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WORKSHOP REPORT ON QUANTIFYING ENVIRONMENTAL DAMAGE FROM ENERGY ACTIVITIES

ASPEN, COLORADO
September 11-13, 1977

Edited by
P.D. Moskowitz, M.D. Rowe, S.C. Morris, and L.D. Hamilton

OFFICE OF ENVIRONMENTAL POLICY
NATIONAL CENTER FOR ANALYSIS OF ENERGY SYSTEMS

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ABSTRACT

In September 1977, 29 natural and social scientists attended a three-day workshop to evaluate data and methods for quantifying environmental damage from energy activities. The participants held informal discussions designed to identify (1) the types and amounts of pollutants emitted by energy technologies that may affect the environment adversely, (2) methods of estimating spatial and temporal changes in air and water quality resulting from these emissions, (3) spatial and temporal distributions of ecosystems at risk, (4) dose-response functions for pollutants and ecosystems at risk, and (5) environmental and economic variables to be used to measure damage.

Emphasis was on available data and on several methods for quantitative estimation of effects of energy on the environment. This report lists damage functions that could be used to quantitate effects of ozone and sulfur oxide on agricultural crops and trees (Table 4, Section 2.2), effects of altered stream depth and velocity patterns on river fish species (Figure 4, Section 3.2.G), and sensitivities of lake chemistry (Figure 3, Section 3.2.B) and biology to acid rainfall (Table 6, Section 3.2.B). Also described are methods for estimating effects of carbon dioxide, sulfur dioxide, ozone, and several other atmospheric pollutants on selected terrestrial communities by using computer modeling techniques. With these techniques, quantitative estimates of the effects of energy on the environment could be developed within one to two years.

This report also contains brief discussions about effects of nutrient and trace metal discharges on terrestrial ecosystems and about impacts of petroleum hydrocarbon, heat, biocides, and entrainment on aquatic ecosystems. These data cannot now be used to develop quantitative impact estimates except in narrowly localized areas but they could be used to indicate thresholds above which envi-

ronmental damage could be expected. Such analyses could be made with about one year of effort. More detailed assessments would depend on advances in knowledge.

Two techniques for converting effects of emissions into common metrics are also described. (1) Estimates of directly quantifiable damage, such as reduced agricultural production are converted into direct dollar costs by using market value estimates. (2) Various societal weighting functions are used to translate damage into common metrics by a variety of approaches, e.g., estimation of pollution abatement costs as a measure of societal value, estimation of perceived valuation of impacts by means of questionnaire surveys, or calculations of inherent energy values associated with an activity. Such analyses could be completed with about one year of effort.

The workshop participants recognized that all methods of impact estimation are fraught with uncertainty, but they agreed that application of the above techniques can be useful to policy makers in directing energy programs and can ensure adequate attention to environmental, as well as to economic and social, considerations.

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1.0 INTRODUCTION

Rising concern over adverse effects of energy-related activities on the environment during the past decade has led to a large number of comprehensive studies aimed at identifying environmental costs and benefits of such activities. These studies have generally provided quantitative estimates of pollutants discharged by various energy activities and qualitative estimates of the effects of these pollutants on the environment. Government agencies and others have found these studies helpful in formulating and directing energy policy decisions, but they need quantitative estimates of the effects of energy-related activities on the environment to assure that environmental impacts receive adequate attention along with pressing economic and social considerations.

In view of this need, the Office of Environmental Policy (OEP) at Brookhaven National Laboratory (BNL), in cooperation with the U.S. Department of Energy (DOE), sponsored three-day workshop, in September 1977, to characterize and evaluate data and methods for quantifying environmental damage from energy activities. The 29 participants, from the fields of natural and social sciences held informal discussions designed specifically to identify: (1) the types and amounts of pollutants emitted by energy technologies that may affect the environment adversely, (2) methods of estimating spatial and temporal changes in air and water quality resulting from these emissions, (3) spatial and temporal distributions of ecosystems at risk, (4) dose-response functions for pollutants and ecosystems at risk, and (5) environmental and economic variables to be used to measure damage. Following introductory informal discussions on these issues, groups on commercial and natural ecosystems (Table 1) were formed to focus participant efforts. After the group discussions, an individual report was prepared by each participant, and these were combined to produce this volume.

Table 1

Workshop Groups

Leonard D. Hamilton
Chairman

Commercial Ecosystems

Walter W. Heck - Chairman
Michael D. Rowe - Rapporteur
Robert J. Anderson
Harris M. Benedict
Thomas D. Crocker
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Carl L. Schofield
Richard D. Urban

The brief duration of the workshop precluded an exhaustive review of the literature; instead, the expertise of the participants was relied upon to summarize the state of the art. When barriers to quantitating environmental impacts were identified by participants, summaries of pertinent data were presented with the hope that they could be integrated into future assessment efforts.

1.1 Background

Before the workshop, a working paper was sent to each participant outlining existing capabilities for damage quantification at BNL and highlighting concerns that would need to be addressed. The working paper, summarized below, provided a framework for workshop discussions for the quantification of impacts.

1.2 Identification of Pollutants

The Environmental Protection Agency, as required by Section 109 of the Clean Air Act of 1970 (U.S. Congress 1970), defined criteria air pollutants to include photochemical oxidants, sulfur dioxide (SO_2), nitrogen oxides (NO_x), hydrocarbons (HC), particulates (TSP), and carbon monoxide (CO) (U.S. EPA 1971). Similarly, EPA established water quality criteria and standards for many organic and inorganic compounds as required by various sections of the Federal Water Pollution Control Act Amendments (U.S. Congress 1972). Many pollutants identified by EPA documents arise from energy activities.

At BNL, the discharge by various energy technologies of selected pollutants (including many EPA air and water criteria pollutants) can be quantified by using the following computer models: Brookhaven Energy System Optimization Model (BESOM) (Cherniavsky 1974); Brookhaven Energy System Network Simulator (ESNS) (Sevian 1975); Regional Energy System Optimization Model (RESOM) (Goettle et al. 1977); and the EPA Regional Emission Projection System (REPS) (Booz-Allen & Hamilton 1975). BESOM, RESOM, and ESNS estimate emissions from fuel and end-use cycles for entire energy scenarios (Table 2). BESOM and ESNS produce quantitative estimates of air and water emissions from alternative national energy scenarios. RESOM estimates air and water emissions for the nine census regions (Figure 1). REPS is a computerized model for projecting air pollution emissions at the level of air quality control regions (AQCR) (Figure 2). It combines exogenous national and regional economic forecasts with point and area source emission inventories to project annual emissions of the five criteria pollutants for each AQCR.

Table 2	
Some BESOM, RESOM, and ESNS Outputs	
<u>Water Pollutants</u>	<u>Radioactive Effluents</u>
Bases	Solids
Nitrates	Krypton-85
Other dissolved solids	Tritium
Nondegradable organics	Liquids
Biochemical Oxygen Demand	
Chemical Oxygen Demand	
<u>Air Pollutants</u>	<u>Heat</u>
Carbon dioxide (CO ₂)	
Carbon monoxide (CO)	<u>Solid Waste</u>
Nitrogen oxides (NO _x)	
Sulfur dioxide (SO ₂)	<u>Land Use</u>
Particulates (TSP)	
Hydrocarbons (HC)	
Aldehydes (Ald)	

1.3 Spatial and Temporal Changes in Air and Water Quality

Effects of some kinds of emissions on air quality can be estimated at the local level by using simple wind-rose dispersion models (Muschett 1977) or at the regional level by using long-range transport models (Meyers et al. 1978). These models generally trace wind trajectories and calculate dispersion, deposition, and transformation of a number of atmospheric pollutants (including SO₂, TSP, and sulfates) to estimate local and regional changes in air quality. The BNL transport model has also been used to estimate spatial and temporal alterations in ozone (O₃) and oxidant concentrations (Wolf et al. 1977).

Regional alterations in water quality from energy activities are more difficult to predict because of limitations imposed by variations in circulatory patterns, land surfaces and groundwater flows. While many models exist to estimate

Figure 1. U.S. map showing the RESOM census regions.

Figure 2. U.S. air quality control regions.

local or semi-regional water quality impacts (Hinson 1977), the site specificity of these models and their detailed data requirements limit their usefulness in most large scale policy analyses. Studies of the effects of consumptive water withdrawal impacts on river flows are less complex and have been completed by Samuels (1976) and others.

1.4 Species and Communities at Risk

Assessing damage to human health, which involves a single species, is difficult enough, but assessing damage to plants and animals is considerably more complex. Data are sparse on the present temporal and spatial distributions of most species except for agricultural crops (U.S. Bureau of the Census 1974) and forests (Little 1976; U.S. Dept. of Agriculture 1974). Since species and communities sensitivities to pollutants vary considerably indicator communities or species that are ubiquitously distributed and studied in sufficient detail to identify their response to pollutants may prove to be most valuable in impact assessments.

1.5 Dose-Response Relationships

Dose is a function of pollutant concentration and exposure duration; response is the reaction of the community or species at risk to a given dose. Chronic effects may include, but are not limited to, carcinogenicity, teratogenicity, mutagenicity, and changes in productivity, behavior, chemical structure, species composition at the community level, and disease susceptibility. Chronic effects due to low concentrations over extended times are difficult to identify and measure in plants and animals. Historically, chronic air pollution effects on plants have generally been measured in terms of lowered productivity.

Acute effects due to higher concentrations over shorter times are generally caused by direct toxicity; they often result in visible damage to individual species or destruction of parts of the community at risk. Severe, acute effects have been the principal means for identifying environmental damage caused by air and water pollutants, but recently considerable interest has arisen in identifying damage over wider intervals of time and space.

1.6 Environmental Variables to Measure Impacts

Several attempts have been made to predict and quantify responses of systems at risk to changes in air and water quality due to energy production and use at local, regional, or national levels. Most were concerned with damage to agricultural crops and forests by air pollution (Benedict et al. 1973; Waddell 1974; Taylor and Miller 1973). The approaches used ranged from post-event surveys of crop or forest damage to computer modeling of effects on forest community structure. The work on aquatic systems has included estimation of effects on water quality and quantity (Council on Environmental Quality 1976), post-event surveys of fish kills (U.S. EPA 1974b; Sharma et al. 1977), and computer modeling of selected ecosystems (U.S. AEC 1972b).

