PROFILE

Risk Assessment as an Environmental Management Tool: Considerations for Freshwater Wetlands

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ABSTRACT /This paper presents a foundation for improving the risk assessment process for freshwater wetlands. Integrating wetland science, i.e., use of an ecosystem-based approach, is the key concept. Each biotic and abiotic wetland component should be identified and its contribution to ecosystem functions and societal values determined when deciding whether a stressor poses an unreasonable risk to the sustainability of a particular wetland. Understanding the major external and internal factors that regulate the operational conditions of wetlands is critical to risk characterization, Determining the linkages between these factors, and how they influence the way stressors affect wetlands, is the basis for an ecosystem approach. Adequate consideration of wetland ecology, hydrology, geomorphology, and soils can greatly reduce the level of uncertainty associated with risk assessment and lead to more effective risk management. In order to formulate effective solutions, wetland problems must be considered at watershed, landscape, and ecosystem scales. Application of an ecosystem approach can be greatly facilitated if wetland scientists and risk assessors work together to develop a common understanding of the principles of both disciplines.

In the United States, regulations such as the Clean Water Act, CERCLA (Comprehensive Environmental Response, Compensation, and Liability Act), and FIFRA (Federal Insecticide, Fungicide, and Rodenticide Act) make risk assessment an important part of wetland management at the federal, state, and local levels. Many different approaches have been developed to assess ecological risks (US EPA 1992, Suter 1993), but none of these were developed specifically for application to freshwater wetlands. Each has been molded to assure their implementation for risk assessments mandated by law and regulation. For example, some strategies give explicit instructions for evaluating physical stressors that may have impacted a wetland as a consequence of changes in land-use practices, but they offer no guidance for assessing exposure of biota to chemical stressors. Others fail to address important parameters that are unique to freshwater wetlands. These methods could be improved by expanding them into a more comprehensive, ecosystem-based approach.

It is necessary to place the risk assessment process into an ecosystem context in order to identify key linkages between stressors and wetland responses and to develop the most effective risk management strategies. This involves understanding the four principal factors (ecology, hydrology, geomorphology, soils) that determine the structural and functional characteristics of wetlands and then using this information to identify where, when, how, and to what extent stressors are, or could be, causing adverse effects, i.e., effectively integrating wetland science into risk assessment.

Despite the inseparable linkage between wetland science and risk assessment, there is often a lack of communication and understanding between the wetlands research community and the risk assessment community. Wetland science and ecology are often quite foreign to risk assessors and, conversely, the guiding principles for risk assessment are often completely unknown to wetland scientists. This has deterred effective integration of wetland ecosystem analysis into the risk assessment process. Historically, there has been little inherent need for communication between these two disciplines. Today, however, the situation is quite different. The future of many freshwater wetlands will be determined by the outcome of environmental risk analyses, with or without the proper integration of wetland science and ecosystem-based assessment.

Planning and conducting environmentally sound risk analyses depends, in part, on bridging the gap between wetland science and risk assessment. The gap may be more apparent in some instances than others but in most cases the potential for ineffective risk management exists. It is essential that all of the individuals contributing to the risk assessment process have a common understanding of some basic principles from both disciplines. This paper was written to facilitate that common understanding.

KEY WORDS: Ecological risk assessment; Freshwater wetlands; Environmental pollution; Chemical stressors; Physical stressors; Biological stressors

Important Principles of Wetland Science

Recognizing Baseline Conditions

One of the most fundamental principles is recognizing that most of the remaining freshwater wetlands in the United States are altered from their natural state because of changes in hydrology and surrounding land use. Thus, what is perceived as the baseline or normal condition for a wetland may be something quite different than the true baseline that would exist in the absence of disturbance. In many cases the fact that an altered state exists, and the direct linkage between human activity and wetland modification, may not be readily apparent. For example, surface and groundwater extractions and diversions for urban and agricultural water supply have affected the hydrology of many wetlands and changed their water quality, vegetation, and animal life subtly, but substantially over a period of years (Thompson and Merritt 1988, Lemly 1994). Development of wetlands for other land uses has fragmented large wetland complexes into small remnant wetlands that cannot maintain their original functions of hydrologic flux and water storage or habitat for wildlife (Frayer and others 1989, Moore and others 1990). Dredging and channelization for navigational purposes has disrupted the hydrologic balance necessary for riparian wetlands to effectively intercept and moderate flows and water-quality degradation associated with stormwater and agricultural runoff (Lowrance and others 1984, Philips 1989, Richardson 1994, Culotta 1995). These physical alterations constitute a chronic stress that influences the way wetland ecosystems respond to new or added stress. On a regional and national scale, physical alterations are having a far greater impact on the integrity of wetlands than are chemical and biological threats.

Identifying the baseline set of conditions is the starting point for determining risks and threats from stressors that impinge on wetlands. In identifying the baseline, it is important to remember that the salient characteristics of wetland ecosystems are embodied in the integration of local climate with geology and hydrodynamics. The results of this integration over geologic time are evident in the soils, vegetation, and biota. Thus, the wetland ecosystem is a result of the interaction of specific abiotic factors (climate, geology, and hydrodynamics) and various organisms over a long period of time, as modified by recent human disturbances. However, can abiotic traits alone be used to determine what processes and functions a specific wetland may have? The answer is: only in a general context. For instance, a depressional wetland would not be expected to be involved in carbon transport, to actively transport pollutants or nutrients out of the system, or to be highly productive. However, the depression could have a groundwater source rich in nutrients. In that case its productivity might be high, and thus it could act as an efficient buffer or transformer of chemical stressors. Additionally, the amount of organic matter in the soil would influence microorganisms for decomposition and other soil reactions, benthic detrital macroinvertebrates, vegetation types, and potential for exposure of wildlife to food-chain contaminants. This example illustrates the need to consider both biotic and abiotic processes that impinge on wetlands when identifying baseline conditions, i.e., use of an ecosystem-wide assessment.

