few benefits would have to be borne prior to reacquiring the benefits of a relatively nonacidified state. Thus , given limited restoration resources, it may no longer appear worthwhile to restore the nonacidified state. As a result, decisions to control acid precipitation may have strong "all-or-nothing" elements: inter-mediate control measures can lead to burdensome control costs while generating few environmental benefits. Literal interpretations of prices applying to initial states lying at and to the right of C in Figure 2 would guarantee high levels of acidification: the ecosystem destruction wrought has been so great that the benefits from reduced acidification appear minor. To use an extreme example, the benefits from increasing fish recruitment cannot appear large when there are no fish around who can reproduce. Only by undertaking the far more arduous and complex task of empirically accounting for the ecosystem and economic adjustments and consequent changes in price structure resulting from

Figure 3.3
Possible Time Path of Acid Precipitation Effects
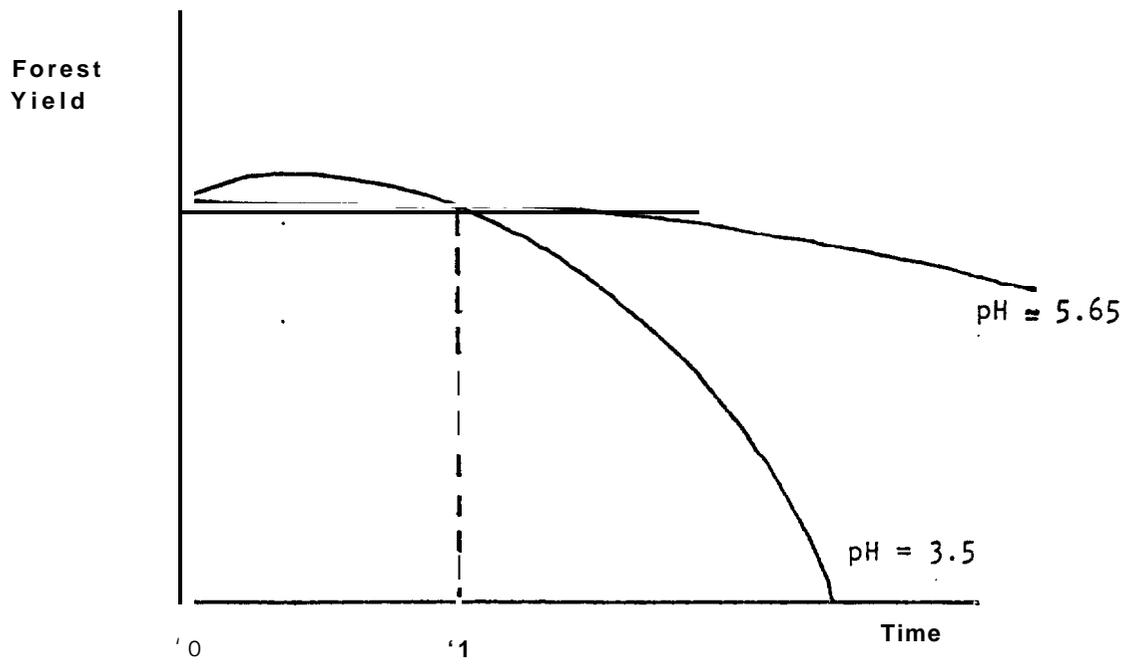
a large move from an initial state at or to the right of C to a state in the vicinity of B could the benefits of reduced acidification be captured. [3/]

The Now-or-Never Feature: Irreversibilities

In the above, we have remarked on the distorted picture of reality that market or market-like price signals can introduce when the incremental damages of acidification within a period are declining. It is argued that if a state of high acidification is reached, the decisionmaker must expand the scope of his vision and analysis to include discrete rather than marginal alterations in existing states. Given this scope, we have presumed that he is able to reverse the current and future consequences of current and past acidification so that acidification levels henceforth remain in the vicinity of B. In short, we have presumed that the marginal damage function in Figure 2 is invariant with respect to both the status quo point and the direction of movement. The presumption appears to be incorrect for the effects of acid precipitation upon many components of forest and aquatic ecosystems.

Figure 3 is consistent with findings which attribute via soil amendments stimulator [Lee and Webber (1979); and Maugh (1979)] and debilitating [Jonnsson and Sundberg (1972); Tamm (1976)] effects upon plant growth to acid precipitation. In Figure 3, it is assumed that over some decade-or-longer period, a forested region is annually subjected to precipitation averaging pH ⁼3.5. The line labelled ∿5.65 refers to a "no acid precipitation regime. It decays slowly because of the natural tendency over millennia of soils in humid regions to become acidified. Under an acid precipitation regime, where forest management practices and influential factors other than acid precipitation are assumed invariant, the line labelled pH = 3.5 becomes relevant. Acid precipitation thus accelerates the natural tendency over time of soils to become acidified, as McFee, et al. (1976) and Peterson (1980) emphasize.

Over the t_1 - t_0 interval, the acid precipitation (or acidifying deposition) is neutral with respect to or contributes positively to forest yields. The sulfur and nitrogen compounds in the precipitation can directly and indirectly enhance the nutrient content of the forest soils. After t_1, however, the atmospheric inputs of positively charged hydrogen ions are greater than the ability of the forest soils to neutralize them. Organic and mineral nutrients are then leached from the forest soils at a rate more rapid than they can be replaced from atmospheric, decomposition, weathering, and microbial sources [Likens (1977)]. Simultaneously, phytotoxic metals, such as soluble inorganic aluminum and iron, are made more available and organic matter accumulates to seal the upper layers of the soil column while permitting various phytotoxins to be formed from the matter [Brady (1974, Chap. 14)]. As time passes, with the frequency and intensity of acid precipitation invariant, the rate of nutrient leaching and phytotoxin

formation accelerates [McFee (1978, p. 66)]. In turn, this is expected to cause forest yields to decline at an increasing rate. Moreover, since water bodies serve as catchment basins for land areas, they too are expected to experience increases in hydrogen ion concentrations and heavy metals.

Once levels of forest soil (and water body) acidification beyond t_1 are widespread, there is no evidence that large-scale reverses are economically (or even technically) feasible in anything other than geologic time. The addition of lime to acidified soils and water bodies is the only widely considered technical remedy. It is, of course, a commonly used remedy in agriculture. However, as Tisdale and Nelson (1976, p. 428) note, limestone particles cannot move in the soil and must therefore be placed where they are needed in the soil column. Rorison (1980, p. 206) remarks that isolated additions of lime to acid sulfate soils are of no lasting value. Tilling lime into extensive areas of forest soils with standing trees would seem an economic, if not a technical, impossibility. $\underline{4/}$ Moreover, Tamm (1976, p. 338) adds that when lime has been added to forest soils in small-scale experiments, tree growth rates have typically not been enhanced. He attributes this to the tendency of the lime to immobilize the nitrogen in organic matter and thereby reduce its availability to trees. Abrahamsen, et al. (1980, p. 357) found that soil animal populations nearly always failed to increase when soil acidity was reduced by liming.

The practicality of large-scale liming to resolve the problems life forms have in acidified water bodies appears to be no better than for forest soils. As Holden (1979, p. 11) emphasizes, the effective use of lime to raise the pH of natural water bodies requires a great deal of information about the hydrological and chemical properties of each body of water. He notes that most of the added lime is flushed out, fails to dissolve, or remains in the sediment. Reactivity of the lime will vary with its purity, particle-size, hardness, magnesium content, chemical constituents and stratifications of the water body by season, and a host of other factors.

Finally, according to Dickson (1978, p. 58) and Bengtsson (1980), care must be taken when raising aquatic pH levels to ensure that they are not allowed to persist in the 4.5-6.0 range. Heavy metals, particularly the inorganic aluminum that acidified waters contain in abundance, becomes especially toxic to ol-der fish. Thus liming must be calibrated for the state of the fish as well as for the state of the water. This toxicity is dramatically illustrated in Bengtsson's (1980) report on the successes of Swedish lake liming experiments. His data indicate that the perch in one lake before liming were all large and mature individuals. After liming, the number of perch increased by a factor of 100 but the size of the representative individual had declined by a factor of 10. Liming redresses the balance of harm by destroying the older and larger fish surviving the original acidification. As Bengtsson (1980, p.35) states, "when liming an acid lake the organisms suffer

a transition period before the metals have been precipitated . . . liming has even killed salmon and trout when the aim was to save the fish." This problem is further substantiated by Professor Harold Harvey a zoologist at the University of Toronto. In a July 15 article on the "Acid Lakes" in the Toronto Globe and Mail, Professor Harvey is quoted as saying, "Noone knows to what degree of certainty what liming will do." He adds that indiscriminate liming "may improve the pH and end up killing all the fish" by setting off a chemical reaction involving lime and heavy metals in the lake. If continuing acidification requires intermittent liming, Bengtsson's (1980) data are therefore consistent with an inability to retrace the damage function relevant to declining pH.[5/]

This litany of complexities in deciding how much lime an acidified water body requires and when this lime is required by no means suggests that wide-spread and large-scale intermittent or continuing liming is likely to be economically attractive. At best, the litany suggests that a subtraction of the expected costs of successful liming from the marginal damage (marginal benefit) curve of Figure 3 will drastically lower the curve when the status quo point is less than pH = 6.0. Thus for given marginal costs of reducing actual emissions, even though pH may vary over the same interval, the net benefits of recovering the pH at the high end of the interval will be less than the net benefits of preventing a decline of pH to the low end of the interval.: the marginal benefits of ranising pH are less than the marginal benefits of preventing declines in pH.[6/]

As is true for nonconvexities, the irreversible features of acid precip-itation-induced ecosystem deterioration can mean that current prices will provide misleading signals about the most valuable corrective steps to take.

Standard economic representations of the efficient depletion of environ-mental or other assets require that the present value of the gains from further depletion in any period be equal to the sum of the depletion losses and interest charges. When property rights to the resource are secure, this implies, as Scott (1973) has shown, that the present value of the marginal unit of depletion in each period must be the same in all periods; otherwise, gains could be obtained by shifting units of depletion from periods where their present value is lower to those where it is higher. Delays avoid the costs of engaging in the activity that causes the depletion, but they also require an increased waft for the benefits the activity yields. Given a positive discount rate, if the present value of marginal depletion units is to be the same across all periods, the current undiscounted value of the marginal unit in each subsequent period must be greater by the rate of interest than the current value of the marginal unit in the immediately proceding period. In short, the rate of increase in the value of the resource that remains will tend to approach the rate of interest. [7/]

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The above result explicitly weighs the impact of current depletion activities upon the opportunities for depletion that remain in future periods. This result can be contrasted with a situation where the depletion is reversible. Consider, for example, the agriculturist who acidifies his soils by the use of ammonia fertilizer amendments. He will simply allow acidification to continue until the current period net gain from further acidification no longer exceeds the current period net gain from. restoration accomplished by liming. Since his soil acidity can easily be reduced by liming at any time he wishes, there is no reason for him to weigh the impact of his current fertilization practices upon his future opportunities for growing corps. There are no future opportunity losses for him to count as a cost: he will therefore base his decisions only upon current market prices.

Acid precipitation accelerates the rates of depletion of the buffering capacities of forest soils and water bodies in addition to reducing the current flows of material goods and amenity and life support services from these resources. Any control plan which accounts only for the value of the reduction in current flows, as registered in actual or inferred current market prices, will thus underestimate the damages acid precipitation is causing. Stated in an alternative manner, even though the current net benefits of continuing acid precursor emissions may still be positive, it may be optimal to cease emitting. Some immediate losses must be borne in order to avoid the possibility of even greater losses later on that the current precursor emissions can readily cause.

If the benefits of acidification decline over time relative to the bene-fits of natural environments, the irreversible effects of acidification, when combined with a positive discount rate, lend the acidification issue a now-or-never character. This is most easily seen by assuming perfect foresight and by disregarding any short-term fertilization benefits. In particular with a positive discount rate, any delay in causing above-background acidification will only make it look progressively less attractive. Not only are the relative benefits of acidification declining over time by assumption, but the positive discount rate causes the present value of acidification benefits to be reduced with every delay. In the meantime, since the unacidified ecosystems already exist, the material goods and life support and amenity services they produce continue unabated. Therefore, cet. par., the net gains from ecosystem acidification will never be greater than they now are.

The presence of declining relative benefits of acidification (or increas-ing relative benefits for preserving natural environments) is a necessary condition for the above conclusion. As set forth by Smith (1974) and others, **two key propositions lead to a prediction of increasing relative benefits** for **natural environments.** First, environments that have remained unsullied by man's activities and artifacts are superior goods. That is, as real incomes

increase, the willingness-to-pay for natural environments increases at an even greater rate. Man-made substitutes become progressively less attractive. Second, because of the imperfect reproducibility of natural phenomena, technological change tends to reduce the supply prices for man-made goods relative to the supply prices for natural environments. The obvious general conclusion is that both the relative cost of supplying and the willingness-to-pay for natural environments are going to increase progressively over time. Both of these countervailing supply and demand forces imply that the citizenry will attach increasing values to natural environments relative to fabricated goods.

When combined with the fact that at present very little is known about many of the social, environmental, and financial consequences of ecosystem acidification, the irreversibility phenomenon introduces yet another basis for expecting declines over time in the relative benefits of acidification. As Arrow and Fisher (1974) have demonstrated, the possibility that current actions might burden and constrain (deplete) future opportunities must be counted as a cost against the current action. In short, the irredeemable nature of current acidification may foreclose valuable future options, whether due to currently unknown technologies, price structures, or changing tastes. Since new information can be exploited only if irreversible consequences have been avoided, the consequences of a decision to acidify cannot be undone even if the new information suggests that the decision was mistaken. Thus if acidification is ultimately discovered to have only trivial irreversible and undesired consequences, a delay in the decision to acidify can only mean that the present value of its ultimate benefits is reduced. On the other hand, if these undesirable consequences will actually be present, delay serves to enhance the probability they will be discovered, thus making a decision to acidify appear less attractive than it now does. Of course, the chances of discovering nontrivial adverse consequences of acidification might reasonably be directly related to the completeness of the existing state-of-knowledge about these consequences and the prospects for rapid advances in this knowledge. 8/ If this relation is direct, it follows that the expected decline in the relative benefits of acidification will be less than otherwise: delays in the acidification decision are then made to appear more favorable.

Summary and Conclusions

We have discussed two decision problems in the optimal control of acid precipitation. The first problem concerns the shape of the dose-response function while the second concerns a possible ratchet-effect associated with movements along a given dose-response function.

The shape of the dose-response function determines the shape of the marginal damage (benefit) function associated with varying levels of acidity. Studies of the impact of acidity on fish species suggest that the relevant

marginal damage curves are nonmonotonic functions of the acid level. As a
result there may exist more than one level of acidity and associated price
structure which balances the marginal benefits and marginal costs of reducing
acidity. Not all of these levels and price structures will be welfare max-
imizing, some may be welfare minimizing in the sense that small deviations in
either direction may improve welfare.

The analysis presented with respect to the shape of the dose-response
function did not explicitly consider the dynamic costs of adjustment. If an
ecosystem becomes highly acidified as a result of having made decisions based
upon prices in the n.eighborhood of a minimum point, then it may become optimal
to forget about attempts to control. This result occurs because the possibly
large short-run costs of control dominate the benefits of the action. This
result was obtained. by Forster (1975) in a study of water pollution control.
For sufficiently high levels of pollution, it is economically optimal to allow
a waterway to become biologically dead. This result depends upon a high rate
of discount. As the discount rate approaches zero, the result evaporates,
This is not surprising since the adjustment costs loom large in the short-run
while the discount rate shrinks the present value of future benefits. We will
not enter into a debate over the appropriate discount rate--but its importance
should be noted.

The ratchet-effect of the dose-response function refers to the possible
irreversibility of the environmental disruption caused by increased acidity. A
given increase in acidity reduces natural resources by an amount ΔR determined
by the dose-response function. Subsequent equivalent reductions in acidity
may not increase natural resources by as much as AR or may not increase them
at all. As long as this relationship is known and understood by all, then
market price signals should correctly reflect the net benefits of the
situation. If the relationships are not known, however, then current market
prices will not reflect the future costs of current actions and acidification
will proceed too far.

These decision problems suggest certain research and policy strategies
for acid precipitation. The potential irreversibilities dictate that systems
which are on the verge of acidification be subjected to immediate research to
ascertain the properties of their dose-response functions. Fisher and
Krutilla (1974) have argued that uncertainty regarding any irreversible
destruction that might be caused by an activity may be sufficient reason to
justify postponing the activity until the relevant information has been
collected. In the case of disruption due to acid precipitation, the activity
is now occurring and may intensify even while the information gathering is
taking place.

The discussion regarding the shape of the dose-response function suggests

a possible policy strategy for control in aquatic ecosystems. It suggests that systems may be classified into two groups according to their current acidity levels. The first group consists of those systems whose pH levels are sufficiently high (to the left of B in Figure 2) to make them worth saving. The second group consists of those that are not worth saving--the adjustment costs dominate potential benefits. [9/] This negative prescription may be put in a more positive light. Much work needs to be done quickly to keep more systems from moving from the first group to the second. At the same time, the frequency with which this nonconvexity issue occurs in the responses of eco-systems must be more completely identified. Table 2 raises the possibility that it could be a creature of ecosystem diversity. It is to the problem of valuing this diversity that we turn in the next chapter.

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1/ Although waters that are acidified from mine drainage perhaps contain more iron compounds than waters derived from acid precipitation, mine drainage otherwise seems to have a chemistry generally similar to that of precipitation-derived water. The high concentrations of iron hydroxides present in acid mine drainage are known to intensify the harmful effects of pH $\leq$ 6.5.

The situation in lakes may be different than in streams. Streams may be more capable of flushing themselves than lakes. Also the ability of fish to migrate in order to avoid high levels of acidity may be greater in steams than in lakes.

Fromm (1980) warns that "Data relating to the specific effect of low pH on growth of freshwater fishes are ambiguous." It is significant for the purposes of the present paper which seeks to point out potential difficulties in the control of acid rain to fird one example of the type presented. However, the frequency with which this nonconvexity issue occurs in the responses of ecosystems must be more completely identified.

2/ Arthur Okun has an interesting comment on economic optimization:

"The wise economist knows, however, that merely finding a marginal- that is not sufficient for an evaluation. A rigidly incrementalist approach can lose sight of major opportunities. Locating the least soggy spot in a swamp is not optimizing if high ground is accessible outside the swamp."

The Political Economy of Prosperity (Brookings Institution, 1970), p. 4.

3/ There is some evidence that there may be nonconvexities present in the benefit function for improved forest aesthetics. For example, in a psychological study of individuals' responses to insect and damaged southern pine forests, Buhyoff and Leuschner (1978) found that "visual preference" dropped rapidly as damages increased to 10 percent of the forested area but that preference declines were minor thereafter.

4/ In principle, the spreading of lime on top of forest soils might raise precipitation pH before it moves down the soil column. We are unaware of any commentaries on either the technical or the economic feasibility of this

practice.

5/   The tone of Bengtsson (1980) is optimistic with respect to the
practicality of large-scale liming to restore acidified water bodies. He
tends, however, to abstract from details that might compromise the optimistic
tone.   Barnes (1979, p. 1.232) speaks approvingly of the prospects for
neutralizing to be obtained by placing a lime column in a river bank.
Andersson (1980, p. 6) reports that six-fold higher neutralization effects
have been acquired using sodium hydroxide rather than lime in laboratory
experiments.

6/   Houck (1977) provides a technique for specifying and estimating
nonreversible functions.

7/   Let the extraction of the marginal depletion unit be delayed from t_0 to
t_1. This delay would cause the undiscounted net return in t_1 to be $(p_0 - \dot{p}t_1 - c)$,
where the dot indicates the time derivative of price.   Assuming $\dot{c}=0$,   if
instead the depletion unit is extracted in t_0 and the net returns $(p_0 c)$
invested in some other asset, the value of the investment in t_1 would be $(p_0 -
c)(1+rk)$, where k is the time interval between t_1 and $t_{0'}$ and r is the rate of
interest. Upon equating $(p_0 - \dot{p}t_1 - c)$ and $(p_0 c)(1+rk)$, and simplifying, we are
left with $\dot{p} = r(p_0 c)$, which says that in a regime of secure property rights in
the resource, its market price increases at the rate of interest over time.

