CURRENT PERCEPTIONS AND APPLICABILITY OF ECOSYSTEM ANALYSIS TO IMPACT ASSESSMENT[1,2]

STANLEY I. AUERBACH, Environmental Sciences Division, Oak Ridge National Laboratory,[3] Oak Ridge, TN 37830

Abstract. A framework is presented for defining the environmental impact of a project on an ecosystem. The difficulties in assessing impacts at the ecosystem level are illustrated with examples drawn from theoretical considerations and nutrient cycling studies. The need for rigorous, quantitative analysis of ecosystem deviations from homeostasis and the subsequent implications of this deviation over long periods of time is illustrated and discussed in terms of individual and societal value judgments.

OHIO J. SCI. 78(4): 163, 1978

I recognize that the term *stress effects* in the title of this symposium is not unique with the organizers. I cannot help wondering, however, how Hans Selye, one of the founders of modern endocrinology, would have reacted to the concept of "stress effects on natural ecosystems." I am sure that some thought, such as the following, might have gone through his mind: Does an ecosystem respond by the production of certain biochemical moieties if it is stressed? Are there ecological hormones? Is there an ecosystem equivalent of adrenaline? Being an excellent scientist, Dr. Selye might not have given voice to such questions, but I am sure they might have at least flitted through his thinking when confronted with the title. Words are powerful image creators and convey vastly different meanings to different individuals according to their usage. Ecologists have been struggling for many years to develop a vocabulary which adequately conveys the scientific concepts and perceptions that we are attempting to address or define in our particular areas of interest. Occasionally, ecologists have been prone to fall into traps of terminology which have caused misunderstanding by scientists other than ecologists. On the other hand, such terms have stimulated research which has led to a far better understanding of the processes and phenomena that we are attempting to understand in a holistic context. The term *stress* has strong anthropomorphic connotations. Is there an operational definition for ecosystem stress? Should it be viewed as the analog of endocrinological stress—which is commonly viewed as deviation from homeostasis? Can we apply the same constraint to ecosystems? If so, what are the conceptual and operational problems associated with such an application?

Let me point out at this time that I am not negative to the concept of stress effects in ecosystems; particularly with respect to the problems that are faced in environmental assessment. I think that there are major challenges and research opportunities, as well as social needs, facing us in this area. I hope to show some of the difficulties, both experimental and conceptual, that we face in attempting to look at stresses at the ecosystem level and will describe some findings which illustrate potential avenues of investigation and opportunity. My caution is that we not delude ourselves into allowing some words, which have strong imagery, to lead us into generalizations to which we have fallen prey in the past.

[1]Manuscript received June 6, 1977 and in revised form January 30, 1978 (#77-45).

[2]Publication No. 1062, Environmental Sciences Division, ORNL. By acceptance of this article, the publisher or recipient acknowledges the U.S. Government's right to retain a non-exclusive, royalty-free license in and to any copyright covering the article.

[3]Operated by Union Carbide Corporation for the Energy Research and Development Administration.

One of the most aggressively pursued areas of ecosystem analysis, in relation to stress and impacts of technology, has been the concept of stability. This concept has given rise to many conceptual questions, as well as experimental and theoretical research. Questions such as: To what extent may impacts to an ecosystem be absorbed before disrupting essential homeostatic mechanisms? Which ecosystems are the most susceptible to disruptions? What measurements of the ecosystem will predict disruption? Van Voris (1976) recently reviewed ecological stability in an ecosystem perspective:

"The concept of ecological stability has been approached from many perspectives in recent history. The concept has progressed from a level of intuitive understanding through diversity-stability and connectivity-stability at the population and community level to dynamic stability of system models. All approaches to stability have relied heavily on techniques of measuring system complexity and its relation to ecosystem stability."

"In most cases, the difficulty with stability concepts arises from the misinterpretation of the concept of ecosystems. Most ecologists consider only animal or plant populations, and generally only the aboveground portion(s). An ecosystem contains both biotic and abiotic components and is an externally forced, open system. Considering only the aboveground component ignores the fact that in some forested ecosystems 80% to 90% of net forest primary productivity is cycled directly to the decomposer. Most attempts have dealt with isolated portions of ecosystems and have not yet come to grips with either system complexity nor the ecosystem-level concept. It is interesting to note that classical analyses were concerned with the aesthetics of the environment, and we are still very close to this position today in our understanding of ecosystem stability."

Most ecosystem analysts agree that ecosystems have numerous properties or traits which should be measurable in a dynamic sense. The questions then arise, what are those dynamic properties and how can they be measured? Once these questions have been answered, ecosystem stability can be investigated forcefully. Some researchers, however, have expressed doubts that such properties can be defined and measured. Levins (1974) stated, "It is clear that *none* [referring to systems properties] can be measured by a single variable." May (1974) stated, "I have gloomy doubts as to the feasbility of providing any simple recipe whereby practical people may characterize the resilience of any ecosystem."