Climate, Geohydrology, and Soils

One of the most basic and important factors to consider is climate. Regional climate influences not only temperature, which mediates many biological processes within the ecosystem, but also the amount, form, and timing of precipitation. For wetlands to occur, there must be excess water, generally coming as runoff from upland drainage areas. In wetlands, evapotranspiration tends to dominate the water balance and during some periods of the year it may exceed precipitation so that a water deficit develops, leading to seasonal wetlands. Depending on the climate and water balance, wetlands may be highly evident during dormant growth periods and less so during the growing season. Conversely, if water excesses occur primarily during the summer months the wetlands will be most evident during the growing season. Time of year can thus be an important consideration in identifying and characterizing the wetlands as well as conducting the risk assessment itself. Climatic data are generally available through state and federal agencies, and they provide valuable clues as to the temporal nature of wetlands within a region.

How these climatic variables are expressed on the landscape depends in significant part on regional and local geomorphic setting and soils. Of particular importance to wetlands is the way in which these external factors interact with each other to determine the hydrologic setting. For example, precipitation, evapotranspiration, and geomorphology all interact to regulate runoff and thus strongly influence hydrologic controls. Wetlands provide a critical link between uplands and aquatic systems (streams, rivers, and lakes) whether the connection is across the surface or through the groundwater. It is this critical linkage that in part determines the importance of wetlands as biogeochemical filters or transformers buffering flows from uplands to aquatic systems. In addition, it is this critical linkage that often places wetlands at risk from seemingly isolated stressors and makes them an important component of many toxicological evaluations.

Geomorphology is also a critical factor to consider. It is the landscape position or geomorphic setting that accommodates the runoff and storage of water (Brinson 1993a). As a consequence, geomorphology generally has a strong influence on the potential for chemical stressors to be transported, stored, and cycled in a wetland. There are depressional, riverine, and fringe categories of geomorphic settings, each of which has unique characteristics that are important to risk assessment.

Depressional wetlands include such landforms as kettles, potholes, vernal pools, and Carolina bays. They frequently occur high in drainage systems; thus they typically depend heavily upon local precipitation when compared to other geomorphic settings. In climates where evapotranspiration exceeds precipitation, depressions tend to be dry much of the time, for example, vernal pools, or they depend upon groundwater (Zedler 1987, Brinson 1993a). In climatic regions where runoff (precipitation minus storage and evapotranspiration) is greater than zero for a significant portion of the year, depressions may accumulate sufficient peat to develop a domed topographic relief. These types of wetlands receive their water from precipitation as contrasted to groundwater or overbank flooding. Peatlands may cover large areas such that the peat substrate dominates the movement and storage of water, the mineral nutrition of the plants, and patterns of the landscape. Extensive peat formations caused by paludification across the landscape may develop surface patterns that are independent of the underlying topography. As a consequence, there is a gradient from the headwater ombrotrophic wetlands with diffuse outlets to ones further downstream with fenlike characteristics (Moore and Bellamy 1974, Siegel and Glaser 1987).

Riverine wetlands form as linear strips parallel to streams but are generally separated from the stream channel by natural levees. A riverine wetland may occupy extensive floodplains adjacent to large rivers and high-order streams but may be very small or nonexistent in the case of low-order streams (Theriot 1988, Hook and others 1994). Hydroperiods range from short and flashy in low-order streams to long and steady in higher-order streams. The slope of the stream channel determines whether a given section of the floodplain is predominantly erosional or depositional. Channel morphology can influence how adjacent wetlands interact with upgradient sources of contaminants and other stressors.

Freshwater fringe wetlands are restricted to freshwater tidal zones associated with estuaries. These types of wetlands are generally riverine (alluvial) in nature but some may be headwaters (nonalluvial). The latter occur in small drainages that feed into rivers near estuaries. The geomorphic setting provides important information for understanding how stressors will impinge on a particular wetland.

It is also important to know the hydrodynamics of wetlands in order to evaluate transport and potential exposure of animal and plant life to chemical stressors. The source of water for freshwater wetlands may be precipitation, groundwater discharge, surface or near-surface inflows, or any combination of these. Many depressional wetlands receive their water from precipitation runoff. These types of wetlands occupy depressions in the landscape that are typically above the water table. They are generally separated from the water table by a layer of relatively impermeable soil that restricts the rate of water movement downward through the soil. Therefore, the dynamics of the water table are vertical; it moves up when it receives runoff and down primarily due to evapotranspiration. Depressions generally have no inlets or outlets or, if they are present, they receive or drain water only during or after storm events. They tend to be disconnected hydrologically from the surrounding landscape and the substrate below the restrictive layer. However, during high water events some water may spill out of the depression beyond the restrictive layer and come into contact with the substrate below. For example, research in Florida has shown that the cypress domes may be more interconnected than originally thought (Riekerk 1993). Depending on the size, geomorphology, and regional location, depressional wetlands may develop distinct zonational vegetation and structural patterns in relation to the time and duration of inundation and fluctuation of the water table. Nutrient input into these systems is primarily by precipitation. On a relative scale they tend to have low primary productivity. However, productivity may vary with the geology, climatic conditions, and types of soils and vegetation that develop. Some depressional wetlands receive groundwater in addition to runoff from precipitation. If the groundwater table intersects the slope at or within the depression, water enters from below as well as from runoff.