1.7 Economic Variables for Estimating Environmental Impacts

Estimating economic costs of effects of pollution on the environment may take at least three forms. (1) The most obvious is estimation of costs of altered productivity of economically valuable plants and animals. This has clear economic significance, but it captures only a small portion of total environmental impact because it ignores plants and animals that do not have direct economic significance. It also ignores important, but difficult to estimate, functions of ecosystems such as maintaining clean air, pure water, a green earth, and a balance of organisms. (2) Estimating the cost of not creating envi-

ronmental impacts, that is, the cost of pollution control, has no relationship to the problem of actual environmental damage but indicates the size of the solution. (3) Estimation of the cost of providing in some alternative way the "services" or functions of damaged systems (e.g., the cost of nitrate fertilization as a surrogate for decreased nitrogen fixation by microorganisms) suffers from the same limitations as the first method, because it is clearly impossible to determine all the losses due to alterations of environmental functions.

These methods, moreover, apply to a small set of all environmental damages. Many types of damage, such as elimination of endangered species, have not been analyzed to determine the human (social) consequences. In such cases, policy analysts have assumed that, if ecological damage must be estimated, weighted values should be established and incorporated in traditional cost analyses. It may well be, however, that a large set of ecological damages should simply be identified in analyses with no attempt to assign economic costs to them.

2.0 COMMERCIAL ECOSYSTEMS

2.1 Introduction

Many atmospheric pollutants from energy sources are known to affect some agricultural crops adversely, but only O_3 and SO_2 are sufficiently important and well understood (Heck and Brandt 1977; Heck et al. 1977) to allow dose-response estimation for many commercial crops, including cultivated trees (see Table 3).

Exposure of sensitive vegetation to SO_2 and O_3 can visibly injure leaves and reduce growth, yield, or quality and in extreme cases can increase mortality (Heck and Brandt 1977). Visible injury to leaves can be used for estimation of reduction in yield or quality of plants whose leaves are the marketable product such as tobacco, alfalfa, leafy vegetables, and ornamentals. For plants whose

fruit, seeds, stems, or roots are the marketable product, the relationship between leaf injury and value reduction is less well understood and documented. Generally, for most crops or timber species, >5% of leaf area must be affected for growth and/or yield to be reduced (op. cit.).

Table 3
Level of Knowledge of Impacts on Agricultural Ecosystems
From Energy-Related Pollutants

(Adequacy of knowledge: 0 = none, to 5 = complete)

Level of knowledge			
Time	Known to be of National Importance	Known to be of Regional or Local Importance	Known to be important but significance not known
Now	O ₃ (3)	Hydrogen Fluoride (4)	Acid rain (2)
	SO ₂ (3)	Peroxyacetylnitrate (3)	Cl ⁻ (1)
			CO ₂ (1)
			Ethylene (2)
			NO _x (2)
			Radioactivity (4)
			SO ₄ (1)
			Trace Metals (1)
<10 years	O ₃ (4)	Hydrogen Fluoride (>4)	Acid rain (3)
	SO ₂ (4)	Peroxyacetylnitrate (4)	Cl ⁻ (3)
			CO ₂ (3)
			Ethylene (3)
			NO _x (3)
			Radioactivity (4)
			SO ₄ ²⁻ (3)
			Trace Metals (2)
>10 years	O ₃ (>4)	Hydrogen Fluoride (>4)	Acid rain (4)
	SO ₂ (>4)	Peroxyacetylnitrate (4)	Cl ⁻ (3)
			CO ₂ (4)
			Ethylene (3)
			NO _x (3)
			Radioactivity (4)
			SO ₄ ²⁻ (4)
			Trace Metals (3)

2.2 Dose-Response Functions

Dose-response functions for selected agricultural crops sensitive to SO_2 and O_3 are shown in Table 4. These estimates were made by workshop participants on the basis of the following assumptions.

(1) Effects of long-term exposures to O_3 or SO_2 are the cumulative result of acute effects of short-term exposures.

(2) Times of 1 or 3 h (SO_2) and 1 or 8 h (O_3) are realistic average durations of episodic exposures.

(3) The concentrations listed represent thresholds for 10 to 20% reductions in yield. Lower reductions that are statistically significant are difficult to detect in the field.

(4) These concentrations may occur once to several times over a growing season. The frequency and intensity of lesser exposures can be judged from the concentrations specified.

(5) Plants are growing in the stage of maximum sensitivity and are not subject to other significant environmental stresses.

(6) Interactive effects of combinations of pollutants are neither synergistic nor antagonistic on a regional or national scale. This assumption, although possibly incorrect for individual crops and pollutant combinations, seems necessary because research has shown that there is no uniformly applicable principle.

(7) Many factors (such as interactions with other stresses due to weather or pathogens), although possibly important, were not considered because of the lack of quantitative dose-response information about them.

(8) Values are given across cultivars (varieties) within a species, and in some cases across plant types.

(9) Usable plant responses are (a) reduction in yield or productivity or (b) a response that can be converted, however imprecisely, into such a reduction, e.g., effects on photosynthesis, plant growth, or reproduction or a reduction in functional leaf area.

(10) Best judgement and intuition are used to define thresholds for yield reduction because information of the necessary scope, quality, and depth is rarely available.

Table 4					
Estimates of Sulfur Dioxide and Ozone Doses Producing 10 to 20% Yield Reduction in Selected Crops					
(Concentrations given are maxima occurring once or twice a year) (see pages 11 - 12)					
	Max O ₃ conc. (parts per hundred million)		Max SO ₂ conc. (parts per million)		Sensitive days per planting
	1 h	8 h	1 h	3 h	
Potatoes	15	8	1.5	0.5-1.0	42
Alfalfa*	15	8	1.5	0.4-1.0	All
Grains	15	8	1.0	0.5	?
Tomatoes	15	8	1.5	0.5-1.0	All
Leafy veg.	15	8	---	---	All
Soybeans	20	10	1.0	0.5	28-42
Cotton	20	10	1.0	0.5	56
Snap Beans	20	10	1.0	0.5	21
Tobacco	20	10	1.5	1.0	All
Grape	20	10	?	?	All
Corn	20	10	Very resistant		?
Ornamentals*	20	10	1.5	0.8	All
Ponderosa pine	20	10	?	?	All
Southern pine	25	15	1.0	0.5	All
Sugar beet	25	15	?	?	42
Citrus**	35	15	?	?	?

*Leaf damage 5%.

**Very resistant.

2.3 Yield Reduction Estimates

The thresholds shown in Table 4 represent a consensus of the workshop participants about extreme conditions, but they are not very useful for predicting environmental impacts because emission--air quality models seldom generate data of this kind, more commonly estimating average conditions. Described below is a method for converting the values shown in Table 4 into continuous yield reduction estimates as functions of mean exposure.

The probability, α , that a particular concentration measurement for a given mean, variance, and averaging time will exceed the threshold, T , is given by

$$\alpha = \int_T^{\infty} f(x|\bar{x},\sigma)dx$$

where $f(x)$ is the cumulative distribution function of the pollutant in time and space. The probability that the threshold value will be exceeded at least once in a given length of time is

$$P(x>T) = 1 - (1 - \alpha)^t$$

where t is the number of averaging periods in the length of time considered. The expected value of the percent reduction in yield produced by pollution is then the product of the yield reduction if the threshold is exceeded (i.e., 10 to 20% reduction, midpoint = 15%) and the probability that it will be exceeded, $P(x>T)$. It can be assumed that atmosphere pollutants are lognormally distributed (Larson 1971). Values of α can then be obtained from standard tables of statistical distributions (Rohlf and Sokal 1969).

In this method two assumptions are made regarding time: (1) that the mean value of the distribution is constant in time, and (2) that pollution concentrations from one averaging time to the next are independent. Neither assumption is valid; both would be unreasonable under some conditions of weather and pollution source characteristics. For crops that are sensitive to pollutants through a long growing period, however, they are probably sufficiently valid to keep the resulting uncertainty of the estimate within the uncertainty range of the underlying relationship between threshold and yield reduction. For crops that are sensitive to pollutants for a relatively short time, the influence of seasonally abnormal conditions on the structure of the pollution concentration distribution should be examined carefully.

If pollution systems are not independent, the above method of calculating yield reduction as a function of mean pollution concentration and variance systematically underestimates actual yield reduction. This happens not only because lack of independence means that a single "incident" increases the probability of another, but also because a number of "incidents" at close intervals may have more than an additive effect on yield reduction.

2.4 Damage Index

Crude approximations of impacts on agricultural crops can be made by means of a "plant impact surrogate," developed at the workshop, in which national and regional emission projections are used to estimate air pollution levels. In the surrogate it is assumed that about 90% of all crop damage from air pollution is attributable to O_3 , and of the remaining 10%, most is attributable to SO_2 ; therefore, in estimating potential crop damage at the national level, O_3 and SO_2 production should be weighted in the ratio 9:1. Although no simple way of estimating the total O_3 produced by HC and NO_x emissions is known, the ratio of

changes in O_3 production over long periods of time can be approximated as follows

$$O_{3(T=1)} = \frac{RH_{(T=1)} \cdot NO_{x(T=1)} \cdot OH^-_{(T=1)}}{RH_{(T=2)} \cdot NO_{x(T=2)} \cdot OH^-_{(T=2)}}$$

where RH means reactive hydrocarbons and OH^- arises from ionization by sunlight. By assuming that the ratios of RH to total HC, and of nitric oxide (NO) to NO_x , remain constant, and that over the long term the amount of sunlight produced OH^- remains constant, the ratio of ozone productions at different times can be expressed as the ratio of hydrocarbon concentration times NO_x concentration at different times. For nationwide estimates, the ozone ratio can be based on emissions of the latter pollutants.

