ECOLOGICAL ASSESSMENT: ECOSYSTEM MANAGEMENT CURRENT STATUS AND IMPLICATIONS FOR NEW JERSEY

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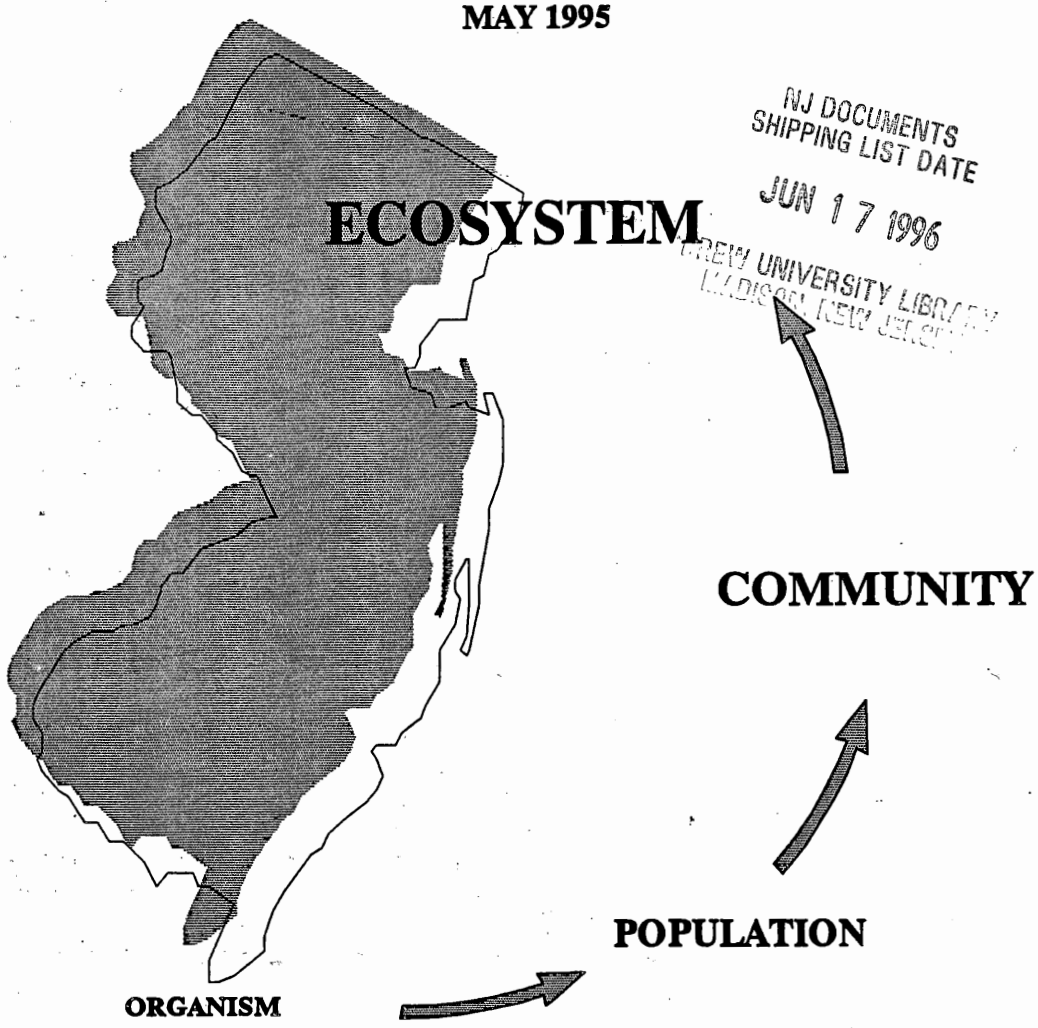

New Jersey Department of Environmental Protection
Division of Science & Research

Vision: The Department of Environmental Protection is committed to providing a high quality of life for the residents of New Jersey.

Mission: To assist the residents of New Jersey in preserving, sustaining, protecting and enhancing the environment to ensure the integration of high environmental quality, public health and economic vitality.

We will accomplish our mission in partnership with the general public, business, environmental community and all levels of government by:

- ❑ Developing and integrating an environmental master plan to assist the Department and our partners in decision-making through increased availability of resource data on the Geographic Information System.

- ❑ Defining and publishing reasonable, clear and predictable scientifically-based standards.

- ❑ Achieving the Department's goals in a manner that encour ages compliance and innovation.

- ❑ Employing a decision-making process that is open, comprehensive, timely, predictable and efficient.

- ❑ Providing residents and visitors with affordable access to safe and clean open space, historic and natural resources.

- ❑ Assuring that pollution is prevented in the most efficient and practical way possible.

- ❑ Assuring that the best technology is planned and applied to achieve long-term goals.

- ❑ Assuring that non-treatable wastes are isolated, managed and controlled.

- ❑ Enhancing environmental awareness and stewardship through education and communication.

- ❑ Fostering a work environment that attracts and retains dedicated and talented people.

- ❑ Committing to an ongoing evaluation of the Department's progress toward achieving our mission.

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ABSTRACT

A general overview of ecological assessment, with an emphasis on risk assessment from a toxicologic perspective, is provided including background on theory and related concepts. Ecological assessment is considered as linked with ecosystem management. Current ecological assessment and ecosystem management programs in the United States are identified at both the federal and state levels. The intent of this research has been: to examine various approaches for assessing ecological effects; to bring New Jersey Department of Environmental Protection (NJDEP) programs up-to-date in this rapidly developing field; to foster links between programs; to identify outstanding issues with some suggested resolutions; and to outline a framework for addressing these issues in New Jersey.

CONTENTS

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I. Executive Summary

Although New Jersey is only 7,509 square miles in size (the fifth smallest land area of the fifty states), it is the most densely populated state in the country, first in pharmaceutical, and second in chemical manufacturing. Yet, New Jersey contains a landscape high in biodiversity ranging from the Northwestern ridgetops to the Middle Atlantic Coastal Plain with 2,600 known plant species, 335 known resident vertebrate species, and countless invertebrate species. However, over 600 of the plant species (23%), 128 vertebrates (38%), 183 invertebrates, and 38 natural communities are classified as either rare or endangered in New Jersey.

Precisely because New Jersey contains high species diversity, the anthropogenic environmental stresses associated with such a high population density and industrial base, as well as habitats of critical importance to numerous species (including more than a million migrating birds), it is important that ecosystem assessment and its associated management be scientifically grounded, thorough, and thoughtful.

This review of ecoassessment emphasizes risk from a toxicologic environmental input approach usually applied by programs bound by pollutant-specific regulations. A more holistic approach is usually taken by natural and historic resource protection programs focusing on habitat, biodiversity, rare and threatened species protection, etc. Many of the concepts described in the General Methodology, Environmental Indicators, and other sections, however, can be applied to ecosystem assessment overall, regardless of the approach.

The objectives of this research have been to: examine the various approaches for assessing ecological effects; identify outstanding issues; and develop a protocol for addressing these outstanding issues.

Future research should examine the holistic approach further; examine overall where methods from the toxicologic and holistic approaches can be knitted together to establish an ideal paradigm for ecological assessment; and examine how such a paradigm might be useful for improving individual program assessments and needs.

Chapter II. Ecological Assessment/Ecosystem Management

General Methodology Within Predictive and Retrospective Approaches

Chapter II is a detailed overview of ecosystem assessment/ecosystem management methods. Some perspective is given on ecoassessment, particularly in light of its differences with human health assessment. A general ecorisk paradigm could include: 1) Problem Formulation; 2) Analysis; and 3) Risk Characterization. Analysis includes Stressor-Response Analysis and Exposure Analysis.

The general paradigm is discussed within the context of predictive ecological risk assessment and retrospective risk assessment. Discussion of ascertaining effects at the organism, population, community, or ecosystem level are presented within the section on predictive ecological risk assessment, but are also applicable for retrospective assessment. Numerous empirical, modeled, or combinations of approaches at each ecological assessment level are presented.

In general, as the level of ecosystem assessment increases in scope (from organism to population to community to ecosystem), the more complex the assessment. Approaches for assessing ecosystems can be landscaped-based (freshwater, marine, estuarine, terrestrial, wetlands, etc.), function-based (energy sources, nutrient cycling, ability to recover), or some combination of both.

Models specific to New Jersey ecosystems are discussed including food chain models by Thomann (1989) and Thomann et al. (1989; 1992) developed with data from the Hudson River.

Retrospective assessment is distinguished from predictive assessment in that risk is evaluated based upon a prior event or activity. Epidemiologic criteria for establishing causation within the context of human health assessment may be adapted as guiding principles for establishing causation within an ecological assessment.

Ecological Risk Assessment and Genetic Engineering

The general ecological risk assessment paradigm can be modified for assessing risk from genetically engineered organisms.

Regional Ecological Risk Assessment
The general paradigms for predictive and retrospective ecological assessment could be applied over regional scales defined by natural features (such as watersheds, physiographic provinces, population ranges, etc.), by anthropogenic features (such as pollutant sources using industrial ecology/industrial metabolism approaches), or some combination of both. The industrial ecology approach allows for regional trend analysis considering sources of contaminants, their loading into various media, and related effects considering past, current, and future activity.

Operational Considerations
Operational considerations for deriving acceptable contaminant exposure levels for aquatic and terrestrial species are discussed within various media exposure scenarios. Federal and New Jersey approaches are included. For the most part, these methods are single pollutant:target species toxicity-based approaches.

Uncertainty Factors. The use of uncertainty factors in ecological assessment is not consistent and there is not yet scientific consensus as to the most acceptable application of such factors. In New Jersey, some specific factors and guidances exist in the site remediation and surface water programs.

Environmental Indicators
Indicators can be used to ascertain regulatory compliance and regulatory success; as measures of the environment's health; and as inputs for conducting ecoassessments. Since indicator data are often developed through regulatory compliance, monitoring, or natural resource protection programs, environmental indicators can provide useful data often unavailable in the published literature. The New Jersey Department of Environmental Protection's (NJDEP) Division of Science and Research (DSR) is currently preparing a compendium of approximately 40 NJDEP ambient monitoring programs and evaluating a subset of these for developing indicators.

Chapter III. Federal and State Ecological Assessment Programs

Chapter III provides a comprehensive review of ecological assessment programs within the Federal government and within New Jersey. Limited discussion is given to ecoassessment programs within other states.

U.S. Environmental Protection Agency (EPA). EPA has been working toward a uniform ecological risk assessment paradigm. Although there is currently no standard process for ecological assessment throughout EPA, individual program offices have legislative and regulatory authorities for conducting ecological assessments and have tailored guidances specific to their programs. These authorities include: CERCLA/SARA for site remediation; Clean Air Act; Clean Water Act/Water Quality Act of 1987; FIFRA; National Acid Precipitation Assessment Program; and NEPA.

U.S. Department of The Interior (DOI). DOI's National Biological Service (NBS) was established in 1993 to develop an appropriate survey of basic biologic data and trends to prevent ecosystem endangerment.

NBS's GAP Analysis Project is looking at biodiversity through geographic display of vertebrate and plant species in concert with land use. DSR has been working with NJDEP's Natural Heritage Program, Division of Fish, Game, and Wildlife, and Geographic Information System to facilitate New Jersey's participation in the GAP Analysis Project with NBS representatives.

Other NBS initiatives include: State Partnership Program; Species-At-Risk; Museum Initiative; Ecosystem Projects; Status and Trends Reports; and a National Biological Information Infrastructure.

National Oceanic and Atmospheric Administration (NOAA). NOAA, provides technical support to USEPA and States to regain costs for cleanup and restoration of marine and coastal resources. NOAA also conducts a number of monitoring programs through its National Status and Trends Program for Marine Environmental Quality including its Mussel Watch Project and Benthic Surveillance Project. In New Jersey, these programs link with the NY/NJ Harbor Estuary and Delaware Estuary Programs.

U.S. Department of Agriculture (USDA). Legislative authorities for USDA regulation of genetically-altered organisms include: Federal Plant Pest Act; Plant Quarantine Act; Federal Meat Inspection Act; Poultry Products Inspection Act;

Virus-Serum-Toxin Act; and Food Security Act of 1985.

Ecosystem Management in New Jersey. Ecosystem management has been consciously fostered for quite some time in New Jersey. Some program and project highlights include:

- Creation of the Pinelands National Reserve in 1978.

- Current DSR-sponsored Pinelands research project focusing on impacts to native vegetative species and ground and surface waters.

- DSR and Geographic Information System (GIS) participation in the Delaware Estuary and NY/NJ Harbor Estuary Programs.

- GIS inventory of natural resources data including freshwater wetlands, land use/land cover, etc.

- DSR investigations of mercury in fish; finfish resource assessment in New Jersey estuarine waters; and pollutant fate, transport, and accumulation in Barnegat Bay.

- New Jersey and Delaware Chapters of The Nature Conservancy implementation of a strategic plan for preservation of the Delaware Bayshores Bioreserve.

- Creation of the Highlands Trust Advisory Board emphasizing watershed protection.

- NJDEP's Whippany River Watershed Pilot Study for ecosystem-wide watershed management.

- NJDEP Division of Fish, Game and Wildlife's Endangered and Nongame Species Program (ENSP) research including projects on: contaminant effects in bird populations; species habitat; population decline; and landscape approaches to rare species protection.

- ENSP's ecosystem management including: population restoration; management plans for specific ecosystems; and monitoring and management of beach nesting bird sites.

- ENSP's revision and publication of the endangered and nongame species list for New Jersey.

- NJDEP Division of Parks and Forestry, Office of Natural Lands Management (ONLM) programs that focus on habitat protection and passive recreation including: Natural Heritage Program and Endangered Plant Species List; Natural Areas Program; Natural Lands Trust; Open Lands Management Program; Trails Program; and Wild and Scenic River Program.

- ONLM-sponsored research on endangered plant species in New Jersey and integration of these results with the Natural Heritage Database and Endangered Plant Species List.

- NJDEP's Bureau of Forest Management forest health monitoring, urban and community forestry projects, forest tree nursery, conservation education, application processing and inspection of private land forestry operations, park and forest management, and forest research projects.

- Parks and Forestry and DSR are supporting research on Atlantic white cedar regeneration to address the decline of this species and develop management strategies.

Ecological Site Assessment in New Jersey. Specific NJDEP program offices in which site or region-specific ecological assessments are conducted include the Environmental Toxicology and Risk Assessment (ETRA) Unit within the Division of Publicly Funded Site Remediation and the Office of Natural Resource Damage Assessment within Natural and Historic Resources.

Ecological Assessment in Deriving Standards and Criteria in New Jersey. In New Jersey, ecological endpoints are considered in surface water quality criteria, soil standards, standards for sewage sludge and sludge-derived materials, and requirements under the Clean Air Act.

Environmental Indicators and Data Bases in New Jersey. In addition to those being developed via the DSR research previously described, other NJDEP indicator programs and sources include:

- Water Monitoring Management Program within DSR including: ambient stream monitoring network; ambient ground water monitoring network; lakes sampling and assessment program; ambient biomonitoring of watersheds; ecoregion

biomonitoring program; coastal plankton monitoring; zebra mussel monitoring; and shellfish and sediment monitoring.

- Biennial New Jersey Water Quality Inventory Report assessing water quality in New Jersey's rivers, lakes, estuaries, oceans, and ground water.

- Yearly New Jersey air quality report published by the NJDEP Bureau of Air Monitoring.

- Notable Information on New Jersey Animals (NINJA) data base on New Jersey's fresh water and terrestrial wildlife resources.

- New Jersey Natural Heritage Database of rare plants and animal species and representative natural communities in New Jersey.

- ENSP survey projects including shorebirds, raptors, beach nesting birds, herptiles, and various other vertebrate and invertebrate surveys.

- Data on deer herd health, habitat loss, and damage collected by NJDEP's Deer Management Program.

- Data on hard and soft clams, invertebrates, polychaetes and mollusks, as well as water quality collected by NJDEP's Bureau of Shellfisheries.

- Fish, invertebrate, and water quality data tracked by NJDEP's Bureau of Marine Fisheries.

- Bacteria, toxicology, and pathology data for fish and wildlife collected by NJDEP's Office of Fish and Wildlife Health and Forensics.

Other New Jersey sources of indicator data include non-DEP monitoring, survey, and sampling data from naturalist societies, academics, volunteer monitors, industry, public works agencies (such as water purveyors), etc.

NJDEP's Geographic Information System (GIS). GIS is a sophisticated tool for mapping a wide range of ecological data. NJDEP's GIS is actively mapping both natural resource and stressor data. The GIS can be consulted and incorporated into ecological assessment where appropriate.

Current Initiatives for Integrating Ecosystem Assessment:

- DSR together with the Division of Fish, Game and Wildlife (FGW), initiated a series of meetings in 1994 to bring together scientists from throughout NJDEP to discuss ecosystem assessment issues, program needs, and explore collaborative efforts.

- Rutgers University sponsored meetings in late 1994 to explore issues, opportunities, and action steps for developing a biodiversity initiative in New Jersey, including a New Jersey Biological Survey.

- March 1995, DSR established an Environmental Research Partnership among the academic, research, government, non-governmental, business, and environmental communities to facilitate coordination of basic data, data needs, and research for ecosystem management in New Jersey.

Chapter IV. Outstanding Issues in Ecological Assessment

Some of the major interrelated scientific and policy issues that remain outstanding as the field of ecological assessment continues to develop include:

- Defining ecological assessment
- Defining ecosystem health
- Data availability
- Data applicability
- Scientific uncertainty
- Risk characterization
- Lack of uniformity

In light of these overriding issues, several suggestions can be offered:

1) Problem formulation and assessment design must be thorough, should consider indirect effects, magnitude of response, and temporal variations.

2) Develop criteria for defining ecosystem health and implement steps necessary for evaluating system health based on these criteria. Consensus as to what level of impairment is acceptable in defining ecosystem health may be necessary.

3) Develop statewide indicator program with two prongs: a) indicators for ecological assessments required within individual regulatory programs; and b) ecosystem level indicator program to address measures of overall ecosystem health on a system-

wide basis.

4) Develop comprehensive ecosystem data program to identify extant data sources and directories for New Jersey systems, including indicators data, and couple this to a coordinated statewide research program to fill data gaps.

5) Explore feasibility of uniform ecological assessment guidelines. Include: study design; hierarchy of appropriate data types; criteria for selecting a critical study, endpoints, models, and data extrapolation; protocol for use of uncertainty factors; and incorporation of qualitative aspects.

6) Statewide working group(s) on ecological assessment could begin to discuss the issues of concern and develop consensus about scientifically valid approaches.

7) Consider establishing a set of scientifically-based ecological quality criteria for all media: air, water, soil, and sediments. Identify constraints to individual programs, how these constraints might effect ecosystem evaluation, and possible remedies.

8) Identify and undertake priority ecological assessment research projects. Those already identified include: trophic transfer studies of bioaccumulative contaminants through an estuarine food chain from sediment to apex level species to develop trophic transfer and biota-to-sediment accumulation factors for risk assessment and standard setting; development of valid mixture assessment methods to ascertain biological potency of chemical mixtures present in the environment; and development of ecological toxicity profiles (including guidance on deriving no-adverse-effects concentrations) for compounds routinely encountered at New Jersey sites but for which comprehensive data are not currently available in a readily accessible form.

Other research needs include: contaminant fate and transport including speciation and associated toxicity; cumulative contaminant impact analyses; efficacy of current media-specific criteria and remediation programs through media and receptor sampling; and environmental and human health benefits of native species landscaping and restoration. Also, the GAP Analysis for New Jersey should consider expansion to include a better level of precision, aquatic organisms, and application to programs such as site remediation and resource damage assessment.

9) Develop data, data directories, indicators, and research that is spatially-referenced for geographic display and analysis.

10) Establish an ecological risk communication working group to define place-based assessment; identify economic, public health, and environmental benefits of place-based assessment; identify risks associated with assessments that are not place-based; identify audiences; develop a communications strategy including products and action steps that allows various sectors to participate in preserving New Jersey systems.

11) Couple communication with on-going programs and initiatives, including research, to provide public with precise information for decisionmaking and participation.

Chapter V. Framework For Issue Resolution

A New Jersey Ecological Survey or Ecological Data Program could form a coordinated scientific basis for ecosystem management in New Jersey incorporating three interrelated elements: ecological data; environmental indicators; and ecosystem research. A schematic of this framework is depicted in Figure 1.

An environmental data resource guide could be developed including directories of researchers, bibliographies, data sources, data bases, data needs, data applications, specialized equipment, and training opportunities for New Jersey and adjacent state ecosystems. This can be linked to NJDEP's current Environmental Research Partnership initiative in which regional and issue-specific symposia are being developed for New Jersey ecosystem research.

Some environmental indicators have been developed for New Jersey, and research for developing others are currently underway. Further expansion into a coordinated statewide indicator program, would build upon current expertise in New Jersey.

New Jersey has numerous scientists conducting environmental research through the auspices of various governmental, nongovernmental, academic, and private organizations. NJDEP's active

environmental research programs can continue to maintain, facilitate, and expand coordinated research with scientists knowledgeable about New Jersey ecosystems. One mechanism for such coordination might be through the Environmental Research Partnership initiative.

Data, indicators, and research elements could be spatially-referenced to allow for geographic display and analysis whenever possible. As the NJGIS is made available to the public (via local libraries, watershed associations, non-governmental organizations, etc.), students, citizens, businesses, etc. will be able to access environmental information for a particular geographic location in New Jersey.

These elements could be coupled to a risk communication, outreach, and public participation strategy that includes various partners from government, nongovernmental organizations, academia, business and industry, environmental groups, etc.

The three survey or data elements could form the basis for a State of New Jersey's Environment Report on a periodic cycle. A State of New Jersey's Environment Report could include both ecological and human health components. Such a report could assist New Jersey in determining how effective its regulatory programs are in protecting ecological and human health. It could also enable New Jersey to identify what we know about our environment, how our environment has changed over time, gaps in our knowledge, and provide input for the development of data and research priorities. Such a report could also assist in coordinating priorities for regulatory, conservation, monitoring, and research program adjustments within New Jersey and neighboring states.

Many of the pieces to such a report already exist throughout the various program areas. Therefore, this umbrella framework could be achieved by linking existing efforts via a central liaison or coordination mechanism.

II. Ecological Assessment/Ecosystem Management

A. Introduction and Concepts

Despite some statistics which might indicate otherwise, New Jersey has remarkable plant and animal species diversity. Here are the statistics: New Jersey is only 7,509 square miles in size (the fifth smallest land area of the fifty United States); is the most densely populated state in the United States (with more persons per square mile than India, Japan, or the Netherlands); is a manufacturing center (leading the nation in pharmaceuticals and second in chemicals); and has more miles of highway per square mile than any other state in the United States (Collins and Anderson 1994).

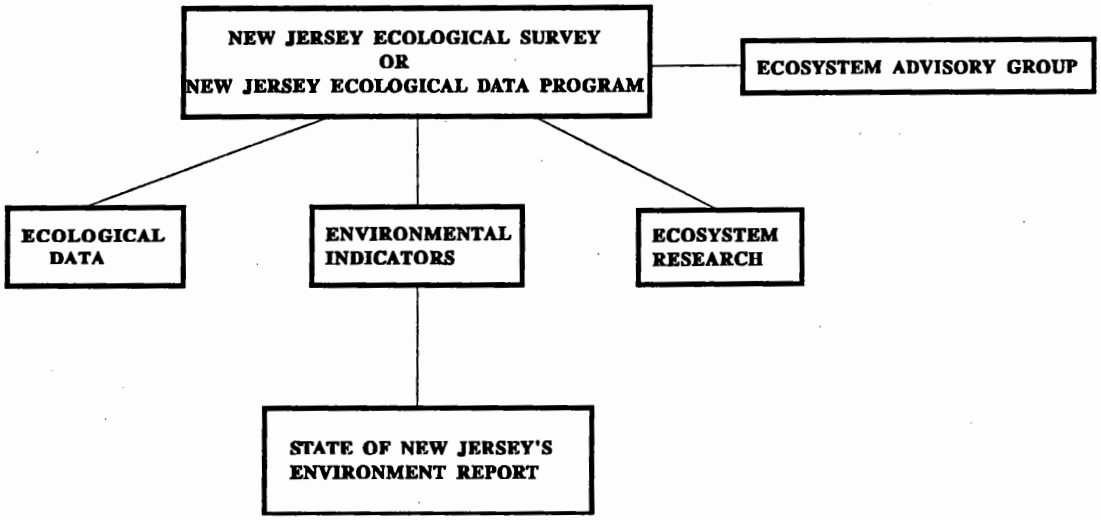
Still, amid all of this development, there are over 500,000 acres of State-designated open space in New Jersey (including 300,000 acres of state parks, forests, and recreation areas) and over 85,000 acres of Federally-designated open space in New Jersey (recreation areas, wildlife refuges, historic sites, etc.) (Thornton 1995; Coutros 1994). Taken together, approximately 12% of New Jersey's acreage is designated as open space by the State and Federal governments. These figures do not include local, county, or privately held open spaces. In addition, approximately 18% of New Jersey's land is used for agriculture (Collins and Anderson 1994).

New Jersey's landscape ranges from the Northwestern ridgetops to the hardwood and evergreen forests in the Northern Piedmont to the Middle Atlantic Coastal Plain in the Southeastern region (Collins and Anderson 1994; NJDEPE 1994b). In addition, New Jersey contains more than 2,600 known plant species, 335 known resident vertebrate species, and countless invertebrate and microbial species (TNC 1994; Breden 1995). Unfortunately, over 600 (23%) of these plant species, 128 (38%) vertebrate species, 183 invertebrates, and 38 natural communities are rare or endangered in New Jersey (TNC 1994; Breden 1995).

Precisely because New Jersey contains high species diversity, the anthropogenic environmental stresses associated with such a high population density and industrial base, as well as habitats of critical importance to more than a million migrating birds, it is important that ecosystem assessment and its associated management be scientifically grounded, thorough, and thoughtful.

