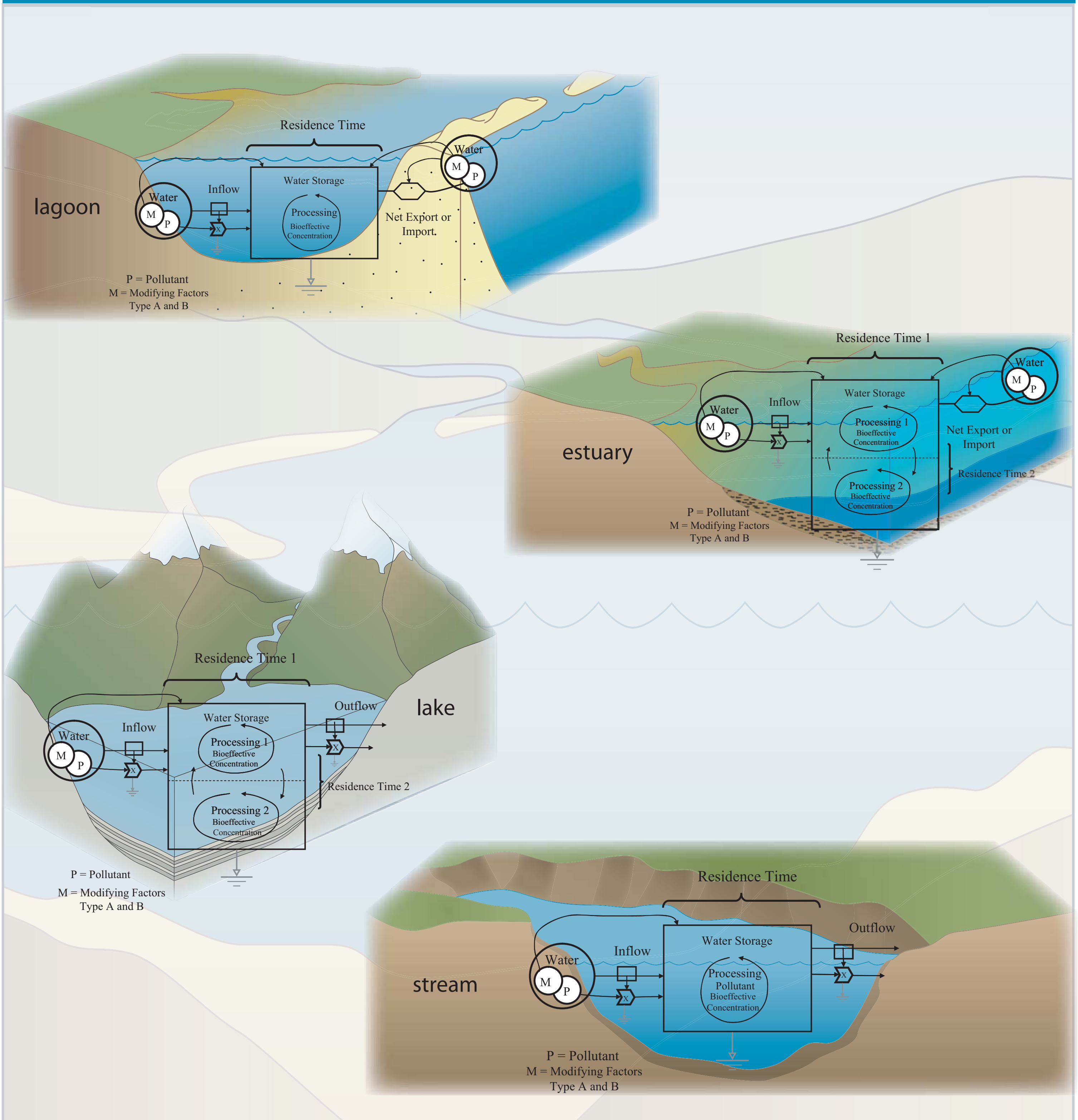
United States Environmental Protection Agency

Conceptual Models and Methods to Guide Diagnostic Research



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Conceptual Models and Methods to Guide Diagnostic Research

U.S. Environmental Protection Agency
Office of Research and Development
National Health and Environmental Effects Research Laboratory
Research Triangle Park, NC 27711

Notice

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Executive Summary

This report contains conceptual methods and models to guide research on and development of tools for diagnosing the causes of biological impairment within the aquatic ecosystems of the United States. It was produced to satisfy requirements in the U.S. Environmental Protection Agency Aquatic Stressors Framework (USEPA 2002b). The goal of the National Health and Environmental Effects Research Laboratory's (NHEERL) Diagnostics Research Program is (1) to provide tools to diagnose the causes of biological impairment in aquatic ecosystems, (2) to develop a classification system that simplifies the process of developing Total Maximum Daily Loads (TMDLs) or other regulatory programs for the myriad of water bodies requiring them, and (3) to support the States and Tribes in determining the causes of impairment of water bodies to be placed on their 303(d) lists. To accomplish these goals, NHEERL convened a Diagnostic Research Work-group. This workgroup developed an overview conceptual model of the factors controlling the action of pollutants and detailed conceptual models for four aquatic stressors: nutrients, suspended and bedded sediments, toxic chemicals and altered habitat that were identified as the stressors of major concern in the Aquatic Stressors Framework (USEPA, 2002b). Four canonical forms of the overview conceptual model were used as the framework for developing detailed conceptual models of the four stressors and as a basis for classifying aquatic systems according to their sensitivity to each stressor of concern. The proposed classification framework is designed to simplify the development of TMDLs by grouping aquatic systems according to similarity in their response to a particular stressor, In addition, classification may enable a more refined approach for quantifying stressor-response relationships for criteria development (USEPA 2004).

The approach to determining the causes of impairment used here consists of a linked set of hierarchical, modular methods and models and a proposed classification scheme for aquatic ecosystems. In addition, an Energy Systems Theory (Odum 1994) framework for developing causal network models of stressor action in aquatic ecosystems is presented in a series of text boxes as a parallel discussion. These two parallel approaches are brought together in the final section of the paper where detailed energy systems models of the main factors controlling the action of the four major aquatic stressors are given. The end result of our research will be guidelines for diagnosing the causes of biological impairment within aquatic ecosystems of the United States.

The suite of methods and tools under development (Figure ES-1) bridges the critical link between the assignment of a water body to the 303(d) list and the initiation of a TMDL (USEPA 2002a). This link includes the determination of a definitive cause or causes for the observed impairment that placed the water body on the 303(d) list in the first place, as well as, the development of tools that will allow us to understand how to restore the impaired ecosystem. To forge this link we identified six critical research elements that need to be accomplished to make a definitive diagnosis of the cause of impairment and to ensure the development of an effective TMDL or other regulatory program that will successfully control a pollutant. These research tasks are:

(1) Link sources, which result from human activities, to stressors and biological effects that occur in receiving water bodies in a manner that supports the development of effective corrective action.
(2) Formulate a set of simple, standardized models (*canonical* models) that incorporate the fundamental causal mechanisms determining biological condition and the observed impairment. Canonical models serve as a starting point for classification and as a frame for the development of detailed models of stressor action, which in turn may serve as a guide to the development of integrative diagnostic indicators.
(3) Development methods that use the hierarchical structure of watersheds and human activities when determining the scope of control needed for a successful TMDL.

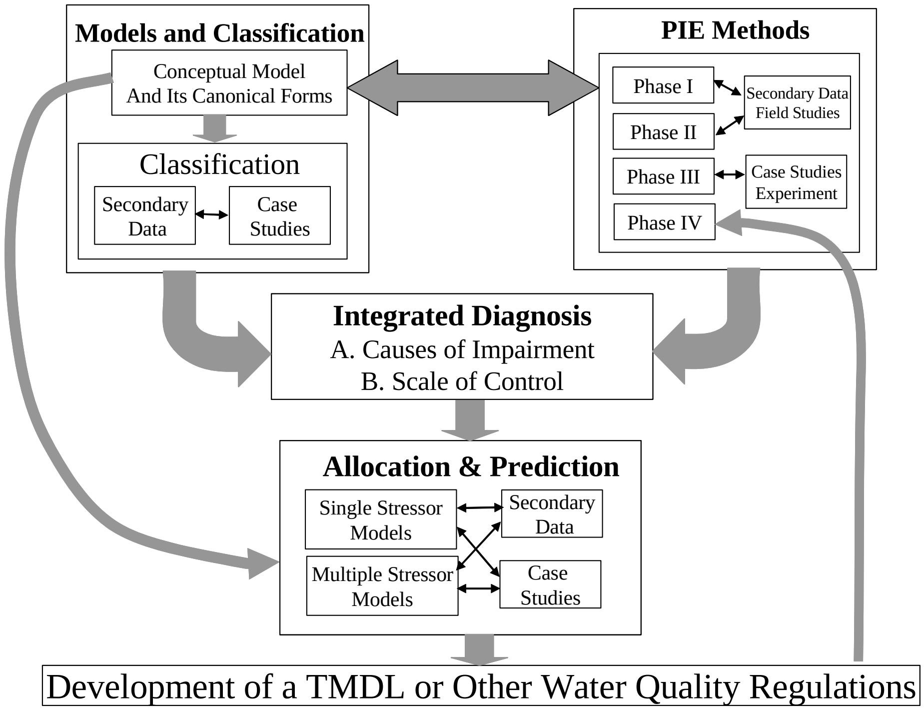


Figure ES-1. Organization of the elements of the Diagnostics Research Program showing how conceptual models, classification, and Pollutant Identification Evaluation (PIE) methods are used together to make an integrated diagnosis of the causes of impairment. Information on the results of implementing a TMDL feeds back to Phase IV in the PIE module, where it is evaluated to confirm or deny the original diagnosis.

(4) Extend the logical methods of deduction and elimination used to develop techniques in Toxicity Identification and Evaluation, TIE, (Burgess 2000, Ho et al. 2002) so that they can be applied to other pollutants. We have named this method, now under development, Pollutant Identification Evaluation (PIE). These tools, when fully developed, will allow the States and Tribes to make a definitive diagnosis of the causes of impairment to aquatic ecosystems. The general logical methods of causal analysis and the approach to problem solving described in the Stressor Identification Guidance Document (USEPA 2000a) are used in the development of PIE tools and methods.

(5) Construct detailed energy systems models of the action of single and multiple stressors within aquatic ecosystems to serve as the precursor of simulation models that will predict system behavior and allow the allocation of observed effects among multiple demonstrated causes.

(6) Classify ecosystems based on their response to stressors to simplify the process of diagnosis and the subsequent development of regulatory programs.

When used together, the methods and models developed within these six research elements will allow us to diagnose biological impairment in a simpler and more accurate manner, thereby accomplishing our goal of taking a water body from its 303(d) listing to the successful implementation of a TMDL.

The overview conceptual model that was used as a starting point for all of our research consists of three factors that we hypothesize control the action of pollutants in aquatic ecosystems.

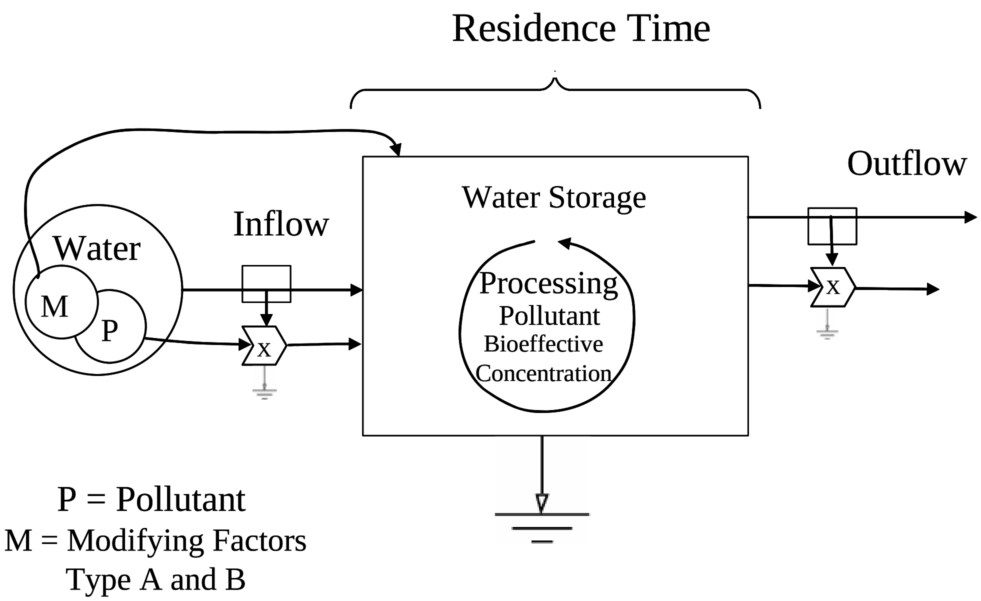


Figure ES-2. Conceptual model shown as an Energy Systems Language module of the three primary factors controlling the action of pollutants in aquatic ecosystems.

They are:

(1) the residence time of water and the pollutant in the system,
(2) the natural processing capacity of the system for the pollutant including the pathways that decompose, take-up, or sequester the material, and
(3) ancillary factors that modify the form of a pollutant, (i.e., the rate of processing) or the kind of action the pollutant exerts within the ecosystem (Figure ES-2).

These three factors are evaluated in a manner that quantitatively determines the effective dose of a pollutant for different types of ecosystems. Also, in this report, we hypothesize that characteristic properties of aquatic systems related to residence time, processing capacity, and modifying factors can be used to differentiate classes of ecosystems that develop different biologically effective concentrations of a material when loaded with a given quantity of a pollutant. The classification problem may be further simplified by grouping pollutants according to their mode of action such that an ecosystem processes all members of a class in a similar manner. In this case, we can express the bioeffective concentration in aggregate units (e.g., standard toxicity units). The conceptual models and methods presented in this document need to be mathematically formulated and evaluated with data from laboratory experiments and field studies before their veracity can be demonstrated and their full potential realized.

The development of PIE tools and methods as outlined in this document holds considerable promise for the diagnosis of the causes of impairment to estuaries and other coastal ecosystem. The diagnosis of the causes of impairment to coastal ecosystems is particularly difficult because of the complex flow regime in estuaries and the presence of multiple stressors in the marine environment, which commonly receives pollutants from many sources.

The overview conceptual model and its canonical forms can serve as a guide to the design of a stressor-based classification system, the development of scale independent indicators that give the expected condition of a system based on its class, and the identification of the scale of control needed for successful regulation of a given pollutant (a database tool to accomplish these tasks is under development). In addition, the overview conceptual model and its canonical forms serve as a frame for the development of detailed energy systems models for the individual stressors and their commonly occurring co-stressors. When quantitatively evaluated and programmed on a computer, these detailed conceptual models can simulate various scenarios of stressor-loading, giving results that can be used to allocate an observed effect among multiple active stressors. This document may be of immediate use to scientists in the process of developing initial problem formulations for risk assessments in support of the development of a TMDL or other regulatory program for one of the four classes of aquatic stressors covered by our research.

1.0 Introduction

In this report, we propose methods and conceptual models to guide research on the development of tools for diagnosing the causes of biological impairment within *aquatic ecosystems* of the United States. The approach to determining the causes of impairment used here consists of a linked set of hierarchical, modular methods and models and a proposed classification scheme for aquatic ecosystems. In addition, an energy systems framework for developing causal network models of stressor action in aquatic ecosystems is presented as a parallel discussion in a series of text boxes. These two parallel approaches are brought together in the final section of the paper where detailed energy systems models of the main factors controlling the action of four classes of aquatic stressors, nutrients, suspended and bedded sediments (SABS), toxic chemicals, and habitat alteration, are given. These four classes of aquatic stressors were identified as high priority areas for research by the Aquatic Stressors Framework (USEPA 2002b). A simple model of the main factors controlling the action of the three classes of aquatic stressors that are pollutants (nutrients, toxic chemicals, and SABS) was proposed as the linchpin to hold together the proposed methods and models needed to diagnose impairment to aquatic ecosystems of all kinds. This model also serves as the basis for determining the scale of management needed to develop an effective total maximum daily load (TMDL) for a pollutant at the scale of watersheds as well as water bodies. A detailed conceptual model of habitat alternation was also developed and related to the simple conceptual model controlling pollutants. A preliminary classification framework for coastal systems is presented in another report (US EPA 2004); however, our thinking on conceptual models and classification proceeded in parallel; therefore, the connection between the need to classify aquatic ecosystems and the development of conceptual models is reported here. When fully developed we believe these tools will simplify and improve the accuracy of water body evaluations currently being carried out by the states under Sections 305(b) and 303(d) of the Clean Water Act. Ultimately, our goal is to ensure the success of water body restoration programs such as those that control the TMDL of a pollutant (USEPA 2002a)

The models and methods presented in this paper are diagnostic in that they are all concerned with or support the determination of the cause or nature of a particular phenomenon B an observed impairment of ecological condition in an aquatic ecosystem. These methods include the use of characteristic signs and symptoms which suggest a cause; however, this evidence is strengthened by the use of definitive tests of causality whenever possible. The goal of diagnosis is to establish cause as unambiguously as possible under any given set of circumstances. Classification schemes are also diagnostic in the broad sense because the classification of systems based on their behavior under stress allows us to use the logical process of inference to impute causality and simplify the diagnostic process. Imputed causes of impairment can be verified using the definitive diagnostic tools that we are developing. Another diagnostic method is to focus on specific organisms that have been shown to be sensitive to or tolerant of a particular stressor. Specific diagnostic indicators are not discussed in this report, but they may be developed in the course of our future research. A final method of determining cause is to construct and analyze models of causal networks based on demonstrated and hypothesized mechanisms of interaction that are linked to the transformation of energy (Odum 1994). Whenever possible and especially in difficult or ambiguous cases, more than one diagnostic method should be used to determine active cause(s) of an observed impairment. When the results of several methods agree, we have greater confidence in the resulting diagnosis.

Our research program bridges the critical link (Figure 1) between assigning a water body to the 303(d) list and the initiation of a TMDL (USEPA 2002a). This key link includes the determination of a definitive cause or causes for the observed impairment that placed the water body on the 303(d) list in the first place, as well as, the development of tools that will allow us to understand how to restore the impaired ecosystem. To

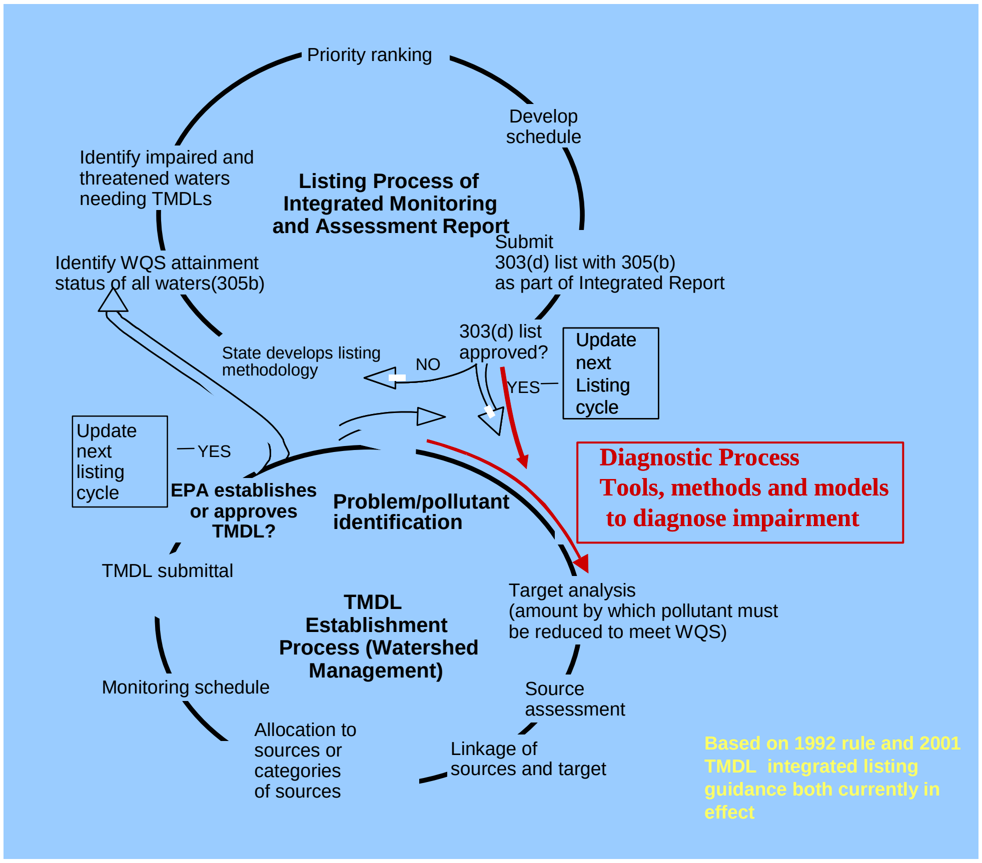


Figure 1. Conceptual model showing the links between the listing process in the monitoring and assessment report and the process for establishing a Total Maximum Daily Load (TMDL) for watershed management (USEPA) 2002a)

forge this link we identified six critical research objectives that need to be accomplished to make a definitive diagnosis of the cause of impairment and to ensure the development of an effective TMDL or other regulatory program that will successfully control a pollutant. These research tasks are

(1) Link sources, which result from human activities, to stressors and biological effects that occur in receiving water bodies in a manner that supports the development of effective corrective action.
(2) Formulate a set of simple, standardized models (*canonical* models) that incorporate the fundamental causal mechanisms determining ecosystem condition and observed impairment to serve as a basis for classification, a guide to the development of integrative diagnostic indicators and as the starting point for the development of detailed models of *stressor* action.
(3) Develop methods to account for the hierarchical structure of watersheds and human activities when determining the scope of a TMDL.
(4) Extend the logical methods of deduction and elimination used to develop techniques in TIE, Toxic Identification and Evaluation (Burgess 2000, Ho *et al*. 2002) to other *pollutants* to allow the states to make a definitive diagnosis of the causes of impairment to aquatic ecosystems. We have named this method, now under development, Pollutant Identification and Evaluation (PIE). The general logical methods of causal analysis and the approach to problem solving described in the Stressor Identification Guidance Document (USEPA 2000a) provide a logical framework for these methods.
(5) Construct detailed simulation models of the action of single and multiple stressors within aquatic ecosystems to predict system behavior

and to allocate observed effects among multiple demonstrated causes.

(6) Classify ecosystems based on their response to stressors to simplify the process of diagnosis and the subsequent development of a TMDL.

When used together, the methods and models developed within these six research areas will allow us to diagnose biological impairment in a simpler and more accurate manner, thereby accomplishing our goal of taking a water body from 303(d) listing to the successful implementation of a TMDL (Figure 1). For example, development of the PIE diagnostic methods (4) and the development of detailed models of stressor action (5) provide a means for identifying, evaluating and interpreting cause and effect relationships by relying first on PIE methods that use field data, experiment, and logic to definitively demonstrate the presence of causal links between source, stressor and effect within an aquatic ecosystem. When source, stressor and effect are shown to exist and the linkages between source and stressor and stressor and effect have been demonstrated the pollutant is designated as a demonstrated cause of the impairment. Next the detailed conceptual models for the four stressors, *i.e.*, nutrients, toxic chemicals, suspended and bedded sediments and habitat alteration (2), are used to create simulation models (5) to predict impairment and allocate observed effects among multiple demonstrated causes, thereby allowing us to set priorities for restoration. The development of tools to link stressors and effects in watersheds (3) will allow us to trace the causal links between watershed activities (sources) and individual pollutants (1) over at least three scales of watershed organization (*e.g.*, stream orders and receiving water bodies within their nested watersheds) to determine the appropriate spatial scale at which a TMDL needs to be implemented to control the concentrations of a particular *pollutant* in hierarchically organized aquatic and human systems. Together, the canonical models (2) and the classification system (6), now under development, will be used to simplify and improve the accuracy of impairment decisions by grouping aquatic systems according to similarities in their behavior under *stress*. Classification categories will identify systems that are functionally similar and this similarity may serve as both a basis to impute causality across the class and as a guide for restoration of individual systems within the class.

In this report, we have used the principles and methods of systems ecology, specifically Energy Systems Theory, EST (Odum 1983, 1994), to develop and trace pathways of causality in ecological networks and to synthesize information and develop models for the action of stressors in aquatic ecosystems. This method is an integrating thread that flows through this work and it provides either the primary method or an alternative method for accomplishing all but one of the six research tasks. However, other approaches and methods were drawn upon to design our research, including the methods of Toxic Identification and Evaluation and Stressor Identification. Many readers will be unfamiliar with energy systems concepts and for this reason supporting information is provided in a series of text boxes that present topics that are relevant to accomplishing our research objectives.

1.1. Present Context for Aquatic Ecosystem Protection in the United States

The dilemma faced by modern industrial societies that accounts for the need to protect natural environments is discussed in the text box (A) and diagramed in Figure 2 using the Energy Systems Language (Odum 1971, 1983). The United States government has tried to solve this dilemma by promulgating environmental laws and regulations aimed at limiting environmental degradation. The Clean Water Act (CWA) is foremost among the laws passed to protect aquatic ecosystems. Among other things, this law charges the states with periodically assessing the condition of all fresh and salt waters within the state's boundaries (Section 305 [b]) and with reporting those water bodies that do not meet water quality standards (Section 303 [d]). This assessment is performed within the context of a regulatory structure (water quality standards, criteria, designated uses, etc.) that is intended to

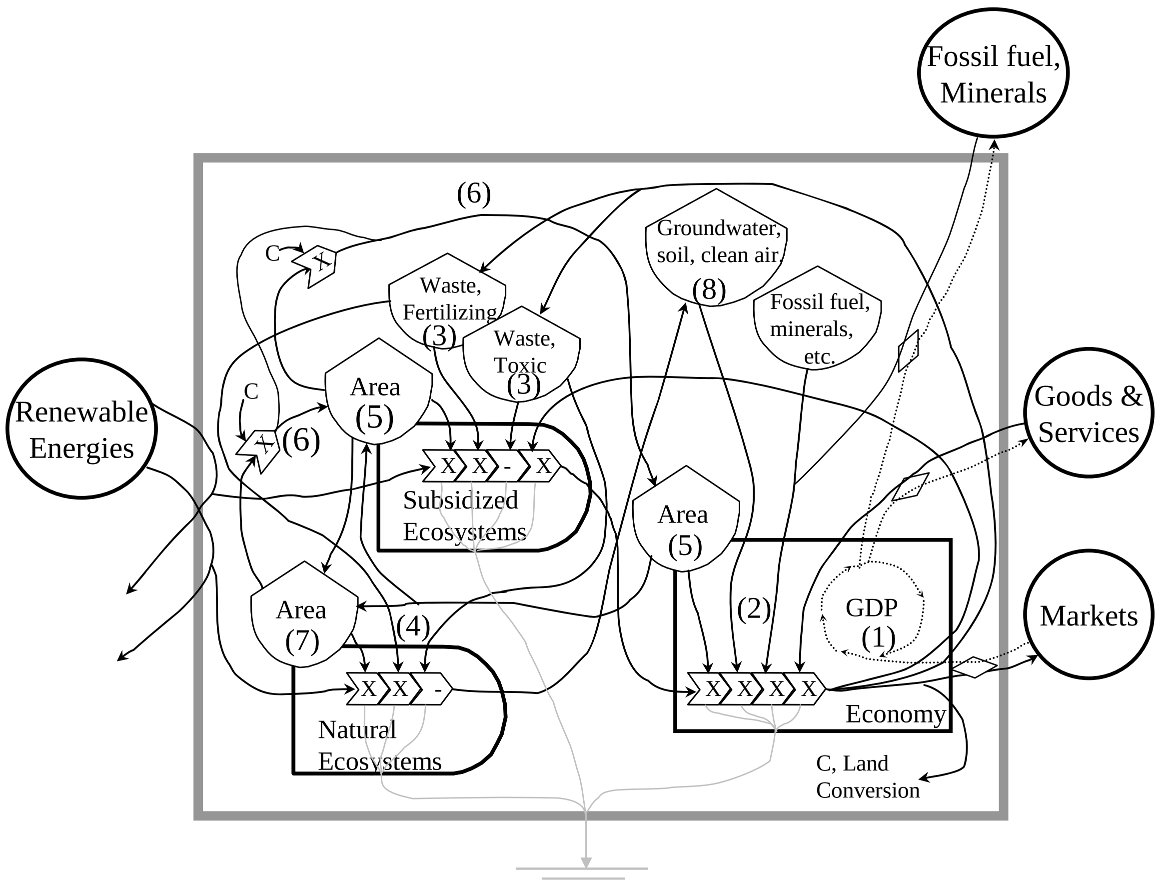


Figure 2. Diagram of the environmental system of a region, in which lines with arrows show the chain of connections that link economic activity (circles on the right) with environmental degradation (see Figure 3).

ensure that the quality of aquatic environments is maintained within limits that are acceptable to society.

Currently, the states and the federal government deal with the difficult tradeoff between economic prosperity and environmental quality by assigning designated uses to the various water bodies. The capacity to choose the best use for a particular water body makes it possible to set physical, chemical, and biological water quality standards that correspond to those expected for a certain degree of desired economic development. However, the law does not allow designated uses to be created only to support economic development, in fact, economic feasibility and the water body's pre-existing condition at the time the Clean Water Act was passed are only two of many considerations. The anti-degradation clause of the CWA is intended to prevent backsliding. However, all these legal safe-guards may not be sufficient to counter the tendency for economic priorities to drive society's expectations for the condition of the environment.

The periodic monitoring and assessment of the condition of all water bodies under Clean Water Act Provision 305(b) is intended to ascertain whether a given water body has attained the desired status for all applicable water quality standards. If applicable water quality standards are achieved, no further work must be done. In the event that a physical, chemical, or biological water quality standard has been consistently violated, a

(A) The Inherent Conflict Between Economic Prosperity and Environmental Quality

The ecological and economic context for society's practical need to establish regulatory mechanisms is illustrated by a simple energy systems model shown in Figure 2 (Campbell 2001a). This model illustrates the dilemma faced by modern society on every scale of human activity: A fundamental conflict within society arises because economic prosperity (1) is not obtained without the use of resources (2) which inevitably leads to the creation of wastes (3) and degradation of the natural environment (4). Furthermore, resource use promotes the expansion of the human population which requires more support area (5) causing formerly natural lands to be appropriated for human use (6). As natural lands (7) are converted into farmlands and cities and the functioning of the remaining natural areas is compromised by pollution, the critical life-supporting services that the environment provides to individuals and society (8) are gradually lost. When total system productivity begins to decline as a consequence of this lost support capacity, people perceive that their standard of living is declining and they look for ways to mitigate environmental degradation without sacrificing economic productivity. Establishing Total Maximum Daily Loads for a pollutant and other regulatory mechanisms are ways that society has chosen to try to restore a balance between human and natural use of resources. More permanent solutions may be found by changing system designs to make human systems more closely reflect natural systems in their structure and function (Holmgren 2002).

process is set in motion to restore conditions to meet the standard. This process includes designating the water body as impaired by placing it on the 303 (d) list, diagnosing the cause of impairment, allocating observed impairments among multiple demonstrated causes, acting on the diagnosis by establishing a TMDL or other regulatory program to control the stressor or stressors of concern, and monitoring the mitigation of adverse effects, which may confirm the diagnosis. Continued monitoring of state waters is used to determine if the actions taken are sufficient to restore the water body to the water quality standards that accompany a given designated use.