Combining estimated HC and NO_x production and total SO_2 emissions yields an index, I of potential crop damage:

$$I = 0.9 \frac{HC_{(T=1)}}{HC_{(T=2)}} \cdot \frac{NO_{x(T=1)}}{NO_{x(T=2)}} + 0.1 \frac{SO_{2(T=1)}}{SO_{2(T=2)}}$$

where HC, NO_x , and SO_2 are estimates of total amounts of emissions, nationwide, at two different times. This index in no way represents actual crop damage; it expresses an approximate relationship between potentials for pollution-induced crop damage at different times. A factor for converting the index to actual crop damage might be found by calculating the value of the index for a year in which total crop damage was measured, but, so far, measurements of total nationwide crop damage due to pollution are so inconsistent that they are of limited use in providing a base with which to compare future estimates.

3.0 NATURAL ECOSYSTEMS

3.1 Terrestrial

A. Introduction

Energy-related effects on terrestrial ecosystems selected by workshop participants as being sufficiently important to require quantitative estimation of their impacts included CO₂, SO₂, trace elements, acid rain, oxidants, nutrients, and direct damage due to land stripping. These impacts differ from those on commercial ecosystems principally because of differences in time scales of effects, in management practices, and in available data. Described below for each category are the major expected effects at the ecosystem level and the dose-response functions and models available for impact quantification.

For each category, the workshop participants generally agreed that, to quantitate impacts, dose-response functions would have to be coupled with various ecosystem models. In particular, they thought that the U.S. International Biological Programs (IBP) biome study models for grasslands (Innis 1978; Parton and Ellis 1978), forest (Sollins et al. 1973; O'Neill et al. 1972), deserts (Goodall 1973), and tundra (Miller and Tiezen 1972), the Botkin forest ecosystem model (Botkin and Miller 1974), and the University of California forest succession model (Miller et al. 1977) could be adapted for use in such quantitation. These models generally simulate growth on successional trends in complete ecosystems and could be modified to accommodate stresses imposed by the pollutants selected. In cases for which appropriate data or models for impact quantification do not now exist, workshop participants presented data regarding effects on indicator organisms that can be expected as pollutant levels increase. Such data do not permit quantification at this time.

B. Carbon Dioxide

Chlorophyll-containing plants fix CO_2 from the air to produce primary sugars. For many plants, fixation of CO_2 increases as ambient CO_2 level increases. Thus, increases in ambient CO_2 levels tend to increase the rate of photosynthesis and therefore the rate of biomass production. The extent of this effect depends strongly on ecosystem type, availability of nutrients and other limiting variables. (Some studies suggest that increased atmospheric CO_2 levels can be harmful to plant systems, but this issue requires further research.) Dose-response functions relating CO_2 levels with photosynthetic rate are available for crop species (Rosenberg 1974) but not for noncrop species. Botkin et al. (1973) modeled the effect of increased CO_2 levels on forest systems by bypassing the dose-response function and assuming linear increases in photosynthetic rates. Hypothetical dose-response functions could be generated for indicator terrestrial plant species on the basis of theory and crop species data to give preliminary results, and appropriate experimental work could provide dose-response functions within a few years.

Although some specific information could be readily incorporated into existing computer models, analysis of other direct effects of increasing CO_2 levels, such as interaction with effects of other pollutants, may be a larger task. For example, increased CO_2 concentrations may inhibit the effect of O_3 exposure (Heck and Dunning 1967). The major influence of CO_2 increases is apt to be an indirect one via global climatic change. At present, no method is known for predicting quantitatively how ecosystems will adapt to increasing CO_2 accompanied by climatic change (U.S. Dept of Energy 1977).

C. Sulfur Dioxide

It has been recognized for over a century that sulfur oxides affect plants (Stoeckhhandt 1871). Since the first report, many other studies have documented sulfur oxides effects (U.S. NAPCA 1969; U.S. EPA 1973). The effects of sulfur oxides on natural ecosystems are similar to those described above for agricultural crops, but the data needed to quantitate them are more limited. The dose-response data in Table 4 can be applied to the vegetation in natural ecosystems and coupled with various ecosystem models, such as the Shugart and West (1977) and Innis (1978) models of forest and grassland ecosystems, to estimate SO₂ impacts. Minimum SO₂ levels at which damage to plant system can be expected are available from studies on lichens (Rao and LeBlanc 1967; Skye 1968; LeBlanc and DeSloover 1970).

D. Trace Metals

The possible ecological impacts of trace elements emissions from coal combustion have been reviewed by Berry and Wallace (1974), Vaughan et al. (1975), Van Hook and Shults (1976), and Braunstein et al. (1976). Although data are available concerning trace element accumulation and toxicology for a large variety of plants and animals, they provide little basis for quantitative conversion of trace element concentrations to organism or system effects. Estimates of ambient trace element concentration in terrestrial ecosystems, calculated from emission-dispersion models, can be compared with toxicological data to estimate potential threshold concentrations at which biological effects could be expected. Vaughan et al. (1975) have made such analyses.

E. Acid Precipitation

Acidity of precipitation in the northeastern United States has increased in the last several decades. In general, the increase has been attrib-

uted to sulfuric and nitric acids produced by oxidation and hydration of sulfur and nitrogen compounds from anthropogenic sources (Likens et al. 1972; Brosset 1973).

Likens et al. (1977) have suggested that acidic precipitation and/or drought are responsible for reduction in tree growth of the Hubbard Brook Forest of New Hampshire. Experiments on effects of acid rains on forests conducted by Scandanavian investigators, however, have not confirmed significant reductions in forest productivity (Johnson and Sundberg 1972). Recent results reported by Botkin and Aber (1977), obtained by computer modeling of a mixed forest ecosystem, suggest that acid rainfall affects species composition. Experiments with differential leaching of nutrients from plant foliage after exposure to simulated acidic precipitation and acidic particulates suggest that long-term alterations in overstory and understory plant succession may result from differential nutrient loading and leaching by acidic precipitation (Wood and Borman 1975; Fairfax and Lepp 1975; Hindawi et al. 1977). Alterations in nutrient storage capabilities among organisms are also suggested (Woodwell and Whittaker 1968).

F. Oxidants

Specific effects of oxidant exposure on natural ecosystems include changes in primary productivity, reproductive capability, host-parasite relationships, detritus degradation rates, nutrient storage capability, succession patterns or rates, and growth, reproduction and death rates of animals. The changes can be relatively rapid (e.g., accelerated mortality of overstory pine trees in a mixed conifer forest) and, depending on the site and pollutant stress, may be irreversible or slowly reversible. Table 5 indicates the current

level of knowledge regarding the above seven significant effects of oxidant stress on several ecosystems.

Table 5					
Level of Knowledge of Effects of Oxidant Stress on Terrestrial Ecosystems					
(Adequacy of knowledge: 0 = none, to 5 = complete)					
	Forest				
	Grasslands	Conifer	Deciduous	Chaparral	Desert
Primary productivity	2	3	2	1	1
Reproductive capacity	2	2	0	1	1
Host-parasite relationships	0	3	0	0	0
Detrital degradation	0	2	0	0	0
Nutrient storage	0	2	3	0	0
Succession patterns	0	3	3	1	1
Animal morbidity	0	1	0	0	0

Some preliminary estimates of the extent of these physical changes as a function of O_3 dose are possible. With respect to grasslands, Treshow and Stewart (1973) have suggested a dose threshold for foliar injury to a number of species comprising a grass-oak woodland, and Price (1973) have provided data on the effect of O_3 on growth and reproduction of selected grass species. An existing model of grassland productivity (Cole 1976) may be modified to make further estimates of changes induced by O_3 .

Western mixed conifer forests dominated by ponderosa and jeffrey pines have received the most attention to date. Miller et al. (1977) give a preliminary estimate of the relationship between O_3 dose and foliar injury for the entire growing season. This is currently being modified to include the influence of environmental factors of needle tissue sensitivity to ozone.

Concurrently, a forest succession model is being prepared at the University of California to simulate long-term effects of chronic O_3 exposure. The foliar injury and forest succession models for this type of forest may be available in preliminary form within 8 to 12 months and in a further validated form within 20 to 24 months (Miller et al. 1977). The applicability of these models to other forest ecosystems is not yet known, but the dose-response of single species such as eastern white pine may be estimated by adapting the ponderosa-jeffrey pine model.

G. Nutrient Loading

Nitrogen and sulfur oxide deposition from the atmosphere over many years can add significant nutrients to terrestrial ecosystems. Depending on geographic location, roughly half the inputs are dry deposited and half wet deposited. Little work is available from which to draw conclusions on specific effects on ecosystem structure and function, but possible long-term responses may be similar to the eutrophication of lakes in which increased successional rates and changes in species might be facilitated by the removal of nutrient constraints on primary productivity. The specific impact is likely to be affected by the extent to which the system is nutrient deficient and the degree to which other factors exert more significant controls on system functioning. Elevated concentrations of sulfur in plant materials may also affect the quality of forage available to animals grazing there (Schwartz et al. 1978). Certainly no simple dose-response relationships can be formulated. Effects lie embedded within a dynamic system of interacting components. Ecosystems models that include nutrient cycling submodels are the most useful methodology available for examining such effects. Some work has begun in the Hubbard Brook study on examining possible influences on nitrogen and sulfur cycling by empirical input-output models

(Likens et al. 1977). The U.S. IBP grassland biome study models are more mechanistic, featuring nitrogen cycling submodels ideally suited for such application (Cole 1976).

Recently, a sulfur cycling submodel was constructed for use with an improved version of the IBP grasslands model to assess possible effects of SO_2 dry deposition on a northern greatplain ecosystem (Coughenour 1978). The model was validated by using data collected within the EPA zonal air pollution system study on a mixed prairie in southeastern Montana (Dodd et al. 1978). These data include sulfur contents of various plant parts and soil. Similar data may be collected for nitrogen and sulfur in other ecosystems to give some indication of enrichment, but baseline or control data must be available for comparison.