This review of ecoassessment emphasizes risk from a toxicologic environmental input approach usually applied by programs bound by pollutant-specific

FIGURE 1.
PROPOSED NEW JERSEY ECOSYSTEM MANAGEMENT FRAMEWORK



regulations. A more holistic approach is usually taken by natural and historic resource protection programs focusing on habitat, biodiversity, rare and threatened species protection, etc. Many of the concepts described in the General Methodology, Environmental Indicators, and other sections, however, can be applied to ecosystem assessment overall, regardless of the approach.

The objectives of this research have been to: examine the various approaches for assessing ecological effects; identify outstanding issues; and develop a protocol for addressing these outstanding issues.

Future research should examine the holistic approach further; examine overall where methods from the toxicologic and holistic approaches can be knitted together to establish an ideal paradigm for ecological assessment; and examine how such a paradigm might be useful for improving individual program assessments and needs.

Ecological assessments of various types have been conducted over the years to address a variety of environmental concerns. These evaluations are very often synoptic assessments; scientists ascertain what the status of the system is at a particular point in time and use that information for determining what the impact of a proposed activity could be or what the impact of something such as a contaminant release has been. Ecological risk assessment is a form of ecological assessment that attempts to quantify the magnitude and probability of adverse effects from human activities or natural disasters to the nonhuman environment.

What distinguishes ecological risk assessment from ecological assessment in general are several things: 1) the somewhat erroneous perception that ecorisk is a new concept meant to finally address the nonhuman ecological parameters at risk from environmental exposures; 2) the search for a uniform approach to conducting an ecological risk assessment, much in the same way that there is a uniform set of guidelines for conducting human health risk assessment; and 3) the predictive nature of the assessment in which models for estimating exposure may be applied in an attempt to quantify or approximate risk.

Ecological assessment is not a new idea, but it is a concept that has recently received renewed attention. In the United States, our more recent past has emphasized human health risk assessment related to environmental exposure. Legislation calling for the protection of human health and the environment did not ignore ecological effects, but regulatory programs requiring risk assessment focused resources on protection of human health. The spectrum has now come full circle with the realization that ecological risk assessment is as important as the anthropocentric focus of human health assessment, that ecosystems are complex webs of interrelated organism structure

and function, and that shorter term adverse effects on nonhuman organisms or systems may ultimately have devastating consequences for humans, as well.

Therefore, developing a more formalized structure for conducting ecological risk assessment has lagged behind that of human health risk assessment. Given the complexity of ecosystems and our relative dearth of knowledge about how they function and the effects of contaminants on so many species (as compared to our knowledge of human health effects), as well as our renewed consciousness about the need for addressing ecological effects, it is not surprising that the field of ecorisk assessment is rapidly evolving.

Ecological assessment and ecological risk assessment tend to be terms that are tossed about together. In some circles, ecorisk assessment is perceived as a formal process to ascertain risk at a specific site defined by specific boundaries. Regardless of the semantics, a qualitative or quantitative ecological assessment can provide important data to contribute to ecosystem management, even if the intent of the assessment is very limited by site boundaries.

Ecosystem assessment should be an independent evaluation of natural resources and impacts upon them using a multi-media systems approach incorporating biotic and abiotic data. The objective scientific assessment should be used to formulate ecosystem management approaches. Usually, management approaches also consider institutional and societal constraints. Yet, because ecosystems are complex webs of interrelated organisms, ecological assessments should be designed to provide scientifically sound answers to system-wide management concerns.

What makes ecological assessment such a challenge is that unlike human health effects, ecosystem effects can occur at the organism, population, community or ecosystem level. Defining ecosystem health or integrity is a critical concept. Can we define ecological health with its human health corollary as the absence of disease (Schaeffer et al. 1988)? Is ecological health correlated to "wellness" or the ability to resist disease or injury? Is it system integrity? Can we define an acceptable condition or threshold for a particular ecological parameter, i.e., what effect or endpoint are we trying to measure? These are all questions to which answers are being developed as ecological assessment evolves.

Human health risk assessment has developed into a somewhat formalized process of ascertaining the risk or incidence of human death or disease from an exposure. Like human health assessment, ecological assessment can be a tool for protecting and managing the environment. In contrast, the ecological assessment process is less polished at the current time, and dependent not only upon endpoint definition, but also upon appropriate methods for endpoint measurement.

> Difficulties in predicting ecological responses and recovery from anthropogenic disturbances derive from several factors. First, there are a wide array of types of ecosystems that are of concern to humans. Second, unlike the human organism, which is tightly organized and highly homeostatic, ecosystems are only loosely interconnected, with many different mechanisms for positive and negative feedbacks and relatively low levels of homeostasis. Third, there is no universal, simple measure (endpoint) of ecological health that is analogous to those measures used in human health risk assessment. (EPA 1992b)

In addition, human health risk assessment usually addresses a spectrum of disease resulting from a range of exposures to a toxicant. Inherent in this analysis may be identification of a subset of the population that would be particularly sensitive to exposure such as children, women of reproductive age, elderly people, persons with specific risk factors, etc. Calabrese and Baldwin (1993) describe this as assessing risk to individuals in the population, rather than overall population effects. They argue that sensitive subpopulations are usually not considered in ecological assessment unless the assessment accounts for sensitive life-stage evaluation or endangered species (Calabrese and Baldwin 1993).

Suter (1993) provides a good basis for understanding some very fundamental distinctions between ecological assessment and human health assessment. These distinctions are set forth in Table 1.

Similar to human health assessment, the extent to which an ecological effect must be quantified varies and is often dependent upon the confines of a legislative or regulatory directive. Suter (1993) describes a measurement progression for ascertaining a desired endpoint or policy goal that generally follows the scheme:

Environmental Indicator	→	Measurement Endpoint	→	Assessment Endpoint	→	Hazard/Policy Goal

Table 1. Ecological Assessment Considerations[1]

- Nonhuman exposure may occur through routes not pertinent to humans such as preening feathers, respiring water, drinking from contaminated ambient water sources, or root uptake.

- Nonhuman species may be more sensitive than humans to certain chemicals, possibly for mechanistic reasons that are not applicable to humans. Examples include eggshell thinning by DDT or stomatal closure in plants exposed to SO_2.

- Ecological effects can occur at various system levels ranging from organism to population to community to ecosystem, as opposed to human health effects which occur at the organism level. An example would include eutrophication of a lake ecosystem by nutrient loading.

- Nonhuman species may be more highly exposed to compounds than humans even when the route of exposure is the same because of inherent differences in behavior and habitat. For example, humans may vary their diet while a shorebird might eat only fish for each meal, consuming the entire fish, rather than just relatively uncontaminated muscle tissue.

- Dose per body mass in birds and mammals may be greater than humans because their higher metabolic rates would require greater consumption of food and water, and greater respiration of air than humans.

- Nonhuman organisms may be at greater risk for exposure to biologically significant doses of compounds designed to harm certain target species of organisms that are physiologically and ecologically similar to "innocent" non-target organisms.

- Secondary effects of chemical exposure may be more significant for nonhuman organisms that do not have access to alternate sources necessary to sustain life. For example, a compound may not be biologically problematic for a nonhuman organism, yet it might destroy organism food supply or physical habitat. Unlike humans, which have greater access to alternate habitats and food supplies, nonhuman organisms may not be able to adjust.

[1]From Suter (1993).

Measurements are often made through toxicity tests or monitoring, either in the field or the laboratory. These tests or monitoring points are environmental indicators.

The formal, usually quantitative, expression of results of testing or monitoring is a measurement endpoint. An example might be determination of the LC_{50} in a fish species. In addition to environmental indicators, multi-dimensional descriptive models, such as those which describe concentration-response functions, are very important measurement endpoints for assessing ecological risk (Suter 1993).

According to Suter (1993), assessment endpoints refer to population and ecosystem characteristics defined over large scales. Since assessment endpoints are frequently predictive in nature, they often require extrapolation from field or laboratory measurement endpoints.

An example from Suter (1993): an ecological assessment to ascertain the hazard of an agricultural pesticide with the policy goal of no unacceptable reductions in avian populations. Working backwards, an assessment endpoint might be determination of rates of decline of bird populations within the region of use. One measurement endpoint could include number of bird carcasses per hectare that would be determined based on the environmental indicator of avian field toxicity. Another measurement endpoint could be an avian species LC_{50} as determined by that species' laboratory toxicity (i.e., environmental indicator). These pieces of data would be used to determine whether that pesticide contributed to changes in bird populations.

Suter (1993) describes two primary methods of assessment: physical methods and quantitative methods. Physical methods can be material representations of some system, such as a laboratory study or an analog study based on an existing environmental disturbance.

Quantitative methods include statistical or mechanistic models. Statistical models are used to derive generalizations from experimental or observational data. In risk assessment, statistical models are used for hypothesis testing, description, and extrapolation (Suter 1993). Suter (1993) notes that hypothesis testing is not often appropriate for environmental science applications because comparison of contaminated sites to control sites for example, may not show statistically significant differences, yet adverse effects could indeed result from conditions at the sites of concern. An example cited by Suter (1993) is that of comparing background soil concentrations to contaminant soil concentrations that might not be different statistically, yet the

contaminant soil concentrations would be biologically significant in terms of toxicity.

Statistical extrapolation of data includes range extrapolations and data extrapolations. Extrapolation of data from points within a data set to a range outside of a data set (as in low-dose extrapolation) is range extrapolation. Range extrapolation assumes the relationship between dependent and independent variables outside the data set is the same as for those variables within the data set.

Data extrapolation transfers relationships from one set of conditions to another, as in extrapolation of rodent data to human systems or data from laboratory conditions to field conditions. Data extrapolation assumes the systems between which extrapolation occurs respond similarly to the conditions under assessment. (Suter 1993)

Statistical models can also be used to describe data sets. These descriptive results are often used to generate hypotheses which can then be tested experimentally.

Unlike statistical models which are primarily mathematical, "mechanistic models describe quantitatively the relationship between some phenomenon and its underlying causes" based on independent measurement and operational definition (Suter 1993). Mechanistic models are fate and effects models. They can include contaminant dispersion transport models or toxicokinetic models which relate uptake and effects of contaminants to their physical and chemical properties. Mechanistic models can be used for predicting effects of very complicated exposure scenarios and for integrating data sets collected under varied conditions.

B. Ecological Risk Assessment: General Methodology

The human health risk assessment process includes the four-step approach published by the National Academy of Sciences (NAS) in 1983: 1) hazard identification; 2) dose-response assessment; 3) exposure assessment; and 4) risk characterization. Based on the inherent distinctions between assessing risk to human health and risk to ecological systems as already outlined in Table 1, the standard NAS paradigm is not completely apropos for ecological risk assessment.

A general approach for ecorisk analysis has been developed:

1. Problem Formulation

2. Analysis
   a. Stressor-Response Analysis
   b. Exposure Analysis

3. Risk Characterization (EPA 1992a).

Stressor response analysis somewhat parallels dose-response assessment in human health risk assessment. Often, the objective of a stressor response analysis is to derive acceptable exposure levels for terrestrial and aquatic species as well as acceptable toxicant concentrations in water, soils, and sediments. For the most part, air concentrations have not been considered much in ecological assessment except for assessments pertaining to climate change. Stressor response values can include maximum acceptable tissue concentrations, maximum acceptable toxicant concentrations, toxicity reference values, apparent effects thresholds, no-observed-adverse-effects levels, water quality criteria, or other types of values.

It is important to recognize that chemical toxicity is based on both inherent toxicity and dose. However, depending upon the conditions and the compound of interest, one or the other parameter (inherent toxicity or net concentration) may drive the response in a particular organism.

For instance, a compound with a high-octanol water partition coefficient (K_{ow}) may require a residence time longer than 96 hours (standard acute LC_{50} bioassay) before absorption and subsequent demonstration of a toxic response. Comparing K_{ow}s for various organisms using this type of assay is more a comparison of compound quantity than inherent toxicity. Standardization of exposures to account for internal concentration, rather than environmental exposure concentrations, would be a more realistic method for toxicity comparisons, although such data are difficult to find in the literature. Therefore, when evaluating toxicity data, especially that which incorporates a model, it is important to recognize the pharmacokinetic (absorption, distribution and elimination processes) and pharmacodynamic (receptor response) variability in different classes of

chemical compounds. It is also important to recognize the potential for bioconcentration from dietary and ambient sources and bioaccumulation through food chain transfer. (Calabrese and Baldwin 1993).

Exposure analysis is a method to quantify the amount of contaminant in the receptor organism. Characterizing exposure includes data on exposure sources, species consumption patterns, species range, body weight, etc. Once exposure is quantified, this value is compared to the stressor response value to characterize risk.

The most reliable exposure analysis would obviously be one in which actual data for the agent of concern in the species of concern at the site of concern were available. Since this is usually not the case, exposure analysis is often a combination of measured values and assumptions as input parameters to exposure models.

Data on characteristics of the species of concern can be gleaned from the literature. EPA's sourcebook of exposure data for wildlife (birds, mammals, reptiles, and amphibians) can be useful for estimating wildlife contaminant intakes for screening level purposes; however, original literature should be reviewed to supplement the exposure factors handbook (EPA 1993e; 1993f).

The EPA Wildlife Exposure Factors Handbook includes: exposure factors for selected vertebrate species; allometric models for extrapolation of data between species; and algorithms for calculating exposure. Species-specific exposure factors include: normalizing factors (body weight, growth rate, metabolic rate); exposure route data (food ingestion rates, dietary composition, water ingestion rates, soil and sediment intakes, inhalation rates, dermal exposure, etc.); population dynamics (range, territory, social organization, population density, annual fecundity and mortality); and seasonal activity (EPA 1993e).

The handbook does not contain information on fish, aquatic or terrestrial invertebrates, or chemical-specific data such as bioavailability factors (EPA 1993e). Geographic variability of species and the presence of native species not included in the EPA handbook, indicate the importance of ascertaining as much local information as possible for site-specific assessments. Obviously, other data sources to estimate exposure in other vertebrates, invertebrates, and vegetative species should be consulted as they pertain to the assessment. One sourcebook on vegetative species distribution specific to New Jersey is that of Collins and Anderson (1993).

In concert with data on species of concern, environmental fate modeling predicts exposure concentrations of materials throughout the environment including persistence, distribution, magnitude of exposure, and dominant fate processes.

Various models predict contaminant concentration in receptors including one and two-compartment models, bioaccumulation models (which incorporate environmental contaminant concentrations with dietary concentrations), food-chain transfer models (which go beyond bioaccumulation models by looking at contaminant concentrations through various trophic levels i.e., examine bioconcentration for trophic level plus chemical contribution from feeding on next lower level).

However, assessors should be aware that under certain conditions compounds of concern can undergo chemical transformations such as speciation and complexation reactions. For example, inorganic mercury in sediments can undergo biotransformation to methyl mercury via bacterial transformation. Upon ingestion of mercury-contaminated fish, piscivorous wildlife respond with adverse reproductive outcomes.

Limits to models include their lack of data on sediment-water interactions and lack of model verification (Calabrese and Baldwin 1993). Also, uncertainties in input parameters may lead to overestimation or underestimation of contaminant concentrations. For example, in food chain transfer models, errors in estimates of assimilation efficiency of the agent from food (i.e., absorption) or in the elimination of the agent from an organism based upon depuration and growth rate (which reduces the relative concentration of the contaminant in the organism) can significantly effect model predictions of contaminant concentration.

Evaluating stressor-response in concert with exposure analysis is the basis of risk characterization. The assessor needs to be aware of the limits of the data, how they were derived, and the assumptions that go

into applying data to models when characterizing risk.

Data acquisition, verification, and monitoring can occur during any of the ecorisk steps. Risk characterization is followed by risk management and further data acquisition, verification, and monitoring.

The general framework cited above may be the most practical organizational structure for ecological assessment possible at the current time. One must understand the complex web of variables that form the context of an ecological assessment. These include: the stresses which are being assessed (anthropogenic chemical stresses such as toxics, pesticides, nutrients or non-chemical stresses such as climate change, biological depletion, habitat alteration); the level of biological organization at which the assessment is being conducted (organism, population, community, system); the ecosystem type or classification (freshwater, marine/estuarine, terrestrial, wetlands); the spatial area over which the assessment is concerned; and the time scale over which the assessment will endure (EPA 1992b). It is because of this complexity that, like human health assessment at times, determining causation in ecological assessment may rely more in identifying trends and distinguishing patterns of response in species or activity of chemical compounds than in explicit cause and effect relationships.

A fundamental consideration in problem formulation is identification of ecosystem scale; i.e., is the system being assessed on the biosphere, regional or localized level? Is it possible or desirable to define a universal system for ecosystem classification? Formulation of the problem and assessment design may be ultimately bound by scientific, regulatory, policy, and economic constraints; however, can an idealized classification scheme be developed?

For example, ecosystems have been traditionally classified based on landscape characteristics as lake, stream, forest, grassland, or freshwater; however, a classification scheme based on system function has been proposed for consideration in ecological assessments (EPA 1992b). This alternative would specify an ecosystem into one of 64 categories based on continuums within six functional areas as described below. Inherent in this classification scheme is a function-by-function assessment of the effects of stresses within an ecological system.

The six ecosystem classification areas proposed by EPA workshop participants (EPA 1992b) include:

1. Stresses via gaseous/terrestrial phases versus aquatic phases in which systems would be classified along a continuum from pelagic to upland.

2. Significance of energy sources as they relate to the effects of stresses on primary producers. This would follow a continuum from indigenous energy sources to non-native energy sources.

3. Degree of energy degradation to heat following a continuum from consumers of living resources (biophagous) to consumers of nonliving resources (saprophagous). This dichotomy signifies the relationship between stresses and consumers based on previous trophic level productivity (biophagous) and stresses and consumers not directly affected by prior trophic level productivity.

4. Evaluation of the ecosystem as an energy source (passage to an adjacent system) versus an energy sink (storage). The effects of stressors on energy potential is particularly significant with respect to coupled ecosystems.

5. Degree of nutrient recycling versus nutrient recycling and replacement (flux); what aspect of nutrient cycling in a system might be affected by the stressor of interest?

6. Frequency and duration of system disturbance with respect to ability to recover. Ecosystems in which catastrophic disturbances are more common have evolved such that native species are better able to recover than systems with less environmental variation.

The EPA workshop members acknowledged that this proposed classification approach requires further development (EPA 1992b). It is however, a logical approach that may be very useful for ascertaining ecological effects that would not be observed by a classification system based solely on landscape-type parameters.

Other important considerations in assessing ecological risk include evaluation of impacts with respect to intensity of effects; nature of effects; degree of certainty of projections; and time scale for recovery (Harwell and Kelly 1986).

Nature of effects can include considerations such as: community structure (diversity, trophic structure changes); ecological processes (primary production, nutrient cycling rates, decomposition rates); species of direct importance to humans (aesthetic and or economic, endangered or threatened); and potential for the ecosystem to function as a vector for human exposure (Harwell and Kelly 1986).

Technical issues in ecological assessment which require further resolve have been identified by participants in an EPA workshop on ecological assessment (EPA 1992b):

1) Should ecosystem classification be based on traditional landscape distinctions? on function as described above? or perhaps some combination of factors?

2) Should ecosystems be further classified by frequency of natural disturbances, i.e., where material and energy processing rates are similar?

3) How do we classify managed ecosystems such as farmland?

4) How do we account for natural variability in assessment of effects?

5) How do we determine what effects are acceptable or unacceptable?

In all likelihood, these and many other questions will be debated for some time. Some general consensus within the scientific community may develop as the art of ecological assessment evolves.

The following sections describe various assessment paradigms based for the most part on time and spatial parameters. These include: predictive and retrospective assessments, as well as more specific types of assessments (such as those for genetically engineered organisms and regional assessments).

It is important to note that the individual subsections presented below in the discussion on "predictive" ecorisk assessment are not restricted to "predictive" ecorisk assessment. Many of the ecorisk parameters described below are applicable to retrospective or other types of ecorisk assessments. In a sense, all risk assessments are predictive in that we are estimating or predicting risk whether we are concerned with an activity or event that has not occurred or with one that has occurred. Therefore, assessments which are either predictive or retrospective in nature, will contain elements of the subsections described below (endpoint selection, habitat description, source terms, exposure assessment, effects assessment, risk characterization, etc.) These sections are included under the heading of "Predictive Ecological Risk Assessment" mostly for the sake of context as well as convenience.

Additionally, operational considerations are discussed separately, including specific information on some of the mechanics applied to various types of ecoassessments in New Jersey. Finally, a section on environmental indicators is presented.

C. Predictive Ecological Risk Assessment

Predictive ecorisk estimates risk associated with a new chemical, a new source, or a potential accident. Predictive ecorisk parallels the NAS paradigm in some respects. Endpoint selection (parallels hazard identification) includes diverse endpoints (population affected) or hierarchical endpoints (organizational level affected). Dose-response and exposure assessment steps in human health assessment are paralleled by the effects assessment (use effects models or predictive models to ascertain effects of agent on endpoint over time) and habitat description (obtain source terms such as release rates, disturbance rate, temporal and spatial information), respectively. Integration of the source terms and habitat description form the exposure assessment. Next, integration of the effects assessment and the exposure assessment is the risk characterization.

1. Endpoint Selection

Suter (1993) suggests a series of tools to assist in endpoint selection. These aids are intended to identify appropriate endpoints based on technical grounds, especially to prevent omission of an important scientific aspect of an assessment restricted by legislative or regulatory constraints. These include:

1) develop a matrix of actions being assessed and

environmental receptors potentially affected with a possible scoring scheme for intensity of exposure and sensitivity of receptor;

2) perform receptor identification that would be a preliminary qualitative or quantitative assessment of media predicted to be most contaminated and what receptors (communities, trophic groups, populations, life stages, etc.) would be most exposed to those media;

3) develop trees of causal linkages between emissions and environmental components; and

4) review existing data to determine species or process sensitivity to contaminants or similar contaminants (Suter 1993).

Once hazards have been identified, Suter (1993) explains that each assessment endpoint must be "operationally defined" in terms of what characteristic of a specific endpoint will be quantitatively evaluated. An assessment may define an effect in terms of a probability of occurrence or prediction that there is a risk of an effect.

Measurement endpoints are often used as assessment endpoints, but Suter (1993) suggests that it is preferable to "choose measurement endpoints for their relationship to a defined assessment endpoint", rather than as de facto assessment endpoints. Suter (1993) gives the example of using an LC_{50} for an appropriate species of fish as a predictor of fish kills from exposure to a specific compound.

2. Environmental/Habitat Description

Ecological assessment requires definition of the environment in which exposure is to be considered. The environment may be a specific site that actually exists or one that is conceptualized. Characterization of the habitat may be limited by regulatory boundaries (such as surface soil on or directly adjacent to a site) or driven by intrinsic assessment goals (such as defining the boundaries as those areas in which concentrations exceed a biologically significant threshold). Suter (1993) explains that the latter case may require identification of boundaries through an iterative process.

3. Source Terms

"Source terms are estimates of the rate and the spatial and temporal pattern of release of a chemical from a source or set of sources" (Suter 1993). These values may be obtained from direct emissions measurements or from indirect emissions (such as leachate). Suter (1993) explains that in many instances, source terms are the greatest areas of uncertainty in ecological assessment.

4. Exposure Assessment

Predictive exposure assessment translates source term estimates into estimates of receptor dose. Models that simulate contaminant transport (partitioning and volatilization), transformation (hydrolysis, photolysis, biotransformation), degradation, and partitioning are used to predict pollutant concentration in the receiving environment. Characteristics of the environment (habitat), as well as those of the pollutant are significant input parameters to exposure assessment models.

Suter (1993) points out that exposure assessment requires knowledge of the organisms within a system, as well as pollutant characteristics. For instance, organism exposure may occur via more than one route of exposure or via more than one source of exposure. In addition, some organisms, assumed to be subject to exposure, may in fact be repelled by a contaminant, or others may be especially attracted to it. Therefore, when possible, exposure assessment should include background, incremental, and total exposure, to aid in risk management decisions (Suter 1993).

5. Effects Assessment

Evaluating the relationship between contaminant exposure and potential hazard to assessment endpoints is the effects assessment. One way to evaluate this relationship is to extrapolate effects models from retrospective assessments of field sites with the same contaminants.

A more common approach to effects assessment is to use toxicity data or conduct toxicity tests to derive an exposure-response model. The exposure-response model is then related to the assessment endpoints to

derive an effects model. The effects models are then extrapolated to appropriate population and ecosystem processes to derive a "function relating the level of effects on the assessment endpoint to exposure" (Suter 1993).

6. Risk Characterization

Risk characterization integrates the exposure assessment and effects assessment to ascertain risk and, if possible, its magnitude.

McCarty and Mackay (1993) reiterate a logical paradigm for ecorisk assessment with the observation that this paradigm is improved and "feasible" because of improvements in quantitative methods. The ecorisk paradigm links chemical fate modeling (used to predict a range of contaminant concentrations in the ecosystem) with food chain/web modeling (used to estimate biologic concentrations in target organism tissue). Tissue concentrations are then compared to acute and chronic toxicity data to estimate effects.

What distinguishes their approach from the more conventional ecorisk methodology is their idea of a CBR or critical body residue method for determining the relationship between exposure and toxicity (McCarty and Mackay 1993). McCarty and Mackay (1993) argue that critical body residue comparison to tissue concentrations is more advantageous than the more common method of relating exposure concentrations with environmental concentrations known to cause toxicity (e.g., relating ambient water concentrations with LC_{50s}). Some advantages include consideration of: bioavailability, accumulation kinetics, metabolic effects, and mixture effects (McCarty and Mackay 1993).

McCarty and Mackay (1993) emphasize that more recently developed environmental fate models are capable of a mass balance approach for estimating contaminant discharge rates via transport and transformation mechanisms throughout an ecosystem. Quantification of the tendency for a contaminant to transfer across media (known as fugacity) can be coupled with a trophic transfer model which allows for calculation of contaminant tissue residue levels in target organisms. Instead of using surrogate measures of exposure concentrations to indicate target site toxicity levels, McCarty and Mackay (1993) argue that body residue levels are more accurate measures of toxicity level.