Groundwater may enter wetlands or create wetlands on slopes where the water table intersects the soil surface. Such areas can best be visualized as seeps or sprmgs. However, relatively large wetlands can occur on slopes. If groundwater enters a wetland, it has been in contact with the mineral content of an aquifer or soil. Depending on the time of contact and the composition of the lithology, such water normally has higher mineral content than water derived from precipitation. Consequently, plant communities in wetlands that receive groundwater discharge tend to be more productive than rainwater wetlands (depressions). Furthermore, the hydrodynamics of the system are apt to be more stable than in precipitation-driven wetlands (i.e., dry downs may not be as severe and as rapid). The dynamics of the water table in these types of wetlands tend to be vertical in relation to water inputs and outputs.

The source of water in riverine wetlands may be from overbank flooding, groundwater, and precipitation. The dominant water source is not always evident even after extensive exploration. A study in the Piedmont of South Carolina showed that a fourth-order stream received periodic overbank flooding on average about three times per year during the dormant season. However, during the growing season, the wetland was driven entirely by precipitation (Hook and others 1994). In contrast, in a fifth-order stream in coastal Georgia, water came from overbank flooding during the growing season as well as the dormant season, but between major rainfall events in the watershed, precipitation and groundwater influenced the wetland to varying degrees depending on topographic relief. Piesiometric studies showed that microtopography had important influences on drainage patterns and sources of water between flood events (Saul 1995).

The water in a flood plain tends to flow unidirectionally downstream but, depending on topography and the presence of depressions in the floodplain, it may take on vertical dynamics when the river is not in flood stage. In the lower reaches of rivers influenced by tides, the fringe wetlands may be subjected to bidirectional flow similar to those in estuaries. The variation in hydrodynamics among wetlands and within localities of a wetland must be carefully considered if risk assessments are to successfully identify key transport and exposure pathways to biota.

When a wetland has two or more water sources, it can be difficult to separate their relative contributions. For riverine systems, records of time, frequency, depth, and duration of overbank flooding are necessary to evaluate the extent of individual contributions, effects of overbank flooding on the wetland, and how chemical stressors may be delivered, retained, and transported. Some rivers may be gauged to monitor discharge (a function of the Geological Survey in the United States). Such records are invaluable for ecological and toxicological studies and evaluation of various wetland functions. In the absence of such records, stream flow or peisometric studies are necessary to quantify many characteristics of a wetland. Problems arise in determining how long monitoring must have occurred to be useful. For example, a 38-year record for one wetland in eastern South Carolina demonstrated that depending on which three- or five-year period was selected for measurement, the site could either be classified as a wetland or nonwetland using jurisdictional criteria (Hook and others 1994).

Wetland soils and soil-plant associations can also yield information that is important to risk assessment. Many, if not most, counties in the United States have surveys of the soils. The surveys contain general traits that will help determine the potential characteristics of a specific wetland. They identify soils by series and drainage class, and provide information on productivity, amount of organic matter, general information on the degree of soil saturation or flooding, times of hydroperiods, and occasionally the duration of hydro events. In addition, if the wetland is forested, the data bank may include information on site index for various tree species. This provides another clue as to the relative productivity of the wetland (site index is the height that a tree will reach at a specified age and has proven to be a very good measure of the productivity of the site). Again, these are general traits for a soil series, but they provide the researcher with a fairly extensive array of characteristics about the wetland site in question. It is necessary to verify whether the soil information is truly indicative of the site by examining the soil profile and other salient characteristics of the site. Is the vegetation natural or has it been altered? Has the hydrology been altered by drainage and blockage of drainages? Assistance with this process can usually be found close by. For example, in the United States, the Department of Agriculture, Natural Resources Conservation Service, generally has offices in each county with trained personnel that can help interpret soil survey information and sometimes assist with actual field checks. County agents and university extension personnel also may be available to help interpret the data or provide guidance on where to seek help.

Biological Processes and Ecosystem Functioning

Integrating biotic and abiotic factors is the focal point of an ecosystem approach to risk assessment (Figure 1). Ecosystem responses to external, abiotic factors yield a complex set of interactions between the biota (organisms, species, populations, communities), the critical processes they perform (photosynthesis, microbial action, decomposition, etc.), and the way these organisms and their processes are expressed through ecosystem functions (production, biomass accumulation, biogeochemical processes, etc.). To a large extent, the complex structure and function of wetlands reflect the divergent properties of their biota. Most wetlands are dominated by a flora of vascular plants that