8/   With the possible exception of limnology, where experimental means have
been extensively used to study the behavior of ecosystem functions such as
productivity and decomposition. [Vervelde and Ringelberg (1977)], many
ecologists view the prospects for rapid accumulation of new information as
unfavorable. Most of the relevant ecological disciplines lack a corpus of
empirically testable propositions derived from a broadly encompassing
analytical structure as well as quantitative bits of information that have
been related to or associated with each other [Clark, Jones, and Helling
(1979)] . Resort, therefore, has been either to simulation models or to the
real-time tracking of the behavior of a system under stress.

9/   Allen Kneese has reminded us that the West Germans approximate this
policy in their assignment of separate rivers and streams to pristine and
highly polluting uses. Note also that the PSD program of the 1977 Clean Air
Act Amendments in the United States is consistent with this policy approach.

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IV. VALUING ECOSYSTEM FUNCTIONS: THE EFFECTS OF ACIDIFICATION

Iv.production

Population growth and human territorial expansion are placing unprec-
edented burdens on ecosystems. Farmlands are being converted to suburbs,
while forests are being converted to farmlands. The Amazon forest, earth's
richest biological region, is losing to development each year an area half the
size of Great Britian [Prance, (1977)]. Pollution is now recognized as a
global problem with particular emphasis on acid precipitation and the
greenhouse effect. Estimates of species lost to extinction worldwide are as
high as 1000 per year [Myers (1979)].

But what values are reflected by this and similar data on our dwindling
natural environment? Part of the answer can come from a study of ecological
systems placed in an economic framework. Ecological systems must be reduced
to tractable analytical. frameworks which can then be incorporated into
economic models that are able to ascertain benefits and costs. For example,
in environmental economics, studies have estimated the willingness to pay for
trout fishing along a particular stream. These studies could then be used to
estimate the value of the effect of acid precipitation on trout populations.
Trout have value to people, and if the trout were to vanish so would the
benefits of the fishing. But trout are only one species in a complex
ecosystem. By removing other species, say certain insects that may appear to
be of no value, the trout may also vanish. Thus, a proper valuation of an
ecosvstem entails not just the valuation of end products like trout, but a
recognition of the interactions between trout and other species so that the
value of these other species can be established. By doing this, better
estimates can then be made of the uncompensated costs associated with
population growth and industrial expansion which affect the sources of
pleasure and life support services that ecosystems provide.

Ecosystems are incredibly complex. They may be composed of thousands of
species interacting in diverse ways. Each species fills a niche in the
overall system, and depends on one or more of the other species for survival.
But complex systems are not foreign to economists who have the difficult task
of sorting out complex economies. Notions such as short-run and long-run
equilibriums, steady states, and exogenous shocks appear to be applicable to
both ecosystems and economies. In addition, the same type of questions arise

in either system. For an economy, the economist uses models to determine the effect a tax in one sector has on other economic sectors. For an ecosystem, the ecologist (and the economist) may need to know the effect a particular pollutant that harms one insect species will have on all other species.

The parallels between ecosystems and economics suggest that similar models may be used for each. Moreover, if this can be accomplished, then linking ecosystems with economies may be possible. Such a linkage would permit not only detailed descriptions of how a pollutant will effect an ecosystem, but how the changes brought about in the ecosystem will effect the economy and, in turn, how these changes in the economy will influence the ecosystem.

Ecologists attempt to answer such questions by using energy as a unit of value. By measuring the flow of energy through an ecosystem, one can determine how an exogenous shock might affect that energy flow [Grodzinski (1975)]. The effect is then evaluated using some pecuniary value placed on an energy unit. Some support for this approach once was found among economists. The English economist, J.A. Hobson (1929) has remarked that:

> "... all serviceable organic activities consume tissue and expend energy, the biological costs of the services they render. Though this economy may not correspond in close quantitative fashion to a pleasure and pain economy or to any conscious valuation, it must be taken as the groundwork for that conscious valuation. For most economic purposes we are well--advised to prefer the organic test to any other test of welfare, bearing in mind that many organic costs do not register themselves easily or adequately in terms of conscious pain or disutility, while organic gains are not always interpretable in conscious enjoyment." (p. xxi)

According to one's perspective, Hobson's statement can be taken as support for an energetic basis of value, and as a plea for economists to devote more attention to the workings of the biological world and its implications for human welfare, both as a source of pleasure and as a life-support system. Hobson's first point has been received warmly by ecologists such as H.T. Odum (1971), to the point where it has been enshrined alongside cost-benefit analysis as a means of evaluating proposed energy technologies [Energy Research and Development Agency (1975)]. However, it has been coldly received by modern economists. Georgescu-Roegan (1979) neatly expresses the economists' source of difficulty with energy as the unit of value for the satisfaction of human wants:

> "The entropic nature of the economic process notwithstanding, it would be a great mistake to think that it may be represented by a vast system of

thermodynamic equations . . . . The entropic process moves through an intricate web of anthropomorphic categories, of utility and labor above all. Its true product is not a physical flow of dissipated matter and energy, but the enjoyment of life. . ..pleasure is not related by a definite quantitative law to the low entropy consumed." (p. 1042)

Thus the value of energy varies with its use. The correct approach is therefore to include the ecosystem in the economy where the uses of the ecosystem can be evaluated relative to all other goods.

Hobson's second point, that economics should give deeper consideration to the role of biosphere in human affairs, has suffered from neglect. With the exception of the work inspired by Boulding (1966) and Krutilla (1967), the economics discipline continues to be notable for its inability to capture many of the concerns of biological scientists, particularly ecologists, about the impacts of human activities upon ecosystems and, via these ecosystem impacts, ultimately upon human welfare. Perhaps economists have dismissed these themes simply because the economics discipline has lacked a means of fitting them into the framework of economic analysis.

The purpose of this paper is to develop a link between ecosystem and economy that will allow an economic evaluation of ecosystem structure. We try to broaden traditional approaches to environmental economic problems by encompassing bioenergetics, but without resorting to the use of energy as the unit of value used by humans. There are two main phases of the development. First, an ecosystem model is described using the notions of production functions, optimization, and equilibria. Humans are absent from this phase. All energy input into the model derives from the sun. In the second phase, humans are introduced under the familiar guise of utility maximizer. This I.cads to behavior that interferes with the ecosystem through changes in the sources and uses of energy.

The second section develops a model of the optimizing behavior of a single organism in an ecosystem. The third section extends this idea to multiple organisms and to ecosystem equilibrium. In the fourth section, common ecological- themes are discussed as they relate to the model. Human perspectives of the ecosystem enter in the fifth section. The sixth section uses the developments of previous sections to address questions about the value of pollution impacts upon ecosystem structure. The seventh section is a simple general-equilibrium model incorporating the concepts of previous sections.

Optimization by Individual Organisms

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Initially, we develop a model of an ecosystem where humans have no influence. The model can be considered a depiction of prehistoric times or of very remote areas in modern times. In this world, all energy is derived from the sun.$\underline{^{1/}}$ Organisms may use this energy directly, in the case of plants, or indirectly, in the case of herbivores and carnivores. Each organism is a member of a particular trophic level, where a trophic level is defined as ". ..a collection of species which feed from the same set of sources and which do not produce for each other" [Harmon, (1976, p. 260)]. In essence, each trophic level can be thought of as a stratem in a food pyramid.$\underline{^{2/}}$ The objective is to link mathematically the trophic levels. This will provide a framework for discussing equilibria in the ecosystem.

Before deriving the links, however, the actions of the individual organisms must be described. In a general equilibrium model. of an economy, individual consumers and firms are usually described as maximizers. But in an ecosystem, do nonhuman organisms maximize? Most people do not credit a weasel " with thoughtful preference revelation when it raids the chicken coop instead of ferreting out a mouse or two. ". ..men consciously optimize, animals do not - they survive by adopting successful strategies 'as if' conscious optimization takes place" [Hirschleifer (1977, p. 4)]. This "as if" assumption is sufficient to capture much of the behavior of nonhuman organisms, and, thereby, establish a fruitful model: if one always remembers that these organisms are not human, it can be worthwhile to treat them as solving human-like problems.

Various suggestions have been made as to what it is that nonhuman organisms maximize, or behave as if they are maximizing. Lotka (1925) developed a model where the maximand is the rate of increase of the species. This rate is a function of food capture, shelter, and other physical needs. Obtaining these needs requires energy expenditure. If a species is to be successful, then the energy expended on the needs must be less than or equal to the energy acquired. Lotka characterizes a maximum in this system with a set of equations where the marginal productivity (i.e. , an increase in the species) of an energy expenditure equals the marginal loss (i.e., a decrease in the species) of that energy expenditure.$\underline{^{3/}}$ Modern work has emphasized the role of energy more directly in the search for a maximand. Odum (1971, p. 90) points out that life requires power and "... the maximum and most economical collection, transmission, and utilization of power must be one of the principal selective criteria. .." Finally, Hannon (1976) develops a model using stored energy as the maximand. Stored energy is simply the energy squired by the organism less the energy needed to maintain itself. Hannon argues for the reasonableness of this objective based on general observation, and on the increased organism stability it provides during periods of fluctuating inputs.

The stored energy approach is used here. It does not seem to differ significantly from Lotka's approach, particularly since he viewed organisms as energy transformers. As indicated in the next section, if organisms of a species are successful in storing energy, this is interpreted as leading to an increase in the species. Hence, the stored energy approach appears acceptable to modern ecologists, and consistent with the pioneering work of Lotka.

For specificity, suppose the organism is a fox which, as an energy transformer, gathers all its energy from food, and then assimilates this energy for various purposes. All input energy must be accounted for as output energy in the form of waste heat, metabolism, growth, reproduction, losses to predators, detritus, mechanical activities, and storage. Let x_i and ε'_i, $i = 1, \dots, n$, be the mass flow from the ith source to the organism and the energy content per unit of mass i respectively. The x_i may be various species of small mammals preyed upon by the fox. Total input energy is then:

$$\sum_{i=1}^{n} \varepsilon'_i x_i \tag{1}$$

Let ε''_i be the energy spent to obtain a unit of x_i, so that the net input of energy from a unit of x_i is $E_i = E'_i - E''_i$. Therefore, net input energy is:

$$\sum_{i=1}^{n} E_i x_i \tag{2}$$

The energy outputs are given by x_k, $k = n+1, \dots, m$ and the energy content per unit of x_k is ε_k. For example, x_k may be the activity of searching for a den, and ε_k is the energy spent per unit of searching. For some inputs such as heat loss, x_k is measured in energy and $\varepsilon_k = 1$; however, no loss of generality results from using "k"   Total energy output is:

$$\sum_{k=n+1}^{m} \varepsilon_k x_k \tag{3}$$

Stored energy is the difference between input and output. It represents energy in excess of what is needed for viability. Let r be this energy. Then, using (2) and (3):

$$r = \sum_{i=1}^{n} \varepsilon_i x_i - \sum_{k=n+1}^{m} \varepsilon_k x_k \tag{4}$$

For convenience, all inputs and outputs will henceforth be denoted x_j,

j=1, ..., m+n, where $x_j \geq 0$ for inputs and $x_j \leq 0$ for outputs. Each $\varepsilon_j x_j$ is interpreted now as a net input of energy. Thus, if index j is heat loss, the net energy input from heat loss is $-\varepsilon_j x_j$. Expression (4) can be rewritten as

$$r = \sum_{j=i}^{m+n} \varepsilon_j x_j \qquad (5)$$

The objective of the fox is to maximize expression (5).

A bundle of net inputs for the organism is represented by the m+n real numbers $x = (x_1, \ldots, x_{m+n})$. Not all bundles are feasible for the organism. The fox cannot continually catch squirrels without ever losing heat energy. The set of feasible bundles will be called the physiology set.[4/] In essence, this set places constraints on what is achievable for the organism by describing the physiological processes which convert inputs to outputs. For example, as a general rule of ecology, in order for the organism to use ingested material, it must oxidize the organic molecules in the material it ingests. [See Morawitz (1968, Chap. 5)]. This creates useful energy, but some formerly useful energy is also lost as heat. The physiology set also will depend on ambient temperature, time of year, and other environmental conditions. Human activities may influence this feasible set. Acid precipitation is a good example of a human activity that interacts with an ecosystem via alterations in physiology sets.

A simple diagram illustrates these notions. Suppose for the fox there is only one input, squirrels, and one output, mechanical activity. Figure 1 shows the physiology set as the shaded region. The set is entirely within the second quadrant where squirrels are consumed in positive quantities and mechanical activity is a loss or a negative quantity. With mechanical activity of $\hat{x}_2$, the fox can attain a quantity of squirrels x, a quantity $\hat{x}$, or any amount between $\hat{x}$ and the horizontal axis. Bundle $\hat{x}$ repesents the greatest amount of squirrels attainable for $\hat{x}_2$. For this reason, $\hat{x}$ is labelled an efficient point of the physiology set; and all points along the heavy curved border of the set are referred to as the physiologically efficient points.

> Definition: 'A bundle $\hat{x} = (\hat{x}_1, \ldots, \hat{x}_{n+m})$ in the physiology set X is physiologically efficient if there does not exist an alternative bundle $x = (x_1, \ldots x_{n+m})$ in x such that $x_j \geq \hat{x}_j$, $j = 1, \ldots, m+n$, and $x_j > \hat{x}_j$ for at least one j.

Thus, a physiologically efficient bundle is one where greater amounts of energy cannot be attained without even greater losses of energy. Note that points along the nonheavy border in Figure 1 are, therefore, not

physiologically efficent.

The dependence of the physiological set on environmental conditions is depicted in Figure 2. The upper cross-hatched area may represent the physiological set of a lake trout prior to the occurrence of acid precipitation, while the lower cross-hatched region represents the trout's set subsequent to the acid precipitation. This change clearly indicates a detrimental effect from the pollution.

The fox behaving as a stored energy maximizer can be illustrated in the simple diagram as well. With one input and one output, the fox maximizes the expression from (5)

$$r \equiv \varepsilon_1 x_1 + \varepsilon_2 x_2 \qquad (6)$$

For a fixed level of stored energy, r, (6) can be plotted as the line in Figure 3 labelled $\bar{r}$. A higher level of stored energy is shown by the line $\bar{\bar{r}}$. The vertical and horizontal intercepts indicate the stored energy attainable, and the further the line from the origin in the first quadrant, the greater the stored energy. Given a particular point, say $\hat{x}$, and energies ε_1 and ε_2, the stored energy is given by $\bar{r}$. The slope of the line is the ratio $-\varepsilon_2/\varepsilon_1$, or the rate at which squirrels can be transformed into mechanical energy in the ecosystem. Thus , the ε_1's are the energy prices the fox faces.

The fox is assumed to take ε_1 and ε_2 as given.; that is, he has no control over these values. They are parameters in his maximization problem. The point of maximum stored energy will be given by that stored energy line that is furthest above the origin, but still having at least one point in common with the physiology set. Obviously, this point will be one that is physiologically efficient. Figure 4 illustrates maximums of $\hat{r}$ for values $\hat{\varepsilon}_1$ and $\hat{\varepsilon}_2$, and r for values $\bar{\varepsilon}_1$ and $\bar{\varepsilon}_2$. The maximizing solution depends on the shape of the physiological set and the values of ε_1 and ε_2. At $\hat{x}$, greater levels of mechanical activity and squirrels prevail, because squirrels have more energy content ($\hat{\varepsilon}_1 > \bar{\varepsilon}_1$) and/or mechanical activity results in less

Figure 4.1
The Physiology Set

squirrels

$\hat{x}$

$\hat{x}_1$

x

$\hat{x}_2$

'1

'1

o

Mechanical activity

'2
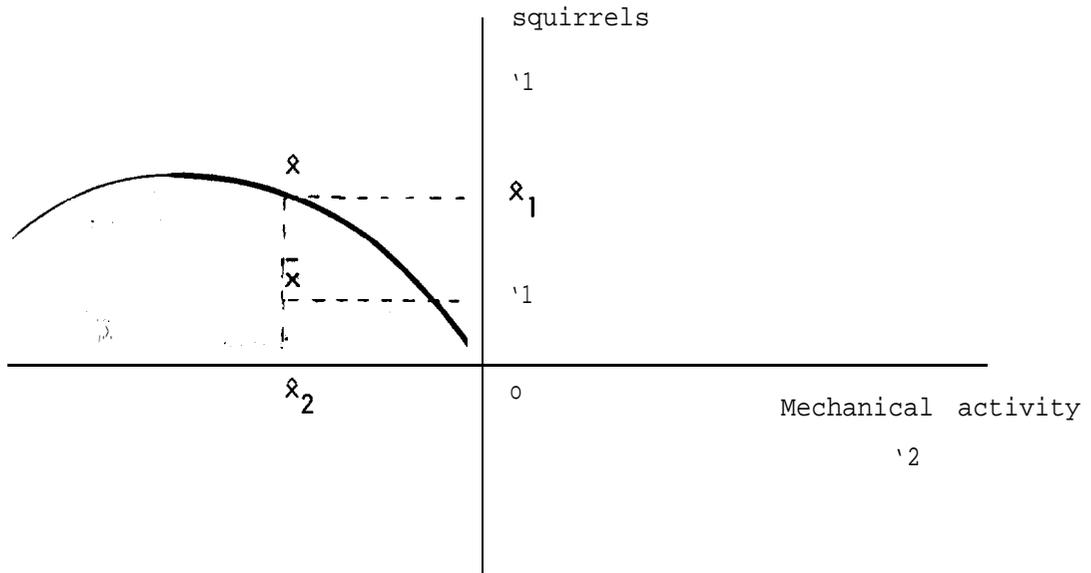
Figure 4.2
Effect of Environmental Conditions

output

Input
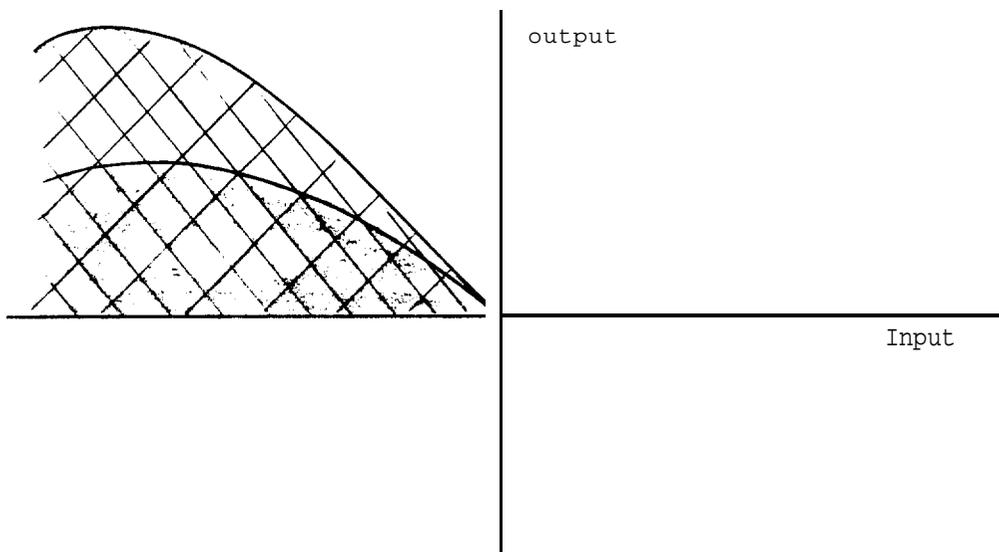
Figure 4.3
Attainable Stored Energy