The authors describe various models that demonstrate ways to predict chronic toxicity based upon critical body residue which can incorporate modifying factors such as metabolic transformation of a contaminant; however, they do cite the need for better understanding of the residue:effect relationship (McCarty and Mackay 1993). One proposal is to develop a single bioassay protocol that would be expanded to include simultaneous analysis of exposure levels and tissue residue. McCarty and Mackay (1993) argue that enhanced bioassay design would increase knowledge concerning bioavailability, transformation, and ultimately decrease uncertainty in assessments by reducing reliance on safety factors in extrapolating from literature values based on current protocols.

It appears that this approach is most applicable in situations where the toxicity results from a fairly constant body residue. The authors do recognize that the critical body residue approach may not be applicable for scenarios in which exposure is acute or intermittent and death or irreversible injury results; hence, the environmental exposure approach is better.

7. Effects Assessment and Risk Characterization At Various Ecological Levels

Ecological risk assessment is a relatively new field. Although ecological assessment might be able to ascertain that one or more variables have a causal relationship to a particular effect, it may be difficult to quantify or predict the risk of an adverse ecological effect based on such factors. Very often, ecological risk assessment is a discrete evaluation: risk is predicted to occur or not occur at a specified level of exposure, especially at the organism level.

Numerous models have been developed and continue to be developed for predicting ecological effects; however, as is always the case in risk assessment, qualitative evaluation and scientific judgment are critical in the utilization of models. Care must be taken to insure application of biologically-appropriate models. For example, a population model described by Suter (1993), which employs discrete variables, rather than continuous variables, should be applied to species with seasonal reproductive cycles and not to species with continuous reproductive cycles.

Effects can be ascertained at the organism, population, or system level; however, as Odum (1971) explains, there are no distinct lines or breaks in the spectrum from genes to cells to organs to organisms to populations to communities to ecosystems; rather, there is a continuum in which the levels are interdependent upon one another. One might argue, therefore, that an assessment at any one level is merely an artificial boundary necessary for some point of reference. It is precisely because of these quasi-boundaries that sensitivity in recognizing an effect may be lacking and it is important that assessments be designed to examine multiple levels of biological organization whenever possible.

At the organism level, an assessment can provide the quantitative value for exposure at which such an organism is assumed to be protected from toxicity. Suter (1993) points out that development of an exposure limit beyond the organism level, however, accepts some level of toxicity. Essentially, an assessment beyond the organism level (population, community, ecosystem) may attempt to quantify a limit of exposure that would not affect population, community or ecosystem viability. In addition, effects at the population, community, or ecosystem level may require complex models which attempt to predict effects at these levels. Assessments that are based on only one level may be inconclusive if effects indeed occur at a different organizational level. Again, study design is critical as negative effects when assessing population level dynamics, for instance, may not preclude effects at the ecosystem level.

Organism Level Effects

Organism level effects can include algae, aquatic plants, fish, aquatic invertebrates, terrestrial plants, terrestrial invertebrates, and terrestrial vertebrates (Suter 1993). Organism level effects can be ascertained through laboratory or field toxicity tests and histopathology. Specific examples might be sediment bioassays, clam burrowing behavior, phytoplankton growth, tissue/tumor/lesion examination (EPA 1993b).

Organism level responses alone are subject to various factors of uncertainty: test species, test endpoints, and extrapolation of test data to other species. Each of these are described further below.

Taxonomic differences between species means that extrapolation of data specific from one species to another will inherently involve some level of uncertainty. Suter (1993) cites evidence to support taxonomic extrapolation to some degree, noting that extrapolations between species within the same family can be made with better certainty than extrapolations between species in broader taxonomic groups (i.e., orders, classes, or phyla). Suter (1993) is careful to point out that if those characteristics which determine species sensitivity to an exposure source are "highly modified by the adaptions to the specific environments and ecological roles that distinguish species and subspecies of that taxonomic group, closely related species will be as dissimilar as distantly related species". An example cited by Suter (1993) is the lack of clear taxonomic response to air pollutants by plant species because of adaption to humidity or aridity depending upon geographic region. In addition, Suter (1993) points to research showing prediction of whole taxa based on a test species may be limited. For example, regressions of aquatic animal test data found that deviations from the analysis increased with increasing toxicity (Thurston et al. 1985 and Holcombe et al. 1988 as cited by Suter 1993).

One common taxonomic extrapolation method is to estimate risk for a particular species based upon data from another. Another extrapolation method is to estimate sensitivity among members of a population using test data for one or more species. Referring to the latter as the generic taxonomic approach, Suter (1993) explains such an approach assumes once the most sensitive species are identified in a community and protected for, the other members of that community will be protected.

Test endpoints may not always correlate well with the exposure scenario of concern. For example, ascertainment of fish toxicity may be most useful quantitatively if constant exposure conditions are maintained; however, under some scenarios (such as exposure to a spill) assessment of fish toxicity would be more valid if exposure conditions simulated the spill, rather than constant exposure.(Suter 1993).

Dose-scaling is used to reduce uncertainty when extrapolating data from one species to another. Dose-scaling can include dose per unit mass (mg/kg -- which assumes effects occur at a certain organism concentration); dose per unit surface area (mg/m^2 --

which assumes thermoregulated metabolic demands such as oxygen consumption, blood volume, renal function and water requirements are constant when scaled to surface area); and life expectancy (which assumes response rates for effects such as cancer induction are proportional to life expectancy) (Suter 1993).

Additional approaches to extrapolation between species are mentioned by Suter (1993) as possible alternatives to taxonomic or physical dose scaling methods. Such alternative extrapolation techniques might include comparison of species based upon food source and trophic level (e.g., sensitivity of herbivores within a trophic group) or comparison based upon guilds of organisms which utilize a common resource (e.g., sensitivity of bird species based upon dietary source: nectar, seeds, vertebrates, etc.). Other ideas Suter (1993) mentions include predictive models based on species adaptation (some examples: plant transpiration rates, plant height as related to air pollutant exposure, species longevity and therefore susceptibility to chronic pollutants) or biochemical traits.

An important limitation in taxonomic-based species extrapolation is the emphasis on "single life stage and test condition, implicitly ignoring the possibility that intertaxa relationships may not be constant" (Suter 1993). For example, Suter (1993) cites data that show sensitivity of fish eggs to be more variable than the sensitivity of fish larvae, making a strong argument for consideration of other factors in addition to taxonomic ones for interspecies extrapolation.

Other variables which can influence assessment of organism level effects include life stage and size (is there a critical exposure period or most sensitive exposure stage?); temporal dynamics of exposure (including daily and seasonal fluctuations; reaching contaminant equilibrium; or loss of contaminant over time in an experimental system); exposure tolerance via acclimation (physiological and biochemical adjustments) or exposure tolerance via adaptation (genetic adjustments); routes of exposure; physical conditions (pH, temperature, salinity, hardness, etc.); severity of effects (i.e., are we missing effects because we only measure lethality?); and organism condition (Suter 1993). Complicating matters further is the issue of chemical mixtures and the ability to determine organism level effects for chemical mixtures.

Various models may be utilized for organism-level assessment when field or laboratory data are not available. Such models include: dose-response models, structure activity models, and effects extrapolation models. Using models, however, rather than actual measurement data, will increase assessment uncertainty.

Dose-response models are predictors of effects at varying exposure concentrations. Structure-activity models attempt to predict the behavior of a compound based upon some common physical or chemical property of that compound for which surrogate compound toxicity data are available. Effects extrapolation models are used to predict effects in a species of interest from the response in another species, lifestage, or test type.

Effects extrapolation models include factor models, statistical extrapolation models, and toxicodynamics models. Factor models incorporate various uncertainty, safety, or modifying factors to account for differences between the applied data set and the endpoints under assessment. Statistical extrapolation models are regression models which attempt to define the relationship between exposure and effect based on observed data. Toxicodynamic models assume effects occur at a certain internal concentration or activity and therefore consider temporal variation over time as well as metabolic and size differences. (Suter 1993)

Risk at the organism level can be characterized using a quotient method; a probabilistic method based on extrapolation (Analysis of Extrapolation Error); or simulation modeling using uncertainty analysis (Monte Carlo Analysis).

The quotient method relates expected environmental concentration to a toxic endpoint concentration. If the quotient of that ratio is less than a safety or uncertainty factor (or product of factors), the environmental exposure is considered safe. If the quotient exceeds the safety factor, the environmental exposure is not safe. Quotients that are not less than or greater than the safety factor imply further assessment is warranted; however, Suter (1993) points out that since such further assessment is often not possible, if the quotient is not less than the safety factor, the release is not considered safe.

Probability density functions plot distributions of an effect with respect to exposure concentration using models fitted to surrogate endpoint concentrations derived from laboratory data extrapolated to field conditions. This approach has been called Analysis of Extrapolation Error and requires a model that generates a statistical distribution of effects (Suter 1993). Risk using this approach is the probability that the expected environmental concentration will be greater than the surrogate endpoint concentration. Suter (1993) asserts that probability density functions offer a more complete portrayal of the assessor's understanding than the quotient/safety factor characterization and also do not rely on extremely conservative uncertainty factors.

Monte Carlo simulation is another risk characterization technique in which a distribution of output values can be used to determine the probability of any particular value given the uncertainty in the input parameters (Suter 1993).

Population Level Effects

As Suter (1993) explains, unlike organism level assessment, death or impairment to an individual organism is not significant when ascertaining population level effects; some level of effect is expected when deriving allowable exposure levels. Instead, population level effects center on abundance, production, and persistence of groups (populations) inferred from collective data for individual organisms as related to population theory (Suter 1993).

Population characteristics of importance can include: biomass (total number); rates of growth or decline; and composition based on age, sex, size or genotype (Suter 1993). These characteristics vary based upon inherent organism life processes (individual and life stage differences, birth and death processes, development processes, etc.), physical environmental processes, interactions with other organisms, and human activity (Suter 1993).

Population level effects can be ascertained through laboratory toxicity tests and field study. Examples of assessing population level effects include resident fish species survival, growth, reproduction, species occurrence, population density, and number of nests hatched (EPA 1993b).

Models accounting for inherent variability within a population (including environmental variability) can be utilized to predict probability of extinction of a population. Appropriate management strategies can then be developed to prevent extinction or overuse of a resource such as limitations on fishing.

Population level effects can be quantified using various models: estimates of reproductive potential, projection matrices, aggregated models, and individual-based models. Appropriate models must be used to reflect biological systems. For instance, models that employ discrete variables should be used for systems in which activity is discrete, such as seasonal reproduction. Similarly, models that employ continuous variables should be used for continuous activity, such as continuous reproduction. Additionally, as with any modeling approach, population dynamics models alone may not be adequate in ascertaining risk, especially if effects are at the ecosystem level.

Reproductive potential can be estimated given values for the fraction of organisms surviving from one age to the next and the average number of offspring of an organism at a given age. Such a model assumes that if left undisturbed a population will reach a stable age distribution that remains constant from generation to generation. Estimates of the rate of population change can be made from measurements of daily survival and reproduction. Alterations in the rate of population change upon chemical exposure in various organisms can indicate toxicity (Suter 1993).

Projection matrices can also be used for predicting the change in population abundance over time but in addition to age distribution, a projection matrix can be developed to include life stages. Suter (1993) explains that these models can be applicable for ecological assessment particularly because "plants and invertebrates have complex life cycles in which population dynamics are more strongly influenced by size and developmental stage than by age".

Suter (1993) also describes aggregate models in which organisms are grouped primarily based on age structure (examples include population size or parents and offspring).

Finally, Suter (1993) describes individual-based models. Unlike the previous types of population-level models which are assessments based on numeric quantification in the population (births, deaths, age

structure), individual-based models attempt to incorporate physiological, behavioral, or other biological properties which may partially account for these numeric differences in populations (Suter 1993). For instance, such a model might incorporate data for a number of individual organism parameters and their overall response be calculated for a distribution of exogenous environmental stressors as the sum of their individual responses to these stressors.

Community and Ecosystem Level Effects

Community and ecosystem level assessments may not only focus on direct effects to some property of an ecosystem, but also may be useful in complementing an assessment at another level, such as incorporating effects at the organism level to predict effects at the ecosystem level. Community and ecosystem level effects may include: direct effects on interactions between populations of different species (e.g., inability to avoid predators); indirect population effects because of effects on another related population (e.g., loss of prey species will indirectly effect abundance of predator); system structural changes (number of species or trophic level); and changes in system functional properties (e.g., changes in primary production) (Suter 1993).

Community and ecosystem level effects can be obtained through field studies or modeling or some combination of data sources as inputs to models. Community level assessments center on the interaction of population groups. Odum (1971) describes a community as an assemblage of populations living in a prescribed physical habitat that functions through coupled metabolic transformations.

Ecosystems are distinguished from communities in that they include both the abiotic and biotic components of a community. An ecosystem is defined as the organisms or community within an area "interacting with the physical environment so that a flow of energy leads to clearly defined trophic structure, biotic diversity, and material cycles (i.e., exchange of materials between living and nonliving parts) within the system" (Odum 1971).

Examples of assessing ecosystem effects would be forest succession models or wetland evaluation of function and value. A pond study on pesticide effects to an aquatic community would be an example of a community level assessment (EPA 1993).

Suter (1993) astutely points out that some scientists argue that true ecological assessments center on ecosystem level effects, as opposed to effects at the organism level based on toxicity tests or population level studies. In reality, however, there are significant barriers to conducting pure ecosystem level assessments. Suter (1993) cites the following hindrances to ecosystem level assessments:

- In general, ecosystem level tests are costly and time-consuming.

- Consensus is lacking about appropriate ecosystem level study design.

- In general, it is not known how variation in test conditions affects outcome at the ecosystem level.

- In general, it is not known how the range in test conditions compares to the range in exposed natural systems.

- Toxicity can be difficult to detect because of low replication and high variance among replicates.

- Functional redundancy in an ecosystem may not show system-level effects, yet, may mask organism or population level toxicity at low exposures.

Therefore, several population or organism level responses should be incorporated into an ecosystem level assessment to ensure that actual effects are not being missed. Even so, it is important to recognize that even population or organism level responses can miss effects because the necessity of using indicator species may mask effects in a non-indicator organism.

Scientists may differ as to the extent of an ecosystem boundary or habitat. As pointed out in Section II.B. (Ecological Risk Assessment: General Methodology), defining ecosystems by function (energy source, nutrient cycling, ability to recover, etc.) may be an alternative to landscape-based system definition.

The most logical approach for defining and then assessing a system might be to identify the system based upon a combination of landscape classification and function. This might require a number of iterations in which a system would be defined first by landscape classification and then refined based upon initial assessment of functional parameters.

As previously mentioned, landscape-based systems can include freshwater, marine, estuarine, terrestrial, wetlands, etc. while function-based systems can center on energy sources, energy potential, nutrient cycling, and the ability of a system to recover (EPA 1992b). Suter (1993) identifies general ecosystem assessment boundary categories based on a functional approach and their related models: energy; material cycling; composition; and stability.

Energy-based assessments may center upon the significance of environmental stressors with respect to rate of energy capture, production/respiration ratio, or energy transfer within the system food web (Suter 1993). The effects of stressors on patterns of material cycling in systems may focus on nitrogen, phosphorous, or other nutrients which for example, could have implications on primary production (Suter 1993).

Composition-based assessments (referred to by Suter [1993] as a naturalist perspective) would focus on stressor effects on species composition and perhaps species diversity as fundamentally significant with respect to ecosystem effects. Although these are basically population level assessments, one assumes that they are indicators of significant ecosystem level effects. Examples of adverse system-wide effects given by Suter (1993) include loss of native species or invasion of non-native species.

Finally, ecosystem level assessments may center on what Suter (1993) describes as ecosystem stability: resistance and resilience. Obviously a key component of such an assessment would be basic understanding of a system's natural variability. Baseline data is sorely needed to establish the background or reference information for biotic and abiotic ecosystem components.

Ecosystem Models

Ecosystem models for predicting risk are distinguished from other ecoassessment models in that they incorporate both biotic and abiotic components of ecosystems and represent populations from one or more trophic levels. There are numerous models that have been developed which can be applied for ecosystem assessment: some for specific assessments and others adapted from basic research models. Suter (1993) provides a brief overview of approximately 20 different ecosystem assessment models.

Food web models are a type of ecosystem model that may be used to identify an effect at the organism, population, community, or system-wide level. For instance, food web models can be used as a basis for quantifying trophic level exposure for site-specific assessments. Species characteristics (such as mean body weight and food and water consumption patterns) as well as contaminant data are input parameters to food web models. For site specific ecological assessments, these data may be quite distinct, while for assessments that are establishing general standards, these data would be based on more universal assumptions.

Ecosystem models have been developed which are specific to New Jersey. One model pertinent to New Jersey is a Delaware Estuary stream reach model utilized for water quality management. Kelly and Spofford (1977) considered tributary, municipal, and industrial inputs into 22 segments comprising the Delaware Estuary. Their model included eight biotic and three abiotic ecosystem elements. The biotic compartments were based on trophic level (algae as primary producers, zooplankton as herbivores, fish as omnivores, and bacteria as decomposers) to address pollutant pathways and nutrients (nitrogen and phosphorous), organic matter (BOD), and oxygen to address energy and material transfer across trophic levels. Abiotic compartments included turbidity, toxicity, and temperature. Model equations describe the relationship of upstream inputs, feeding, toxics, temperature, and oxygen on consumers and resulting change including death, respiration, excretion, downstream loss, and predation. Nutrient uptake for alga is distinguished from the consumer compartments because of unique considerations: light controls photosynthesis rates and alga require simultaneous nutrient sources.

Kelly and Spofford (1977) reported mixed results in that their predictions for oxygen, BOD, phosphorous, and nitrogen for some periods fit well to reported data, while for other periods, the data do not fit the model very well.

Thomann (1989) developed a model for calculating organic chemical concentration in a "generic" four-level food chain (phytoplankton → zooplankton → small fish → top predator). Thomann (1989) reported organic chemical uptake, assimilation, and excretion

to be a function of the octanol-water partition coefficient. Other, non-lipid factors were assumed to have minimal influence on chemical partitioning to lipids. Growth rate and variation in uptake efficiency are two parameters that significantly influence the bioconcentration factor calculation and, ultimately concentration estimates in receptors via trophic transfer (Thomann 1989). Thomann (1989) acknowledged the limits of his model: at log K_{ow} greater than 6, using K_{ow} for estimating bioconcentration factors can lead to a significant underestimation of tissue concentrations. In addition, at log K_{ow} 5-7, observed chemical concentrations in upper trophic levels are significantly higher than calculated bioconcentration factors (Thomann 1989).

To address the utility of PCB controls, Thomann et al. (1989) specified a two stage framework of PCB fate, transport, and bioaccumulation in the Hudson River estuary. The first stage is a physio-chemical model which predicts dissolved PCB concentrations throughout the estuary considering partitioning, settling, resuspension, sediment exchange, volatilization, and decay. The PCB concentrations drive a food web model which considers fish uptake, depuration, bioaccumulation, and migration. Combined, this framework predicts PCB concentrations in striped bass under alternative remedial scenarios: no removal of PCBs from sediment or complete removal of PCBs from upstream.

Thomann et al. (1992) have extended the four-level food chain model to include sediment interaction as a component of chemical accumulation in aquatic food webs. The model addresses bioavailability of chemicals from sediments and their potential for trophic transfer through a fifth food web component: benthic invertebrates (Thomann et al. 1992). One pathway the model does not appear to address is direct ingestion of zooplankton by benthic invertebrates.

Thomann's work is discussed here not only because it is specific to a New Jersey ecosystem, but also because it is considered to be one of the best food web models available. Note, however, that Thomann's models have been designed for determining food web transfer of organics and are not considered to be appropriate for many compounds, such as metals. (The New Jersey Department of Environmental Protection, Division of Science and Research [NJDEP DSR] is currently exploring a collaborative effort with Thomann to develop credible fate and bioaccumulation models for metals discharged to surface waters [Hazen 1994]). Although regulatory agencies do not always endorse the use of Thomann's model *in toto* for developing criteria, food chain multipliers based upon Thomann's work have been used by EPA and state regulatory agencies to quantify exposure (EPA 1993c; 1993d).

Section III.B.1 of this report describes the Delaware Estuary and New York-New Jersey Harbor Estuary Programs which have been sponsoring research since 1989. Where appropriate, data generated from these programs might be tested against the estuary-specific models described above to ascertain their utility. (Obviously, there may be some question as to whether the Kelly and Spofford model is dated: the data used by Kelly and Spofford are almost 25 years old while the model itself was published in 1977.)

Although ecosystem models have been evolving for some time, there does not appear to be scientific consensus about which models are the most appropriate for using in ecosystem assessment. In addition, there is a lack of a standard set of criteria for evaluating the utility of any one particular ecosystem model. As the level of sophistication or utility of ecosystem models improves, scientific consensus as to the appropriateness of any one model for a certain scenario or scenarios should develop. In the meantime, it might be useful for NJDEP, as a regulatory agency, to establish a set of general criteria that should be met prior to using an ecosystem assessment model as the basis for a regulatory action or recommendation.

Empirical Approaches for System-wide Assessment

Suter (1993) identifies testing approaches for ecosystem level assessment in the following categories: interactions between species; microcosm tests; mesocosm tests; and field tests. Although there are few data to show that stressors directly affect species interactions, Suter (1993) points out that this may be because of poorly designed studies. Species interactions for assessment might include predation, parasitism, or competition.

Microcosms are laboratory setups that simulate an

ecosystem or subsystem (Suter 1993). These test systems can be assembled from components of the environment which they are intended to simulate or can be excised directly from the environment into a laboratory setting. Assembled systems have the advantage of standardization, while excised systems have the advantage of being more realistic in terms of actual exposure. In addition to intent for identifying system-wide responses, microcosm studies may be useful to "reveal mechanisms of response which are then incorporated into conceptual or mathematical models of responses of field ecosystems" (Suter 1993).

Mesocosms are external systems that can be assembled or enclosed portions of existing outdoor systems. The advantage of these systems is that they compose some portion of an outdoor system subject to the actual physical parameters present in nature; however, this type of system is also subject to damage from weather or other disturbances (Suter 1993).

Field tests are experiments in "an unconfined system conducted at approximately the same scale as the actual contamination" (Suter 1993). The advantage of field tests is that they are exposure scenarios of interest, rather than representative of an exposure scenario via a laboratory study or assembled mesocosm study. Like mesocosm studies, field tests are subject to damage and also can be more costly than laboratory studies (Suter 1993).

According to Suter (1993), the most common method for extrapolation of ecosystem test data to ecosystem assessment has been to assume that ecosystem test data are representative of responses in a similarly exposed ecosystem. This assumption is utilized for lack of any other approach but is limited in that test measurements may be indicators of assessment endpoints (rather than the actual assessment endpoints themselves), may show effects at a different level of exposure than the assessment endpoints, or may respond in a manner different than assessment endpoints.

Suter (1993) examines other potential methods for extrapolation of ecosystem test data to an ecosystem of concern. One method might be to apply the stochastic approach of species sensitivity (in which species response is represented as a statistical distribution in relation to a test endpoint) to ecosystem sensitivity. A model might be used to predict the probability of a response at various exposure levels. Obviously, such an approach is limited by the use of fitting test data to statistical models, but such an approach offers greater confidence by using a range of endpoint responses rather than a point estimate of response. Suter (1993) points out, unfortunately, that such an approach may not be possible at this time because of the lack of uniformity in ecosystem study design.

Another approach described by Suter (1993) is to classify ecosystems based on functional parameters and relate field data to an assessment based on classification. Suter (1993) suggests developing classification criteria and methods for applying these classifications in assessments. Examples of functional parameters for classification could include number of species per trophic level, nutrient limitations, or soil or water chemistry.

Suter (1993) also explains that effects on ecosystem functional processes may be a plausible method of ecosystem assessment if examined at the microcommunity process level, rather than assessments extrapolating data from the organism or population level. Assessments of this type (e.g., rate of decomposition), however, must include careful understanding of how the processes in one microcommunity compare to those in another (if indeed they are comparable), as well as the natural variability within the system itself. For example, a system which may show a decrease in decomposition is not necessarily an adverse effect as leaf litter in a system is involved in a host of system interactions (enhancement or inhibition of seed sprouting, erosion control, etc.) that are constantly in flux (Suter 1993).

Currently, it appears there is no standard approach for extrapolating ecosystem response data to risk assessment. Suter (1993) points out that because the chemical and physical processes that govern chemical fate at the ecosystem level are better known than biologic processes, ecosystem-level assessments use fate and effects models of functional parameters for characterizing chemical responses at the ecosystem level, while biologic responses at the ecosystem level are usually determined by extrapolating from organism-level data. A no-observed-effects concentration or lowest-observed-effects concentration from an organism-level assessment may be applied for ecosystem level effects when no other system-

wide assessment is available. Again, however, it is important to qualitatively account for how individual species fit into the trophic structure or web of a system. Although there may not appear to be an effect in a test organism, trophic transfer into another species might reveal definitive effects that could be missed by applying organism-level data to a system-wide assessment.

Obviously, continued work in the area of ecosystem-level assessment should help to clarify the limitations cited above. In the meantime, it is up to risk assessors and risk managers to work together to evaluate the broadest body of scientific knowledge so that risk management can proceed taking into account current limitations, while moving forward with research into those critical areas identified during the risk assessment process.

8. Risk Management

Risk management is the process of evaluating the independent ecological risk assessment and developing directives for the resource. Qualitative judgment is very important in risk management, especially as it relates to any models that may have been utilized. As Suter (1993) aptly describes, for example, some population-based models may require the incorporation of density dependence on the one hand, otherwise predictions can only assume population extinction or population growth without bounds. On the other hand, incorporating density dependence may lead to a false sense of certainty in that very often these models cannot account for the extreme sensitivity that populations would actually exhibit in response to stressors. Therefore, as Suter (1993) points out, management strategies developed based on these models have met with mixed results. Obviously, it would be prudent to develop management strategies based on a comprehensive body of knowledge, best scientific judgment, and continued refinement as assessment methods increase in precision.