1.2 Use of Conceptual Models in Monitoring and Assessing Ecosystems

An understanding of the interrelationships between ecosystem components and processes and environmental conditions is critical for successfully linking stressor sources to biological effects in water bodies receiving a pollutant. In addition, knowledge of the cause and effect relationships operating within an ecosystem is a critical element of successful diagnosis and of successful monitoring and assessment programs. It may be helpful to examine the usefulness of conceptual model building in guiding monitoring research as a means for understanding how we might use conceptual models to guide diagnostic research.

Manley *et al*. (2000) argued that conceptual models are the foundation for scientifically-based, ecologically-focused monitoring programs. Conceptual models are viewed as "working hypotheses about ecosystem form and function" (Manley *et al*. 2000), and as "qualitative or quantitative statements concerning the nature of ecological risk" (Gentile *et al*. 2001). Conceptual models are used to inform resource managers and scientists about the critical elements of ecosystems and their relationships to environmental stressors, and they are applied to guide the scope and scale of monitoring programs. Grant *et al*. (1997) and Gentile *et al*. (2001) list several stages of conceptual model development: defining goals and objectives, delineating spatial, temporal and ecological scales and boundaries, identifying sources of natural and anthropogenic stressors, identifying critical ecosystem components at risk from the stressors, identifying causal relationships between system components and stressors, developing a graphical model, and describing expected patterns of model behavior.

Ecosystems have been defined on the basis of both structure and function. Ecologists are somewhat artificially divided into two schools of thought regarding ecosystems: those who view ecosystems as collections of populations or communities and those who view ecosystems as collections of processes and functions (O'Neill *et al*. 1986). The population-community approach views ecosystems as networks of interacting populations. The process-function approach views ecosystems as the sum of the physical, chemical, and biological processes active within a space-time unit. While it is tempting to emphasize one approach over the other, there is danger in assuming that relevant system dynamics can be understood through only one approach.

Manley *et al*. (2000) addressed the structure vs. function problem in developing conceptual models for the Sierra Nevada Ecosystems Project. Their approach clearly emphasizes ecosystem processes, because of the ability of energy and material flows to integrate system components through space and time, while also including broad structural components. Gentile *et al*. (2001) used a risk-based approach to develop conceptual models for managing South Florida ecosystems. Their approach, while considering ecosystem structure and function, also emphasizes ecological values (ecosystem sustainability), endpoints (attributes of ecological and/or societal importance), and measures (stressor-response relationships). The goal of Gentile *et al*.'s risk-based approach is to "establish a parsimonious set of endpoints for each ecosystem of concern, such that any change in the structure or function would be manifested as a change in one or more of the endpoints" (Gentile *et al*. 2001). Similarly, our approach to developing conceptual models will incorporate both structural and functional elements and endpoints.

2.0 Methods for Developing Conceptual Models

Because energy transformation is the underlying cause of all phenomena, the structural and functional elements of ecosystems can be integrated and related to external forcing functions and stressors by constructing networks of energy flow using the Energy Systems Language (Odum 1994). For this reason, one approach that we used to develop conceptual models of stressor action within ecosystems began by first considering the energy transformations that are the proximal cause of the behavior of the system variables that we want to understand and predict (*e.g.*, the metabolic processing of materials by organisms affects *in situ* concentrations of nitrogen, carbon, phosphorus, and oxygen in a water body). Next, we consider energy transformations of the larger system, which often account for the origin of stressors. The larger system provides the context for understanding and interpreting stressor action on the ecosystems of a water body. First, the larger context for the water body of concern is established by characterizing the natural processes and human activities of the next larger system (usually the watershed); then we formulate specific methods and models to address diagnostic research needs and determine the causes of impairment within those water bodies. These research needs (*i.e.,* methods, models and other tools) are defined from a study of the requirements of existing water quality regulations and the current methods that states, regions, and tribes use to comply with these regulations.

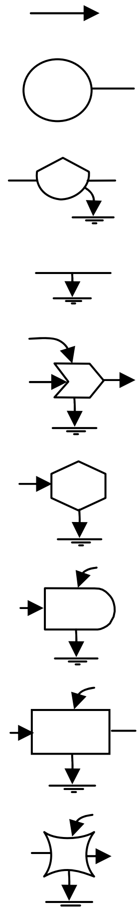
In this study, model development was initially performed using narrative descriptions and simple summary diagrams. These verbal descriptions were formulated as detailed conceptual models using the Energy Systems Language, ESL (Odum 1983, 1994). The ESL (Figure 3) is a symbolic language that is ideal for building conceptual models of ecological and environmental systems because the symbolic objects of the language correspond to components and processes found in ecological and economic systems and are mathematically defined. Conceptual models created in ESL are easily converted into quantitative mathematical models for simulation and prediction of ecological variables.

2.1 The Context for Diagnosing Impairment | to Aquatic Ecosystems

The key concepts needed to gain an overview of the diagnostic process were derived directly by considering the implications of the words in our charge given in the Aquatic Stressors Framework (USEPA 2002b), which was to understand and predict the actions of *stressors* on *aquatic ecosystems.* In addition to sorting out the effects of multiple stressors and determining the cause of observed ecological impairments, we recognized that the states and tribes needed a way to simplify the process of determining the causes of impairment to efficiently deal with an overwhelming number of impaired water bodies. We propose to accomplish this end by classifying ecosystems based on differences in their response to stressors. The need to produce a stressor-based classification system was a primary consideration in the development of our conceptual models, thus, the conceptual overview described here provides the basis for developing and testing our classification system as well as a context for the development of diagnostic methods and models. The ideas that provide a context for our research are discussed in more detail below.

2.1.1 Aquatic Ecosystems and the Hydrologic Cycle

All systems of concern in this document are *aquatic ecosystems* and as such, they are the result of water flows in the global hydrological cycle (Figure 4). Aquatic ecosystems can be visualized as a network of water storage units of different kinds and sizes (*e.g.,* streams, wetlands, lakes, estuaries) that are interconnected by water flows. Aquatic ecosystem units are arrayed over the surface of the continents from pole to pole and from sea level to the highest mountains (Bailey 1998). Stream flow is unidirectional and each storage unit is connected to the one below it in the network (Figure 4). Aquatic system units are also linked by bidirectional water flow in estuaries seaward of the head of tide. The dissipation of the energy carried by the flow of water over the landscape creates a hierarchical



Energy circuit A pathway whose flow is proportional to the storage or source upstream.

Source A forcing function or outside source of energy that delivers forces according to a program controlled from outside.

Tank A compartment or state variable within the system that stores a quantity as the balance of inflows and outflows.

Heat sink Dispersion of potential energy into heat accompanies all real transformation processes and storages. The energy leaving the system at this point is no longer usable by the system.

Interaction Interactive intersection of two pathways that are coupled to produce an outflow in proportion to a function of both; a work gate.

Consumer An autocatalytic unit that transforms energy, stores it and feeds it back to improve inflow.

Producer Unit that collects and transforms low-quality energy under the control of high quality flows.

Box Miscellaneous A symbol to use for whatever unit or function is needed.

Switching Action A symbol that indicates one or more switching actions controlled by a logic program.

Figure 3. Definitions for the basic symbols of the Energy Systems Language (modified from Odum 1994).

network of interconnected storage units and flows that each have their own unique characteristics for transporting and processing energy and materials.

2.1.2 Diagnosis and Management at the Watershed Scale

Water flows organize the natural landscape hierarchically so that aquatic ecosystems are distributed in space and linked by converging water flow (Figure 5). Water flows that connect one ecosystem unit to another on the landscape transfer materials in excess of each ecosystem’s processing capacity from one system to the next converging materials and creating a three dimensional hierarchy of landform and energy and material flows. In Figure 6, this complex process is visualized as a longitudinal series of connected water bodies, in which the excess nutrients supplied to an ecosystem cause production, P, to exceed respiration, R, producing an excess of fixed carbon over that which the ecosystem can process. This excess is transferred by water flows to the next system in the series where it stimulates additional respiration, thereby increasing nutrient remineralization. Any excess nutrients are in turn transferred to the next system in the longitudinal series, where they may stimulate increased production (Odum 1971). The transfer of materials is more complicated in systems with bidirectional flow, but the same model will apply wherever there is a net transfer of nutrients. In summary, aquatic ecosystem system functions are linked across multiple scales in the nested hierarchy of watershed organization, and this condition must be considered when diagnosing the causes of impairment within a particular water body and when promulgating an effective TMDL for a pollutant with hierarchically structured sources and flows.

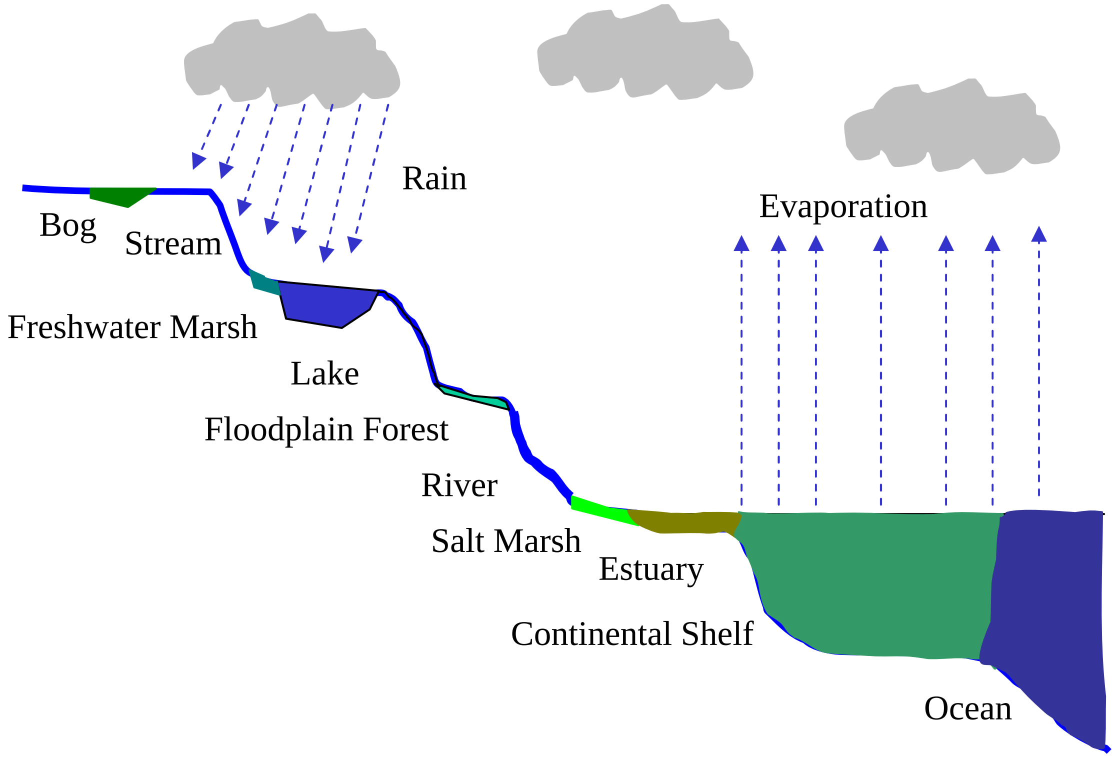


Figure 4. Aquatic ecosystems storing water within the hydrologic cycle.

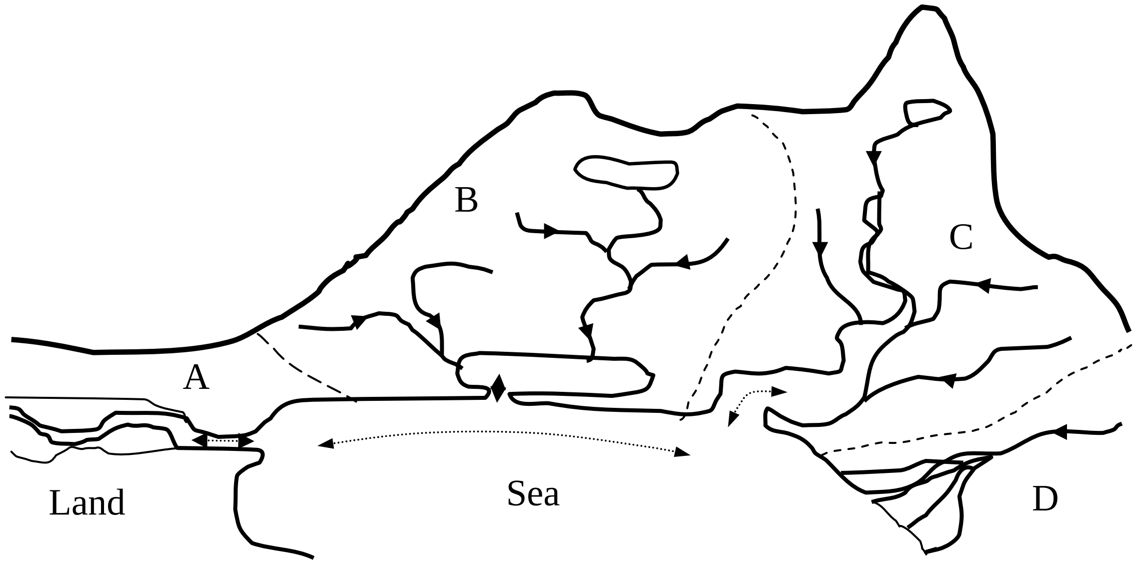


Figure 5. Networks of ecosystem units arranged on the landscape within fundamental watersheds A, B, C, and D each of which contains a river system that empties into the sea. Dashed lines indicate watershed boundaries.

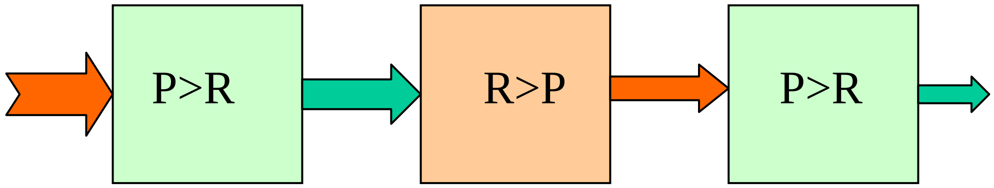


Figure 6. Aquatic ecosystems are arranged on the landscape in a longitudinal series of processing units dominated by production (P) or respiration (R). Excess production in one system sets the stage for excess consumption in the next and vice versa.

(**B) Definitions of Ecological Stress and Emergy**.

Stress is an energy drain caused by an injury or impairment to an ecosystem that results from the overuse of one or more ecosystem components or processes compared to a typical or original functional state (Odum 1968). In general, stress is the result of a change or perturbation in the long term or "normal" *emergy signature* (see Text Box C) of a place. In many places, the long term emergy signature is primarily comprised of inputs from the natural environment. The change in external forcing is often a product of human activities and it invariably results in a change in the "normal" or expected functioning of the ecosystem under the original signature.

Emergy is the available energy of one kind, previously used up, directly and indirectly, to make a product or service (Odum 1996). Its unit is the emjoule. Em- is an acronym for energy memory that indicates the energy associated with a quantity has been used in the past (Scienceman 1987). Emergy can use any kind of energy as the base, for example coal joules, solar joules, etc. However, in evaluating environmental systems, we commonly use solar energy as the base unit. *Solar emergy* is the available solar energy previously used up to make a product or service. Its unit is the solar emjoule (abbreviated sej). Available energy is energy with the capacity to do work, sometimes called exergy. Available energies in different products and processes do work of different kinds when used in a system network. In calculating the emergy of an item, the available energy of different kinds must be converted to a single kind to make them comparable. These transformed values are the emergy contributions of each required input to the production process and they can be summed to determine the emergy of the item. Transformities are the coefficients by which the available energies input to a production process are multiplied to get emergy. Solar *transformity* is the solar energy required to make one joule of a product or service. Its units are solar emjoules per joule (sej/J). The *transformity* of a product is its solar emergy divided by its available energy. Thus, the fundamental equation of emergy analysis is:

Emergy = Transformity X Available Energy (exergy).

The change in *empower* (emergy per unit time) that occurs in an ecosystem under stress is a measure of the ecological cost or benefit that results from the change in forcing inputs. This change can take various forms depending on the frequency, magnitude, and duration of the change in forcing functions and the properties of the system under stress (Campbell 2000, Holling 1986).

2.1.3 Stress in Aquatic Ecosystems

One common anthropogenic cause of *stress* (see text boxes A and B) in ecosystems is the production of wastes of various kinds. Wastes are produced as a consequence of human activities on the landscape (Figure 2) and transported through aquatic environments (Figure 4) which are changed as a result. Three of the four stressors that we are examining, *i.e*., suspended and bedded sediments, nutrients, and toxic chemicals, are wastes resulting from human activities, which often have stressful effects on the biota when introduced into aquatic ecosystems at concentrations greater than those the system is able to process. A conceptual representation of the impact pathway that results in stress in an ecosystem is a simple chain of cause and effect:

Human activities → pollutant sources → presence of the stressor in the environment (*e.g.*, the concentration of a pollutant) → observed effect (*e.g.*, a biological impact).

Given this causal chain and the state of our present knowledge of aquatic ecosystems (USEPA 1998, 2000b), we asked two questions, “What tools do we need to make a definitive diagnosis of the cause of stress observed in aquatic ecosystems?” and “What do we need to know to develop effective management plans to control the impact of stressors on aquatic ecosystems?” These questions and an examination of the process that the states and tribes currently use to answer them (GAO 2002) led us to propose the development of five diagnostic tools and a classification scheme.

2.2 Development of Diagnostic Tools

Diagnostic tools need to be developed that will allow us to meet the research objectives. A brief description of the diagnostic tools proposed to address these tasks is given below.

(1) Causal webs: Restoration of aquatic ecosystems and the mitigation of observed effects require us to demonstrate the link between sources of a pollutant in human activities on the landscape and concentrations of that pollutant in the environment. The construction of causal webs from both top-down and bottom-up perspectives was proposed as a way to understand and trace the links between human activities in the environment, pollutant sources, stressor concentrations and observed effects.

(2) Canonical models: A *canonical* model is a simple diagrammatic and/or mathematical model that captures the key characteristics of a system. For example, the canonical model of bidirectional flow describes an estuary, whereas, unidirectional flow describes a stream. Where sources of pollutants and their effects are present in aquatic ecosystems, we need to determine the mechanisms of stressor action and control. To address this need we developed an overview conceptual model and a series of standard models with simple properties (canonical models) that were used to guide the development of detailed models and a classification system and to estimate the strength of links within a spatially distributed network of aquatic systems having different residence times and processing capacities for the pollutant.

(3)Watershed scale for diagnosis and control: When sources of pollutants and their effects are present in watersheds and in their associated aquatic ecosystems distributed over a landscape, there is a need to determine the appropriate scale of control (water body, local watershed, next larger watershed, etc) that is required to develop an effective regulatory program to control a given stressor or combination of stressors. To address this need, we developed a conceptual overview of the way that aquatic ecosystems are linked on the landscape and proposed a method for determining system connectivity and the scale of regulation needed to implement a successful TMDL using a linked network of canonical models.

(4) Pollutant Identification and Evaluation (PIE): A definitive diagnosis of the cause of an observed impairment requires that we demonstrate the operation of the impact pathway at the site of the impairment. This is accomplished by establishing the presence of source, stressor and effect and then demonstrating the link between stressor and effect by showing that the observed concentration of the pollutant can produce the observed

biological effect under the prevailing conditions at a given location. This need will be answered for estuaries by the PIE field and laboratory methods under development.

(5) Models for the allocation of effects: The canonical models serve as a starting point for constructing more detailed quantitative models for further development, evaluation and simulation of particular indicators. In this study, detailed conceptual models of the action of individual stressors within an ecosystem were constructed using Energy Systems Language (Odum 1971, 1983, 1994). Where multiple stressors are shown to be causing an observed impairment, quantitative versions of these more detailed energy systems models for computer simulation can be developed to allocate observed effects among several active causes, i.e., stressors for which source, stressor and effect and the links between source and stressor and stressor and effect have been demonstrated using PIE or other methods.

(6) Response-based classification: The overview conceptual model of the factors controlling the action of stressors in aquatic ecosystems and the series of canonical models based upon it will serve as the basis for defining classification criteria and developing and testing model-based and statistically-determined classification systems.

2.2.1 Causal Webs: Bottom-Up and Top-Down

Linkages within a network can be shown by using the weight of evidence approach, which relies on the specification and documentation of verified impact pathway models to trace the production of a stressor or stressful condition (i.e., an observed impairment to an aquatic ecosystem) to human activities. These conceptual models may be built using either a top-down or bottom-up approach and include transport mechanisms and other factors linking human activities, sources, stressors, and effects on the landscape. Figure 7 presents a conceptual model showing how these impact pathways may be delineated from both the top-down and from the bottom-up perspectives. For any given analysis of landscape patterns, each link in the chain connecting the stressor with the effect must be shown to be present and capable of producing the hypothesized action. For example, lawn chemical application must be sufficient in a certain urban area to account for the observed concentrations of pesticide in sediments when transport and microbial break down of the substance have been accounted for. Because resources and their concomitant human activity patterns are often clustered as by-products of an underlying process of organization, multiple impact pathways often operate at the same time and this complicates the diagnostic process. Where more than one impact pathway can be shown to be potentially active the PIE diagnostic methods can be applied to a random selection of sites to determine the active agents of impairment. If any step in the impact pathway model can be shown, on the basis of data, to be insufficient to result in the observed distribution of a stressor; additional research is begun to find sources that can account for the observed distributions.

For understanding the causes of impairment in ecosystems, one may use either inductive (bottom-up, diagnostic) or deductive (top-down, predictive) reasoning (Ziemer 2004). Environmental monitoring has traditionally used the inductive, diagnostic approach based on observation and correlation. For example, the loss of species abundance in streams is noted and appears to be related to the loss of habitat used by that species. Habitat loss is correlated with increased sediment loads in streams. The sediment load appears to be related to increased erosion from watersheds, which have been logged. Using an inductive approach and this chain of evidence, we can conclude that declines in species abundance or condition are the result of timber harvesting (Figure 8a).

Causality can also be investigated by reversing the usual diagnostic path, which is inductive. The top-down, deductive approach attempts to predict the impacts to ecosystems based on known or anticipated responses of the systems components to stressors. Using the timber harvesting example above, we predict that loss of forest cover in the watershed due to clear-cutting will result in exposure of the soils to weathering, leading to increased losses of soil from the watershed. This soil enters the streams

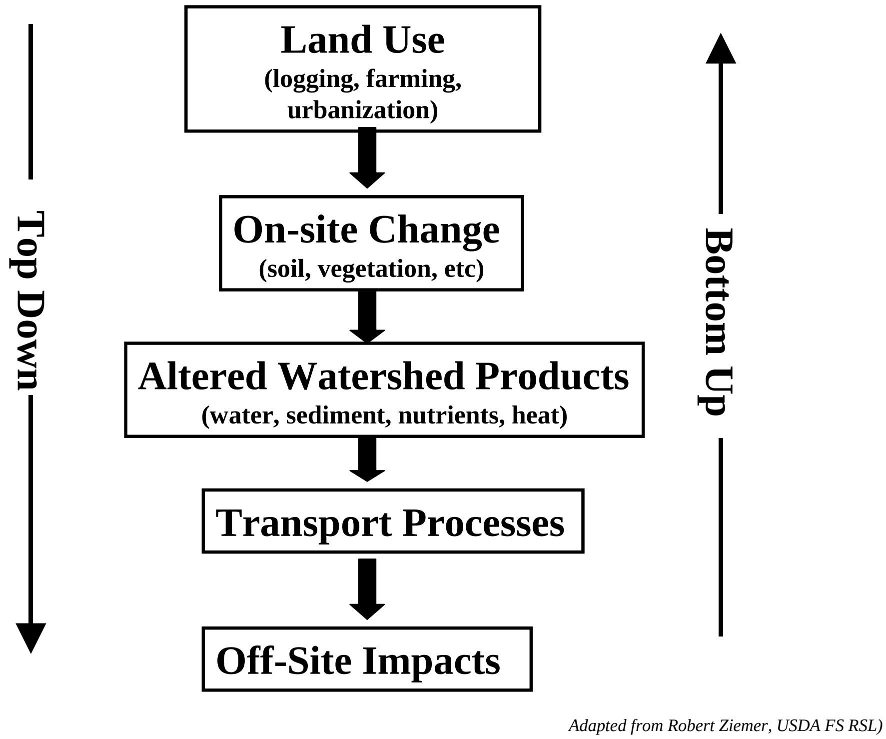


Figure 7. Illustration of the top-down and bottom up approaches to tracing causal pathways (Ziemer 2004).

where it accumulates altering stream hydraulics and habitat that is necessary for the biota, leading to a decline in species abundance or condition (Figure 8b).

Either approach may illuminate the cause of impairment, if ecosystem processes are adequately understood and sufficient stressor-response data are available. In practice, the level of understanding and adequacy of data are rarely met and neither approach should be used exclusively. This is especially true when multiple stressors and complex systems are involved. Our method uses both the top-down and the bottom-up perspectives, and in so doing combines the strengths of predictive modeling based on first principles with the empirical verification of mechanisms upon which accurate diagnoses depend.

2.2.2 Canonical Models

In general, we can conceptualize the connection between contiguous waters using canonical energy systems models (Figure 9). A canonical model captures a process or represents a system of interactions in a simple form. The canonical models given in this paper are used to represent the storage units of aquatic systems on the landscape and the links between them. Source, stressor and effect are dynamically linked together by energy systems models in their fully developed form. If a canonical or other model is applied to every grid square of a spatial field, or in every element of a connected network, it is termed a *unit model* (Odum 1994). The network of *ecosystems* described above may be simply modeled as an interconnected series of canonical energy systems models that are inserted into each cell, such that the model components and flows can assume different values in every cell based on the ecosystem characteristics and the forcing functions at the boundaries. In its simplest form the canonical model for a stressor acting on a stream reach in a single unit of the landscape accounts for inputs of the pollutant to the stream reach from upstream, the addition of the pollutant over the length of the stream segment from its watershed, and the assimilation or break down of the stressor by chemical and biological processes taking place within the water body (Fig. 9a). The

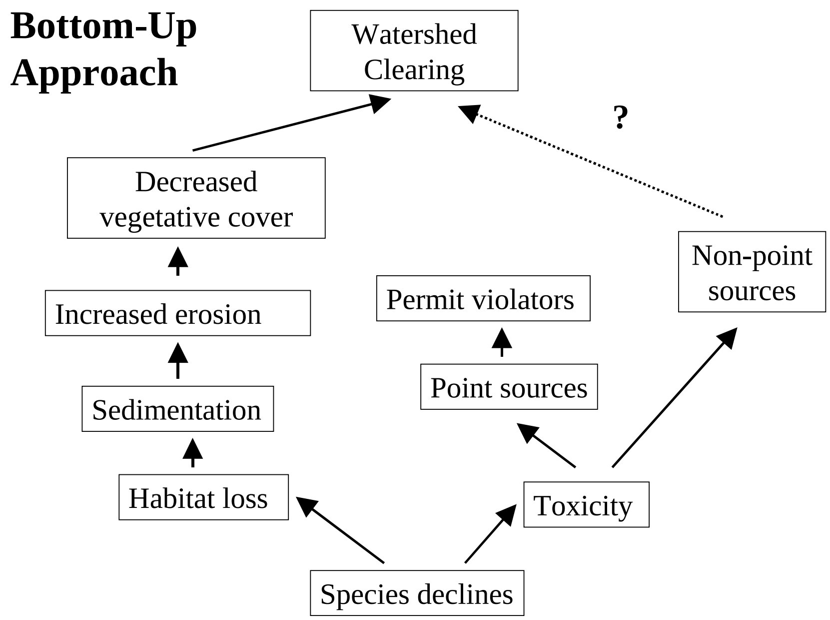


Figure 8a. Conceptual model of the bottom-up approach to tracing causality through a web of interactions. In this example, the observation of a species decline initiates model construction.

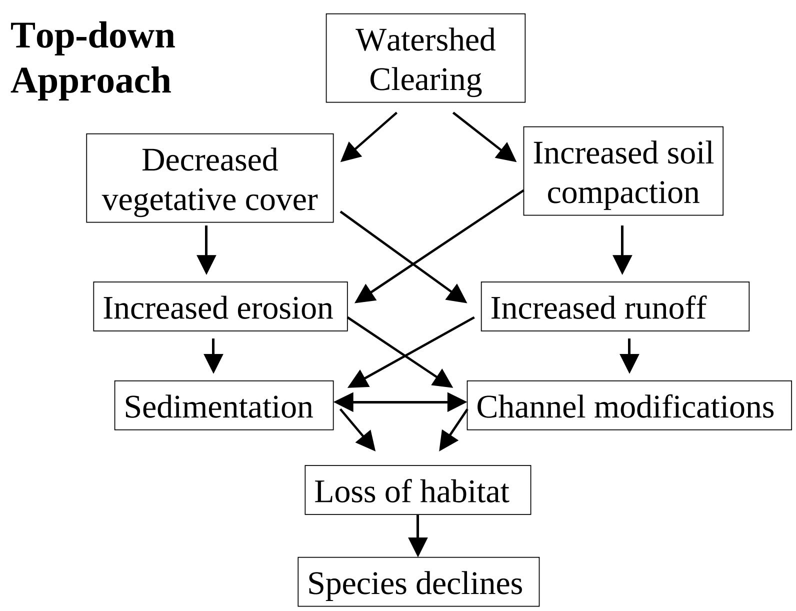


Figure 8b. Conceptual model of the top-down approach to tracing causal pathways through a web. In this example, the observation of watershed clearing initiates model construction.

output variable for this basic model is a flow of the pollutant moving into the next stream segment or downstream water body.