$$x_1 = \frac{r}{\iota_1} - \frac{\iota_2}{\iota_1}\,\iota_2$$

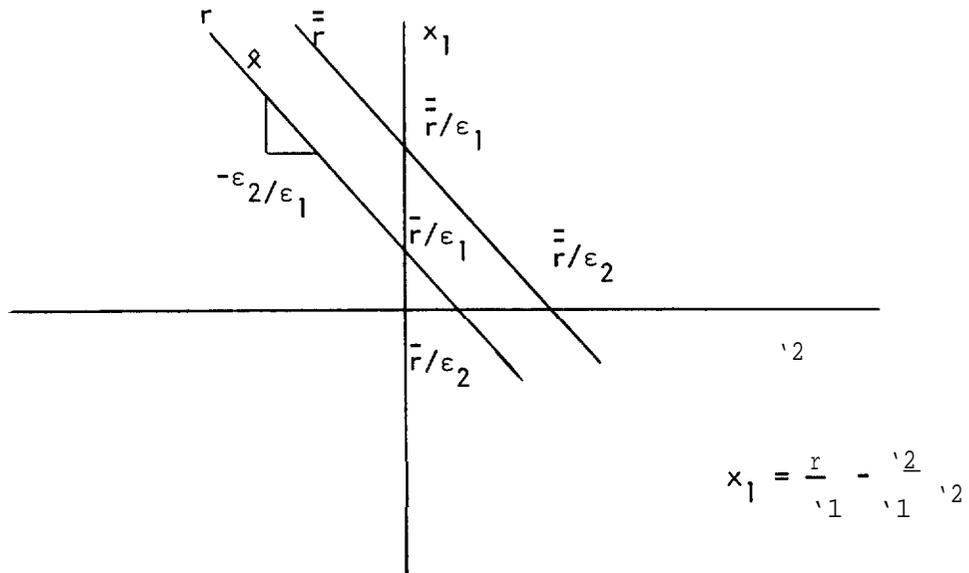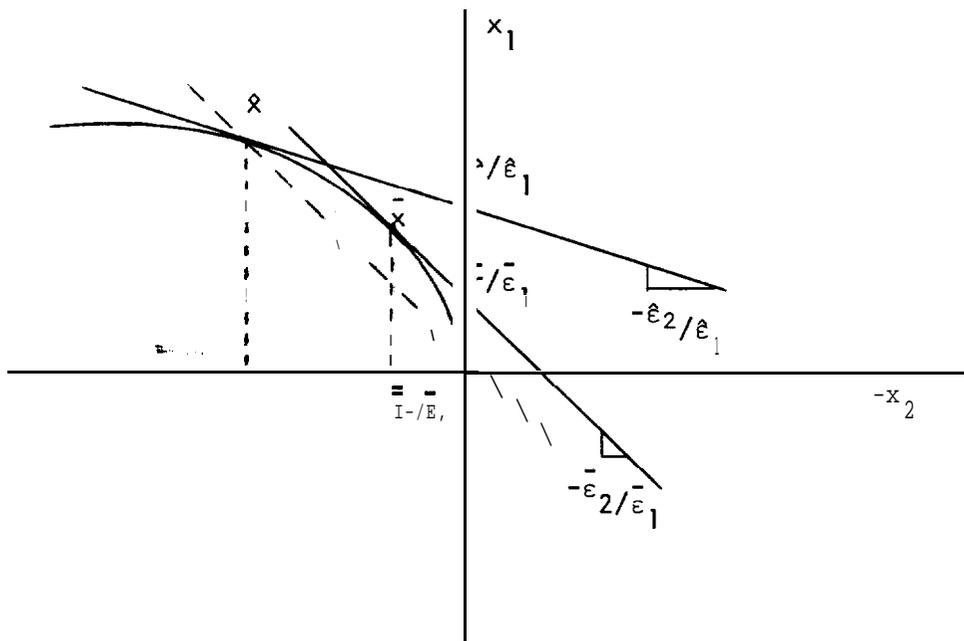## Figure 4.4
The Maximizing Solution

energy loss

$(\hat{\varepsilon}_1 < \bar{\varepsilon}_1)$. For values $\bar{\varepsilon}_1$ and $\bar{\varepsilon}_2$, the fox would not move beyond point $\bar{x}$. To do so would mean more mechanical activity and more squirrels, but the energy gained would be less than the energy lost. Moving from $\bar{x}$ to $\hat{x}$ would mean a drop in stored energy from $\bar{r}$ to r.

A maximum will exist provided certain restrictions are placed on the physiology set. In particular, the set must be bounded and include its boundaries. These restrictions do not seem unrealistic. Figure 5 illustrates a set that is not bounded. For positive ε_1 and ε_2, maximum stored energy is infinite. The shape of the set must be left to experiments, observations, and statistical analysis, and it can be expected to vary significantly among organisms. Research into these shapes is necessary to apply the theory presented here.

Further insight into the maximization model can be gained by returning to the general case with n+m variables. To do this, the concept of a physiology function is introduced using the physiology set. For any set of values of all but one of the net flows, x_j, there is only one value of x_j that is compatable with physiological efficiency. This is obvious for the two variable case from the figures above. For n+m variables, let $x^{-j} = (x_1,\ldots,x_{j-1},\ x_{j+1},\ldots,x_{n+m})$, then there is a one-to-one correspondence between the n+m-1 dimension vector x^{-j} and the scalar x_j. In functional form,

$$x_j = f(x^{-j})$$

or equivalently

$$F(x) = x_j - f(x^{-j}) = 0 \tag{7}$$

The function F(x) is the physiology function, and, by construction, it embodies physiological efficiency. That is, $\hat{x}$ is physiologically efficient if and and only if $F(\hat{x}) = 0$. In two dimensions, $F(\hat{x}) = 0$ implies that $\hat{x}$ is on the border of the physiology set.

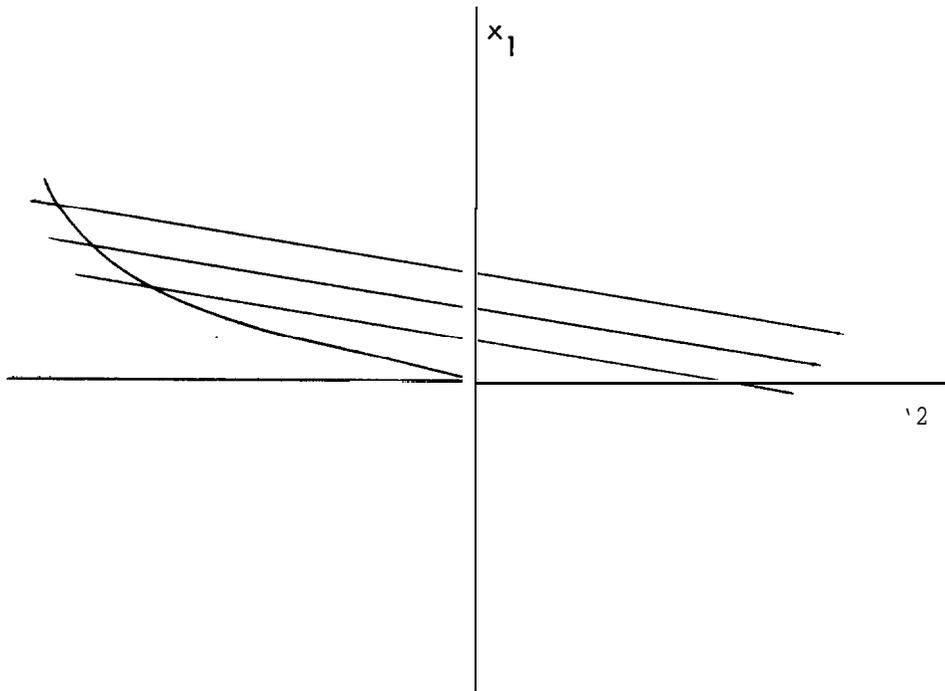The maximization problem can be restated as

$$\max r \equiv \sum_{j=1}^{m+n} \varepsilon_j x_j \tag{8}$$

$$\text{subject to} \quad F(x) = 0$$

where F(x) is assumed to be twice differentiable and the physiology set is

Figure 4.5
. . . An Unbounded Physiology Set

assumed to be **strictly** convex. Strict convexity assures that the second-order sufficiency conditions of the maximization problem are satisfied, and that there is a unique maximum. The **Lagrangian** for problem (8) is:

$$L(x;\lambda) = \sum_{j=1}^{m+n} \varepsilon_j x_j + \lambda F(x) \tag{9}$$

and the first-order conditions for a maximum are

$$x_j: \qquad E_j + \lambda \frac{\partial F(x)}{\partial x_j} = 0 \qquad j = 1,\ldots,m+n \tag{10}$$

$$g: \qquad F(x) = 0 \tag{11}$$

Dividing any two conditions in (10) by each other yields

$$\frac{\partial F(x)/\partial x_i}{\partial F(x)/\partial x_j} = \frac{\varepsilon_i}{\varepsilon_j} \tag{12}$$

so that for a maximum, the ratio of partial derivatives of $F(x)$ must be equal to the ratio of **energy** prices. Using (7),

$$F(x_1, \ldots, x_{j-1}, f(x^{-j}), x_{j+1}, \ldots, x_{m+n}) = 0$$

and differentiation with respect to x_i, $i \neq j$, yields

$$- \frac{\partial f(x^{-j})}{\partial x_i} = \frac{\partial F(x)/\partial x_i}{\partial F(x)/\partial x_j} \tag{13}$$

Thus, the left-hand-side of (12) can be interpreted as the rate at which x_j must be substituted for x_i while **all** other values are held constant. Or, for the fox's predatory behavior, (12) states that the rate at which he can trade squirrels for rabbits while maintaining stored **energy** must equal the rate at which he can exchange squirrel energy for rabbit energy in the ecosystem.[6/] Alternatively, **(12)** and (13) can be used to obtain

$$\frac{\partial \varepsilon_j f(x^{-j})}{\partial \varepsilon_i x_i} = 1 \tag{14}$$

The left-hand-side of (14) is the rate at which energy from source j must be traded for energy from source i in order to remain physiologically efficient.

Or, substituting squirrels for rabbits must lower the input of rabbit energy at the same rate squirrel energy is increased.

The conditions for a maximum given by (12) can. be related to the earlier figures. Condition (12) for the one input-one output case is shown by the tangency in Figure 4; The left-hand-side of (12) is the slope of the physiology set border, and the right-hand-side of (12) is the slope of the stored energy line.

The first-order maximum conditions given by (10) and (11) constitute m+n+1 equations which can be solved for the optimum values of the x_i and λ as functions of the energy prices. A solution is guaranteed by the assumption of a convex physiology set. Thus , there exist the functions:

$$x_j = \phi_j(\epsilon) \qquad\qquad j = 1,\ldots,m+n \qquad\qquad (15a)$$

$$\lambda = \phi_\lambda(\epsilon) \qquad\qquad\qquad (15b)$$

The function $\phi_j(\epsilon)$ indicates the amount of the jth input acquired or j^{th} output spent, given the energy prices of all inputs and outputs. Substituting these amounts back into the objective function gives the maximum stored energy,

$$r = \sum_{j=1}^{m+n} \epsilon_j \phi_j(\epsilon) \qquad\qquad (16)$$

If j represents rabbits, $\phi_j(\epsilon)$, can be thought of as the fox's demand for rabbits at prices ε.

Finally, the $\phi_j(\epsilon)$ terms can be substituted into (10) and (11), and derivatives can be taken with respect to the ϵ_j. This yields the system of equations:

$$1 + \sum_{k=1}^{m+n} \phi_\lambda \epsilon \frac{\partial F(x)}{\partial x_j \partial x_k} \frac{\partial \phi_k(\epsilon)}{\partial \epsilon_j} + \frac{\partial \phi_\lambda(\epsilon)}{\partial \epsilon_j} \frac{\partial F(x)}{\partial x_j} = 0 \qquad\qquad (10')$$

$$j,k = 1,\ldots,m+n$$

$$\sum_{k=1}^{m+n} \frac{\partial F(x)}{a x_k} \quad \frac{\partial \phi_k(\epsilon)}{\partial \epsilon_j} = 0 \qquad j = 1,\ldots,m+n \qquad (11')$$

This system can be·used to solve for the $\partial \phi_k(\epsilon)/\partial \epsilon_j$, values, and, by the second-order conditions,

$$\frac{\partial \phi_j(\epsilon)}{\partial \epsilon_j} > 0 \qquad j = 1,\ldots, m+n \qquad (17)$$

The interpretation of (17) is that an increase in the energy price of a net input results in an increase in the use of that input. If the net energy the fox could obtain from a rabbit were to increase while the net energy obtained from a squirrel remained the same, the fox would chase more rabbits and fewer squirrels. A similar interpretation holds on the output side.

Before closing this section, a brief comparison between this model and economic models is worthwhile. The energy storage maximizing organism is analogous to the profit maximizing firm. The firm uses inputs (capital, labor, etc.) to produce outputs (guns, butter, etc.). The firm's technology set consists of net outputs, so that inputs are negative and outputs positive. This is opposite to the organism whose physiology set is made up of net inputs. Moreover, the firm pays money to buy inputs, and collects money in selling outputs. This also is opposite, since the organism collects energy from inputs, and pays energy for outputs. Inequality (17) is, however, the same for the firm and the organism since the two opposites cancel..

Multiple Organisms

An ecosystem. comprises manv stored energy maximizers which must be linked to provide a. complete picture. Each organism belongs to a species, and sets of species form trophic levels. The trophic levels are links in a food chain or levels in a hierarchy. Each species feeds on species in lower trophic levels, and in turn provides food for species in higher trophic levels. Some hierarchies may be considerably more complex than others in that some species may interact with other species from. many different trophic levels. Thus the inputs and outputs of the previous section represent inputs from other organisms and outputs to other organisms.

At the bottom of the hierarchy are the simplest plants who derive all their input energy from the sun. In fact, in an ultimate sense, the sun supplies all the energy consumed by the ecosystem. This provides one equation in the ecosystem model: total output energy in the form of heat which is lost

in the ecosystem equals total input energy from the sun.

By responding to energy prices, ε, each organism behaves as the stored energy maximizer of the previous section. We assume each organism to be inconsequential with regards to its effect on the ecosystem, since there are so many other organisms. From this we infer that each organism has no control over the energy prices. This is consistent with the maximization process discussed above. However, the relative energy prices are determined by the activities of the organisms in the ecosystem. The fox's energy price for acquiring rabbits will depend on the availability of rabbits. If an exogenous shock were to reduce the number of rabbits drastically, we would expect the energy price to increase for the fox, causing a decrease in the fox's stored energy.

The existence of an equilibrium ecosystem, given the number of interacting maximizers, and given a set of initial conditions or initial numbers of organisms and environmental surroundings, requires a set of energy prices such that all organisms are maximizing stored energy while at the same time inputs are consistent with outputs and total energy is conserved. Existence will depend on the forms of the physiology sets and on any threshold conditions that may prevail. For instance, too few individuals of a certain species may lead to a total collapse of the species. There is also the possibility of multiple equilibria. That is, equilibrium, if it exists, may not be unique. Different equilibria may consist of a variety of configurations of species numbers.

In accordance with Hannon (1976), stored energy is zero for all organisms in the equilibrium ecosystem. Recall that stored energy is energy above and beyond what is needed to survive. This is analogous to all firms making zero profit in a perfectly competitive economy. To see why this is, suppose an equilibrium exists and all species have zero stored energy; then consider an exogenous change that causes foxes to have positive stored energy. The foxes are healthy, vigorous, and increasing in numbers. But this means that each fox will now face greater competion in his search for energy inputs. Numb ers of rabbits will decline, and the energy price of rabbit inputs will increase. This increase will cause a decrease in the foxs' stored energy, until zero is again attained. A new equilibrium is established, although it mav be one with more foxs and fewer rabbits than before. The same type of scenario can be used to show how the system responds to negative stored energies.

Setting up a mathematical model to study this ecosystem equilibrium is similar to the problem of setting up a general equilibrium, competitive model of an economy. The mathematics of existence can be complex, and will not be pursued here. However, efforts along these lines should be rewarding.

Insights could be had regarding: 1) whether the stored energy behavior concept is consistent with observed equilibria; 2) those restrictions on the physiology sets consistent with equilibria and with field and experimental observations; and 3) the effects exogenous shocks, such as human induced, acid precipitation, have on these equilibria.

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Common Ecological Themes

Watt (1973, p. 34) sets forth the following as a fundamental principle of ecological science: the diversity of any ecosystem is directly proportional to its biomass divided by its productivity. That is:

$$D = k(\frac{B}{P}) , \qquad\qquad (18)$$

where D is a diversity measure directly related (Pielou, 1977, Chap. 19) to the number of species in a given habitat and the relative abundances of each species; $\underline{7/}$ B is the total weight or standing biomass of living organisms in a habitat; P is the amount of new living tissue produced per unit time; and k is a constant differing from one habitat to another. Thus, for a given biomass, system diversity and system productivity are inversely related.

Within a given habitat, d(B/P)/dt > 0, implying that in the early life of an ecosystem, the production of new tissue is very large compared to the amount of biomass. This high relative productivity is the source of biomass growth. It is achieved by introducing into an abiotic or stressed environment a small number of pioneer species (e.g., weeds) with rapid growth rates, short and simple life cycles, arid high rates of reproduction. In the mature stages of an ecosystem, a wider variety of organisms that grow more slowly and have longer life spans is present. Net production or "yield" is lower in a mature system because most energy is invested in maintenance of the standing biomass. Thus, whereas energy in the pioneer stage is used to increase biomass, so that a relatively empty habitat can be filled, all the captured energy coming into a fully mature system is employed to maintain and operate the existing biomass, which already occupies all the habitat territory available.

Ecosystems that must live under intermittent or continued severe stress exhibit the attributes of immature systems: they have relatively low diversity and biomass but high throughput of energy and thus high yields.

Ecologists traditionally prefer ecosystems with large biomass and diversity. This preference for mature ecosystems appears to rest on two positions: the maximization of system energy capture; and the maximization of system stability. In the first case, more energy is captured per unit biomass in a mature system because less energy has to be "wasted" in growth and

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reproduction activities. The distinction is similar to Boulding's (1966) description of the "cowboy economy" and "the spaceship economy", where the former maximizes throughput and therefore energy diffusion, while the latter maximizes incoming energy concentration and fixation. According to Margalef (1968), the immature or stressed system expends more energy per unit biomass in reproduction in order to make up for its more frequent loss of individuals. In addition, because of its relatively small energy recycling capacity and its relative inability to alter and to renew its environment in ways favorable to its sustenance, it must expend relatively more energy per unit biomass in food gathering activities. The immature system thus expends relatively more energy in producing new tissue to replace that which has disappeared (depreciated). In contrast, the mature system expends most of its incoming energy in keeping what it has already developed: it is durable. Because it sustains a greater biomass per unit energy, the mature system is frequently said to be more "efficient" (B.P. Odom, 1971, p. 76).

Although exceptions appear to exist [May (1971), Jorgensen and Mejer, (1..979)], the greater efficiency of mature ecosystems is associated in ecological thought with greater stability, where stability is variously interpreted to mean system resiliency to exogenous shocks or infrequent fluctuations in standing stock. This stability is thought to originate in a set of homeostatic controls present in greater number and variety in mature systems, thus providing a greater number of avenues through which the system can recover from damages to one or more of its components. The greater simplicity of the immature system is thought to increase the likelihood that if anything goes wrong, everything goes wrong. Thus monoculture, which are by definition the simplest and least diverse of ecosystems, are susceptible to being wiped out by any single pest or event to which they are sensitive. Incoming energy flows only through one or a small number of pathways; when this pathway is degraded, no means to capture energy remains. The system therefore collapses unless energy subsidies (e.g., fertilizers) are provided from outside. These subsidies are of course a further source of the low biomass supported per unit incoming energy that is characteristic of immature ecosystems.

The human dilemma posed by the ecologists then involves a tradeoff between high yield but risky immature systems with undifferentiated components, and low yield, reasonably secure systems with a variety of components. Even if the requisite energy subsidies were usually available, an earth covered with cornfields would be dangerous. Moreover, given, as Scitovsky (1976) convincingly argues, the human taste for variety and novelty, a world of cornfields would be exceedingly dull. Nevertheless, flowers and butterflies nourish only the human psyche; they provide little relief to an empty stomach. Human activities increase biological yields by accelerating

energy flows through ecosystems. In terms of the model of the previous two
sections, these activities increase overall energy prices. To accomplish
this, they simplify ecosystem structures, either by keeping them in a
perpetual state of immaturity or by impoverishing the energy flows their
habitats can produce.

. . .

In the context of the above perspective, pollution, such as acid precip-
itation, harms human welfare by reducing yields of the material scaffold of
wood, fish, and corn and by increasing ecosystem simplicity: yields are
reduced and monotony is increased. Woodwell (1970) notes that by elimination
of sensitive species. SO_2 air pollution around the Sudbury smelter in Ontario
first resulted in a reduction in the diversity and biomass of the surrounding
forest. Finally the canopy was eliminated with only resistant shrubs and
herbs surviving the assault. He also notes that chronic pollution reduces
plant photosynthesis without having much effect upon respiration requirements.
As a result, large plants, which have high respiration requirements, are
placed at a disadvantage relative to small plants. In a vivid image, he
posits the replacement of the great variety of phytoplankton of the open ocean
by the algae of the sewage plants that are insensitive to just about any
stress.