I do not wish to succumb to predictions based on a somewhat less than holistic point of view. Nevertheless, we do have problems in viewing the impact of technology on ecosystems and this is especially true if we wish to quantify ecosystem impacts so that they can be factored into cost-benefit analyses.

Stress effects and environmental assessment no longer can be viewed solely on the basis of their scientific merit and need. We are in the age of cost-benefit analyses, an area no longer the arcane territory of the economist. Environmental matters, particularly because of NEPA, now weight heavily on the budgetary calculations, not only of government, but of an increasingly larger number of private and corporate entities. It is, therefore, appropriate that any general treatment of this field also be viewed in the context of the costs versus the benefits.

Table 1 illustrates certain gradients of difficulty in the process of analyzing impacts (Van Winkle *et al* 1976). The difficulties of either predicting or measuring effects (and therefore their costs) increase: (1) as one moves from a short time frame to a long time frame, (2) as one moves from considering effects in the immediate vicinity of a point source of impact to considering effects far removed from the source, and (3) as the scope of the investigation is expanded from effects on an individuals, or populations, to communities or ecosystems.

The specific difficulties which increase (due largely to limitations in the state-of-the art) are related to the cost, the time, and the level of effort required both to obtain and to analyze the needed information. Uncertainty also increases as one moves toward long-term, far-field predictions involving several species.

The direct impact from a power plant, for example, on an ecosystem may be manifested as short-term, near-field effects on individuals of many species. Far-field effects of increasing importance over a longer time frame should persist primarily for mobile species having a long

TABLE 1

*Time, Space, and Biological Complexity Gradients Associated with Analyzing Impacts.**

AS MOVE FROM

SHORT-TERM	TIME →	LONG-TERM
NEAR-FIELD	SPACE →	FAR-FIELD
INDIVIDUAL	LEVEL OF BIOLOGICAL ORGANIZATION →	ECOSYSTEM

GRADIENT OF INCREASING DIFFICULTY OF → EITHER MEASURING OR PREDICTING EFFECTS

DUE TO

LOW	COST →	HIGH
SHORT	TIME REQUIRED →	LONG
RELATIVELY SMALL	UNCERTAINTY →	RELATIVELY GREAT

*Van Winkle *et al* (1966).

generation time and life span, particularly if their reproductivity is concentrated spatially in the vincinity of the source of impact. A logical way to analyze an impact is to begin at the left-hand side of the gradients for time, space, and level of biological organization (table 1), where the impact would be most amenable to prediction via the quantitative application of laboratory and field studies. Consideration of the right-hand side of these gradients, for which simulation modeling is becoming an increasingly important tool, can then proceed as far as data, time, and money will allow. Because only limited information is available concerning community and ecosystem effects in field situations, the population level is currently the focus of assessment. The winter flounder population in the vicinity of Mill Stone Point, CT, and the striped bass populations which spawn in the Chesapeake and Delaware Canal and in the Hudson River are examples of populations which have been the subject of impact assessments at the population level (Christensen *et al* 1976).

My introduction of the concept of cost-benefit in relation to ecological assessment at this point is quite deliberate. Irrespective of whether you, as ecologists, or other environmental scientists have aims toward performing studies related to the need for improving our knowledge of stress effects or whether you are involved in the multiple combination of scientific, social, and legalistic activities that comprise impact assessments, each should be aware of the duality of your roles in this area. First, ecologists have a responsibility to supply scientifically sound and objective predictions of a potential impact of a project on an ecosystem. Our aim as ecologists is the improved fulfillment of this role, which, in turn, requires both basic and applied research directed at better understanding and forecasting the dynamics of entire ecosystems and the important components of ecosystems.

Since it is not practical to evaluate the impact for every ecosystem state variable, the professional ecologist, given constraints of cost, time, and uncertainty, bears the responsibility in selecting the temporal scale, the spatial scale, and the level of biological organization to be used in assessing the environmental impact of a stress. For example, the ecologist may proceed by carrying out a detailed assessment of the potential impact for several representative and important species, or for a group of the species that are important in the functioning of the cosystem. Because the assessment of an environmental impact involves forecasting effects, the ecologist has an obligation to predict effects quantitatively, at least with respect to direction and order of magnitude, by means of past experience, simulation models, and his intuition and judgment as a professional ecologist. Shugart (1976) pointed out that a valuable side effect of variable quantification is that controversies tend to become less emotional.

The ecologist must be wary that the regulatory bodies do not transform ecological concepts and methods into rigid procedures. The establishment of standard procedures and requirements is to be expected in the name of efficiency, uniformity, and ease of regulation. Since there are unique features to each project and to each site, however, such standardization may hinder the making of responsible decisions based on scientifically sound and objective predictions of the potential environmental impact. In this role, the ecologist must also be wary of special interest environmental groups focusing mainly on a restricted set of ecosystem state variables.