For estuaries, the rules that govern transport will be more complicated, but the unit model will be similar in structure. The common and distinguishing properties of the four canonical models used to represent aquatic ecosystems with different hydrological properties are shown in Figure 9. Three prominent factors are derived by inspection from the structure of energy systems models of aquatic ecosystems and these are hypothesized to be the primary factors controlling the stressful actions of pollutants in aquatic ecosystems. They are (1) the residence time of water and pollutant in the system, (2) the natural processing capacity of the system for the pollutant including the pathways that decompose, take-up, or sequester the material, and thereby determine its turnover time, and (3) additional factors that modify the form of the exposure-effect relationship or the kind or intensity of action that the pollutant exerts within the ecosystem. Processing capacity can be altered by the presence or absence of other materials changing the biologically effective concentration of the pollutant. These factors are designated as type A modifiers, *i.e.*, they alter the biologically available concentration of the pollutant. Other additional factors (3) change the form or character of the relationship between the biologically available concentration and the observed effect. They are designated as type B modifiers, e.g., turbidity is a type B modifier of nitrogen availability, because it alters the primary production obtained by adding an additional unit of nitrogen.

These three factors can be evaluated in a manner that quantitatively determines the effective exposure to a pollutant that is experienced by an ecosystem. For example, the product of the average ambient biologically effective concentration of a pollutant and the average time that an entity is exposed to a molecule of that material determines the dose received by the entity, *e.g*., an ecosystem or one of its components. We hypothesize that different ecosystems will have characteristic properties related to residence time, processing capacity, and modifying factors, which can be used to differentiate classes of ecosystems, and that that these classes may be distinguished by the different effective exposures that will occur when the systems are loaded with a given quantity of the pollutant. The problem of diagnosis can be further simplified if pollutants can be grouped or classified according to their activity such that an ecosystem type processes all members of a class of pollutant in a similar manner. In this case, the bioavailable concentration might be expressed in aggregate units, *i.e.*, standard toxicity units for a given class of material. A consideration when using source, stressor, and effect within a classification scheme is that the average bioavailable concentration and residence times for the pollutant must match the time periods that are relevant to producing an effect. For example, readily available data from standard monitoring programs may not be sufficient to account for the effects of an extremely toxic pollutant buried in the sediments, if its concentration is suddenly increased due to roiling of the sediment during a storm. In this case, the relevant time period for determining source, stressor and biological effects might be hours rather than weeks or months. In streams, episodic events such as floods often govern ecological responses.

When quantitatively evaluated, residence time and processing will give an expression for the exposure of the ecosystem to biologically active concentrations of a particular stressor:

Residence Time (days)* Bioavailable Concentration (g m^{-3}) = Exposure (g m^{-3} - days)

In addition, Type B modifying factors shift the relationship along the exposure axis, causing a different effect than that expected based on the typical stressor-response relationship. The application of Type B modifying factors to the calculated exposures gives an estimate of *effective exposure*, which is the variable that drives model-based classification.

We recognized the need for two configurations of the basic model to account for uni and bi-directional water flow and two additional configurations to represent systems of each flow type where processing capacity and/or residence time is not homogenous. These four canonical models are unidirectional flow (Fig. 9a), bidirectional flow (Fig. 9b), unidirectional flow, stratified (Fig. 9c), and bidirectional flow,

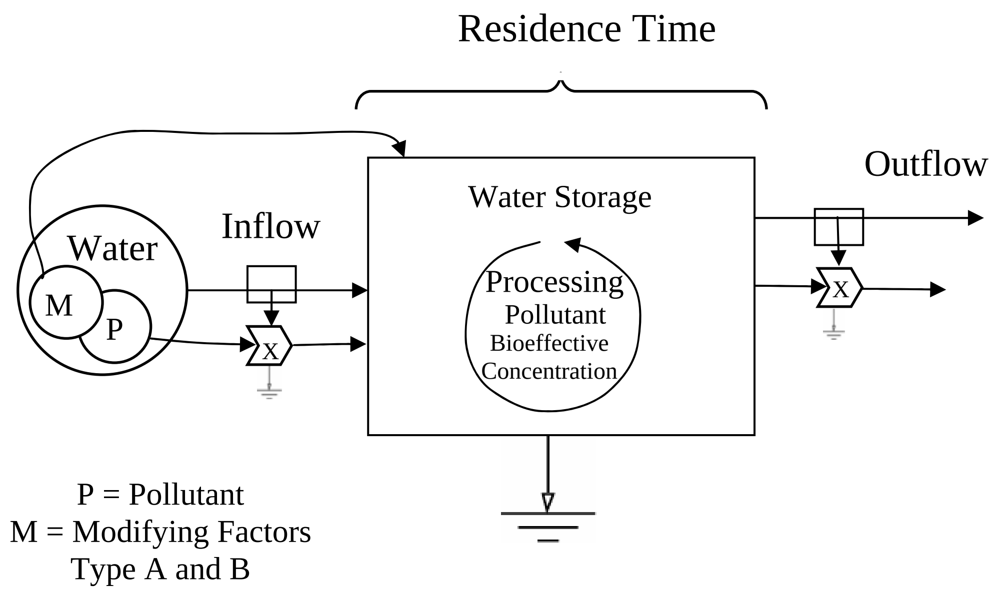


Figure 9a. Model of the factors controlling the action of stressors in aquatic ecosystems within a water storage unit with unidirectional flow.

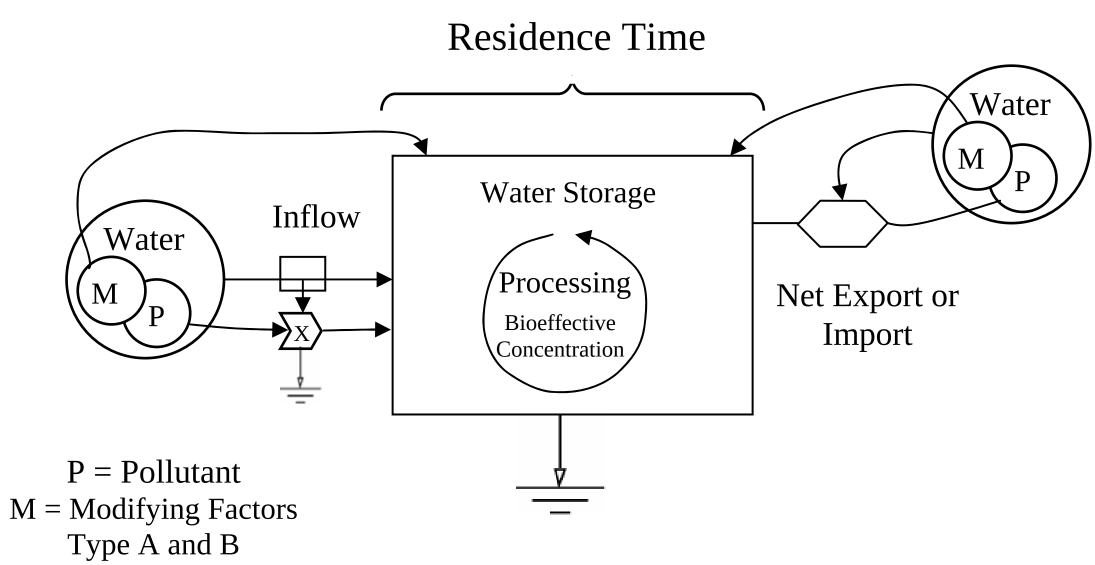


Figure 9b. Model of the factors controlling the action of stressors in aquatic ecosystems within water storage units with bi-directional flows

stratified (Fig. 9d). Additional or other internal differences in flow and/or processing capacity can be accommodated by adding compartments to the stratified modules. For example, sediment and water might be the two relevant compartments in the processing of toxic chemicals, or in broad estuaries lateral flows can be considered by adding adjacent units. When quantitatively evaluated, these canonical models will provide, as output, the effective dose of a pollutant that we expect to be related to the observations of biological effects and ecosystem condition (Figure 10).

The four canonical models given above were applied to develop more detailed models that include multiple stressors (Section 3). Transfers from one canonical module to another can be used to determine the appropriate scale for effective management of a pollutant as described below. In taking this approach, we assume and will try to verify that observed effects are a function of the effective exposure of the biota to a pollutant according to a relationship that is characteristic for an individual stressor (Figure 10). Also, we assume that a molecule of a material will be processed through similar pathways regardless of the ecosystem in which it is found, *e.g.*, denitrifying bacteria will always reduce nitrate to nitrous oxide and diatomic nitrogen gas. A known or demonstrated relationship between effective exposure and an observed biological effect allows us to use calculations of effective exposure as a classification variable to determine groupings. Other factors in the canonical models like the amount of a pollutant that is processed annually might also prove to be good general classification variables.

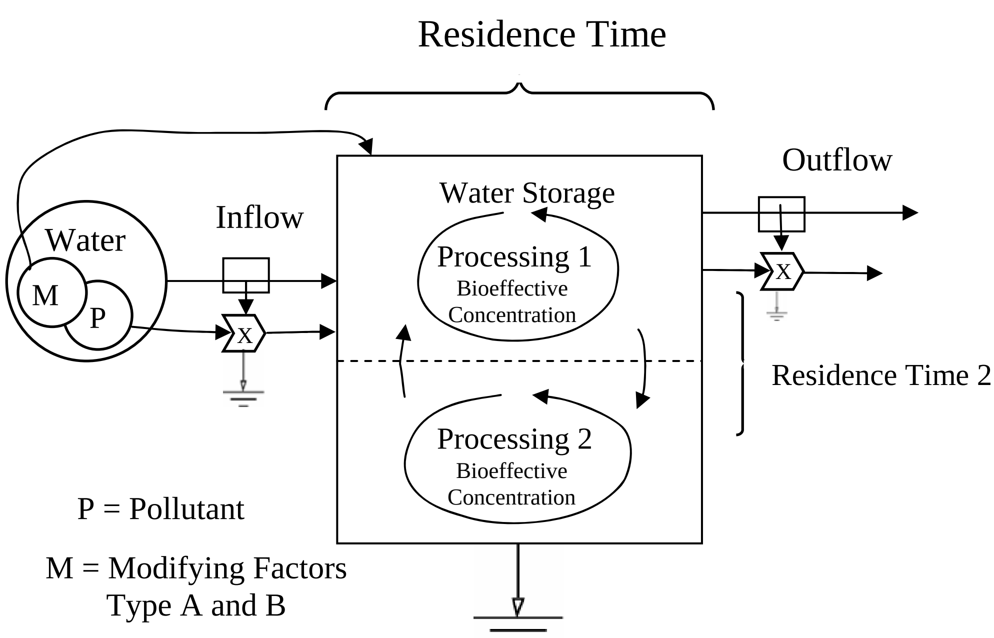


Figure 9c. Model of the factors controlling the action of stressors in aquatic ecosystems within water storage units with unidirectional horizontal flow and two different processing capacities, e.g., stratified systems or sediments and water column.

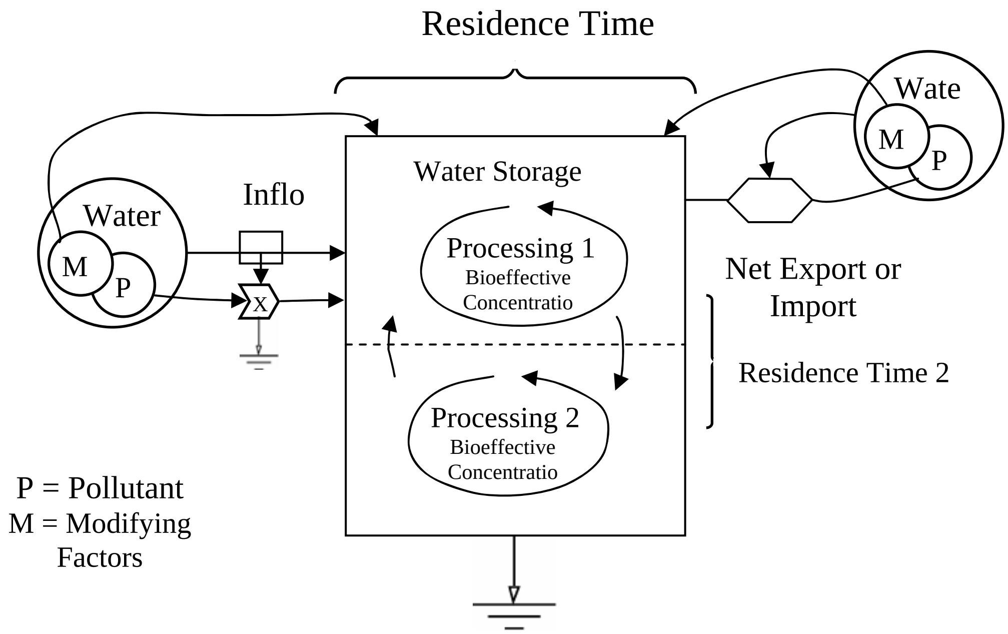


Figure 9d. Model of the factors controlling the action of stressors in aquatic ecosystems within water storage units with bidirectional flows and two processing capacities, *e.g.*, stratified systems or sediments and water column.

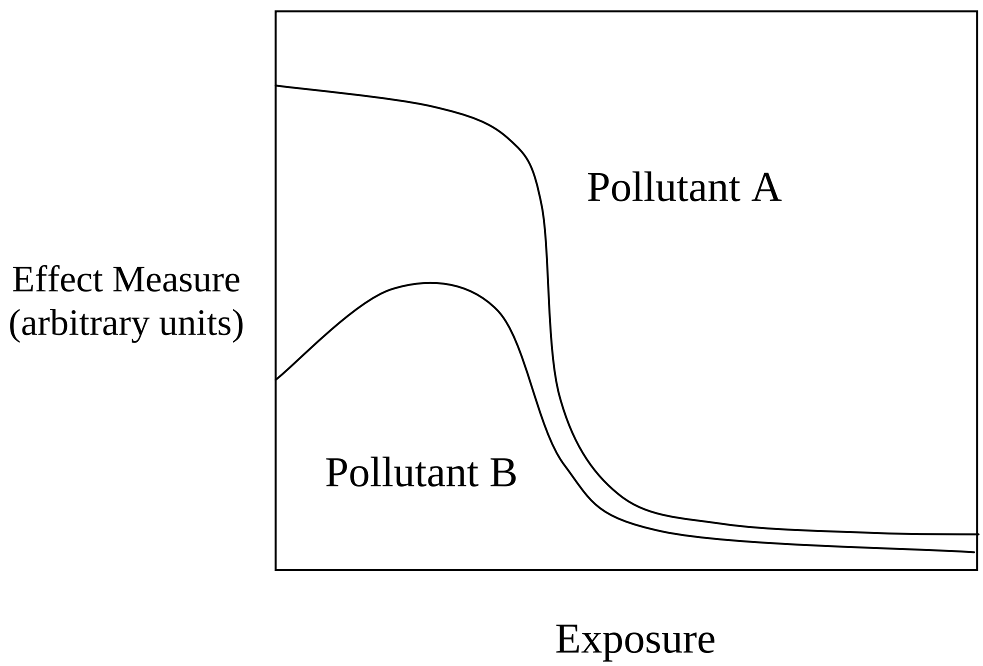


Figure 10. Two hypothetical exposure-effect relationships for two different pollutants.

2.2.3 Watershed Scale for Diagnosis and Control

Often watersheds, resource distributions, and human activity patterns do not follow political boundaries. Therefore, it is important that the states use a consistent protocol and method for the collection and analysis of data when compiling their 305(b) and 303(d) lists, so that comparable states of impairment can be determined, shared water bodies accurately assessed, and TMDLs or other regulatory programs implemented using boundaries determined by the controlling processes rather than political divisions (GAO 2002, USEPA 2002c). Consistent data on the condition of state waters displayed in a geographic format allows the national distribution of the condition of aquatic resources (water bodies, streams, wetlands), human activities, water quality variables, biological variables, etc. to be seen and studied (for example, see www.epa.gov/waters). The accurate assessment of shared water bodies requires that the states coordinate their assessment activities with adjacent states and share the relevant 305(b) data, so that distributions of the condition variables may be plotted for an entire watershed, wherever watershed boundaries extend across the borders of two or more states. When the distributions of stressors and/or effects are clustered together on the landscape, underlying factors may be responsible for their convergence, and there is a need for a broader analysis of the underlying factors, pollutant transport, and the connections between tributary stream segments and the receiving water bodies.

The fate and transport of a pollutant within the aquatic ecosystems of hierarchically nested watersheds is the key factor in determining if a TMDL must be implemented at a scale larger than the individual water body. The evaluation of a linked series of fate and transport modules provides the conceptual basis for a method to determine the appropriate boundaries over which TMDLs must be developed for a connected network of water bodies. In the *unit model* given above, the two primary factors that determine the residence time of the stressor within the system are the ecosystem's processing capacity for the pollutant in the water body and the transport of the pollutant through the system. If pollutant sources from a larger system contribute more than a certain percentage of the loading to a water body, a TMDL being developed for the water body would need to consider the pollutant loading from the larger system to be effective.

Existing water quality analysis programs, such as the Office of Water's BASINS (see http://www.epa. gov/OST/BASINS/) and its associated models, Qual2e and Hydrological Simulation Program-Fortran (HSPF) are able to make detailed analyses of the transport and assimilative capacity of streams and to define the areas that an effective TMDL must consider. Coupling these watershed models with a 2 or 3 dimensional estuary model allows seaward sources of the polluting materials to be considered. One problem with such models is that they are data intensive and may require considerable research effort and expense gathering the needed data, modeling and analyzing the results. This makes an accurate diagnosis of the causes of impairment imperative before large amounts of money are spent on a detailed model analysis to set a TMDL. A coupled set of the simple canonical models presented above may allow us to make a quick first order determination of the watershed scale necessary for the regulation and control of a particular pollutant and help focus diagnostic research on the primary cause of impairment. Further analyses using detailed numerical modeling programs can be employed to develop or refine TMDLs for stressors shown to be the primary cause of impairment.

The problem of diagnosing the causes of impairment at the watershed scale can be approached by first compiling a data base for stream reaches and other water bodies covering the area within a fundamental watershed. *Fundamental watersheds* are defined here as networks of water bodies and their associated aquatic ecosystems that are linked by flows of water and have a terminal connection to the open sea or to one of the Great Lakes (Figure 5). If pollutant inputs from the open sea and the atmosphere are small relative to watershed fluxes, fundamental watersheds represent the largest area within which wetlands, stream segments, lakes, estuaries and their watersheds must be managed to ensure that limits established for pollutants and habitat alteration will be effective. The data on land use, pollutant sources, mass transport, pollutant concentrations, and biological conditions in the water bodies and stream reaches within a fundamental watershed can be mapped to show the distributions of the pollutant in the individual water bodies, the transport of the pollutant from one system to the next, and the association between human use patterns, pollutant sources and observed impairments. The evidence supporting the presence of a causal chain linking source, stressor, and effect can be assembled by evaluating the degree of overlap between overlays of the spatial plots, showing each link in the impact chain. The water bodies influenced by point and non-point sources of a pollutant can be determined using simple fate and transport models. Once zones of association among the three links in the causal chain have been established using the assembled data, a diagnosis of the watershed management scale needed to control the pollutant can be made based on the coincidence of patterns of source, stressor, and effect and the magnitude of the pollutant flows from one system to the next. This diagnosis can be checked by identifying random locations within areas of coincidence in the individual systems and then performing PIE diagnostic methods (described in the next section) on samples from these stations and at these locations in the water bodies of concern. A pollutant shown to be responsible for impairment over a portion of the area of the fundamental watershed would have a TMDL determined for the affected portion of the watershed, which defines the scale of effective management for that pollutant.

This method can also be used to make preliminary diagnoses in more complicated cases where multiple stressors are involved, although the probability that PIE methods (see section 2.2.4) will be needed to make a definitive diagnosis is higher than in the simpler single stressor case. For example, the hypothetical stressors of acid mine drainage and nutrient enrichment are active in a West Virginia watershed. Assume that old coal mines dot the northwest quarter of the watershed, where many stream reaches have low pH. This region drains into the lower watershed. The entire northeast quarter of the watershed is forested. Acidity is a problem in the lower half of the watershed although no mines drain from that area. Nutrient enrichment is also a problem in the lower watershed where several municipal sewage treatment plants discharge into the river. Biological impairment of stream biota occurs throughout the northwest quarter and in the lower watershed but not in the northeast quarter. No other stressors have been measured in the region; however, there is a nearby atmospheric monitoring station that shows elevated SO_4 in rain fall. Applying the spatial coincidence method proposed above and the rules of logic proposed by Hill (1965) and USEPA (2000a), we observe from our overlapping maps that either acid mine drainage (AMD) and/or acid rain might be responsible for biological impairments in the northwest quarter. The fact that no impacts are observed in streams in the Northeast quarter allows us to eliminate acid deposition pointing to AMD as the most probable cause of biological impairment in the streams of the Northwest quarter. In this case, the overlapping patterns show the presence of two possible sources, a stressor, and an effect, but one source could be eliminated based on the evidence of co-occurrence. The information given is not sufficient to make a similar diagnosis for the cause of impairment in the lower watershed, because we are unable to separate nutrient enrichment from acidity, both of which have source, stressor, and effect present. In this case, direct cause and effect tools that link source and stressor and stressor and effect such as those under development in PIE methods can be applied to try and sort out the cause of biological impairment in the lower watershed.

2.2.4 Pollutant Identification and Evaluation

Pollutant Identification and Evaluation is a method for the diagnosis of the causes of impairment to aquatic ecosystems in estuaries and other water bodies. PIE is based on the experimental methods and logical rules of analysis used to demonstrate the causes of toxicity that have been developed over the past 15 years in the Toxic Identification and Evaluation, TIE, research program. Recently, a general logical approach to determine the most probable cause of an observed biological impairment in an aquatic ecosystem has been promulgated by the USEPA. Stressor Identification, SI, (USEPA

2000a) provides a framework for determining the cause of biological impairments observed in aquatic ecosystems. Currently, it has been applied primarily to fresh water streams; however, the same logical techniques, *e.g.*, elimination, association, and weight of evidence, can be applied to determine the causes of impairment in other water bodies such as lakes and estuaries.

2.2.4.1 Methods for Discerning Cause and Effect

The basic principles used to identify the most probable cause of a condition from observed associations are elucidated by Hill (1965) for problems in epidemiology. Such an approach forms the basis for demonstrating probable cause in many scientific disciplines. SI, TIE and PIE use the elements of Hill's causal criteria in arguing for causation from observations of association. PIE uses these logical rules to determine the most probable cause of an observed condition; therefore, it is compatible with SI and operates within its logical framework, but it also goes beyond the methods used in SI by developing laboratory and field methods to directly demonstrate the presence of causal mechanisms.

Energy Systems Models approach the problem of causality somewhat differently by constructing causal networks based on proven or hypothesized relationships among system variables. The fundamental notion underlying this approach is that the transformation of energy is the proximal cause of all action. These models can be further analyzed to rank causes and allocate effects among multiple causes. Hill (1965) stated that his rules to make the case for causation from observed correlation are a second best, though practical, alternative to the detailed scientific research and experimentation that is often necessary to demonstrate the mechanisms of cause and effect. Modeling causal networks and developing tools to demonstrate causal mechanisms deal directly with causation. Hill's postulates are logically consistent with such direct research and make a contribution where such direct causal analyses do not exist or are difficult to implement. It is our position that the USEPA needs both kinds of research and tools to effectively deal with cause and effect in environmental systems. Both SI and PIE are needed and can be used to support the 303(d) listing process and the development of water quality regulatory programs such as TMDLs. Energy systems models of causal networks are perhaps best used in the allocation of effects among multiple causes. SI, PIE, and energy systems models are not identical approaches and all three provide unique tools needed in the cause and effect toolbox. Some of the characteristics of and differences between SI and PIE are given in the following paragraph along with the ways in which PIE and SI augment and support each other.

PIE uses a succinct logical sequence, the impact pathway, to demonstrate causality and it is focused on the development of particular tools for the identification of the causes of impairment in estuaries, where multiple active causes are the rule rather than the exception. SI summarizes and codifies the logical rules of causal analysis in a method that can be used in the application of PIE and other analyses. SI relies largely, but not exclusively, on existing information and the weight-of-evidence to determine probable cause based on observed associations. The development of particular tools to fit within the framework is encouraged although not a part of SI *per se.* An SI analysis is triggered by the observation of a biological impairment, whereas, a PIE can be triggered by any violation of a water quality standard. This is important because the pathway that a state or local government follows into a causal analysis is not always through the observation of a biological disturbance, *e.g.*, the largest category of impairment on the 303(d) lists is unknown. SI is designed to be most effective in determining the cause of impairment when the weight of evidence points to a single most probable cause among many candidate causes, whereas, PIE focuses on developing tools that will allow diagnoses of the causes of impairment in estuaries where multiple stressors are the rule and where several stressors may be actively causing biological impairment in most systems. PIE and SI researchers have met several times over the past two years to compare approaches and exchange information. For example, the PIE method considers the sources of stress early in the diagnostic process and SI adopted early consideration of source, because it proved useful. PIE complements SI through the development of specific diagnostic tools for estuaries that can be used within the context of the SI approach. At present, SI has been applied primarily to determine the cause of biological impairments in fresh water stream systems, where it has proved to be an effective vehicle for communication and diagnosis. Currently, PIE tools and methods are being developed and tested in case studies of two estuaries and in a retrospective case study of New Bedford Harbor, MA.

2.2.4.2 Description of the PIE Method

The Pollutant Identification and Evaluation method has four phases with costs and complexity increasing from the first to the fourth: Phase 1: *Screening* in which we compile existing data for verification and diagnosis, Phase 2: *Identification* in which we establish the presence of source, stressor, and effect, Phase 3: *Linkages* in which we demonstrate the connections between source and stressor and stressor and effect, and Phase 4: *Confirmation*, in which time and circumstance are relied upon to demonstrate the accuracy of our diagnoses. Pollutant Identification and Evaluation uses a phased approach where a number of candidate causes are possible and specific tools are used to prove or disprove logical links and/or evidence is used to develop a cause and effect relationship or show the lack of one.

In most cases, the water bodies under consideration are on the 303(d) list; therefore, the screening and verification phase should be able to find evidence that confirms the earlier assessment of impairment performed by the State. For the purposes of our diagnostic research, an ‘effect’ is defined as an actual biological or ecological impairment observed at some level of organization in the water body under investigation. Therefore, for us the presence of the stressor alone is not sufficient cause for concluding an effect is occurring or likely. Because State and Regional practices sometimes assign water bodies to the 303(d) list based on the presence of a stressor (*e.g.*, exceeding a Water Quality Criteria) and without observing a biological effect, our method could find that no biological effects are present at a 303(d) listed site. We believe a scientifically defensible method must be able to link directly the presence of a stressor to an effect before making a diagnosis. The only exception to this definition is when pathogens are the suspected stressor. In this case, the presence of pathogens in numbers known to represent a risk to human health is sufficient evidence to conclude that an effect is present or likely. Thus, we have assumed that an effect on aquatic biota must be detectable for us to make a diagnosis of the cause of impairment, but we do not assume that this affect will necessarily be known before the analysis is carried out.

Phase 1 analysis is triggered once a water body has been placed on the 303(d) list for a water quality standard violation, an observed biological impairment, or other observed impairment. In Phase 1, data on the water body, its immediate watershed, and the next larger system, of which the watershed is a part, are compiled and used to evaluate the system and, if possible, make a diagnosis of the cause of the observed impairments (see Phase 1 in Table 1), when the evidence is strong enough (See SI guidance, USEPA 2000a). Screening provides information on human activities in the watershed and may provide clues to show how these activities are linked to the production of pollutants and to physical alteration of the water bodies. Phase 1 data may be organized into causal webs using the top-down, bottom-up or energy systems approaches to evaluate the interactions that link human activities with stressor sources and stressor concentrations within watersheds. If possible, Phase 1 verifies the probable cause of impairment by using existing data to demonstrate that sources of a pollutant are present and that related effects have been observed in the 303(d) listed water body in question. If a biological effect cannot be verified from the existing data, subsequent phases (2&3) of the PIE method provide a framework for gathering evidence and demonstrating that an effect is or is not present (Table 1). On one hand, if the existing records are complete and extensive and demonstrate impairment, a given water body might go directly into Phase 3 based on Phase 1 results. On the other hand, if such complete records show no impairment is present, the analysis can stop after *Phase 1*. A final task in the Screening Phase is to collect available information on the cross boundary flows of the stressors. This information will be used as described above to determine if diagnosis must be pursued on a scale larger than the water body to result in an effective TMDL.

The compiled data for an individual water body and its watershed is represented spatially to help develop sampling plans for the work done in Phase 2 and 3 and to look for patterns as described in the preceding section on watershed diagnosis. The screening process may also identify data gaps to focus on in Phase 2 and 3. The list of diagnostic tools shown in Table 1 are under development for use in Phases 1, 2, and 3 of a PIE; however, this list should not be viewed as a comprehensive listing of all tools that may be needed or prove to be useful in establishing the causes of an ecological impairment. Other tools may be added to this list to be developed and tested in the future.

Phase 2 seeks to identify biological effects in a water body and the stressors that might be responsible for the observed impairments. Phase 2 tools (Table 1) provide data to show the presence of source, stressor, and effect along the causal pathway discussed above. In addition, Phase 2 tools can corroborate evidence from the state and regional assessments and the screening and verification phase. Table 2 identifies the possible outcomes of Phase 2 and it contains guidance on how to proceed given these outcomes. Generally, we recommend that evidence for all three components of the impact pathway be present to advance to Phase 3, but we also recommend courses of action when a specific line of evidence is absent. In the case where both a source and a stressor are evident, but no effect can be found, the stressor might still be moved into Phase 3 due to the preliminary nature of the search for effects performed in Phase 2. In all cases the strength of the evidence should be used along with the logical rules of causal analysis in eliminating a stressor in Phase 2 or in moving it ahead to Phase 3.

In Phase 3, diagnosis moves from developing lines of evidence for source, stressor, and effect, to a clear and testable demonstration of cause and effect links between these lines of evidence. Phase 3 tools (Table 1) seek to provide a definitive demonstration that (a) the causal chain between the source of a stressor and the observed effect is unbroken and (b) the stressor is capable of producing the observed effect under controlled conditions and under the prevailing conditions present in the water body under analysis. Table 2 gives the conditions that may occur at the beginning of Phase 3 and suggests actions to confirm or eliminate the stressor. If (a) is false and no other cause of the impairment can be found, the stressor is designated for further research. If (b) is false, the stressor is eliminated from further consideration. A stressor is considered to be an active, but not necessarily the only, cause of an observed effect if the above two conditions linking source, stressor and effect can be demonstrated to be true. Table 3 gives possible results at the end of Phase 3 and the suggested actions that follow from each sequence of events. The strength of the evidence from the Phase 3 tests should be used along with the logical rules of causal analysis in the elimination of a stressor or in the diagnosis of a stressor as an active cause of impairment.