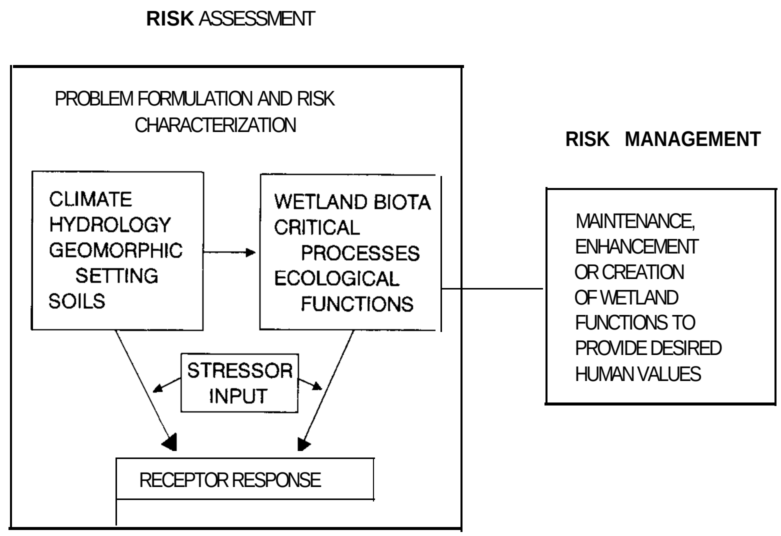

Figure 1. Conceptual framework for an ecosystem approach to risk assessment in freshwater wetlands. The integration of wetland science is important in problem formulation and risk characterization.

are adapted to a greater or lesser extent to flooded conditions but that are, in most respects, structurally and physiologically similar to their terrestrial relatives. Yet, wetlands may also have features similar to deep-water aquatic ecosystems, including sediment biogeochemical and biotic processes mediated through predominantly anoxic conditions and aquatic food webs of algae, invertebrates, and vertebrates.

Although wetlands show structural and functional overlap with terrestrial and aquatic systems, they often serve as the interface between these two systems. Wetland structure, internal critical processes, and ecosystem functions are sufficiently different from terrestrial and aquatic systems to require a knowledge base specific to wetlands. This knowledge is essential for planning and conducting a risk assessment using an ecosystem approach. Key biological processes and ecosystem functions that must be considered are discussed here.

Wetlands can best be viewed as complex temporal and spatial mosaics of habitats with distinct structural and functional characteristics. Variation in vegetation structure represents one of the most striking examples of spatial and temporal pattern in wetland habitat. Depending upon the type of wetland, the system may be dominated by emergent herbaceous or woody macrophytes with open water relegated to relatively small areas among blades of emergent plants or to small open patches within the emergent stand. However, regardless of the dominant vegetation, horizontal zonation is a common feature of wetland ecosystems, and in most nonriverine wetlands, relatively distinct, often concentric bands of vegetation develop in relation to water depth. Bottomland hardwood forests and prairie pothole wetlands provide excellent illustrations of zonation in two very divergent wetland types (van der Valk 1989, Mitsch and Gosselink 1993).

Wetlands may display dramatic temporal shifts in zonation patterns in response to changing hydrology. Entire systems may shift between predominantly emergent and open-water zones. In periods of little or no water, some wetlands may temporarily become almost terrestrial in form and function. Yet, the same system in other years or in other seasons of the same year may be flooded to the extent that the system becomes, in small or significant part, aquatic in nature. Temporal patterns are in fact important characteristics of many wetland types. Seasonal cycles are a major feature of floodplain forests, for example. High-order stream systems are typically flooded during winter and spring periods of high streamflow and bankfull discharge but are typically dry by mid- to late summer due to drainage and evapotranspiration. Longer-term cycles are a major feature of prairie pothole wetlands that undergo dramatic, more or less cyclic changes in response to a variety of environmental factors including water-level fluctuations and grazing (van der Valk 1989, Mitsch and Gosselink 1993). As a result, these systems may exhibit major year-to-year variations in vegetation structure and distribution and in the relative importance of vegetated and open-water zones.

Given the complex temporal and spatial structure of wetlands, it is important to understand the critical habitat characteristics that exert control over major aspects of wetland function. These are the trigger points at which stressors operate to disrupt wetland processes and cause adverse effects. In comparison to our understanding of vegetation dynamics, there is relatively little information regarding the influence of vegetation on wetland environments. However, it is clear that vegetation structure has dramatic effects on the physical, chemical, and biological attributes of wetland habitats. Wetland macrophytes affect environmental attributes and biogeochemical processes in a variety of ways, including reducing light available to algae and/or submersed macrophytes, reducing water temperatures (due to shading), reducing circulation of the water column with resultant effects on gas exchange and material transport, increasing inputs of detrital carbon, enhancing transport of gases to and from the sediment (rhizosphere), and either reducing or enhancing mineral uptake and release. In addition to direct and indirect effects on biogeochemistry, vegetation structure is one of the most important factors affecting food-web structure and bioenergetics in wetland ecosystems. Despite the obvious oversimplification, it is useful to distinguish three broad classes of primary producers in wetlands with regard to food web dynamics: (1) emergent macrophytes, (2) submergent and floating leaved macrophytes, and (3) planktonic and periphytic algae.

Emergent macrophytes are similar to terrestrial plants in that their biomass is high in structural components such as cellulose and lignin. Their leaves and stems have the low nutrient content and high carbon to nitrogen ratios typical of terrestrial plants of similar growth form, and their food value is relatively low. In general, herbivory on emergent macrophytes is very low and most of their production is transferred to the detrital pool. Nonetheless, the impact of herbivore activity may at times be extensive. For example, the complete destruction of emergent vegetation by muskrats in freshwater marshes has been documented numerous times (van der Valk 1989). However, even during these events, muskrats prefer roots and shoot bases and rarely consume leaves and stems of emergent macrophytes. These tougher materials are instead discarded or used to build lodges, thus entering the detrital pool. Due to the prevalence of structural compounds such as cellulose and lignin, detritus derived from emergent macrophytes is relatively resistant to digestion or decomposition, especially under anaerobic conditions. Nutrient content is even lower and carbon to nitrogen ratios higher than in the living plants and, as a result, decomposition frequently requires nutrient subsidy.