D. Retrospective Ecological Risk Assessment

Retrospective assessment is determination of risk based upon past activity. Since contamination of a system has already occurred, there is some initial sense at assessment outset of sources, exposures, or effects. Retrospective ecorisk assessment attempts to define the relationship between contaminant source and ecological effect through environmental exposure pathways. Measurements or other data relating to one or more of these assessment parameters allow for inferences about other parameters as the assessment proceeds.

Retrospective assessments can be achieved by relating contaminant distribution in the environment to both measured field responses and modeled responses (especially in cases where adverse effects may not have yet manifest themselves). Establishing cause and effect is best achieved through field measurements using an epidemiologic approach. Toxicity-based retrospective risk assessment can be used when effects are not evident, but data from effluent, media, and other laboratory tests can be correlated with implied effects (EPA 1991b). EPA (1991b) stated that when field observations of effects or field measurements for toxicity correlations are not possible model-based retrospective assessment should be used.

Suter (1993) points out that unlike a hypothetical boundary developed for predictive assessment, a retrospective assessment will be bound by either the source, affected species or system, or extent of exposure depending upon which parameters are known and therefore, drive the assessment.

A basic approach for retrospective risk assessment is: hazard definition; endpoint measurement and estimation; and risk characterization.

1. Hazard Definition

Defining hazard is determined by which parameters drive the assessment and the distribution of those parameters in the environment. Source-driven assessments are defined by the distribution of contaminant sources throughout the environment; effects-driven assessments are defined by the distribution of environmental effects; and exposure-driven assessments are defined by the distribution of environmental exposure. (Suter 1993).

2. Endpoint Measurement

Endpoints may be determined through direct

measurement of contaminant sources, exposure in target organisms, or effects in the organisms, population, community, or system of concern. Direct measurements may not be possible because of rapid contaminant degradation, inaccessibility, or other constraint. Indirect measurements such as indicator measures, modeled values, or a combination of measured and modeled values may be used for estimating sources, exposures, or effects.

Establishing cause and effect may not be straightforward, especially for assessments at the population, system, or community level where natural variability may not be well-understood or difficult to discern. Testing the effects of contaminated media on an organism of concern or indicator organism may be *in situ* or in the laboratory. Comparisons are made to reference sites.

Inherent difficulties in testing media already contaminated as opposed to predictive assessment testing include potential for compositional changes during sample collection, transport, storage, and preparation, as well as variability of the contaminants in the environment over time (Suter 1993). For example, known pesticide contamination in ground water may shift dramatically from one sample date to another in an area subject to wide fluctuations based on tidal surges. Therefore, one might miss high levels of a contaminant present during critical exposure periods.

3. Risk Characterization

In characterizing risk for retrospective assessment, the focus is on whether an effect is real, and if so, what is its cause and how great is it. When effects can be measured in the environment, epidemiologic methods, although developed for assessment of human health, may be applicable for characterizing ecological risk. Epidemiology is the study of the occurrence and distribution of diseases and other conditions in populations; assessment of diseases and conditions in nonhuman populations may be achieved in some part via epidemiologic approaches.

EPA's Risk Assessment Forum (February 1991) report cites epidemiological methods as most applicable to ecorisk because determination of whether an effect is real requires knowledge of normal variability for a particular site. Knowledge of normal variability allows comparison of differences for that particular site or to limit the chosen endpoints to those that are "genuinely aberrant" (EPA 1991b).

Normal variability is a major area of uncertainty in ecorisk; especially since wide variability within a system may be inherent and therefore, determining real effects may not be so straightforward. Baseline information about system variability is an area in which scientists would probably agree that very basic research is necessary.

Epidemiologic approaches in which systems are assessed through comparison with reference sites are used in some ecorisk assessments. Although some scientists might argue (Suter 1993) that statistical methods in epidemiology are not useful in general for ecological risk assessment, because they often rely on dichotomous variables, it may be possible to apply methods which look at comparison of group means or the statistical significance of related variables (Golden 1994).

Determining an appropriate reference site can be another prime area of uncertainty. Some scientists might argue that since sites are so variable from one another, there are no appropriate reference sites. Other scientists might advocate various methods for defining "normality for a particular community type in a particular region" (Suter 1993). Such methods could include: using aggregate data to develop a regional index for a specific attribute in an undisturbed system; using multivariate statistics to determine differences between exposed and unexposed communities; analyses of similarity, etc. (Suter 1993).

Habitat evaluation is a technique that can be used for determining normality in a community; that is, what species would one expect to find in a particular community under "natural" conditions. The absence of a species from a particular habitat may indicate an effect (Suter 1993). Models have been developed for some types of systems, such as streams, which have been used to predict the probability of species occurrence.

Suter (1993) points out that if a system has been studied over time prior to a known or apparent effect, then it might be possible to estimate deviation from system normality.

If no reference community is available or a system's standard state cannot be ascertained, a more empiric intuitive approach would be necessary for determining an effect. For example, clearly abnormal endpoints, such as the absence of any wildlife or vegetation, high mortality, or high reproductive failures, would be clues that something irregular had occurred.

Models can also be used to determine if an observed ecological effect is consistent with a suspected cause. A model representing the hypothesized cause can be used to calculate the magnitude of source required for the observed effect. "If the required model inputs are consistent with the observed source, then one obtains partial confirmation that the source could have caused the observed change (Suter 1993)."

Important to any study design or method are: careful definition of what ecological endpoint is of concern; what are the best measures of that endpoint; and what are the potential causes based on the most current scientific information including mechanisms of action, exposure routes, etc. One pitfall in ecological assessment is that the more one measures potential causes of an effect in exposed vs. unexposed populations, the greater the likelihood of finding a statistical correlation by chance alone. Certainly, there may be multiple factors leading to an adverse ecological effect; yet, it is important to distinguish causation from the problem of multiple testing.

The standard proof of toxic effect is demonstration of an increase in effect with an increase in exposure...rather than testing the null hypothesis that exposed communities are the same as unexposed communities, the assessor should hypothesize and statistically test a model describing the relationship between the measurement endpoints and the hypothesized causes of variation in the measurement endpoints...(Suter 1993)

Epidemiology originally developed in response to diseases of infectious etiology. Efforts in the 19th century to establish infectious disease etiology evolved into the Henle-Koch postulates of causation. More recently, epidemiologists have expanded these postulates into criteria for causality which have been applied to chronic diseases of non-infectious etiology.

A summary of the criteria for causation is given in Table 2. Assuming one can ascertain normal variability for a particular system, it may be possible to apply these criteria for causation, developed in the realm of epidemiology, for determining ecological risk. Establishing a temporal association of causality may be possible through historical biological data. For example, paleologic information may provide stratified and/or baseline data for the status of algae, arthropods, macrophytes, and chemical compounds in these organisms, as well as sediments. Tree rings can also provide important historic information in relation to chemical exposure. (Suter 1993). Keep in mind when reviewing Table 2 that the endpoint of concern might be an ecologic effect rather than disease, the traditional endpoint of an epidemiologic study.

Quantifying the magnitude of risk is the ultimate objective in ecological risk assessment. One approach is to estimate the magnitude of effects by adjusting toxicity test model results to the results of effects observed in the field. In other instances, it may be more appropriate to develop a model based on field data. Estimates of risk should also account for spatial distribution from the source of exposure, as well as temporal distribution to account for episodic exposures.

Although an important consideration, quantifying recovery of an ecosystem from an exposure is difficult to predict because of the inherent fluctuations in any one system. Therefore, perhaps the best approach for recovery is to estimate when a system might exhibit characteristics of one that had never been exposed to the contaminant based on a confidence interval of predicted time to recovery or adaptation. (Suter 1993)

As previously mentioned, when contamination is known but effects are not initially observed in the field, retrospective assessment can use toxicity test data for ascertaining effects. This form of retrospective assessment is very common in ascertaining ecological risk at contaminated sites. The assessor relates toxicity test data with field data to predict effects from exposure. Risk is usually qualitative, rather than a quantitative characterization; risk for a particular endpoint is either predicted to occur or not to occur based upon a quotient of environmental contaminant concentration in relation to an acceptable exposure concentration. Acceptable exposure concentrations are derived from toxicity test data using a variety of methods generally accepted in the regulatory community.

Table 2. Criteria For Causation[1]

1. Hypothesized cause distributed in population in same manner as disease (or ecologic effect).

2. Incidence of disease (or ecologic effect) significantly higher in those exposed to hypothesized cause than in unexposed.

3. Frequency of exposure to hypothesized cause should be greater in those with the disease (or ecologic effect) than in controls.

4. Disease (or ecologic effect) should follow exposure to hypothesized agent (temporally).

5. The greater the dose or length of exposure, the greater the likelihood of disease occurrence (or ecologic effect).

6. For some diseases (or ecologic effects) a spectrum of responses should follow exposure to the hypothesized agent along a logical biologic gradient from mild to severe.

7. Association between the hypothesized cause and disease (or ecologic effect) should be found in various populations (or locales) using different methods of study.

8. Other explanations for the association should be ruled out.

9. Elimination or modification of the hypothesized cause or of the vector carrying it should decrease the incidence of the disease (or ecologic effect).

10. Prevention or modification of the receptor's response on exposure to the hypothesized cause should decrease or eliminate the disease (or ecologic effect).

11. When possible, in experimental settings the disease (or ecologic effect) should occur more frequently in animals or man (or other defined parameter) appropriately exposed to the hypothesized cause than in those not exposed.

12. All findings should make biologic and epidemiologic (and ecologic) sense.

[1]From Kelsey, Thompson, and Evans 1986.

Toxicity testing (including field studies, mesocosm studies, or microcosm studies) can also be performed to observe effects from actual environmental exposure. In this instance, quantification of risk may be possible.

Prediction based on retrospective assessment extrapolates future risk from continued exposure beyond the point of assessment field observations. Modeling future risk from retrospective assessment is a type of predictive risk assessment, but may be more precise than predictive assessments for potential sources of exposure because contamination has already occurred. Therefore, input parameters are based on actual contaminant levels (not predicted levels) and, other model parameters can be more site specific because site characteristics are known.

E. Ecological Assessment and Genetic Engineering

Assessment of risk from the environmental release of genetically-altered organisms attempts to address the following issues: escape of a bioengineered organism into ecological niches and overcoming natural species such that it becomes a pest; disruption of delicate environmental balance; direct risks to humans or wildlife; threat to gene stability or risk of gene transfer; and evolutionary impacts (OTA 1992).

A single approach for assessing risk from planned introductions of genetically-engineered organisms has not been universally developed or accepted; however, because living organisms reproduce, have the potential to exchange genes, and can relocate, ecological assessment of this type of material is particularly unique (OTA 1992).

OTA (1992) proposes a methodology for risk assessment of genetically-engineered organisms drawing upon Risk-Source Characterization, Exposure Assessment, and Dose-Response Assessment. Ecological aspects of this proposed assessment process would include properties of the bioengineered organism and its recipient environment including genetic stability, likelihood for gene transfer, and interactions between the organism and biologic parameters of its new environment (OTA 1992). OTA (1992) further cites the need to consider possible means of containment and monitoring in the event that adverse effects could result, as these organisms cannot be recovered upon release.

Inherent in the assessment of risk from bioengineered organisms is our limited understanding on the actual effects of introducing bioengineered species. How far can we extrapolate data from laboratory and field trials to large scale applications? What do we know about the rate of spread of introduced organisms? Are the recombined genes stable? Do they have the

potential for transfer into other organisms? Do they place evolutionary pressures on other organisms? Can we monitor recombined organisms once released into the environment?

Gene stability is an important aspect of ecological assessment for bioengineered organisms and requires data on the individual gene as well as knowledge of its manipulation into the host organism. For instance, the relative stability of a gene can be greater if inserted into chromosomal areas subject to lower frequencies of recombination (OTA 1992).

Potential for gene transfer into wild species should be assessed based upon the organism in which the gene is introduced as well as the original gene transfer mechanism. Information about the potential for gene transfer in the naturally occurring bacterial environment is important, particularly given the fact that many genetically-engineered organisms are themselves bacteria. Gene transfer may be mediated by vectors (transduction) which can move freely among organisms, incorporating genetic material from one to the other. Gene transfer may also be via direct transformation of freely existing DNA into cells or via conjugation between organisms (OTA 1992). At the current time, gene transfer among species in the naturally occurring environment is not well-defined (OTA 1992).

These are just some of the data gaps and scientific uncertainty in ecological assessment of bioengineered organisms. Data in these areas have been and continue to be generated through laboratory research and field trials of these organisms. As the discipline of bioengineering and assessing its risks continues to evolve and as ecologists and biochemists become more familiar with one another, the ability to assess risk in the field of biotechnology will continue to improve.

OTA (1992) points out that some research on bioengineered organisms to date has shown that genetically-modified microorganisms may not persist at levels great enough to proliferate and therefore disrupt existing communities; however, given the limited data, large scale uses should proceed with caution until more is known about the fate and effects of these organisms. In addition, because of the intrinsic subtleties in genetic material, risk assessments should focus on the specific product.

F. Regional Ecological Risk Assessment

Assessment of ecological risk for regional areas considers the same paradigms as predictive or retrospective risk assessment, yet the spatial and temporal area over which the assessment is concerned can be orders of magnitude from that which many ecological assessments have focused on to date. Since regional assessments are concerned with effects at relatively large scales and include multiple activities, conducting a regional assessment can be quite complex.

Suter (1993) points out that ideally, regions should be defined in terms of natural features such as watersheds, air basins, physiographic provinces, and ranges of populations of concern. Natural delineations allow for more biological and physical uniformity and therefore, similar regions are more likely to respond to stressors in a similar manner.

In some instances, anthropogenic delineation may be most useful for regional definition. For example, regions can be defined by sources such as pollutant deposition (Suter 1993). Regional ecorisk assessment endpoints could include what Suter (1993) describes as "traditional" ecological endpoints (population, community, and system-wide effects) as well as regional endpoints based on anthropocentric factors. Regional endpoints include species range; soil or nutrient loss; pollutant export; climatic effects; and susceptibility of the system to phenomena such as pests, fire, floods, etc (Suter 1993).

The concept of industrial metabolism or industrial ecology is a systems approach that considers all sources of contaminants, their pathways through the industrial economy, and mechanisms of transformation into outputs absorbed and processed by the environment (Stigliani and Jaffe 1992). Such regional approaches to ascertaining chronic contaminant inputs to ecosystems should be linked to physiographic regions and indicators of the health of that ecosystem.

A paradigm for system-wide assessment would be to couple this regional approach to anthropogenic sources and their environmental fate, transport, and transformation with indicators of ecosystem health. The industrial ecology approach allows for system-wide assessment using trend analysis regarding anthropogenic inputs. This trend analysis can provide

decisionmakers with a point of reference for assessing past, current, and future activities within an ecosystem.

Recent work by Stigliani, Jaffe, and Anderberg (1993) in the Rhine Basin has centered on an historical analysis of cadmium, lead, and zinc pollution from 1950 extrapolated through 2010. Atmospheric and aqueous loadings of cadmium, lead, and zinc peaked in the Rhine Basin in the mid-1960's and have declined since then as the result of several factors: use of unleaded gasoline; reduction in iron and steel production; increased use of pollution control equipment; cleaner manufacturing technologies; better solid waste management practices; and increased recycling of wastes. Although total emissions of these metals decreased, the source allocations have changed over time. For example, incineration of waste consumer products had been a minor source of cadmium and zinc atmospheric emissions in the 1960's but by 1988, waste incineration became a major source of cadmium and zinc emissions (Stigliani, Jaffe, and Anderberg 1993).

In addition to sources, Stigliani, Jaffe, and Anderberg (1993) provide a trend analysis of environmental loads into soil and surface water. By linking such information to the potential effects of such loads on receptors (animal, vegetative, or human), one can assess environmental impacts and potential for future impacts based upon current practices and sources of contaminants on a regional basis. The authors cite the utility of this technique for planning purposes. As an example, decisions about converting agricultural land for other purposes may consider the potential for heavy metal mobilization once pH levels are no longer maintained at 6.0 or higher in light of current metals loadings to soils, sources, and their related effects (Stigliani, Jaffe, and Anderberg 1993).

As Omernik (1987) points out, the ecoregion concept is not new, and actually dates back at least to 1905 when Herbertson (as cited by Omernik 1987) published an essay on the "major natural regions" and considered human development via land use. Ecoregion definition might be useful for addressing issues such as nonpoint source pollution, eutrophication, and acidification (Omernik 1987).

Omernik derives ecoregions for areas in the United States based on relative homogeneity of soils, land use, land surface form, and potential natural variation (Omernik 1987). Omernik's (1987) ecoregion size varies from 15,000 km^2 to 330,000 km^2; the author suggests a minimum size of 500 km^2 to encompass topographic watersheds.

Omernik et al. (1988) have clarified regional patterns of phosphorous deposition as a measure of lake trophic state in Wisconsin, Minnesota, and Michigan. The authors defined regions of lake quality and compared them with variables such as land use, physiography, forest type, geology, vegetation, and soils. High phosphorus values tended to be associated with high percentages of land use in agriculture, and lakes with low phosphorus values tended to be in areas with little to no agricultural potential, were heavily forested, and/or had sandy soils (Omernik et al. 1988).

The Ecoregion Subcommittee of EPA's Science Advisory Board (EPA 1991a) has endorsed the ecoregion concept as a defensible classification technique for large areas (covering one or more states) that is superior to the classification methods currently in use. States such as Arkansas, Ohio, and Minnesota (EPA 1989) have used the ecoregion approach as a water quality management tool by developing regional specific criteria for water quality assessment. The subcommittee concluded that states need assistance from EPA in developing water quality criteria and standards using the ecoregion concept.

In addition, the subcommittee identified other potential applications of the ecoregion concept including land, wildlife, and timber management, as well as consideration of identifying designated areas for chemical releases or multimedia regulatory approaches (EPA 1991a). The subcommittee recommended that whatever research and evaluation of this concept continues, it should be coordinated with EPA's EMAP program.

Assessing risk at the ecoregion level should consider multiple stressors on a system. Changes in land use and resource management over time may be very important variables to consider in concert with chemical stressors on a system. Suter (1993) cites types of cumulative impacts that can be very significant as a whole, on ecosystems. Such cumulative impacts result from a number of similar small incremental effects; cumulative effects that occur when actions are so close in time that the

system does not have the chance to recover between stresses; cumulative effects that occur because of physical overlap; and cumulative response of indirect effects from stressors (Suter 1993).

G. Ecorisk Assessment: Operational Considerations

The following sections discuss operational aspects of ecorisk assessment for deriving acceptable exposure levels. Theoretical operations are discussed, as well as operations specific to deriving criteria and standards in New Jersey. In addition, uncertainty factors in ecological assessment are discussed.

1. Using Stressor-Response Analysis and Exposure Analysis To Derive Acceptable Exposure Levels

Calabrese and Baldwin (1993) provide a number of operational discussions, procedures, and recommendations for calculating and analyzing stressor response, exposure, and acceptable exposure levels. Much of their analysis is based upon aquatic data but they also propose methods for deriving terrestrial equivalents. The organism-level approaches are, for the most part, chemical-specific. This parallels much of risk assessment, which is usually chemical-by-chemical analysis, rather than addressing mixtures. Some discussion on mixtures is provided; however, assessing risk of multiple contaminants usually relies on some qualitative evaluation in the absence of quantitative data and methods.

There is no single method (or protocol for selecting a method) that has been adopted or endorsed by government or the scientific community, as a whole, for deriving acceptable exposure levels in ecological risk assessment. As the field of ecological assessment progresses, scientific consensus should begin to develop about the appropriateness of various models, data, and uncertainty factors, including those identified and recommended by Calabrese and Baldwin (1993).

Maximum Acceptable Tissue Concentration

One assessment technique is to determine a Maximum Acceptable Tissue Concentration (MATissueC) using a food web model. This approach is applicable to environmentally persistent lipophilic compounds which bioaccumulate and biomagnify. It can be used if steady-state has been established, if enough data exist to show tissue or whole body residues predict toxicity, and if enough data demonstrate that interindividual or interspecies variability in susceptibility are due to differences in pharmacokinetics rather than pharmacodynamics (response) (Calabrese and Baldwin 1993).

The food web model allows for estimating exposure at various trophic levels and also to estimate tissue concentrations at known environmental levels. One can also use this approach to derive acceptable environmental concentrations of contaminants. The assessor should be aware that often estimated tissue concentrations are compared with toxicity data; however, toxicity data are given in dose rates (mass contaminant/mass tissue per day) while tissue concentrations in mass/mass (mg/kg). Therefore, concentrations must be translated to dose rates or vice versa to prevent confusion.

A food web model calculates a biomagnification factor (BMF) for contaminants. Environmental source concentrations of contaminants are calculated based on the BMFs and tissue concentrations in exposed biota. If an MATissueC is determined, this can be used in concert with a BMF to calculate a maximum acceptable source concentration.

An MATissueC can be determined by identifying an acceptable exposure rate (from the literature) and exposing biota to that exposure level. Once an MATissueC for each trophic level is derived, a BMF can be derived by measuring the concentration of the agent in the body divided by the concentration of the agent in food. A PBC (probabilistic body criterion) for that medium can be calculated based on the MATissueC/BMF. In a regulatory sense, this PBC may serve as an acceptable environmental concentration for a compound in a specific medium.

Calabrese and Baldwin (1993) explain that extrapolating an MATissueC value from one species to another may be acceptable if the variation in species susceptibility is pharmacokinetic (uptake-, distribution-, elimination-based) rather than pharmacodynamic (response-based). If the variation is pharmacokinetic, then one would expect similar responses in the receptor at steady-state tissue levels. Again, the MATissueC may not be appropriate if

enough of a literature base does not exist to confirm exposure concentrations with tissue concentrations. In addition, an assessor must be aware of other factors and exposures which could cause other effects or contribute to those observed or predicted to occur.

Terrestrial Reference Values

Calabrese and Baldwin (1993) review methods to derive Terrestrial Reference Values (TRVs) when the MATissueC approach is not applicable (e.g., compounds that do not bioaccumulate or unavailability of tissue samples for endangered species or migratory birds). TRVs are considered by some to be the terrestrial non-human equivalent of Reference Doses (RfDs). Endpoints of concern can include growth, maintenance, reproduction, and development.

Uncertainty must also be taken into account when deriving TRVs. Examples of uncertainty factors include intra- and interspecies variation, LOAEL to NOAEL extrapolation, less-than-lifetime to chronic exposure extrapolation, and an additional modifying factor for endangered species protection. (A more detailed discussion of uncertainty factors is given below).

The TRV methods reviewed by Calabrese and Baldwin (1993) are species-specific and nonspecies-specific methods proposed (and partially used) by a government contractor for site cleanup at the Rocky Mountain Arsenal site. The authors conclude the methodology is useful to generate meaningful discussion about an appropriate methodology for deriving TRVs; however, they do not endorse its specific use at this time because the basis for application of many uncertainty factors is not clearly documented. Therefore, the TRV methodology could result in reference values that would be less conservative (or in some instances overly conservative) in the absence of a scientific basis for their derivation.

The TRV methodology attempts to address those areas in which there are distinct differences between human health assessment and ecological assessment. Human health assessment, in general, accepts an interspecies UF of 10 when extrapolating from a test species to humans. Ecological assessment, however, is a method which is concerned with all non-human species. Therefore, a generic interspecies scaling factor may be difficult to ascertain. Interspecies scaling may be based on phylogenetic relatedness between the test species and the species of concern. (Further discussion of Calabrese and Baldwin's (1993) recommendations for interspecies scaling is provided below. Again, they reject the TRV proposals based on lack of documentation).

The TRV method also includes scaling factors based on trophic level differences (which assumes biomagnification), chemical partitioning into body compartments, and multi-media exposures. Calabrese and Baldwin (1993) consider trophic level differences between species and chemical partitioning as exposure assessment issues, rather than UF issues and recommend addressing these issues through exposure factors and food web modeling, respectively. They point out that multi-media exposures are addressed through additivity of exposure sources, rather than via a UF (Calabrese and Baldwin 1993).

The critiqued TRV method accounts for uncertainty in laboratory-to-field extrapolation; however, Calabrese and Baldwin (1993) argue that "no convincing case yet exists to support the use of a generic UF". Human health risk assessment does not generally incorporate a UF for laboratory-to-field extrapolation.

The TRV method proposes UFs based on the metabolic data available (phase 1 oxygenation or phase 2 conjugation UFs). Calabrese and Baldwin (1993) argue that metabolic information is inherently accounted for in the dose-response estimation from toxicity data and application of an UF is not necessary. Finally, Calabrese and Baldwin (1993) argue that applying a UF based on a wide range of toxicity values in the literature is not used in other risk assessment methodologies and is deficient in its basis.

In reality, regulatory agencies often utilize a modified TRV approach for terrestrial species. Often, acceptable exposure levels for a species of interest or related species are derived from a LOAEL or NOAEL value identified via a literature review.

Terrestrial Organism Evaluation in New Jersey

Derivation of acceptable exposure levels for site cleanup in New Jersey relies on EPA guidance and professional judgment. Literature extrapolations are most credible when they pertain to the same species exposed to the same compound as that for which acceptable exposure levels are being derived. If the same species are not available, assessors look to those most closely related; i.e., literature comparisons should at least be for species within the same phylum and trophic level (producers, consumers, decomposers). In addition, data should be for organisms within the same feeding guild (omnivore, carnivore, herbivore, graminivore, etc.).

Finally, species should live within similar habitats; i.e., terrestrial, aquatic, or wetland/saltmarsh for appropriate comparison. If these criteria are not met, then the basis of comparison may be irrelevant and the assessment invalid. (Sacco 1994).