Phase 4 confirms that the diagnosis of the cause or causes of impairment is correct. PIE methods are used to make a diagnosis of impairment for a water body or watershed based on tests carried out in the field and in the laboratory. Unlike a controlled experiment, which can be duplicated in the laboratory and where the results of a treatment can be statistically confirmed, ascertaining the results of manipulating ecological systems in the field may be difficult and involve relatively long time delays before the results of the manipulation are known. Generally, confirmation of an initial diagnosis cannot be made until after a TMDL has been performed and the water body monitored for some time to determine if the anticipated recovery takes place. In most cases, it will require several years of follow-up monitoring to demonstrate that an initial diagnosis was correct. Successful classification holds some hope for hastening the confirmation phase. If water bodies can be classified into groups with similar behavior under loading with a pollutant, the results of past TMDLs could be used to establish an expectation of confirmation for systems with similar parameters. In this case, similar circumstances allow an early peek into the probable outcome of the confirmation phase. A final confirmation can only be made after the system has been monitored for a sufficient time to demonstrate recovery of the biological variables originally judged to be impaired.

Table 1. Proposed diagnostic tools used in the development of Pollutant Identification and Evaluation (PIE) methods for determining the causes of impairment to the aquatic ecosystems in estuaries. Phase 1 is screening and verification. Phase 2 is the identification of source, stressor, and effect. Phase 3 is establishing the linkage between source and stressor and stressor and effect. Phase 4 is confirmation of the diagnosis.

Phase	Stressor	Diagnostic Tool	Diagnostic Information	Status of Tool
1	All	GIS maps. High quality environmental data sets, *e.g.*, NAWQA, EMAP. The scientific literature.	Possible sources. Determine reference sites. Plot data on GIS maps to give spatial relationships for sources and stressors. Identify data gaps.	GIS maps and data need to be checked for accuracy and compatibility of data.
2	Nutrients	Grain size normalized TOC.	Indicator of eutrophic conditions (nutrient loading with C, N, P).	Needs to be validated. May also give information on clean sediment and toxic chemicals.
		River DIN/DOC/PIN and residence time analysis.	Indicator of normal or excessive DIN/PIN loadings.	Relationships need to be developed between DIN, PIN and benthic effects.
		Analysis of dissolved oxygen concentrations. Dawn-dusk and continuous.	Indicator of organic enrichment resulting in eutrophic conditions. Effect measure for N, P, and C.	Sampling must be complete to avoid erroneous conclusions due to natural variability
	Toxic Chemicals	Toxicity testing and limited chemical analyses.	Integrator for all toxic chemicals.	May not detect toxicants that have a chronic effect, dietary route of exposure, or for which organisms may not be sensitive (*e.g.* Hg, dioxins) Increasing classes of species tested or exposure duration may increase sensitivity.
	Habitat Alteration* by pollutants.	Reduction-oxidation potential (RPD) assessment.	Effect measure for nutrients, suspended and bedded sediments, toxic chemicals or physical change. Oxygen availability in sediments.	This measure is based upon Pearson and Rosenberg (1970) ‘better’ conditions have a deeper RPD than impacted areas. Some variability is inherent in measurements. Validation of habitat quality based on RPD depths for different grain size habitats needs to be performed.
	Clean Sediments	River particulate load analysis. Turbidity.	Measure total suspended particulate inflow	Criteria for suspended and bedded sediments in marine ecosystems have not been established. Turbidity may measure biological particles as well as sediment. Use dry weight/ash weight

Phase	Stressor	Diagnostic Tool	Diagnostic Information	Status of Tool
Table 1 cont'd		Grain size analysis.	Measurement of grain size can act as a surrogate measure of energy regime.	Current measurements may be needed to verify flow conditions.
	Pathogens	Bacteria measurements. For example: *E. coli*	Indicator of the presence of bacteria from vertebrate sources.	Does not indicate whether the source of bacteria is natural (wildlife) or from human/domestic or agricultural activities.
	All stressors (Nutrients *Habitat alteration** Toxic chemical)	Benthic community analysis.	General state of benthic community. May validate impairment assessment relative to an appropriate reference system.	Reference systems may be difficult to find. Does not currently diagnose a single stressor. Relationships between benthic community condition and 'natural' forcing factors need to be developed.
3	All stressors	Models	Provide information on mechanistic, organism and ecological pathways that support linkages between sources, stressors and effects	Specific to the stressor modeled.
	Nutrients	Measurement of N, P, chlorophyll a and DO over appropriate time periods. Stable isotope analysis.	Provides temporal information on the amounts of nutrients present. Linkages to source.	Correct interpretation and quality control of data is critical to avoid false positives. Reference estuarine systems may be difficult to find. Nutrient criteria for marine systems have not been established.
	Toxic Chemicals	Toxicity Identification Evaluations (TIEs)	Determines which toxic chemicals are causing observed toxic effects	Methods require further development and validation. Because these methods use toxicity tests, all the potential issues of toxicity tests are inherent in TIEs. See Phase 1
		Physical and chemical measurements of suspected toxic chemicals. Calculate bioavailability using models.	Provides information on the amounts of bioavailable toxic chemicals present in water and sediments. May provide links between source, stressor, and effect.	Correct interpretation and quality control of data is critical to avoid false positives. New analytes need method development.
	Habitat Alteration*	Physical and chemical measurements Grain size, salinity and TOC analyses.	Provides information on the amounts of nutrients, toxic chemicals, suspended and bedded sediments and the physical properties of water and sediment samples.	Interpretation of data relative to a reference, baseline or threshold measures is critical to avoid false positives.

Phase	Stressor	Diagnostic Tool	Diagnostic Information	Status of Tool
Table 1 cont'd	Suspended and bedded sediments	Calculate and/or measure energy regime from grain size distributions or current meter data.	Grain size distribution may provide information about energy regime. May provide historical change in type of deposition as a result of agricultural/construction activities.	Reference or historical deposition rates and associated grain size may not be available. Criteria for suspended and bedded sediments in marine ecosystems have not been established.
		Deposition analysis.	Rate of sedimentation. May link stressor with source.	Determining deposition is difficult. Criteria for clean sediment reference sites in marine ecosystems have not been established.
	Pathogens	Bacteria source analysis. Genetic finger printing.	Distinguish between human and non-human *E.coli*, as the presence of the bacterium often automatically indicates a violation of many 'fishable' and 'swimmable' criteria.	Determine which tests are accurate and available for extramural purchase.

*Habitat Alteration The stress from habitat alteration occurs in two forms: direct and indirect. Examples of direct effects include anthropogenic activities that result in physical changes to the habitat like damming, dredging, paving and filling. Indirect habitat alteration results from the adverse effects of a pollutant, such as, excess toxic chemicals, suspended and bedded sediments, or nutrients. Both direct and indirect habitat alterations affect benthic communities. However, the PIE method is only concerned with diagnosing habitat alteration that results from the effects of pollutants. The direct effects of habitat alteration on benthic communities will be removed through the choice of reference systems.

Table 2. A Decision Table for evaluating outcomes in Phase 2 of a PIE. For each stressor considered in Phase 2, there should be evidence of a source, the presence of a stressor, and an effect to move to Phase 3. An X indicates presence.

Source	Stressor	Effect	Guidelines for Action
--	--	--	Eliminate stressor from consideration
X	X	X	Move stressor ahead to Phase 3
--	X	X	Look for source, look to a larger system to ensure all possible sources are considered. Look for a precursor to the stressor that may have a source. Repeat Phase 2. If the source is not found, one may want to move the stressor to Phase 3 to determine if a strong linkage can be developed between the stressor and the effect. If a strong linkage can be developed, one may want to invest more resources to determine if a source exists.
X	--	X	May be the result of temporal loading, repeat Phase 2 with attention to temporal scales. The source may emit a different stressor with a similar effect. Repeat Phase 2 with attention to all possible stressors. The source may be spatially linked to a proximal stressor, if one is certain the proximal stressor is in the causal pathway, *e.g.*, monitoring low DO or chlorophyll events in the absence of nutrient concentrations. If ultimately no evidence of a stressor or proximal stressor is found, eliminate stressor from consideration.
X	X	--	Need to ensure that the endpoint measured is a sensitive effect endpoint and/or the stressor is bioavailable. Given the preliminary nature of Phase 2 effect endpoints, one may want to move this stressor forward to Phase 3. If no effect is found, eliminate stressor from consideration.
X	--	--	Eliminate stressor from consideration. Continue assessment of other stressors.
--	X	--	Need to ensure that the endpoint measured is a sensitive effect endpoint. Need to consider possible lag time for the stressor to have a measurable effect, or if the stressor is ephemeral. Need to look at the larger system to ensure all possible sources are considered. Look at the assimilative capacity and residence time of the pollutant in the system. Repeat Phase 2.
--	--	X	Consider other stressors that may have similar effect endpoints. Consider possible lag time for the stressor to have a measurable effect, or if the stressor is ephemeral. Also, look to a larger system to ensure all possible sources are considered. Repeat Phase 2.

Table 3. A Decision Table for possible outcomes of Phase 3 showing the presence or absence of linkages between source, stressor and effect. In all cases, the strength of the weight-of-evidence should be used along with professional judgment in elimination or diagnosis of a stressor. An X represents an affirmative result.

Source and Stressor Linked	Stressor and Effect Linked	Guidelines for Action
--	--	Eliminate stressor from consideration.
X	X	Links established between source, stressor and effect. Weight of evidence may include two independent lines of evidence that point to the same stressor, and laboratory and field measures. Diagnosis completed.
--	X	Look for source, need to look to larger systems (increase spatial scale) to ensure all possible sources are considered. Repeat Phase 3
X	--	May be a result of temporal loading, repeat Phase 3 with attention to temporal scales. Source may emit several different stressors, repeat Phase 3 with attention to all possible stressors. If ultimately no link between a stressor and an effect can be established, eliminate stressor from consideration.

2.2.5 Modeling Methods and the Allocation of Effects

A systematic and exact framework based on physical laws, *e.g.*, the laws of thermodynamics, is helpful in modeling, understanding, and predicting patterns and processes in the environment. In this paper, we used such a framework for model development. Energy Systems Theory (Odum 1971, 1983, 1994) was chosen to guide the development of conceptual models of the action of stressors in aquatic ecosystems (See Text Box D). Energy Systems Language is a vehicle for model-building that allows the symbolic and mathematical representation of the energetic and kinetic aspects of stressor-response relationships in an integrated manner. Energy transformation is the underlying cause of all action and can be used to trace causal pathways within a network organization. Thus, a consideration of energy transformations was a key factor in gaining an integrated understanding of those factors controlling the fate and transport of pollutants within an ecosystem, including the metabolic processing of these materials, and the action of modifying factors in determining bioavailable concentrations and the biological impacts of stressor action.

We developed a hierarchical, modular framework for constructing conceptual models. At the highest level of aggregation we designed a set of four canonical models which can be used to describe the links between aquatic ecosystems on the landscape. The more detailed generic conceptual models for a particular pollutant, which are described below, can be plugged into the processing box of the appropriate canonical model to represent a particular system in more detail. The appropriate forcing functions must be added to accurately represent these more complex systems. The canonical models serve as a frame within which more complex pieces can be inserted to evaluate the details of a particular pollutant’s action within a particular ecosystem type.

In this paper, we used a progressive process for the development of detailed conceptual models. The first step in this process was writing narrative descriptions of the stressors identified in the Aquatic Stressors Implementation Plan (USEPA 2002b). This process allowed everyone to contribute to the conceptualization process and it drew upon the combined expertise of the group in assembling knowledge about each stressor. These narrative descriptions were produced in a standard format that facilitated comparison and the development of overview figures of the factors controlling each stressor. The information from the narrative descriptions and overview figures was then used to construct detailed models for each stressor. We used ESL to develop the detailed conceptual models for each stressor including the physical forces and flows that control the residence time of pollutants in the system, the biological and chemical factors that determine processing capacity (type A modifiers) and the modifying forcing functions and materials (type B modifiers) that alter stressor action.

The detailed conceptual models serve as a guide to building quantitative simulation models for the allocation of effects among several causes when more than one cause is active. In addition, the detailed conceptual models can be used as a checklist of the important processes that might affect the action of a stressor in an aquatic ecosystem. These more complex models include the factors that we believe to be important in controlling the action of the four stressors, but they should not be construed to be a comprehensive representation of the effects of a particular pollutant in a particular aquatic ecosystem. Rather, they are a guide to thinking about such problems. As always, good quantitative models and analysis will depend on the perspicacity of the investigators. Nevertheless, with some practice and the detailed examples as a guide, we hope that many investigators will be able to use this model development process and simplification through functional aggregation (Odum 1976) to develop their own conceptual models to guide diagnostic research. The conversion of simplified versions of these detailed models of stressor action into computer programs for simulation will make it possible to predict the behavior of particular ecosystem variables under various levels of pollutant loading. The special character of habitat alteration as a modifying factor on the action of other stressors was recognized and addressed in its conceptual model. A sensitivity analysis of the effects of increasing concentrations of the several stressors that are demonstrated causes of impairment in a place allows observed biological effects to be allocated among the stressors.

2.2.6 Response-Based Classification

Our work on model-based classification is in its inchoate stages; therefore, only our research strategy is reported here. A report on the statistical classification of coastal ecosystems of the 48 contiguous states using physical variables alone has been completed (USEPA 2004). To simplify the problem of diagnosing the causes of impairment, a classification scheme must be based on similarities and differences in the way that aquatic ecosystems respond to the loading of a pollutant. If all systems respond to a particular concentration of a pollutant in the same manner, then all systems are in a single class and classification does not help. However, if ecosystem types correspond to coherent groupings of effective exposures, aquatic ecosystems may be partitioned into stressor-based classes that will aid the process of diagnosis and restoration. Responses to habitat alterations, which are not the result of the addition of a pollutant, are considered separately.

We hypothesize that many measurable ecosystem variables can be quantitatively related to three key system characteristics (residence time, processing capacity, and modifying factors) that control the effective exposure of an ecosystem to a pollutant. One way to identify classes is to list all the variables that are related to each one of the three key factors controlling effective exposure, and then group these variables based on quantitative or pseudo-quantitative relationships to ecosystem condition variables. A brief description of how we will use models to develop a classification scheme follows: (1) One of the four canonical models representing a water body type will be used as a template for evaluating the activity of individual pollutants within the water body. (2) For a given water body or a relevant functional division of that water body, we will evaluate each of the 3 factors in our model to determine effective exposure. (3) We will then look for differences in the classification variables (obtained above) that can characterize water body types associated with ranges of effective exposure. (4) We hope to define stressor-based classes of aquatic systems from the data on residence time, processing capacity, and additional factors by looking for patterns and differences in the relationship between effective exposure to a stressor and the observed behavior of condition variables within the aquatic ecosystem.

As mentioned above, exposure is a function of the biologically effective concentration of pollutant molecules and the time that those molecules reside in the ecosystem. Residence time of the pollutant in turn depends on the residence time of the water and the capacity of that ecosystem to process the polluting material. To verify the assumed relationship between exposure and biological effects, we will use existing data, wherever possible, to demonstrate that exposure to the pollutant is directly related to ecological effects through a quantified dose-response relationship. The efficacy of this model in classifying ecosystems according to different effective exposure regimes verified by observed differences in response is currently being tested.

The eco-energetic system of Odum *et al*. (1974) is an alternative classification method for coastal water bodies that will be updated using emergy signatures (see text box C) determined from our database. By documenting the anthropic and natural parts of the emergy signatures of coastal water bodies (Odum *et al*. 1977, Campbell 2000) along with the ecological organization found in those systems, we may be able to establish a characteristic relationship between the qualitative and/or quantitative features of the emergy signature and the ecological organization to be expected in the coastal system receiving that signature. Stressor-based and non-stressor-based classes of coastal systems may be identified in this way and compared with the classes of coastal systems developed by statistical and model based methods.

(C) Energy and Emergy Signatures: **An Alternative Method for Classifying Ecosystems**

If the external forcing functions of a system are arranged in categories along the abscissa of a graph in order of increasing *transformity* and their magnitudes converted to energy and plotted on the ordinate, the plot is called the *energy signature* of the system (Odum *et al*. 1977). If these energy values are then multiplied by their appropriate *transformities* and the results plotted as before, the resulting graph is the *emergy signature* of the system (Odum 1996). Developing a library of energy and emergy signatures provides a new approach to the old problem of predicting the ecosystems that develop under the influence of known environmental forcing functions. Assembling data on the forcing functions from many systems and grouping them into categories based on the emergy supplied may serve as a robust means for predicting ecological structure and function. Emergy signatures can also provide a unique method for characterizing ecosystems using the quantitative and qualitative differences in the signatures. Campbell (2000) demonstrated that the emergy signatures of a fluvial, a lagoon, and a macrotidal estuary were different. If the ecological processes and species which can utilize the dominant energies of the signature are also different, the features of the energy signature may serve as a means for predicting ecological organization.

The “natural” or expected emergy signature of a system is a baseline for estimating potential impacts. An emergy deficit or excess in any category in the signature gives a quantitative measure of the stress on a system adapted to the original inputs. The total of subsidizing (+) and stressful (-) emergies may be an integrated measure of overall stress on an ecosystem. Also, the impact of stress on ecosystems may be measured by changes in ecological organization and network power flow where both are

(D) The Energy Systems Language

The Energy Systems Language, ESL (Figure 3), is particularly suited for developing diagnostic models because the transformation of available energy is used to trace causality in the ecosystem upon which both diagnosis and prediction depend. ESL can be used in both a general form for building conceptual models and in a specific form which allows complete mathematical expression of the model and simulation (Odum and Odum 2000). The development of simulation models facilitates the allocation of observed effects among multiple demonstrated causes through a sensitivity analysis of the models. ESL is very rich allowing, the modeler to represent the complex energetic and kinetic aspects of a system in a parsimonious manner. Campbell and Wroblewski (1986) showed that box diagrams (an alternative model system) required considerable written explanation to approximate the system description that was achieved with models constructed using ESL. A disadvantage of using ESL is that it requires some training for a scientist to become fluent in the specific form of the language. For conceptual model development, a familiarity with the general meaning of the symbols and the rules of syntax for building diagrams is sufficient, because translation of the network into mathematical expressions and equations is done later. Understanding the more complex conceptual models does require some patience in tracing interaction pathways on the diagrams and using labels to key the numbers on the pathways to the associated tables.

The ELS symbols and their mathematical definitions are widely available in the literature (Odum, 1971, 1983, 1994), thus only a brief description of the language and symbols is presented here (Figure 3). Systems are viewed as being composed of components and processes that are interconnected by energy flows and interact within a network that is influenced by one or more external forcing functions. External forcing functions (circles) describe energy inflows to the system (both matter and information have energy equivalents); ecosystem components are producers (bullets), consumers (hexagons), storages (tanks); processes are shown as interaction symbols (the unidirectional double-pointed rectangular arrow) or work gates where energy is being transformed and where some energy is degraded into unusable form; longitudinal dispersion/vertical mixing (a rectangular arrow with points in opposite directions); logic programs (symbol with four concave sides) handle logic and switching functions; pathways carry energy, matter, or information (lines with arrows) and connect forcing functions and components directly or through the work gates. The symbol with an arrow going to ground is the heat sink carrying energy that is no longer available for use within the system. A rectangular box is used to show system boundaries or to delineate subsystems.

3.0 Development of Detailed Conceptual Models

Detailed conceptual models were developed for each of the four classes of aquatic stressors. Each detailed model considers: (1) information available on the action of the primary stressor within an ecosystem, (2) the major stressors interacting with the primary stressor, and (3) the network of stressor interactions affecting biological components and flows in the aquatic ecosystem. These detailed models are the starting point for developing computer simulation models capable of allocating biological effects among multiple demonstrated causes of impairment. As a first step in developing these detailed conceptual models, we wrote narrative descriptions of the four stressor classes listed in the Aquatic Stressors Framework (USEPA 2002b). In this section, the narrative descriptions of the stressors are presented using a similar outline for each, along with the translation of these narratives into detailed ESL models for each stressor.

3.1 Narrative Descriptions of the Stressors

Narrative descriptions and summary diagrams for the four classes of aquatic stressors that are of particular concern in the Aquatic Stressors Framework are presented in this section. The stressors considered are as follows: (1) nutrients, (2) metal and organic toxicants, (3) suspended and bedded sediments, and (4) altered habitat.

3.1.1 Nutrients

Definition: Nutrients are nourishing elements required by all living systems for normal functioning. The primary elements associated with nutrient over enrichment and cultural eutrophication are nitrogen and phosphorous. Nitrogen is important in causing and controlling eutrophication in marine systems, while phosphorous is generally recognized as the controlling element for nutrient enrichment in most freshwater systems. The concurrent addition of nitrogen and phosphorous to ecosystems can have synergistic effects, stimulating the growth of organisms at the base of the trophic web, such as phytoplankton. At times, other elements, like silica and iron, may regulate phytoplankton growth through their role as limiting factors (*e.g.*, the occurrence of diatom and toxic dinoflagellate blooms in coastal waters) but their importance with respect to nutrient enrichment is often secondary to nitrogen. Characteristic features often observed in eutrophic ecosystems are the accumulation of organic carbon in sediments, increased abundance of certain algal and macroalgal species, *e.g.*, blue greens and *Ulva*, overgrowth of other species that can tolerate the new conditions, and low dissolved oxygen in the sediments and in the overlying water.

Natural occurrence: Living systems require many elements for their normal functioning and over time they have adapted to ensure that as far as possible (within the constraints imposed by climate and the geochemistry of the substrate) optimum supplies of these materials will be available for use. In the sea, phytoplankton require approximately 16 moles of N for every mole of P they assimilate, the Redfield ratio (Redfield, 1958). If the ratio of available N to P is less than 16:1, primary production in the ocean will be limited by N; if the ratio is higher, production will be P limited. The availability of nutrients is affected by N and P inputs, storage and recycling (denitrification) and nitrogen fixation. Because living systems adapt to prevailing conditions, we find a range of nutrient states that are characteristic variations within many kinds of ecosystems, *e.g.*, lakes, streams, wetlands, *etc*. In general these variations manifest as different structural and functional states of the ecosystems, which are determined by adaptation to the different levels of nutrient supply. These states are termed oligotrophic, for low nutrient inflow, mesotrophic for moderate inflow, and eutrophic for high inflow. In the absence of human activities ecosystems are expected to manifest a range of characteristic structural and functional conditions that correspond to the natural range of nutrient inflows into the system.

Changes caused by human activities: One of the chief characteristics of human economic and cultural use of natural resources is to disrupt the natural cycles followed by the chemical elements, including nutrients (Odum et al. 2000). The addition of excess nutrients also disturbs the expected pattern of production (P) and consumption (R) within ecosystems (see discussion on P and R, Fig. 6). In a process called cultural eutrophication, human economic and social activities concentrate nutrient elements on the landscape and then redistribute them in patterns that differ from those that were formerly established by the natural biogeochemical cycles. Most often, the ecosystems that receive these concentrated nutrient materials are not adapted to process them efficiently. Those animals and plants that can process the excess nutrient inflows flourish, moving the structural and functional characteristics of the original ecosystem toward those of a more eutrophic system.

Mechanisms of stressor action: The general mechanism of action for excess nutrient is to over-stimulate plant growth. If sunlight, nutrients or any other environmental factor is inadequate to support further growth of organisms capable of primary production, the condition is said to be "limiting growth". When nutrients are unlimited, photosynthetic organisms grow and multiply until a new limitation is encountered. Given adequate nutrients, phytoplankton will multiply until their density limits further growth by self-shading (light limitation). In general, those plants that are best adapted for nutrient uptake are the ones that flourish under conditions of nutrient enrichment, *e.g.*, duckweed on the surface of standing water in a wetland receiving sewage. The actual effects on plant growth of any nutrient in excess will also depend on the other factors that control plant production. For example, if light or a required nutrient is not present in sufficient quantity, the increased growth of plants, which is expected to follow from an excess of the nutrient may not be observed. In addition, the spatial pattern of excess plant growth follows from the geometry of the system that determines the availability of light. For example, in shallow aquatic systems, light strikes the surface of the water first. If the water is torpid, a surface dwelling plant such as duckweed may proliferate and shade out all plants that might grow below the surface. If the water is in motion breaking-up surface films, light next extends into the water column supporting the growth of phytoplankton, which in turn can attain such high densities that rooted aquatic plants are cut off from the light and decline. The long term effect of excess plant production is the accumulation of organic matter on the bottom and the overgrowth of the benthic animals that are best adapted to process this excess organic material. Prolonged nutrient enrichment or extremely high nutrient loads cause systemic problems for ecosystems as a result of the overproduction of organic matter and a subsequent increase in oxygen consumption by microbes. The conditions that accompany the hypoxia and then anoxia that follows can make usually beneficial nutrients, such as ammonia, toxic. Even though a range of sediment oxidation levels is expected based on the different ability of oxygenated water to penetrate sediments of different grain size, increased organic matter supply to the sediments increases the utilization of oxygen by microorganisms there and lowers the oxygen penetration expected for any particular grain-size, which results in the loss of organisms and the decay of sediment structure. A definitive determination of cultural eutrophication can be difficult, because reference systems are often rare or nonexistent.

Expected scale of stressor induced effects: The location and intensity of cultural eutrophication on the landscape is almost entirely due to the form, extent and intensity of human activities. The space time boundaries for establishing a successful TMDL or other regulatory program can be determined by examining the distribution of processes that are concentrating and distributing nutrients over an entire watershed connected to the open sea. Where external nutrient supplies from human activities account for more than a small fraction of the total nutrient supplied to an ecosystem, we can expect to observe the effects of nutrient enrichment. Internal stores of nutrients and organic matter in the sediments, which are the result of past loading patterns, will, in part, determine the scale and intensity of eutrophication in a water body.

Factors to consider in making a diagnosis: It is often difficult to decide if an ecosystem is impaired by excess nutrient inputs, because nutrients are ubiquitous constituents of living systems naturally present in varying amounts that lead to different structural and functional states. Extreme cases of nutrient loading in aquatic environments can result in chronically low dissolved oxygen concentrations and the complete deterioration of the conditions necessary to support aerobic forms of life. Even though areas of hypoxia and anoxia develop naturally in some ecosystems; increasing the intensity, extent or frequency of hypoxic or anoxic conditions in an area or finding these conditions where they are not expected, are events often associated with human alteration of the environment. The effects of carbon loading on aquatic environments are often apparent in the increased organic matter found in sediments of a given grain size; however, it is more difficult to determine with certainty the origin of the carbon increase. Often a chain of intermediate results separate nutrient loading from an observed effect, such as a low oxygen event or a change in benthic species distributions. In making a diagnosis that nutrients are the cause of any given impairment, the investigator should ascertain that the impact pathway is as completely documented as possible, using the weight of evidence to decide if nutrients are the most probable cause of an observed effect.

3.1.2 Toxic Chemicals

Definition: Toxic chemicals are defined as compounds or elements that elicit a dose-dependant toxic response in a biological system (Rand and Petrocelli 1985, Manahan 1989). The affected biological system may range over organizational scales from the sub-cellular to the ecosystem. Toxic response endpoints range from necrosis of cells, through mortality of individual organisms, to significant changes in community composition. Toxic chemicals may be anthropogenic (*e.g.*, polychorinated biphenyls or PCBs, dioxins, pesticides) or natural (copper, cadmium, nickel, hydrogen sulfide, ammonia) in origin. Some toxic chemicals like the metals copper, nickel and lead are naturally-occurring but their concentrations in the environment are greatly elevated by human activity, *e.g.*, mining (O'Neill 1985).

Natural occurrence: Many potentially toxic chemicals occur naturally in aquatic environments and are only toxic when their concentration is increased beyond a threshold. For example, all metals have a geological source, while organic compounds such as polycyclic aromatic hydrocarbons (PAHs) are produced both through natural and anthropogenic processes (O'Neill 1985, Burgess et al. 2003). In nature, these chemicals are not often found at high enough concentrations to be toxic. Exceptions include localized oil seeps or easily eroded mineral deposits. However, under these circumstances, it is likely that local ecosystems have adapted over thousands of years to the presence of these otherwise toxic chemicals, just as populations of organisms have been shown to adapt to more recent contamination (Nacci et al. 1999, 2002). Some toxicants such as copper or ammonia are required by living systems at low concentrations (O'Neill 1985). However, these same chemicals at elevated concentrations can cause toxicity. As *Paracelsus* (1493-1541) said, "therapeutic and toxic properties often differ by just the dose" (Doull et al., 1980). Toxic chemicals can not be easily grouped into one large stressor category, because of their common natural occurrence and their ability to be both therapeutic and toxic depending on dose; therefore, they are best considered one at a time or in functionally similar groups or subcategories (Doull et al. 1980).

Changes caused by human activities: Human activities have two major effects on the occurrence of toxic chemicals in the environment: disruption of the natural cycles of some toxic chemicals and synthesis of 'new' toxic substances. First, a chief characteristic of the human economic and cultural use of natural resources is to extract raw materials and in so doing disrupt the natural material cycles. Concomitant waste production and disposal practices often result in the addition of elevated concentrations of many elements (*e.g.*, Cu, Pb, Hg) and compounds (*e.g.*, ammonia, hydrogen sulfide) in the environment. Furthermore, human activities cause the distribution of these materials into patterns that differ from those established by natural biogeochemical processes. Often, ecosystems that receive concentrated doses of toxic chemicals (*e.g.*, mine tailings, manufacturing waste) have not adapted to effectively process them resulting in potentially severe disruption of ecosystem structure and function. Second, over the last 100 years human activities, especially those related to the chemical and pharmaceutical industries, have created many new synthetic toxicants (Colborn et al. 1996, Carson, 1962). Some of these synthetic toxic chemicals have been designed purposefully to be toxic (*e.g.*, herbicides, pesticides) while others were designed for other uses (*e.g.*, PCBs as electrical component insulators) but have had environmentally deleterious effects (Ho et al. 1997, West et al. 2001).

Mechanisms of stressor action: Toxic chemicals may have many adverse effects that result from a variety of mechanisms; therefore, it is difficult to develop a simple scheme for accurately categorizing their impacts. The most commonly measured effect in aquatic toxicology is mortality; in addition, sublethal effects include reproductive changes, alteration of growth, competition and behavior, *e.g.*, being an efficient predator or an elusive prey (Rand and Petrocelli 1985). Beyond effects on individual organisms, ecological effects include changes in communities such as alterations of diversity, biomass, functional processes, and species composition and in ecosystem properties such as the health and integrity of systems and changes in spatial (habitat heterogeneity) and temporal (nutrient regeneration rates) dynamics (Rand and Petrocelli 1985). Current research at NHEERL's Atlantic Ecology Division is focused on extrapolating from the individual to population, community and ecosystem endpoints (Kuhn et al. 2000, 2002).