In contrast to emergent macrophytes, submergent and floating leaved macrophytes have substantially less structural material. Their tissues generally have higher nutrient content (percentage) and lower carbon to nitrogen ratios. Due to their higher nutrient content, the food value of submergent and floating leaved plants can be relatively high in comparison to emergent macrophytes. Herbivory on submergent and floating leaved macrophytes is highly variable, but in comparison to emergent macrophytes, a larger portion of their production may be consumed by herbivores rather than being transferred directly to the detrital pool. The principal herbivores consuming submergent and floating leaved macrophytes include waterfowl, macroinvertebrates, and fish. Due to the relative paucity of structural compounds, detritus derived from submergent and floating leaved macrophytes is relatively labile and relatively easily digested or decomposed.

Planktonic and periphytic algae, of course, have very little structural material. Their tissues have very high nutrient content (%) and low carbon to nitrogen ratios. Algae have very high food value and are easily consumed and digested by a wide range of herbivores including microzooplankton, macroinvertebrates, and fish. Although grazing rates vary, a significant portion of algal production in wetlands is consumed by herbivores rather than being transferred directly to the detrital pool, significantly more than in the case of aquatic macrophytes or even submergent macrophytes. Detritus derived from algae is very labile and easily digested or decomposed.

Most freshwater wetlands are assumed to be dominated to a greater or lesser extent by a food chain that is weblike and detritus-based (Mitsch and Gosselink, 1993). However, it is clear that spatial heterogeneity in vegetation structure can result in a mixture of detritus-based and producer-herbivore based food webs. For example, emergent macrophytes dominate production in the emergent zone of freshwater marshes. Most of this production could be expected to enter the detrital pool, with relatively little consumption by herbivores. In contrast, phytoplankton dominate production in the open-water zone of freshwater marshes, and much of their production would probably be consumed directly by herbivores. In wetland zones dominated by submergent and floating leaved macrophytes, these macrophytes and their attached algae might both contribute significantly to total production. In either case, a significant proportion of the total production would probably be consumed directly by herbivores. Given these relationships, it is probably better to characterize the food webs of freshwater marshes and most other wetlands not as either detritus-based or producer-herbivore based but rather as complex mosaics of habitats with distinct food webs. It is important to recognize that seasonal as well as longer-term shifts in habitat mosaics and in their associated food webs and biogeochemistry are fundamental aspects of the character of many wetland ecosystems.

Understanding the major external and internal factors that determine the operational conditions of wetlands is critical to risk characterization. Determining the linkages between these factors, and how they influence the way stressors affect wetlands, is the basis for an ecosystem approach (Figure 1). Adequate consideration of wetland ecology, hydrology, geomorphology, and soils can greatly reduce the level of uncertainty associated with risk assessment by providing multiple lines of evidence. These four ecosystem components embody the key indicators of wetland status and should be a major focus of risk assessment regardless of whether or not the assessment end points are biologically or ecologically based. Applying an ecosystem-wide investigation allows risks to be characterized within the context of functional integrity, i.e., viability and sustainability. This can lead to more effective risk management.

Important Principles of Risk Assessment

Terminology

Risk assessment for freshwater wetlands may focus on chemical (nutrients, contaminants, etc.), as well as nonchemical issues. As used here, risk assessment is the estimation of hazard or threat posed by stressors (chemical, physical, or biological) to the biotic and/or abiotic components of wetlands. Ecological risk assessment connotes a biologically driven assessment focused on relationships between the biotic and abiotic components. In either case, risk estimates may be quantitative or qualitative and may be expressed as levels of hazard (high, medium, low, etc.) according to definitions given by the investigator or expressed in probabilistic terms (i.e., there is an 80% chance that some event will occur, such as, certain biota will die, waterborne concentrations of nitrate will exceed an acceptable threshold, microbial mineralization will be reduced significantly, etc.). In practice, chemical and physical stressors generally impinge on wetlands simultaneously. In evaluating the effects of chemical, physical, and biological stressors, various issues must be resolved during the problem-formulation and risk-characterization phases of the risk assessment process. For adequate technical support, management and policy input must be clearly positioned prior to the risk analysis activities associated with exposure and ecological effects assessment, including the resolution of questions revolving around two interrelated issues focused on data interpretation (performance-based versus criteria-based practices) and the distinction among risk analysis (characterization), risk assessment, and risk management (Figure 1).

Evaluations of wetlands may incorporate the concepts of performance-based or criteria-based practices to varying degrees, depending on whether the system being assessed is a naturally occurring or a constructed wetlands, and the regulatory context that may be associated with the risk assessment. Performance-based criteria are those that specify some design-focused criteria for evaluating wetlands; for example, a naturally occurring or constructed wetland may be considered an effective remediation measure if it decreases heavy-metal concentrations in mine tailings runoff by 80%. Criteria-based evaluation practices most frequently assess wetland water quality by some numeric value developed as a consequence of a regulatory objective, e.g., water discharged from a remediation wetland must meet the drinking water standards for heavy metals. Regardless of the data sources being used in the risk assessment (e.g., historic data or data derived from designed studies), technical data collections must be applied within the data quality objectives that are developed from either performance-based or criteria-based needs.

The relationship of risk analysis, risk assessment, and risk management activities to wetlands may be markedly different, especially as they relate to a technical characterization of wetland functions versus a more risk assessment-like consideration of wetland values. The role these potential differences play in evaluating threats and impacts of anthropogenic activities on wetlands depends upon how clearly these terms are distinguished and mutually understood by the risk assessor and risk manager.