Valuing Diversity and Yield

In accordance with the treatments of Hannon (1979), Mauersberger (1979),
and sections two and three of this chapter, the ecosystems refered to in the
following development are long-run equilibria sustainable with various
combinations of energy from solar, biogeochemical, and subsidy sources.
Contrary to much of the ecological literature, day-to-day transient states in
the relative abundances of various species are disregarded. This permits us
to concentrate upon a small number of key expressions and basic principles,
thereby avoiding the bewildering black-box flow diagrams often used by
ecologists. We wish to gain insight into two questions. First, what is the
economic value of the quantity of each species that a location is producing?
For our purposes, a location is simply a set of map coordinates. Second, what
is the economic value of the assortment or bundle of species that the location
is producing? That is, what is the value of a particular ecosystem design?
For a particular species assortment, the first question is usually answerable,
given that market (not energy) prices of each species unit are readily
observed or inferred.[8/] However, the second question, whether treated singly
or in combination with the first, has not yet been grappled with insofar as
ecological questions are concerned. We adapt a model. of Lancaster's (1975) to
deal simultaneously with the two questions.

To analyze these two questions, we need a model permitting us to trace

through the impact upon the economic benefits derived from ecosystems of changes in specie quantities and assortments caused by changes in energy IIlOWS. The first step in doing this is to define an ecosystem, e_i, as a set of species, where these species are in fixed proportions to one another. Expression (19) identifies ecosystem i with n species and

$$e_i \equiv (r_1^i, r_2^i, \ldots, r_n^i) \tag{19}$$

where r_4^i is the quantity of species j. Biomass is used to normalize the measure of different species. An ecosystem thus contains different species in a particular proportion at a single location. Ecosystems that contain species in different proportions are considered to be different ecosystems. Given the linearity of (19), the species content of x units of an ecosystem is simply x times the content of each species in an ecosystem unit.

Allow some time interval sufficiently long to permit each feasible eco-system to attain a long-run equilibrium defined in accordance with the model of sections two and three. Assume that a given amount of energy, $\bar{E}$, from solar, biogeochemical, and subsidy sources is available for this time interval at the location in question. Included in the biogeochemical energy source is the energy currently stored in the standing biomass. With $\bar{E}$, a variety of ecosystems can be established, the range of the variety being determined by the physiology sets of each species and the ways in which the species interact with each other.

Note that our notion of long-run equilibrium need not be a climax bio-logical equilibrium; that is, it includes other sustainable states as well. In particular, by including energy subsidies and biogeochemical energy in available energy, we allow immature ecosystems to be formed and sustained. For example, an energy subsidy is being provided a vegetable garden when it is weeded and when it is harvested. The weeding prevents the garden from "reverting" to field, woods or prairie; the harvesting prevents the standing stock of vegetable plants from suffering the effects of congestion. This standing stock will produce, period after period, a unique sustainable flow of new biomass or yield as long as the requisite biogeochemical energy and energy subsidies are provided. Similarly, with enough of an energy subsidy (as with a greenhouse) in Wyoming, one can sustain a banana-mango ecosystem with its associated flow of bananas and mangoes. We assume, whether reference is to an entire ecosystem or to a particular species within that system, that the sustainable yield measure is an order preserving transformation of the standing stock measure.

For a particular quantity of incoming energy, there will be some maximum amount of each ecosystem that a particular location can produce.[9/] Let the

minimum energy requirements for producing an ecosystem be given by:

$$E = E(e(r)) = \phi(r),$$ (20)

where the elements of the r-vector are sustainable yields per unit time. $\phi(r)$ will be called a diversity possibilities function. It shows the maximum quantities of various species combinations that a location can sustain with given available energy each period. We assume that $\phi(r)$ is homothetic and convex, and that $\phi' > 0$. For a given energy flow at a particular location. Figure 6 illustrates a diversity possibilities function for grass and corn.

In Figure 6, four ecosystems are depicted, one of which, e , contains only grass, and another of which e_4, contains only cows. Two ecosystems, e_2, and e_3, containing grass and cows in different combinations, are also depicted. If enough alternative ecosystems are possible, a continuous diversity possibilities frontier, E, can be formed, as we assumed in (20). For given energy availability, each point on the frontier, E, represents the maximum quantity of one species that can be produced with a particular quantity of the other species being produced. Since cows probably use relatively less, if any, solar radiation directly, a progressively greater proportion of biogeochemical energy and energy subsidies will be included in E as one moves from the vertical axis to the horizontal axis.

The convexity of the frontier follows from an ecological version of the economic law of diminishing returns known as Mitscherlich's law [Watt (1973, p. 21)]. As progressively more energy is diverted from grass production to cow production at the location in question, the increment to the latter will decline. Similarly, the diversion of energy from cows to grass will result in declining increments to grass production. Since in Figure 6, the cows could feed upon the grass, the convexity of the feasible region is also attributable to the less biologically efficient use of the given available energy by cows than by grass. As a food chain lengthens, the amount of original. energy used for production by species distant from the original energy input tends to decrease at an increasing rate (E.P. Odom, 1971, Chap. 3). Of course, as Tullock (1971) recognizes, the croppings and droppings of the cows may recycle some of the energy originally embodied in the grass and cause both grass and yields to increase over some portion of the frontier. However, as grass becomes scarce, the cows must expend progressively more energy in search for it, if it is to remain a part of their food supply. Finally any cow grazing . whatsoever might be so harmful to grass that the frontier bows inward, causing a nonconvexity problem for applications of economic optimization techniques.

The assumptions of homotheticity and $\phi' > 0$ for (20) imply that: $\phi(\lambda,r) = F(\lambda)\phi(r)$ for all $\lambda, r > 0$. In terms of Figure 6, these assumptions mean that
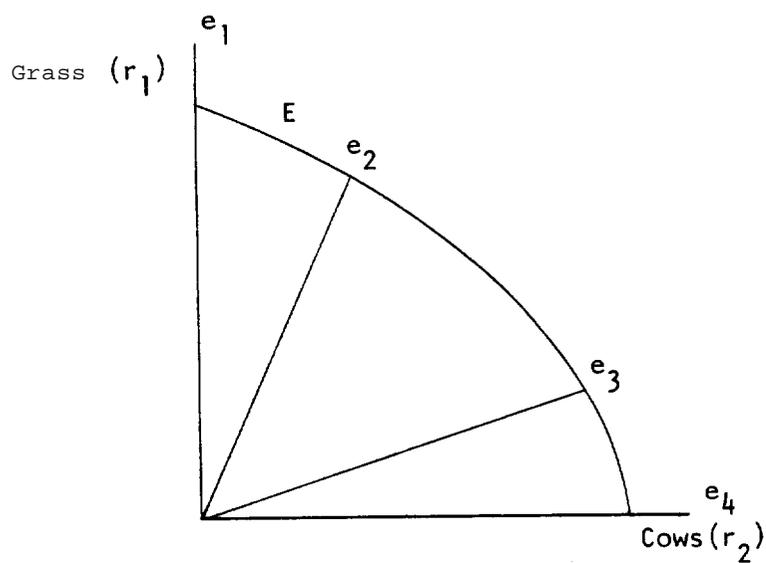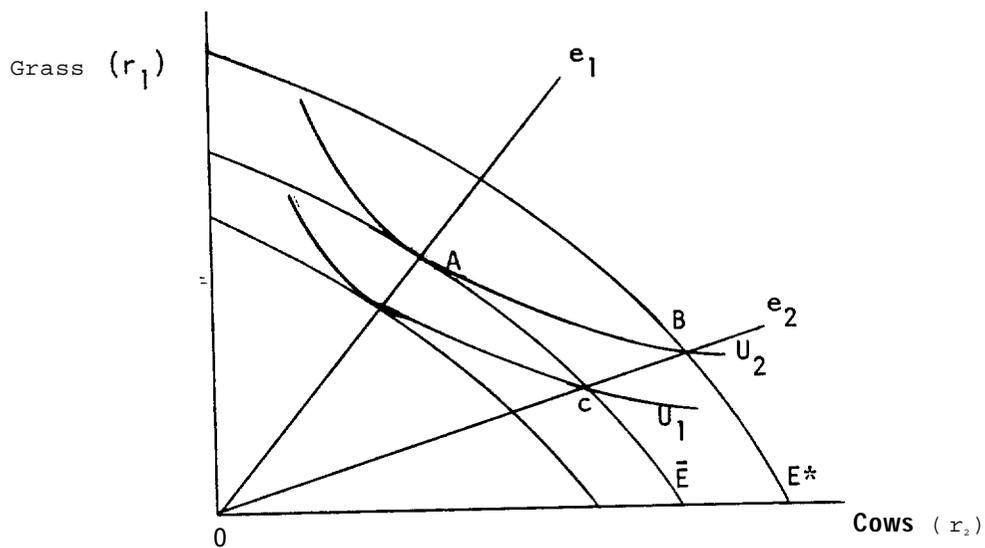
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Figure 4.6
A Diversity Possibilities Frontier

Grass (r_1)

e_1

E

e_2

e_3

e_4

Cows (r_2)

Figure 4.7
The Compensating Function

Grass (r_1)

e_1

A

B

e_2

U_2

c

U_1

$\bar{E}$

E*

Cows (r_2)

0

there could exist a series of similar diversity possibility frontiers, one for each level of **energy** availability. The greater the level of energy availability, the farther would be the associated frontier from the origin. Therefore the biomass of any species obtained in a particular ecosystem to which greater quantities of **energy** are made available will increase but not necessarily on a one-to-one basis with the increase in available energy.

To make different ecosystems comparable, we define the solar radiation to which the location in question is exposed per period as the unit amount of energy, E_o. Each of the ecosystems that can be produced by this unit energy are therefore comparable in terms of the **biomasses** of each species embodied in them. We shall **call** them unit ecosystems. Keeping in mind that an ecosystem is defined as embodying species in fixed proportions, an altered quantity of an ecosystem is a simple multiple of the **quantity of** any species appearing to some positive degree in the unit ecosystem.

To complete the most fundamental parts of our analytical apparatus, we introduce a well-behaved utility function, $U(r)$, for a representative person. **Assuming others, energy** subsidies to the relevant location to be predetermined, the **Lagrangian** of this individual's decision problem then can be stated as:

$$L = U(r) + \mu(\bar{E} - \phi(r)).$$

(21)

The first-order necessary conditions for a maximum of (21) are,

$$\frac{\partial U}{\partial r} - \mu \frac{\partial \phi}{\partial r} = 0$$

(22)

and the constraint expressing the available energy. Expression (22) states that the individual will equate the marginal utility he obtains from an additional unit of a species to the marginal cost of expending the energy to acquire that additional unit. Figure 7 is a diagrammatic representation of (22) for two types of ecosystems, e_1, and e_2, and two indifference curves U_1, and U_2. With available energy, $\bar{E}$, the individual's utility-maximizing choice is clearly at A, which corresponds to (22). We shall therefore call any ecosystem which conforms to (22) the ideal ecosystem. This is the ecosystem having that species assortment most preferred by the individual.

Assume that our representative individual, perhaps because he is unable to exercise enough influence over land use, cannot have the e_1 ecosystem. Instead, he must face the e_2 system, a system containing substantially more cows and less grass. The **latter** system may be considered to be less "natural" since its maintenance likely requires substantial man-supplied energy

subsidies. With the available energy, $\bar{E}$, the individual will be worse off with the e_2 system since the highest utility level he will be able to reach is U_1 at C. If he were to be as well off with the e_2 system as he would be with the ideal system at A, he would have to be at B. The attainment of B, however, requires more input **energy** as indicated by the diversity possibilities frontier, E^*. Since OA and OC both require $\bar{E}$ units of **energy**, while OB requires E^* ʻenergy units, the **energy quantity** required to compensate the individual for the fact of the e_2 system is $E^* - \bar{E}$ along the e_2-ray. The compensating ratio, $OB/OC \geq 1$, is then the quantity of the existing system relative to the quantity of the ideal. system that keeps the individual at the original utility level. Since OB and OC are each defined in energy units, the compensating ratio is a pure number. A glance at Figure 7 makes it obvious that this compensating ratio will be greater, the less substitutable the two systems are for one another, the steeper the slopes of the diversity possibility frontiers, and the wider the difference between the ideal ecosystem and the actual ecosystem. In addition to depending upon underlying preferences and production conditions, this ratio is obviously a function $h(e,e^*)$, where e^* is the species ratio in the ideal ecosystem and e is the species ratio in the existing system. Lancaster (1975, p. 57) describes the properties of this compensating function, which must be convex.

If all existing ecosystems are not to be ideal ecosystems, the preceding framework implies that in the real world there are some ecosystems produced under conditions of increasing returns-to-scale. If decreasing **returns-to-scale** were universal, less energy would be used by producing fewer units of a greater variety of ecosystems. In the extreme, each individual would have his ideal ecosystem available to him. Similarly, under constant **returns-to-scale**, the quantity of energy used to produce a quantity of an ecosystem is directly proportional. Thus , with decreasing or constant returns-to-scale, any individual. who does not have his ideal ecosystem available is using more input energy to attain a particular utility level than would be required with his ideal ecosystem. Casual observation suggests that everyone is not happy with the ecosystems they have available. One plausible reason for this is the presence of increasing returns-to-scale in the production of **ecosystems.**[10/] That is, the presence of increasing returns-to-scale for some ecosystems may force the individual to choose between an ideal diversity of ecosystem components and reduced energy consumption per unit of production for some smaller set of these components.

Let us momentarily return to (20), which gives the amount of input energy required to produce some amount of a particular ecosystem. Because of our use of energy to bring the unit quantities of different ecosystems to the same measure, and because of the properties we have assigned to the diversity possibilities frontier, if Q_1, and Q_2 represent quantities of different

ecosystems, e_1 and e_2, then $f_1(Q_1) = f_2(Q_2)$ when $Q_1 = Q_2$. This allows 's 'º
perform the analysis in terms of a single input function:

$$E = f(Q) \tag{23}$$

The energy required to produce quantity Q_1 of e_1 and quantity Q_2 of e_2 is
given by the sum of the two input functions:

$$E = f_1(Q_1) + f_2(Q_2), \tag{24}$$

and not the sum of the quantities of $(Q_1 + Q_2)$. If $f_1(Q_1) + f_2(Q_2) = f(Q_1 + Q_2)$, then constant returns-to-scale would exist. As usual, we assume
$f(Q) > 0$, and $f'(Q) > 0$, but we need not assume that all incoming energy
results in additional biomass, nor need we attach any sign to $f''(0)$.

Now define a degree of economies-of-scale parameter, $\theta(Q)$, which is the
ratio of the average energy input requirement to the marginal energy input
requirement. This is simply the inverse of the elasticity of (23), or:

$$\theta(Q) = \frac{f(Q)}{Qf'(Q)} = \frac{f}{Q} f' \tag{25}$$

If θ is a constant, $f(Q)$ will then have the form:

$$E = E_0 Q^{1/\theta}, \tag{26}$$

the inverse of which is

$$Q = aE^{\theta} \tag{27}$$

This last expression is immediately recongizable as a homogeneous function of
degree 0. If $\theta > 1$, there are increasing returns-to-scale; if $\theta = 1$, there
are constant returns-to-scale, and if $\theta < 1$, there are decreasing
returns-to-scale.

In expressions (21) - (22), we derived the representative individual's
ideal diversity of ecosystem components, assuming that he faced no tradeoffs
between this ideal and lowered unit energy costs of ecosystem production. We
are now prepared to consider this question of the optimal deviation of the
actual ecosystem available to the individual from the individual's ideal
ecosystem.

Assume we wish to enable the individual to reach some predetermined
arbitrary utility level with minimum use of energy.[11] Let Q* be the
quantity of an. ideal ecosystem, e*, that is required for the individual to

reach this predetermined utility level. If the available ecosystem, e, is nonideal, the individual will have to be compensated by being provided more than Q* of the available system. According to our previous definition of the compensating function, h(e, e*), the amount of the available eco-system required to bring the individual up to the predetermined utility level will be Q*h(e, e*). Since the input function (23) is independent of the species ratios (by the assumed homotheticity of production and the definition of unit quantities), the optimal ecosystem is that which minimizes the quantity, Q, required to reach the predetermined utility level. That is, we wish to minimize:

$$Q = C/*h(e, e*) \tag{28}$$

This minimum is given by:

$$Q* \frac{\partial h}{\partial e} = 0 \tag{29}$$

which obviously corresponds to (22). This result is relatively trivial but it does serve as a necessary prelude to determination of the optimal deviation of the available ecosystem from the ideal ecosystem.

Suppose there are n-1 less-than-ideal feasible ecosystems, the deviation of each less-than-ideal system from the ideal system being given by $x_i = e* - e_i$. Then the quantity of the ith ecosystem required to reach the predetermined utility level is given by: $Q_i = Q*h(x_i)$. The total energy inputs required to reach this utility level for all systems, whether ideal or not, are then:

$$E = f[Q*h(x_i)], \tag{30}$$

where the x_i are the variables of the problem. From (30) is obtained:

$$\frac{dE}{dx_i} = Q* \frac{df}{dh} \frac{dh}{dx_i} = Q*f'h' \tag{31a}$$

or

$$Q*h' = \frac{1}{f'} \tag{31b}$$

for a minimum expenditure of energy.

The interpretation of (31b) in economic terms is quite easy. The l.h.s. of the expression shows the increase in the quantity of the ith ecosystem required to maintain the predetermined utility level if there is a one unit biomass increase in the deviation of the available ecosystem from the ideal

ecosystem. The denominator of the term on the r.h.s. shows the increase in. the available quantity of the ith ecosystem to be obtained with a one unit increase in input energy. Thus (31b) says that the optimal deviation of the available ecosystem from the ideal ecosystem occurs when the change in the compensating ratio is equal to the reciprocal. of the additional energy required to produce more of the ith ecosystem. As the available ecosystem deviates less from the ideal system, the compensating ratio decreases. If the energy inputs required to reach the predetermined utility level also decrease, then the ideal system would clearly be optimal. However, if the compensating ratio increases and, due perhaps to economies-of-scale in production with simplified ecosystems, energy inputs per unit of yield decrease, then the achievement of an optimum requires that the tradeoff between the two be recognized.

The optimum condition (31b) can be clarified when stated in elasticity terms. Upon defining the elasticity of compensating function as $\eta_h = xh'/h$ and substituting this and the elasticity, (23), of the input function into (31b), we have

$$\eta_h \left(\frac{h}{x}\right) = \frac{Q*\theta}{f} \; , \tag{32}$$

which if f, h, and Q are fixed is simply

$$\eta_h(x) = \theta \; . \tag{32b}$$

Thus the optimal deviation of the available ecosystem from the ideal ecosystem occurs where the elasticity of the compensating function, $\eta_h(x)$, is equal to the degree, θ, of economics of scale in production. If x were such that $\eta_h(x) > \theta$, a one percent decrease in deviation of the available ecosystem would require η_h percent less in ecosystem quantity (remembering that all ecosystems are measured in the same units because they are defined relative to a unit ecosystem) and require $\eta_h(x)/\theta > 1$ percent less energy resources, so that energy inputs would be made smaller by reducing the extent Of deviation from the ideal system. However, if $\eta_h(x)/\theta < 1$, an increase in the extent of deviation would reduce energy inputs. Thus when $\eta_h(x) = \theta$, the deviation. is optimal. The welfare loss from an increase in the deviation of the available ecosystem from the ideal ecosystem is balanced by the increased ecosystem quantity obtained for a given energy input.

The Impact of Pollution

In the previous section, we have presumed that over some interval of the input function, (23), there exists increasing returns-to-scale: that is, as

more energy is devoted to the production of a particular ecosystem, the ecosystem yield per unit of energy is increasing. When there are feasible monocultural ecosystems that yield an output (e.g. beef) highly valued for consumptive purposes, or as an input (e.g., sawtimber) for a fabricated good, and if these ecosystems exhibit increasing returns-to-scale, then some deviation of the available ecosystem from the ideal ecosystem may be optimal. The condition for optimality is $\theta * h' = (f')^{-1}$ or, in elasticity terms, $\eta_h(x) = \theta$ It is thus apparent that the extent of optimal deviation will vary with the parameters that influence the above conditions. The elasticity, η_h, is determined by the properties of the compensating function, h. The economies-of-scale parameter, θ, is either an exogenous parameter (with homogenous production) or is a function of yield, and thus of the compensating function.