Broad classes of mechanisms of toxic action that produce stressful effects include: narcosis a reversible effect that reduces metabolic activity (van Wezel and Opperhuizen 1995), alteration of chemicals to more toxic forms via the mixed function oxidases (MFO) enzyme system (Doull et al., 1980, Coakley et al., 2001), exceeding the detoxification capacity of metallothionien proteins (Kagi and Schaffer, 1988), and the broad class of mechanisms that promote and produce cancer (Ashby and Tennant, 1988, Lijinsky, 1989). Regardless of the mechanism, toxic chemicals damage the metabolic structure and function of the organism. All organisms have repair mechanisms; however, if adverse effects are manifested, the repair mechanisms have been overwhelmed.

Expected scale of stressor induced effects: Anthropogenic activities have distributed toxic chemicals globally (LaFlamme and Hites 1978, Valette-Silver 1993, Muir et al. 2000); however, because most toxic chemical discharges are concentrated at point sources, toxic chemicals most commonly manifest adverse effects radiating out from the point of discharge within an individual water body. The majority of toxic sediments are found in close proximity to industrial, municipal or agricultural sources (Long 1992, Daskalakis and O'Connor 1995, Long et al. 1996); however, one should not rule out widely-distributed areas of contamination simply because concentrations are not extremely elevated or areas of elevated contamination not directly associated with obvious sources (*e.g.*, illegal disposal of hazardous wastes). Examples of toxic pollutants that have wide distributions include mercury released into the atmosphere by coal burning power plants (Renner 2002) and the atmospheric transport and deposition of PCBs that through trophic transfer and biomagnification appear in high concentrations in the fat and milk of high-latitude mammals including indigenous peoples (Muir et al. 2000, Swain, 1988).

Factors to consider in making a diagnosis: Extensive data are available on the toxicity of many classes of chemicals and our understanding of the toxic mechanisms of many chemicals allows prediction of their bioavailability and effects. Toxicity testing is a valuable tool but to date it has focused on organisms as endpoints, while many of the effects considered in developing a TMDL include impacts on populations, communities and ecosystems. Also, traditional toxicity testing and endpoints may lack sensitivity to certain toxic chemicals that bioaccumulate (*e.g.*, dioxins, mercury). In this case, tissue residues are a better measure of the stressor's potential effects. Instrumental analysis of environmental samples for toxic chemical concentrations represents a viable secondary approach for determining the exposure of organisms to these chemicals. However, such analyses can be prohibitively expensive, are not capable of detecting all toxic chemicals, and may erroneously over- or underestimate the bioavailability of target toxic chemicals. Linking toxicity testing and instrumental analysis in a TIE approach captures the strengths of both methods.

3.1.3 Suspended and Bedded Sediments

Definition: Suspended and bedded sediments act as a stressor in the environment when they have negative effects on organisms and /or habitats due to their presence in excess quantity, *i.e.*, they are present in quantities that the ecosystem is not adapted to process. Suspended and bedded sediments fall into two broad categories: suspended sediments and bedload. Suspended sediments are those particles that are suspended in the water at any given time because the turbulent energy of the water is sufficient to prevent them from settling to the bottom under the influence of gravity. Bedload describes those particles moving through the aquatic system via sliding, rolling or saltating on or near the bed (channel, or lake bottom). Whether a particle is moved as suspended sediment or bedload depends on the complex interactions of particle size and shear stress, with shear stress being a function of the interaction of channel slope, gravity, bottom roughness, and current speed. The direct measurement of bedload transport is difficult, therefore patterns of mud, sand, gravel, and current speed are commonly used to estimate its magnitude and occurrence.

Natural occurrence: The most common source of sediment entering streams is the weathering (by wind and precipitation) of parent material in the catchment, which is subsequently moved down slope by gravity into the receiving waters. This process may be accelerated by natural disturbances (*e.g.* storm events) within the catchment. Thus, sediment in the receiving waters is an expected result of catchment evolution and weathering processes. Background concentrations of sediments in receiving waters are variable, depending on the age, geologic composition and history of the catchment. The balance of erosion and deposition resulting from the interactions of physical forces and conditions changes the nature of subsequent interactions, so that sediment features on the bottom grow and decline over time. The quality and quantity of sediment movements is closely tied to natural episodic events such as floods, tsunamis, and hurricanes.

Changes caused by human activities: Anthropogenic activities have greatly accelerated the rate of sediment movement from catchments to receiving waters because of land use practices that increase erosion and runoff. In general, the conversion of natural lands within a catchment to human uses will result in increased sediment transport and deposition in the receiving waters of the systems. Specifically, row crop agriculture, livestock grazing, forestry, mining, and urban development have all been linked to increases in sediments in streams, lakes, reservoirs, wetlands and estuaries.

Mechanisms of stressor action: Suspended and bedload sediments have two major avenues of action in aquatic systems: 1) direct effects on biota and 2) direct effects on physical habitat, which results in indirect effects on biota. Examples of direct effects on biota include suppression of photosynthesis by shading primary producers; increased drifting of, and predation on, benthic invertebrates; and shifts to turbidity-tolerant fish communities. Indirect effects on biota will occur as the biotic assemblages that rely upon aquatic habitat for reproduction, feeding, and cover are adversely affected by habitat loss or degradation.

Expected scale of stressor induced effects: The effects of suspended and bedload sediments span the scales of biological organization from individuals to ecosystems. The biological responses to this stressor at a site are related to site-specific effects (turbidity shading, substrate embeddedness) and to the cumulative loadings of sediments from the catchment above the site. In addition, the cumulative effects of these biological responses (failed reproduction or reduced habitat) at sites are additive over the entire catchment, so that catchment-wide stressor impacts are possible based on the cumulative nature of the stressor.

Factors to consider in making a diagnosis: The stress of excess suspended and bedded sediments on receiving systems rarely acts alone. Other stressors associated with accelerated clean sediment loadings are increased temperature, which usually results from riparian canopy losses, and increased nutrients inflows from clear cutting and agriculture in the catchment. Often, these various stressors are the result of the same anthropogenic activities, *e.g.*, land conversion and human disturbance.

3.1.4 Habitat

Definition: Habitat describes the environment in which particular organisms reside and includes both physical attributes (*e.g.*, structure, temperature, hydrological regime) and chemical attributes (*e.g.*, dissolved oxygen, salinity). The physical structure of habitat includes both abiotic components (*e.g.*, geologic substrate, temperature, oxygen, *etc.*) and biotic components (*e.g.*, vegetative structure, oyster shells, worm tubes, *etc.*) and thus its extent can be measured in two or three dimensions (area or volume, respectively) and its quality ascertained by a wide variety of measures. The scale and components of habitat must be defined in relation to a particular population or group of populations. For the purposes of illustration, habitat will be defined with respect to fish populations in the conceptual model discussed here.

The words “habitat” and “suitable habitat” are often used interchangeably, giving rise to the concept that habitat for a population must encompass a minimum set of conditions that are associated with a finite probability of survival. Habitat can be examined at a range of scales, from micro-habitats (at the scale of the organism) to habitat patches (homogeneous units related to populations) to habitat mosaics (a collection of habitat patches, often dispersed in a background matrix of unsuitable habitat). Habitat quality, or the probability that habitat will support a self-sustaining population, can also be defined at each of these scales. Factors affecting temporal and spatial scales of habitat variability are described in Figure 11.

Natural occurrence: In general, aquatic habitat is of natural origin, although it is possible to create artificial habitat (*e.g.*, artificial reefs, sewage treatment systems). Natural regimes (spatial and temporal patterns) exist for temperature, water, geologic substrate, biological structure, dissolved oxygen, and salinity, and these differ over ecosystem types and geographic regions. The hierarchical structure of biological and ecological organization must be considered to understand the role of natural habitat in the survival of individuals and populations. If the set of energy inputs to an ecosystem provides the available energy to organize its abiotic and biotic structures and functions upon which habitat depends; habitats, whether natural or produced by human activities, may be effectively distinguished by the incoming emergy signature (see text box C) of the place.

Changes caused by human activities: Both the quality and quantity of natural habitat can be altered by human activity. Potential changes include 1) shifts in the spatial and temporal patterns of water level regimes or flow, temperature and dissolved oxygen fluctuations, or gradients of salinity, 2) changes in the physical structure of habitat, both biotic (vegetation) and abiotic (substrate particle size, organic matter content), 3) loss of area of suitable habitat, and 4) changes in the spatial configuration of suitable habitats on the landscape. In addition, human activities are changing the natural emergy signatures of many places on earth, and indeed of the earth as a whole (Brown and Ulgiati 1999). Many new aquatic habitats are being created by altering the natural energy and material flows and storage units and by creating new materials and discharging them to the environment in sufficient quantities to change the character of a place, and thereby the unique suite of species that the ecosystem can support there. For example, Odum et al. (1974) mention emerging new systems associated with pulp mill wastes, thermal pollution, sugar cane wastes, oil shores, brine pollution, *etc*.

Mechanisms of stressor action: We will distinguish between attributes of habitat that can act as stressors on individual organisms (temperature, hydrologic regime, dissolved oxygen, salinity) and those attributes required for physical structure. The physical structure of habitat can provide a substrate for attachment, substrates for growth of primary producers or prey items, shelter from disturbance such as wave action, and/or a refuge from predators. As such, habitat structure can modify the availability of food resources or reduce population loss rates from predation or physical damage. Vegetative habitat can also modify the chemical characteristics of surrounding waters through photosynthesis or through limiting turbulence and exchange with the atmosphere or with flowing water.

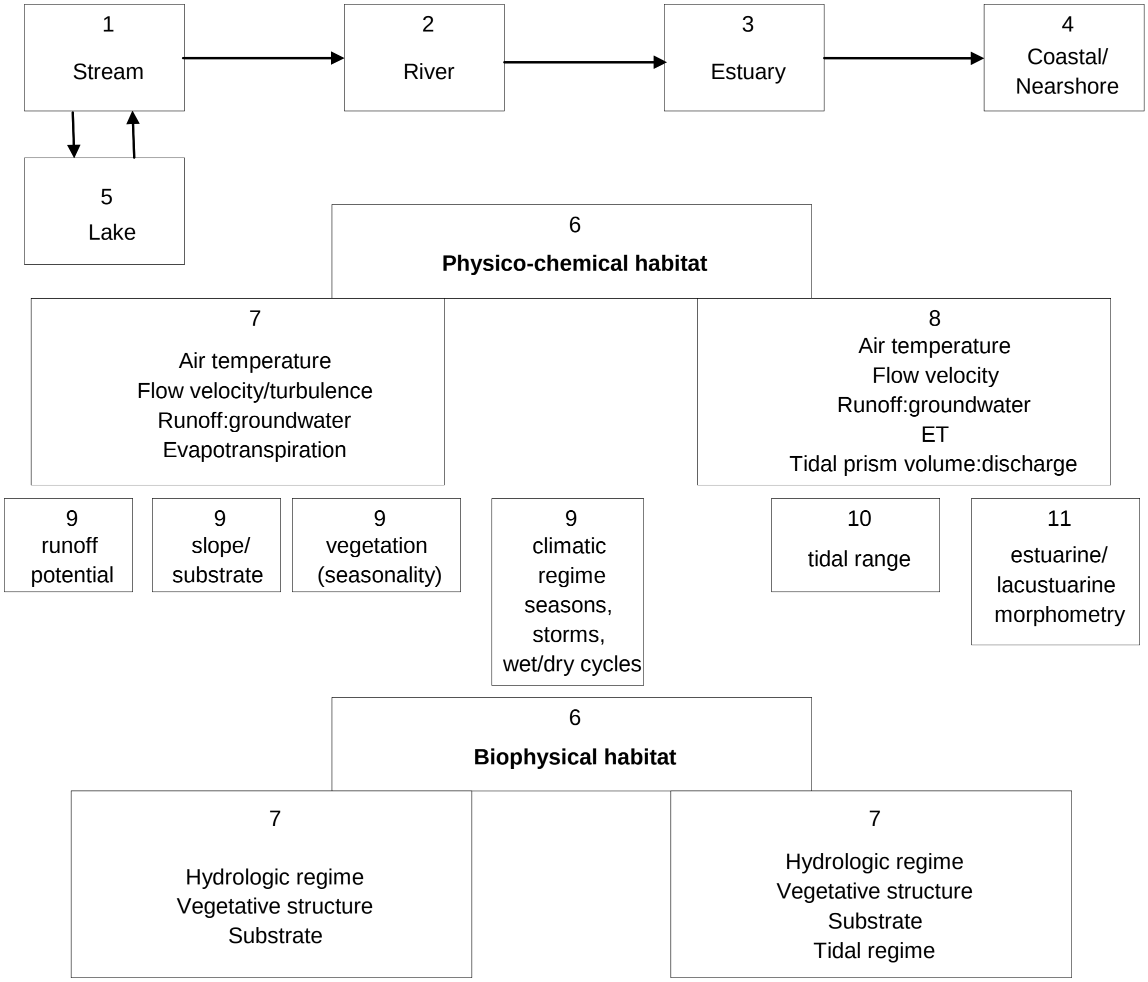


Figure 11. Summary diagram of the principle factors that control habitat of a species and its alteration.

Mechanisms of stressor action associated with temperatures outside of the metabolically-suitable tolerance range of a species include direct mortality, altered growth, reproductive changes and emigration or movement out of the formerly suitable habitat. Altered temperature regimes will be manifested directly as structural changes within a population and compositional changes within the fish community. Indirectly, altered temperatures can affect fish through changes in prey availability and changes in interspecies competition based on differences in optimum temperature or tolerance ranges.

Direct mechanisms of action associated with altered hydrologic regimes include physical disturbance affecting individuals (scouring, increased drift), and populations (desiccation, and loss of suitable instream habitat volume). Indirect mechanisms of effect associated with altered hydrologic regimes that apply to both individuals and populations include: 1) changes in vegetative structure based on selection for plants with different growth forms and life history strategies, 2) changes in sediment characteristics (*e.g.*, siltation, embeddedness) due to changes in flow, 3) changes in thermal regime related to the balance between groundwater and surface water inputs, 4) decreases in dissolved oxygen associated with stagnation and reduced turbulence, 5) changes in salinity gradients and stratification patterns in estuaries, and 6) changes in retention time and associated processing rates that affect the concentrations of nutrients and toxins. Tertiary effects include changes in competitive ability related to changing physical and chemical conditions, in some cases selecting for invasive species.

Mechanisms of stressor action associated with altered dissolved oxygen regimes include massive mortality, altered growth, and avoidance of areas with unsuitable habitat. As with other habitat stressors, effects will be observed at the individual, population, and community levels. Mechanisms of action associated with altered salinity gradients include loss or gain of suitable habitat with resulting effects on growth and survival, emigration or immigration resulting in changes in community structure and composition.

Expected scale of stressor-induced effects: Habitat quality can be affected at the scale of organisms (microhabitat), at the scale of habitat patches, or at the scale of landscapes. The total area, spatial configuration and density of habitat patches can all affect the survivorship of populations. For example, the density of habitat patches could affect the rate of colonization and the number of populations experiencing periodic recruitment failures. Spatial configuration of habitat patches can be particularly important where different habitat types are critical for different life stages of an organism. The scale to be considered depends on the range of the species of interest, both within a year, and between different life history stages.

Factors to consider in making a diagnosis: A reference condition for habitat structure, habitat extent, and the distribution of habitat types must be established to determine whether changes have occurred. Historic changes in habitat type, area, and distribution can be measured or inferred through use of mapped inventories (*e.g.*, National Wetlands Inventory, http://www.nwi.fws.gov/), aerial photographs, or through the use of indicators such as the association of hydric soil complexes with different wetland vegetation types. Information on the historic range and biogeographic constraints for individual species may help distinguish between the effects of habitat loss and change in the quality of habitat. The effects of changes in habitat area can be inferred through development of empirical species-area curves, or production-area relationships. Responses to loss of suitable habitat area are more likely to be a function of organism requirements than of habitat features. The effects of changes in the mosaic of habitat structure can be examined through pattern descriptors such as patch density or diversity, patch cohesion, dispersion, and perimeter: area ratios, and the development of relationships between these measures and biotic endpoints. In addition, the effect of changes in the mosaic structure of habitats can be simulated through use of spatially-explicit population models.

Species are adapted to utilize variations in naturally occurring ecological conditions (niches), such as different thermal, dissolved oxygen, and hydrologic regimes. The presence and absence of species often can be directly related to their requirements for a specific habitat. For example, guilds of fish species adapted to different physical regimes have been identified based on commonalities in physiological adaptations, behavior, or life history traits, and they can be used to infer that changes in these parameters are associated with community-level shifts. For temperature and dissolved oxygen, limits associated with mortality have been determined in laboratory tests, while sublethal effects may be predicted through bioenergetics models. More accurate limits on fish species distributions in streams can be derived through development of empirical associations between parameters that describe properties of these regimes (*e.g.*, 7-day low flow, 21-day average maximum temperature) and fish presence/absence.

3.1.5 Special Characteristics of Aquatic Ecosystems Applicable to all Stressors

Several aspects of aquatic ecosystems are particularly relevant to understanding habitat alteration, but they also apply to the other three stressors and should be considered in developing and analyzing conceptual models of stressor action. These general factors will be discussed in this section with reference to habitat alteration, because this stressor is the most complex. Similar considerations might be applied to the other three stressors (toxic chemicals, nutrients, and suspended and bedded sediments). The general characteristics considered here are as follows: 1) Biological components, stressors, and modifying factors are all linked in a single interacting network. 2) Spatial and temporal variation of stressor actions and effects is common, *i.e.*, stressor interactions and effects are dynamic in space and time. In addition, the subject, classification and assessment of habitats, is briefly considered in this section.

Factors modifying altered habitat are linked: Many variables determining suitable habitat are linked and, thus cannot be examined in isolation. Each of the factors that alter habitat can be modified by the state of the other modifying factors. Some examples of such interactions follow: a) increased heat inflow to an estuary will cause a larger or smaller change in the average water temperature depending on water flow and vertical mixing in the estuary; b) the response of an organism to a habitat variable can also change in the presence of a pollutant, which can be considered as a modifying factor on habitat (for example, increased sedimentation in a stream ecosystem can alter the response of the biota to changes in flow regime); c) shifts in the spatial and temporal patterns of water level, regimes or flow, temperature, and dissolved oxygen fluctuations, or gradients of salinity can each affect the biological consequences of a change in one or more of the others; and d) the structure of habitat itself might alter the biological effects of a change in one of the modifying factors as discussed below.

Spatial and temporal variations in habitat quality: Over short time periods at the micro-habitat scale, the effects of shifts in the mean or the temporal patterns of water level, flow, temperature, dissolved oxygen, or salinity on organisms should depend only on the mean and variance of conditions in the system without impacts, *i.e.*, those conditions to which resident organisms are adapted. In this case, the micro-habitat scale is defined relative to the spatial grain or heterogeneity in those environmental parameters defining suitable versus unsuitable conditions (leading to death or to a net loss of energy). The temporal scale (“short” time period) is defined relative to the time required for an organism to move from an unsuitable habitat to a suitable habitat.

Over longer time periods and at spatial scales consistent with the range of an organism, organism sensitivity to changes in the mean or in the spatiotemporal variation of environmental parameters will depend on the spatial or temporal distribution of refugia (*i.e*., suitable habitat). An example of refuge from scouring flows would be the hyporheic zone or backwater habitat in a stream. Gradients in the physical structure of habitat, both biotic (vegetation) and abiotic (substrate particle size, organic matter content), organize habitats in space. In general, the spatial and temporal variability of habitat under normal disturbance regimes will determine the sensitivity of populations within the habitat to change. For example, some ecosystems naturally go through stages of succession, or periodic shifts in vegetation structure related to wet and dry periods (van der Valk 1981). These ecosystems are more likely to have seed banks for vegetation adapted to different phases of the wet-dry cycle, resting stages of other organisms (e.g., ephipia) adapted to changing conditions, and/or to a relatively high proportion of colonizing species, which allow populations to recover following periods of stress.

The effects of spatial and temporal variability in habitat quality on landscape-scale population dynamics have been generalized by Turner et al. (1993; see Figure 2). Turner and colleagues simulated a terrestrial landscape with 8 seral stages of succession based on the assumption that seed sources would not be limiting and captured the results in a state-space diagram. Landscapes can be compared by scaling the disturbance interval to the recovery time, and scaling disturbance extent to landscape extent. To judge the general effects of spatial and temporal variability in habitat on animal population stability, this type of simulation would be repeated and the results summarized with disturbance interval scaled to the time to recolonize patches and disturbance extent scaled to organism’s home range on the landscape. Turner’s simulations were based on the spatial and temporal dynamics associated with terrestrial succession. In an aquatic landscape, the concept could be extended to cover dynamics of daily migrations (between suitable and unsuitable microhabitat based on diurnal variations), movement over a breeding season (annual migrations), and movement between breeding seasons. An upper limit to regional population viability could be expressed as a function of disturbance extent relative to critical dispersal distance and disturbance interval relative to organism longevity or length of viability of resting stages (*e.g*, egg bank) where appropriate (See Turner et al. 1993, Figure 3).

Meta-population models or spatially-explicit population models can be used to predict the sensitivity of different systems to changes in the spatial configuration of suitable habitats on the landscape. Sensitivity to change will depend on organism life history traits and the ability to migrate across the landscape as well as the density of suitable habitat patches (Gibbs 1993). Island biogeography theory (MacArthur and Wilson 1963) suggests that recovery of populations will occur more rapidly when suitable habitat patches are less fragmented or dispersed (Gibbs 1993). Due to a lack of long-term data, this hypothesis has been tested for relatively few aquatic systems (Niemi et al. 1990, Detenbeck et al. 1992). Recovery time for aquatic populations depends on the distance of refugia from the point of other stressor impacts (Detenbeck et al. 1992). Simulation models have been developed that can mimic the movement of organisms across landscapes or potentially across riparian zones (Gardner et al. 1992). These models predict that a critical change in landscape structure occurs as the ratio of inner to outer edges (or fractal dimension of a landscape) changes. In general, the predictions of these models have not been verified against actual data.

Classification and Habitat Assessment: Existing habitat classification schemes such as Cowardin's scheme for wetlands and deepwater habitat tend to categorize habitat into patches of similar environmental conditions and substrate type, and thus would not necessarily predict sensitivity to a change in environmental variables (Cowardin et al. 1979). Classification schemes or parts of classification schemes describing the normal or reference range of variation in abiotic environmental variables and biotic structure (vegetation) might provide a framework for predicting sensitivity to change in habitat quality. Examples of such classification schemes include use of the hydrology modifier in Cowardin's classification scheme for wetlands or Poff's concept of flow regimes for streams (Poff and Ward 1989). For lakes, temporal and spatial variability of dissolved oxygen and temperature regimes has been described as a function of lake morphometry (surface area, maximum depth) and trophic state (Stefan et al. 1995, 1996).

Classification schemes describing variation in the hydrologic regime are likely to explain the relative sensitivity of different aquatic ecosystems to changes in the biotic component of physical habitat, whether natural or unnatural. For systems in which gradients of physical and chemical conditions exist (*e.g.*, estuaries), the shift in these gradients relative to suitable biophysical habitat such as emergent vegetation will determine sensitivity to change (Sklar and Browder 1989). Thus overall sensitivity will depend both on the steepness of biophysical gradients and estuarine morphometry; more gently sloping systems will probably have a broader band of suitable habitat types such as emergent vegetation. In general, spatial variability will be a function of source characteristics, topography or bathymetry, and mixing factors (surface area/depth, salinity ratios).

3.2 Generic Conceptual Models of the Four Classes of Aquatic Stressors

This section presents generic models for the four classes of stressors; however, these stressors do not exist within aquatic ecosystem alone, but in the company of one or more of the other stressors, which may affect the action of the stressor being modeled. For example, suspended and bedded sediments and nutrients are natural components of all aquatic ecosystems and will co-occur with each other and with toxic materials such as anthropogenic organic chemicals. To meet this challenge, an energy system perspective (Odum 1994) was used in developing conceptual models of the individual stressors. Therefore, each stressor was represented with the complex of factors that determine its actions within the ecosystem, including factors that might be considered stressors in their own right. Thus, the detailed conceptual model for a single stressor includes information on other stressors, where they are important modifiers on the actions of the stressor under study. In this manner, both single and multiple stressor problems will be addressed with a unified approach. In cases where additional stressors are important, the single stressor models may be modified or coupled with other single stressor models to examine joint effects.

Modeling each stressor within the context of the larger aquatic ecosystem is important, because the result of performing a PIE or other method of causal analysis in marine ecosystems often presents us with situations where more than one stressor is shown to be actively causing impairment to the biota. In this situation, we need models that incorporate accurate information on causal mechanisms to allocate observed effects among multiple stressors. Sensitivity analyses of ecosystem simulation models can be used to set priorities for mitigating a stressor by determining the pollutant to which the ecosystem output variables (measurement endpoints) are most sensitive. In addition, these models can serve other useful purposes, such as predicting the threshold for an effect and determining joint or bundled criteria for combinations of stressors interacting within an ecosystem.

The first step in producing pollutant-specific, quantitative models for eventual conversion into computer simulation models is the construction of a conceptual model of stressor action. This process is the means for putting our mental models of stressor action into a concrete form. In this section, the narrative descriptions of the major classes of stressors given in Section 3.1 are used to create ESL diagrams showing stressor action within the context of an ecosystem. The starting point for developing more complex models for the individual stressors is to apply the narrative descriptions within the appropriate canonical model described in Section 2.2.2. Once created, a detailed conceptual model can be written as a set of simultaneous differential equations, and then translated into finite difference equations and programmed for computer simulation. The outcome of these simulations is the prediction of stressor behavior in the ecosystem These models are specific for an individual pollutant, so the processing pathways and modifying factors may be somewhat different for each material and ecosystem type; however, the water flows governing residence time will be similar within each of the canonical models.

3.2.1 A Generic Energy Systems Model for Habitat Alteration

The mechanisms of habitat alteration are somewhat different from and more complex than the other three stressors considered in this paper. For this reason and because habitat alteration has just been discussed, the generic energy systems model of habitat is examined first. Nutrients, suspended and bedded sediments, and toxic substances are all materials that act as pollutants, when excess amounts are added to an ecosystem. Habitat alteration can be caused by a pollutant, such as suspended and bedded sediments or a toxicant, but it can also result from a physical or biological change in the environment. For example, diminished water flow, temperature change, and oxygen depletion are physical changes in a water body that may affect the suitable habitat available to a particular species. Habitat is always defined in relation to the needs of a particular species or group of species. The biological space, or conditions (habitat) available to support that species is also a function of its competitors and predators, thus, the introduction or invasion of a new species may alter the habitat available to established species. In general, the many factors that alter habitat can be viewed as modifying factors that affect the actions of a particular pollutant in the ecosystem.

Studies of habitat change are customarily focused at the population level of biological organization and are usually performed with the enhancement or preservation of a particular species in mind. The health of a particular population will depend on the existence of habitat of sufficient quality to support growth and reproduction. From an energy systems perspective, analyses that focus only on the habitat needs of a single population or species are necessary, but may not be sufficient to answer questions related to the long term health and survival of that population. This is true because the ecological requirements of species are interconnected and knowledge of the effects of habitat alteration on one population or species may not reflect the cumulative effects of habitat change in the whole system. For these reasons single species studies are not sufficient to answer the important questions about the consequences of broad scale habitat alteration that confront society. A better understanding of the ultimate fate of an individual species or population can be found by examining change in the unique emergy signatures supporting the various required habitats for that species and its coevolved prey, predators, and competetors. The emergy signatures of habitats within the system are determined by changes in the system forcing functions, which in turn are driven by the dynamics of the next larger system. Therefore, analysis at one level of organization in a hierarchical system is seldom able to achieve an understanding of the ultimate causes and remedies for phenomena observed at that scale of organization. In this model, we consider the effects of habitat alteration within an ecosystem context, viewing multiple levels of organization and using the principles of energy systems theory to trace causality.

3.2.1.1 A Comprehensive Measure of Impairment

The word “habitat” refers to the area and type of environment in which a particular organism or population normally lives. Therefore, the concept of habitat is tied to a space (area or volume) within an ecosystem. The emergy that enters this area through its forcing functions determines the ecological organization that can develop there, and may be an integrative measure of habitat quality for the species that can thrive under the prevailing conditions. The change in the empower density developed by a particular species within an ecosystem that results from a change in environmental forcing may be an integrative measure for evaluating the overall effects of habitat change on that species. Establishing that a habitat has been altered is a prerequisite to determining the cause of that alteration. The degree of habitat alternation can be assessed by measuring the empower flowing through the habitat area including the species of interest and comparing it to the empower of a reference state. Empower of the habitat is an integrated measure of ecological functioning that gives an estimate of the overall effects of alteration on the species of concern and on the ecosystem as a whole. Areas of the landscape are or were suitable habitat because they receive or have received in the past a suite of forcing functions (an *emergy signature*) that establishes ranges of variables and/or a suite of energy flows within the area (niches) which are suitable for the survival and reproduction of the various species that are found there. Over greater or lesser periods of time, the operation of the same or different external forcing functions have been responsible for creating the stored emergy in structures, *e.g.*, vegetation, stream morphology, etc., found in the area and necessary for the persistence of a particular species. Habitat change occurs when the emergy signature changes as a result of changes in anthropogenic or natural forces impinging on the area causing a loss of structural components (*e.g.* species) or an alteration of processes and thus a change in empower of the system.