The second role that ecologists play is related to their responsibility, in collaboration with professionals from other fields, to recommend the acceptability of the predicted impact. The ecologist is commonly pressured to judge for non-ecologists (e.g., members of a hearing board) the meaning, seriousness, and acceptability of the predicted impact. Not infrequently, the recommendations of an ecologist provide a major part of the basis for decision-making. In playing the second role, however, an ecologist must be aware that he is offering his opinion as a citizen professional. His recommendations are based, in part, on his own values and biases, including those of his professional discipline and those of the party which he represents in the impact assessment procedure. It is not easy to separate professional judgment from personal value judgment; this is probably one of the most challenging obstacles we as ecologists face and is one of the strongest reasons for the further development of the quantitative aspects of our science.

One must distinguish between point sources and non-point sources in dealing with the concept and problems of stress effects on total ecosystems (see table 1). Much of the activity in environmental assessment currently underway in this country deals essentially with individual point sources (e.g., power stations, dams, refineries, or tanker ports). These are technologies whose effects are immediate and impinge on a small part of the landscape. Obviously, many of these individual point sources can operate cumulatively to have a generalized non-point source stress. We are still in the very early stages of trying to deal with the assessments of cumulative point sources. In practice, it has been essential from a number of points of view (scientific, legal, social and economic) to look primarily at the response of species and populations and organisms to these potential sources of ecological damage.

Non-point sources such as acid rain, diseases, or regionally dispersed chemical pollutants impinge on large areas. Once delivered, these sources permeate the entire system so that their potential for damage may include the physical destruction of key structural components of the system. Most types of non-point source insults are subtle in their visible effects and their assessment requires a sophisticated understanding of how ecosystems function. In order to assess non-point sources, we must use not only the kinds of empirical knowledge employed in population studies but also the advanced, theoretical, and mathematical techniques which enable us to make useful inferences, if not direct forecasts, about the consequences of a non-point stress at the ecosystem level.

We at Oak Ridge, in our Ecosystem Analysis program, have had an interest in the study of ecosystem parameters which might serve to measure properties of the total system. Early efforts in this area were highlighted by the work of the brothers Odum, wherein ecosystem parameters were based on power and energy-flow concepts (e.g., community metabolism, or photosynthesis to respiration ratios). These parameters have proven useful in both aquatic and terrestrial ecosystem analyses. Interest in ecosystem parameters based on nutrient cycling resulted from the Hubbard Brook work of Likens and co-workers (1970), which showed that disruption of system integrity by clear-cutting led to a major nutrient loss. Recent theoretical studies (O'Neill *et al* 1975, Reichle *et al* 1975) have indicated that nutrient cycling parameters might be logical candidates for measures of total system system dynamics. Shugart and co-investigators (1976) showed that the nutrient concentration in soil water would be an optimal monitoring point.

Interest in ecosystem parameters led to a desire to understand how changes in component populations are reflected in measures of overall dynamics. Consierable progress has been made in understanding populations in competitive or trophic interactions as summarized by May (1973). Pimental and co-workers (1975) have shown that plant-herbivore systems can co-evolve toward more stable energy dynamics, but these studies have not attempted to relate population phenomena to any overall measure of system performance.

Recently, Levins (1974) has shown that graphical representation of the interaction between system components could provide insight into the dynamics of the overall system. Levins' work has led to a hypothesis (fig. 1) from which the Oak Ridge systems ecology group put together a concept which illustrates to possible relationships between population interactions and system parameters. Populations competing for a limited resource (e.g., light or nutrients) are essentially in a parallel configuration. Disturbance to a single population would result in other populations being freed from competition and growing to utilize the resource made available. Consider a system parameter, such as total photosynthesis or total respiration, which results from the summation of the activity of the individual populations. Since the limited resource tends to constrain the potential of the entire system, the expected result is that the system parameter would remain relatively constant, even though the population changed. In this concept, the various species populations which comprise the sensitivity parameters

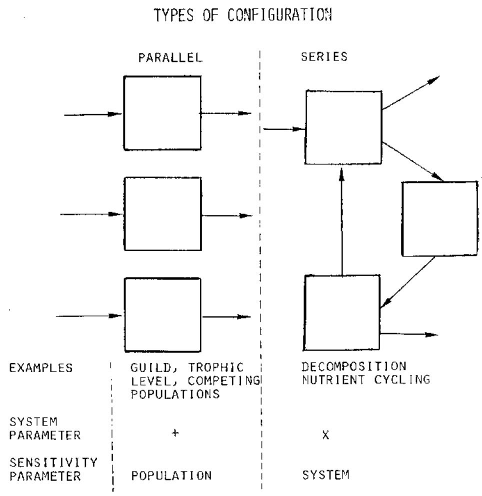

FIGURE 1. Graphical representation of 2 possible relationships between population interactions and system parameters. Populations in parallel would tend to compete for a limited resource such as light or nutrients. Populations in series must process a resource before it can be utilized by other populations in the system, typical of nutrient cycling of decomposition. (R. V. O'Neill, personal communication)

are considered as functional entities and can be expected to have their functional roles maintained, even though the specific gene pools represented by particular species may in themselves be altered or eliminated.