In some cases, ecological assessments in New Jersey will rely on toxicity data for a surrogate compound other than the compound of concern at a site. In this instance, professional judgment is used to allow for compounds with physico-chemical characteristics similar to that of the compound of concern. Review of ecological assessments involves a team of scientists as a check. (Sacco 1994).

Aquatic Assessment: Maximum Acceptable Toxicant Concentration

For aquatic species in which the MATissueC approach cannot be applied, Calabrese and Baldwin (1993) provide an operational protocol for deriving a chemical-specific Maximum Acceptable Toxicant Concentration (MATC) within a hierarchy based on data availability. Although developed primarily on aquatic toxicologic principles, the authors contend their framework is applicable for nonaquatic organisms, as well. This protocol prescribes a method for MATC calculation under the following six data availability conditions:

1. Life cycle test data for species of interest
2. Life cycle test data for alternative species
3. Partial life cycle toxicity test data for species of interest
4. Partial life cycle toxicity test data for alternative species
5. LC_{50} for species of interest
6. LC_{50} for alternative species.

When no other data are available, Calabrese and Baldwin (1993) recommend actual testing be required. They do not recommend extrapolation based on structure-activity relationships because of "extensive uncertainty" (Calabrese and Baldwin 1993).

Aquatic Assessment: Surface Water Quality

EPA's surface toxics control regulation established requirements for an integrated approach in water-based toxics control to achieve and maintain water quality standards (54 Fed Reg 23868, June 2, 1989). A triad of control approaches includes: chemical specific criteria, whole effluent toxicity testing, and integration of biological criteria/biological assessment (EPA 1991d). In theory, this three-part approach should be a system of checks to ensure a water body sustains a healthy ecological community. Currently, New Jersey regulations address chemical specific criteria and whole effluent toxicity testing (N.J.A.C. 7:9B). NJDEP's Bureau of Water Monitoring is in the process of developing biological reference stations for water quality assessment (Mumman 1994).

New Jersey surface water quality standards (N.J.A.C. 7:9B) are based upon chemical specific water quality criteria in concert with stream classification and stream flow. In addition to human health protection, water quality criteria have been developed for protection of aquatic life from both acute and chronic effects of toxics (NJDEPE 1992). The water quality criteria form the basis for limiting specific chemicals in surface water discharge permits, as well as provide a yardstick for determining acceptable exposure levels in surface water systems.

For aquatic life protection, there are four types of water quality criteria: acute marine; acute freshwater; chronic marine; and chronic freshwater. Salinity at 3.5 ppt or greater designates a marine system. Chemical specific criteria (for protection of human health and/or aquatic life) exist for approximately 130 compounds primarily derived from EPA's extensive AQUIRE and IRIS data bases. (NJDEPE 1994a; Lavery and Mallepalle 1994).

Toxicity testing data from a specified distribution of families (minimum of 8 species) are required to derive acute aquatic life criteria for fresh and saline waters. Species data are combined into a species geometric mean. Genus data are combined into a genus geometric mean. The four most sensitive genus means are input into an EPA-derived algorithm to develop a Final Acute Value (FAV). The algorithm is intended to protect for 95% of the organisms in surface water environments. An application factor of 2 is applied to the FAV to derive an acute criterion.

The acute criterion is checked by comparing it to the most sensitive species mean values. If the calculated acute criterion is higher than the most sensitive species mean value and that species is judged by the assessor to be commercially or recreationally important, then the acute criterion defaults to the most sensitive species mean value. (NJDEPE 1994a; Lavery and Mallepalle 1994).

Acute criteria are based on lethal concentrations or effect concentrations (observable adverse effects such as death or immobilization) usually within 96 hours of exposure. In contrast, chronic criteria are concerned with long-term exposure (possibly 1/10th the lifespan of the organism or more) and involve

endpoints such as reduction in growth, reproduction, or survival (EPA 1991d).

The preferred method for calculating chronic criteria is to use chronic data for 8 families. Since data sets for a distribution of 8 families are usually not available, chronic criteria are derived from a combination of chronic data in at least 3 fish species and their respective acute toxicity values. The chronic criterion is the geometric mean of the acute-to-chronic ratio. A final method for deriving chronic criteria is to divide the acute criterion value by 2 as specified in 1985 EPA guidelines.

The chronic criterion value is checked against data on bioconcentration or bioaccumulation factors and aquatic plant toxicity with the most stringent of the three possible values as being the ultimate chronic criterion (Lavery and Mallepalle 1994). Although few data exist to make such comparisons, data sources for the bioconcentration or bioaccumulation factors (BCF or BAF) and plant toxicity comparisons include EPA's AQUIRE data base and EPA water quality criteria documents (Mallepalle 1995).

If BCF or BAF information and either a Food and Drug Administration (FDA) Action Level or a maximum permissible tissue concentration has been derived, then a residue value can be calculated and compared with the chronic criterion (Mallepalle 1995). Residue values are calculated by dividing the action level or permissible tissue concentration by the product of the geometric mean BCF multiplied by % lipids for that species. (Mallepalle 1995). If the residue value is more sensitive than the calculated chronic criterion value or plant toxicity data, the residue value becomes the basis for the final chronic value. Examining BCF data incorporates longer-term data than calculation of a chronic criterion from acute toxicity data because BCF studies require relatively long-term exposure for BCF determination (Mallepalle 1994).

Plant toxicity comparison is directly based upon a 96 hour LC_{50} with an alga or chronic test with an aquatic vascular plant (Mallepalle 1995). In some instances, aquatic plants may be more sensitive to chemical contaminants than other aquatic life. This comparison takes into account the relative sensitivity of aquatic plants and animals. Again, not enough data on BCFs and plant toxicity are available to incorporate in chronic criterion derivation.

Current New Jersey chemical-specific water quality criteria are the same as EPA's chemical-specific water quality criteria. NJDEP has, however, been evaluating the data for aquatic species indigenous to New Jersey waters to determine if enough fish and invertebrate species data exist to develop New Jersey-specific water quality criteria. Where enough data do exist for New Jersey species, NJDEP is developing New Jersey specific water quality criteria. In the coming months, NJDEP anticipates proposing New Jersey-specific water quality criteria for approximately 10 metals and ammonia (Lavery and Mallepalle 1995).

In addition to chemical-specific criteria, water quality criteria exist for conventional pollutants such as dissolved oxygen, temperature, total suspended solids, etc. These criteria are based upon the tolerance of the most sensitive species expected in New Jersey waters (Lavery and Mallepalle 1995). In addition, narrative water quality standards require surface waters to be free from toxics in toxic amounts. Whole effluent toxicity tests allows for testing the aggregate effects of an industrial or municipal discharge on surface water.

Standardized whole effluent toxicity tests have been developed using freshwater or marine plants, invertebrates, and vertebrates. Acute toxicity is a 96 hour lethality or immobilization test; chronic toxicity measures sublethal effects such as survival, growth, and reproduction and does not include teratogenic effects or bioaccumulation (Boros-Russo 1995). Many EPA tests are short-term chronic tests in that they are only 7 days in duration with a focus on the most sensitive life-cycle stage. (EPA 1991d).

Acute toxicity test data are adjusted by using either an effluent-specific acute-to-chronic ratio derived from whole effluent toxicity or a default application factor of 0.05. Calculated acute toxicity limits are compared with those derived from chronic data and the more stringent limit is the basis for the whole effluent toxicity limit (Boros-Russo 1995).

Although not yet fully integrated into the surface water regulatory framework, NJDEP is developing a biological criteria monitoring program using an ecoregion approach.

Biological criteria are numerical values or narrative statements describing the reference biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use...Biological criteria quantitatively are developed by identifying unimpaired or least-impacted reference waters that operationally represent best attainable conditions. Once candidate references are identified, integrated biological surveys are used to characterize the resident community. (EPA 1991d).

Biological criteria can supplement chemical-specific criteria or provide an alternate when chemical-specific criteria cannot be derived as a measure of

overall biological integrity in comparison to reference sites. They are intended to support designated aquatic life use classifications for state standards. Aquatic community structure and function should be used to establish biological criteria. Once established, a protocol needs to be defined and followed for ascertaining whether biota in a waterway are impaired in comparison to the reference sites. (EPA 1991d).

NJDEP is using an ecoregion framework for updating a reference station data base. These reference stations can be used as control sites for assessments using indigenous aquatic biota, i.e., benthic macroinvertebrates. "The ecoregion concept assumes that waterbodies reflect the lands they drain and that similar lands should produce similar waterbodies...Indigenous stream habitats can thus be categorized according to their respective ecoregions" (NJDEPE 1994b).

Benthic macroinvertebrates such as aquatic insects, worms, snails, mollusks, and crustaceans are relatively good indicators of water quality over time because of their limited mobility and (for the most part) limited response to temporal changes in comparison to other organisms such as plankton. In addition, macroinvertebrates can help assess non-chemical impacts such as siltation, thermal pollution, or habitat degradation (NJDEPE 1994b). A rapid bioassessment protocol using a suite of eight possible indices is employed to rate sites as non-impaired, moderately impaired, or severely impaired (Korndoerfer 1995). Sites must be monitored seasonally (four times per year) and be non-impaired for 3 out of 4 seasons to qualify as reference sites (Korndoerfer 1995).

Aquatic Assessment: Sediment Values

Although the water quality assessment compartments described above (water quality criteria, whole effluent toxicity testing, and biological criteria) indirectly apply to sediment toxicity, addressing sediments as a distinct ecological assessment component is important because sediments: act as a reservoir for contaminants; act as a source of contaminants into the water column; accumulate contaminants; and contain contaminants which are toxic to sediment-associated organisms (Calabrese and Baldwin 1993). Calabrese and Baldwin (1993) provide an important review of methods for determining sediment quality criteria. Inherent in criteria development is the assessment of ecological endpoints in sediment-associated organisms. Some of these methods rely on water quality criteria, which are described elsewhere. An assessor will usually compare a sediment quality criterion for a particular compound with the concentration of that material present at a site to characterize risk.

Assessing risk from sediment exposure very often relates to equilibrium partitioning of the contaminant from sediment into a receptor. Calabrese and Baldwin (1993) explain that equilibrium partitioning is related to sediment organic carbon concentrations because "the bioavailability of nonpolar organic contaminants in sediment is a function of the organic carbon content of the sediment, the lipid concentration of the organism, and the relative affinities of the agent for sediment organic carbon as compared to organism lipid".

A reference approach establishes sediment quality criteria based on reference area contaminant concentrations (Calabrese and Baldwin 1993). Reference value selection is obviously a function of scientific judgment about what denotes an acceptable level of contamination.

Sediment quality criteria can be based on EPA water quality criteria: sediment pore water concentrations are compared to water quality criteria (Calabrese and Baldwin 1993). This approach, however, does not address the toxicity of the sediments themselves to sediment-associated organisms via direct exposure or food chain transfer and assumes the water quality criteria are up-to-date.

Equilibrium partitioning between nonpolar organics in pore water and the organic matter in sediments can form the basis for sediment quality criteria. A partitioning model correlates EPA water quality criteria for pore water contaminants with sediment values for contaminants of concern. This approach calculates site-specific sediment quality criteria based on organic carbon content of the sediment. Input values include the organic carbon content of the sediment, the K_{oc} (organic carbon-water partitioning coefficient), and the water quality criterion for that particular compound (Calabrese and Baldwin 1993). Again, this approach assumes the water quality criteria are updated, toxicity is related to pore water and not directly to total sediment concentration, and the K_{oc} value can be adequately calculated from K_{ow} (octanol-water partition coefficients) values. This method is limited in its application to nonpolar organic compounds.

The sediment-biota approach establishes sediment quality criteria based on acceptable levels of contaminants in benthic organisms. Sediment levels that correspond to the contaminant levels in benthic organisms are derived via equilibrium partitioning

(Calabrese and Baldwin 1993). Determination of acceptable contaminant levels in benthic organisms and the limits of the partitioning model used will both drive, as well as limit, the assessment.

A comprehensive sediment quality approach would incorporate models to derive limits based upon partitioning to pore water as well as acceptable levels to benthic organisms.

A bioassay approach determines relationships between field-collected or *in situ* contaminated sediments and test organism responses (Calabrese and Baldwin 1993). This approach may be costly because it requires site-specific field work; however, it may yield the most valid site specific assessment because it is based on actual, rather than modeled, data. This approach can be particularly useful in cases where the literature are limited or for sites with multiple contaminants in which it is difficult to relate literature values for individual compounds to mixtures that might behave differently because of antagonism, synergy, etc.

Another bioassay approach is one in which test organisms are exposed to known amounts of contaminants which have been added to sediments. Dose-response relationships can be determined using this method to establish sediment quality criteria (Calabrese and Baldwin 1993).

Comparing sediment concentration to a screening level concentration may be a first cut to ascertain whether a sediment is hazardous. The screening level approach approximates the concentration of a nonpolar organic compound in sediment which can be tolerated by 95% of the benthic species (Calabrese and Baldwin 1994). A minimum of 20 observations of a species at varying sediment contaminant concentrations is required. In addition, total organic carbon for the sediment is required. Contaminant concentrations at each station are normalized to total organic carbon to yield an organic carbon-normalized concentration of contaminant. The normalized contaminant concentration is plotted against sites where species are present from least to most contaminated station. Multiple species screening levels will be plotted per contaminant. Rank ordering of the screening levels and interpolation identifies the tolerant concentration for 95% of the benthic species (Calabrese and Baldwin 1993).

Since values can be obtained through the literature, if literature values are used, the screening level method is limited by the quality of the data in the literature. In addition, the effect of multiple compounds on the measured outcome may not be discerned through such an approach. A problem with the screening level approach is that field data from contaminated sites may be representative of contaminant-tolerant species only; species that would be absent because of contamination might be underrepresented by the screening level methodology if enough uncontaminated sites were not used to develop the screening levels.

Apparent Effects Threshold (AET) is that sediment contaminant concentration above which statistically significant adverse effects would be expected in exposed biota. The AET is derived from field data on exposure (sediment concentration) and at least one indicator of a biological effect (such as histopathological abnormality, bioaccumulation, mortality, etc. in benthic organisms) in comparison to reference sites (Calabrese and Baldwin 1993). Some limits to this method are that it may not establish cause and effect (especially if the analysis does not account for other contaminants) and it may also miss chronic effects.

A sediment quality triad approach uses three sediment measures: chemistry, bioassays, and *in situ* biotic effects to ascertain the relationship between contaminant concentration and biotic effects and contaminated sites (Calabrese and Baldwin 1993). This approach is similar to the guidance for sediment quality evaluation applied to waste sites in New Jersey.

Guidance for Sediment Quality Evaluation in New Jersey

Ecological assessment in New Jersey includes sediment quality evaluation for marine, estuarine, and freshwater systems. A framework for evaluating contaminated sediments at uncontrolled waste sites in New Jersey employs a sediment triad approach (NJDEPE 1991).

The basic components of the New Jersey framework include: sediment bulk chemistry to identify contaminant concentrations; tissue analysis and bioassays; and community bioassessment (an assessment of resident biota). The bulk chemistry analysis also includes analysis for sediment grain size (a habitat parameter indicating the type of species expected in the sediment and a first indicator of bioavailability of compounds) and total organic carbon content (a physico-chemical parameter which is used to normalize sediment concentrations for comparison purposes and also to calculate K_{oc}).

A sediment quality evaluation should be designed appropriately; i.e., "if the greatest contamination occurred in the past, it is likely that recent actions have resulted in the deposition of a layer of relatively uncontaminated sediment on top of the sediments of concern" (NJDEPE 1991). In such cases, sediment cores analyzed for various layers, rather than near-surface sediments only, may be the most appropriate samples.

New Jersey recommends a tiered approach in which sediment contaminant levels are compared with established screening level criteria. The screening approach for organics uses an equilibrium partitioning approach based on sediment contaminant concentration and organic carbon content. Screening for inorganics, some pesticides, and PAHs are related to a review of adverse biological effect data prepared by NOAA (National Oceanic and Atmospheric Administration) (NJDEPE 1991). More current screening levels for marine and estuarine sediments (not freshwater) have been developed for some compounds (EPA 1992g). These more recent screening levels supersede the values in the 1991 NJDEPE guidance at marine and estuarine sites for those compounds which are evaluated in both guidances (Demarest 1994). These screening values have also been published in Long et al. (1995).

If screening levels are exceeded, then further evaluation is required to ascertain the magnitude of effects from sediments. Further evaluation includes biological sampling (tissue analysis; physical appearance evaluation, etc.) which can provide information on direct effects, as well as bioconcentration, bioaccumulation and biomagnification (NJDEPE 1991). Bioassays may be required to directly measure acute and chronic response of biota to contaminated sediments. System salinity and sediment grain size will dictate what organisms will be selected for bioassays. For example, a bioassay of an estuarine system with a high variation in salinity should utilize organisms that can tolerate a similar range in salinity (Sacco 1994). A clean-up number can be best derived if a dilution series is incorporated into the bioassay design (this applies to soils and surface water as well) (Sacco 1994). Bioassessments of community structure (taxonomic distribution) and community function (trophic level and morphological assessment) can also evaluate the ecological or biological integrity of the system. (NJDEPE 1991).

Federal Guidance For Sediments

Sediment evaluation is also critical with respect to proposed dredging and subsequent ocean disposal of contaminated dredge materials. An evaluation protocol for dredge material and reference site sediment has been developed by the U.S. Army Corps of Engineers and U.S. Environmental Protection Agency (1992). Along with these federal agencies, NJDEP scientists participate in workgroup meetings to develop site-specific guidance and assessment of ecological effects from the dredging and disposal of these materials. Permits for dredging and disposal cannot be issued without evaluation of the environmental impact of these operations.

The dredge material guidance consists of four tiers of dredge material/sediment testing and evaluation. The tier requirements are more complex as data for the previous tier do not provide enough information to satisfy criteria for permit approval.

Tier I - assesses existing information of proposed dredge material including contaminants of concern.

Tier II - water column evaluation of chemical and physical parameters input to models to determine if water quality criteria are met.

Tier III - water column evaluation using acute toxicity testing on dissolved and suspended material remaining after dredge material discharged; benthic toxicity tests; and bioaccumulation testing in benthic organisms.

Tier IV - case-specific study for unusual contamination problems not addressed in other tiers. (U.S. Army Corps of Engineers and U.S. Environmental Protection Agency 1992).

U.S. EPA has developed a Contaminated Sediment Management Strategy (EPA 1994) to streamline EPA's management of contaminated sediments. Prior to this initiative, sediments have been addressed by more than 10 federal statutes and various EPA program offices (EPA 1994). This strategy is intended to provide consistent sediment assessment methods, characterization of risks, and management approaches, as well as to coordinate research needs. EPA intends this strategy to be the "keystone" of the larger federal strategy for contaminated sediment management as required via the Water Resources Development Act of 1992 which established the National Contaminated Sediment Task Force (co-chaired by EPA and the U.S. Army Corps of Engineers).

EPA is developing sediment quality criteria which may be incorporated by states for water quality

criteria and used in developing discharge permit limits or for other programs such as waste site assessment (EPA 1994). EPA's sediment criteria for nonionic organic compounds uses an equilibrium partitioning approach. Draft sediment criteria have been developed for acenaphthene, dieldrin, endrin, fluoranthene. A methodology for developing sediment criteria for metals is anticipated in 1995 (EPA 1994).

A research component of EPA's strategy will continue to collect sediment monitoring data to incorporate into a National Sediment Inventory. In addition, EPA intends to develop bioassay methods, fate and transport models for sediments and dredged materials, and remediation technologies.

To date, EPA's Sediment Tiered Testing Committee has selected solid phase acute sediment and bioaccumulation tests for Agency-wide use. These include: ten day freshwater acute toxicity tests using amphipods, scuds, and midges; twenty-eight day freshwater bioaccumulation tests using oligochaete worms; ten day marine and estuarine toxicity tests using amphipods; and twenty-eight day marine bioaccumulation tests using clams and polychaete worms (EPA 1994). EPA is in the process of developing standard chronic sediment toxicity test protocols; a tiered testing framework; and guidance for statistical analyses and interpretation of results (EPA 1994).

An important ingredient in the EPA coordinated strategy is the link between EPA's intended research program (including testing protocols and development of tissue residue thresholds) and development of criteria. Criteria should be updated as research results become available.

Consideration of Air Contaminants

New and modified sources of air pollution which meet the requirements for an air pollution control preconstruction permit (N.J.A.C. 7:27-8) are subject to a risk assessment if they emit certain contaminants regarded as air toxics; however, the risk assessment is based on human health endpoints and currently does not consider ecological endpoints (NJDEP 1994f; Held 1995).

Limited ecological endpoints are considered as part of the preconstruction new source review (for new major stationary sources of air pollution and major modifications to major stationary sources of air pollution) for facilities in areas in which National Ambient Air Quality Standards (NAAQS) are attained (EPA 1990d). Facilities located in areas in which NAAQS are not attained are not required to address ecological endpoints as part of the permitting process. Currently there are only 6 NAAQS pollutants: ozone, carbon monoxide, particulates, nitrogen dioxide, sulfur dioxide, and lead (Held 1995). Permit applicants in attainment areas must analyze potential emissions impacts to soils, vegetation, and visibility (EPA 1990d).

Ecosystem Level Approaches For Deriving Acceptable Exposure Levels

The previous discussion on stressor-response and exposure analysis for assessing ecorisk centered on organism-level assessment. Very often, organism-level assessment assumes ecosystem protection: protecting for the most sensitive species protects the ecosystem. In reality, however, such an approach might miss a system-wide effect that is more subtle, but not necessarily less adverse to the system. In addition, organism-level approaches very often rely on what data are available in the literature and do not systematically characterize species distribution for a system. Therefore, approaches for ecosystem level assessment should also be considered.

Like organism-level assessments, ecosystem-level assessments can be chemical specific or for multiple contaminants. Calabrese and Baldwin (1993) review a number of ecosystem-level approaches. In theory, these approaches make sense; currently, however, enough data may not be available to actually perform the cited approaches.

One method is to determine the most sensitive species in the system and develop an acceptable level for that species. This approach is similar to the organism-level approach, except that to claim it is a systemwide approach, one must identify all possible species in a system and select representative species for testing. Were baseline census data available for a system, then theoretically such an approach would be possible.

Another approach is an estimation of hazardous concentration affecting a percentage of species in a community (Van Straalen and Denneman 1989 as cited by Calabrese and Baldwin 1993). The problem with this approach is that it assumes some percentage of species within a community is expendable. A similar approach by Kooimjan (1987 as cited by Calabrese and Baldwin) proposed that the number of species can be mathematically predicted by habitat surface area. Both of these methods could be limited by their ability to identify the appropriate representative species in a population. Therefore, they could miss more sensitive species, thus

underestimating a hazardous concentration to more than either the percentage deemed expendable or that presumed to be most sensitive.

One EPA approach develops a community threshold level for effects assuming 95% of genera are protected based upon chronic effects values for animals and plants, and residue values (Stephan et al. 1985 as cited by Calabrese and Baldwin 1993). Another EPA approach is to derive a chemical-specific ecosystem acceptable concentration by extrapolating available data using prescribed uncertainty factors called "assessment factors" (EPA 1984 as cited by Calabrese and Baldwin 1993).

Other ecosystem methods consider trophic transfer, bioaccumulation, and biomagnification. Numerous models and techniques are utilized by various assessors. EPA incorporated ecosystem pathway analysis in its development of standards for metals in sewage sludge to be land applied for agronomic or other reuse purposes (EPA 1992h). As mentioned previously, NJDEP utilized this approach in proposed soil standards for copper and zinc (24 N.J.R. 373-403). Calabrese and Baldwin (1993) review some other methods and their associated safety factors for trophic level transfer. An expanded discussion on trophic transfer uncertainty factors is found a bit further on in this report. In addition, previous sections discussed ecosystem modeling in greater detail.

Another ecosystem level approach centers on the keystone species concept. Keystone species are "individual populations that play a central and determining role in community structure, integrity, and stability" (Calabrese and Baldwin 1993). Contaminant levels that would have an adverse effect on a keystone species would lead to a significant alteration in ecosystem community structure. Fairly extensive knowledge of predator-prey relationships and community structure is necessary for any type of qualitative assessment based upon effects to keystone species. In addition, such an approach is fairly site specific.

As explained before, scientific consensus has not yet evolved over which is the best approach for ecosystem-level assessment. Although not without flaws, ecosystem-level assessments may be valuable in identifying some more subtle system-wide effect that is missed through organism-level assessment. Therefore, their utility should not be discounted if used in concert with organism-level approaches.

2. Uncertainty Factors in Ecological Assessment

Uncertainty factors (UFs) are widely used in human health assessment to account for incompleteness in the data used for calculating safe levels of exposure or estimating risk. Adjustments to data sets include application of UFs: to account for variability in susceptibility within a species; to extrapolate data between species; to extrapolate data from acute or subchronic exposures to a chronic exposure scenario; and to identify a no-observable-adverse-effect level (NOAEL) from a lowest-observable-adverse-effect level (LOAEL). UFs are widely accepted and applied in human health assessment and supported by biological data (Dourson and Stara 1983).

Unlike human health assessment, the use of uncertainty factors in ecological assessment is not consistent and no methodology has yet been developed or accepted as standard convention. Calabrese and Baldwin (1993) provide a review of uncertainty factors in ecological assessment and offer some recommendations. In some ecological assessments, UFs are referred to as application factors (AFs). Some of these UFs are discussed in more detail below. Although some recommendations of Calabrese and Baldwin (1993) are provided, one should be aware that further discussion within the scientific community is warranted before acceptance of any as convention.

An important point made by Calabrese and Baldwin (1993) centers on selection of the appropriate test endpoint. Much of their analysis of ecological data for UF determination is applied for derivation of an MATC based on aquatic toxicity. Care must be given to ensure the most sensitive endpoint is used as the basis for the MATC. Calabrese and Baldwin (1993) cite the work of Suter et al. (1987) which proposes that reduction in fecundity is a significantly more sensitive endpoint than other sensitive life-stage endpoints such as hatching success or larval survival. Analysis of toxicity data for a less sensitive endpoint has numerous ramifications on the calculation of an acceptable exposure level. Similarly, determination of uncertainty in extrapolation within species, from acute to chronic conditions, between species, and from LOAELs to NOAELs also affect acceptable exposure level calculation.