3.2.1.2 Model Description

An energy systems model to examine the effects of habitat change was constructed (Figure 12), beginning with the narrative description of altered habitat as a stressor and the summary diagram presented in Figure 11. All forcing functions, components and pathways in Figure 12 are defined in Table 4. The boundaries of the system to be evaluated are shown as a large box that contains the ecosystem components, including the population or populations utilizing the habitat. The choice of the system boundary establishes the spatial scale (*e.g.*, an estuary or stream reach) that is needed to examine the important factors controlling the growth and survival of a species or group of species. The choice of boundaries also establishes the geometry of the system (i.e., average depth, volume, etc.) and the habitats present within it. Usually, *state variables* within the ecosystem are evaluated per unit area or on the basis of the system as a whole. We set the system boundaries in Fig.12 to correspond with those of a stream reach used by the states in 305 (b) assessments to illustrate the process of model development and to show how habitat alteration might affect an example species, Q_i, found within this system. This model assumes that the ecosystem contains habitat features (*e.g.*, plant structure, streamform, prey items) supporting species Q_i; however, these features can be changed as needed to represent the particular species and ecosystem and generalized to consider any number of species (i =1 to n) and the competitors and predators of that species, $C_{ij,}$ (i =1 to n, J = 1 to m). The processes and components of the stream ecosystem modeled here can be viewed as a generic representation of the classes of processes and

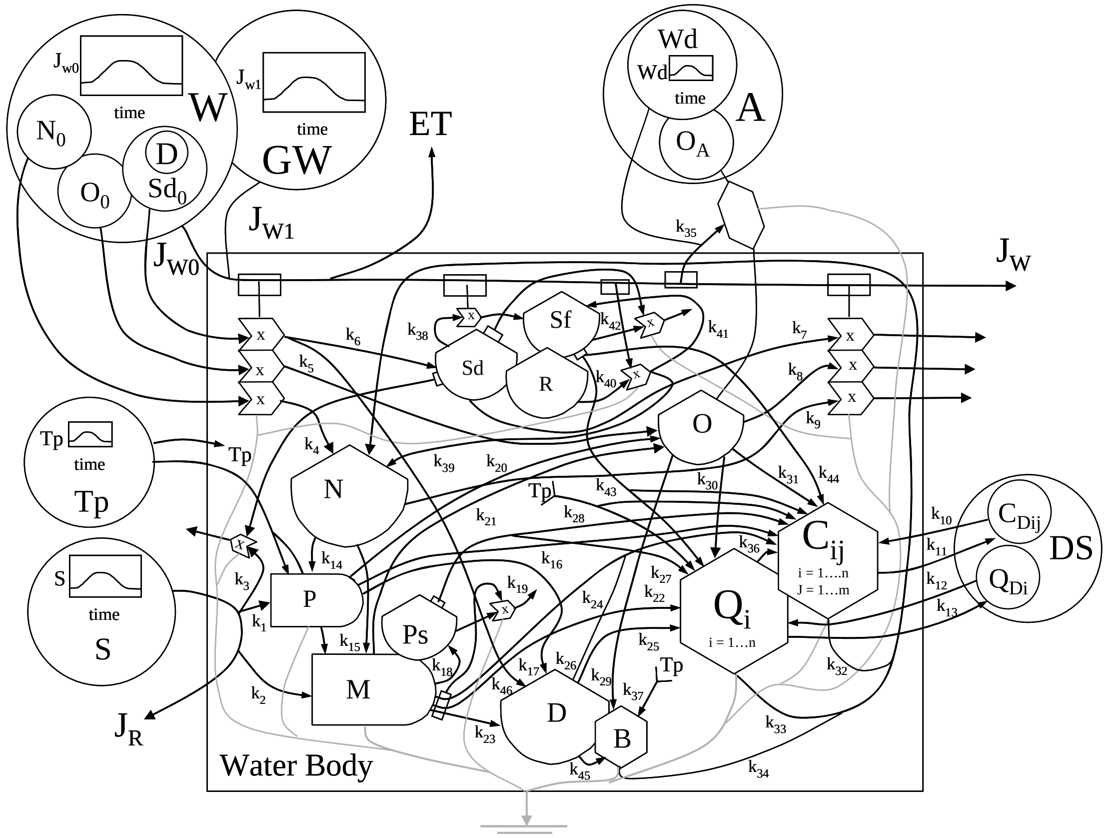


Figure 12. Energy systems model of the effects of habitat alteration on a species, Q_i, in an aquatic ecosystem. Forcing functions, components, and pathway flows are defined in Table 4.

components that we believe are important in determining the effects of habitat alteration in most aquatic ecosystems. For example, to convert this model for use in an estuary set it within the canonical module for bidirectional flow and add of subtract the appropriate state variables (*e.g.*, add salinity and subtract streamform) and their interactions.

Once the system and its boundaries are chosen, the next step in building the model is to specify the important external forcing functions. Forcing functions include the energy, material, and information sources that drive trends in the internal components or state variables of the system. In general, these external forcing functions are state variables of a larger system that can be understood through a diagrammatic representation of the larger system at its level of organization. The important forcing functions for the example (Figure 12) are solar radiation, S, temperature, T_P, runoff, J_{W0}, groundwater base flow, J_{W1}, wind, Wd, nutrient concentration in runoff, N_0, oxygen concentration in runoff, O_0, suspended mineral solids concentration in runoff, Sd_0, the suspended detritus (organic matter) in the runoff, D_0, the oxygen concentration at saturation in the overlying water, O_A, and animal immigration from and emigration to downstream ecosystems, DS. The external forcing functions comprise the energy and emergy signatures of the system; they are arranged around the outside of the box that delineates ecosystem boundaries and are shown from left to right in order of increasing transformity. In general, all the habitats used by a species during the stages of its life cycle should be included in the model, if the aggregate reproductive success of a species is of concern. Where critical habitats are widely separated in space or are of very different character, modeling and the determination of the energy flows supporting the species, Q_i, become more complicated. In such cases, a linked series of models should be used to evaluate the species of concern.

Table 4. Definition of the forcing functions, components, and pathways in the generic energy systems model to evaluate the effects of habitat alteration on aquatic ecosystems.

Symbol	Definition
Forcing Functions	
S	Solar insolation as a time series
J_R	Solar radiation that remains unused (albedo)
T_P	Temperature as a time series
W	Watershed (time series of water flow)
N_0	Nutrient concentration in the runoff (can be a time series)
O_0	Oxygen concentration in the runoff (can be a time series)
Sd_0	Sediment concentration in the runoff (can be a time series)
D_0	Detritus (organic matter) concentration in runoff (can be a time series)
GW	Groundwater Aquifer and its characteristics
J_{W0}	Water flowing in as runoff
J_{W1}	Groundwater base flow
ET	Water evapotranspired in the system
A	Atmosphere system
Wd	Wind as a time series
O_A	Concentration of oxygen in the air
DS	Downstream ecosystems
Components	
N	Nutrient
P	Phytoplankton
M	Macrophytes
Ps	Plant structure (critical to a species and used by competitors)
Sd	Sediment in the stream
Sf	Streamform (physical structure of the stream bed used by a species and/or its competitors)
R	Rock in the stream bed
O	Concentration of oxygen in the stream
D	Detritus on the stream bottom
B	Benthic bacteria
Q	A species for which the effects of habitat change is to be determined
C	Competitors and predators of Q, the species of interest.
Pathways and Flows	
J_W	Water flowing out of the system
k_1	Light used by phytoplankton (the producer symbol implies GPP, NPP respiration.)
k_2	Light used by macrophytes (the producer symbol implies GPP, NPP respiration.)
k_3	Light attenuated by turbidity in the water
k_4	Nutrient inflow in runoff
k_5	Oxygen inflowing in runoff.
k_6	Suspended particulate matter inflowing in runoff.
k_7	Suspended particulate matter flowing out in streamflow.
k_8	Oxygen flowing out in streamflow.
k_9	Nutrients flowing out in streamflow.

Symbol	Definition
k_{10}	Immigration of competitors and predators from downstream ecosystems
k_{11}	Emigration of competitors and predators to downstream ecosystems
k_{12}	Immigration of species Q from downstream ecosystems
k_{13}	Emigration of species Q to downstream ecosystems
k_{14}	Nutrient uptake by phytoplankton
k_{15}	Nutrient uptake by macrophytes
k_{16}	Phytoplankton eaten by competitors
k_{17}	Phytoplankton death to detritus
k_{18}	Contribution of macrophytes to plant structure
k_{19}	Plant structure lost as a result of macrophyte losses
k_{20}	Oxygen produced by phytoplankton
k_{21}	Oxygen produced by macrophytes
k_{22}	Macrophytes eaten by species, Q
k_{23}	Macrophyte death to detritus
k_{24}	Macrophytes eaten by competitors and predators
k_{25}	Detritus eaten by species Q
k_{26}	Detritus eaten by competitors
k_{27}	Plant structure effects on species Q
k_{28}	Plant structure effects on competitors
k_{29}	Oxygen used by benthic bacteria.
k_{30}	Oxygen used by species Q.
k_{31}	Oxygen used by competitors and predators.
k_{32}	Nutrients recycled by competitors and predators.
k_{33}	Nutrients recycled by species Q.
k_{34}	Nutrients recycled by benthic bacteria.
k_{35}	Exchange of oxygen with the atmosphere.
k_{36}	Species Q consumed by predators.
k_{37}	Temperature effects on bacterial respiration.
k_{38}	Sediment building streamform.
k_{39}	Nutrient dissolved from rocks.
k_{40}	Rock dissolved by the water currents.
k_{41}	Currents building streamform from rock.
k_{42}	Stream structure lost by sedimentation (burial).
k_{43}	Effects of streamform on survival and growth of species Q.
k_{44}	Effects of streamform on survival and growth of competitors.
k_{45}	Detritus consumed by bacteria
k_{46}	Detritus (organic matter) supplied from outside the system

After specifying the forcing functions, the internal system components are specified. The system components should include all the internal state variables and processes that are necessary for the survival and reproduction of the relevant population or populations of concern and their competitors and predators.

In Figure 12, the system components are substrate including rock, R, bedded sediments, Sd, streamform, Sf, water flow, J_W, vegetation as phytoplankton, P, and aquatic macrophytes, M, plant structure, Ps, oxygen, O, nutrient, N, detritus, D, and bacteria, B, the species of interest, Q_i; and the competitors and predators of the species of interest including all other important species in the ecosystem aggregated according to similar function, C_{ij}. All components are measured in appropriate units, usually mass or energy. Structural metrics like streamform and plant structure are expressed as measurable quantities or attributes that correspond to the properties of these variables that are important aspects of habitat for the species of concern and/or its competitors and predators.

In Figure 12, the network of interactions that controls the growth and survival of species, Q_i, is shown by the lines connecting components and processes. Each line represents a flow of energy, material or information and is identified by a pathway coefficient, k_i, as defined in Table 4. Briefly, the network shows runoff carrying in flows of nutrient, k_4, oxygen, k_5, and suspended sediment, k_6, according to the streamflow and the material concentrations that result from activities in the surrounding watershed. Concentrations of these same materials within the system are removed by the stream outflow, J_W, as fluxes of sediments, k_7, oxygen, k_8, and nutrients, k_9. In this model, suspended sediments, in addition to those entering in inflowing water, are assumed to be resuspended in proportion to the sediments accumulated on the bottom. The light attenuated by suspended sediments is shown on the pathway designated, k_3. The light absorbed by phytoplankton and aquatic macrophytes is designated by pathways, k_1 and k_2, respectively. The internal structure of the producers and consumers, *i.e.*, pathways of gross and net production and respiration, are included in the definition of the hierarchical producer and consumer symbols, but they are not shown explicitly in the model. Pathways k_{10} through k_{13} show the emigration and immigration of species Q_i and the species included in C_{ij} to and from downstream ecosystems. The uptake of nutrients by phytoplankton and aquatic macrophytes is shown by pathways k_{14} and k_{15}, respectively. Phytoplankton is consumed by competitors of species Q_i on pathway k_{16} and phytoplankton die and sink to the bottom on k_{17} forming detritus. Pathways k_{20} and k_{21} represent the oxygen produced by phytoplankton and aquatic macrophytes, respectively. The increase in plant structure occurs on pathway, k_{18}, and plant structure is lost as vegetation is consumed or dies to become detritus, according to the sum of pathways k_{22}, k_{23}, k_{24}. Detritus consumed by species Q_i and its competitors is shown on pathways, k_{25} and k_{26}, respectively. The role of plant structure in promoting growth and survival of species Q_i and its competitors is shown on pathways, k_{27} and k_{28}, respectively. The oxygen consumed by bacteria, species Q_i, and competitors and predators is given on pathways, k_{29}, k_{30}, and k_{31}, respectively. The nutrients recycled by the metabolism of the C_{ij}'s, Q_i, and B are shown on pathways, k_{32}, k_{33} and k_{34}. The exchange of oxygen with the atmosphere occurs along pathway, k_{35}, and is driven by the oxygen concentration gradient between air and water and the mixing energy supplied by wind and current. The amount of species Q_i consumed by its predators is shown on pathway k_{36} and the effects of temperature driving bacterial action is governed by the pathway coefficient, k_{37}. Sediments are worked into stream structure along pathway, k_{38}, and rock dissolves, k_{40}, to yield nutrients and form stream structure on pathways k_{39} and k_{41}, respectively. The structure of the stream is lost on pathway, k_{42}, as sediments accumulate on the bottom. The effects of stream-form providing habitat that promotes the growth and survival of species Q_i and the competitors and predators of species Q_i are represented by pathways, k_{43} and k_{44}, respectively. Competitors and predators might be separated in this model by adding state variables. Detritus is consumed by bacteria on pathway k_{45}. The biological oxygen demand added to the system in the runoff is represented by the flow on pathway k_{46}.

If Q_i is a particular species of fish whose habitat is controlled by temperature and water flow, this model could be considerably reduced in complexity. In this example, the effects of temperature on species, Q_i, and its competitors, C_{ij}, is explicitly included but the affects of flow regime are modulated through other factors such as streamform, plant structure, and oxygen concentration. If streamflow itself accounts for most of the control action, a much simpler model could be built and plugged into the canonical model of unidirectional flow (Fig. 9a). In this case, the effects of streamform, plant structure and oxygen would be aggregated into the single variable streamflow and the simpler model evaluated against existing data.

The generic energy systems models that we have developed in this paper are intended to capture the most complexity that will be needed to describe the action of a stressor in the majority of cases. In this way, these models can serve as a checklist for the completeness of an analysis and as a guide to simplification through functional aggregation. We envision that this generic model might be used to guide analysis of the effects of altered habitat in any aquatic ecosystem. Any modifications required to address a particular question or system can be accomplished using the methods and overview models presented above and the expert knowledge of local scientists about the particular system to be evaluated. An example of an evaluated energy systems model for an estuary can be found in Campbell (2005).

3.2.1.3 Evaluating Temporal Aspects of Habitat Alteration

Species may have requirements for special conditions in time as well as in space. The temporal dimension of an energy systems model is described by the time series of values used to specify the forcing functions. For example, the complete definition of a forcing function entering a system includes a time series of values whose length determines the maximum time of the simulation. The time series of a forcing variable may have different frequencies or pseudo-frequencies of oscillation. The highest frequency of oscillation that can be investigated in the model will be determined by the measurement interval of the observations. A change in the frequency of forcing events, *e.g.*, the frequency of floods of a given magnitude, may constitute a change in the suitable habitat for species whose life cycles are interrupted by the alteration of inundation regimes. In general, the dynamic properties of a system in time can be most easily investigated using model simulations validated with temporal data or with data gathered in space for time substitutions. The relationship between the frequency of a disturbance and the time needed to recover from that disturbance has been proposed as a sensitive indicator of the risk to a given species from sporadic or repeated habitat change (Turner et al. 1993). Simulation models are ideal tools for investigating the temporal dimension of habitat change.

3.2.1.4 Examples of Habitat Change

The critical parameters of a habitat may be altered by: 1) A qualitative change in the emergy signature of the place caused by the addition or removal of an emergy source. For example, a battery factory built on the edge of a small estuary or wetland suddenly adds the pollutants Pb, Cd, Zn, and Ni to the system (Odum et al. 2000), or a hurricane or large storm moves enough sediment to close the breach way to a coastal pond, thereby removing the tidal forcing supplied from the sea as seen in some of Rhode Island’s coastal ponds. The stored structures, *e.g.*, the salt content of the waters, sand bars, etc. built by the work of energy sources from the sea will remain for varying times depending on the unique turnover characteristics of the structure and subsequent catastrophic events. 2) Habitat may be altered by a change in magnitude of a forcing function which carries a system variable outside of the range suitable for a species. For example, natural or anthropogenic climate change may result in sufficient warming in the Gulf of Maine to preclude successful reproduction of cold water species such as the sea scallop, *Placopectin magellanicus*, over much of its area, conversely, warmer waters can result in increased survival for species like *Homarus americanus* that are near the northern limits of their range (Dow 1977). 3) Structural changes in a system might result from direct or indirect affects of a qualitative or quantitative change in the suite of forcing functions. For example, road or bridge construction in the Florida Keys could increase turbidity to a point where sea grass habitat would be destroyed, or the nitrogen loading to an estuary from a new municipal treatment plant could increase primary production to the point where the microbial consumption of fixed carbon settling to the bottom depletes the oxygen concentration to lethal levels. (4) Habitat may also be altered by a regular or persistent shift in the timing or frequency of forcing events. For example, climate change might result in a change in the frequency of occurrence of 100 year hurricanes in Florida and Louisiana, which would result in increased risk for people and property residing in coastal habitats and lowland areas. Warmer winters in the Gulf of Maine could more consistently support development of zooplankton populations early in the year which would decrease the frequency of large pulses of phytoplankton biomass delivered to the benthos, and there by, diminish the number of dominant year classes in demersal fish populations of the region (Townsend and Cammen 1988, Townsend et al. 1994).

In general, an existing habitat will be altered by the introduction of a material across its boundaries, when that material is concentrated above a threshold for action. Once the material exceeds this threshold, it becomes an additional source of available energy for the system and will have an effect. The available energy will either be used by existing organisms or by organisms that are carried to the system from other places, or it will act as an energy drain (a stress) that exacts a metabolic cost from the organisms that process it and also from those that fail to process it. When the emergy sources of an incoming signature are balanced (Campbell 2000) and when the ecosystem with its concomitant species and populations have had sufficient time to adapt to these inputs, usually many types of organisms are present and their numbers are fairly well-balanced. When a new energy source with either a positive or negative effect is added, the growth of some species are favored over others, the number of species is often diminished, and those species best adapted to use or resist the effects of the new source overgrow the others and appear in greater numbers in the ecosystem (Yount 1956). When a stress is large enough to be close to the edge of the niche space for all organisms few survive and both numbers and species richness will be diminished.

3.2.2 A Generic Energy Systems Model for Suspended and Bedded Sediments

An energy systems model of the action of suspended and bedded sediments in a stream ecosystem was constructed (Figure 14) by beginning with the narrative description of the characteristics of suspended and bedded sediments as a stressor and using the overview diagram presented in Figure 13. All forcing functions, components and pathways in Figure 14 are defined in Table 5. An area corresponding to the boundaries of the system to be evaluated was delineated, in this case a stream reach. The system boundary is shown as a box enclosing the ecosystem components including the populations and stream features affected by suspended and bedded sediments (Figure 14). The processes and components of the fresh water ecosystem modeled here can be viewed as a generic representation of the classes of processes and components that are important in determining the effects of suspended and bedded sediments in any aquatic ecosystem. In large rivers and estuaries dredging is a major source of problems related to the deposition of suspended and bedded sediments and this process could be added to the models for those systems. This generic model can be used to guide the allocation of biological effects to suspended and bedded sediments in any aquatic system with appropriate modifications, guided by the canonical models presented above, and the expert knowledge of local scientists about the particular system to be evaluated. For example, to evaluate a shallow estuary the canonical model for bidirectional flow (Figure 9b) would be used to structure water flows and turnover times of the system rather than the unidirectional flow model used here and clean sediment processes of concern, such as dredging and/or the disposal of dredged materials would be added to the model.

The important forcing functions for the stream reach shown in Figure 14 are solar radiation, S, temperature, T_P, runoff, J_{W0}, groundwater base flow, J_{W1}, wind, Wd, nutrient concentration in runoff, N_0, bedload transport into the system by streamflow, Sb_0, suspended sediment concentration in runoff, Ss_0. The immigration of pelagic and benthic fish from downstream ecosystems, DS, into the stream reach is shown as proportional to the downstream populations, BF_D and PF_D. By convention, the external forcing functions are arranged around the outside of the box delineating ecosystem boundaries in order of increasing transformity.

The system components should include all the internal storage units that are important in determining the effects of suspended and bedded sediments in the stream. The important internal system components in the clean sediment model are shown on Figure 14 and identified in Table 5. The system components are substrate including rock, R, bedded sediments, Sb, streamform, Sf, streamflow, J_W, kinetic energy in the water, W_{ke}, suspended sediments, Ss, vegetation as phytoplankton, P, aquatic macrophytes, M, nutrient, N, detritus, D, pelagic invertebrates, PI, benthic invertebrates, BI, pelagic fish, PF, and benthic fish, BF.

The network of interactions is represented by the lines connecting the components and forcing functions through processes. Each line represents a flow of energy, material or information and is identified by a pathway coefficient, k_i, in Figure 14. The network includes runoff carrying inflows of nutrient, k_4, bedload sediment, k_5, and suspended sediment, k_6, according to the concentrations supplied by activities in the surrounding watershed. These same concentrations are removed by the stream outflow, J_W, as fluxes of suspended sediments, k_7, nutrients, k_8, and bedload sediments k_9. In this model some kinetic energy of the water goes into vertical mixing, k_{39}, mediating the resuspension and settling of suspended sediment along pathway k_{25}. The light attenuated by suspended sediments is shown on the pathway designated, k_3. The light absorbed by phytoplankton and aquatic macrophytes is designated on pathways k_1, and k_2, respectively. Pathways k_{10} through k_{13}, show the emigration and immigration of benthic and pelagic fish to and from downstream ecosystems. The uptake of nutrients by phytoplankton and aquatic macrophytes is shown by pathways k_{14} and k_{15}, respectively. Phytoplankton is consumed by benthic invertebrates on pathway k_{16} and k_{17} shows the phytoplankton consumed by pelagic invertebrates. On pathway k_{18}, phytoplankton dies and sinks to the bottom forming detritus. Pathway k_{19} shows the negative effects of abrasion and scour on aquatic macrophytes and pathway k_{20} carries the dead macrophyte biomass into the detritus pool. The positive support given to macrophyte growth by well-developed streamform acts on pathway k_{22}. Pathway k_{23} shows the detritus consumed by benthic invertebrates and pathway k_{24} represents the detritus lost by burial. Additional pathways of external supply and downstream washout can be added if loading with organic material is an important factor accompanying loading with clean sediment. Pathway k_{25} shows the net vertical flux of suspended sediments governed by the turbulent mixing energy in the water. Pathway k_{26} represents the action of water column energy building streamform. Note that this process is modeled as a push-pull interaction (Odum 1994) and that the kinetic energy in the water can result in a decrease in streamform, if it exceeds the optimum amount needed to build form in a particular stream. Pathways k_{27} and k_{28} represent the positive effects of the expected streamform on the growth and survival of benthic fish and invertebrates, respectively. The effect of bottom roughness on the production of turbulent energy in the water is represented by pathway k_{29}. Pathway k_{30} represents the loss of stream structure as a result of burial by and accumulation of excess clean sediment. Note that many ecosystem components, including macrophytes, benthic invertebrates, benthic fish, and detritus will be affected negatively by the loss of streamform. Pathway k_{31} carries the nutrient dissolved from rock by the flowing waters and pathway k_{32} represents the streamform created as a result of this action. The negative effects of scour and abrasion on benthic invertebrates are shown on pathway k_{33}. The benthic invertebrates eaten by benthic fish and pelagic fish are shown on pathways k_{34} and k_{35}, respectively. The pelagic invertebrates eaten by benthic fish and pelagic fish are shown on pathways k_{36} and k_{37}, respectively. Pathway k_{38} represents the effects of abrasion on pelagic invertebrates. The production of turbulent kinetic energy in the water is shown on pathway k_{39} and the kinetic energy used to drive vertical circulation and resuspension is shown on pathway k_{40}. The action of the wind in generating kinetic energy in the water is represented on pathway k_{41}, and the

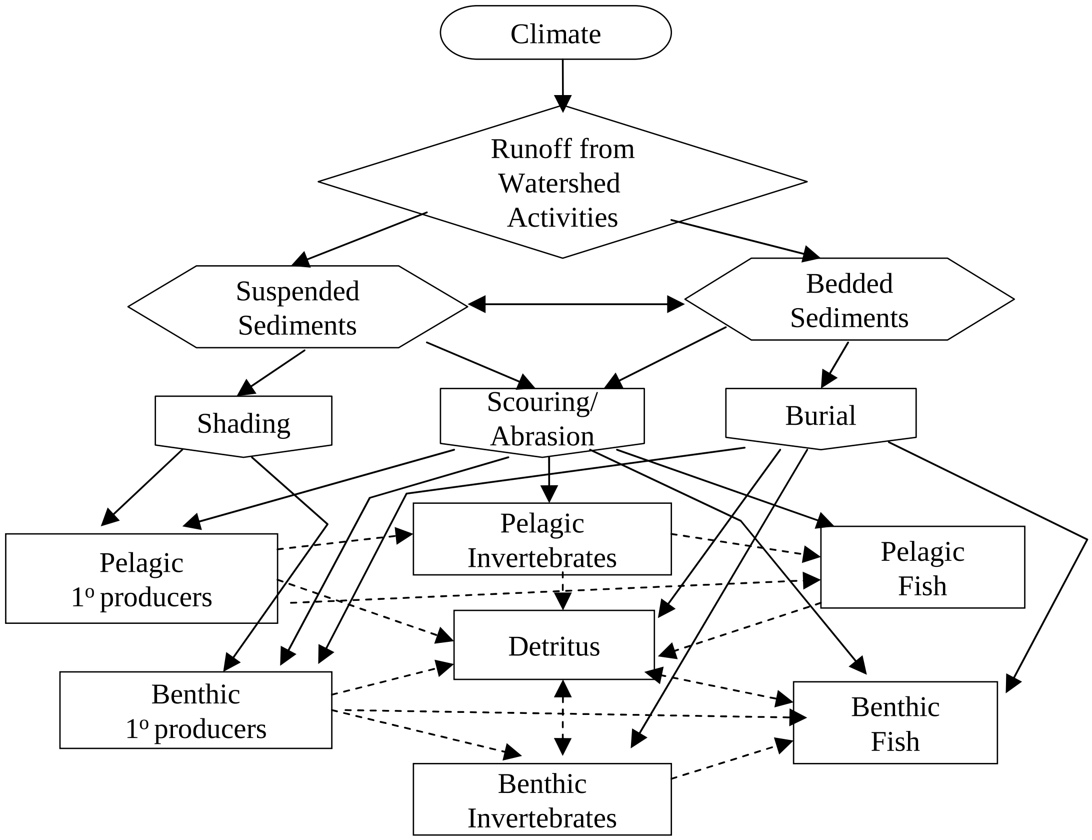


Figure 13. Summary diagram showing the factors that control the effects of suspended and bedded sediments in aquatic ecosystems.

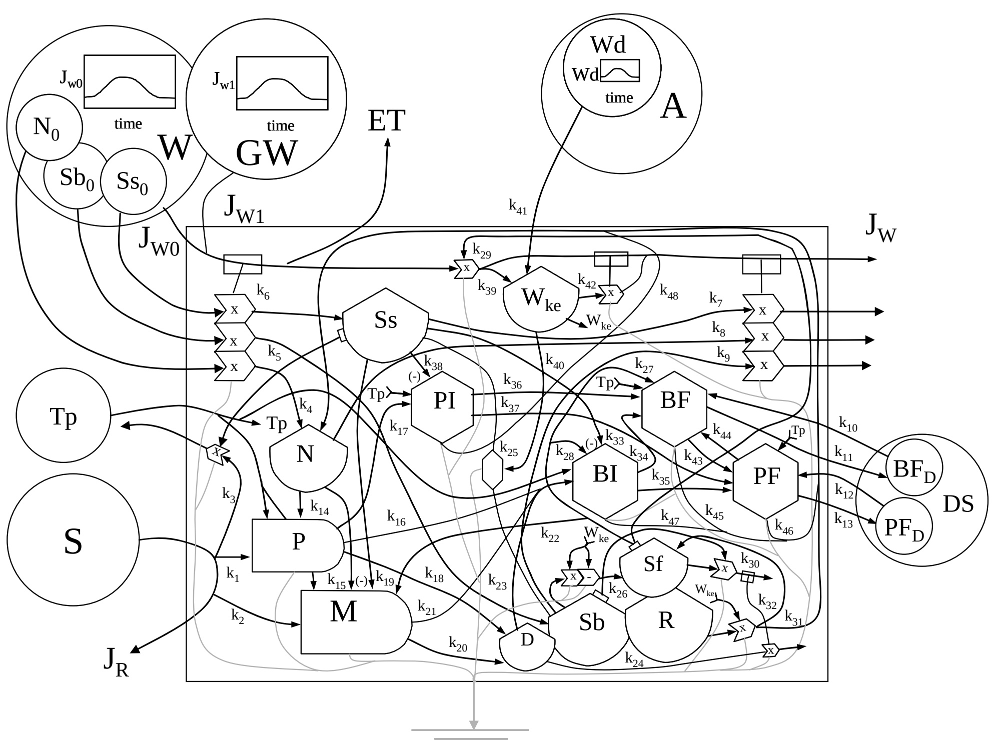


Figure 14. An energy systems model of the effects of suspended and bedded sediments on aquatic ecosystems. Forcing functions, components, and pathway flows are defined in Table 5.

Table 5. Definition of the forcing functions, components, and pathways in the generic energy systems model to evaluate the effects of suspended and bedded sediments on aquatic ecosystems.

Symbol	Definition
Forcing Functions	
S	Solar insolation as a time series
J_R	Solar radiation that remains unused (albedo)
T_P	Temperature as a time series
W	Watershed (time series of water flow)
Ss_0	Suspended sediments in the runoff (can be a time series)
Sb_0	Bedload sediments from upstream (can be a time series)
N_0	Nutrient concentration in the runoff (can be a time series)
GW	Groundwater aquifer and is characteristics
J_{W0}	Water flowing in as runoff
J_{W1}	Groundwater base flow
ET	Water evapotranspired
A	Atmosphere system
Wd	Wind as a time series
DS	Downstream ecosystems
Components	
N	Nutrient
P	Phytoplankton
M	Macrophytes
W_{ke}	Kinetic energy in the water
Ss	Suspended sediment in the stream
Sf	Streamform (physical structure of the stream bed used by a species and/or its competitors)
Sb	Bedded sediments
R	Rock in the stream bed
D	Detritus on the stream bottom
PI	Pelagic invertebrates
PF	Pelagic fish
BI	Benthic invertebrates
BF	Benthic fish
Pathways and flows	
J_W	Water flowing out of the system
k_1	Light used by phytoplankton (the producer symbol implies GPP, NPP respiration.)
k_2	Light used by macrophytes (the producer symbol implies GPP, NPP respiration.)
k_3	Light attenuated by turbidity in the water (shading)
k_4	Nutrient inflow in runoff
k_5	Bedload transport into the system.
k_6	Suspended sediments inflowing in runoff.
k_7	Suspended sediments flowing out in streamflow.
k_8	Nutrients flowing out in streamflow.
k_9	Bedload transport out of the system.
k_{10}	Immigration of benthic fish from downstream ecosystems

Symbol	Definition
k_{11}	Emigration of benthic fish to downstream ecosystems
k_{12}	Immigration of pelagic fish from downstream ecosystems
k_{13}	Emigration of pelagic fish to downstream ecosystems
k_{14}	Nutrient uptake by phytoplankton
k_{15}	Nutrient uptake by macrophytes
k_{16}	Phytoplankton eaten by benthic invertebrates
k_{17}	Phytoplankton eaten by pelagic invertebrates
k_{18}	Phytoplankton settling to detritus
k_{19}	Scour and abrasive effect of suspended sediments on macrophytes
k_{20}	Injured and dead macrophyte biomass going to detritus
k_{21}	Macrophyte biomass eaten by benthic invertebrates
k_{22}	Positive effect of streamform on macrophytes
k_{23}	Detritus eaten by benthic invertebrates
k_{24}	Detritus buried by sediments
k_{25}	Resuspension of bedded sediments and settling of suspended solids
k_{26}	Kinetic energy of water building streamform from sediments
k_{27}	Positive effect of streamform on benthic fish
k_{28}	Positive effect of streamform on benthic invertebrates
k_{29}	Roughness of streamform creating turbulence
k_{30}	Loss of streamform by burial
k_{31}	Nutrients dissolved from rock
k_{32}	Streamform built by kinetic energy acting on rock in the stream bed
k_{33}	Scour and abrasive effects of suspended sediments on benthic invertebrates
k_{34}	Benthic invertebrates eaten by benthic fish
k_{35}	Benthic invertebrates eaten by pelagic fish
k_{36}	Pelagic invertebrates eaten by benthic fish
k_{37}	Pelagic invertebrates eaten by pelagic fish
k_{38}	Scour and abrasion effects of suspended sediments on pelagic invertebrates.
k_{39}	Kinetic energy transported into and generated by fluid flow in the system.
k_{40}	Kinetic energy dissipated in mixing.
k_{41}	Kinetic energy generated by wind.
k_{42}	Kinetic energy transported out of the system by streamflow.
k_{43}	Benthic fish eaten by pelagic fish.
k_{44}	Pelagic fish eaten by benthic fish.
k_{45}	Nitrogen recycled by benthic fish.
k_{46}	Nitrogen recycled by pelagic fish.
k_{47}	Nitrogen recycled by benthic invertebrates.
k_{48}	Nitrogen recycled by pelagic invertebrates.