If the populations are operating in a series (or cycling) configuration, then each population must process a resource before it can be utilized by other populations in the system. Such a configuration would be expected in nutrient cycling or in the decomposition of complex substrates. If the system parameter of interest is the loss of nutrients, then the loss can occur from any point in the system. Losses from populations might occur, even though significant population mortality would not be detected, because a simple change in the rate processing could disrupt the synchronization of the system. Thus, one component may release soluble products more rapidly than neighboring populations can process them. Ecological systems can be expected to contain both parallel and series configurations. When the system is disturbed, some parameters (e.g., photosynthesis) will be relatively constant, while other parameters (e.g., nutrient loss) will be relatively sensitive.

This approach enables one to consider the ecosystem as an integrated system with homeostasis achieved through interaction among functional groups of organisms and abiotic components. For example, nutrient cycling results from the functional synchrony of autotroph and heterotroph and soil-organic matter. By focusing on such mechanisms, it may now be possible to identify monitoring points that reflect changes in the total ecosystem.

NUTRIENT RESPONSE STUDIES

The rate at which nutrients are leached from the soil has potential for terrestrial monitoring. Because of the large number of interacting components, detrimental increases and nutrient loss might be detected, irrespective of which specific organisms or processes were being affected. In some cases, it may not be possible to predict the exact mode of action of a new pollutant. In other cases, it may be difficult or impossible to measure the direct effect on specific organisms or processes. Thus, physiological changes might affect the rates and timing of processes without a measurable increase in population mortality. Controlled theory studies (Shugart *et al* 1976) have suggested that soil nutrients can be an optional monitoring point. O'Neill and co-investigators (1977) have looked at this possibility utilizing data from three studies on the effects of toxic substances on different systems (table 2). The first study involved a series of experiments using soil-core microcosms which were excised from the field. Vegetation was removed and the cores maintained in a growth chamber. Aqueous sodium arsenate was applied to the surface of treatment cores in an amount equivalent to 100 ppm arsenic. The microcosms in this experiment had the smallest unit size, no autotrophs, and the most complete spectrum of population community and ecosystem parameters measured.

Table 3 shows the results obtained with the soil core microcosms at the end of 6 weeks. No significant differences were detected in the population parameters. While the parameters given do not monitor a single population, they are indicators of the activity of a single functional group, namely the microbial decomposers. The significant increase was determined in the quantity of nutrients leached from the soil core—in this case calcium and phosphorus—which indicates that a system-level parameter is detectable but not in the array of soil organism measurements.

The second microcosm experiment had a larger unit size and an autotrophic component was included. The design consisted of six boxes of intact Emory silt-loam soil, each 2 m high containing one red maple sapling, with associated herbaceous ground cover. Three replicates were treated with primary lead smelter stack emissions equivalent to 12 mg Pb/cm^2; three boxes served as controls. Soil leachates were collected bi-

TABLE 2

*Experimental designs and parameters for three test systems of various sizes and complexity.**

Exper. unit complexity	Size	Perturbation	Population parameters	System parameters
Soil core, with no autotroph	20 cm^3	Arsenic	Soil microbe density, activity	Nutrients in leachate
Excised soil block with *Acer rubrum*	0.06 m^3	Primary Pb smelter emissions	Aboveground autotrophic growth	Nutrients in leachate
Forested watershed	466 ha	Primary Pb smelter emissions**	Litter arthropod, community diversity and biomass	Litter nutrient and mass pools

*From O'Neill *et al* (1977).
**Includes Cd, Cu, Pb, and Zn.

TABLE 3

*Population and system level parameters measured in soil core microcosms treated with 100 ppm Na_2HAsO_4.**

Treatment	Population parameter		System parameter	
Level (mg/cm)	ATP concentration[a] (μg/g soil)	Bacterial density[b] (10^4/g soil)	Nutrient leached†	
			Ca (μg/ml)	NO^3-N (μg/ml)
0	1.58±0.33	0.50±0.11	20.3±1.7	32.4± 4.8
100	1.52±0.30	0.63±0.18	29.4±2.9	132.2±26.0

*From O'Neill *et al* (1977). Values represent mean±standard error.
[a]Chloroform extract stabilized in sodium bicarbonate buffer at pH = 7.2; analyzed using reaction of glucose with ATP in presence of hexokinase, measured spectrophotometrically at 340 nm.
[b]Standard dilution plating technique, 1:50,000 dilution.
†Measured in leachate from soil core following water addition; analyses for Ca by atomic absorption for NO_3.

weekly by adding excess distilled, demineralized water to the soil.