H. Direct Damage

Strip-mining provides the most dramatic example of direct damage to ecosystems: It removes natural vegetation and destroys wildlife habitats. Its impact may be reduced in the future since federal regulation now requires reclamation of strip-mined lands. The procedure usually includes contouring the overburden, redistribution of the original surface soil, establishment of a cover crop to reduce erosion, and reseedling with plant species similar to those found prior to the disturbance. Long-term effects depend on the success of this procedure. Important physical changes include aquifer disturbance and alteration of topography, stream locations, soil structure, and water quality. These changes can have a large impact on the biological community.

Although descriptions of the impact of strip-mining on natural ecosystems have been published by the U.S. Fish and Wildlife Service, quantitative information about the impact of strip-mining on natural ecosystems is lacking. Grassland and forest ecosystem models could be useful in quantifying some of the

impacts. The U.S. IBP grassland ecosystem model (Parton and Ellis 1978) can be used to simulate impacts of various reclamation policies on primary and secondary production and nutrient cycling in grassland systems. The IBP forest models should be useful in demonstrating impacts of soil structure and nutrient levels on primary productivity in forest systems. Work on expanding the application of these models, coupled with continued experimental work, could provide an adequate basis for quantitative assessment of ecosystem level impacts of strip-mining plus reclamation within 1 to 3 years.

3.2 Aquatic

A. Introduction

Water-borne pollutants or stresses on aquatic ecosystems selected by workshop participants for review included acidification, petroleum hydrocarbons, heat, biocides, entrainment, and consumptive withdrawal. Since the participants generally agreed no data or models exist for converting emissions into quantitated biologic impacts, only critical pollutant levels and past research efforts are presented. The participants thought that critical pollutant levels could be coupled with local and regional water quality models to estimate effects on water quality and to highlight threshold levels at which effects on aquatic organisms could be expected. Other materials have been presented with the hope that they may be incorporated into future assessment efforts.

B. Acidification

A major impact of energy production has been acidification of precipitation by increased emission of SO_2 and NO_x which are converted in the atmosphere to H_2SO_4 and HNO_3 . Acid precipitation has caused the acidification of thousands of lakes and several major rivers in Scandinavia (Braekke 1976) and hundreds of lakes in the Adirondack State Park in New York (Schofield 1976b). Severe damage

to aquatic ecosystems has resulted. The rate of acidification of freshwater ecosystems is dependent on two parameters, deposition rate of acids and buffering capacity of the receiving area.

1. Deposition

Two processes transfer material from the atmosphere to aquatic ecosystems; precipitation (rain and snow) and dry deposition (particulate fallout impaction of aerosols and gaseous absorption). To determine the rate of acid deposition to aquatic ecosystems, the amount and the composition of the deposited material must be known. Extensive data on the amount of precipitation in the United States are available from the National Weather Service, but quantitative data on the rate of dry deposition are essentially unavailable. It is unlikely that anything other than qualitative estimates on dry deposition rate will be available in the next year.

Determinations of major inorganic components of precipitation (H^+ , NH_4^+ , Ca^{2+} , Mg^{2+} , Na^+ , K^+ , SO_4^{2-} , NO_3^- , Cl^-) are currently being performed at more than 50 laboratories (federal, private, and academic) (Miller and Wisniewski 1976). A serious deficiency in this data base is that most of the studies are not coordinated; and no national data bank for precipitation composition exists except for the precipitation chemistry section of the DOE MAP 3S program (MacCracken 1978). However, a national precipitation chemistry network sponsored by the USDA Agricultural Experiment Stations is scheduled to begin operation in early 1978 (Galloway and Cowling 1977). Within two years, precipitation composition data on a national level will be available.

2. Buffering Capacity

Once the acid-deposition rate is known, sensitivity of the receiving areas to acidification must be determined at two levels: (1) regions

within the North American Continent and (2) individual watersheds within the regions.

Regions of North America containing lakes sensitive to acidification have been identified (Figure 3). The primary criteria used in determining regional sensitivity are lake alkalinity and bedrock geology (Galloway and Cowling 1977). The identification of individual watersheds sensitive to a given input of acid is dependent on bedrock geology, altitude, soil type, vegetation type, and watershed and lake morphology. The specific interrelationship of these variables in determining sensitivity to acidification is not known. Alkalinity of a lake, however, is the master variable; the lower the alkalinity, the more sensitive a lake is to acidification. Therefore, given estimates of acid inputs and lake alkalinities, a first order approximation of which lakes will acidify can be made.

Hydrogen ion concentration is the primary variable used to assess water quality effects in freshwater ecosystems exposed to acid precipitation. The change in pH of waters exposed to any given level of acid input is determined by the alkalinity of the receiving water body. Alkalinity may be conceptually defined in terms of the equivalent balance of major base-forming cations and acid-forming anions:

(1) alkalinity (equivalents per liter) =

$$2(\text{Ca}^{2+}) + 2(\text{Mg}^{2+}) + (\text{Na}^{+}) + (\text{K}^{+}) - 2(\text{SO}_4^{2-}) - (\text{NO}_3^{-}) - (\text{Cl}^{-}).$$

The base-forming cations in the positive portion of equation (1) are derived primarily from soil and rock weathering in the watershed, whereas the acid-forming anions in the negative portion come primarily from strong acids in precipitation

Figure 3. Regions in North America containing lakes that are sensitive to acidification by acid precipitation.

(except in regions with significant sulfur mineralization). Alkalinity of water reaching a stream or a lake is thus determined by the degree of exchange of hydrogen ion in precipitation for metallic cations from the soils.

Empirical relationships between strong acid anion concentrations (or loads) in precipitation and lake-stream concentrations (loads) have been developed for watersheds in eastern North America and in Scandinavia (Likens et al. 1969; Schindler et al. 1976; Wright and Henriksen 1978), but at this time base-forming cation levels resulting from weathering and H^+ exchange in various geologic provinces cannot be precisely quantified. Hydrogen ion activity may, however, be derived from the following relationship

(2) alkalinity =

$$[HCO_3^-] + 2[CO_3^{2-}] + [OH^-] - [H^+] =$$

$$\frac{K_1 K_H pCO_2}{[H^+]} + 2 \frac{K_1 K_2 K_H pCO_2}{[H^+]^2} + \frac{K_W}{[H^+]} - [H^+]$$

This expression of alkalinity is equivalent to that presented in equation (1) and enables calculation of pH change for given levels of strong acid input, temperature, ionic strength, and carbon dioxide partial pressure. As a first approximation to the changes in pH that might result from given loadings of strong acid, this equation could be used in conjunction with available surface water alkalinity data and existing hydrologic models to project water impacts (Schofield 1976b).

Acidification of lake ecosystems appears to affect aquatic organisms both through solubilization of potentially toxic substances (e.g., trace

metals including Al, Cd, Cr, Cu, Hg, Pb, Zn) and through changes in pH (Wright and Snekvik 1978; Schofield et al. 1978). As an example of the role of lake acidification in metal dynamics, studies in eastern Canada (Brouzes et al. 1977), Sweden (Landner and Larsson 1972), and the Adirondack region of New York (Schofield et al. 1978) have revealed levels of methyl mercury in fish from acidified regions that are higher than the Food and Drug Administration limit of 0.5 ppm. On the basis of their chemical properties, it can be assumed that other metals also increase in concentration with decreasing pH of lake water. Except for the work by Schofield on aluminum in acid lakes (Schofield et al. 1978), effects of increases in metal concentration on the biota are not known.

Table 6 lists general impacts associated with lake acidification based on field observations (Almer et al. 1974; Grahn et al. 1974; Hendrey and Wright 1976; Leivestad et al. 1976; Schofield 1976a; ØKland 1969; Sutcliffe and Carrick 1973). It is important to note that these impacts do not indicate the quantitative nature of the impact, nor the degree of interference on interactions among organisms in aquatic ecosystems.

Effects that cannot now be quantified but are nevertheless of concern include impacts on bacterial and algae productivity and biomass, and the replacement of bacteria by fungi as the dominant saprotrophs. As a consequence of these changes, mineralization is greatly retarded and organic debris accumulates (Hendrey and Wright 1976; Hendrey et al. 1976).

Table 6

Impacts on Biota Associated With Lake Acidification

Long-term changes of <0.5 pH unit in the range 8.0 to 6.0 may alter the biotic composition of freshwaters to some degree, but the significance of such slight changes is not great.

A decrease of 0.5 to 1.0 pH unit in the range 8.0 to 6.0 may cause detectable alterations in community composition: productivity of competing organisms will vary, and some species will be eliminated.

Decreasing pH from 6.0 to 5.5 will cause a reduction in species numbers and, among remaining species, significant alterations in ability to withstand stress.

At pH <5.0 many species will be eliminated, including molluscs, amphipods, most mayfly species, and many stone fly species; in contrast, several pH-tolerant invertebrates will become abundant, especially air-breathing forms (e.g., Gyrinidae, Corixidae), those with tough cuticles that prevent ion losses (Sialis lutaria), and some forms that live within the sediments (Chiromidae and Tubificidae). Overall, invertebrate biomass will be greatly reduced.

At pH <4.5 all the above changes will be exacerbated, the more tolerant invertebrates will begin to be eliminated and all fishes will be eliminated.

C. Petroleum Hydrocarbons

Petroleum hydrocarbons enter the world's oceans at a rate of 6.1×10^6 tons per year, inputs to United States marine waters being about 25% of the total (National Academy of Sciences 1975; Farrington 1977). Man's use of fossil fuels is already contributing most of the hydrocarbons present in surface sediments of several estuaries and nearshore coastal regions (Farrington and Tripp 1977; Goldberg 1976). Man-generated petroleum hydrocarbons are also clearly incorporated into marine food webs (Goldberg 1976).