In the case of extrapolation of acute aquatic toxicity data to chronic data, the UF or AF is referred to as the acute to chronic ratio (ACR). The ACR is the ratio of the 96 hour LC_{50} concentration to the no-observed-effects concentration (NOEC) for chronic exposure to a toxicant in the same species under similar conditions. This experimentally-derived AF is then applied to other species for which only acute data are available.

Calabrese and Baldwin (1993) cite data of Kenaga (1982) in which valuation of acute to chronic ratios for 84 compounds in 9 species of fish and 2 aquatic invertebrates ranged from 1 to 18,100. Since 99% of all agents showed an ACR of $\leq$ 1,000, one recommendation has been to apply an UF of 1,000 for extrapolation of an acute LOAEL to a chronic NOAEL (ESE 1989 as cited by Calabrese and Baldwin 1993). Such practice, however, would result in stringent limits for some compounds in which the ACR was much lower than 1,000. Conversely, for compounds with ACRs higher than 1,000, the limits may not be protective for chronic effects.

Calabrese and Baldwin (1993) also review models for developing estimates of acute to chronic aquatic extrapolation factors. They recommend using an UF of 50 based upon regression modeling (citing Barnthouse et al. 1990) which would protect 95% of the populations in their analysis. Unfortunately, the acute to chronic ratio for some of the data sets show a 200-fold to 2,000-fold difference for some very sensitive endpoints. It may be premature to determine a generic application factor for extrapolation of acute toxicity data to chronic conditions.

Calabrese and Baldwin (1993) recommend a 10-fold UF to apply less-than-lifetime exposure studies to chronic risk scenarios for non-aquatic species. They propose that chronic studies be defined as those in which the minimum exposure duration is 15% of the normal adult post-weaning life-span. Calabrese and Baldwin (1993) propose this interval because it would give the species a chance to reproduce, thereby allowing for analysis of potentially sensitive reproductive endpoints such as fecundity, which is usually not assessed in the sensitive lifestage studies.

Calabrese and Baldwin (1993) raise the point that an assessor should be aware that observed toxicity may vary depending upon the kinetics of an agent; some compounds require longer residence time or time to achieve steady-state before chronic effects occur. They suggest that an adjustment to the UF for acute-to-chronic ratios or less-than-lifetime exposure is theoretically valid, yet difficult to ascertain if the toxicity data on which the UF is based does not consider enough time to reach steady-state or does not consider that time beyond the study duration when toxicant level continues to rise in tissues.

Extrapolation of LOAEL data to identify a NOAEL has not had wide use in ecological assessment and varies among assessors and regulatory programs. Calabrese and Baldwin (1993) purport that since past emphasis of ecological assessments has been on acute toxicity, there has not been as great a need for assessing chronic exposures resulting in less data application for derivation of a NOAEL. Calabrese and Baldwin (1993) suggest using some generic UF or deriving one based on regression modeling.

Unlike human health assessment, Calabrese and Baldwin (1993) cite the lack of scientific consensus on the appropriate UF for interspecies variation. The difficulty in developing an UF for interspecies variation is inherently more complex in ecological assessment because of the broad range of taxa which are included in ecological assessment. As one might expect, susceptibility varies among species in according to taxonomic relationship and the agent of concern. Most of the analyses of interspecies comparisons for ecological endpoints are based on aquatic species. Calabrese and Baldwin (1993) cite Sloof et al. (1986) as finding a higher positive correlation for freshwater species within the same phylogenic grouping in response to toxics as opposed to species more distantly related.

An extensive analysis of toxicity in fish species showed an uncertainty factor of 10 for species within genera but increased up to a 32-fold uncertainty when comparing orders within class (Barnthouse et al. 1990 as cited in Calabrese and Baldwin 1993). Calabrese and Baldwin (1993) suggest using an UF of 10 for species within genera and one of up to 65 or approximately 100 for orders within class. The authors recommend a 1,000-fold UF for interspecies uncertainty between classes within phylum.

Intraspecies variability is taken into account for differences in life stage sensitivity. An uncertainty factor is applied to derive an acceptable exposure level in an adult based on data from a sensitive early life stage study. This accounts for less than lifetime exposure, as well as individual differences in susceptibility, because of differences in age. Other aspects of individual variability such as genetic, sex, or nutritional differences are not addressed by a life stage uncertainty factor.

Calabrese and Baldwin (1993) observe that since ecological assessment focuses more on population parameters, than the individual, the life stage UF does not address the differential susceptibility "as long as reproductive success is assumed". (This argument is their rationale as to why less-than-lifetime exposure requires an additional UF as described previously.) The authors recommend a 10-fold UF for sensitive life-stage scaling.

Human health assessment is somewhat less complex than ecological assessment because of its focus on

one species. Uncertainty factors are utilized to account for individual differences in susceptibility, in particular, sensitive subgroups of the human population. As mentioned above, in contrast, ecological assessment primarily focuses on population parameters. Calabrese and Baldwin (1993) suggest that ecological assessments for endangered species require a distinct application of UFs because endangered species legislation is designed to ensure protection for all individuals in the population. Intraspecies differences for endangered organisms should consider chronic toxicity as well as individual differences in susceptibility. Calabrese and Baldwin (1993) propose a UF greater than the 10-fold intraspecies factor for humans because endangered species protection is distinct in focusing on all individuals in the population, rather than human health which focuses on sensitive subpopulations. They propose a generic 20-fold UF: 10-fold to account for intraspecies variability and 10-fold to account for less-than-lifetime exposure. The authors do state this as an area requiring further development of criteria for deriving UFs.

As in human health assessment, knowledge about both bioavailability and absorption is important but often one or both are not well known. Bioavailability pertains to the ability of a material to transfer from an environmental medium into a receptor organism. This should not be confused with absorption, which is the ability of that material to be systemically transported within the receptor and assimilated within its tissue. Assumptions about bioavailability and absorption are usually made, yet these assumptions are usually taken into account in exposure assessment, and not normally through the use of uncertainty factors, as suggested by some assessors (HLA 1991 as cited by Calabrese and Baldwin 1993).

Uncertainty Factors in New Jersey Ecological Assessments

Ecological assessments are conducted under varying auspices within NJDEP (site assessments, development of standards and criteria, etc.). Currently, there is no agency-wide protocol for conducting ecological assessment and employment of uncertainty factors in these assessments. In the preceding sections, use of various uncertainty or application factors was described within each of the specific media program protocols. A brief summary gleaned from the prior discussion is included here.

For the most part, ecological assessments in the site remediation program rely on best scientific judgment. Using a 10-fold UF is generally accepted when deriving a NOAEL from a LOAEL. The use of other UFs are left to professional judgment (Sacco 1994). Scientific reviews do, however, involve a team of professionals to ensure the analysis makes scientific sense and is protective of ecological endpoints.

In surface water criteria development, application factors are used. An application factor of 2 is used with final acute values to derive acute water quality criteria (Lavery and Mallepalle 1994). One method for developing chronic criteria is to apply an uncertainty factor of 2 to the acute water quality criteria (Lavery and Mallepalle 1994). Keep in mind, however, that these calculated values are compared to other data to ensure the most sensitive species for which data are available are protected.

Application factors can be utilized by NJDEP scientists when developing water quality based whole effluent toxicity limitations (NJDEPE 1994a). Application factors are applied to extrapolate acute toxicity data (LC_{50} or EC_{50}) to any aquatic organism lifestage (NJDEPE 1994a). If effluent-specific acute-to-chronic ratio data are not available for deriving whole effluent toxicity limits, a default application factor of 0.05 is used (Boros-Russo 1995).

H. Environmental Indicators

Environmental surveillance and monitoring programs have, for the most part, been designed to ascertain regulatory compliance; however, data generated from these programs can be used as a regulatory yardstick as well as determinants of trends and status of the environment's health. Often referred to as environmental indicators, surveillance and monitoring data are used to measure the health or progress of a regulatory program. For example, has an air pollution regulatory program reduced emissions of the pollutants it intended to?

It is important to understand the distinction between an environmental indicator being used as a data point for an ecological assessment and an environmental indicator as a measure or a concept related to a policy yardstick. This distinction is important because the term environmental indicator can be used differently depending upon the situation, as well as the policy maker, regulator, or scientist using the term. Again, environmental indicators are used for both strict ecological assessment, as well as measures of policy.

In terms of ecological assessment, environmental indicators can provide useful status and trend information often lacking in the published literature.

Some of these data are required by regulatory agencies within permit compliance authorities. Other indicator data are developed by natural resource inventory programs.

Indicators can also provide the impetus behind an ecological risk assessment. If the status of an indicator or a trend analysis for a particular indicator signifies an environmental hazard or potential hazard, a risk assessment (ecological and/or human health) may be warranted.

With the recognition that environmental indicators are important in both the regulatory policy arena, as well as ecological assessment area, New Jersey is working with EPA to evaluate its environmental indicator data base. This specific program is described more fully in Section III.B.4. Environmental Indicators in New Jersey.

EPA's Environmental Monitoring and Assessment Program (EMAP) has delineated environmental indicators by category. The actual environmental effect is measured via response indicators but assessed in relation to the other two categories of indicators: exposure and habitat indicators and stressor indicators.

1. Response Indicators

Response indicators quantify the response of ecosystems to anthropogenic stress at the organism, population, community, or ecosystem level (EPA 1990b; 1990c). Examples include signs of gross pathology or indices of community structure and biodiversity (EPA 1990b). These indicators can be used to describe ecological conditions (EPA 1990c).

2. Exposure and Habitat Indicators

Exposure and habitat indicators are diagnostic indicators that are measured in conjunction with response indicators (EPA 1990c). They show whether ecosystems have been exposed to pollutants, habitat degradation, or other causes of poor conditions (EPA 1990b). Exposure-Habitat Indicators examples include ambient pollutant concentrations, acidic deposition rates, bioaccumulation of toxics in plant and animal tissues and field bioassays (EPA 1990b). These can be used to identify plausible explanations for observed differences and changes in response indicators (EPA 1990c).

3. Stressor Indicators

Stressor indicators are socio-economic, demographic, and regulatory compliance measurements that are suggestive of environmental stress (EPA 1990b). Examples include coal production, population figures, pollutant emissions inventories, and land use (EPA 1990b). These can be used to identify possible causes for changes in exposure and habitat indicators (EPA 1990c).

4. EMAP Indicators

EMAP indicators have been described for six ecological resource categories: inland surface waters, wetlands, forests, near-coastal waters, agroecosystems, and arid lands (EPA 1990c).

III. Federal and State Ecological Assessment Programs

A. Ecological Risk Assessment At the Federal Level

The U.S. Environmental Protection Agency (EPA) has taken a leading role in developing an ecological risk paradigm; however, the U.S. Department of the Interior (DOI) is also moving forward with an ambitious ecosystem approach to DOI lands. Ecosystem assessment is also conducted by other agencies including the National Oceanic and Atmospheric Administration (NOAA) and the U.S. Department of Agriculture (USDA). Representatives from these various agencies, as well as the U.S. Department of Defense and the U.S. Department of Energy, are now involved in an Interagency Ecosystem Management Coordination Group. The primary role of this group is to encourage coordination, cooperation, collaboration, and communication among governmental and nongovernmental agencies for ecosystem management nationally and internationally.

1. Ecological Assessment: U.S. Environmental Protection Agency

Development of Agency Guidance

A comparison of risks associated with major environmental problems was conducted by a task force of EPA senior career managers and technical experts in an effort to prioritize these problems in light of finite Federal resources. One component of this comparative risk project was to initiate an EPA focus on ecological risk.

EPA established an intra-agency Ecological Risk Workgroup that sought outside review and evaluation of its comparative risk ranking methodology. This expert review was attempted in a two-day workshop assembled by the Ecosystems Research Center at Cornell University (Harwell and Kelly 1986). The Cornell workshop was also intended to provide comment on EPA's role in addressing ecological issues.

The first EPA report, known as "Unfinished Business" (EPA 1987), found there to be a "general lack of information on and attention to welfare and ecological effects," as well as "no applicable methodology for ecological risk assessment."

As a follow-up to the comparative risk project, EPA's Science Advisory Board (SAB) reviewed "Unfinished Business" in an effort to assess and compare different environmental risks in light of the most recent scientific data. The SAB report (EPA 1990a) highlighted 10 recommendations. The second recommendation was:

> EPA should attach as much importance to reducing ecological risk as it does to reducing human health risk. Because productive natural ecosystems are essential to human health and to sustainable, long-term economic growth, and because they are intrinsically valuable in their own right, EPA should be as concerned about protecting ecosystems as it is about protecting human health (EPA 1990a).

As a result, the Risk Assessment Forum of U.S. EPA has been reviewing the science of ecological risk assessment to develop guidelines for ecological risk assessment through a series of workgroups (EPA 1991b; 1992a; 1992b; 1992c). In addition to providing oversight of the workgroups, the Risk Assessment Forum develops consensus within EPA on risk assessment issues. Forum recommendations, analyses, and conclusions are presented to the Risk Assessment Council for consideration as EPA policy. Forum products are also transmitted to EPA's Science Advisory Board, as well as generally receive external peer review. If the Risk Assessment Council recommends the guidelines to the EPA Administrator and the Administrator concurs, the guidelines become Agency policy.

EPA's ecorisk development process has included an analysis of whether ecorisk assessment can and should fit into the human health risk assessment paradigm published by the National Academy of Sciences (NAS) in 1983.

EPA's peer review panel concluded that the NAS paradigm can be used to a point but it should not be adopted directly because of the many differences between ecological and human health risk. For example, ecorisk includes stressors other than chemicals (such as climate change) and concerns multiple species, populations, communities, and ecosystem levels. (EPA 1992c)

The Risk Assessment Forum has released a series of issue papers on topics in ecological assessment (EPA 1993g). These papers were peer reviewed as part of an EPA workshop; a workshop report is anticipated in early Spring 1995 (van der Schalie 1994). Eventually, these papers and report should form some of the basis and background for ecological risk assessment guidelines which, like human health risk assessment guidelines, are to be published by U.S. EPA in the Federal Register (van der Schalie 1993).

Ecological Assessment Within EPA Program Offices

Although there is currently no standard process for ecological assessment throughout EPA, individual program offices conduct ecological assessments within the bounds of their legislative and regulatory mandates. For example, EPA has developed a series of guidances specific for sites listed under the Superfund program (EPA 1989a, 1989b,1989c). Below are listed the various legislative authorities for several EPA programs in which ecological assessments are conducted.

Clean Air Act: statutory authority to establish National Ambient Air Quality Standards for any air pollutant which may reasonably be anticipated to endanger public health or welfare and whose presence in the air results from mobile or stationary sources. Requires periodic review of relevant scientific literature leading EPA to establish National Crop Loss Assessment Network to study impact of ozone on agricultural resources.

Comprehensive Environmental Response Compensation and Liability Act and Superfund Amendment and Reauthorization Act (CERCLA/SARA): statutory authority for selection of remedial action at a hazardous waste site that is protective of human health and the environment which takes into account persistence, toxicity, mobility, and propensity to bioaccumulate of such hazardous materials and their constituents and conduct an assessment of permanent solutions that will result in a permanent and significant decrease in their toxicity or mobility. Remedial Investigation/ Feasibility Study is required to define risks to public health and the environment.

Clean Water Act/Water Quality Act of 1987: statutory authority for protection of wetlands and requires the maintenance and restoration of the chemical, physical, and biological integrity of the Nation's waters. Requires development of water quality criteria used by states in establishing water quality standards for surface water. Water quality criteria should protect species necessary to support an aquatic community. Water Quality Act of 1987 requires EPA to establish standards for use of sewage sludge based on potential adverse effect to public health and the environment.

Federal Insecticide Fungicide and Rodenticide Act (FIFRA): statutory authority for prevention of pesticide sale or distribution if pesticide presents unreasonable adverse effects on the environment or causes unreasonable risk to man or the environment accounting for economic, social, and environmental costs and benefits.

National Acid Precipitation Assessment Program: statutory authority under Acid Precipitation Act of 1980 to conduct 10 year research and assessment program on acid deposition.

National Environmental Policy Act (NEPA): establishes national policy and goals for protection, maintenance, and enhancement of environment including Environmental Impact Statement process to assess impacts of federal projects.

EPA's EMAP Program

EPA has undertaken an ambitious initiative known as the Environmental Monitoring and Assessment Program (EMAP). EMAP "represents a long-term commitment to assess and document periodically the condition of the Nation's ecological resources" (EPA 1990b). Although EMAP's primary purpose is to assess environmental regulatory program goals through measures of environmental indicators, EMAP is also intended to provide scientists with broad-based ecosystem data. Initiated in 1990, EMAP will attempt to determine current status, extent, and geographic distribution of ecological resources over periods of years to decades in a statistically-based design.

EMAP's initial activities include: 1) indicator evaluation and testing to collectively describe the overall condition of an ecosystem; 2) network design for statistically sound data collection within national, regional, or subregional sub-grids to define ecosystems and adjust to system changes over time; 3) landscape characterization in GIS format to establish baselines of the physical environment, biological composition, and human activity patterns for quantitative assessment over time; and 4) a near coastal demonstration project in the Mid-Atlantic region to define estuarine resources in coordination with NOAA, U.S. Fish and Wildlife Service, and USGS. These four primary activities are inter-related. For example, the landscape characterization activity will collect remote sensing data that will also be used for network design, as well as identification of stressor indicators such as mining.

Practically speaking, it may be difficult to utilize any data generated via EMAP in the short-term, as sampling is designed through a four-year cycle in which each site will be sampled every fourth year (EPA 1990c). EPA (1990c) observes that trends may only be detected after 10 to 15 site visits, therefore, assessment of conditions at any particular site may not be possible for 40 to 60 years. Also of note, EPA's EMAP program is currently undergoing review and reorganization (Hoang 1995).

2. Ecological Assessment: U.S. Department of the Interior

In February 1993, U.S. Department of the Interior (DOI) Secretary Bruce Babbitt announced plans for the creation of a new bureau within DOI, the National Biological Survey (NBS) [now referred to as the National Biological Service] (Stevens 1993a).
A primary rationale for creation of the NBS is to develop an appropriate survey of basic biologic data and trends to prevent ecosystem endangerment. Under the Endangered Species Act, a conservation plan must be put in place for a species listed as endangered; however, no plan is currently required for prevention of species or ecosystem endangerment (Stevens 1993a, 1993b).

NBS is intended to be a comprehensive inventory of biological resources in the United States. Just as the U.S. Geological Survey maps water and geologic resources nationally, the NBS will attempt to map biological information. Since a primary goal of the NBS is to move toward ecosystem management to prevent resource management problems from reaching crisis proportions, an important aspect is establishment of national status and trends for ecosystems beginning with already existing data to help identify declining species and ecosystems (DOI 1993a, 1993b).

The NBS has three main components: 1) biological research; 2) the development of biological inventories and monitoring systems; and 3) information transfer

to resource managers and other scientists. Basic biologic research includes: bench and field research; inventories; monitoring; and status and trends surveys on plants and animals, including their systematics, physiology, behavior, ecology, habitats, diversity, and ecosystem processes and functions. The NBS centers on the collection and generation of basic scientific information using hypothesis testing and other basic research approaches, independent of management; however, the independent data generated via the NBS is intended to be used in decisionmaking (DOI 1993a, 1993b).

The National Research Council (NRC) Committee on the Formation of the National Biological Survey cited the importance of the NBS in addressing the following issues:

- Identifying methods to preserve biologic resources.
- Sustainable management of biological resources.
- Determining ecosystem operation for continuation of functions such as maintenance of water supply, erosion and flood control, and climate amelioration.
- Comprehending impact of human activity on biologic resources.
- Continuing current biotic contributions to quality of life.
- Comprehending anthropogenic and natural effects of climate change on biologic resources.
- Identifying new economic advantages from biologic resources.
- Restoring degraded environments (NRC 1993).

The NRC committee recommended that the NBS should have the following scientific objectives:

- Identify existing biological specimens and data throughout U.S. institutional collections.
- Discover, describe, classify, and map U.S. species of selected taxa.
- Establish taxonomic specialists, collections and data bases.
- Study biology of selected species of importance.
- Develop classification systems for ecological units and set of core ecosystem attributes and protocols.
- Develop predictive models for sustainable management.
- Research the restoration of degraded environments.
- Perform research to develop biological protocols for pollution and to identify biological indicators of ecological trends.
- Establish collaborative pilot projects for research on biological resources in regions of the U.S., targeting areas that are changing rapidly because of human activity, are highly biodiverse, or are unique ecologically (NRC 1993).

The NRC committee also recommended that a National Biotic Resource Information System be established for ready data collection and dissemination (NRC 1993).

NBS maintains a research program that provides objective scientific data for issues relevant to the Interior Department. Divisions within the research program include species biology, population dynamics, and ecosystem research. Cooperative Research Units are located throughout the country and combine Federal, State and university resources for scientific research. NBS regional field units implement NBS programs developed in cooperation with the Secretary of the Interior, Congress, and other partners. Some field units have specific scientific capabilities such as aquatic sciences or terrestrial ecology.

Other NBS initiatives include its State Partnership Program to develop state-level capability for increased access to and integration of biological information. Coupled with this is NBS's National Biological Information Infrastructure initiative to facilitate a network of data bases and information sources on biological resources.

NBS maintains an Inventory and Monitoring Program. Current efforts are to identify existing biological information. NBS is developing a series of Status and Trends Reports of U.S. biological resources to create a format for biennial reporting, trend data, and to identify data gaps.

Monitoring aspects include species and ecosystem trends. A specific project through the Inventory and Monitoring Program is the Gap Analysis Program. Gap is a national research program to look at biodiversity through geographic display of vertebrate and plant species in concert with land use. In particular, it attempts to identify whether or not managed lands are maintaining species diversity and

preventing species from becoming threatened or endangered (Scott et al. 1993). A proposed Gap project for New Jersey is discussed in III.B. below.

The Success With Species-At-Risk initiative is to conduct studies through partnerships with land managers (government and non-government) and academics for scientifically-grounded management decisions. NBS also has a series of specific Ecosystem Projects to coordinate regional information exchange and scientific activities among government, non-government, academic, and private organizations.

The NBS Museum Initiative is to bring together various organizations with museums for better information exchange regarding the biological information available within museum collections.

3. Ecological Assessment: NOAA

The National Oceanic and Atmospheric Administration (NOAA) is involved in ecological assessment in remediation and restoration from environmental releases, as well as in development of baseline ecological data. As a trustee of natural resources, NOAA, provides technical support to USEPA and States under the purview of CERCLA/SARA, the Clean Water Act, and the Oil Prevention Act to regain costs for cleanup and restoration of marine and coastal resources including the range of migratory fish. NOAA's focus is on both assessment of releases as they impact wetlands, as well as impacts of the remedy (Wehner 1993).

NOAA is also involved in baseline ecological assessment and bioeffects monitoring. NOAA conducts the National Status and Trends Program for Marine Environmental Quality to monitor spatial distributions and temporal trends of contaminant concentrations and respective biological responses in coastal and estuarine regions of the U.S. Contaminant concentration and distribution is measured in bivalves and sediments through NOAA's Mussel Watch Project at 200 sites nationwide. Contaminant concentration and distribution are also measured in bottomfish and sediments through NOAA's Benthic Surveillance Project at 80 sites nationwide. Evaluation of data from these programs identifies areas where more intensive study is warranted. In New Jersey, this includes the NY/NJ Harbor Estuary and Delaware Estuary Programs in collaboration with other state and federal agencies.

4. Ecological Assessment: USDA

Ecological assessment applies to the field of agriculture in numerous ways, often integrating with other federal, state, and local programs. Managed agronomic practices may be less "random" in potentially adverse releases to the environment than unplanned environmental releases or those releases we are more apt to characterize as pollutant releases, such as intentional waste disposal. The difference may be that agronomic applications of potentially harmful materials to the environment are intentional because we are trying to derive some agronomic benefit. Agronomic impacts from pesticide or fertilizer applications however, may be as ecologically significant or more so than unplanned releases if their risks have not been appropriately assessed prior to release.

Although most agronomic practices have not undergone ecological risk assessment, as stated by the Congressional Office of Technology Assessment (OTA):

> Biotechnology has appeared on the scene during a time of intense environmental and political scrutiny of new technology...planned introductions of recombinant DNA-modified organisms will occur in a regulatory climate vastly different from that which existed as dramatically new crop varieties were introduced in the past. (OTA 1992).

Agricultural biotechnology products intended for use in the environment include transgenic cows, insect resistant field crops, microbial pesticides on field crops, and transgenic fish in aquaculture (OTA 1992). The overriding ecological concern with respect to genetically modified organisms is the uncertainty of their impacts on local species. USDA oversees biotechnology programs in the areas of plants, veterinary biologics, and animals.

USDA currently issues permits for field testing of genetically-engineered organisms; however, USDA coordinates with EPA in areas where the two agencies have overlapping jurisdiction. This overlap applies to pesticides regulated by EPA under FIFRA (for manufacture, processing, distribution, and use of pesticides) and the Food, Drug, and Cosmetic Act (where EPA has authority to set pesticide tolerance levels in food and feed), and new chemicals regulated by EPA under the Toxic Substances Control Act.

The primary regulatory authority used as the basis for USDA regulation of plants with respect to environmental releases of genetically-altered organisms are the Federal Plant Pest Act and the Plant Quarantine Act. These statutes are intended to prevent the entry into or dissemination within the U.S. of living organisms considered dangerous to agriculture (OTA 1992). An Environmental

Assessment (EA) is prepared by the USDA Animal and Plant Health Inspection Service (APHIS) in determination of whether and under what conditions a genetically manipulated organism may be released into the plant environment. The EA is based upon information submitted to APHIS by the entity requiring permission for such a release. The Environmental Assessment contains procedural and physical precautions against risk, environmental effects, and background biology (OTA 1992).