Table 4 shows the results with the larger soil microcosm containing an intact autotrophic component. After 9 months of monitoring, no significant difference between controls and treatment could be detected in aboveground growth, measured as the sum of branch growth from the previous year's bud-scale scar. The mean values for soil leachates, however, were significantly higher for the treated system. Because the experimental treatment was initiated following bud burst, and most of the nutrients required for new growth are taken up before bud burst, it is not anticipated that significant changes in tree growth will be detected until next year. The relevant point is that a significant effect on nutrient leaching is already evident and measurable.

The third example is based on results from plot samples taken from a smelter-impacted watershed and expands the analysis to a large spatial scale under natural field conditions. These results are from a one-year project, initiated in March 1974, to measure heavy metal impact on the litter components of Crooked Creek Watershed located near a lead smelter in the New Lead Belt region of Southeastern Missouri. This watershed is unique in that surrounding vegetation remains structurally intact, even though it has received smelter emissions containing lead, cadmium, copper and zinc since 1968. Sub-samples were taken from seasonal littler collections along a directional transect running northwest from the smelter stack.

Table 5 illustrates the findings under field conditions of arthropod diversity, using classical techniques, and did not indicate a significant effect at a site 2 km from the smelter when compared to a 21-km control site. System-level effects, however, were significant, as illustrated

TABLE 4

Population and system level parameters from microcosms of Emory silt loam soil containing Acer rubrum *seedlings.**

Treatment	Population parameter	System parameter	
Smelter emissions (Pb mg/cm²)	Annual branch growth (cm)	Leachate concentration (μg ml^{-1})	
		Ca	NO_3-N
0	347±42.8	6.6±0.4	1.4± 1.4
11	346±67.7	10.0±0.8	11.0±10.9

*From O'Neill *et al* (1977). Values represent mean ± standard error.

by litter mass and nutrient pools in litter. While the measurement of the magnesium pool immobilized in litter is not a measure of nutrient loss, it does indicate a disturbance of the recycling process which is detectable 2 km from the smelter. Differences in litter fall between control and 2-km sites did not account for differences in litter mass. Instead, there appears to be a reduction in decomposition rates. At sampling sites closer to the smelter, there was direct evidence of nutrient loss, as well as evidence of the expected trend toward lower arthropod diversity.

tem. Under this concept, the ecosystem can be viewed as homeostatic which indicates that the end-point of succession and evolution is toward some type of homeostasis. The degree of displacement from homeostasis is one of the key concerns that we have in the assessment process. What are the consequences of this displacement? Will it result in acceptable or unacceptable damage to the system? And how do we define damage, and under what time parameters? Obviously, these questions involve the development and presentation of quantified data which illustrate that there has been an impact on the system and the judgment as to the extent to which this impact results in acceptable or unacceptable damage to the system. Such judgments, both the scientific and value judgments, must take time into account.

TABLE 5

*Population and system parameters measured in O_2 litter on Crooked Creek Watershed at 2.0 and 21 (control) km from a Pb smelter in southeastern Missouri.**

	Population parameter	System parameters	
Distance from smelter (km)	Litter invertebrate diversity**	Litter mass (g/m^2)	Mg Pool in litter (g/m^2)
2.0	2.5 ± 0.23	1595 ± 171	1.9 ± 0.3
21.0 (control)	2.3 ± 0.15	1008 ± 110	1.1 ± 0.2

*From O'Neill *et al* (1977). Error term is standard deviation.
**Shannon Index of general diversity: $\left(H = - \sum_{i=1}^{s} \rho_i \ln \rho_i\right)$

Thus, in 3 independent studies, disturbances were detected in nutrient cycling, but not in population/community parameters. This, of course, does not prove that sensitive population parameters do not exist. It is not immediately apparent, however, which population parameters could have been chosen. Furthermore, changes in specific populations might imply very little about effects on the total ecosystem. Perhaps the most tantalizing aspects of these studies concern the mechanism resulting in nutrient leakage. In some cases, the effect may be due largely to the disruption of physicochemical mechanisms in the soil. In other cases, the effects may be on the metabolism of biotic components of the system resulting in physiological changes which have not yet been manifested in significant mortality.

These data suggest that the metabolism of the ecosystem operates independently of the populations that comprise the sys-

LONG TERM IMPACT PREDICTIONS

The timing of ecological phenomena is a major intellectual and institutional challenge to the ecologist involved in the arena of stress assessment. Ecological phenomena may be on the time scale of centuries. Institutional time scales, however, range from the 2-year election cycle of a member of the House of Representatives, to the 5- to 10-year cycle of the business economist, to the 30-year cycle used by some demographers. Timing of phenomena presents 2 challenges: one is the development and acceptability of valid forecasting of consequences over a considerable period; the other is the willingness of social institutions to factor these delayed consequences into planning. We are making progress in the first aspect—namely, our ability to forecast—as illustrated in this Symposium. I am not sure, however, that we have made much progress yet in the aspect of social planning.