Modification of benthic community structure is probably the most significant impact of petroleum hydrocarbons in estuaries and coastal areas. Oil adversely affects the benthic community because of the tendency for all frac-

tions of oil to become attached to particulates and settle out. This property combined with the toxicity of petroleum hydrocarbons (particularly refined products) to invertebrates has resulted in several documented benthos kills (Sanders 1977). Limited measurements in New York Bight sediments show that oil concentrations of 0.2 to 2 mg/g dry weight are consistently associated with seriously stressed benthic communities, i.e., communities markedly degraded in comparison with those of less polluted sediments in the same region. These oily sediments also contain high concentrations of heavy metals, halogenated hydrocarbons, and other toxins, but hydrocarbon levels of 1 to 10 mg/g dry weight sediment also caused substantial benthic mortality and ecosystem-level effects in otherwise uncontaminated sediments off Massachusetts (Sanders 1977). Even if it is assumed that oil is the sole contaminant in sediments of the New York area, concentrations are probably high enough to prevent establishment of normal benthic communities throughout the New York harbors and over 500 square miles of the New York Bight and Long Island Sound. On the basis of the situation in the New York coastal zone, it may be appropriate to consider contiguous areas of 200 square miles with sediments approaching 0.2 mg/g dry weight of oils to represent a serious concern.

D. Heat

Steam condensation during electric power generation liberates large quantities of heat, which are generally discharged to aquatic ecosystems. This may affect ecosystems in a variety of ways including alterations in (1) the physical properties of water such as density, viscosity, vapor pressure, and solubility of gases in it, (2) rates of chemical reactions, (3) biological processes such as reproduction, development, and metabolism, (4) community structure and

stability, and (5) susceptibility of the biota to disease and parasites. Our present ability to quantitate these effects is limited.

Estimates of effects of thermal discharges on biological communities usually arise from coupling of physical models for estimating water quality impacts with dose-response mortality data for selected organisms. Field-tested three-dimensional dispersion models for assessing the impact of thermal loadings on surface waters (Tennessee Valley Authority 1976; 1977) use site-specific data and are therefore of limited value for assessing regional impacts. Field-tested one- and two-dimensional dispersion models are available for regional evaluations of thermal loadings (op. cit.). Carter et al. (1977) have advanced such analytical capabilities by combining existing hydrodynamic models with a thermal response model to assess the hypothetical effects of thermal exposures on organisms subjected to heat discharges.

Specific quantitative impacts of thermal loading on marine ecosystems have been modeled by several investigators. Studies by Clark and Brownell (1973), for example, suggest that a temperature rise of 4° to 5° F over a 35-mile reach of the Hudson Estuary would be sufficient to kill a significant portion of delicate striped bass larvae. Similarly, U.S. AEC studies (1972a) suggest that, in shallow water systems, rooted grass beds may be extensively damaged by thermal pollution.

E. Biocides

Biocides, principally chlorine, are added to power-plant cooling-water systems to control fouling on inside surfaces of condenser tubes. In most cooling systems, enough chlorine is added to produce free available chlorine levels of 0.1 to 0.6 mg/l in waters leaving the condensor (U.S. EPA 1974a). Both chlo-

rine and chloramines (a reaction byproduct of chlorine and ammonia) are toxic to aquatic organisms, frequently at extremely low concentrations.

Studies of freshwater fishes indicate that residual chlorine concentrations of 0.008 to 0.23 mg/l may produce significant mortalities with exposures longer than several days. Lethal short-term (1-h) exposures for fish vary from 0.74 to 0.88 mg/l. The critical level of free chlorine is 0.05 mg/l for young Pacific salmon; it is <0.1 mg/l for Chinook salmon and coho salmon with 72-hr exposures. Studies of chlorine impacts on the giant kelp suggest that exposures of 5 to 10 mg/l reduce photosynthetic capacity of bottom fronds by 10 to 15% after two days and by 50 to 70% after 5 to 7 days (U.S. EPA 1977). Similarly, studies of phytoplankton in Long Island Sound suggest that chlorine at 0.1 mg/l may reduce productivity by as much as 79% (Carpenter et al. 1972). Benthic vertebrates appear to be relatively resistant to the levels of chlorine discharged by power plants (U.S. EPA 1977). The EPA recommends that total ambient residual chlorine levels not exceed 3.0 mg/l to protect salmonid fish and 10.0 mg/l to protect other freshwater and marine organisms (op. cit.). EPA may be revising the salmonid fish standard to 0.02 mg/l.

F. Entrainment

Entrainment may be a particularly important consideration because of the large quantities of water withdrawn for cooling purposes. Its major effects are due to heat and to turbulence, abrasion, and other mechanical shocks to organisms passing through the cooling system.

Several studies suggest that 20 to 70% of the organisms passing through a cooling system are killed (U.S. AEC 1972b; Carpenter et al. 1974). Data from site-specific studies of the effects of entrainment on the striped bass population of the Hudson River Estuary have been used to obtain population impacts by

computer modeling techniques. Results suggest that >30% of the annual brood of an estuarine spawning fish population may be killed by the operation of a single 1000 -MW plant located in a vital nursery area on a river (U.S. AEC 1972b).

Likewise, a pump-storage plant pumping up to 1.7 billion cubic feet of water per day in the same region for hydroelectric generation could potentially eliminate 25 to 75% of the young striped bass of each year's brood (Ray 1973).

Site-specific studies of the quantitative effects of entrainment on copepod populations in Long Island Sound showed that ~70% of the copepods entering the cooling cycle of a nuclear power plant were killed. Combining these mortality estimates with estimated copepod production in the immediate region led to the conclusion that about 0.1 to 0.3% of the copepod production in eastern Long Island Sound could be eliminated (Carpenter et al. 1974).

G. Consumptive Withdrawal

Consumptive withdrawals of water for cooling or process use modify physical characteristics of streams and lakes and adversely affect organisms inhabiting them, including freshwater fish. Their impacts can be assessed by using a detailed river systems model developed by the U.S. Fish and Wildlife Service (FWS) (1977a,b). This model is specifically designed to analyze effects of water use on stream flow, temperature, and sediment transport and deposition and resulting impacts on 60 flow-sensitive stream fish species inhabiting the 18 major watersheds designated by the U.S. Water Resources Council (1968). In this model, probability curves are used to convert alterations in temperature, depth, velocity of flow, and substrate into effects on each of the selected species. (Probability curves have not been completed for all fish species, but should be available in 1978.) Examples of such curves are shown in Figure 4. Outputs of the model include changes in fish production expressed as average biomass of

Figure 4. Probability curves used to convert alterations in depth and velocity of hypothetical stream to effect on brown trout.

4.0 ECONOMIC VALUATION OF ENVIRONMENTAL IMPACTS

4.1 Introduction

The value of environmental conditions can be expressed only in human terms, and the meaning (or definition) of value can be diverse: measurement of value can range from a wholly nonquantitative assessment to the most explicit and detailed mode of quantification. At least three types of valuations have been described: (1) value measured in terms of personal preferences, (2) value reflected in human behavior, (3) value as an absolute (often expressed as moral absolutes).

These represent mental constructs of value which are quite different in terms of measurement, understanding, and usefulness to policy analysis. For the purpose of comparing values of different types of things, it is important to develop a standard metric independent of these differences. Historically the metric has been economic value, but many others might be used. The workshop participants chose economic value as the principle metric and applied themselves to the problem of translating value systems into money. Three methods were considered.

(1) Estimation of the money value of loss in yield of marketed plants and animals.

(2) Analysis of consumer preference as disclosed through questionnaires, interviews, and observed responses to changed or different environmental quality.

(3) Analysis of valuations implicit in laws and regulations.

Another metric, energy content as a resource of value was also considered.

4.2 Valuing Effects on Commercially Produced Goods and Services

Some principles and methods that can be used to value effects of energy-related pollutants on commercially produced goods and services are described here. Emphasis is on methods of obtaining empirical estimates of costs and benefits rather than on the economic theory that provides a methodological justification for the calculations. The methodological foundations are fully discussed in Mishan's book (1976), where it is also shown how the simple analysis described here can be extended to more complicated real situations.

Our approach requires estimates of physical and biological effects of energy-related pollutants. On the basis of these statements, the following steps are performed: (1) estimation of the influence of physical/biological effects on market supply and demand relationships and on market prices and quantities and (2) estimation of the economic costs and benefits associated with these supply and demand shifts.

To illustrate how these steps could be performed, consider the following hypothetical and highly simplified evaluation of the economic loss associated with a pollution-induced reduction of 4% in the yield of wheat. A simple model of the wheat market before the 4% reduction is as follows:

(1) $Y = 25$ where Y = yield per acre per year, in seeded bushels.

(2) $A = (5 \times 10^7) p^{0.5}$ where A = acreage seeded and p = price per bushel.

(3) $Q = Y \cdot A$ where Q = quantity produced.

(4) $Q^d = (2.375 \times 10^9) p^{-0.4}$ where Q^d = quantity demand.

Solution of equations (1) through (4) provides the following predictions for market price and quantity in the absence of a yield reduction:

$p \cong \$2.05$ per bushel,

$Q \cong Q = 1.78 \times 10^9$ bushels,

$$A \hat{=} 7.13 \times 10^7 \text{ acres.}$$

To estimate effects of projected physical/biological impacts on market supply and demand relationships, the yield relationship is modified to reflect a 4% reduction as follows:

$$(1a) \quad Y = 25(0.96) = 24.$$

This changes the market price and quantity:

$$P \hat{=} \$2.14,$$

$$Q \hat{=} Q_d = 1.46 \times 10^9,$$

$$A \hat{=} 7.31 \times 10^7.$$

Note the increase in price, decrease in production and consumption and increase in acreage seeded to wheat.

The second step of our approach is to convert estimates of effects on supply-demand prices and quantities into estimates of the economic costs of pollution. Our measure of economic cost is the dollar loss of consumer surplus plus the loss of the change in the increase in producer rents. For this illustration, we assume that the change in producer rents is zero. Then the economic cost of the yield reduction, measured by the loss in consumer surplus, is given by

$$\begin{aligned} \text{econ. cost} &= \int_{2.05}^{2.14} (2.375 \times 10^9) P^{-0.4} dP = 2.375 \times 10^9 \left(\frac{P^{0.6}}{0.6} \right) \bigg|_{2.05}^{2.14} \\ &= 9.5 \times 10^7. \end{aligned}$$

This is the estimated cost in one year of a single event 4% yield reduction (assuming all else constant).