Consider a pollutant, α, which might, in principle, effect h' , f' , or both. For example, a pollutant stresses ecosystems, making them immature, and thus less diverse. In addition, for at least some of the ecosystems remaining viable after the introduction of a pollutant, their yields are less than they would be without the presence of the pollutant, i.e. , the level of ecosystem yield obtainable with any given provision of energy is reduced. Thus, in terms of Figure 7, the diversity reduction would be reflected in a rotation of the available ecosystem toward one or the other axes, while the reduction of yield of whatever ecosystem was ultimately available would register in a shift of the diversity possibility frontiers toward the origin. If the ideal ecosystem is unchanged, and if the reduction in diversity represents a movement away from this ideal system, then the individual will require additional compensation if he is to remain at the original utility level. A similar result occurs if f' (the additional energy input required to obtain an additional unit of an ecosystem) increases. In both cases, an increase in the deviation of the optimal from the ideal ecosystem occurs. The effect of a variation in a on the optimal deviation is easily found by differentiating either (31b) or (32b).

Upon differentiating (32b) with respect to a, we get:

$$\frac{dx}{da} = \frac{(d\theta/d\alpha) - (d\eta_h/d\alpha)}{(d\eta_h/dx) - (d\theta/dQ)} \tag{33}$$

Given the convexity of the indifference curves, the $d\eta_h/dx$ term in the denominator must be positive. If the degree of economies-of-scale is fixed or declines with increases in the level of output, the $d\theta/dQ$ term in the denominator must be negative. Thus the denominator in (33) will be unambiguously positive. The sign for (33) will therefore depend solely upon the terms of

the numerator.   If the ideal ecosystem has high diversity, the sign of $d\eta_h/d\alpha$ will be positive since the convexity of the indifference curve requires that reduced ecosystem simplification imply increased responsiveness of the necessary compensation to further simplification.

The sign of $d\theta/d\alpha$ in (33) is less easily determined.   Remembering that θ = (f)f'/(Q), it is plausible that increases in α would increase only f', implying that $d\theta/d\alpha$ would be positive, but leaving the sign of the numerator in (33) dependent on the relative magnitudes of $d\theta/d\alpha$ and $d\eta_h/d\alpha$. It is of course possible that pollution would reduce the yields obtainable for every ecosystem for all output levels.   This event would be reflected in a reduction in f, implying that $d\theta/d\alpha < 0$, for a given f' and Q.   In this case, the increase in pollution would reduce rather than increase the optimal deviation of the available ecosystem from the ideal ecosystem!

These results obviously imply that economic analyses which concentrate only on the ecosystem yield effects of pollution can be seriously misleading. In cases where pollution reduces both yields and diversity, the analyses will tend to underestimate the economic losses from the effects.   Similarly, if there exist cases where diversity is decreased while yields are increased, the usual analyses might not perceive any losses.   However, in some cases, the usual analyses will exaggerate the severity of the losses.   Harkov and Brennan (1979 pp. 157-158) conclude, for example, "... that slower growing trees, which often typify late successional communities, are less susceptible to oxidant damage than rapid-growing tree species, which are commonly early successional species." Assuming that the ideal ecosystem is more diverse than was the available ecosystem before the increase in pollution, the increase in pollution could reduce f',θ, or both.   In either circumstance, more incoming energy would be required than before to obtain a given yield with the immature ecosystem.   The pollution may therefore reduce the optimal deviation of the available ecosystem from the ideal system.   In short, pollution can enhance rather than hinder the willingness of individuals to live with mature biological communities! Obviously, in this case, any economic analysis which neglected the increase in diversity would overestimate the economic damages attributable to the pollution.

A Simple General Equilibrium Model.

A simple general equilibrium model of an economy and ecosystem will now be presented that in some respects captures more dimensions of our basic concerns than do preceding sections, but which does so at the cost of neglecting some dimensions that the preceding sections feature.   The ecosystem will be represented by the single stored energy variable r. Of course, this masks

many interesting questions (e.g. diversity vs. scale economies) due to the level of aggregation taking place. Nevertheless, the ecosystem solves the one-input problem

$$\max\; r = \epsilon_1 x_1 - \epsilon_2 x_2$$

$$\text{St.}\quad x_2 = g(x_1; E_r) \tag{34}$$

where E_r is a parameter indicating the amount of human supplied energy into the ecosystem. In the second section, we saw that $E_r = 0$. The solution to the problem is characterized by the first-order condition,

$$\frac{\epsilon_2}{\epsilon_1} = \frac{\partial g(x_1; E_r)}{\partial x_1} \tag{35}$$

This is the analogue of (12). If the ecosystem is in equilibrium, with no human interaction (i.e., $E_r = 0$), $r = 0$.

In order to capture a general equilibrium setting, we now introduce a Hicksian composite good, z, into the individual's utility function. Thus human preferences are given by:

$$U(z,r) \tag{36}$$

The term r appears in the utility function to indicate the human preference for a natural environment. Ideally, that environment should be pollution free with little trace of intervention. In other words, for some z value, zero is an optimum value of r. As intervention increases through increased E_r, r increases and utility decreases for fixed z. Consumer preferences are shown by the indifference curves of Figure 8. The arrow shows the direction of preference.

The production of z is given by the function

$$z = f(E_z, r) \tag{37}$$

where E_z is the energy used in the production of z. Stored energy enters z since it represents that part of the ecosystem which is cropped to provide goods in the economy.

The human problem is to maximize (36), subject to (35), (37), and the resource constraint on total available energy.

$$E_r + E_z \cdot \bar{E} \tag{38}$$
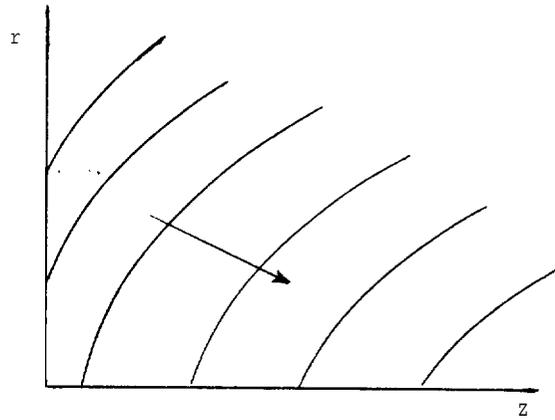
Figure 4.8
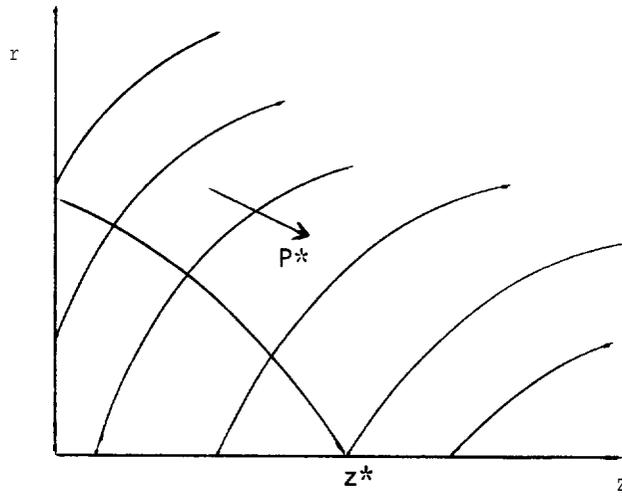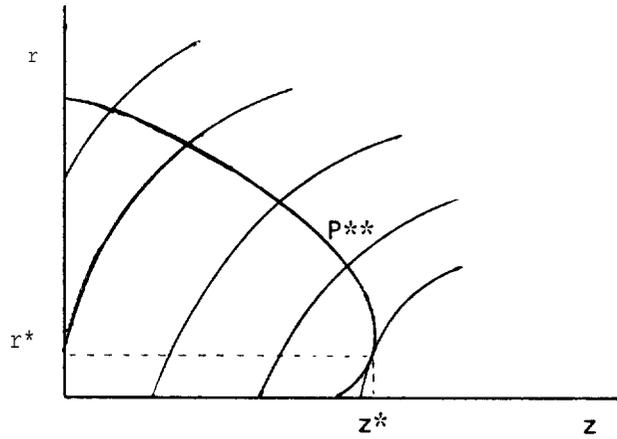Consumer Preferences

Figure 4.9
A Natural State Optimum

Figure 4.10
An Interventionist Optimum

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The solution is shown graphically by the two possibility curves in Figures 9 and 10. In Figure 9, curve P* is the production possibility frontier. As we move from z*, incoming energy is being diverted from the production of z to r, and, therefore, more r is produced. Greater r means more natural environment is available for producing z. However, the shape of P* indicates that the increase in r does not make up for the decrease in E_z in the production of z. The optimum is z* where the ecosystem is in a natural state.

The second possibility is curve P** in Figure 10. Again, energy is being diverted to r. But now in producing z, the increase in r more than makes up for the loss of energy E as shown by the shape of P**. The optimum is now r**, z** where intervention in the ecosystem is justified. Examples of these possibilities may be forest harvesting since most would agree that harvesting forests for lumber is a worthwhile task. The first case may be harvesting baby harp seals, since many argue that the goods made from the seals can be made inexpensively using synthetics.

While this is a very simple example, it is a useful means of displaying the potential for describing the links between economies and ecosystems. Questions of optimum exploitation and extinction can be inferred from sophisticated versions of the analyses in Figures 8 through 10. But research is needed to determine the shape of the possibility frontiers, which means that research into physiology sets of ecosystems and the technology sets of economies will be required.

Summa ry and Conclusions

We have tried to demonstrate how the application of economic analysis to bioenergetics, a framework with some degree of acceptance in ecology, can be used to describe the behavior of ecosystems. Moreover, we have indicated how the descriptions thereby obtained can be made an integral part of a model adapted from Lancaster (1975) that, in principle, can be used to value both the yield and the diversity impacts of stresses upon ecosystems. We are by no means the first to express the thought that the human-induced ecosystem effects for which one may feel secure using the conventional methods of benefit-cost analysis may be those having the least long-term economic significance. The conventional analysis disregards mayflies because their contribution to the food supply of trout has been untraceable. We believe further attempts to combine bioenergetics and economic analysis might make this neglect untenable. Neglect of the life support services that mayflies and their peers provide for trout may mean that the ultimate effects of pollution on trout, via mayflies, may go unrecognized and therefore unaccounted.

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Just as the conventional analysis disregards the life support services provided by soil microbes, dung beetles, and caddisflies, it focuses upon an (incomplete) item-by-item listing of organisms in the ecosystem while failing to consider how the proportions in which these organisms are present might be sources of human pleasure. The ecologist, even though he has lacked an acceptable means to value ecosystem diversity, seems to have been more sensitive to this source of welfare than has the benefit-cost analyst. Economic efficiency, narrowly interpreted as minimizing the inferred or observed cost of producing a given quantity of ecosystem yields (and thereby taking advantage of all scale economies), need not result in maximum human welfare if there exists diversity in tastes among individuals for types of ecosystems or if ecosystem components are not valued independently of the environmental state from which they come. We speculate that traditional benefit-cost analysis, to the extent that the information it generates has been used for decision purposes, may occasionally have fostered Pareto- losses rather than Pareto-improvements. At a minimum, it has probably brought about wealth transfers from those who value ecosystem diversity and variety to those who possess the machinery for producing and maintaining ecosystem homogeneity.

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1/ This ignores other possibilities like geothermal systems or tides.

2/ This is somewhat simplified in that it ignores more complex chains.

3/ Lotka likens the development of this model to the work of Jevans and the marginalist school of economists. He recognizes that this maximal is not appropriate for humans. Borrowing from Pareto, he describes humans as maximizers of pleasure. This is consistent with maximizing species growth only if the marginal pleasures (i.e., marginal utilities) are proportional to the marginal productivities of the physical needs. Thus, Lotka essentially denies the validity of an energy theory of value which, as pointed out earlier, has been propounded by many modern day ecologists.

4/ The physiology set is analogous to the firm's technology set often used in economics. The development of the model presented here closely parallels the development of the economic model in Russell and Wilkinson (1979, Chapter 7).

5/ This is paraphrased for Russell and Wilkinson's (1979, p. 129) definition of technologically efficient bundles.

6/ Condition (12) is analogous to the geometric solutions of Rapport (1971) where he determines the optimum selection of two different preys. His indifference curves represent two net inputs and one net output in. the model used here.

7/ The numbers of a particular species are capable of interbreeding.

8/ See Freeman (1979) for a thorough survey of available techniques for answering this question.

9/ The work of Bigelow and his colleagues (1977) is a detailed account of the ecosystem possibilities in a Dutch estuary. Odom (1971) and other ecology texts are replete with other examples.

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10/ Other plausible reasons exist. For example, a process through which the individual can register his ecosystem preferences may be lacking.

11/ The envelope theorem (Shephard's lemma) assures us that the solution to this problem is equivalent to the solution of the utility maximization problem.

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v.   NATURAL SCIENCE RESEARCH USEFUL TO THE ECONOMIST

Introduction

Throughout the preceding chapters, we have emphasized the necessity of knowing the influence of various physical and biological factors upon some ecosystem variable of interest if economic methods for assessing the benefits of controlling acid precipitation are to be applied.[1]   At the same time, we have formulated several analytical and empirical characterizations of the acid precipitation problem intended to be helpful in deciding which of these relations are likely to be worthy of more immediate research attention. For example, our discussion of nonconvexities and irreversibilities in Chapter 111 leads to the conclusion that the very early stages of ecosystem acidification often have the greatest economic consequences. The devotion of research resources to understandings of the behaviors of already highly acidified systems may, therefore, yield little information that is economically important. However, before abandoning or greatly reducing research on already highly acidified systems, it is obviously important to establish accurately the temporal and spatial frequencies of the nonconvexity and irreversibility issues.   If these issues appear with considerable frequency, then an allocation of research resources that accords with the ordering of current annual sectoral control benefits estimated in the "first exercise" of Chapter II might well be mistaken. The economic import of a unit of information on indirect ecosystem effects could presently be much higher than would more information on materials damages or direct agricultural effects.

The treatment in Chapter IV is intended to reinforce the theme that the (relatively) easily observed current direct economic effects of acid precipitation could readily have the least long-term economic significance. By providing a skeleton for combining economic analysis with ecological energetic that is built upon resource allocation processes, we have tried to establish a basis for valuing the possible effects of acid precipitation upon the life support services and human pleasures that ecosystems supply. Traditional economic assessment methods, as set forth in Chapter I, disregard these services except insofar as they are valued independently of the environmental states that produced them. Any empirical implementation of the skeleton set forth in Chapter IV that captures at least some features of the values of these life support services will clearly require substantial contributions from that part of ecology which describes the combinations and

quantities of ecosystem components resulting from various quantities of available energy.

Although knowledge of the response of some result to various mixes and magnitudes of inputs is central to the concerns of previous chapters, we have as yet discussed few criteria for deciding when a particular response, given limited research resoürces, is worthy of attention. In succeeding sections of this last chapter, we present some qualitative criteria for deciding when attention is warranted. We also shall point to some factors that might determine the relative benefits and costs of alternative research efforts into particular ecosystem responses to acid precipitation. In economic language, the concern of this chapter is with the value of research into the effects of acid precipitation upon ecosystem production functions or response surfaces. Because the economist's concept of the production function often differs in subtle but economically important ways from the natural scientist's idea of a dose- response function, we take a brief respite in the next section from the central purpose of the chapter to present a brief overview of concepts in production theory particularly relevant to later discussion.

The Production Function

All results or outputs require at least two kinds of causative agents or inputs. Usually many more than two inputs are required. In general:

$$Y = f(X_1, X_2, \ldots, X_n),\tag{1}$$

where Y is the quantity in similar units of an output rather than the number of possibly dissimilar individuals in some biological. population, the X_i (i = 1,...,n) are input quantities which may themselves be an output of some other production process, and Y, $X_h \geq 0$. without exception. It is usually, but need not be, assumed that (1) is twice differentiable, with $\partial Y/\partial X_i > 0$, $\partial^2 y/\partial x_i^2 < 0$, and $\Sigma(X_i/Y)(\partial Y/\partial X_i) < 1$. Negative inputs such as acid precipitation can be defined so that redactions in their levels constitute positive inputs. The first two assumptions are typically referred to respectively as positive but diminishing marginal products, while the third assumption represents decreasing returns-to-scale. The expression (1) is typically viewed as being perfectly reversible, where reversibility is defined as the absence of asymmetrical changes with respect to the status quo point and the direction. of movement. Rarely are any restrictions placed upon the sign of $\partial^2 Y/\partial X_i \partial X_j$ for i ≠ j.

Expression (1) implies that all the X_i are variable and of relevance for determining the value of Y. However, there are many instances where the influence of an X_i upon a Y is trivial or nonexistent either because the X_i is

fixed or has so little influence that it can be disregarded. Thus if n-m inputs are fixed or considered to be trivial, (1) can be written as:

$$Y = f(X_1, \ldots X_m; X_{m+1}, \ldots, x_n),$$ (2)

with the X's to the right of the semicolon being treated as irrelevant for the problem at hand.

Neither (1) nor (2) are necessarily concerned with growth in terms of the number of individuals in some biological population. Temporal considerations may nevertheless be introduced by treating time as one of the inputs or by treating the inputs themselves as functions of time. However, most economic treatments treat the time interval as fixed and emphasize various relations between and among the biophysical and human inputs and between these inputs and the outputs. These latter relations, rather than population dynamics considerations, tend to be emphasized because they are the key to most applications of the economic assessment methodologies outlined in Chapter I.

For a <u>particular</u> level of output, Y, rates of substitution, dX_1/dX_2, between any pair of inputs, X_1 and X_2, can be determined by total implicit differentiation of $\bar{Y} = f(X_1, X_2)$. Thus, since $X_1 = f(X_2, \bar{Y})$, we have:

$$\frac{\partial Y}{\partial X_1} \frac{dX_1}{dX_2} + \frac{\partial Y}{\partial X_2} = 0$$

and therefore:

$$\frac{dX_1}{dX_2} \qquad \frac{\partial Y/\partial X_2}{\partial Y/\partial X_1}$$ (3)

where, as before, the numerator and the denominator on the right-hand-side are the marginal products of the respective inputs. If the marginal products are positive, (3) means that the level curve or isoquant depicting dX_1/dX_2 for a particular $\bar{Y}$ must have a negative slope as in Figure 1. The isoquant, $\bar{Y}$, in Figure 1. does not represent the rate of substitution of X_1 for X_2 in any basic biochemical or physiological process or production technique. It merely displays the fact that within limits the same quantity of output can be obtained from various combinations of possibly very diverse inputs. For example, there are probably numerous combinations of reductions in acid precipitation and liming of forest soils which will result in identical standing stocks of timber. 'The underlying physiological processes are of interest only insofar as they contribute to comprehension of the effects of input mixes and magnitudes upon an output or result that has economic

relevance.

Given that there are positive marginal products for all inputs, there will exist a series of isoquants like those depicted in Figure 1. Levels of output are increasing as one moves away from the origin. The set of all such isoquants is a response surface. If all inputs but one are fixed, say at X_2 in Figure 1, then the response of Y to various applications of X_1 is a response function.

If the marginal products of each input are someplace positive but diminishing ($\partial Y/\partial X_i > 0$, $\partial^2 Y/\partial X_i^2 < 0$), then some portion of a level curve or isoquant depicting dX_1/dX_2 for a particular Y will have a convex shape as in Figure 1. 2/ This implies that as one moves up (down) the isoquant, it becomes progressively more difficult to substitute $X_2(X_1)$ for X1(X2); that is, a larger and larger quantity of $X_2(X_1)$ is required to replace the loss of a unit of $X_1(X_2)$ if the level of output is to remain. unchanged. There is, of course, no reason why the isoquant could not be depicted as in Figure 2, where the concave interval ABCD implies either that the marginal product of one or the other inputs has become negative (the intervals AB and DC), or that the marginal products of both inputs are negative (the interval BC). Whether reference is to human decisions or to the behavior of a nonhuman organism, if the isoquant were everywhere concave, only one input would ever be used since the marginal benefits of use of the first input would decrease the more of the other input was used. The use of only one input does not usually accord with experience in either the human or natural worlds, thus implying convexity of the level curves. Production objectives would be ill-served by operating in the concave portion of the isoquant (the interval ABCD): the same level of output could be obtained by using less of both inputs or less of one input and no more of the other input. 3/

In Figure 2, we see that the concave portion (the interval ABCD) of an isoquant need not be described in any detail because these portions ill-serve any organism that acts "as if" it wishes to minimize the resources that must be expended to reach a given level of an objective. For example, a human might wish to minimize the costly resources he must use to achieve a given goal, and a nonhuman organism might behave so as to minimize the available energy it must expend to acquire a particular amount of nutrition. If only those portions of the response surface are studied where all inputs have positive marginal products, one may rest assured that concave portions are being avoided.