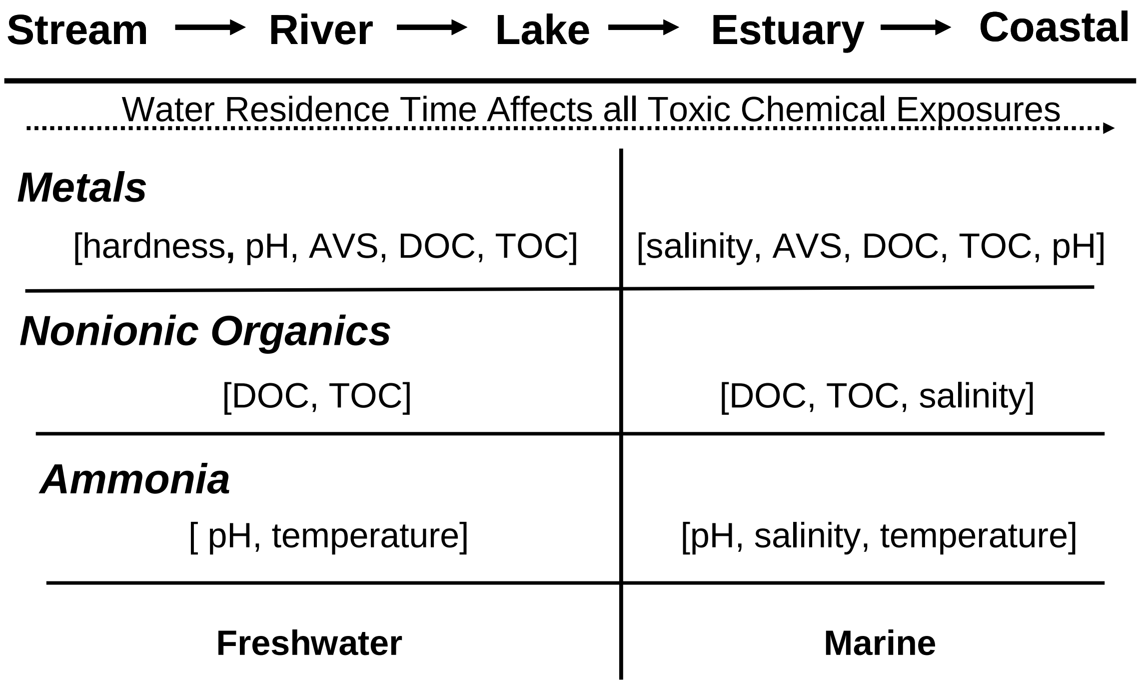


Figure 15a. Factors that affect the bioavailability of toxic chemicals in freshwater and marine systems.

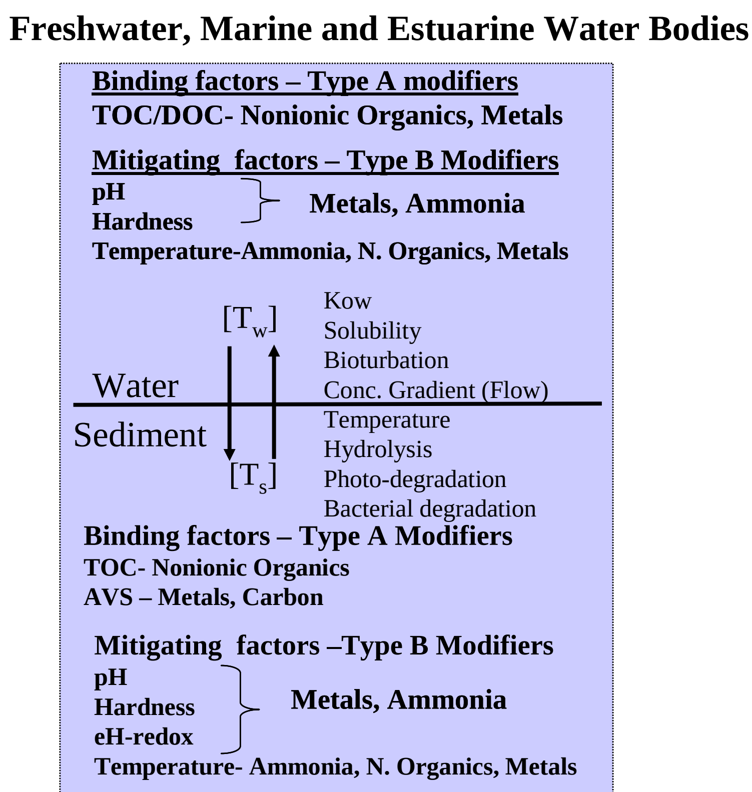


Figure 15 b. Factors that affect the availability of toxic chemicals in the water column and in sediments. T is the concentration of bioavailable toxin.

removal of kinetic energy in streamflow is shown as pathway k_{42}. Pelagic fish eat benthic fish, k_{43}, and in turn benthic fish eat pelagic fish, k_{44}. Nutrients are recycled by the metabolism of the benthic fish, pelagic fish, benthic invertebrates, and pelagic invertebrates, as shown on pathways, k_{45}, k_{46}, k_{47}, and k_{48}, respectively. The reader should also examine Kaufmann et al. (1999) for guidance on developing clean sediment models for stream ecosystems.

3.2.3 A Generic Energy Systems Model for Toxic Chemicals

An energy systems model of the actions of toxic chemicals in aquatic ecosystems was constructed (Figure 16) by beginning with the narrative description of the characteristics of toxic chemicals as a stressor and using the conceptual summary presented in Figure 15a and b. The construction of a generic energy systems model that covered all toxic chemicals was a challenge, because toxic actions are diverse and the number of toxic chemicals is large. Nevertheless, at a high level of abstraction, we believe that all toxicants demonstrate some basic similarities in behavior. The narrative description and Figure 15 summarize the salient factors controlling toxic action in fresh water and marine systems. We propose that most toxic chemicals can be modeled as a subsystem containing state variables for available toxin and unavailable toxin (Figure 16). An available toxicant is capable of causing toxic effects to organisms, populations, and the ecosystem. An unavailable toxic chemical has been sequestered by one or more possible mechanisms that depend on the particular chemical. For example, organic toxic chemicals like PCBs partition into organic material adsorbed to particulate matter, and positively charged metal ions are neutralized by sulfates and other negatively charged ligands. Our generic model of toxic action in an ecosystem is centered on the processes controlling the availability of the toxic chemical in sediments and the overlying water. We also hypothesize that despite the diversity of mechanisms of toxic action all forms of toxicity act as an energy drain on biological organization (Odum 1968). These energy drains can manifest as direct mortality or they can impose an additional metabolic cost on organisms that diminishes growth and reproduction. These common characteristics of toxic substances and toxic action are represented in the energy systems model presented in Figure 16. The details of the model constructed for a particular pollutant will be different depending on the chemical and its properties, but we believe that this highly aggregated conceptual model can serve as a useful guide in organizing thinking and structuring analyses of the effects of toxic chemicals on aquatic ecosystems.

All forcing functions, components and pathways in Figure 16 are defined in Table 6. An area corresponding to the boundaries of the system to be evaluated is delineated along with its accompanying geometric characteristics, in this case the area, average length, width, depth, etc. of the stream reach. The system boundary is shown as a box enclosing the ecosystem components including the plant and animal populations affected by toxic loadings. The formulations given in this model are more precise than those given for the other three stressors, because the specifics of some component interactions are shown within the aggregate symbols for producers and consumers. This amount of detail allows us to write a set of simultaneous differential equations describing the network of interactions directly from the energy circuit diagram. We have not given the equations because the purpose of these models is to provide a conceptual overview and not the mathematical formulations that would be used in simulation. The rate processes governing the equilibrium between bioavailable and sequestered toxic chemical are given by pathway coefficients for the toxic chemical subsystem in the water column. Similar rate processes are diagramed in the sediments, but have not been explicitly identified with coefficients. However, these processes mirror those given for the water column and the reader can easily identify parallel processes in the sediment diagram. This generic model can guide the allocation of biological effects to toxic chemicals in any aquatic system with appropriate modifications, guided by the canonical models and the expert knowledge of

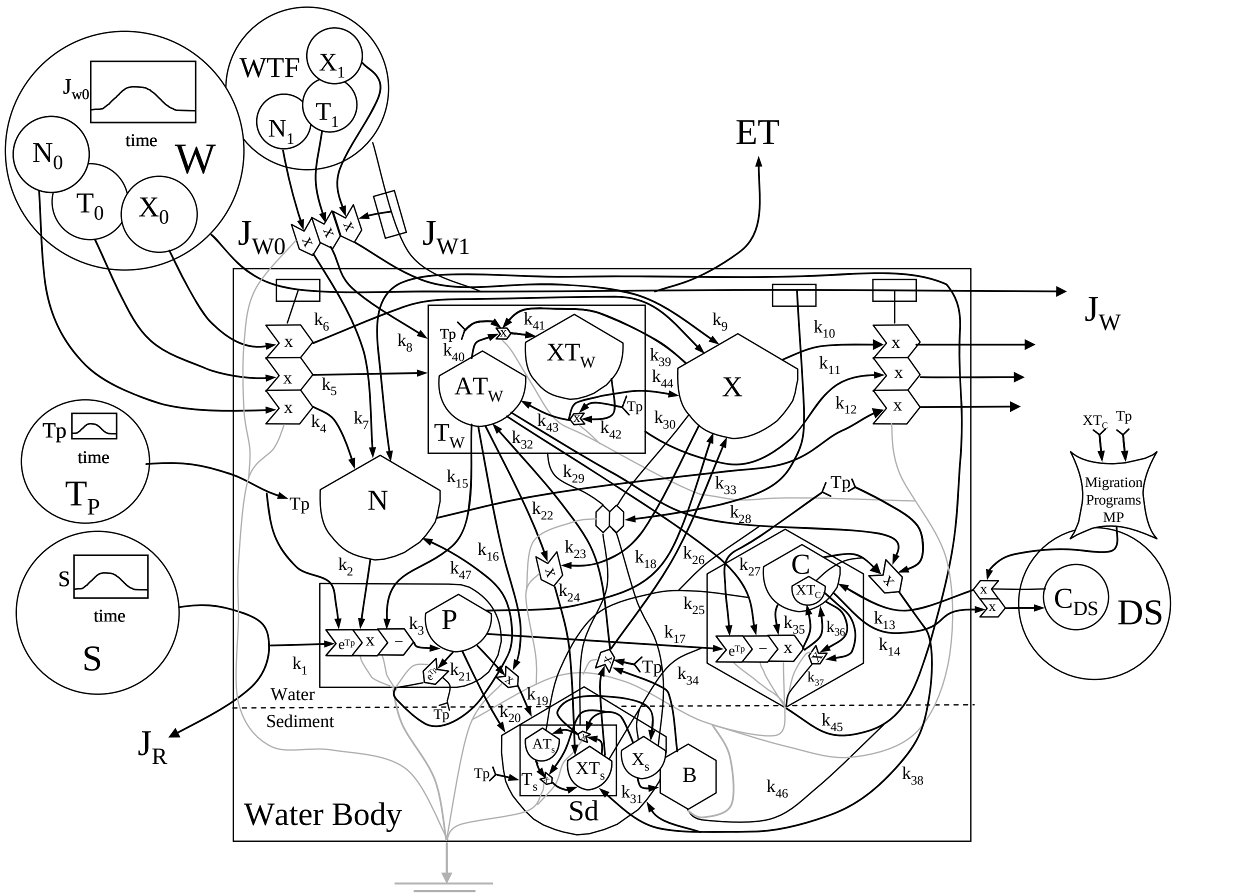


Figure 16. An energy systems model for the effects of a toxic chemical, T, on an aquatic ecosystem. Forcing functions, components, and pathway flows are defined in Table 6.

local scientists about the particular system to be evaluated.

The important forcing functions for the model of the effects of toxic chemicals on a stream reach (Figure 16) are solar radiation, S, temperature, T_P, runoff, J_{W0}, and waste water inflow, J_{W1}. All of these forcing functions are given as a time series of values when fully specified. Each of the water inflows can carry concentrations of nutrients, chemicals, and other materials determined by the characteristics of upstream flow and land use in the immediate watershed of the reach, W, or by the characteristics of the waste stream from waste water treatment facilities, WTF. The nutrient concentration, N_0, the concentration of toxic chemical, T_0, and the concentrations of modifying chemicals, X_0, in streamflow and immediate runoff enter the stream reach. The nutrient, toxic chemical, and modifying chemical loadings in the waste stream are given as the concentration of a single nutrient, N_1, toxic chemical, T_1, and modifying chemical, X_1, respectively. Where more than one pollutant is important in the process under evaluation, the model can be expanded to consider these additional materials. A final forcing function is the immigration and emigration of consumers from downstream ecosystems, DS, into the stream reach under evaluation. Consumer movements are proportional to the downstream populations, C_{DS}, and seasonal or other behavioral programming, MP. The program controlling migration was made more complex by adding temperature to provide a temporal cue for migration and by using the burden of toxic chemicals carried by the organisms, XT_C, to make the probability of migration more or less likely by affecting the organism's behavior. The program controlling the migratory behavior of consumers can be duplicated on the upstream boundary where this interaction is important.

The system components should include all internal state variables that are thought to be

Table 6. Definition of the forcing functions, components, and pathways in the generic energy systems model designed to evaluate the effects of toxic chemicals on aquatic ecosystems.

Symbol	Definition
Forcing Functions	
S	Solar insolation as a time series
J_R	Solar radiation that remains unused (albedo)
T_P	Temperature as a time series
W	Watershed (time series of water flow)
J_{W0}	Water flowing in as runoff
N_0	Nutrient concentration in the runoff (can be a time series)
T_0	Concentration of the toxic chemical in the runoff (can be a time series)
X_0	Concentration of chemicals that modify toxicity (can be time series)
WTF	Outflow from waste water treatment facilities (can be a time series)
J_{W1}	Waste water inflow
N_1	Nutrient concentration in the waste water (can be a time series)
T_1	Concentration of the toxic chemical in waste water (can be a time series)
X_1	Concentration of modifying chemicals in waste water (can be a time series)
ET	Water evapotranspired in the system
DS	Downstream ecosystems
C_{DS}	Downstream populations of migrating consumers
MP	Behavioral programs controlling animal migrations
Components	
N	Nutrient
P	Primary producers
X	Modifying chemicals in the water
T_W	Toxic chemical subsystem
AT_W	Available toxin in the water
XT_W	Unavailable toxin in the water (*i.e.*, chemically bound or neutralized)
C	Consumers
XT_C	Toxic chemicals in the tissue of consumers
Sd	Sediments
B	Bacteria
T_S	Toxic chemical subsystem in the sediments
AT_S	Available toxin in the sediment pore water
XT_S	Unavailable toxin in the sediments (*i.e.*, chemically bound or neutralized)
X_S	Modifying chemicals in the sediment
Pathways	
J_W	Water flowing out of the system
k_1	Light used by primary producers
k_2	Nutrient used by the primary producers
k_3	Gross primary production
k_4	Nutrient inflow in runoff
k_5	Toxic chemical inflowing in runoff.
k_6	Modifying chemicals inflowing in runoff.
k_7	Nutrient added in waste water.
k_8	Toxic chemical added in waste water.

Symbol	Definition
k_9	Modifying chemicals added in waste water.
k_{10}	Modifying chemicals flowing out in water leaving the system.
k_{11}	Toxic chemical flowing out in water leaving the system.
k_{12}	Nutrients flowing out in water leaving the system.
k_{13}	Immigration of consumers from downstream or seaward ecosystems
k_{14}	Emigration of consumers to downstream or seaward ecosystems
k_{15}	Available toxin used up in decreasing plant growth.
k_{16}	Available toxin used up in causing direct mortality of plants
k_{17}	Consumption of primary producers by consumers.
k_{18}	Modifying chemicals produced by plants
k_{19}	Plant death from direct mortality caused by available toxin
k_{20}	Plant death from other causes
k_{21}	Plant respiration
k_{22}	Toxic chemical bound and precipitated to the bottom
k_{23}	Modifying chemicals used to bind the toxin
k_{24}	Bound toxin settling to the bottom
k_{25}	Available toxin in the sediment used in decreasing consumer growth
k_{26}	Available toxin in the sediment used to cause direct mortality of consumers
k_{27}	Available toxin in the water used in decreasing consumer growth
k_{28}	Available toxin in the water used to cause direct mortality of consumers
k_{29}	Resuspension and settling of the toxic chemical subsystem
k_{30}	Resuspension and settling of the modifying chemicals
k_{31}	Bacterial consumption of modifying chemicals if appropriate
k_{32}	Bacterial decomposition and release of bound toxin from the sediment
k_{33}	Bacterial release of modifying chemicals from the sediment
k_{34}	Toxic chemicals eaten by consumers
k_{35}	Toxic chemical incorporated into consumer biomass
k_{36}	Carbon assimilated by consumers
k_{37}	Toxic chemical and carbon metabolized by consumers
k_{38}	Loss of biomass and toxins in mortality caused by available toxin.
k_{39}	Modifying chemicals interacting with available toxin in the water
k_{40}	Available toxin bound by modifying chemicals in the water
k_{41}	Bound toxin produced in the water
k_{42}	Bound toxin decomposed in the water column
k_{43}	Available toxin generated in the water by decomposition of bound toxin
k_{44}	Modifying chemicals generated by the decomposition of bound toxin
k_{45}	Nutrients recycled by consumer respiration.
k_{46}	Nutrients recycled by bacterial respiration.
k_{47}	Nutrients recycled by plant respiration.

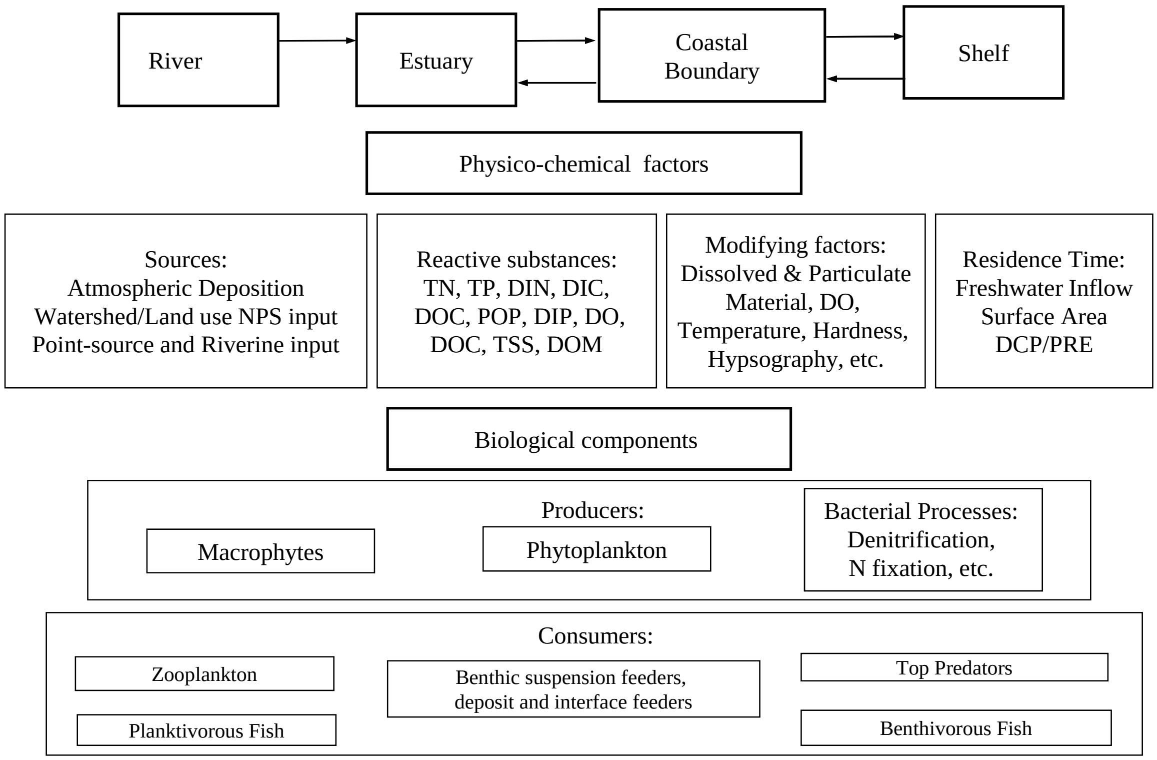


Figure 17. Factors that control the effects of nutrients in aquatic ecosystems.

important in determining the effects of the toxic chemical in the stream. Where new knowledge or additional research indicates that other components are important they must be added to the model structure. In energy systems models, components are combined according to function as shown by the broadly aggregated producer and consumer components in Figure 16. Where analysis shows that there are relevant differences in behavior within an aggregate group, it can be broken down according to functional differences to provide the additional detail needed to explain observations. Each toxic chemical and modifying chemical combination will require the particular details of the relationship to be substituted into the general form of the model given here.

The ecosystem components found in the water are nutrients, N, the toxic chemical subsystem, T_W, including bioavailable toxic chemical, AT_W, and bound or sequestered toxic chemical, XT_W; modifying chemicals, X, primary producers, P, and consumers, C. In addition, the model includes a sediment phase containing sediments, Sd, the toxic chemical subsystem in the sediments, T_S, including bioavailable toxic chemical, AT_S, and bound or sequestered toxic chemical, XT_S, modifying chemicals, X_S, and bacteria, B.

The network of interactions is represented by the lines connecting the components and forcing functions through processes as shown in Figure 16. Each line represents a flow of energy, material or information and is identified by a pathway coefficient, the k_is. The light absorbed by phytoplankton and aquatic macrophytes is designated by pathway k_1. Pathway k_2 shows the nutrient taken up in primary production. The gross primary production, GPP, of the plants is given on the pathway designated k_3. Temperature drives the metabolic processes of plants, consumers, and bacteria in the ecosystem using an Arrhenius formulation (e^{kT}) to control the rate. Runoff carries in flows of nutrients, k_4, toxic chemicals, k_5, and modifying chemicals, k_6, according to the concentrations supplied by activities in the surrounding watershed. Nutrients, toxic chemicals, and modifying chemicals (type A modifiers) can also enter the system in the waste water stream as shown, respectively, on pathways, k_7, k_8, and k_9. These same concentrations are removed by the stream outflow, J_W, as fluxes of modifying chemicals, k_{10}, toxic chemicals, k_{11}, and nutrients, k_{12}. Pathways k_{13} and k_{14} show the immigration and emigration of consumers to and from downstream ecosystems under the control of behavioral migration programs that are modified by temperature and the accumulation of toxic chemical in animal tissue.

The negative effects of available toxic chemical on primary production are shown on pathway k_{15}. The available toxin taken up in causing direct mortality to plants is represented by pathway k_{16}. The plants eaten by consumers are shown on pathway k_{17} and the modifying chemicals, *e.g.*, dissolved organic carbon, DOC, released by plants are given on pathway k_{18}. Pathway k_{19} shows plant biomass that dies from toxic exposure and sinks to the bottom forming detritus, while pathway k_{20} represents plants that die from natural causes. Plant respiration, shown on pathway k_{21}, is a function of temperature. Pathways k_{22}, k_{23}, and k_{24}, show the interaction of toxic chemicals and modifying chemicals in the water to make the toxic chemical unavailable. Pathway k_{25} shows the adverse effects of available toxic chemicals in the sediment on the growth of consumers and pathway k_{26} represents direct mortality of animals caused by available toxins in the sediment. Similarly, pathways k_{27} and k_{28} show the decreased growth and direct mortality of consumers caused by available toxicant in the water.

Mixing in this stream ecosystem is assumed to be proportional to water flow and the concentration gradient between sediments and the water column. The net resuspension or settling of materials in the toxic chemical subsystem is shown on pathway k_{29}. A similar balance of resuspension and settling for the modifying chemicals is given on pathway k_{30}. The modifying chemicals consumed by bacteria are shown on pathway k_{31}. Bacteria breakdown unavailable toxic chemical bound in the sediment and recycle it to the water column on pathways k_{32} and k_{33}. The consumption of bound toxic chemical and modifying chemicals from the sediment by consumers is given on pathway k_{34}. Consumer biomass growth is shown on pathway k_{36} and the concomitant incorporation of toxic chemicals into animal tissue is given on pathway k_{35}. Pathway k_{37} shows the respiration of consumers and pathway k_{38} gives the consumer biomass that dies and returns to the sediments with its body burden of toxic chemical. The transitions between available and unavailable forms of the toxic chemical within the toxic chemical subsystem in the water column are given by the next six pathway coefficients (Table 6). A similar set of coefficients governs these transitions in the sediment and they can be easily derived by following the pattern used for the water column. Pathway k_{39} is the quantity of modifying chemical, X, that interacts with a quantity of available toxic chemical, k_{40}, to form unavailable toxicant in the water. Conversely, a quantity of unavailable toxic chemical in the water, k_{42}, decomposes both biologically and chemically under the influence of temperature to form a quantity of available toxic chemical on pathway, k_{43}, and modifying chemicals on pathway k_{44}. Nutrients are recycled into the water by consumers, k_{45}, bacteria, k_{46}, and plants, k_{47}.

3.2.4 A Generic Energy Systems Model for Nutrients

An energy systems model of the actions of nutrient enrichment in aquatic ecosystems was constructed (Figure 18) using the narrative description of nutrient enrichment and the overview diagram presented in Figure 17. Figure 18 like the generic energy systems models for the other three stressors is structured using a fresh water stream ecosystem as the example. The intent is that these generic models serve as a useful guide to understanding the stressor's action over a range of aquatic ecosystems (see the spectrum of system types shown at the top of Figures 11, 13, 15a, and 17). As pointed out in the narrative description, phosphorus is often the limiting nutrient in fresh water and terrestrial ecosystems, whereas, nitrogen is more frequently limiting in coastal and marine waters.

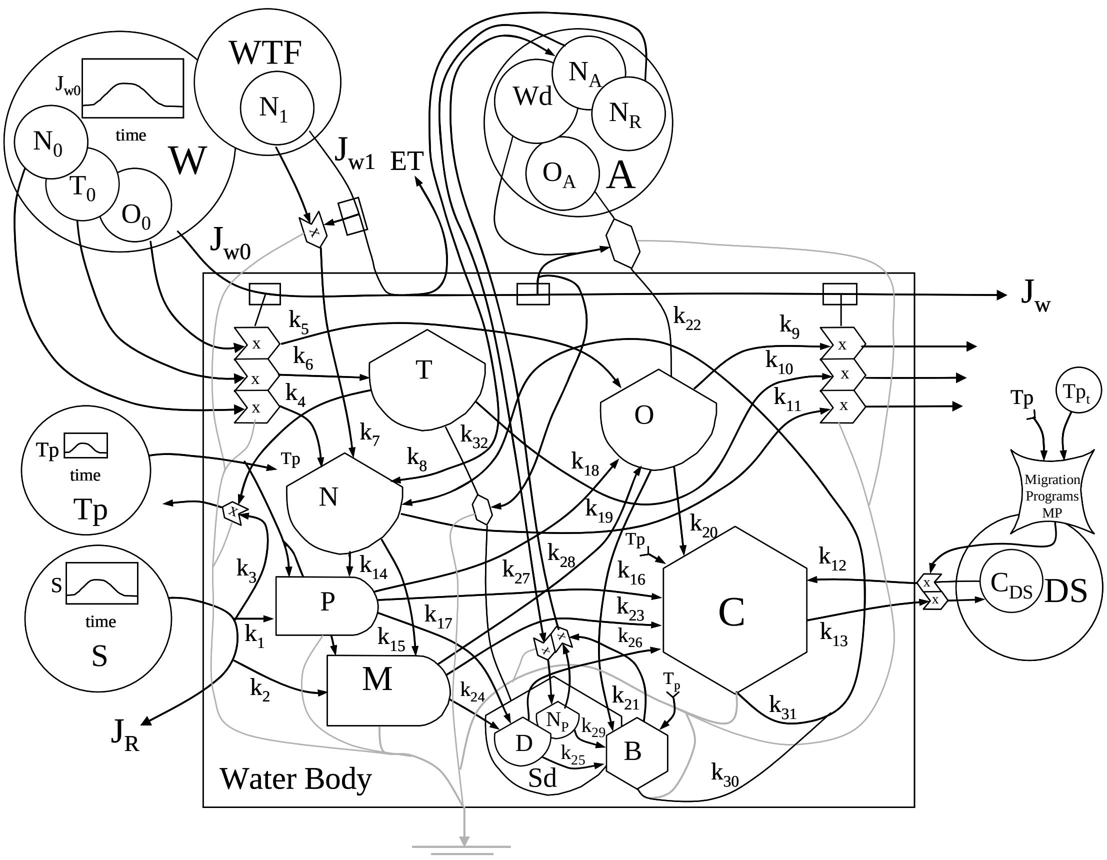


Figure 18. An energy systems model of the effects of excess nutrients on an aquatic ecosystem. Forcing functions, storages, and flows are defined in Table 7.

Ecosystem processes controlling nitrogen are included in the generic model for nutrient enrichment, because nitrogen can be limiting in fresh water stream systems and the pathways are needed to model eutrophication in coastal and marine ecosystems. All forcing functions, components and pathways in Figure 18 are defined in Table 7.