Environmental risks from veterinary biologics are assessed under the Virus-Serum-Toxin Act as amended by the Food Security Act of 1985. APHIS conducts case-by-case reviews of applications for field trials, such as the field release of a genetically-engineered rabies vaccine. Recombinant DNA products must be shown to be pure, safe, potent, and efficacious, and not worthless, contaminated, dangerous, or harmful, with assurance of lack of negative effects on the environment and human and animal health prior to licensing. APHIS requires information on human safety, ecological concerns, vaccine virus characterization, and animal safety data. Ecological assessment includes biochemical characterization of the biologic organism, characterization of the environment into which it will be released, its host range, and its potential effects on other species (OTA 1992).

Environmental risks from transgenic animals or animals treated with recombinant DNA materials can be assessed via the Federal Meat Inspection Act and the Poultry Products Inspection Act under USDA's Food Safety and Inspection Service for the safety, wholesomeness, and proper labeling of food products made from domestic poultry and livestock. Livestock or poultry intended for research purposes cannot be slaughtered until sufficient data demonstrate that the edible products derived from such animals are safe for human consumption (OTA 1992).

B. Ecological Assessment/Ecosystem Management In New Jersey

In New Jersey, there are numerous ecosystem management initiatives and applications of ecological risk assessment. Many of these activities integrate with scientists at the federal level, as well as scientists in academia and members of the public. The ecosystem management programs which are relatively large in geographic scope involve multiple assessments or compartments, while assessments for individual sites usually center on a much more localized assessment of a single site.

Ecological assessment is also used in deriving regulatory standards and criteria in New Jersey. The distinction in ecological assessment applied to standard setting is that universal exposure assumptions and/or data sets for various environmental pathways of concern are used, as opposed to site specific exposure characterization.

Surveillance and monitoring data is collected through various NJDEP programs, much of it for regulatory compliance purposes. These data, sometimes can be utilized as indicators of regulatory success, as well as input data for ecological assessment.

Discussion of these major areas, as well as a description of the NJDEP Geographic Information System (GIS), is provided. Some of the more recent initiatives to integrate ecosystem management programs throughout the state are discussed.

Scattered throughout the following discussions on current ecosystem programs in New Jersey are some examples of the essential research that is taking place to coincide with such programs.

[Note that this is by no means a comprehensive listing of all the ecosystem programs in New Jersey; rather, it is a sampling of some of the more recent initiatives with an emphasis on NJDEP programs or NJDEP-sponsored projects. There are numerous ecosystem programs which have been ongoing through the generous commitment of various environmental, academic, and other groups throughout New Jersey. These programs include activities such as preservation, restoration, acquisition, research, education, data collection, clean-up, etc.]

1. Ecosystem Management in New Jersey

In New Jersey, ecosystem management has been consciously fostered for quite some time. Additionally, there are a number of continuous research programs which catalogue biological data as a basis for effective ecosystem management. Again, although not an exhaustive list, some of these programs are briefly described below.

Creation of the Pinelands National Reserve in 1978 was a landmark effort to conserve an entire ecosystem. Through both a federal law and a state law the Pinelands Commission was created to protect Pinelands resources and manage growth in the one million acre Pinelands area.

NJDEP's Division of Science and Research (DSR) is

currently participating in a study to ascertain the appropriate agronomic, yet environmentally safe, rate of land application of compost in the Pinelands for mining site reclamation and revegetation. A specific component of this research is to ascertain whether or not there will be any significant impacts to native vegetative species in the Pinelands, as well as impacts to ground and surface waters, from the application of this biosolid-derived material in comparison to synthetic chemical fertilizers to the unique Pinelands environment. Principal scientists in this research are from the USGS and Rutgers University. Scientific review includes scientists from the Pinelands Commission, USEPA, and the USDA.

NJDEP participates in the Delaware Estuary and NY/NJ Harbor Estuary Programs through our Division of Science and Research and Geographic Information System. These major ecosystem management initiatives were developed in response to EPA's designation of these two systems as estuaries of national significance. They have been on-going for the past six years. Research is being undertaken in the areas of sediments, biota, habit, toxics, pathogens, and nutrients. Cooperating agencies include NJDEP, representatives of Delaware, New York, Pennsylvania, U.S. EPA, and NOAA. Draft management plans were approved in December 1994. Implementation of these plans will coincide with the plan objectives to protect the water quality of the living resources and maintain the ecosystem for beneficial uses (Gastrich 1995).

As previously mentioned, if feasible, application of the data generated from these two programs to already-developed models specific to these ecosystems (Kelly and Spofford 1977; Thomann et al. 1989) could be useful for establishing the suitability of such models to New Jersey ecosystems.

NJDEP's Division of Science and Research is sponsoring several investigations concerning fisheries, water resources, and contaminants. These include a study on mercury in fish, a finfish resource assessment in New Jersey estuarine waters, and pollutant fate, transport, and accumulation in Barnegat Bay (NJDEP 1994e).

As part of its 1991 "Last Great Places" initiative, The Nature Conservancy (an international, private, nonprofit organization) targeted the Delaware Bayshores as one of 12 ecosystems for initial protection (TNC 1993). The New Jersey and Delaware Chapters of The Nature Conservancy are implementing a strategic plan for preservation of the Delaware Bayshores Bioreserve (Heasly 1995). The plan includes: protection of critical habitats; encouragement of appropriate land management; promotion of research; public outreach; and cooperation with the local business community to address economic concerns (TNC 1993). The plan is being implemented in cooperation with numerous public and private partner organizations (TNC 1993; Heasly 1995).

Another ecosystem management effort here in New Jersey has been the 1993 creation of the Highlands Trust Advisory Board with primary emphasis on watershed protection and flood control (Galli 1994). The Highlands is approximately .75 million acres in size (or approximately 15% of New Jersey's land area) and supplies drinking water for over 4 million residents of New Jersey (Galli 1994). The Advisory Board, comprised of local, regional, and state representatives, is to identify priority land tracts for conservation and respective management practices.

NJDEP is moving toward comprehensive ecosystem-wide watershed management, rather than the current framework of addressing specific system components such as ground water, surface water, or individual dischargers (NJDEPE 1994c). A watershed is the geographic region in which water, sediments, and dissolved materials drain to a certain receiving water body (NJDEPE 1994c).

Comprehensive watershed management is the integration of current water resource protection efforts with new and innovative methods within a flexible framework that accounts for the dynamic relationships that exist within a watershed and which sustain natural resources and their beneficial uses. (NJDEPE 1994c)

Watershed planning can be achieved through five steps: data base development (mapping boundaries, hydrology, geologic data, land use, infrastructure, and ambient water monitoring); data assessment (analysis of water quality and flooding problems, inventory of pollutant sources, mapping future land use, and identifying infrastructure needs); goal setting (developing models to ascertain loading goals); strategy development and implementation; and monitoring and evaluation to update and fine-tune the watershed plan (NJDEPE 1994c).

NJDEP has initiated the Whippany River Watershed Pilot Study to test the methodology cited above, as well as integrate planning, permitting, modeling, monitoring, financing and enforcement (NJDEPE 1994c). Since the objective is to develop regional goals, the Whippany project involves all the community stakeholders with an interest in water resources. A project strategy has been developed and a water quality analysis plan has been drafted (NJDEP 1994b).

Within NJDEP's Division of Fish, Game, and Wildlife, the Endangered and Nongame Species Program (ENSP) directs a number of research, survey, and management activities that address its mission of actively conserving the state's biological diversity by maintaining and enhancing the state's endangered and nongame wildlife populations within healthy, functioning ecosystems (NJDEPE 1993e).

ENSP research projects include studies on: the effects of contaminants on various populations of birds; habitat of various species; and the causes of decline of various populations. Survey projects include shorebirds, raptors, beach nesting birds, herptiles, and various other vertebrate and invertebrate surveys. Management activities include population restoration, development of management plans for specific ecosystems, and monitoring and management of beach nesting bird sites, as just a few examples of the many ENSP management programs. (NJDEP 1990; NJDEPE 1993c).

The ENSP program also oversees revision and publication of the endangered and nongame species list for New Jersey and integrates this information with the Natural Heritage Database and Notable Information on New Jersey Animals (discussed below in III.B.4).

The ENSP is conducting a Landscape Project with the ultimate goal of long-term protection of rare animal species. Principles of landscape ecology are being used to define landscape protection areas (LPAs). The first LPA being addressed by the Landscape Project is the Coastal Plain (including Cape May and Cumberland Counties). An assessment of rare species (including endangered, threatened, and species with special habitat needs) is being conducted for the LPA based on existing data and field surveys for herptiles, birds, invertebrates, and fishes. Critical habitats within the landscape protection area will be mapped and guidelines for long-term species protection will be developed. Ultimately, these work products will be provided to land use regulators for integration with management decisions to ensure species protection. (Bowers-Altman 1994).

The NJDEP Division of Parks and Forestry, Office of Natural Lands Management, administers a number of programs that coincide with its mission of natural resource protection through habitat management (NJDEPE FY 1994). These programs focus on habitat protection and passive recreation including the Natural Heritage Program and Endangered Plant Species List (discussed below in III.B.4); Natural Areas Program; Natural Lands Trust; Open Lands Management Program; Trails Program; and Wild and Scenic River Program (NJDEPE FY 1994).

New Jersey maintains over 40 state-designated Natural Areas (approximately 30,000 acres) through the Natural Areas Program (NJDEPE 1993d). Originally established by the New Jersey Legislature in 1961, the goal of the New Jersey Natural Areas System is to preserve and protect lands that support endangered and threatened plants and animals, significant ecosystems, and important wildlife habitats through acquisition, maintenance, and preservation (NJDEPE 1993d; NJDEP 1994g). Natural Area management plans are designed to foster ecosystem and species preservation with the assistance of local naturalists and university researchers (NJDEPE undated).

The Office of Natural Lands Management also sponsors research on endangered plant species in New Jersey, including identification of threats to such species and development of protection plans (NJDEP 1994g). Research results are to be incorporated into the Natural Heritage Database, as well as in developing the Endangered Plant Species List (N.J.A.C. 7:5C-2.1).

The Natural Lands Trust is a state-funded land preservation organization to secure permanent protection of land and diversity preservation. The Open Lands Management Program provides financial assistance to private property owners who allow their land to be used for public passive recreation. The Trails Program studies and recommends sites for acquisition and trail development and initiates planning for existing trail management. The Wild and Scenic Rivers Program focuses on river protection and preservation for environmental, recreational and aesthetic values; identifies resources and recreational opportunities within a river corridor; and develops river management plans (NJDEPE FY 94).

NJDEP's Bureau of Forest Management operates within the Division of Parks and Forestry. The Bureau of Forest Management monitors forest health, participates in urban and community forestry projects, oversees a forest tree nursery, conducts conservation education, processes applications and inspects private land forestry operations, manages approximately 300,000 acres of state parks and forests, and conducts forest research projects (Coutros 1994).

NJDEP has recently released a draft resource management plan for the Bass River State Forest (NJDEP 1994a). Plan objectives include identifying old growth stands and ecologically significant

communities and developing management strategies to provide the recommended level of biological diversity for the 16,798 acres of forest (NJDEP 1994a). The plan also addresses watershed protection, and designates management areas considering soils, forest cover types, watershed, rare species and activity or purpose (NJDEP 1994a).

Along with the Division of Parks and Forestry, NJDEP's Division of Science and Research is supporting research on Atlantic white cedar regeneration to address the decline of this species and develop management strategies.

In addition to the specific projects cited above, DSR is also cooperating on a number of ecologically-related investigations including developing methods for contaminant measurement, identifying contaminants in various media, and developing methods for remediating contaminated resources (NJDEP 1994e).

2. Ecological Assessment at Known Contaminated Sites in New Jersey

Specific NJDEP program offices in which site or region-specific ecological assessments are conducted or reviewed include the Environmental Toxicology and Risk Assessment (ETRA) Unit within the Division of Publicly Funded Site Remediation and the Office of Natural Resource Damage Assessment within Natural and Historic Resources.

Ecological assessments reviewed by ETRA are required under the CERCLA Remedial Investigation and Feasibility Study (RI/FS) process using the EPA protocol established in the Risk Assessment Guidance for Superfund (also known as RAGS). NJDEP provides review and oversight of assessments for these federal Superfund sites through the auspices of the Biological Technical Assessment Group: a regional group composed of representatives from EPA, NOAA, Department of the Interior, NJDEP, and New York State Department of Environmental Conservation.

Pursuant to P.L. 1993, c.139 (S-1070) and Technical Requirements for Site Remediation (N.J.A.C. 7:26E, to be reproposed 12/95), the NJDEP Site Remediation Program shall address ecological concerns at all non-CERCLA contaminated sites on a case-by-case basis until an Environment Advisory Task Force established by S-1070 makes recommendations regarding ecological standards (Hamill and Neumann 1995).

Staff experienced in the use of the RAGS document (cited above), ensure that all sites are evaluated for potential ecological impacts using a phased approach. A Baseline Environmental Evaluation is conducted during the initial investigation phase using qualitative screening techniques to assess whether contaminants of ecological concern exist on site, whether a "designated natural resource" (as defined in N.J.A.C. 7:26E-1.8) exists on or adjacent to the site, and whether a potential contaminant migration pathway to the designated natural resource exists or is apparent.

A case team determines whether further investigation via an ecological risk assessment is required. These more rigorous assessments are typically required if contaminant concentrations exceed pre-determined screening levels such as surface water quality criteria or sediment quality guidelines, or a potential for risk exists via a terrestrial exposure pathway. Such ecological risk assessments are conducted and reviewed in accordance with U.S. EPA guidelines (Hamill and Neumann 1995).

A more detailed assessment of ecological effects may require an on-site habitat and wildlife inventory, sediment toxicity testing, benthic macroinvertebrate surveys and tissue analysis (Hamill and Neumann 1995). Exposure modeling for assessing food chain effects generally include 3 rodent species (herbivore, carnivore, and omnivore), small mammals, soil arthropods, and native vegetation (Hamill and Neumann 1995; Sacco 1993). For example, in cases where deer are a concern, vegetation that deer would consume, such as blueberries, will be included in the assessment for pathway analysis. (Sacco 1993).

ETRA scientists primarily assess risk using a hazard quotient approach. Data from the field are compared to literature values for No-Observed-Adverse-Effects-Levels. If the quotient is less than 1, ecological risk is not assumed to be present. Ecologically-based clean-up levels are established based on acceptable risk for the species currently on-site or for target species representative of a specific trophic level (Sacco 1993).

The Office of Natural Resource Damage Assessment (ONRDA) within the NJDEP Natural and Historic Resources Program is responsible for assessing damages to resources owned or managed by the state and federal government from releases or threatened releases to these resources. NJDEP works with various federal agencies (including EPA and NOAA) to assess resource damage in economic terms and to recover costs for current injury, costs for potential injuries from a remedy, and costs for restoration of the resource. Authority for the reclamation of costs

comes from federal statutes such as CERCLA and the Oil Pollution Act of 1990, as well as the New Jersey Spill Act. Assessment of natural resource damage is often limited by lack of baseline data concerning the status of resources (species existence, habitat data, and their respective economic value) prior to environmental releases (McHugh 1993).

3. Ecological Assessment in Deriving Standards and Criteria in New Jersey

Ecological endpoints are considered in various standards and criteria specific to New Jersey. These include surface water quality criteria, soil standards, standards for sewage sludge and sludge-derived materials, and requirements under the Clean Air Act. Prior sections of this report addressed some of the operational considerations in the development of ecological-based criteria for surface water and sediments as currently applied in New Jersey.

New Jersey is in the process of establishing soil standards for the cleanup of contaminated sites. Two sets of standards are to be developed: one to address ecological effects and another to address human health endpoints. The first proposal of the standards (February 1992) primarily addressed human health endpoints; however, because of their phytotoxicity, copper and zinc standards were proposed based upon this ecological endpoint (24 N.J.R. 376). The proposed cleanup standards were not adopted and NJDEP is currently using soil cleanup criteria when evaluating hazardous waste sites. For the most part, the soil cleanup criteria are the same as the proposed soil standards with a few exceptions.

The 1993 Industrial Site Recovery Act (P.L.1993, c.139) established an Environment Advisory Task Force to recommend soil remediation standards protective of the environment including ecological receptors and an Environmental Risk Assessment and Risk Management Study Commission to develop recommendations concerning human health standards.

Future revisions to the soil cleanup criteria will not include the phytotoxicity values for copper and zinc because the legislative mandate requires separate standards for ecological and human health endpoints to be consistent with recommendations by the Environmental Advisory Task Force and Risk Assessment Study Commission, respectively. The Environmental Risk Assessment and Risk Management Commission is preparing a final report on human health risk assessment methodologies; however, selection of Environment Advisory Task Force members to address ecological effects has not yet been completed.

NJDEP is also in the process of adopting revised EPA standards for the beneficial use of sewage sludge-derived materials in land application practices. These standards also account for multiple human health and ecological pathways. With respect to land application of sludge, phytotoxicity was the limiting pathway for chromium, copper, nickel, and zinc (58 Fed. Reg. 9248-9415).

4. Environmental Indicators and Data Bases in New Jersey

Environmental indicators can be used to assess status and trends in the environment's ability to support ecological and human health. This information can provide environmental managers, elected officials, and the public with information about the state of the environment, and hence its utility in ecological assessment.

The NJDEP Division of Science and Research (DSR) is conducting a research project to evaluate New Jersey's ambient monitoring programs and develop environmental indicators for New Jersey. DSR has coordinated its effort with the 1992-1993 effort of the NJDEP Office of Policy and Planning Sabbatical Team. Together, DSR and the Sabbatical Team, conducted an environmental information survey throughout NJDEP in October 1992. The Sabbatical Team sponsored a March 1993 Northeast regional conference (NJDEPE 1993a), as well as a June 1993 workshop on environmental indicators.

The DSR project builds upon the original environmental information survey as part of an effort to compile detailed information on New Jersey's ambient monitoring network. A compendium of approximately 40 NJDEP ambient monitoring programs is being developed. A subset of these monitoring programs are being evaluated for development of environmental indicators. In addition, the Ambient Stream Monitoring Network is being evaluated as a case study. Some of the water column parameters that are being developed into indicators include dissolved oxygen, fecal coliform, ammonia, nitrate, chloride, copper, mercury, and lead. Sediment indicators may include copper, lead, mercury, DDT and metabolites, PCBs, and phosphorous.

In addition to the Ambient Stream Monitoring Network, the Water Monitoring Management Program within DSR maintains a series of monitoring programs which are used to evaluate status and

trends, classify waterways, identify environmental health problems (human and ecological), and to devise strategies for enhancing water quality. These data are generated through the ambient ground water monitoring network, a lakes sampling and assessment program, ambient biomonitoring of watersheds, the ecoregion biomonitoring program (described previously), coastal plankton monitoring, zebra mussel monitoring, and shellfish and sediment monitoring.

An ongoing environmental indicator assessment is the New Jersey Water Quality Inventory Report which is prepared every two years. The report assesses water quality in New Jersey's rivers, lakes, estuaries, oceans, and ground water including trends in water quality, pollution problems and sources, and recommendations for improving water quality in New Jersey (NJDEPE 1993b).

A summary of New Jersey air quality can be found in the yearly air quality report published by the NJDEP Bureau of Air Monitoring (NJDEPE 1994d). Data are collected via air monitoring networks throughout the state. Parameters measured include the NAAQS criteria air pollutants, benzo(a)pyrene, some trace metals (arsenic, barium, cadmium, chromium, copper, iron, magnesium, manganese, nickel, potassium, vanadium, and zinc), acid precipitates, etc. (NJDEPE 1994d).

NJDEP maintains several natural resource data bases which can be utilized for assessing biologic status and trends. Notable Information on New Jersey Animals (NINJA) is a central repository of data on New Jersey's fresh water and terrestrial wildlife resources. NINJA contains information on over 550 New Jersey species including taxonomy, life history, habitat, and distribution. Information from the NINJA data base can be used for numerous purposes including evaluation of species habitat preferences, occurrence, and various management decisions. (Herrighty 1994). Data generated from research such as New Jersey's black bear tracking program is input to NINJA (McConnell 1994; Herrighty 1994).

The NJ Natural Heritage Database is an inventory of rare plants and animal species and representative natural communities in New Jersey. The data base tracks over 1,000 species of plants and animals and more than 50 natural communities including species listed at the federal and state levels as endangered, as well as rare species. The data base is part of an international network of data bases throughout the U.S., Latin America, and Canada. Data can be used for management decisions, as well as priority setting for preservation. (Cartica 1994a; 1994b)

The Natural Heritage Database is developed through New Jersey's Natural Heritage Program. Amendments to New Jersey's Endangered Plant Species List can be made based on new or updated information in the Natural Heritage Database (N.J.A.C. 7:5C-2.1).

The Natural Heritage Program interfaces with the ENSP to integrate ENSP animal data into the Natural Heritage Database. In addition, the continuous survey and research projects for numerous species conducted through the Endangered and Nongame Species Program can be considered valuable environmental indicator data sources.

Deer management in New Jersey includes data on herd health, habitat loss, and deer damage. These data are tracked through NJDEP's Deer Management Program (Ferrigno 1994).

Data on hard and soft clams, invertebrates, polychaetes and mollusks, as well as water quality are collected by NJDEP's Bureau of Shellfisheries (Joseph 1994).

Fish, invertebrate, and water quality data are tracked by NJDEP Division of Fish, Game and Wildlife's Bureau of Marine Fisheries. This includes an ocean stock assessment and a tagging program for flounder, sturgeon, and striped bass (NJDEP 1994d).

The Office of Fish and Wildlife Health and Forensics maintains data on bacteria, toxicology, and pathology of fish and wildlife in New Jersey to ascertain cause of death and track disease occurrence (NJDEP 1994d).

Data on conventional pollutants, metals, and organic chemicals are submitted to NJDEP through various permitting programs and are usually media-specific. In addition, the site remediation program receives data pertaining to site cleanups for soils, sediments, surface and ground water (Hamill 1994).

Various other data sources that could be used as environmental indicators exist throughout NJDEP, as well as throughout New Jersey (academic researchers, permitted dischargers, volunteer monitors, naturalist club surveys, water purveyors, etc.). A comprehensive list has not been developed.

5. The NJDEP Geographic Information System

The NJDEP Geographic Information System (GIS) is a sophisticated tool for mapping a wide range of ecological data. At the current time, the GIS

contains an enormous quantity of New Jersey specific digitized information in three general areas: ambient monitoring; regulatory compliance; and natural resource baseline data.

Ambient monitoring data includes: air quality monitoring stations; nuclear engineering monitoring stations; ambient surface water quality monitoring; coastal water quality; and groundwater quality monitoring.

Examples of regulatory compliance data available on the GIS are the New Jersey Right-to-Know Toxic Release Inventory and Hazardous Waste Site data.

Natural resource baseline data comprises a wide range of information including: shellfish distribution; endangered and non-game species; fisheries data; migratory waterfowl staging areas; open space; land use/land cover; wetlands; Atlantic white cedar stands; environmentally sensitive site data from NOAA maps; and water supply boundaries.

The NJDEP GIS has recently completed a User's Guide for New Jersey geographic information that is being translated for posting within the World Wide Web. As NJGIS is made available to the public (via libraries, non-governmental organizations, and other venues), students, citizens, businesses, etc. will have access to environmental information for a particular geographic location in New Jersey. GIS should be consulted and incorporated into ecological assessment initiatives wherever possible.

6. Current Initiatives for Integrating Ecosystem Assessment

Several statewide initiatives have begun for integrating ecosystem assessment throughout New Jersey. With ecological risk assessment, ecological impact assessment, and ecological management cutting across numerous programs within NJDEP, the Division of Science and Research, together with the Division of Fish, Game and Wildlife, has initiated a series of meetings to bring together scientists from throughout NJDEP to discuss these issues, program needs, and explore how the programs might integrate (NJDEP 1994c; 1994d). Relationships among staff scientists throughout NJDEP have developed and increased information sharing has resulted as an outcome of these meetings.

The U.S. Fish and Wildlife Service is pursuing a cooperative project with NJDEP's Office of Natural Lands Management, Geographic Information System, and Division of Science and Research on a Gap Analysis for New Jersey. As described in a prior section, Gap is a project to geographically display terrestrial and vegetative species data with land use data in an attempt to identify habitat types and species that are not represented adequately in existing biodiversity management areas (Scott 1993).

New Jersey might be able to expand the current Gap protocol by improving the scale of the assessment to a better level of precision using the NJGIS land use/land cover data layers. In addition, the project could be developed at the outset with input from the Division of Site Remediation and the Office of Natural Resource Damage Assessment as these programs are in need of baseline and reference data. A Gap Analysis project would therefore, serve multiple purposes.

Another statewide initiative currently in development is a proposed alliance for biodiversity in New Jersey. A December 1994 meeting, hosted by Rutgers University, explored issues, opportunities, and action steps for developing a biodiversity initiative in New Jersey, including a New Jersey Biological Survey. Scientists from academic institutions throughout New Jersey (Princeton, Rutgers, Stockton, etc.) joined with representatives from environmental organizations, industry, agriculture, and government to share ideas and develop proposals. Scientists from NJDEP's Division of Science and Research and Office of Natural Lands Management participated in the sessions.

In March 1995, DSR initiated the New Jersey Environmental Research Partnership to facilitate coordination of basic environmental data and data needs for New Jersey. The partnership, which continues to expand, includes representatives from academia, government, non-governmental organizations, industry, and environmental organizations. Goals include developing an electronically-available resource guide for New Jersey environmental data, linked with related directories. Information could include environmental data and data bases, researchers, scientific applications for decision making in New Jersey, specialized equipment, and training and internships. A series of symposia concerning regional ecosystems throughout the state, as well as concerning issues specific to New Jersey (watershed, wetlands, forest fragments, etc.) will ensure inclusion of local naturalists, volunteer monitors, and other organizations or persons knowledgeable about New Jersey ecosystems.