Economic analysis can be employed to delimit further the portions of the response surface that are worthy of description if organisms behave as if they minimize the resources that must be expended to reach a given level of an

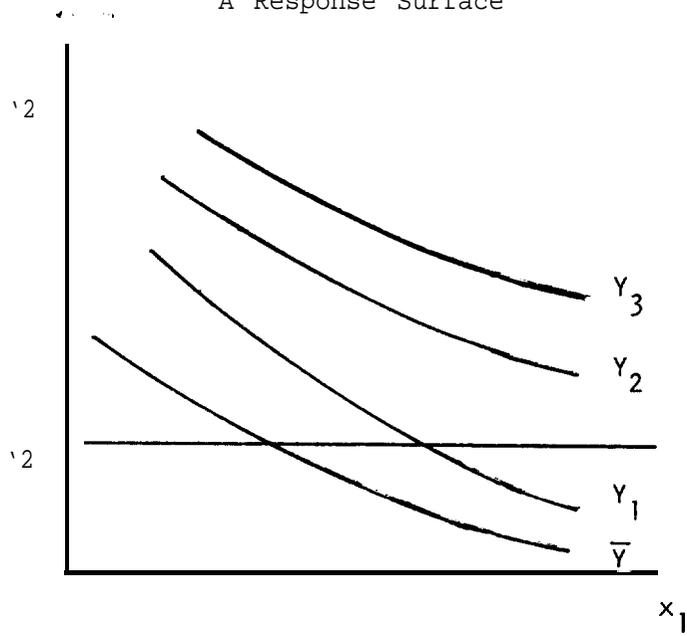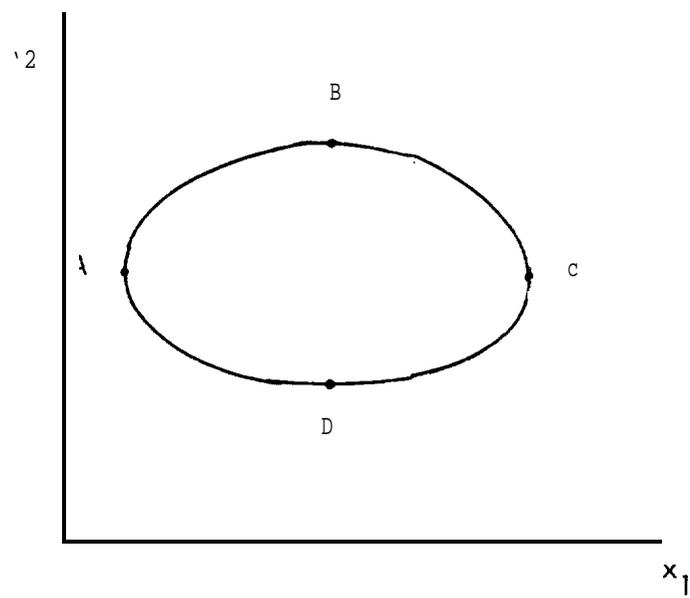Figure 5.1
A Response Surface

Figure 5.2
Convexity and Concavity

objective, or, equivalently, as if they maximize subject to available resources the level of attainment of some objective, whatever this objective might be. Reconsider the fox in Chapter IV who obtained his nourishment from various combinations of rabbits and squirrels. The combinations that he chose and therefore the only steady-state or long-run equilibrium combinations that would be observed in nature would conform to a condition where any reduction in the net input of energy obtained from rabbits (squirrels) would be matched by an increase in the net input of energy obtained from squirrels (rabbits). Thus , if one were trying to describe the effect of pollution upon the feeding habits of foxes with respect to rabbits and squirrels, only those combinations of rabbits and squirrels on the convex portion of each fox isoquant that conformed to the condition under various pollution levels would be of interest. Of course, these combinations may themselves constitute the object of any research effort. Nevertheless , it is likely that an accumulation of research knowledge would ultimately indicate that some rabbit and squirrel combinations on the convex protions of the isoquants are clearly inconsistent with the condition, meaning that their impact upon the well-being of the fox need not be candidates for description. They would certainly be of no concern to the fox, and if the only research object is to describe naturally occurring states, information about them would be of no value to humans. Alternatively, if it is initially thought that any one of the combinations on the convex portion of a particular isoquant could ultimately prove to conform to the condition, information on the state of the fox's well-being under each of these combinations would have some positive value. In short, the researcher, if he is interested in describing naturally occurring states must dismiss consideration of input combinations known to be inconsistent with the behavior of the organism that is the subject of the research. Economic analyses of research allocation processes, as set forth in this and the previous two chapters, can contribute to identifying the aforementioned combinations. Those who refuse to let the behavior of organisms direct their research would apparently perceive no qualitative difference between studying the effect of feeding corn to a beached whale and studying the impact of SO_2 fumigations upon a laboratory plant that is supplied with more nutrients than it could or would acquire in its natural or agricultural state.

The Value of Information and of Alternative Models

Returning momentarily to (1), there are several levels of completeness of knowledge that one might acquire about the effect of pollution on a given production or response surface. Completeness would involve knowledge of the coefficients attached to each of the input variables on the right-hand-side of (1) and of its functional form. In the absence of knowing the values of the coefficients knowledge of whether each input variable has a "strong" or a "weak" influence on the output would be nearly as useful. If this knowledge

is not directly available, **knowlege** of the functional form of (1) can allow deductions to be made about the relative **levels** of influence of particular input variables, given that one has some <u>a priori</u> idea about the plausible bounds for the values of some coefficients. Moreover, knowledge of functional form assists in directing research to those input variables likely to be most influential in determining output magnitudes. However, <u>a priori</u> knowledge of the functional form of (1) is very frequently beyond the analytical powers of the relevant disciplines to obtain. Most often, the specification of functional form must wait for the gradual accretion of empirical experience. Usually well before this empirical experience has been **fully** accumulated, deductive or empirical insight is acquired into the signs of $\partial X_2/\partial X_2$, $\partial Y/\partial X_1$, $\partial^2 Y/\partial x_i$, and $\partial^2 Y/\partial X_i \partial X_j$. As the bodies of theory in many **disciplines**, including macroeconomics and ecology, attest, **knowlege** of these signs can be most helpful. in drawing inferences about the underlying structure of the natural or social system being investigated. Having acquired these structural **insights,** bounds can often be imposed upon functional forms, the relative influences of variable pairs, etc. If **knowlege** of the signs attached to the preceding derivatives cannot be obtained, decisions founded on particular production or response surfaces must resort to simple listings of all or some of the variables thought to enter the right-hand-side of (1). However, unless these listings can ultimately be molded into a theoretical structure, they can contribute little to ultimate knowledge of the production or response surface. Only by sustained and substantial efforts to accumulate empirical experience can this knowledge be acquired. Even then, it must remain. unknown whether the accumulated empirical knowledge is **generalizable** to as vet **unobserved** events or whether different results obtained from seemingly similar settings are reconcilable.

There exist, as is clear from the preceding remarks, two **mutually** reinforcing yet partially substitutable fundamental ways in which **knowlege** about response surfaces can be acquired. Two legs, the theoretical and the empirical, are required to walk well, but for some tasks, one leg can accomplish more than the other. The question nevertheless remains as to how far toward complete specification of the form of the response surface investigation, whether theoretical and/or empirical, must proceed. This question can best be understood within the context of the economics of information. Two concepts, the value of information and the value of alternative models, are central to any research effort into the effects of acid precipitation upon the response surfaces of various ecosystems components.

The results of this research are intended to be of direct use to persons who must make decisions about the control of acid precipitation or to serve as inputs into other research efforts providing results useful to decisionmakers.

Research designed only to reveal a greater understanding of basic biochemical or physiological processes must be evaluated on some basis other than that developed here. In order to establish a framework for evaluating research into the effects of acid precipitation upon the response surfaces of ecosystem components, one must consider together the decision which is at issue and the decisionmaker. As Crocker (1975, p. 342.) remarks, "the choice of a particular research effort or information system implies the use of a particular class of decision models since certain types of information are relevant to some models and not relevant to others. Conversely, the choice of a decision model implies the use of a particular class of information systems yielding the parameters of the model.--/ The decision variable of interest here is the amount of acid precipitation to which an ecosystem component is to be subjected. The payoff from the decision is the net benefits of controlling the acid precipitation, defined as the economic value of the ecosystem component damages prevented less the cost of controlling the acid precipitation. The payoff is related to the decision through some imperfectly understood response surface.

As earlier noted, the arguments of the response surface include a great many other variables in addition to acid precipitation. The imperfectly understood response surface is approximated by some expression such as (1), where some X's might represent a taxonomic system (e.g., soil classes) originally established for an entirely different purpose, other X's might be measures set up specifically for the study of acid precipitation effects upon the ecosystem component of interest, and still other X's are inputs which can be measured but not predicted. Finally at least one X in (1) must represent a residual or error term intended to capture unknown, unacknowledged, and purely stochastic influences on the response surface.

The payoff, π is approximately related to the decision variable as:

$$\Pi = p_1 f(\cdot) - CX_A \qquad (4)$$

where p is the observed or inferred unit price of the ecosystem component of interest, c is the cost of reducing acid precipitation by one unit, and X_A is the number of units of acid precipitation. Since there exist unknown, unacknowledged, and purely stochastic influences upon $f(\cdot)$, and since the values of some other variables cannot be predicted prior to the control decision, for any given level of acid precipitation, the payoff is a random variable.

Whether performed by economists or noneconomists, the standard way to account for the randomness in expressions such as (4) has been to use range sensitivity tests. Waddell (1974), for example, includes upper and lower

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bounds and "best guesses" for various air pollution damage categories. A similar procedure is adopted in most of the ambitious work in d'Arge, et al. (1975) on the economic impact of climatic change as well as in Fisher's, et al. (1979) work on air pollution. damages in the State of California. An alternative but unfortunately rarely used procedure is to generate probability distributions for the random variables or the right-hand-side of (4), and then to aggregate these distributions to produce a probability distribution for the payoff measure.

Two readily un.derstandable examples of this approach, where the Weibull (1951) family of distributions is employed, are Pouliquen (1.970) and Mercer and Morgan (1975). These studies demonstrate that the valuable information made available to the decisionmaker and the researcher can be considerably enhanced: not only is he provided with the range of possible outcomes and payoffs but he is also presented with various common summary statistics allowing him to assign a probability statement to each outcome. These statements can be subjective rather than objective. Accumulated wisdom and intuition can be incorporated in an explicit and communicable fashion. Although many would object to the inclusion of subjective information, the question of real importance is not whether a particular probability assessment is subjective or objective but whether it has important consequences for the decision problem. Rather than fulminating over variables in some particular algebraic specification that fail to have coefficients significantly different from zero, most concern should be displayed about whether the formulation in question predicts better than the next best alternative. Errors of omission would seem no less worthy of critical scrutiny than errors of commission.

Another major advantage of the probability approach is that it does not throw away useful information. For example, in a poorly coordinated group research effort attempting to assess direct acid precipitation damages to commercial crops, the biochemist or agronomist might specify a response function relating some attribute of the crop to acid precipitation. This function, which the economist will employ to perform his assessment tasks, will likely be what the natural scientist considers to be the "best" of a set of several alternatives. In the absence of a thoroughly coordinated research effort in which the economist specifies the variables, units of measure, and sampling procedures the natural scientist is to use, it is likely that the natural scientist's conception of "best" does not coincide with the economist' s.

It is then up to the economist, who usually is only semi-literate in the relevant natural science, to translate the natural scientist's results into something useful for purposes of economic analysis. Moreover, by being asked to present a "best" function, a great deal of the natural scientist's unique

knowledge is being thrown away. Finally, the failure to report the full set of probable outcomes to the economist and thereby the decisionmaker means that yet another decision problem has been introduced: the natural scientist must assess which of the alternative formulations the decisionmaker will find most useful. By requiring that probabilities be assigned to the various plausible outcomes, the force of this decision problem is greatly ameliorated.

Any specification of a reponse surface will, except by chance, always be wrong. The suggested probability approach to the study of acid precipitation-induced response surfaces captures this fact. The implications of this for planning research into these response surfaces can be perceived by considering the investigator who must begin with very little information about the surface to be investigated. Guided by the principle that information should be acquired only as long as its value exceeds the cost of obtaining it, he can search for a finite number of kinds of information in varying quantities. Paraphrasing Marschak and Radner (1971), the value of additional information is the difference between the decisionmaker's current expectations of: (a) the payoff value that will occur if he chooses his act as well as he can without the information; and (b) the payoff value that will occur if he were to obtain the information and then choose his act as well as he can. In short, the value of the information is the increment in expected payoff that can be realized by having the information contribute to the decision.

When additional information is defined as a finer partitioning of some natural state, it may consist of both observations and experiments on a greater number of variables or on a particular variable, and a more discriminating model of the surface, i.e., a model that is better able to distinguish among alternative outcomes. The researcher must decide whether the reduced uncertainty and systematic broadening of identifiable alternatives that more information offers outweighs the costs of acquiring the information. The number of distinctions drawn can be no greater than the number of measurable consequences, if differences in payoffs are distinguishable only insofar as they generate measurably different results. In the next section, we take note of some of the more important aspects from the economist's perspective of this problem.

Issues in Designing Studies of Response Surfaces

Anyone who proposes to engage in estimation of, as opposed to expatiation about, response surfaces must give pragmatic consideration to several practical and interrelated issues. All these issues require compromises with the abstract analytical frameworks of the applicable disciplines. A reasonably complete listing with particular relevance to the study of acid precipitation-ecosystem component response surfaces might be as follows: the

design of response surface experiments; the estimation of these surfaces; the choice of a model to represent the surface; and the sources of discrepancies between response surfaces estimated in controlled or experimental conditions and observed in field conditions. We shall deal with each of these issues in sequence, trying to highlight those features of the issue that seem particularly relevent to studies of the impact of acid precipitation upon response surfaces.

Experimental Design: In situations where an experiment is the biologically appropriate way in which to generate and to test hypotheses about response surfaces, it is highly important that the economically relevant region (as defined in a previous section) of the surface be purposively and systematically covered. The great majority of biological research into response surface questions is of minimal use to the economist because it does no more than use analysis of variance techniques to establish only whether there exist statistically significant differences in the output obtained from a few levels of a single input. Rather than trying to design a systematic coverage of the economically relevant portion of the surface, the traditional emphasis has been and continues to be on replication, as if arbitrarily selected levels of statistical significance could impart structural understanding of system behaviour. Not only is the replication intended to improve the analysis of variance but to measure the variance as well. When the objective is to estimate a response surface, replication is much less essential. Primary concern should be with developing a model that predicts real world outcomes better than the next best alternative rather than testing whether the results of some particular model have statistically significant differences. Predictions are made so that something can be done: they are not first objects of contemplation. The proper object is informed manipulation of the system.

Changes in input mixes and magnitudes can substitute for replications of a particular input mix and magnitude since both types of observati.ons are intended to locate the response surface more accurately. For a given outlay of research resources, the information provided by more observations on output responses to an assortment of economically relevant input. mixes and magnitudes will usually be more valuable than will the information garnered from additional replications using a particular input mix and magnitude. Moreover, if alternative models have similar a priori plausibility as descriptors of a response surface, empirical discrimination among models will obviously be assisted more by increasing the breadth and the density of the sampling coverage of the surface rather than by replication of experiments directed at only one point on the surface. A near-infinity of models is consistent with a single point.

Bluntly put, the traditional experimental designs of biologists inv-
estigating response surfaces have been motivated by the maximization of disci-
plinary integrity. Designs have been structured via the mechanical
application of purely statistical criteria so as to minimize the probability
of accepting a false hypothesis. The result has been an excessive emphasis
upon replication, if the purpose of the research is taken to be the provision
of useful information to economists and to decisionmakers. To pose the point
in an extreme fashion, given that it is well-known that acid precipitation
harms fish, it is ridiculous even to advance for testing purposes the null
hypothesis that fish are unaffected by acid precipitation. Neither the
economist nor the decisionmaker cares whether there is a five per cent or less
chance that a fish-acid precipitation response surface exists. Their problem
is to know the value of the fish that are lost due to acid precipitation.
Thus, if disciplinary custom dictates the supplication of significance tests,
logic, rather than custom, requires instead that their application to the
value-related quantities derived from the response surface be stressed. This
stress would be consistent with our remarks in the previous section about the
desirability of having probability distributions for the payoff measure.

Put in yet another way, because of the reasonable desire of each
specialist to maximize his disciplinary integrity, a tension exists between
the biologist and the economist with respect to the design of response surface
research. The biologist will obtain less approval from his peers if he does
not replicate in accordance with traditional standards. The economist will
obtain less approval from his peers if he tries to draw inferences from a
small undense and narrow sample of the response surface. For the latter
individual, the cost of knowing nothing about large portions of the response
surface will typically greatly outweigh the costs of small errors in estimates
of a single point on that same surface. In design language, the economist is
interested in the magnitudes of differences in treatment effects rather than
in the existence of these differences.

Having pointed out a source of conflict in the desires of biologists and
economists with respect to the design of response surface experiments
conducted with limited research resources, we would like to provide some
specific criteria a neutral observer could use to weigh the tradeoff between
replication and density of coverage. Anderson and Dillon (1968) provide a
detailed treatment of the efficiency conditions for this choice. Conlisk
(1973), Conlisk and Watts (1979), and Morris (1979) extend earlier treatments
of optimal experimental designs to cases where the form of the response
function is unknown and both the research budget and the number of
experimental units are limited. In the absence of a specification of a
particular design problem, the three universal implications of these
conditions for response surface experimental design are rather simple and

apparent.[4/] First, the greater the sensitivity of the system being investigated to variations in exogenous parameters, the greater the desirability of additional replication. Second, the greater the number of factors thought to impinge in nontrivial ways upon system behavior, the more desirable is increased density and breadth of coverage of the economically relevant regions of the response surface. Third, since it is along these portions that outputs are sensitive to input mixes and magnitudes, research resources should be aimed at denser coverage and greater replication along the steeper parts of the economically relevant portions of the response surface, i.e., along those portions where $\partial^2 Y/\partial X_i^2$, $\partial^2 Y/\partial X_i \partial X_j$, $\partial X_i/\partial X_j$, and $\Sigma(X_i/Y)(\partial Y/\partial X_i)$ are substantial in absolute value. These parts have the greatest economic significance.

The preceding remarks with respect to the tradeoff between increased density of coverage of the response surfaces versus increased accuracy of estimation of a point on that surface apply with equal force to spatial and temporal influences. For example, those who determine the allocation of research resources into the ecosystem effects of acid precipitation will be faced with choices about whether it is preferable to study one or a very few locations in depth or to distribute limited research resources over a wide variety of locations. To the extent that the economically relevant portions of response surfaces are susceptible to spatially and temporally distributed factors, it is important to account for them. A one time period, one location experiment will provide little useful information for analysis. Some insight on how response experiments might best be located over space and time so as appraise variability is provided by Anderson (1.973).

Tn general, the essential fact of which the allocator of research resources must be aware is that there likely exist positive but declining marginal payoffs to additional observations drawn from any particular system or for any variable or particular combination of variables in that system thought to influence the response surface: that is, each additional observation adds something to the expected payoff, but these additions get progressively smaller as the number of observations increases. If the cost of research is a monotone increasing function of the number of observations, one obtains the familiar optimality condition determined by the equation of marginal costs and marginal payoffs.