The above model is simplified beyond the point of direct usefulness. It neglects substitution effects such as those related to technological improvements, agricultural methods, or other adjustments that could reduce the impacts of pollution damage; it neglects many other variables that affect price; and it has an unrealistic time structure. However, realistic models of supply and demand exist for most of the commercially produced goods and services identified by our workshop group as receptors at risk. Perhaps the largest single collection of such models appears in a DOT monograph (d'Arge 1975), which also copiously illustrates the use of such models to value environmental effects.

Other sources of models needed for this approach are agricultural economics journals, master's and doctor's degree theses in economics and agricultural economics, models developed and maintained by the USDA, and agricultural models in commercial econometric forecasting series offered by Wharton Econometric Forecasting Associates, Chase Econometric Associates, Data Resources, Inc., and others.

The model presented above applies to environmental impacts over large areas. A similar model is needed for effects that are geographically limited. The obvious model would multiply decrease in yield by market price on the assumption that a decrease in quantity placed on the market would not affect market price. The total loss would be borne by the producer and owners of variable factors of production. However, if the effect was not due to a one-time occurrence, (such as a year of low rainfall or an early frost) but was expected to be maintained into the future, farmers affected would shift crops. Since the next best crop is usually only slightly less economically attractive, the loss would be much smaller than that calculated from decreased yield times market price. The result would be that the return to farming in the affected region

would be decreased by the difference in net revenue between the original crop and the substitute. This difference would be capitalized and reflected in a decrease in the value of agricultural land.

Two models may be required to calculate these costs. One would be used where the effects are localized and market prices would not be affected. A second would have to be constructed for the case in which the shift caused the price of the damaged crop to increase and the price of the substitutes to decrease or increase. Again, the effect on market value of agricultural land would be calculated. Since marginal products of variable factors can be affected, their returns may also be influenced.

4.3 Values as Estimated by Human Behavior

The basic axiom of this approach is that personal preference is revealed by individual choice among alternative allocations of time and money. The objective of this approach is to establish willingness to pay (WTP) and/or willingness to accept payment (WTAP) for changes in environmental quality through the use of compensated demand functions. A compensated demand function for an environmental "good" is the locus of points showing WTP or WTAP as a function of magnitude of change in the availability of the good, under the assumption of unaltered individual real income or wealth.

Because the magnitudes of most environmental goods are not susceptible to manipulation by an individual, the proper measure of value for these goods is equivalent or compensating surplus. Exact analytical definitions of these terms are provided in any technical economics book dealing with cost-benefit analysis (Mishan 1976).

The basic procedure is to estimate the ordinary demand curve and use it to approximate the compensating demand function for an environmental good by

observing the substitution and complementary relations between the environmental good and marketed goods having observable prices. Theorems drawn from the economic theory of the consumer give analytical content to the relations (Baumol 1977).

Three alternative and mutually exclusive approaches for estimating demand functions for environmental goods are available. These approaches do not differ in basic structure but use different sequences of estimation steps and assumptions to derive demand functions. The first approach, the hedonic, has been presented in detail (Anon. 1975). The second the expenditure function approach, is described by Mäler (1974). With the third, or cost function, approach (Diewert 1974), one is able to infer, up to a monotonic transformation, the utility function from observable market data, rather than having to assume a specific form for this function as required with the first two approaches.

Data for analyzing the above alternatives can be generated by direct observation of individual behaviors or by the use of questionnaires regarding ways in which respondents would alter their time and money budget allocations in accordance with change in the magnitude of the environmental good. Questionnaires must be carefully constructed and administered, but they are relatively inexpensive and provide a rapid mechanism for generating data. An example of questionnaire use appears in a recent study of the value of atmospheric visibility in the Four Corners region (Blank et al. 1977).

The following classifications can be used to value demands for nonmarketed environmental goods: (1) recreation-aesthetics, (2) ecological productivity, and (3) existence and option values.

The literature on demand for outdoor recreation is enormous. It has been well reviewed by Smith (1975) for the National Academy of Sciences. With two no-

table exceptions, Burt and Brewer (1971) and Cicchetti (1976), most of this literature fails to account adequately for the recreationist's ability to substitute across sites.

Very little empirical work on environmental aesthetics outside urban areas has yet been done. A number of studies, generally with residential property values used as data bases, have been done with respect to amenities such as tree plantings, open spaces, etc., but the use of property values raises theoretical problems (Freeman 1974).

Studies of the economic effects of climate change on the productivity of ecological systems are available in a DOT monograph (d'Arge 1975). Although highly variable in quality, these studies provide a wealth of references to other efforts and also provide some good illustrations of what can be done. In general, if a natural scientist can specify the relations between an ecologically important but nonmarketed organism or entity and the ecosystem, the economist, through application of various propositions on derived demand, can infer the economic value of the nonmarketed organism.

Prospects for empirical implementation of analytical models of existence and option values are less bright. Definitions and analytical treatments of option value and quasi-option value have been described by Henry (1974). Some empirical efforts have been moderately successful, the best examples probably being those by Krutilla and his colleagues, particularly with respect to construction of dams in the Hells Canyon area (Fisher and Krutilla 1972).

4.4 Values Implicit in Laws and Regulations

As a society we continually define conditions and behaviors as acceptable or unacceptable. For example, the degree of visibility impairment in clean air regions that we are willing to tolerate has been defined by court decisions

upholding the 1970 Clean Air Act (U.S. Congress 1970) requirements to "prevent significant deterioration" of air quality.

Implementation of such definitions requires that certain costs be incurred. These costs implicitly specify society's minimum willingness to pay to reduce or eliminate the undesirable impacts (consumer surplus is not included). Consequently, an analysis of the social valuation of key present and potential impacts implied by the relevant laws and regulations can yield quantitative estimates useful in national and regional energy models.

4.5 Energy Theory of Value

In recent years researchers have put forth a variety of schemes dealing with alternatives to dollar evaluation (Odum 1973; Chapman 1974). One such alternative is an energy theory of valuation presented in terms of net energy analysis (Odum 1973). Net energy in this valuation is defined as the amount of energy that remains for consumer use after the energy costs of exploration development, production, and delivery are accounted for. Since each input (labor, capital, information, and material--including environment) requires some energy for its production, the energy contained in these inputs subsidizes energy output. The exclusive emphasis on energy and the energy content of inputs in net energy analysis rests on the concept of energy as the ultimate limiting factor. This assertion stems from (1) the role of energy as an input required to run an economic system or production process and (2) the fact that energy cannot be recycled without loss (Gilland 1975).

Given the concept of net energy and the unique nature and role of energy, some researchers have advocated use of energy as the physical measure of environmental impacts of material, capital, and labor requirements. This alternative has two primary approaches. (1) Activity can be centered on processes, both nat-

ural and man-made. Briefly stated, process analysis attempts to determine the energy requirements of a production process by determining the direct energy contribution of each element involved in the process. (2) Activity can be centered on the input-output structure of the economy. The economy in the approach is partitioned into a number of sectors, and energy flow is traced from point of entry and subsequently through other sectors, where its contribution is defined as embodied energy.

Net-energy analyses provide alternatives to dollar evaluation and have received much attention from the energy communities but they are plagued with problems, many of which arise from interesting and important questions that have been ignored:

(1) What is the relative worth (cost) of energy use and production to different sectors of the economy? For example, some forms of energy will be more expensive to produce and use in one sector than in another, and Btu measurements alone do not reflect this.

(2) Does the form in which energy is embodied make a difference in terms of consumer preference?

(3) How can the value of energy-using processes which have consumer value and the value of those which do not be adequately separated?

(4) How can the non-energy effects (i.e., social costs and intergenerational equity) be incorporated into the valuation?

5.0 CONCLUSION

This workshop provided the participants an opportunity to examine existing capabilities for quantifying environmental damage from energy activities. The discussions at the workshop, and the reports subsequently prepared, highlighted a variety of models and data which could be used to provide assessments to help decision makers formulate and direct energy activities. Also noted were the limitations imbedded within these approaches and the need for additional research. Our increasing understanding of the behavior and responses of biological systems to stress and the need to assure that environmental impacts receive adequate attention along with pressing economic and social impacts presents interesting and important challenges to natural and social scientists to develop tools which can be used in the decision making process.

REFERENCES

- Almer, B., W. Dickson, C. Ekstrom, and E. Hornstrom (1974) Ambio 3 (1): 30-6.
- Anon (1975) J. Polit. Econ. 83: 255-77.
- Baumol, W. J. (1977) Economic Theory and Operation Analysis, Prentice Hall, Englewood Cliffs, NJ.
- Benedict, H. M., C. J. Miller, and J. S. Smith (1973) Assessment of Economic Impact of Air Pollutants on Vegetation in the United States: 1969 and 1973, U.S. EPA, Research Triangle Park, NC.
- Berry, W. L. and A. Wallace (1974) Trace Elements in the Environment--Their Role and Potential Toxicity as Related to Fossil Fuels, A Preliminary Study, UCLA 12-946.
- Blank, F. M. et al. (1977) Valuation of Aesthetic Preferences: A Case Study of the Economic Value of Visibility, draft report to EPRI, Resource and Environmental Economics Lab., U. of Wyoming, Laramie.
- Booz-Allen & Hamilton, Inc. (1975) Regional Emission Projection System, System Documentation for EPA, Bethesda, MD.
- Botkin, D. B., J. F. Janak, and J. R. Wallis (1973) Brookhaven Symp. Biol. 24: 328-44.
- Botkin, D. B. and R. S. Miller (1974) Am. Sci. 62: 448-53.
- Botkin, D. B. and J. D. Aber (1977) Some Potential Effects of Acid Rain on Forest Ecosystems: Implications of a Computer Simulation, draft report, BNL.
- Braekke, F. H., Editor (1976) Impact of Acid Precipitation on Forest and Freshwater Ecosystems in Norway, FR 6/76. SNSP Project, 1432 Aas-NIH, Norway.
- Braunstein, H. M., E. D. Copenhauer, and H. A. Pfuderer, Editors (1976) Environmental Health and Control Aspects of Coal Conversion: An Informal Overview, Vol. I & II, ORNL/EIS 94 & 95 (draft).
- Brosset, C. (1973) Ambio 2: 2-7.
- Brouzes, R. J., R. A. McLean, and G. H. Tomlinsom (1977) Mercury--The Link Between pH of Natural Waters and the Mercury Content of Fish, report to U.S. NAS-NRC Panel on Mercury.
- Burt, O. R. and P. Brewer (1971) Econometrica 39: 813-27.
- Carpenter, E. J., B. B. Peck, and S.J. Anderson (1972) Mar. Biol. 16: 37-40.