Wetlands are generally considered to have functions centered around hydrologic flux and storage, biological productivity, biogeochemical cycling and storage, decomposition, and community/wildlife habitat (Richardson 1994). Hydrologic functions are characterized by capacity and throughput (input/output relationship), which may define a wetland as a water source or water sink. Habitat functions of wetlands may be nested with subsets of functions related to biological processes such as decomposition, biological productivity, and biogeochemical processing, but these all directly reflect the biological components of wetland structure. For example, wetland vegetation clearly is critical and plays a major role in maintaining biodiversity and species-critical functions such as reproduction, feeding, and dispersal. While not without technical disagreement, wetland function is relatively easy to address within risk analysis, but wetland values are better characterized as assessment end points wherein societal and policy influences become critial to their definition. Wetland values refer to the benefits obtained by society from wetland functions. For example, hydrologic flux and storage is a wetland function. Human values associated with this function include flood control and the economic and recreational benefits derived from hydrologic flux and storage (Table 1).

Without question, understanding the functions and

Table 1. Some important ecosystem functions and human values of freshwater wetlands[a]

Functions	Values
Hydrologic flux and storage	Flood control and protection Erosion control Water supply Visual-cultural
Biological productivity	Timber production Shrub crops (cranberry and blueberry) Food production (shrimp, fish, ducks) Historical and cultural resources
Biogeochemical cycling and storage	Sediment control Nutrient removal Wastewater treatment Water quality
Decomposition	Medicinal (streptomycin) Wastewater treatment Water quality Nutrient supply and regulation
Community/wildlife habitat	Recreation Hunting and fishing Preservation of flora and fauna (refuge) Threatened, rare, and endangered species

[a]Adapted from Richardson (1994)

Table 2. Examples of assessment end points for freshwater wetlands

Assessment end point	Significance
Hydrological	
Maintain natural supply of water to wetland	Key to maintaining proper level of hydration
Provide sediment control	Reduces turbidity and sediment loading to nearby waterbodies
Geomorphological	
Maintain bank stability	Reduces erosion of stream and river banks
Ecological	
Maintain level of primary productivity	Underpins food web stability

values is a pivotal part of risk assessment for freshwater wetlands. This function-value relationship provides an important conceptual framework within which the risk assessor and risk manager can formulate operational goals and objectives. The principal truths underlying this relationship are: (1) All wetlands are not of equal function or equal value on the landscape, (2) a restored or newly constructed wetland may or may not be equal to a natural wetland in terms of ecological function or value, (3) wetland ecosystem functions and values are coupled to other systems on the landscape, and (4) wetlands often provide functions and values beyond their boundaries and far from adjacent ecosystems (Richardson 1994).

The terms from wetland science and related wetland assessment disciplines must be clearly defined and distinguished as assessment end points or measurement end points if the wetland scientist and risk assessor are going to communicate and become effective resource managers. Assessment end points are the functions and associated values to be protected, enhanced, or created through risk management (Tables 1 and 2, Figure 1). For example, an assessment end point might be to improve water quality by controlling sediment and reducing suspended solids below some threshold concentration. Measurement end points are the specific parameters that indicate whether or not an assessment end point has been achieved. For example, a measurement end point for sediment control assessment could be the amount of suspended solids in the outgoing water.

Similarly, the concepts of threats and impacts to wetlands must be established within an ecological risk setting. Within a risk assessment context, threats are anthropogenic activities (planned or unplanned) that have the potential to cause adverse effects, while impacts are effects that resource managers typically characterize as adverse (US EPA 1992, Suter 1993). Risk management objectives must be adequately characterized in order to clearly identify measurement end points that will identify (or eliminate from further analysis) differences between wetlands at risk and their reference environments. In order to develop cost-effective risk analysis programs, the concepts of function and value, as well as threat and impact must be consistently defined by wetland scientists and those in the risk assessment community. Wetland risk assessors and risk managers must clearly define assessment end points to assure that measurement end points driving technical activities support their wetlands risk assessment needs.

An ecosystem-based approach gives the risk assessor and risk manager considerable flexibility in how to address the problem at hand. This is necessary given the diversity of freshwater wetlands that may be encountered and the multitude of factors or stressors that may be at work in the particular wetland under study. At the same time, however, an ecosystem approach serves to maintain the key concept of interlinkage of the wetland components. An additional overarching provision is that data collection and evaluation be tiered (or phased) so that resources are focused effectively, and there is ample opportunity for the risk assessor and risk manager to discuss the scientific and policy implications as the risk assessment proceeds. The following discussion highlights some of the focal points in the application of an ecosystem approach (Figure 1).

Problem Formulation and Identification

When begining a risk assessment it is important to know what information has been developed previously for the wetlands under study. Aerial photographs, historical maps, land-use documents, etc., are all useful in gaining an understanding of the history and current status of the area. It is also important to gain an understanding of the hydrology and geology driving the wetlands under study. As noted earlier, wetlands vary greatly in their structure primarily due to hydrological and geological conditions, both of which will influence sources and trigger points [parameter(s) affected] for stressors and, thereby, the focus of the risk assessment. While there are other issues that may be relevant to understand before beginning the risk assessment, a key aspect is to determine the spatial extent of the area under study. For some wetlands, this will amount to only a few acres; for others, it may encompass an entire watershed or multiwatershed ecosystem comprised of several thousand acres or more. In order to develop effective solutions, wetland problems must be formulated within a landscape context. A landscape approach is especially needed when the cumulative effect of wetlands on stream water quality and quantity is at issue (Johnston and others 1990). Imposing artifical boundaries (political or otherwise) can result in a superficial risk assessment that fails to address some of the important issues.