First, an area corresponding to the boundaries of the system to be evaluated is delineated. The system boundary for a stream is shown as a box enclosing the ecosystem components including the populations and stream features affected by nutrient loading (Figure 18). This generic model can be used as a guide for allocating observed biological effects to nutrient loading in any aquatic system with appropriate modifications, guided by the canonical models and expert knowledge of scientists about the particular system to be evaluated. For example, to evaluate the effects of nutrient loading on a shallow estuary the canonical model for bidirectional flow (Figure 9b) would be used to represent water flow and turnover time of the system rather than the unidirectional flow model used here. For a phosphorus-limited, deep lake, the nitrogen pathways shown in Figure 18 might be omitted and other appropriate pathways added using the canonical model in Figure 9c. The important forcing functions for the stream reach shown in Figure 18 are solar radiation, S, temperature, T_P, runoff, J_{W0}, waste water flow, J_{W1}, wind, Wd. All of these forcing functions are given as a time series of values when fully specified. Each of the water inflows can carry concentrations of nutrients determined by the characteristics of land use in the watershed, W, or by the characteristics of the waste stream from the waste water treatment facility, WTF. The nutrient concentration in runoff, N_0, the turbidity in runoff and streamflow, T_0, and oxygen in streamflow enter the stream reach from

Table 7. Definition of the forcing functions, components, and pathways in the generic energy systems model to evaluate the effects of nutrient loading on aquatic ecosystems.

Symbol	Definition
Forcing Functions	
S	Solar insolation as a time series
J_R	Solar radiation that remains unused (albedo)
T_P	Temperature as a time series
W	Watershed (time series of water flow)
J_{W0}	Water flowing in as runoff
N_0	Nutrient concentration in the runoff (can be a time series)
T_0	Turbidity in the runoff (can be a time series)
O_0	Oxygen concentration in the runoff (can be a time series)
WTF	Outflow from waste water treatment facilities (can be a time series)
J_{W1}	Waste water inflow
N_1	Nutrient concentration in the waste water (can be a time series)
ET	Water evapotranspired in the system
A	Atmosphere system
Wd	Wind as a time series
O_A	Concentration of oxygen in the air
N_A	Diatomic nitrogen concentration in the atmosphere
N_R	Nitrogen in rain (can be an important input to estuaries)
DS	Downstream ecosystems
C_{DS}	Downstream populations of migrating consumers
MP	Behavioral programs controlling animal migrations
Components	
N	Nutrient
P	Phytoplankton
M	Macrophytes
T	Turbidity in the water
O	Concentration of oxygen in the stream
Sd	Sediment on the bottom
D	Detritus
B	Bacteria
N_P	Nitrogen pool in the sediment
C	Consumers
Pathways	
J_W	Water flowing out of the system
k_1	Light used by phytoplankton (the producer symbol implies GPP, NPP respiration.)
k_2	Light used by macrophytes (the producer symbol implies GPP, NPP respiration.)
k_3	Light attenuated by turbidity in the water
k_4	Nutrient inflow in runoff
k_5	Oxygen inflowing in runoff.
k_6	Turbidity inflowing in runoff.
k_7	Nutrient added in waste water.

Symbol	Definition
k_8	Nutrient supplied in rainfall
k_9	Oxygen flowing out in water leaving the system.
k_{10}	Turbidity flowing out in water leaving the system.
k_{11}	Nutrients flowing out in water leaving the system.
k_{12}	Immigration of consumers from downstream or seaward ecosystems
k_{13}	Emigration of consumers to downstream or seaward ecosystems
k_{14}	Nutrient uptake by phytoplankton
k_{15}	Nutrient uptake by macrophytes
k_{16}	Phytoplankton eaten by consumers
k_{17}	Phytoplankton death to detritus
k_{18}	Oxygen produced by phytoplankton
k_{19}	Oxygen produced by macrophytes
k_{20}	Oxygen used by consumers
k_{21}	Oxygen used by bacteria
k_{22}	Oxygen exchange with the atmosphere
k_{23}	Macrophytes eaten by consumers
k_{24}	Macrophyte death to detritus
k_{25}	Detritus consumed by bacteria
k_{26}	Detritus eaten by consumers
k_{27}	Nitrogen fixation
k_{28}	Denitrification
k_{29}	Nitrogen used by bacteria.
k_{30}	Nutrients recycled by bacterial metabolism
k_{31}	Nutrients recycled by the metabolism of consumers
k_{32}	Settling and resuspension of turbidity

upstream. Nutrient loading in the waste stream is given as the concentration of a single pollutant, N_1. Where more than one material is important in the eutrophication process the material inputs and the model storages and interactions can be expanded to consider additional materials. Odum (1994) gives model formulations for two limiting nutrients. Inputs from and interactions with the atmosphere, A, may be needed to model the effects of nutrient enrichment. Wind energy, Wd, along with streamflow, J_W, drive the exchange of oxygen across the water surface in proportion to the difference between the concentration of oxygen at saturation in the atmosphere, O_A, and oxygen concentration in the water, O. In addition, for lakes, wide estuaries, and coastal and shelf waters, the input of nutrient concentrations in rainfall, N_R, can be important (the time series of rainfall might be added to the model where this input is large). Diatomic nitrogen gas in the atmosphere, N_A, serves as a source of nitrogen for bacterial processes fixing nitrogen and receives the nitrogen that results from bacterial denitrification. A final forcing function is the movement of consumers in the stream reach into and out of downstream ecosystems, DS. The movement of consumers into the reach is proportional to the downstream populations, C_{DS}, controlled by seasonal and/or other behavioral programming, MP.

The system components should include all internal state variables or stored quantities that are thought to be important in determining the effects of nutrient loading in the stream. When the analysis shows that there are differences in function or behavior within an aggregate group, the group can be disaggregated to provide the additional detail needed to explain the observed data. The ecosystem components are nutrient, N, turbidity, T, oxygen, O, phytoplankton, P, aquatic macrophytes, M, detritus, D, sediments, Sd, bacteria, B, the sediment nitrogen pool, N_P, and consumers, C, including a broad range of organisms (Figure 18).

The network of interactions is represented by the lines connecting the components and forcing functions through processes as shown in Figure 18. Each line represents a flow of energy, material or information and is identified by a pathway coefficient, the k_i's on Figure 18. The light absorbed by phytoplankton and aquatic macrophytes is designated by pathways k_1 and k_2, respectively. The light attenuated by turbidity in the water is shown on the pathway designated, k_3. Runoff carries in flows of nutrient, k_4, oxygen, k_5, and turbidity, k_6, according to the concentrations supplied by activities in the surrounding watershed. Nutrients also enter the system in the waste water stream, k_7, and in rainfall, k_8. These same concentrations are removed by the stream outflow, J_W, as fluxes of oxygen, k_9, turbidity, k_{10}, and nutrient k_{11}. Pathways k_{12} and k_{13} show the immigration and emigration of consumers to and from downstream ecosystems under the control of temperature driven behavioral migration programs. Migratory movements are controlled by temperature thresholds, Tp_t. The uptakes of nutrients by phytoplankton and aquatic macrophytes are shown on pathways k_{14} and k_{15}, respectively. Phytoplankton eaten by consumers is shown on pathway k_{16} and on pathway k_{17} phytoplankton die and sink to the bottom forming detritus. The oxygen produced by phytoplankton and macroalgae is shown on pathways k_{18} and k_{19}, respectively. The oxygen used by the consumers and bacteria is given on pathways k_{20} and k_{21}. The oxygen exchanged with the atmosphere is shown on pathway k_{22}. Benthic macrophytes eaten by consumers are shown on pathway k_{23} and pathway k_{24} shows the macrophytes that die and fall to the bottom as detritus. The detritus consumed by bacteria is represented by pathway k_{25} and the detritus eaten by consumers is shown as pathway k_{26}. The rates of nitrogen fixation, k_{27}, and denitrification, k_{28}, are mediated by bacterial processes in the sediment. The nitrogen from the sediment nitrogen pool that is processed by bacteria is shown as pathway k_{29}. Nutrients are recycled into the water by bacteria, k_{30}, and consumers, k_{31}. In the model, streamflow determines the balance between settling and resuspension of turbidity in the water along pathway k_{32}.

3.2.5 Caveat on the Detailed Models of Pollutants

The descriptions of the network of interactions in aquatic stream ecosystems controlling suspended and bedded sediments, toxic chemicals and nutrients are not complete listings of all factors that may prove to be important in determining the effects of these stressors on aquatic ecosystems. The detailed models are simply guides to thinking about the particular problem whose details may include some or all of the factors diagramed here or other factors that may prove important in understanding the particular problem. Models are best used as dynamic tools to be continually revised as the circumstances of analysis change or as more knowledge is gained. These conceptual models can be further developed and evaluated to create simulation models suitable for the allocation of biological effects among multiple stressors. The detailed models in their present form may be useful as tools to stimulate thinking about the processes controlling the biological effects of suspended and bedded sediments, toxic chemicals, and on aquatic ecosystems from fresh water streams to the coastal shelf.

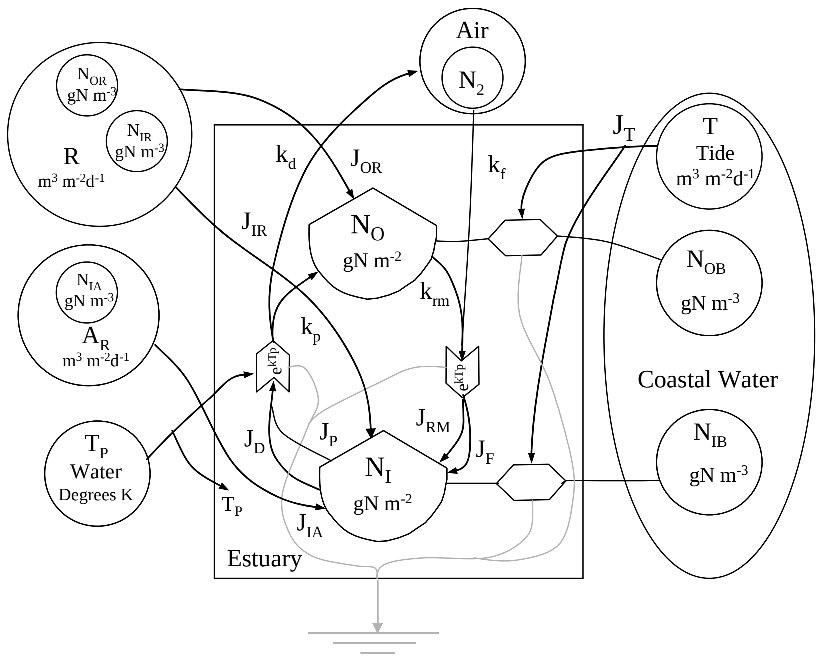
Table 8. Definition of the forcing functions, components, and pathways in the canonical energy systems model of excess nutrient in an estuary.

Symbol	Definition
Forcing Functions	
T_P	Temperature as a time series
A_R	Rainfall from the atmosphere (data as a time series)
N_{IA}	Inorganic nitrogen in rainfall (can be a time series)
R	River inflow (time series of water flow)
N_{IR}	Inorganic nitrogen concentration in river inflow (can be a time series)
N_{OR}	Organic nitrogen in river inflow(can be a time series)
T	Tidal exchange of waters
N_{OB}	Organic nitrogen in the coastal water(can be a time series)
N_{IB}	Inorganic nitrogen in the coastal water(can be a time series)
Components	
N_O	Organic nitrogen in the estuary (state variable)
N_I	Inorganic nitrogen in the estuary (state variable)
Pathway flows	
J_{IA}	Inorganic nitrogen flux in wet and dry deposition
J_{IR}	Inorganic nitrogen flux in river inflow
J_{OR}	Organic nitrogen flux in river inflow
J_{TO}	Organic nitrogen flux in tidal exchange
J_{TI}	Inorganic nitrogen flux in tidal exchange
J_P	Organic nitrogen production in the estuary
J_{RM}	Organic nitrogen remineralized in the estuary
J_D	Denitrification
J_F	Nitrogen fixation

4.0 Development of Quantitative Models

The detailed conceptual models for the four stressor classes will be used as a starting point for developing simple quantitative models for each stressor (Figure 19). These models are highly aggregated versions of the detailed models presented above that nevertheless are able to capture the key features of the stressor's action within an aquatic ecosystem. An advantage to these simpler models is that they have fewer parameters and are therefore easier to evaluate. Figure 19 shows the diagram and equations for a quantitative canonical energy systems model of the action of nutrients within an estuary with two equations and one condition giving the mathematical formulation of the model. In this model the ecosystem is represented simply as an order-disorder loop (Odum 1994) that is loaded with nutrients in both ordered (organic) and disordered (inorganic) form from the land, atmosphere, and sea. N_I is the observed quantity of inorganic nutrient in the water and N_O is the nutrient in complex organic form while $N_T = N_I + N_O$ is the total nutrient in the system. Production is represented by J_P and respiration by J_{RM} in the model. Temperature, Tp, is a factor that modifies the rates of both catabolic and anabolic processing. Loading of inorganic nutrient is given by $J_{IR} + J_{IA} + J_T (N_{IB} - N_I)$ and the net rate of processing ($J_{RM} - J_P$) inorganic nutrient by $k_{rm}N_O e^{k(Tp-Tp1)} - k_p N_I e^{k(Tp-Tp0)}$. Denitrification and nitrogen fixation, respectively augment catabolic and anabolic processing and are included as rate processes driven by temperature. Calibration temperatures for the various processes are given as T0, T1, T2, and T3 in the model equations shown in Figure19. This model allows us to determine an important but difficult to measure variable, the processing capacity of aquatic ecosystems for a particular material (pollutant), by using commonly measured values. We hypothesize that processing capacity will be an important variable in explaining stressor-response relationships. Canonical models such as the one presented in this section can be used to develop and test indicators of whole system function. For example, a persistent imbalance in the nutrient used in anabolic versus catabolic processes over several annual cycles may be an indicator of nutrient stress in the ecosystem. An accumulation of N_T in the system or an increase in the total rate of nutrient processing might be early signs of stress.

The development of detailed conceptual models for the individual stressors is a first step toward constructing simulation models to be used in the analysis and prediction of the combined effects of multiple stressors in aquatic ecosystems. The next step in this process is to construct detailed mathematical formulations of stressor action. These models are operationally defined (Bridgman 1928) so that every element of the model (sources, state variables, and flows) is evaluated with a measured quantity. Model evaluation entails specifying values for all components, processes and forcing functions at an initial time, t_0, when the simulation begins. In addition, the forcing functions must be specified for the time period of the simulation. Data on output functions is needed at points in time over the period of simulation for comparison with model predictions. Model coefficients are determined from data on the initial conditions. Key processes and effects in the model are calibrated using empirical measures or theoretical relations that can predict them. Model predictions of the values for state variables are verified by checking them against the data available for the output variables. A verified model is able to predict the observed values for all output variables with reasonable accuracy. Verified models can be used in sensitivity analyses to determine the degree to which output variables of interest are influenced by changes in the forcing functions. The sensitivity of model outputs to the values chosen for model parameters can also be tested. When model output is verified with an independent data set using the coefficients from the original model and the input functions from the new test case, the model is said to be validated. Our confidence in the original model increases with each independent data set successfully validated. The end result of model development will be the ability to forecast ecological conditions in aquatic ecosystems given known loadings of a stressor into the system.



Equations:

$dN_O/dt = J_{OR} + kpN_Ie^{k(Tp-T0)} - k_{rm}N_Oe^{k(Tp-T0)} + J_T(N_{OB} – (N_O/z))$

$dN_I/dt = J_{IR} + J_{IA} - k_pN_Ie^{k(Tp-T0)} - k_dN_Ie^{k(Tp-T2)} + k_{rm}N_Oe^{k(Tp-T1)} + k_fN_Oe^{k(Tp-T3)}$

$J_T(N_{IB} – (N_I/z))$

Figure 19. Canonical energy systems model of excess nutrients in an estuary along with the equations that describe the behavior of this system. Forcing functions, components, and pathways in the model are defined in Table 8. T_i is the initial temperature where the rate function was evaluated and z is the depth.

5.0 Applications

The conceptual models presented in this document need to be mathematically formulated and evaluated with data from existing studies and field work before their veracity can be demonstrated and their full potential realized. Nonetheless, the present state of our scientific knowledge makes us confident that we are on the right track toward developing a set of useful tools to definitively diagnose the causes of impairment in aquatic ecosystems and to allocate effects among multiple causes when more than one has been demonstrated to be active. This document may be of immediate use to scientists in the process of developing initial problem formulations for risk assessments or other analyses in support of the development of a TMDL or other regulatory program for one of the four classes of aquatic stressors covered by our research. Also we have proposed a plan for applying these conceptual models in the development of a stressor-based classification system for use in the regulatory programs needed to address violations of the water quality standards.

5.1 Use of the Conceptual Models in Classification

We will apply our conceptual models and expand the database used in a preliminary classification of estuaries (USEPA 2004) to develop and test stressor-based classification systems (this database tool is under development). We illustrate a possible approach to model-based classification in Figure 20. The aquatic systems to be classified include water bodies and their watersheds. Any system defined in this way can be classified, *e.g.*, a stream reach and its watershed, an estuary and its drainage area, a lake and its watershed. The classes will be based primarily on water body and associated watershed characteristics and they will be developed specifically for particular stressors. Basic information for the pollutant (stressor) will be determined along with the loading rate from the adjacent watershed, watersheds upstream, the atmosphere, and the ocean for estuaries. In addition, we will determine the stored quantity of the pollutant presently residing in the system.

The classification process is first applied for a unit load of pollutant and the expected biologically effective concentrations of the material are predicted for different classes of aquatic systems. Our initial approach to developing a model based classification scheme is as follows: The first step in our stressor-based method for classification of aquatic systems is to place the system to be classified into one of the four canonical models controlling residence time (Figure 9). Once this is accomplished, we will divide the systems into ranges of average temperature and two classes (continuous or discontinuous) based on the effect of seasonality on the way materials are processed. Thus, we will distinguish between boreal, temperate, and tropical systems at this step. If temperature determines the overall rate of metabolic processing of the pollutant within the system, we can use the information on residence time and the temporal pattern of pollutant processing to determine the average residence time of the material in the system and its relevant range of temporal variation. If residence time of the pollutant varies markedly over the area of the system under study, we will divide the system into subsystems and analyze each subsystem separately. We will separate systems into residence time classes (Abdelrhman 2005) based on the chronic dose-response characteristics of the particular stressor. Knowing the temporal pattern of processing and the variation of turnover with time will allow us to partition a system into residence time classes, if necessary. Once we have divided systems into classes based on residence time of the pollutant, we may split the classes using secondary factors that control processing capacity, *e.g.*, the ratio of wetland area to water body area or volume, concentrations of DOC and AVS and other stressor specific measures.

Wetlands have been characterized as nature's kidneys, because they filter impurities from the waters flowing through them. We hypothesize that the presence or absence of wetlands will be a factor of importance in processing pollutants. Also, we will consider other processing factors at this stage based on the particular pollutant being

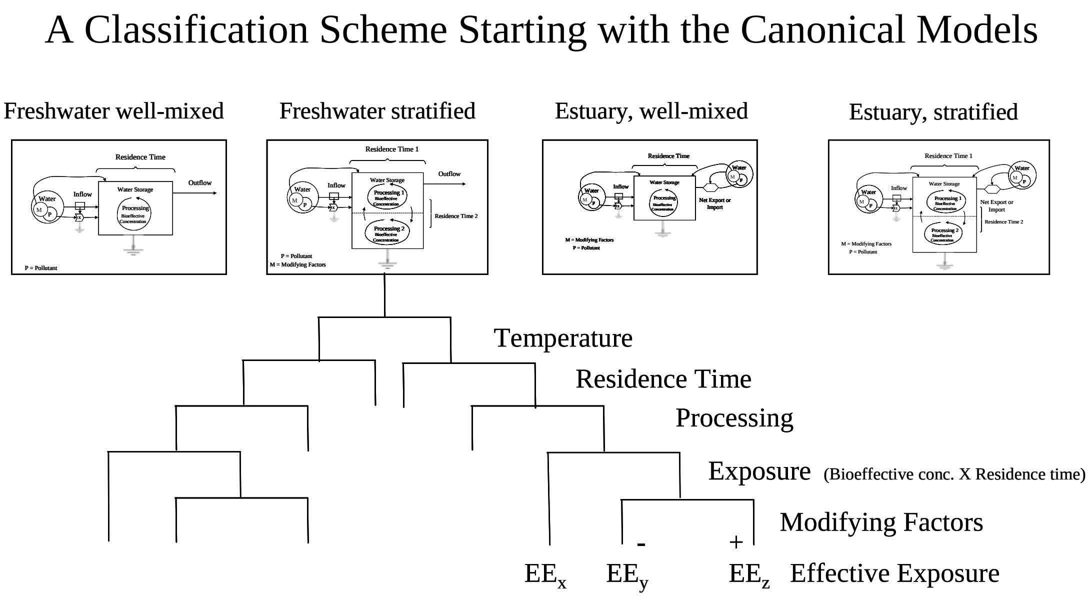


Figure 20. A preliminary classification tree that groups estuaries by effective exposure regimes based on our conceptual model of the factors that control biological impact. This classification based on effective exposure could be applied to any of the stressors with modifications for the particular properties of a given stressor.

evaluated. The result of our analysis of processing capacity will be to estimate the bioeffective concentration of pollutant expected in the class. This value multiplied by the residence time gives the exposure. We will determine the expected exposure-effect relationship for the pollutant from past studies in the literature or from the new data base tool presently under development and predict the effects on biological output variables from the exposures determined for each class.

Next, we will consider the effects of type B modifying factors on the biological impacts expected in particular classes. We will group these modifying factors according to their effects on the pollutant. Those with a positive effect (decrease response) and those with negative effects (increase response) will be combined to estimate the net effect on the biological response. Once we have determined a positive or negative effect, we will adjust the exposures calculated above to determine effective exposures in the system containing modifying factors. If no type B modifying factors are present, the exposure value determined above is the effective exposure and it passes directly to the bottom line in Figure 20. We hypothesize that effective exposure will characterize sets of aquatic systems where similar biological effects will be observed for a unit load of pollutant. Classes will be derived by examining the effective exposure groups to see what combinations of properties are represented by systems in the group.

Next we will apply the actual loads entering the aquatic systems and determine the effective exposures. We will plot the observed values for the biological output variables against the exposures to construct an exposure-effect curve for the pollutant. We will compare this relationship to the one expected from past laboratory and field studies. We expect eco-system classes to plot as a family of curves on the exposure-effect plane or as a single curve on the effective exposure-effect plane, i.e., after type B modifying factors have been considered and the exposure-effect relationship has been normalized. We may also need to consider factors that shift a particular response variable on the ordinate to adjust the y-axis values and further tighten the effective exposure-response relationship. Figure 21 is a hypothetical

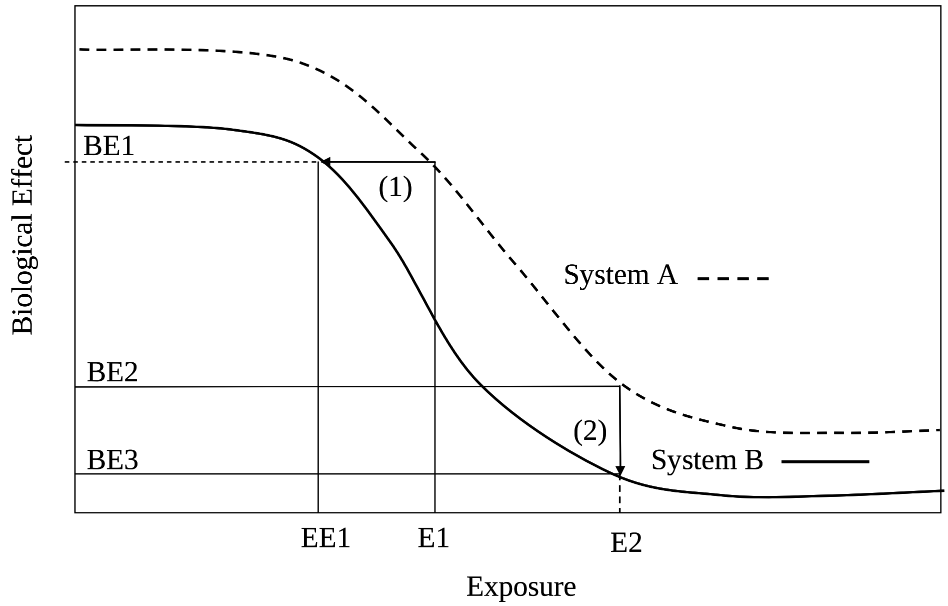


Figure 21. Normalization of the exposure-effect relationship by adjusting both the x and y axis. (1) A type B modifying factor shifts the response variable on the exposure axis. (2) At a standard exposure, different levels of the biological effect variables characterize aquatic ecosystems of different kinds.

example showing how normalization procedures might work. Modifying factors that mitigate the biological response shift the exposure-effect relationship toward lower effective exposure (EE1) for a given level of effect. If the factors controlling the biological response variable can be determined, all effects can be expressed relative to the same baseline exposure-effect relationship (2 in Figure 21). If these relationships can be demonstrated, managers might allow greater loading in a class of aquatic systems that is less sensitive to the pollutant and still attain a given level of condition that is deemed to be acceptable by society

5.2 Use of the Conceptual Models in Risk Assessment Problem Formulation

The general conceptualization of the factors controlling the action of stressors in aquatic systems as an exposure-effect problem follows the approach taken in ecological risk assessment. Our model given in terms of residence time, processing capacity, and modifying factors provides the means to determine effective exposure and it can be of direct use to those performing risk assessments related to the development of TMDLs. If effective exposure can be successfully related to observed effects by accounting for the factors that alter this relationship, we will have a robust method for predicting the effects of pollutant loading in aquatic systems.

When the evaluated and verified energy systems models for the four classes of stressors are simulated, the predicted temporal variations and spatial patterns of risk and effect can be examined. Energy systems models also help organize thinking about a class of stressors and they can serve as a checklist of the important components and processes governing stressor behavior. The simulation of emergy flows in a detailed model of a stressor acting on an ecosystem allows determination of the ecological significance of a given change in the system, which combined with the probability of that change (risk) gives an accurate measure of ecological importance, which is a unified and comprehensive measure of the environmental impacts of a stressor that can be used in decision-making (Campbell 2001b).

The detailed models presented here strive to capture the important processes and interactions of the system components that control the action of multiple reinforcing and agonistic stressors in aquatic ecosystems of different kinds. To the extent that these models are successful in predicting ecosystem behavior under stress they should be useful in identifying and developing specific indicators that could facilitate the diagnosis of the causes of biological impairment in cases where both single and multiple stressor interactions are in play.

6.0 Glossary of Words and Concepts

aquatic ecosystems are material and energy processing units organized by thermodynamic laws and principles that are contained in water bodies of various kinds, which are arrayed over the surface of the earth.

canonical an adjective used to describe the standard form of an energy systems model (an equation or system of equations), especially when the model is simple.

emergy is the available energy of one kind previously used up, directly and indirectly, to make a product or service. It's unit is the emjoule.

energy and emergy signature Energy and material distributions are highly variable in space and time, and thus there are many combinations available to support ecological organization. The energy and material inflows available at a location, when plotted in order of increasing transformity, make up the energy signature of that place. If each energy in the energy signature, is multiplied by its transformity the resulting plot is the emergy signature of the place. There is some indication that qualitative and quantitative differences in the emergy signature correspond to ecosystems of different kinds that have different ecological norms, *i.e.*, expected values for *empower,* the flow of emergy per unit time.

fundamental watershed is a network of ecosystems arrayed on the landscape and linked by water flows that debouch into the open sea or into one of the Great Lakes. These watersheds are commonly the largest system within which wetlands, stream segments, lakes and estuaries must be managed to ensure that limits established for pollutants will be effective. An exception to this rule occurs when atmospheric or oceanic inflows of a pollutant are large.

maximum power (empower) principle states that ecosystems adapt and evolve to use available energy and materials in a manner that maximizes empower in the ecological network. Evolutionary competition puts survival pressure on ecosystems to adapt to use their inputs in a manner that maximizes empower in their ecological network. By definition a natural system is one that has had the time to adjust its structure and function to attain optimum efficiency for maximum power. P and R are often but not always balanced in natural systems that are adapted to their inflows. The theory predicts that system indices will approach a dynamic equilibrium that maximizes power under a given emergy signature.

Paracelsus Axiom: the dose makes the poison. The three pollutant stressors of concern to us, *i.e*., nutrients, toxic substances, and suspended and bedded sediments are hypothesized to be materials that stimulate ecosystem production when present in the quantities to which life has adapted over evolutionary time scales. The frequency of occurrence (relative rarity) of any material in nature is different and the opportunity for organisms to develop the capacity to process any material is proportional to the probability of encountering a given concentration of that substance in nature. Thus, in general, organisms have a greater ability to process more common substances and less for rarer ones (Genoni 1997). Of course there is considerable variation in the concentrations of materials found in nature and there is concomitant variation seen in the life forms that have developed different abilities to process different concentrations of these substances.

pollutant A pollutant is any substance that is present within a system outside of its normally expected range, usually a gaseous, chemical, or organic waste.

state variable Any stored quantity within a system which may be plotted over time. In an energy systems model, there is one 1st order differential equation for each state variable.

stress can be defined as a change in or perturbation of the normal (long term or natural) functioning of a system. Often stress causes energy drains to development within the system. Stress can also cause certain energy flows to exceed there normal ranges. In general, stress is the result of a change in the emergy signature of the place. The usual implication is that this perturbation or change is the result of some human activity. A change in the emergy signature of a processing unit sets in motion the normal adaptive mechanisms of an ecosystem that operate to create a new network design that will maximize power under the changed conditions, if the new conditions become permanent, or return the system to its normal state of operation, if the perturbation is transitory. Ecosystems can adapt to recurring natural perturbations, so that the recurring pulse becomes necessary to the health and preservation of the ecosystem, *e.g.*, fire climax in southeastern U.S. pine forests (Odum 1971). In this case the pulse or perturbation is a stress to which the system has adapted. In cases where ecosystems have adapted to recurring or chronic stress they may be more resistant to the effects of a similar but unnatural change in the emergy signature.

stressor In the context of ecological systems the word "stressor" may be defined as any force that results in an injury to a system (i.e., a decrease in total system empower) often by over use or exertion of some part of the system. Stressful forces can be exerted by physical, chemical, or biological entities and all three are encompassed by the term "stressor".

transformity solar transformity is the solar emergy required to make a joule of a service or product. It is the solar emergy required for a product divided by its heat content in joules. The units of solar transformity are solar emjoules per joule (sej/J).

unit model is an energy systems model or other mathematical formulation applied in every grid cell of a spatial analysis or simulation (some components may be 0 in any particular cell).

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