DSR is coordinating this partnership effort in conjunction with NJDEP's GIS. Ideally, directory information should be geographically based so that

user's can identify site specificity or regional availability of data. GIS has extensive experience in locational data bases and, as mentioned previously, the NJGIS is currently translating its Resource Guide for posting within the World Wide Web.

C. Ecological Assessment in Other States

Other states around the country have begun to develop more formal ecosystem management plans. California has been divided into ten contiguous bioregions in which long-term planning for conservation and economic development will be coordinated by local, state, and federal representatives. An initiative for restoration of the oak savannas in the Midwest has recently begun as a cooperative effort among the U.S. EPA, University of Wisconsin, Northeastern Illinois State University, and the Nature Conservancy. (Stevens 1993c). These are just some examples of the numerous ecosystem management programs which are being pursued around the country.

IV. Outstanding Issues in Ecological Assessment With Suggested Resolutions

There are numerous scientific and policy issues that remain outstanding as the field of ecological assessment continues to develop. Ascertainment of data for resolving many of the scientific issues may take considerable time. Some of the questions may require a general consensus within the scientific community, as opposed to strict scientific determination. Resolution of the policy issues will probably be an iterative process and answers to such questions will be adjusted as analysis of scientific data proceeds, in concert with an increasing level of public understanding and sophistication about ecosystem assessment and management.

Below is a brief discussion of some of the more significant outstanding issues concerning these matters and some suggestions for resolve of these issues. The final section of this document sets forth a framework for issue resolution.

A. What is Ecological Assessment?

Defining ecological assessment is a fundamental issue in ecosystem assessment. Purists might argue that examining toxicity in an individual species or organism, as opposed to population level or ecosystem level effects, does not make for an ecological assessment. However, organism level information is often the only source of data available.

Are we concerned with direct effects only? What about indirect effects, such as the reduction in prey base for a predator species? Have we identified a reference point, reference site, or what magnitude of response signifies an effect? Also, have we accounted for alterations over time?

Suggestions: Problem formulation needs to be thorough. As such, development of the analytical approach is a critical aspect to designing the ecological assessment. If we can only examine an organism level effect, we have to be certain we do so in the context of its relationship to the system as a whole, not just within a simple dichotomy of organism toxicity or no organism toxicity. An assessment design should be checked against the standard of whether the problem as stated will evaluate organism level and systemwide effects as a whole (or at least be some sort of qualitative indicator of the system as a whole). The assessor should also check the design to be sure the stressor-response analysis will address the stated problem.

B. What is Ecosystem Health?

As previously mentioned, a core issue in ecological assessment is determining what makes a healthy ecosystem? How do we define ecosystem health? How do we measure the health of an ecosystem? Again, are measures related to organism level toxicity perhaps missing subtle chronic effects to the system?

Suggestions: An iterative approach may help define ecosystem health. The first aspect would be to develop a set of criteria for defining ecosystem health and implement the steps necessary for evaluating system health based on these criteria. For example, one criterion might be to meet certain requirements established at reference sites. If reference sites were not available, their identification might be required. (As mentioned previously, reference sites are being established for water quality based on benthic macroinvertebrates.) Part of the criteria development may require some statewide consensus as to what level of impairment may or may not be acceptable in defining the health of a system.

It may be possible to develop some classification scheme for landscapes throughout New Jersey (including existing classification schemes such as stream classifications) in concert with policy decisions about the level of degradation, restoration, or maintenance of the existing natural resources within

the region.

Another important aspect in defining ecosystem health is analysis of actual monitoring data to assess system status and trends. Therefore, environmental indicators that could be used for various ecosystem assessment program needs should be developed to ascertain system health. A formal indicator program could be designed with two prongs: one being indicators necessary on a program-by-program basis that can be applied for ecosystem level assessments that are required within the program; and the second being a supplemental indicator program to address those measures of ecosystem health that may be missed via the individual media program requirements.

As previously mentioned, there are numerous indicators already existing within NJDEP including the various species surveys, water quality data, etc., as well as indicators being developed through a DSR research project. Indicators should be developed for receptors via all media (air, soil, sediments, water).

These indicators could be folded into a comprehensive framework for assessing ecosystem health in New Jersey. It may prove useful to develop a State of New Jersey's Environment Report on a four year cycle to ascertain the health of New Jersey systems, as well as provide an analysis of regulatory program efficacy and recommendations for improving such programs.

Finally, it would be important to revisit sites that were identified through various programs as being ecologically unhealthy, or where cleanups were required based on ecological risk, and ascertain the site status to determine the validity of those measures used to assess and define system health.

These approaches could be integrated with one another and refined, especially so that criteria for establishing system health are refined and developed appropriately for future assessment and management purposes.

C. Data Availability

The availability of solid baseline data concerning ecological systems is an outstanding issue at this time. Obviously, this issue is related to a number of other issues such as definition of system health and the ability to define the bounds of an assessment. Hopefully, if the National Biological Service initiatives proceed, a better characterization of the types and quality of existing ecological data should emerge. In addition, perhaps there will be improvements and amendments in the methods for data collection and reporting such that data quality will be ensured in a systematic approach fundamentally understood by scientists within all purviews of ecosystem assessment.

In the meantime, applied ecosystem assessment relies on a combination of original research and data found in the scientific literature. Developing original data is certainly preferable, but the issue of limited literature information, which very often forms the basis for risk assessment and ultimately risk management decisions, remains outstanding.

Suggestions: The issue of data availability in ecosystem assessment is universal. A comprehensive appraisal of this issue has been conducted specific to New Jersey, and echoes many of the universal data availability issues. Therefore, it is most appropriate to reiterate the work of the Strategic Information Needs Survey for New Jersey regarding ecosystem data.

In the Fall of 1992, the Division of Science and Research led a Strategic Information Needs Initiative to identify critical information needs agency-wide. Seven workgroups composed of scientists from throughout NJDEP developed the critical information needs for their respective areas. The Ecological Risk Assessment Group (ERAG), under the leadership of Larry Thornton (Geographic Information System), identified three basic conceptual information needs concerning the status and risks facing ecosystems in New Jersey:

1. Inventory, assess, catalogue and convert (to digital format either data base, scan or Geographic Information System where appropriate) all extant NJDEP ecological information. ERAG estimated that this task could be completed in 18 months.

2. Develop basic reference and inventory information, including digital maps and data bases regarding the ecology and interactions of species and habitats that define New Jersey ecosystems. Once critical baseline data layers were initially mapped, they could be revisited to establish trends. ERAG estimated basic data layer development would take 2 to 3 years to complete.

3. Conduct basic research on the effect of toxic materials on indigenous aquatic and terrestrial species and species communities.

As a supplement to the data needs identified by the NJDEP ERAG group, many of the initiatives

recognized in previous sections of this report could be integrated into a statewide data development initiative. There are numerous federal and state data sources that might be useful in ecological assessment applications but there has been no formal accounting of all data sources. Integration of these data sources could occur on a statewide basis. Briefly, these include:

- Coastal and estuarine data collected by NOAA, Army Corps of Engineers, USGS, USEPA, and NJDEP includes water column, biota, and sediments data.

- Formal data collection effort specifically for ecosystem protection being conducted under the auspices of the Department of Interior, National Biological Service.

- NJDEP-DSR's current environmental indicators project.

- Monitoring data required under state and federal programs that is often not published in scientifically peer reviewed literature.

- Other survey, census, and monitoring data collected throughout New Jersey via NJDEP program offices, naturalists, organizations, researchers, etc.

- New Jersey GIS Resource Guide developed by NJDEP's GIS.

- Other relevant data directories (e.g., Delaware Estuary Program's Regional Information Management Service).

1. Ecoassessment Research Needs for New Jersey

Discussions with colleagues throughout NJDEP has led to a number of research ideas. Where possible, research could be piggybacked onto existing ecosystem programs. For instance, the GAP Analysis for New Jersey could be expanded to help identify ecosystem reference sites useful for both the site remediation and natural resource damage assessment programs (see Section III.B.6).

Initiation of research on bioaccumulation and mixtures should be two priorities in the area of ecological assessment for New Jersey. While estuaries are one of the most productive of all ecosystems, many have been impacted by long-term commercial port activities. Estuarine sediments are sinks for contaminants associated with port and other industrial activities along estuarine systems. A critical need is data for characterization of trophic transfer of bioaccumulative contaminants through an estuarine food chain, from sediment to apex-level ecological receptors. Such a characterization could provide trophic transfer factors and biota-to-sediment accumulation factors that would facilitate estuarine risk assessments and standards-setting.

Valid mixture assessment methods are needed for evaluating environmental matrices contaminated with multiple compounds. While analytes of a complex mixture may be quantified, the biologic potency of such a mixture is not necessarily known because of potential chemical interactions, including the influence of unidentified constituents. Potential mixture assessment methods, such as in vitro enzymatic activity assays, could be evaluated for predicting biological potency.

Another useful research project would be to develop ecological toxicity profiles, including guidance for deriving no-adverse-effects concentrations, for compounds routinely encountered at New Jersey sites but for which comprehensive data are not currently available in a readily accessible form.

An important study could be an investigation of current criteria in relation to environmental samples to determine if indeed the criteria that have been developed for certain media were preventing receptor toxicity. In theory, criteria are predictors of levels that will prevent ecological impacts and are developed via ecological assessment. In practice do they work? Were we to develop a set of criteria to ascertain the "health" of an ecosystem, we could match these criteria to observed field conditions, as well as revisit sites that were remediated to ascertain "health".

Other major scientific gaps in which research should be initiated include: contaminant fate in the environment, including speciation and associated toxicity; systemwide indicators of ecological health; basic toxicity information on wildlife species; dose-response data; reference values; extrapolation of organism-level toxicity assessments to populations or community level impacts; impacts on vegetation and wildlife from emissions of toxic air pollutants; cumulative contaminant impacts including multiple sources and multiple contaminants; and an investigation of remediated sites to ascertain if ecological effects have been mitigated and to develop a method for predicting recovery potential.

In addition, translation of risk to action steps in which various sectors can participate or contribute to reducing risk is often difficult. Research could examine the potential multiple benefits and risk

reduction from using native species for restoration, landscaping, etc. including reduced water consumption, pesticide use, and fertilizer use with related reductions in human and environmental exposure. Would such a program also reduce risks from exotic species invasion, as well as provide the public with an opportunity to contribute to risk reduction statewide?

Research needs, those already identified and those which will develop as a result of our progress in the field of ecological assessment, will need to be prioritized and addressed. In addition, research should be coupled to risk communication, outreach, and participation among various sectors of the community.

Finally, whenever possible, research should be designed to include geospatial references so that data can be both spatially displayed and analyzed.

D. Data Applicability

When data are available, there are fundamental questions about their utility. A key area is the integration of problem formulation and endpoint selection. Even for a less complex site-specific assessment are existing data useful for site-specific ecological assessments?

Literature toxicity values are very often for species related to, but not the same as, those of concern. In addition, environmental toxicity to a specific organism may be very subtle and not observed or reported via a more traditional endpoint associated with effects in humans or mammals. For instance, toxicity or some type of population effect may be indicated and more strongly correlated with a subtle mechanistic effect, such as enzyme induction, rather than actual tumor formation.

Sampling data may be misrepresentative. For example (as identified in NJDEPE's 1991 Guidance for Sediment Quality Evaluation), suppose we are trying to assess ecological effects whereby the greatest contamination to sediments had occurred in the past. Near surface sediment samples would provide data for more recent deposition and would therefore not be representative of the contamination one is trying to assess.

The complexities of an assessment increase when looking at a multiple pathway analysis. For example, in assessing risk to birds from land application of sludge, EPA considered avian ingestion of sludge-exposed earthworms. However, there were no direct data in which sludge-exposed earthworms were fed to birds. There were, however, data on quail ingesting cadmium-enriched earthworms. EPA used the earthworms with cadmium data for quails as a surrogate for birds exposed to sludge but adjusted the exposure using a bioavailability factor from a study of pigs exposed to sludge. Such an approach is certainly less than ideal and may not be considered valid by some scientists. Yet, in the absence of real trophic transfer data, some might argue this approach is better than not considering this pathway at all.

Exposure characterization often incorporates field or literature values with models that can be overly simplistic. Such applications of data may result in overestimation or underestimation of ecological risk.

Suggestions: The following suggestions concern not only data applicability, but also several of the issues raised including data availability, scientific uncertainty, uniformity of approaches, etc.

First, develop guidelines for ecological assessors to follow that would be consistent across program areas. Guidelines could include a hierarchy of priority data types with associated caveats as available data move further away from assessment specifics (e.g., as species for which data are available become less similar taxonomically to species of concern), criteria for choosing a critical study or endpoint, appropriate risk methodology, etc.

Also, a working group on ecological assessment could begin to discuss the issues of concern, including data applicability, and begin to develop consensus about scientifically valid approaches. Such a group should include ecologists and scientists from academia, various government entities, and other organizations.

E. Scientific Uncertainty

As with human health assessment, numerous areas of scientific uncertainty exist. Issues of uncertainty center on:

- Whether or not the models used have been validated with field studies.

- The number of modeling parameters employed and the likelihood that uncertainty will increase as we increase the number of variables.

- Confirmation of causality. Have we identified potential confounders such as multiple stressors to a system?

- What are the appropriate uncertainty factors that should be applied when extrapolating values from the literature to ecological assessment?

Suggestions: As stated above, it would be very useful to develop a standard set of criteria for applying uncertainty factors to the quantitative aspect of an ecological assessment. Guidelines describing qualitative aspects of an assessment and how to consistently address qualitative issues should also be developed. Some of these issues could be discussed and developed through a working group as mentioned above. One issue to tackle could be determination of acceptable models for ecological assessment.

Determining causality is very difficult when hard data are not available. NJDEP is unique because our agency includes both the regulatory programs that limit toxics discharges into the environment, as well as the natural and historic resources programs. Were an indicator program to evolve, it would facilitate further linking of these programs to better characterize ecosystem health, as well as improve causality determinations. Specific data gaps would be identified and planning steps could begin to address these gaps, including initiation of research projects.

F. Risk Characterization

Once we have evaluated the data available for an ecological assessment, can we actually characterize risk either quantitatively or qualitatively? Do we have enough data to make any sort or judgment about risk? On an organism-level basis, we are probably more likely to have some evidence for judgment, or at least we can more readily identify the significant data gaps we face and move toward ascertaining the data we need for characterizing risk. On a population or system-wide basis, it may be more difficult to identify data gaps or find a way to fill such gaps.

Ecosystems are complex webs with natural fluctuations. There are periods of succession, for instance, when we discuss forest systems. Events in nature such as fires or floods require adjustments within systems. On a yearly basis, systems can have seasonal shifts. These sorts of changes over time should be considered on the micro and macro levels when conducting an ecological assessment.

Therefore, a fundamental issue in risk characterization is whether or not the perturbation we are assessing is significant in terms of the natural fluctuations of the system on either the macro or micro level. A question one might ask is whether or not we can identify the likelihood for recovery from a perturbation?

The conclusion of an assessment is significance; does the assessment show there will be a significant impact to the system? In human health assessment, we might quantitate risk from a policy of accepting a one in a million excess risk of cancer or a hazard quotient equal to or greater than 1 for noncancerous effects. For ecological assessment, we have yet to define a policy goal of what an acceptable risk is over what period of time.

In a sense, this issue links to that of how do we measure system health on a macro basis. On a micro scale, this question addresses the fundamental characterization for the more localized assessment such as whether or not a contaminant at a site will result in adverse effects to an organism or what level of a contaminant in surface water will be harmful to aquatic life?

Suggestions: On the larger scale, this can be addressed through defining ecosystem health through indicators, criteria, classification schemes, etc. as discussed in Section B. above. On the smaller scale, this issue can be addressed through establishment of a consistent set of approaches for ecological assessment and possibly through a working group reaching consensus on these approaches.

G. Regulatory/Policy Constraints: Lack of Uniformity

In addition to the limits of data, ecological assessment is bound by policy constraints that can be mandated directly through legislation, regulation, or some type of administrative policy. . Although a regulatory policy or piece of legislation may have an admirable policy goal it may miss the mark scientifically.

For instance, how is ecological risk incorporated into a program mandate? Even though some programs may require ecological assessment, the actual implementation may center more on judgment than on science. For example, permits might be based on achievable technology rather than ecological effects or a selected limit may be chosen based on the assessor's judgment that a species is economically or recreationally important.

Assessments may focus on the hazard from individual chemicals listed in a specific piece of legislation and therefore, if toxicity data are not available for one or more of those compounds, we may not be able to evaluate whether or not a risk exists. In some

instances, an ecologically-based limit may not be developed because not enough data are available to satisfy established requirements for limit development. In addition, compound-by-compound assessment may not address the effects of mixtures.

Further, our assessments are usually fragmented by medium type and the resulting set of criteria or standards that are developed for a particular medium: water, soil, sediments, etc. Each of these criteria have been developed independently of one another with specific algorithms and assumptions that may not evaluate a system with respect to one another. Therefore, cumulative exposure to an organism, population, community or system may not be accounted for in an assessment. If worst-case or reasonable worst-case assumptions are used on a medium-specific basis, decisions that are medium-specific might end up being more conservative than assessments conducted on a cumulative source basis. In addition, in cases where media-specific criteria overlap (e.g., standards for land application of biosolids and soil standards), there may be inconsistencies in methods and ultimately in acceptable media-specific concentrations.

Suggestions: A set of scientifically-based ecological quality criteria or standards might be developed for all media: air, water, soil, and sediments. All regulatory programs could integrate with a consistent set of criteria that may require adjustment depending upon program constraints; however, it would be important to identify what those constraints were, how they may or may not be impeding ecosystem protection, and what remedies might be necessary to adjust the programs to prevent further system degradation. As multi-media facility-wide permitting proceeds, these types of criteria could integrate well.

These criteria could be integrated with our definition of ecosystem health (as described in Section B. above); i.e., we might decide on a certain level of degradation or restoration for New Jersey lands, on a level of population depletion or proliferation for a certain species, or integrate with environmental planners to determine various management tiers that incorporate the criteria plus other management parameters. Gaps in data identified through an integrated approach to standards should be coupled to a research program.

V. Framework For Issue Resolution

Ecological assessment and ecosystem management programs are moving forward nationally and within New Jersey. The issues identified above are currently being discussed and debated by scientists and will continue to be for some time. As the field of ecological assessment continues to evolve, there should be some resolution and increased scientific consensus. In the meantime, New Jersey is well positioned to synthesize some of our existing resources into a coordinated framework for ecosystem assessment and management.

Overall, a framework for addressing these issues could include the following:

- Define ecosystem health and develop a measurement approach.
- Verify scientifically acceptable ecorisk methods.
- Synthesize existing data sources.
- Identify important data gaps.
- Develop a hierarchy of data needs.
- Implement scientific studies to pursue baseline data and other gaps in data, methods, and policy.
- Educate scientists, regulators, the regulated community, and the public about ecological assessment and ecosystem management as the field evolves.
- Develop an ecological risk communication strategy through a working group that would: define place-based assessment; identify economic, public health, and environmental benefits of place-based assessment; identify risks associated with assessments that are not place-based; identify various audiences; develop a communications strategy including products and action steps.
- Develop partnerships with appropriate governmental and non-governmental entities affected within an ecosystem for cohesive and sensible system assessment and management.
- Adjust regulatory programs and applied ecological analysis as the field evolves.
- Pursue ecosystem-wide projects that incorporate other research projects as much as possible.
- Convene working groups as needed to address issues of data needs, methods, appropriate models, uncertainty factors, data applicability, consistent standards, etc.

- Develop a coordinated ecosystem management initiative focused through a New Jersey Ecological Survey or New Jersey Ecological Data Program.

- Three primary elements of a New Jersey Ecological Survey or Ecological Data Program would incorporate basic data, indicators, and research.

- Data developed and synthesized through each data element should be spatially referenced for geographic display and analysis.

- Establish and publish a State of New Jersey's Environment Report.

A. New Jersey Ecological Survey or Ecological Data Program

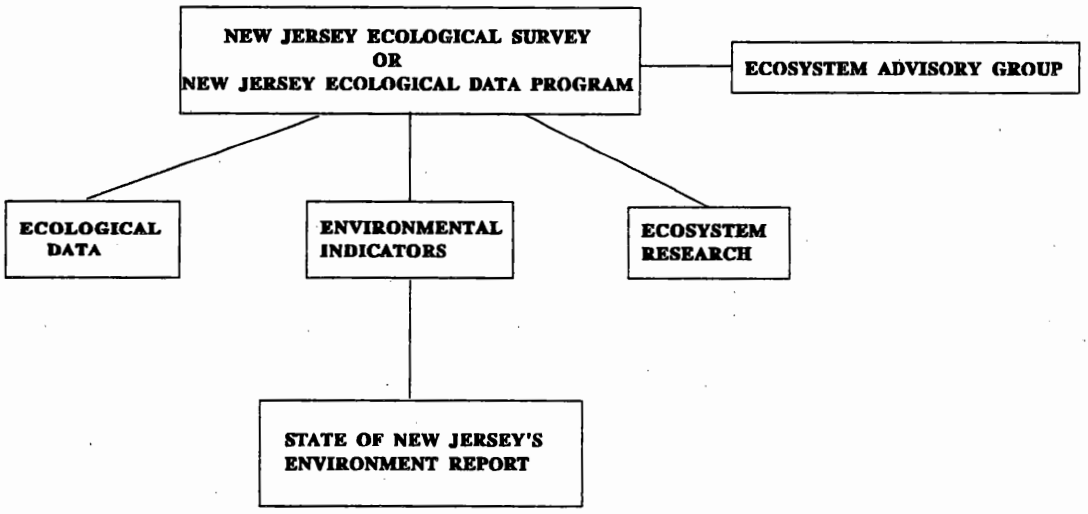
A coordinated ecosystem management framework for New Jersey might consist of a structure as outlined in Figure 1. This proposal is advocating coordination of existing programs via a liaison. This could allow existing programs to continue with their individual missions, while exchanging information via a liaison to meet ecosystem management needs.

A New Jersey Ecological Survey or Ecological Data Program could act as an umbrella for scientific data and research, while members of an Ecosystem Advisory Group could act as counselors, peer reviewers, and in an oversight capacity to ensure that the scientific data were developed in light of the issues identified in Section IV.

A New Jersey Ecological Survey or Ecological Data Program could link data and research programs to form a coordinated scientific basis for ecosystem management in New Jersey. It could be a single clearinghouse for ecological data pertinent to New Jersey ecosystems and could include three elements: ecological data; environmental indicators; and ecosystem research.

Again, these elements could be geospatially referenced whenever possible for geographic display and analysis. As the NJGIS is made available to the public (via local libraries, watershed associations, non-governmental organizations, etc.), students, citizens, businesses, etc. will be able to access environmental information for a particular geographic location in New Jersey.

FIGURE 1.
PROPOSED NEW JERSEY ECOSYSTEM MANAGEMENT FRAMEWORK



1. Ecological Data Element

An ecological data element could provide a comprehensive data base on ecological information for New Jersey. Data could include biotic and abiotic components of the environment. Extant data sources and data could be identified (including data from state, academic, non-profit research organizations, naturalists, etc.). This element could link with the other elements to help develop indicators and identify research priorities.

The ecological data element could link with the New Jersey environmental research and data directory that is anticipated through the New Jersey Environmental Research Partnership (See III.B.6). As previously mentioned, the Partnership is hoping to develop a resource guide including directories of New Jersey environmental researchers, data and data bases, data applications, specialized equipment, and training opportunities for New Jersey and adjacent state ecosystems.

2. Environmental Indicators Element

An environmental indicators element could link existing indicators programs and support development of additional programs to measure status and trends of environmental parameters to assess the state of New Jersey's environment. Indicators could help ascertain what we mean by ecosystem health. In addition, retrospective indicator data may be available, especially for sediments, which might allow us to look back in time to ascertain trends in contaminant deposition from the past to the present. An indicators element could link to the data and research elements to be certain indicators were integrated with these elements.

3. Ecosystem Research Element

An ecosystem research element could coordinate ecosystem researchers and support ecosystem research to provide a scientific basis for ecosystem management in New Jersey. Ongoing research from throughout the state could be disseminated to interested scientists. Cooperation and coordination of projects could be fostered through such an element, perhaps through the Environmental Research Partnership initiative. Priorities for research could be ascertained through linkages with the data and indicators elements, as well as the advisory groups. Section IV.C.1 (Ecoassessment Research Needs For New Jersey) describes specific research projects or needs already identified as a result of this analysis of ecoassessment.

4. State of New Jersey's Environment Report

The three survey or data elements mentioned above could form the basis for a State of New Jersey's Environment Report on a periodic cycle (perhaps every four years). Many of the pieces to such a report already exist through various governmental and non-governmental programs or media-specific program reports (including NJDEP's Air Quality and Water Quality Inventory Reports, monitoring programs, species survey programs, etc.).

A State of New Jersey's Environment Report could include both ecological and human health components. Such a report could assist New Jersey in determining how effective its regulatory programs are in protecting ecological and human health. It could also enable New Jersey to identify what we know about our environment, how our environment has changed over time, gaps in our knowledge, and provide input for the development of data and research priorities. Such a report could also assist in coordinating priorities for regulatory, conservation, monitoring, and research program adjustments within New Jersey and neighboring states.

Workgroups within an Advisory Group could be formed as needed to address specific issues such as ecosystem health, uncertainty, uniform standards and criteria, etc. Participants could include a wide range of scientists and interested parties from throughout New Jersey, as well as specialists (as needed) from outside of New Jersey. Participants in workgroups could also be drawn from New Jersey's Environmental Research Partnership.

An advisory group could link to the Ecological Survey or Ecological Data Program to ensure each of the survey elements were addressing the issues pertinent to ecosystem assessment in New Jersey.

Discussions with colleagues throughout New Jersey and the region has led the author to conclude that New Jersey has a wealth of biological and scientific resources. Coordination of these resources among scientists, planners, and other interested parties could further assist in scientifically-grounded ecosystem assessment and management.

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