Evenson and Kislev (1975) have made use of this condition to distinguish between basic and applied research. They describe the latter as involving drawings from a given probability distribution of the research payoff, while basic research shifts the first moment of the distribution or discovers new distributions from which to draw. A similar distinction might be made between acid precipitation response research which proposes to concentrate on one or a

few locations, and thereby proposes to draw observations from only a very limited number of payoff probability distributions, and response research intended to draw from a variety of distributions by spreading out its available resources over a substantial number of locations. Research bound to one location will, by definition, have to concentrate its observations around one payoff value. There is thus very little chance of discovering different payoffs because the system responses that might yield these payoffs remain unobserved. Consideration of a larger number of spatial and/or temporal settings would bring about a large increase in the sample variance, partly because more natural experiments are likely to appear and partly because a wider range of system input combinations would come under investigation. In many areas of scientific research (e.g. , plant breeding) this wider range of natural experiments and system input combinations has ultimately led to the development of techniques to affect the distributions from which the drawings are taken, and thus to allow the acquisition of information outside the range of historical experience as well as enabling the researchers to limit drawings to those response surfaces of greatest concern. In effect, the ability of decisionmakers who use research results to predict the outcomes of alternative programs is enhanced. Or, equivalently, the range of alternative programs available to the decisionmaker will be systematically narrowed as his inform- ation structure loses its ability to discriminate among different real outcomes. Unlike programs may appear to be similar in terms of their measured results and may thus be mistakenly treated as identical. Given the apparent sensitivity of the ecosystem impacts of acid precipitation to a large number of alternative combinations of biological and geochemical factors, we feel secure in adopting the position that a deaf ear should be turned to scientific counsel that urges the concentration of acid precipitation response surface research to a very limited number of locations. There appears to be insufficient understanding at present of acid precipitation response surfaces to permit the easy transfer of a surface established at one location to other locations.

Estimation of Response Surfaces: Setting aside the issue of the unthinking application of significance tests, the circumstances in which the statistical techniques available for estimating response surfaces in well-controlled experimental settings are appropriate are well understood. Apart from analysis of variance techniques, any good econometrics text such as Kmenta (1971) will provide a detailed and thorough treatment of the subtle issues of estimation that arise in a wide variety of commonly faced contexts, i~.eluding joint outputs, nonlinearities in the parameters, observations which vary cross- sectionally and temporally, systems of equations, non-normality of error terms across experiments on the same response surface, truncated dependent variables, and other matters. Econometrics appears to have little to offer biometrics with respect to useful and correct applications of these

techniques.

However, when the natural scientist uses field data rather than or along with experimental data to arrive at response surfaces, the perspective of the econometrician does have something valuable to offer. In particular, the econometrician will be sensitive to the implications for estimation of the fact that organisms make or behave "as if" they are making choices. Accurate estimation of the response surface parameters thus requires data on the factors that influence these choices. Moreover, an explicit representation of the organism's choice problem must be built into the structure to be estimated. As was argued in Chapter IV, the choice paradigm is potentially as powerful a means of explaining the behavior of monhuman organisms as it has been for human organisms. The importance of accounting for its influence even in a supposedly pure natural science exercise in estimating response surfaces is easily illustrated.

Earlier, we have indicated that if response surface research is to be most helpful to the economist, then it should be limited to what has been defined as the economically relevant portions of the surface. Identification of these relevant portions would likely be enhanced if an economist were to be included in the initial stages of research design. Research resources would be conserved. In the following illustration, inclusion in the original research design of inputs from someone who thinks like an economist is not only desirable. It is imperative if unbiased estimates of response surface parameters are to be obtained.

To make the illustration fully plausible, assume the research problem to be the estimation, through a combination of field and experimental data , of the response of trout populations to acid precipitation. [5/] In implicit form, a good approximation of the expression the natural scientist might apply to the field data collected over a given time interval is:

$$Y = f(X, W, Z, E, \varepsilon) \qquad\qquad (5)$$

where Y is the stock of trout, x is a vector of aquatic ecosystem character- istics, W is a vector of weather characteristics during the period of analysis, Z is a. measure of the fishing pressures imposed by humans upon the trout stock, E is a measure of trout stock exposures to acid precipitation, and ε is a stochastic error. The a priori information that experimental regimens have provided might be used to determine the functional form and the listing of variables on the right-hand-side of (5), to restrict the signs and/or the magnitudes of the coefficients of these variables, and/or to specify the properties of the error term. For simplicity, assume that (5) is linear in the original variables. The coefficient attached to the acid

precipitation variable is then the reduction in trout stocks due to a one unit
increase in acid precipitation. Would it then be reasonable to infer a
dose-response association from the coefficient of this variable?

The aforementioned inference would be correct if and only if it is
possible to alter the acid precipitation exposure without altering the value
of any other explanatory variable in the expression. It is easy to show that
this cannot be done unless the structure of the response surface is presumed
to consist of no more than one relationship. More than one relationship is
present in (5); it contains a variable, Z, the levels of which have been and
continue to be subject to control by fishermen. That is, during the period
over which it is thought acid precipitation effects can occur, the fisherman
can influence by his voluntary choices the fishing pressures applied to the
trout stock. For example, the reduction in trout stocks due to exposures to
acid precipitation might be dependent on the number of mature fish capable of
reproduction that fisherman have caught. In order to explain the trout stock
outcome, the researcher must do more than simply enter the amount of fishing
pressure: he must also explain the structure underlying the choice of the
degree of fishing effort applied. One element in this choice will be the size
of the trout stock. The following simple example shows one way in which trout
stocks and fishing pressures might be jointly determined.

If both the acid precipitation-trout. stock response function and the
fishing activity demand function can be linearly approximated, they can be
written as:

$$Y = \alpha_1 + \alpha_2 E + \alpha_3 X + \alpha_4 Z + \alpha_4 W + \epsilon_1 \qquad (6)$$

$$Z = \beta_1 + \beta_2 Y + \beta_3 I + \beta_4 P + \beta_5 P + \epsilon_2 \qquad (7)$$

Expression (7) states that the quantity of effort the fishermen choose to
expend is related respectively to the trout stock, fishermen income, an index
of the unit prices of substitute recreational activities, and the unit price
of fishing effort.

Solving (6) and (7) for Y, we have:

$$Y = \frac{\alpha_1 + a_4 \beta_1}{1 - a_4 \beta_2} + \frac{\alpha_2}{1 - a_4 \beta_2} E + \frac{\alpha_3}{1 - a_4 \beta_2} X + \frac{\alpha_4 \beta_3}{1 - a_4 \beta_2} I + \frac{\alpha_5}{1 - a_4 \beta_2} w + \frac{\alpha_4 \epsilon_2 \epsilon_1}{1 - \alpha_4 \beta_2} \qquad (8)$$

Consider the coefficient attached to E in (8). If E is acid precipitation,
(8) shows that an estimate of (6) will not yield the response of trout stocks

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to acid precipitation, even though, the dose-response function is "adjusted" for aquatic ecosystem characteristics, weather, and fishing effort. Instead the coefficient for E in (8) will be an amalgam of stock effects due to acid precipitation, fishing effort, and the effects of trout stocks on fishing effort. The product of the coefficients for the latter two effects would have to approach zero in order for the response of trout stocks to acid precipitation alone to be obtained. For this to occur, trout stocks could have no effect upon the amount of fishing effort and/or fishing effort could have no effect on trout stocks. Both assertions are equally implausible. In fact, in the absence of further information, the sign that would be obtained for E when (6) is estimated alone is ambiguous since $\alpha_2 < 0$, $\alpha_4 \leq 0$, and $\beta_2 \geq 0$. It is entirely conceivable, if one were to estimate (6) alone, that one would find that acid precipitation enhances trout stocks. In any case, because the product of α_4 and β_2 is negative in sign, the effect of acid precipitation on trout stocks will be underestimated. However, this negative bias in the response estimate is not predestined. Given (7), a slightly different specification of (6) could readily introduce a negative bias.

It might be reasoned that the difficulty with the preceding example could be removed if the ability of fisherman to influence trout stocks were removed. Expression (6) would not then have any human decision variables in it and would therefore seem amenable to the customary ministrations. These customary ministrations might, however, continue to be incorrect, for the trout, while acting "as if" they maximize net energy storage, are able to alter their food gathering behavior in response to a change in the competition for food. Thus the trout stock and some of the aquatic ecosystem characteristics, X, in (6) are jointly determined: the trout stock helps to determine the competition for food, and the competition for food helps to determine the trout stock.

Arguments similar to those above can readily be constructed for forests, agriculture, materials, and most items and systems thought to be impacted by acid precipitation. For example, productivity of a forest is influenced by the management practices selected by the forest owners, who are reciprocally influenced by the forest's chosen response to the selected practice. The selections of the forest owners are not based upon physical parameters alone but also on the economic factors that influence the benefits and costs of management alternatives. Similarly, the estimated response to acid precipitation of the salmonid species in an aquatic ecosystem is determined not only by the acid precipitation and the fishing pressures applied but also by the price of access for fishermen and the factors that determine the avoidance behavior of the fish.

To attempt to account for the additional factors thought to influence an

organism's response to acid precipitation by simply stringing out variables in a single expression must clearly often be incorrect. During the period in which the response is supposed to occur, organisms can behave so as to influence the magnitudes assumed by certain of these variables. Each variable susceptible to this influence must be explained by an expression of its own if the purpose of the research is to explain the response of the organism to acid precipitation rather than simply to predict its response. Unless circumstances are identical across space and time, predictions based on some version of (8) will err for reasons no one will be able to identify until the response structure is comprehended. Because some human decision variables both influence and are influenced by the response, economic analysis is frequently necessary to impart an interpretable form to response expressions. Purely biological constructs will therefore often be insufficient tools with which to establish acid precipitation response surfaces. Moreover, even when human decision variables have no role to play, the constructs of economic analysis can assist, as was argued in Chapter IV, in explaining the behavioral adjustments that organisms make to changes in acid precipitation exposures.

The above remarks need not lead to the conclusion that research on complex basic biochemical and physiological processes is required for the estimation of response surfaces. Jointly determined variables need be of interest only insofar as they contribute to understanding to the manner in which input mixes and magnitudes act upon outputs and results having economic relevance. Nevertheless, the fact of joint determination does complicate modeling and estimation procedures, occasionally beyond the ability of available analytical and estimation procedures to grasp. For this reason, there is information to be gained by establishing baseline descriptive measurements for a variety of ecosystems and locations thought to be susceptible to acid precipitation-induced effects. These effects can be economically valued even if there is no more that an association between changes in input mixes and magnitudes and changes in levels of the economically relevant outputs. The latter change can be valued whether or not the reasons for the change are comprehended. A demonstration that the economic value of the change, whatever caused it, is great can serve to stimulate research into the causes that might otherwise have been neglected. However, if acid precipitation-induced changes are to be recognized, baselines must be established against which the change can be estimated. These baseline measures must, of course, document seasonal. variances.

Although the economic value of a change in an ecosystem can be established even though there is no more than an association between outputs and inputs, it is important to recognize that the units of analysis must be defined in terms that contribute to the informed manipulation of the system. In particular the research designer must be wary of employing measures which

may be good predictors but which effectively deny the existance of certain substitution possibilities of interest to human and/or nonhuman decisionmakers. These denials are most likely to occur when the researcher aggregates or groups variables. If the aggregate is, for example, a weighted sum of a collection of inputs, there is an infinite number of combinations of the inputs consistent with anv given magnitude of the aggregate. The economically" relevant" substitution possibilities are then impossible to discover. Furthermore, if spatial or temporal comparisons are being made among ecosystems, unregistered changes in input mixes and magnitudes could readily occur. The increases and reductions in the input components could cancel each other out so that no change in the aggregate would take place. In general, therefore, researchers should be extremely reluctant to employ aggregated or grouped input variables when there exist grounds for suspecting that ecosystem components have more than one way available to adjust to the presence of acid precipitation.

Choice of Models: The comparative assessment of alternative models to explain the behavior of identical phenomena is among the most engaging activities of any discipline. The usual criteria applied in models of ecological systems appear to be an amalgam of statistical measures of goodness of fit and significance, a priori considerations relating to the biology and chemistry of the process in question, subjective judgement, and computational tractability. Generalizations about the desirable properties of ecological models, whether of the axiomatic or simulation types, relative to these criteria are very scarce. This is perhaps because model appraisals based on these criteria are bound to be misdirected.

The criteria for choosing among alternative models or theories of ecosystem behavior when stressed by acid precipitation should relate to the value of information they provide. If two models have the same costs in terms of data requirements and application, the preferred model should be that which provides the greatest expected payoff. If the models differ in their costs, this difference should also be allowed for in the payoff appraisal. In general, the important question is not whether any particular type of model is biologically or statistically better than its alternatives, but whether it can better serve the objectives of decisionmakers.

Adoption of the value of information perspective does allow some obvious generalizations to be made about the value of alternative models. 6/ The disciplinarian will usually opt for the analytical delights of ever increasing generality in the specification of the models supporting his empirical analysis. His ultimate objective would be the ability to predict the results of every alternative source of system perturbation without having to alter any of the relations expressed in his model. The generality and realism of the

ideal model would be so great that there would never be any doubt in the researcher's mind as to whether an observed change in some variable was random and thus transitory in nature or whether it was due to changes in the values of fundamental model parameters. However, the greater the progress of the researcher toward this intellectually captivating state, the greater are likely to be the number of variables for which he must make observations, collect and organize data, and establish parameter values. Furthermore, the complexity of relations among these model variables may be so great that estimating techniques are either extremely costly or perhaps even nonexistent. In effect, the elaboration and required detail of the model may be so great relative to the availability of research resources that only superficial attempts can be made to ascertain the true value of any one parameter. The problem in this case is not with a model that involves dangerous simplification of reality but with a model which, given available research resources, is alarmingly complex. The model is insufficiently artificial. Just as one fails to capture the truth when he fails to comprehend the complete structure of a system, he also fails when he is unable to measure with some fair degree of accuracy the parameters of any given comprehension of the structure.

On the other hand, the ideal of many applied scientists is to design an experiment or research effort such that the scientist does not have to think about what the results mean: the answer the experiment gives is unequivocal. Attainment of this state requires that measurement be free from bias. That is, it must be clear that the deviation of the result of any single measurement effort from the mean of the results of repeated applications of measurement effort under the least constrained conditions is purely random. The measurement errors which occur when this condition is not fulfilled can be reduced by devoting more resources to constructing measurement devices. and techniques, by allowing more time for measuremen.ts to be made, and by better training of measurement personnel. But measurement resources are expensive.

Paratt (1961, pp. 109-118) offers the following expression as a device for weighing increased detail of model elaboration against reductions in the error with which model parameters are measured. Let u be a derived property related to the directly measured properties, $x_1, \ldots, x_n$, by $U = \mu(x_1, \ldots x_n)$. For example, u might be a measure of the economic benefits of acid precipitation control. Given that the x's are not independent of each other--they might, for example, be the parameters of a model for estimating the effect of acid precipitation upon soil nutrient content, fresh water pH, and fish populations-- the error in u due to the accumulation of errors in the seperate estimates of the x's is given by:

$$\varepsilon_u = \Sigma \left(\frac{\partial f}{\partial x_i}\right)^2 \left(\frac{\partial \varepsilon}{\partial x_i}\right)^2 + \Sigma_{i \neq j} \Sigma \frac{(\partial f)}{(\partial x_i)} \frac{(\partial f)}{(\partial x_j)} \frac{(\partial \varepsilon)}{(\partial x_i)} \frac{(\partial \varepsilon)}{(\partial x_j)} \; r_{ij} \tag{9}$$

where ε_u is the error in the estimate of u and r_{ij} is the correlation between i and j. The presence of the corrleation coefficient in the above expression makes apparent at least one thing to avoid in the construction and use of complex axiomatic or simulation models in ecology (and economics): do not employ variables in the same model that are highly correlated with one another. Generally, the greater the number of attributes introduced into a model in the form of properties that must be directly measured, the more likely are some pairs of these properties to be highly correlated. Relatively simple models, by definition, require fewer directly measured properties for their solution. In addition, with repeated model applications, a low value of r_{ij} means that overestimates of the payoff are likely to be compensated by underestimates, implying that the average of the expected payoffs will be close to the true average.

Further inspection of (9) readily suggests two more bases for evaluating the tradeoff between model elaboration and errors in measurement. First, the presence of the partial derivatives, $\partial f/\partial x_i$ and $\partial f/\partial x_j$, indicates that measurement resources are more likely to be allocated efficiently if they are assigned to those directly measurable properties thought to have a really significant influence upon the derived property. Since the variables that have a significant influence upon a derived property will frequently be the same in both complex and simple models, the use of the simple model is to be preferred if avoidance of substantial error in the estimate of the derived property is of high priority.

Second, given the presence in (9) of the measurement errors associated with the directly measured properties, it pays to devote resources to reducing the larger of these measurement errors, including those interactive properties (i's and j's) whose products in (9) are greatest. Since in simple models there are fewer estimates of directly measured properties to be obtained, it follows that, to a greater extent than in a complex model, a given stock of measurement resources can be used to reduce the error associated with any one property. Thus, given the cumulative nature of measurement error in models where measured properties are tied together in long chains of reasoning, this rule along with the previous two implies that simple models can be highly advantageous in esitmating ecosystem responses to acid precipitation. The advantages exist apart from the fact that simple models are relatively easy to use and, in spite of the interesting scientific detail they may neglect, they will usually give quick answers to questions.

The preceding statements about the advantages of using simple models to describe response surfaces have not been made in the absence of empirical supporting evidence. For example, Perrin (1976), while studying the responses of various Brazilian crops to fertilizer applications, has contrasted the value to farmers of the information obtained from a simple structure based on Liebig's (1855) "law" of limiting factors $\underline{7/}$ to the information acquired from a multi-input, nonlinear (quadratic) representation commonly favored in much controlled fertilizer response research. Using a set of 28 experiments conducted at various Brazilian sites over a three year period, he compared farmers' implied ex post net revenues from the two distinct models. If soil characteristics were accounted for, the simple one input, linear model based upon Liebig performed equally as well as the nonlinear model.

Empirical. evidence similar to Perrin (1976) is now beginning to appear for the connected black box simulation models so widely favored in much applied ecological research. Stehfest (1978) has compared the payoffs from a simple Streeter-Phelps model of dissolved oxygen and a complex ecological optimal control simulation model with six state variables. Both models were built to provide information on the costs of meeting a water quality standard in a stretch of a West German river. The pavoff was defined in terms of cost minimization. The total annual costs of meeting the standard when the water treatments suggested by the simple model were implemented were 8 per cent lower than would have been the treatments recommended by the more complex model. Of course, the costs of establishing what constituted the recommended treatments were also lower for the simple model. Additional reviews of the performances relative to some objective of simple versus complex models are available in Beck (1978), Griliches (1977), and Young (1978). Outside the econometric literature [Judge, et al. (1980), Chapters 2 and 11], few, if any, implementable rules, other than those of Paratt (1961) already remarked upon, issue forth from these discussions. There is, however, general agreement that although it is naive to view simplicity per se as desirable, the research administrator should place the burden of proof that valuable information will be produced onto the proponents of proposals to build ever more complex ecological and economic models.

Whatever the virtues of model simplicity, it must be admitted that increases in model complexity are worthy attempts, in the absence of information acquisition costs, to improve model robustness, where robustness can be defined as the domain of circumstances where the model can be applied without undergoing structural revision. However, as an alternative to the devotion of more and more research resources to molding, measuring, and manipulating an ever-lengthening string of variables someone reasons or feels may influence what Young (1978) terms a "badly defined system," axiomatic methods can be used. These methods, for which an example building upon

bioenergetics is presented in Chapter IV, permit inferences to be drawn about difficult-to-measure variables by deriving relationships between them and more readily observed variables. In addition, these axiomatic methods, prior to any attempt at measurement, allow discrimination between important and trivial contributors to system behavior. Suggestions for adoption of holistic methods [e.g. Levins (1974), Jorgensen and Mejer (1979)] that recurrently appear in the biological literature are in the spirit of the axiomatic means of introducing information. More broadly yet, the bioenergetics research of Bigelow, et al., (1977), Hannon (1979), and others urges both a holistic, axiomatic approach and a movement away from near-exclusive emphases upon short-run, transient population movements in one or a few species to a concentration upon long-run equilibria for entire systems. The **bioenergetics** framework, when considered in a long-run equilibrium context, has appeal to the economist because it closely accords as a method of reasoning with his approach to the economy, a system perhaps equally as complicated as any ecosystem. In ecological contexts, the system complexity to which ecologists constantly refer is usually imcompatable with "ideal" scientific experiments that remove all responsibility for ex post thinking from the researcher. If ecosystems are equally or more complicated than are economies, the ecologist must be prepared to conceptualize a model that explains the data that is to be and has been observed or generated: he must compose a plausible story having applicability beyond the immediate circumstances being investigated.