- Carpenter, E. J., B. B. Peack, and S. J. Anderson (1974) Mar. Biol. 24: 49-55.
- Carter, H. H., J. R. Schubel, R. E. Wilson, and P. M. J. Woodhead, (1977) A Rationale for Evaluating Thermally Induced Biological Effects Due to Once-Through Cooling Systems, Marine Sci. Res. Center Spec. Rept. 7, Stony Brook, NY.
- Chapman, P. F. (1974) Energy Policy 2: 91-103.
- Cherniavsky, E. (1974) Brookhaven Energy System Optimization Model: A Topical Report, BNL 19569.
- Cicchetti, C. J. et al. (1976) Econometrica 44: 1259-76.
- Clark, J. R. and W. Brownell (1973) Electric Power Plants in the Coastal Zone: Environmental Issues, Am. Littoral Soc. Spec. Publ. 7.
- Cole, G. W. (1976) ELM Version 2.0, Range Sci. Dept. Ser. No 20, Colorado State U., Fort Collins.
- Coughenour, M. B. (1978) Grassland Sulfur Cycle and Ecosystem Responses to Low-Level SO₂, Ph.D. Thesis Colorado State U.
- Council on Environmental Quality (1976) Environmental Quality - 1976 Environmental Quality--7th Ann. Rept. of the CEQ, Washington, DC.
- d'Arge, R. C., Editor (1975) Economic and Social Measures of Biologic and Climatic Change, CIAP Monograph 6, prepared for the Department of Transportation, Climatic Impact Assessment Program, Washington, DC.
- Diewert, W. E. (1974) in Frontiers of Quantitative Economics, Vol. 2, pp. 106-206, M.D. Intriligator and D. A. Kendrick, Editors, North-Holland, Amsterdam.
- Dodd, J. L., W. K. Lavenroth, G. L. Thor (1978) in E. Preston (Editor) Bioenvironmental Impact of a Coal Fired Power Plant 4th Interim Report U.S. EPA, Corvallis, Oregon.
- Fairfax, J. A. and N. W. Lepp (1975) Nature London 255: 324-5.
- Farrington, J. W. and B. W. Tripp (1977) Geochim. Cosmochim. Acta, in press.
- Farrington, J. W. (1977) Oceanus 20(4): 5-14.
- Fisher, A. C. and J. V. Krutilla (1972) Natural Resources J. 12, 417-44.
- Freeman, M. (1974) J. Environ. Econ. Mgt. 1 (1), 74-83.
- Galloway, N. J. and E. B. Cowling (1977) J. Air Pollut. Control Assoc. 28(3), 229-35.

- Gilland, M. W. (1975) Science 189: 1051-6.
- Goettle, R. J., E. A. Cherniavsky, and R. G. Tessmer (1977) Energy Res. Abstr. 2, #24, (BNL 22728).
- Goldberg, E. D. (1976) The Health of the Ocean, UNESCO Press, Paris.
- Goodall, D. W. (1973) Modelling in the US/IBP Desert Biome, Res. Memo 73-52, Ecology Center, Utah State U., Logan.
- Grahn, O., H. Hultberg, and L. Landner (1974) Ambio 3 (2): 93-4.
- Heck, W. W. and J. A. Dunning (1967) J. Air Pollut. Control Assoc. 17: 112-14.
- Heck, W. W. and C. S. Brandt (1977) in Air Pollution, 3rd ed., Vol 2, pp. 401-43, A. C. Stern, Editor, Academic Press, New York.
- Heck, W. W., J. B. Mudd and P. R. Miller (1977) in Ozone and Other Photochemical Oxidants, NAS, Washington, DC.
- Hendrey, G. R., K. Baalsrud, T. Traaen, M. Laake, and G. Raddum (1976) Ambio 5: 224-7.
- Hendrey, G. R. and R. F. Wright (1976) J. Great Lakes Res. 2 (suppl. 1): 192-207.
- Henry, C. (1974) Am. Econ. Rev. 64: 1006-12.
- Hindawi, I. S., J. A. Rea, and W. L. Griffs (1977) 70th Annu. Meet. Air Pollut. Control Assoc., Abstr. 77-30.4, Toronto.
- Hinson, M. O. (1977) Selecting Appropriate Receiving Water Models For Use in Residuals--Environmental Quality Management Studies, Draft prepared by Resources for the Future, Inc., Washington, DC.
- Innis, G. (1978) Grassland Simulation Model, Ecological Series, Vol. 26, Springer-Verlag, NY.
- Johnson, B. and R. Sundberg (1972) in Supporting Studies of Air Pollution Across National Boundaries, The Impact on the Environment of Sulfur in Air and Precipitation, Sweden's Case Study for the UN Conference on the Human Environment, B. Bolin, Editor, Royal Ministry of Foreign Affairs and Royal Ministry of Agriculture, Stockholm.
- Landner, L. and P. O. Larsson (1972) Biological Effects of Mercury Fallout Into Lakes From the Atmosphere, Rept. B-115, Swedish Inst. Water and Air Pollut. Res., Stockholm.
- Larson, R.I. (1971) A Mathematical model for Relating Air Quality Measurement to Air Quality Standards, USWPA, (AP-89) Research Triangle Park, NC.

- LeBlanc, F. and J. DeSloover (1970) Can. J. Bot. 48: 1485-96.
- Leivestad, H., G. Hendrey, I. P. Munig, and E. Snekvik (1976) Chapter 4 in Braekke, F.H (1976).
- Likens, G., F. H. Bormann, R. S. Pierce, and D. W. Fisher (1969) in Proc. Brussels Symp. on Productivity of Forested Ecosystems, pp. 553-63, P. Duvigneaud, Editor, UNESCO. Paris.
- Likens, G., F. H. Borman, and N. M. Johnson (1972) Environment 14: 33-44.
- Likens, G. et al. (1977) Biogeochemistry of a Forested Ecosystem, Springer-Verlag, NY.
- Little, E. L. Jr. (1976) Atlas of United States Trees, Vol. 1-3, USDA Misc. Publ. 1314, Washington, DC.
- MacCracken, M. (1978) MAP3S Program Plan, DOE/EV-0008/1.
- Mäler, K. G. (1974) Environmental Economics: A Theoretical Inquiry, John Hopkins U. Press, Baltimore.
- Meyers, R. E., R. F. Cederwall, and W. D. Ohmstede (1978) in Advances in Environmental Science and Engineering, J. R. Pfafflin and E. N. Zeigler, Editors, Gordon and Breach, NY.
- Miller, P. C. and L. Tieszen (1972) Arct. Alp. Res. 4: 1-18.
- Miller, J. M. and J. Wisniewski (1976) A Review of Precipitation Chemistry Studies, Air Resources Lab., NOAA, Silver Spring, MD.
- Miller, P. R. et al. (1977) Photochemical Oxidant Air Pollutant Effects on a Mixed Conifer Forest Ecosystem--A Progress Report, U.S. EPA, Corvallis, OR.
- Mishan, E. J. (1976) Cost-Benefit Analysis, Praeger, NY.
- Muschett, F. D. (1977) Selecting Appropriate Air Quality Models for Use in Residuals - Environmental Quality Management Studies, Draft prepared by Resources for the Future, Inc., Washington, DC.
- National Academy of Sciences (1975) Petroleum in the Marine Environment, Washington, DC.
- Odum, H. T. (1973) Ambio 2: 220-7.
- Økland, J. (1969) Malacologia 9: 143-51.
- O'Neill R. V. et al. (1972) Terrestrial Ecosystem Energy Model, Memo Report 72-19, ORNL, Tenn.

- Parton, W. J. and J. E. Ellis (1978) Impact of Strip-Mine Reclamation Practices: A Stimulation Study Report for the Western Energy and Land Use Team, USFWS, Fort Collins, CO.
- Price, H. E. (1973) Effects of Ozone on Growth and Reproduction of Grasses, p. 84, Ph.D. Thesis, U. of Utah.
- Rao, D. N. and F. LeBlanc (1967) Bryologist 70: 141-57.
- Ray, D. L. (1973) Letter to Senator A. Ribicoff, Washington, DC.
- Rohlf, F. J. and R. R. Sokal (1969) Statistical Tables, Freeman, San Francisco.
- Rosenberg, N. J. (1974) The Biological Environment, Wiley, NY.
- Samuels, G. (1976) Assessment of Water Resources for Nuclear Energy Centers, ORNL-5097, ORNL, Tenn.
- Sanders, H. L. (1977) Oceanus 20 (4): 15-24.
- Schindler, D. W., R. W. Newbury, K. G. Beaty, and P. Campbell (1976) J. Fish. Res. Board Can. 33: 2526-43.
- Schofield, C. L. (1976 a) Ambio 5: 228-30.
- Schofield, C. L. (1976 b) in Proc. Int. Symp. Acid Precipitation in Forest Ecosystems, 477, L. S. Dochinger and T. A. Seliga, Editors, USDA Forest Service Tech. Rept. NE-23, Washington, DC.
- Schofield, C. L. J. Trojnar, and D. A. Webster (1978) Aluminum Toxicity to Brook Trout (*Salvelinus fontinalis*) in Acidified Waters, (in preparation).
- Schwartz, C. C. et al. (1978) Effects of Controlled levels of SO₂ in the Nutrient Quality of Western Wheatgrass and Prairie Junegrass, (in press).
- Sevian, W. A. (1975) The Energy System Network Simulator: General Description and Sample Analyses, BNL 50492 (Draft).
- Sharma, R. et al. (1977) Fish Impingement, Vol. 1 - 2, ANL Argonne, IL.
- Shugart, H. H. and D. C. West (1977) J. Environ. Mgt. 5(2): 161-80.
- Skye, E. (1968) Acta Phytogeographica Suecia 52: 1-123.
- Smith, V. K. (1975) The Estimation and Use of Models of the Demand for Outdoor Recreation, Working Paper No. 3-75, Economic Growth Institute, SUNY at Binghamton.
- Sollins, R., D. E. Reichle, and J. S. Olson (1973) Organic Matter Budget and Model for a Southern Appalachian Liriodendron Forest, USAEC Report EDFB73-2, ORNL, Tenn.