Once the wetland and its hydrologic linkages are delineated, the problem (stressors) can be described in terms of its source, transport, and potential area of impact. This is the point at which the key stressors and receptors in the wetlands under study should be clearly identified and, if necessary, prioritized in order to guide the risk assessment process that will follow. Problem identification should be an intensive effort to formulate and validate key questions and issues to be addressed by the risk assessment. Moreover, it should determine those wetland functions and human values that are critical to the resolution of the problem. Failure to give adequate attention to this step can undermine the risk assessment and lead to ineffective risk management.

Development of Assessment and Measurement End Points

One of the most important steps in risk analysis is establishing clear assessment end points, i.e., the functions and associated values to be protected, enhanced, or created (Figure 1), because they set the stage for all of the forthcoming effort. In addition, assessment end points specific to freshwater wetlands can take on major significance due to the diversity of potential wetland types that may be encountered. In fact, using an ecosystem approach, the assessment end point(s) may or may not be biologically or ecologically based. For example, the hydrology, geomorphology, soils, or other aspects of the wetlands may be far more important to focus on than some of the biological resources.

For an illustration of assessment end points, assume that the freshwater wetland under study is one that is dependent on a constant supply of high-quality groundwater. One assessment end point might be to protect the supply of high-quality groundwater to the wetlands by preventing exposure to nonchemical stressors. In this example, the focus is on hydrologically driven end points. In other situations, ecologically driven assessment end points may be given top priority. For example, in the study of the Clark Fork River Superfund Site in Montana (Pascoe and DalSoglio 1994, Linder and others 1994), none of the assessment end points for the riparian wetlands or the river itself included protection of the water supply. This does not mean that the risk assessment for this particular wetlands was done incorrectly but that the primary focus was to protect ecological resources rather than the one key component responsible for the wetlands themselves-water. Another important consideration could be ensuring that the groundwater is meeting a minimum quality standard, defined in various ways ranging from a particular range of pH, turbidity, specific conductance, etc., to absence of chemical stressors at some threshold concentrations. The point of these examples is that all of the parameters (hydrological and ecological) that are critical to the long-term sustainability of the wetlands should be considered as possible assessment end points during the development phase. A subset of these may be chosen for use in the risk assessment, but it is important to establish the assessment end points clearly in the context of what is vital to sustaining or improving the health of the freshwater wetland. Some important values and functions of freshwater wetlands, from which assessment end points can be derived, are shown in Table 1. Examples of possible assessment end points specific to freshwater wetlands are shown in Table 2.

Like assessment end points, measurement end points may or may not have a biological or ecological basis, but they must be directly relevant to, and linked with, the assessment end points. In the examples given above, measurement end points could be analytical determinations of contaminant concentrations in the water supplying the wetlands, the specific conductance or suspended solids levels in the water, the flow of water to the

Table 3. Representative technical references for sampling and test methods that may be useful for evaluating risks in wetland habitats

Habitat	Biota	Reference
Freshwater	Vascular plants	Ratsch (1983) Gorsuch and others (1992)
Freshwater/ marine/ estuarine	Algae and vascular plants	Gorsuch and others (1992) Wetzel and Likens (1995)
	Birds	Adamus (1993a,b)
Freshwater	Aquatic vertebrates and invertebrates	Nielsen and Johnson (1983) Rand and Petrocelli (1985) US EPA (1990. 1991) ASTM (1995) Wetzel and Likens (1995)
Marine	Marine/estuarine invertebrates and vertebrates	Nielsen and Johnson (1983) US EPA (1989, 1991) ASTM (1995) Wetzel and Likens (1995)
Freshwater sediments	Epifauna, infauna, and vertebrates	US EPA (1989, 1991) ASTM (1995) Wetzel and Likens (1995)
Marine/ estuarinr sediments	Epifauna, infauna, and vertebrates	US EPA (1989, 1991) ASTM (1995)
Transitional/ upland	Vertebrates and invertebrates	Linder and others (1994)
	Vascular plants	Linder and others (1993)

wetlands, as well as various biological and ecological measures that are relevant to the assessment end points.

Whether qualitative and reliant on published information or quantitative and implemented as part of a designed study, aquatic field surveys and biological tests are the cornerstone for evaluating risks associated with chemical, physical, or biological stressors. Frequently, these tools are used in the measurement or monitoring of wetland populations and communities through structural end points such as relative abundance, species richness, community organization (diversity, evenness, similarity, guild structure, and presence or absence of indicator species), and biomass. Functional end points, such as cellular metabolism, individual or population growth rates, and rates of material or nutrient transfer (e.g., primary production, organic decomposition, or nutrient cycling) are less commonly measured. Functional measurements are important in interpreting the significance of an observed change in population or community structure. However, functional measures are difficult to interpret in the absence of structural information, have not been standardized, and require considerable understanding of the system and processes involved.

There are numerous methods for assessing effects of stressors on biological processes. The response variables used in these tests are the measurement end points that can be used to predict likely impacts and, ultimately, the risk that the functions of the wetland will be impaired. Field and laboratory methods are available for evaluating aquatic habitats, sediments, and soils within wetlands. Sources of information on tests that measure biological responses to stressors are listed in Table 3.