The opportunities and difficulties associated with evaluating a long-term stress on an ecosystem are illustrated by the recent work of Shugart and West (1977). They examined the impact of the chestnut blight on Southeastern forests to analyze the regional effects of stress on ecosystems and to evaluate the use of long-term simulation models of forest ecosystems. The causal organism of chestnut blight, a fungus parasite, was introduced into New York from Asia on imported Asiatic chestnut nursery stock around the turn of the century. Within 30 years of its discovery in 1904, the chestnut blight had destroyed practically all of the mature chestnut trees in the Appalachian region of the eastern United States. This sudden removal of one of the most important canopy tree species in the oak chestnut forest constituted an ecological catastrophe of rare magnitude.

In our context of stress, we have the situation of a non-point source rather quickly eliminating a major structural component of an ecosystem. What effect, if any, did this removal have on the total forest system? A number of investigators have studied the effects of the removal of chestnut, but too little time has passed to dcument the nature of the changes in forest dynamics. The development of ecosystem simulation models, however, has made it possible to develop a rationale and model strategy to look at the long-term projected impact of chestnut removal. Shugart and West (1977) chose to derive their model from Botkin and co-workers (1972), in which the annual change of the model forest stand is simulated by calculating the growth increment of each of the trees growing on the stand, tabulating the addition of new saplings to the stand, and tabulating the death of trees present on the stand. These processes were all considered to be stochastic functions in the model.

The Shugart and West (1977) model was designed to simulate stand development on sites in lower ridge slopes in Tennessee. They compared the post-blight Appalachian forest with pre-blight forest containing chestnut. The model was run for 1000 years both with and without chestnut. Basically, the generation time (turnover time) of the Appalachian hardwood forest is on the order of 580 years. Therefore, to get an accurate picture of the consequences of change induced in that forest by the impact of a stressor—such as loss of chestnut—one needs to look at the impact over several forest lifetimes.

The validation obviously is contingent on the availability of chestnut forest data that antedated the chestnut blight. A number of authors have published on chestnut composition in the Southern forests; many of these studies go back to the period of the chestnut catastrophe. Of particular use to this analysis, however, was a stand table published by Foster and Ashe (1908). This stand table provided a rare and relatively good quantitative summary of forest composition for some counties in East Tennessee. These early stand tables were based on uncut virgin forests. The authors were, thus, able to fit these parameters into their model and simulate forest growth with and without chestnut. Figure 2 illustrates one of these simulations. In the region of the arrows, for a 25- to 50-year period, the 2 conditions are different each year at the 95% level of confidence. These differences in biomass are related to stand-thinning associated with forest canopy closure. Over the next 100 to 150 years, the forest tends to accumulate biomass, but there is as much as 25% greater biomass in forests with chestnuts. After 400 years, the stand biomass tends to remain relatively constant, that in which chestnut is present typically having slightly greater biomass.

Figure 3 illustrates the simulation of a leaf area index. These data illustrate that the forest with chestnut tends to have more leaf area than that without chestnut. The reason underlying this is that chestnut has a greater tendency to reproduce vegetatively, which increases the probability that the individuals of

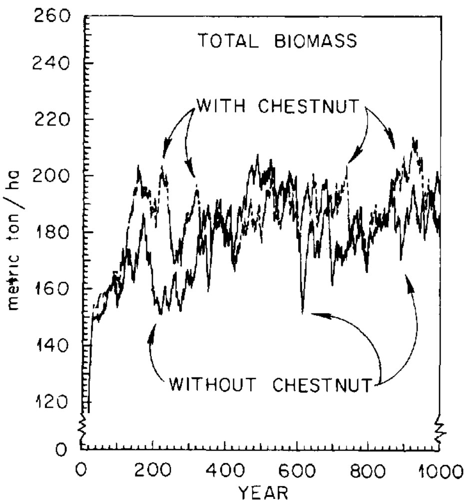

FIGURE 2. Changes in biomass for Appalachian deciduous forests during 1000 years of simulated stand development. Data shown are the mean values for 100 simulated 1/12-ha plots for each of the 1000 years. Simulations all began with an open plot. Dashed lines represent simulations with chestnut as a viable species; solid lines represent simulations without chestnut. (From Shugart and West 1977)

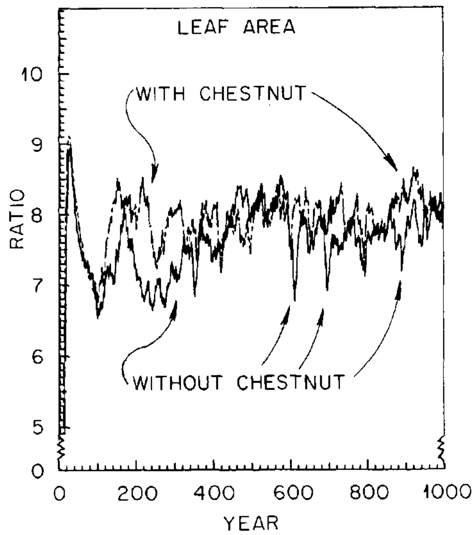

FIGURE 3. Changes in leaf area (m^2 of leaves/m^2 of land area) for Appalachian deciduous forests during 1000 years of simulated stand development with and without chestnut. Data shown are the mean values for 100 simulated 1/12-ha plots for each of the 1000 years. Dashed lines represent simulations with chestnut as a viable species: solid lines represent simulations without chestnut. (From Shugart and West 1977)

this relatively fast-growing shade-tolerant species will be in a position to be in the overstory, should a canopy tree fall.