- Stoeckhandt, A. (1871) Tharandter Forstl. Jahrb. 21: 218-54.
- Sutcliffe, D. W. and T. R. Carrick (1973) Freshwater Biol. 3: 437-62.
- Taylor, O. C. and P. R. Miller (1973) Calif. Air Environ. 4(1): 1-3.
- Tennessee Valley Authority (1976) 316 (a) and 316 (b) Demonstrations--Gallatin Steam Plant, TVA, Muscle Shoals, AL.
- Tennessee Valley Authority (1977) 316 (a) and 316 (b) Demonstrations--John Sevier Steam Plant, TVA, Muscle Shoals, AL
- Treshow, M. and D. Stewart (1973) Biol. Conser. 512 209-14.
- U.S. AEC (1972a) Draft Environmental Impact Statement--Crystal River Nuclear Station Unit 1, Washington, DC.
- U.S. AEC (1972b) Final Environmental Statement--Indian River No. 2, Washington, DC.
- U.S. Bureau of the Census (1974) 1974 Census of Agriculture, Washington, DC.
- U.S. Congress (1970) Clean Air Act (Public Law 91-604), Washington, DC.
- U.S. Congress (1972) Federal Water Pollution Control Act Amendments (Public Law 92-500), Washington, DC.
- U.S. Dept. of Agriculture (1974) The Outlook for Timber in the United States, Forest Resource Rept. 20, Washington, DC.
- U.S. Dept. of Energy (1977) Report of Study Group on the Global Environmental Effects of Carbon Dioxide, (draft)
- U.S. EPA (1971) Federal Register 42CFR410.
- U.S. EPA (1973) in Air Quality Criteria for Sulfur Oxides, EPA R3-73-030, Research Triangle Park, NC.
- U.S. EPA (1974a) Development Document for Effluent Limitations Guidelines and New Source Standards for the Steam Electric Power Generating Point Source Category, Office of Water and Hazardous Materials, Washington, DC.
- U.S. EPA (1974b) Fish Kills Caused by Pollution in 1974, EPA 440/9-75-013, Washington, DC.
- U.S. EPA (1977) Quality Criteria for Water, Office of Water and Hazardous Materials, Washington, DC.
- U.S. Fish and Wildlife Service (1977a) Cooperative Instream Flow Service Group/The First Year, Biological Service Program, FWS/OBS 77/36, Fort Collins, CO.

- U.S. Fish and Wildlife Service (1977b) Methodology and Conceptualized Data System to Determine Instream Flow Needs for Energy Assessment Under Section 13 (a) of the Non-Nuclear Act for the U.S. Water Resource Council, Cooperative Instream Flow Service Group, Fort Collins, CO.
- U.S. NAPCA (1969) Air Quality Criteria for Sulfur Oxides, Natl. Air Pollution Control Admin. Publ. AP-50, Washington, DC.
- U.S. Water Resources Council (1968) The Nation's Water Resources, pp. 2-3, Washington, DC.
- Van Hook, R. I. and W. D. Shults, Editors (1976) Effects of Trace Contaminants From Coal Combustion, Proc. Workshop, Knoxville, Aug. 1976, ERDA Div. Biomed. Environ. Res.
- Vaughan, B. E. et al. (1975) Review of Potential Impact on Health and Environmental Quality From Metals Entering the Environment as a Result of Coal Utilization, Battelle Energy Program Report, Richland, WA.
- Waddell, T. E. (1974) The Economic Damages of Air Pollution, U.S. EPA, Washington, DC.
- Wolf, G. T. et al. (1977) Environ. Sci. Technol. II, 508-10.
- Wood, T. and F. H. Bormann (1975) Ambio 4: 169-71.
- Woodwell, G. M. and R. H. Whittaker (1968) AM. Zool. 8: 19-30.
- Wright, R. F. and A. Henriksen (1978) Limnol. Oceanogr. 23, 487-98.
- Wright, R. F. and E. Snekvik (1978) Int. Ver. Theor. Angew. Limnol. Gerh., in press.

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Research Associate Professor of Environmental Systems Engineering, University of Pittsburgh, 1972

Manager, Special Environmental Projects, CONSAD Research Corporation, 1971-72

Assistant Professor of Economics, Purdue University, 1969-72

Staff Economist, Office of Air Programs, Environmental Protection Agency, 1967-69

Associate Editor, Environmental Perspectives, 1974-

Member: Advisory Committee to Develop a Program of Cooperative International Transportation Energy Research, National Academy of Sciences, 1974-;

Advisory Committee on Transportation and Air Quality of the National Academy of Sciences, 1972-73; Advisory Committee, Institute of Transportation and Traffic Safety, The Pennsylvania State University, 1972-73; Technical Committee on Economic Effects, Air Pollution Control Association, 1971-; Board of Directors, Lafayette Environmental Action Federation, 1970-71.

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Brookhaven National Laboratory: Senior Scientist, Technical Support Organization,
1969-72; Consultant, Economic Analysis Group, 1969-72; Head, Economic Analysis
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Associate Department Head of the Systems Analysis Department, MITRE Corporation,
1961-69

Operations Research Scientist, Massachusetts Institute of Technology, 1957-61
Instructor, University of Connecticut, 1949-57

Consultant, National Science Foundation, Economics of Solar Energy

Visiting Professor, Operations Research, State University of New York at Stony Brook

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1967-

Plant Pathologist, Pacific Southwest Forest and Range Experiment Station, Forest
Service, USDA, 1966-

Committee on Medical and Biologic Effects of Environmental Pollutants, Division of
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Member: American Phytopathological Society; American Association for the Advance-
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Brookhaven National Laboratory: Biomedical and Environmental Assessment Division, Scientist, 1977-; Associate Scientist, 1975-77; Assistant Scientist, 1973-75. State University of New York at Stony Brook, Visiting Assistant Professor, 1976- Carnegie-Mellon University, Adjunct Assistant Professor, 1976- University of Pittsburgh Graduate School of Public Health, Research Associate, 1972-73

Illinois State University, Assistant Professor of Environmental Health Science, 1971-72

U.S. Army: Environmental Health Engineer, 1966-68; Chief, First U.S. Army Environmental Health Engineering Service, 1967-68.

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1977-

Science Monitor, Environmental Defense Fund, 1974-1977

Consultant, National Oceanic and Atmospheric Administration Marine Ecosystems Analysis Program, Stony Brook, NY, 1976-77

Assistant Staff Oceanographer, SUNY at Stony Brook, 1973-75

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Chairman, FEA Environmental Advisory Committee, 1973-

President, Sierra Club, 1973-74

Executive Secretary, Committee on Public Engineering Policy, National Academy of Sciences, 1969-75

White House Fellow and Assistant to the U.S. Secretary of Transportation, 1968-69

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1973-

State University of New York at Stony Brook: Adjunct Professor, 1973-; Associate Professor, Marine Sciences Research Center, 1971-73.

Senior Research Associate, Brookhaven National Laboratory, Upton, NY, 1968-71

Chief, Computer Operations, Division, Technical Information Extension AEC,
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Chairman, editorial board, MESA Environmental Atlas of the N.Y. Bight

Member: Estuarine Research Federation; American Association for the Advancement of Science; American Fisheries Society; American Institute of Biological Sciences; American Limnology and Oceanography; Ecological Society of America; International Association for Ecology; International Association of Theoretical and Applied Limnology, Sigma Xi; Phi Kappa Phi.

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Consultant, National Park Service, 1974

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1975-

Assistant Professor of Biology, University of Puerto Rico, 1972-75

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1976-

University of Pittsburgh Graduate School of Public Health: Teaching Fellow, 1975-76; Research Assistant, 1972-74.

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1976-

Fisheries Research Specialist, Regional Office, Fish and Wildlife Service, Denver, 1975-76

Assistant Unit Leader and Assistant Professor, Utah Cooperative Fishery Research Unit, Utah State University, 1966-75

Member: American Fisheries Society; American Association for the Advancement of Science; American Society of Ichthyologists and Herpetologists; The Wildlife Society; Sigma Xi.

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1975-

Consultant; Council on Environmental Quality, 1975-; Commission on Federal Paperwork, 1976-; Bureau of Social Science Research, Inc., 1974-75.

Visiting Research Scholar, Institute for Social Research, University of Michigan, 1974-75

Analytical Statistician, Statistical Policy Division, Office of Management and Budget, Executive Office of the President, 1970-75

Assistant Economist, National Planning Association, Washington, D.C., 1968-69

Summer Intern, Project Analyst, United Nations Development Programme, 1967

Peace Corps Volunteer, Jasin, Alacca, Malaysia, 1963-65

Member: Advisory Committee on Social Indicators, Office of Management and Budget, 1975-76; Planning Committee on Environmental Indices, National Academy of Sciences, 1973-74; Advisory Committee to the Institute for Social Research, University of Michigan, 1972-74.

U.S. Representative: to meetings of the Organization for Economic Cooperation and Development working party on social indicators, 1971-74; to the United Nations Statistical Commission working party on the development of social indicators, 1974.

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