Experimental versus Field Response Surfaces: The methods of most biological research into response surfaces impede correspondences between surfaces estimated from experimental data and those estimated from data observed in the field. Generally, responses under experimental conditions will significantly exceed in absolute value the responses to be observed under field conditions. 8/ Obviously, until studies are available and quantitative relations established between experimentally-derived and field-observed responses so that suitable adjustments can be made in both experimental designs and analyses, control decisions based soley on experiment-derived response surfaces must be less than fully satisfactory. Indeed, these experimental results might best be viewed as untested hypotheses. They allow firm generalizations to be made about input configurations not found beyond the experiment, in a set of exogenous parameters that nature never replicates. More important perhaps is the fact that the a priori information provided by a combination of experimentation and field observations will frequently make the construction of analytical models an effective means of explaining the discrepancy. The conditions of the experiment and the field observations reduce and define the domain of circumstances which the model must capture. When unexpected and/or unexplained differences exist between experimentally-derived and field-observed outcomes, some worthwhile generalizations about system behavior can usually be made by searching out the

sources of the differences.

The reasons for discrepancies between experimentally-derived and field-observed responses surfaces are probably several. Two come readily to mind. First, as Anderson and Crocker (1971, pp. 146-1.47) point out, so as to remove confounding sources of stress, all factors other than air pollution that might influence behavior in controlled experiments tend to be set at biologically optimal levels. Given that these biologically optimal levels exceed those found in everyday environments, it follows that they are less binding, implying, by the Le Chatelier principle [Silberberg (1978, pp. 293- 298)], that the contribution of an input to the behavior parameter of interest will be greater than it otherwise would be.

A second, less obvious reason arises from the role that risk plays in managed ecosystems, particularly agricultural and forest systems. In strictly controlled experimental settings, all feasible sources of random variation in output levels are excised. However, in field conditions, the system manager must adapt his activities to natural sources of random variation such as weather, insect infestations, and acidifying depositions. As Adams and Crocker (1979) and Just and Pope (1979) demonstrate, the input mixes and magnitudes the system manager selects influence both the level of output in any one time interval and the variability of these levels over time. Thus , for example, if the land area for which a farmer is responsible increases and he has no more inputs (e.g., lime, fertilizers, labor) than before, the susceptibility of his crops to any acid Precipitation events which might occur will also increase. In taking countermeasures to an acid precipitation event, he has to spread the same inputs over a greater area. The implications of this as a source of discrepancies between experimentally-derived and field-observed response surfaces become apparant with the following simple argument extracted from Adams and Crocker (1979).

Consider a risk-neutral, net revenue-maximizing farmer who must make all his input commitments before the start of any single growing season. For simplicity, further assume that acid precipitation over the growing season is expected to be either "high" (α) or "low" (β). If acid precipitation is high, the marginal cost of supplying various crop yields, given the input commitments already made, will be represented by the $(MC|\alpha)$ curve in Figure 3. This curve is the highest of the three marginal cost curves in the figure because the actual occurrence of the α level of acid precipitation will reduce the marginal products of the preselected mix of inputs, and thereby increase the marginal cost of producing any particular yield. On the other hand, if realized acid precipitation levels during the growing season were $, then, in accordance with the $(MC|\beta)$ curve, the marginal. cost of producing various yields would be reduced. The MC^{o} curve is simply the probability weighted average of $(MC|\alpha)$ and $(MC|\beta)$.

Figure 5.3
Effect of Air Pollution Risk Upon Yields
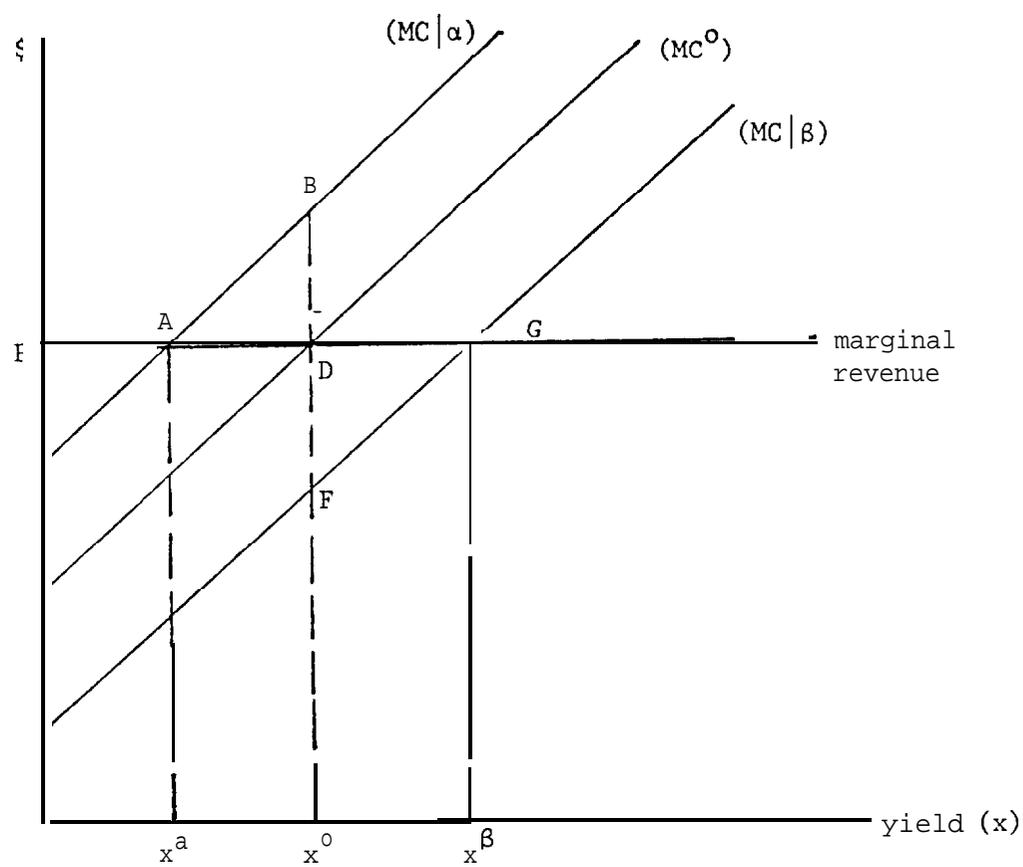
If, for simplicity, the farmer regards the occurrence of either α or β acid precipitation as equally likely, then MC^o is the marginal cost curve associated with the input mix maximizing his expected net revenues. Although this input mix will, on average, yield x", during any one season it will result in yields of either x^α or x^β. Thus if acid precipitation is high during one season, x^α will result, while if it is low, x^β will result. In effect, the variability in levels of acid precipitation causes yields in areas sometimes subjected to acid precipitation to be more variable than in areas where acid precipitation never affects yields or where it is always at a high level. Thus, for given input mixes, the odds of discrepancies between experimentally-derived response surfaces and field-observed response surfaces are greater in regions subject to fluctuating levels of acid precipitation.

If maximum acid precipitation levels have been increasing over time, then one would expect yield variability to increase in those areas where acid precipitation has been increasing. This is because the lowest level of acid precipitation (zero) cannot be altered while the highest level has increased, causing the $(MC|\alpha)$ curve to shift upward. Unless the farmer constantly lives in the darkest depths of despair about the acid precipitation problem, the MC^o curve, which is a probability weighted average of the other two curves, will never shift upward as much as the $(MC|\alpha)$ curve. The result will be increasing yield variability over time. Consequently, discrepancies between experimentally-derived response surfaces and field-observed surfaces are likely to be greater where levels of acid precipitation have historically been increasing.

A Recapitulation

Based or current knowledge, it appears that an ordered, predictable sequence of events follows the deposition of acidifying substances on ecosystems. Acid depositions cause the buffering capacities of ecosystems to decrease, the rates of decrease depending on the buffering capacity at the time of deposition. Systems with low buffering capacities will display relatively rapid decreases, whereas those with high capacities tend to have slow decreases. Also, systems with low buffering capacities generally show relatively rapid negative impacts from increasing hydrogen ion concentrations. Systems with high buffering tend to show initially positive responses from nutrients entering the system with the acidification and from nutrients mobilized by increased hydrogen ion concentrations. Over time, however, the initial positive response to acidifying depositions will reverse as nutrients leach from the system, mobilized metals reach toxic concentrations, hydrogen ion concentrations reach toxic levels, and/or nutrient cycling rates are

reduced as decomposition rates decline.

SO$_x$ and acid particles have harmful direct effects on plants. In general, when deposited on foliage surfaces, the pollutants enter the plants through the stomata. Plant seedlings and meristematic tissues are most sensitive. Therefore, acidification can cause establishment of plant species to be limited to those most tolerant of acid conditions. Over time, selection for tolerant species will simplify terrestrial communities and shift dominance.

Because of their weaker buffering systems, aquatic ecosystems tend to be more sensitive to acidifying depositions than are terrestrial systems. Within the aquatic system fish appear to be the most sensitive group of organisms and the reproductive processes appear to be the sensitive stage of the fish life cycle. Fromm (1980) ranked various reproductive processes in order of decreasing sensitivity: egg production > fry survival > fry growth > egg fertility. With declining environmental pH level, numbers of fish species are continually reduced. Available data indicates that many of the economically most valuable fish species are the most sensitive to depressed pH levels and are the first to be eliminated from the system. Continual depression of pH levels effects reductions in primary production rates, algal biomasses, and invertebrate biomasses. In addition, species diversities are reduced as the most acid tolerant species become dominant. In time, the system can reach a nearly abiotic state.

Acidifying depositions accelerate the decay rates of a wide variety of material artifacts mainly because the presence of acids upon the material surfaces increases the flow across the surfaces of the electric currents that cause corrosion, discoloration, and embrittlement. These processes are intensified for those materials, such as cement, concrete, and some metals, often used in subaqueous and/or high temperature environments.

Because of the water treatment facilities already in place, there is no substantive evidence at this time that the human health effects of acid precipitation are worrisome.

In order for the economist to be able to value the aforementioned effects of acid precipitation upon life and property, the natural scientist must provide him with information on response surfaces (see footnote 1, however). A response surface describes the magnitudes of the influences of various environmental and anthropogenic factors upon something that is valued for its own sake or for its contribution to something that is so valued. Because it emphasizes the description of substitution possibilities among the influential factors, knowledge about the response surface contributes to informed

manipulation of the system of interest. Thus any natural science exercise which fails to make explicit the mapping between the influential factors and the object of value is of no use whatsoever to the economist. A study of the effect of acid precipitation upon leaf necrosis of apple trees is worthless to the economist if the relation between leaf necrosis and apple yields is unknown.

In order for natural science research into response surfaces to be most useful to the economist, it must always have certain properties.

1) Only those portions of the surface where the marginal products of the influential factors (reductions in acid precipitation are a positive input) are positive should be studied. Knowledge about other portions of the surface is economically irrelevant.

2) Only those response surface input combinations consistent with the behavior of any organism that is the object of the research is economically relevant.

3) All economically relevant portions of the surface should be systematically sampled. Coverage of these portions should be as dense as research resources permit. Achieving this broad yet dense coverage will require that substantially fewer research resources than are traditional be devoted to replications of experiments at one or a few points on the surface.

4) Replication should be given greater consideration only when the system being investigated is thought to be extremely sensitive to variations in exogenous parameters.

5) Increased density and breadth of coverage of the economically relevant portions of the surface should be striven for whenever there is a large number of factors thought to impinge in nontrivial ways upon system behavior.

6) Research resources should be aimed at denser coverage and greater replication along the steeper parts of the economically relevant portions of the surface.

7) When the response surface is stochastic, probability distributions should be stated for the random variables that enter. The natural scientist should not leave users of his research with only his "best" estimate.

8) The above remarks apply with equal force to temporal and spatial considerations. In particular, research into the effects of acid precipitation should neither be devoted only to immediate effects nor concentrated only in a small number of locations. Ecological theory cannot often be depended upon to allow empirical findings at one site and/or time to be generalized to other sites and/or times.

Even if the above eight factors are consistently adhered to, there remain factors about which the natural science researcher must be cautioned if he wishes to produce results that are useful to the economist.

9) Jointly determined variables plausibly play a large role in ecosystem response surfaces. Thus attempts to account for the additional factors thought to influence an organism's response to acid precipitation by simply stringing out variables in a single expression will often yield biased estimates. Because some human decision variables both influence and are influenced by the response, economic analysis must often be involved in the initial research design.

10) Baseline descriptive measurements of ecosystem states may now be equally as worthy as research on response surfaces. If researchers are aware of the fact of change, even though they may be unaware of the causes of change, the change can, in principle, be assigned an economic value. Knowledge of the cause of the change is necessary only when one wishes to manipulate the system and/or assign responsibility for the change to human agents.

11) Aggregated or grouped variables to which natural science research is indifferent in terms of informational content may destroy the usefulness of the research for the economist. In general, natural science research should structure its units of analysis so that substitution possibilities are not hidden.

12) The farther is an affected component removed (in the sense of trophic linkages) from something economically valued for its own sake, the less research worthy is the component likely to be. This is because there are more likely to be available substitutes for the component.

We now move from cautionary statements about the performance of natural science (particularly ecological) research into the effects of acid precipitation to a set of aggressive statements about how this research might

be improved to the mutual benefit of the ecologist and the economist.

13) Many ecological models appear to be insufficiently artificial, perhaps because they stress the short-run dynamics of species interactions. Their builders compound errors of measurement by introducing variables that are highly correlated; they seem reluctant to make prior judgements about the significance or the triviality of a variable's influence; and they devote inordinate research resources to reductions in the measurement errors of trivial. variables. These faults are often evident in the confusing connected black box simulation models ecologists frequently use.

14) Ecologists often remark on the great complexity of ecosystems. It is not evident that ecosystems are any more complex than economies. Economists have found that an axiomatic approach which emphasizes comparative static equilibria yields great simplifications of real-world economies at no apparent cost in robustness. The long-run equilibria are used as analytical devices rather than as descriptions of reality. There is recent interest in ecology in viewing ecosystems and their components as solving a resource allocation problem [Rapport and Turner (1977)], where energy is the scarce resource. This organizing principle permits use of the tools of economic analysis as Chapter IV demonstrates. The contribution these tools can make to understanding the ecological effects of acid precipitation should be investigated further. Agricultural systems, because they are immature in ecological terms, and therefore stressed and unstable, might be a worthwhile place for initial research efforts. Note that these systems emphasize growth. It is generally thought that the most active developing tissues in plants are most sensitive to acidifying depositions.

15) Because strictly controlled experiments on response surfaces often are poor facsimiles of the real world, their results are best viewed as untested hypotheses.

Our economic approach to the effects of acid precipitation has yielded more than a set of generalizations about natural science research into response surfaces of all sorts. We have gained some insights into particular economic features of the acid precipitation problem that might be helpful in planning natural. science research into these problems.

16) The current economic value of the ecosystem effects of acid precipitation is very small compared to the value of its direct effects upon materials and perhaps upon agriculture. However, the existing studies of

the materials damages caused by pollution are technically weak in economic terms. New economic approaches to assessing materials damages must be developed before trustworthy results can be obtained.

17) Potentially, the chronic ecosystem effects of acid precipitation almost certainly dominate in economic seriousness the acute effects. Thus natural science research should give greater priority to cumulative acidity issues rather than to episodic acidic events.

18) Careful inventories of the existing stock of buffering capacities must be constructed. The frequency with which ecosystem responses to acid precipitation involve nonconvexities and irreversibilities should be identified. If, as we suspect, one or both appears with substantial frequency, natural science research should concentrate on those systems that are about to or just have exhibited the first symptoms of acidification. This, of course, presumes that good indicators of these first symptoms are available. If not, these indicators must be identified.

19) Studies of already acidified systems should be limited to attempts to establish whether natural recovery times, if any, involve less or more than two or three decades, and whether there exist any human manipulations that can slow decay rates or accelerate recovery. Because of the existence of positive discount rates, recoveries occurring more than two or three decades in the future have little value to the present generation.

20) The measurement of the changes in long-run equilibrium species assortments should be a high priority natural sciences research item because the value that humans attach to the amenities and the life support services that ecosystems provide is often conditional upon the species assortments from which they come.

21) Economists are usually unable to value dung beetles, algae, and assorted other ecosystem components because ecologists have failed to indicate how their contribution to the directly valued components of ecosystems varies with acid precipitation levels. The approach suggested in recommendation (14) might allow these contributions to be specified and thus valued.

Finally, so as to moderate our commentary about the research efforts of the natural sciences into the effects of acid precipitation, we direct a few remarks at our own discipline. We have tried to identify those sets of acid precipitation effects where one may feel resonably secure using the

conventional analysis. We have also tried to identify some possible special features of vegetative and ecosystem damages that appear to require either expansions or even complete replacements of the traditional analysis.$\underline{}^{97/}$ In Chapter IV, we have tried to extend conventional methods to include ecosystem diversity. Unfortunately, we are unable to reject the discomforting notion that the effects for which one may feel secure using the conventional methods are those having the least long-term economic significance. If this is true, it is important, for both scientific and policy reasons, to set the strengths and limits of the conventional analysis, and to design valuation methods that can be extended to phenomena where the analysis either fails or is misleading. At least insofar as the setting of limits is concerned, it is important for obvious reasons that the task not be left soley to economists. However, meaningful participation in this task by noneconcmists means that they must learn the structure and the requirements of the conventional analysis.

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1/ As noted in Chapter I, we presume in this report that Shephard's lemma (the envelope theorem) has limited practical applicability. Nevertheless, the extent to which applications of the envelope theorem might permit assessors of the economic benefits of controlling acid precipitation to avoid having to know these biological and physical influences, awaits some detailed research attention. To see why, consider the restricted profit function of Diewert (1974) and Lau (1976). Let X denote a vector of fixed outputs and inputs, where the inputs are measured as negative quantities, thus allowing both inputs and outputs to be stated in terms of net supplies. In addition, allow p to be a vector of nominal prices of the variable net supplies and let v be a vector of their rates of production or use. The variable profit is then:

$$\pi = p' \qquad\qquad i = 1,...n \qquad\qquad\text{(a)}$$

The maximum variable, or restricted, profit is:

$$IT^* = IT(p, x) \qquad\qquad\text{(b)}$$

Taking the derivatives of π^* with respect to the fixed outputs yields of the negative of the marginal cost. When these derivatives are taken with respect to the fixed inputs the negatives of the marginal valuations or demand prices are yielded. Similarly, the derivatives of π^* with respect to p yield the efficient rates of production or uses of the outputs and inputs. These results are obtained because, under appropriate conditions, every production possibility set defined with at least one fixed input or output implies a unique restricted profit function, and, conversely, every restricted profit function satisfying certain regularity conditions implies a technology. Using these results, given that nominal prices and quantities of inputs and outputs can be observed, knowledge of the exact influence of various physical and biological factors upon ecosystem variables of interest is unnecessary. However, even if these duality techniques ultimately allow economic analyses to proceed without prior knowledge of response surfaces, knowledge of thesurfaces would still prove useful as a means of checking the results obtained from applications of the duality techniques.

2/ This is not strictly true. For the statement to hold without exception even for only two inputs, it must also be true that:

$$\left(\frac{\partial^2 Y}{\partial X_1^2}\right) \left(\frac{\partial^2 Y}{\partial X_2^2}\right) - \frac{\partial^2 Y}{\partial X_1 \partial X_2} > 0.$$

3/ See Chapter III for further discussion of cancavity (nonconvexity). The discussion in that chapter is consistent with activities which operate at either A or D in Figure 2.

4/ See also Anderson and Dillon (1970).

5/ This illustration is an adaptation of a development in Crocker, et al. (1979, pp. 9-12).

6/ This and the subsequent three paragraphs draw extensively upon Crocker (1975).

7/ The "law," as succintly stated by Swanson (1963), says that yields increase at a constant rate with respect to applications of each factor unitl some other factor is limiting.

8/ Insofar as acid precipitation is concerned, nonconvexities, as was argued in Chapter 111, likely constitute an important exception to this statement.

9/ By no means is our listing exhaustive. For example, benefit-cost analysis as presently constituted, is less than robust in its treatment of the benefits and costs of alternative paths of adjustment to an environmental perturbation. Neither is it very helpful in valuing reduced uncertainty about future environmental states. Other items could be added to this listing.

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