IMPLICATIONS OF ECOSYSTEM STRESS ANALYSES

These examples illustrate both opportunity and challenge. If one looks at stress impacts on ecosystems, 2 kinds of scientific input are required. The first involves data collected at the ecosystem level of organization, such as those data resulting from biome studies in the International Biological Program (IBP). The second requires the further development and use of sophisticated ecosystem models which will permit valid predictions of the consequences of induced stresses. The challenge posed to us is that of arriving at a decision (based upon such a projection) as to whether the consequences are acceptable or nonacceptable, either in a scientific or a social sense. We are not in a position to prognosticate the other consequences of the removal of chestnut. As a timber tree, it was not too valuable; some replacement trees have greater value. Its impact on long-term forest management may be important, although here again we are confronted with man's interposing other changes in the system, such as the replacement of hardwoods in many areas by pine plantations. Nevertheless, I do feel that the further development of this type of predictive capability will be increasingly useful in the fulfillment of our dual roles of ecologists and ecologist citizens.

I think the only way we are going to arrive at a valid and ultimately accepted institutional concern over the potential of the consequences of a generalized non-source stressor (e.g., acid rain) is through the combination of models, as illustrated here, together with detailed information on the changes in subtle ecosystem parameters, such as those illustrated earlier on nutrient cycling.

There is another use of ecosystem data and modeling in relation to stress involving the concern regarding the transport of potentially deleterious substances within the ecosystem and their possible future availability to man. Illustrative of this problem are the long-term, long-lived toxic chemicals. Studies are currently underway concerning the management of these in the environment if they are interred as wastes, as is done with the element pultonium. The Environmental Protection Agency (EPA) and the Department of Energy (DOE) are looking at criteria and/or are attempting to establish standards for the long-term interment of plutonium in different parts of the United States. One of the obvious concerns is the behavior of plutonium in ecosystems in situations where there have already been releases due to early atomic operations. Again, fortunately, we now have had a number of years of experience in the development of ecosystem models, which makes it easier for us to provide quantitative projections on the behavior of such an element in the soil (fig. 4).

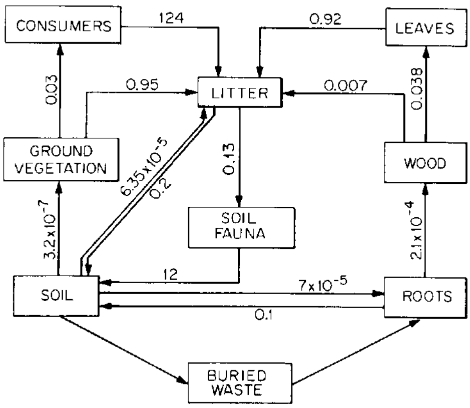

FIGURE 4. Conceptual model of a floodplain forest ecosystem showing compartments and annual transfer coefficients (From Garten *et al* 1977).

Figure 5 projects the buildup of two isotopes of plutonium in a forest over a 120-year time span. It is important to note that we have to be in a position to make projections over periods that go well beyond the normal thinking level of the institutional decision-maker and beyong the experimental time frame of the researcher. With the further development of our knowledge of ecosystem behavior and of our ability to formalize these in models, we will be in a better position to make the kind of projections which do have an important input into the kinds of decision-making that are pertinent to long-term questions.

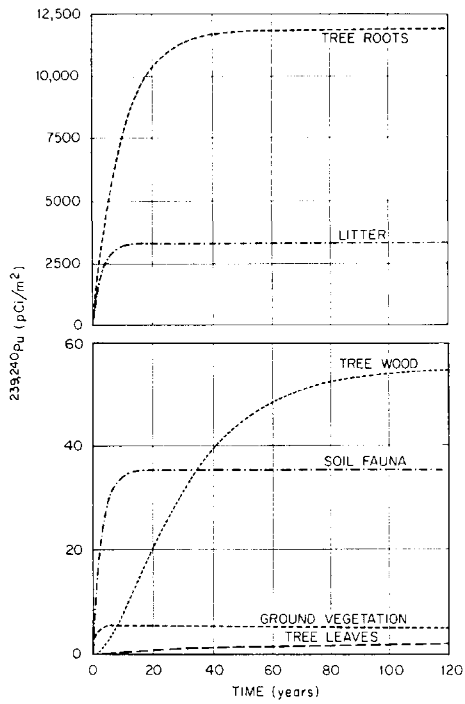

FIGURE 5. Simulated uptake of plutonium from contaminated soil by different biotic components of a floodplain forest ecosystem (From Garten *et al* 1977).

SUMMARY

Let me summarize the salient points that I have attempted to make. One, if we consider stress in the context of a deviation from some homeostatic condition, we do face a number of technical and socially related questions. The technical questions are those concerning the need to define, in rigorous scientific terms, the meaning of ecosystem homeostasis, i.e., the significance, both temporally and spatially, of a deviation from homeostasis and of the elucidation of the acceptability and nonacceptability of such a deviation. The latter, of course, puts us into our role as scientist-citizens. Here we enter the realm of value judgment where we provide only one of many inputs which need to be considered by an institutional decision-maker. These are all tough challenges—scientifically, socially, and institutionally. However, the whole area of environmental assessment, whether it is related to stress, to impacts, or to implementation of NEPA, provides an unparalleled opportunity for ecologists to improve their science and to render it socially useful and acceptable as no other challenge or opportunity has done in the past.

LITERATURE CITED

Botkin, D. B., J. F. Janak and J. R. Wallis 1972 Some ecological consequences of a computer model of forest growth. J. Ecol. 60: 849–873.

Christensen, S. W., W. Van Winkle and J. S. Mattice 1976 Defining and determining the significance of impacts: concepts and methods. p. 191–219. *In* R. K. Sharma *et al* (eds.), Proc. Workshop Biol. Significance Environ. Impacts, NR–CLNF–002, U. S. Nuclear Regulatory Comm., Washington, D. C.

Foster, H. D. and W. W. Ashe 1908 Chestnut Oak in the Southern Appalachians. USDA Forest Serv. Circ. No. 124. 23 pp.

Garten, C. T., Jr., R. C. Dahlman and R. H. Gardner 1978 A linear compartment model of plutonium dynamics in a deciduous forest ecosystem. Health Phys. (in press).

Levins, R. 1974 The qualitative analyses of partially specified systems. Ann. N. Y. Acad. Sci. 231: 124–138.

Likens, G. F., F. H. Bormann, N. M. Johnson, D. W. Fisher and R. S. Pierce 1970 Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook Watershed e osystem. Ecol. Monogr. 40: 23–47.

May, R. M. 1973 Stability and Complexity in Model Ecosystems. Princeton University Press, Princeton. 235 pp.

——— 1974 Stability in ecosystems: Some comments. p. 67. *In:* A. J. Cave (ed.) Proceedings of the First International Congress of Ecology. The Hague, The Netherlands, Centre for Agri. Publ. and Document., Wageningen, Netherlands.

O'Neill, R. V., W. F. Harris, B. S. Ausmus and D. E. Reichle 1975 A theoretical basis for ecosystem analysis with particular reference to element cycling. p. 28–40. *In:* F. G. Howell, J. B. Bentry and M. H. Smith (eds.), Mineral Cycling in Southeastern Ecosystems. ERDA Symposium Series, CONF–740513, Washington, D. C.

———, B. S. Ausmus, D. R. Jackson, R. I. Van Hook, P. Van Voris, C. Washburne and A. P. Watson 1977 Monitoring terrestrial ecosystems by analysis of nutrient export. Water Air Soil Pollut. 8: 271–277.

Pimentel, D., S. A. Levin and A. B. Soans 1975 On the evolution of energy balance in exploiter-victim systems. Ecology 56: 381–309.

Reichle, D. E., R. V. O'Neill and W. F. Harris 1975 Principles of energy and material exchange in ecosystems. *In:* Proc., Plenary Session 1st Int. Cong. Ecol., The Hague, Sept. 8–14, 1974.

Shugart, H. H. 1976 Suitability indices for ecological evaluatioon of alternatives. *In:* R. A. Karem *et al* (eds.), Proceedings of the Short Course on Environmental Impact Statements for Nuclear Power Plants. Georgia Institute of Technology, Altanta, GA, November 26–30. Pergamon Press, New York.

———, D. E. Reichle, N. T. Edwards and J. R. Kercher 1976 A model of calcium-cycling in an East Tennessee *Liriodendron* forest: Model structure, parameters and analysis in the frequency domain. Ecology 57: 99–109.

——— and D. C. West 1977 Development of an Appalachian deciduous forest succession model and its application to assessment of the impact of the chestnut blight. J. Environ. Manage. 5: 161–179.

Van Voris, P. 1976 Ecological stability: An ecosystem perspective. ORNL/TM–5517. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 36 pp.

Van Winkle, W., S. W. Christensen and J. S. Mattice 1976 Two roles of ecologists in defining and determining the acceptability of environmental impacts. Intern. J. Environ. Stud. 9: 247–254.