Exposure Assessment

Exposure assessment refers to the identification of major pathways and media through which wetland receptors come in contact with the stressor. The extent, frequency, and magnitude of the exposure will strongly influence the potential for negative impacts and, thereby, affect risk. Inputs of chemical and nonchemical stresses to freshwater wetlands occur through geological, biological, and hydrological pathways typical of other ecosystems (Mitsch and Gosselink 1993). Geological input from weathering of parent rock can be an important source of exposure in some wetlands (Presser 1994, Presser and others 1994). Biological inputs include photosynthetic uptake of carbon, nitrogen fixation, and biotic transport of materials by animals. Except for gaseous exchanges such as carbon and nitrogen fixation or aerial deposition, however, inputs to wetlands are generally dominated by hydrology. Hydrologic transport to freshwater wetlands may occur through precipitation, surface, and/or groundwater flow. The hydrologic exposure pathways of freshwater wetlands are determined by their flooding regime or by the balance between precipitation and evapotranspiration.

Hydrodynamics will affect exposure levels in both the aquatic and soil/sediment compartment of a wetland because it will to a large extent determine the soil/sedimient chemistry by producing anaerobic or aerobic conditions, importing and removing organic matter, and replenishing nutrients and toxins. Exposure can occur in transition zones between the wetland and surrounding upland areas. It is also important to consider this area when examining potential exposure scenarios.

Ideally, exposure in the wetland ecosystem is assessed based on representative monitoring data. In the absence of measurement data, exposure can be predicted in the context of a wetland-specific hydrogeomorphic, biogeochemical, and ecological setting. In case of a chemical exposure assessment, information on the inherent properties of substance(s) should be used in combination with the wetland characteristics in order to derive exposure concentrations or levels. Describing the level and distribution of a stressor in the wetland environment or organisms, and its changes with time (in concentration and chemical form), is a complex process and needs to include a rigorous evaluation of what drives exposure. In order to ensure that predicted aquatic and sediment exposures are realistic, all available knowledge of the wetland ecosystem should be integrated in the exposure evaluation of a chemical stressor. Some parameters and measurements that can be important when evaluating and predicting exposure to chemical and/or nonchemical stressors in freshwater wetlands are listed in Table 4.

Gompound-specific information on chemicals and biogeochemical processes affecting exposure in the different compartments is usually derived and extrapolated from standard laboratory tests or literature data Applicability of literature data and data from standard tests to freshwater wetland ecosystems requires review and, ideally, field verification.

Table 4. Important parameters for assessing exposure to chemical and nonchemical stressors in freshwater wetlands

Hydrogeomorphic information
Type of water input (capillary, precipitation, etc.)
Type of water flow (surface, subsurface, etc.)
Type of water outputs (percolation, evaporation)
Suspended sediment load and characterization
Sedimentation rate
Biogeochemical information
Soil/sediment origin and characterization
Microbial activity
Oxidation/reduction conditions
Organic matter content of sediments
Ecological information
Plant communities
Aquatic and benthic community structure
Wildlife survey
Structural/functional assessment
Compound-specific information (chemical stressor)
Volatility
Hydrophobicity
Water solubility
Octanol/water partition coefficient
Hydrolysis
Photolysis
Biodegradation

Biological Assessment

Biological assessment is the determination of potential adverse effects to biota and the linkage of exposure to the stressor. The associated level of exposure and resulting response (concentration-response) determination is also part of this assessment. Toxic or other harmful effects may be observed at the species level (as from acute or chronic aquatic toxicity tests, for example) but may also occur at higher levels of organization (population, community). Biological assessments are primarily toxicological tests that can be used in either a field or laboratory setting. While there are many issues related to the conduct and application of ecotoxicological tests, they represent one of the main sources of effects information available to the risk assessor.

In order for the assessment to be on target and provide useful data, the key stressors and receptors in the wetlands under study must be clearly identified. Adequate attention to the problem-formulation step should provide this information. Generally, assessments of biological effects in wetlands should consider toxicity to animals and plants in the overlying water as well in the sediments, provided the stressor is likely to enter and persist in the sediments. In addition, the assessment may not be limited to wetland or aquatic systems but may need to extend to the surrounding transitional zones. Some stressors will impact the terrestrial environment adjacent to the wetlands, and this area too should be evaluated if there are potential pathways for exposure of receptors in the wetlands.

Biological assessments in the aquatic environment should include representative and, ideally, sensitive species of: (1) primary producers, (2) primary consumers, (3) microbial community, (4) saprophages/detritivores, and (5) carnivores. Potential tests for the primary producers could include tests with algae and vascular plants, both submerged and emergent forms. Effects on primary consumers could be evaluated by testing representative species of protozoa, invertebrates, insects, and amphibians. Inhibition of microbial activity could be evaluated by studying the effect on aerobic and/or anaerobic respiration. Toxicity tests with crustacea and insects can be used to assess effects on the saprophage/detritivore community. Finally, standard acute and chronic tests are available to assess the effect on fish (Table 3).

Biological assessment of the benthic communities should take into account pathways of exposure. In addition, it needs to be realized that observed effects will be strongly influenced by sediment/soil biogeochemical conditions such as organic carbon content, particle size distribution, sulfide content, redox potential, and time period allowed for equilibration to occur between dissolved and sorbed fractions of chemical stressors. Available test methods concern detritivores or mixed detritivores/herbivores/carnivores and include insect, annelids, and crustacean species with both acute and chronic end points (Table 3).

Guidelines and tests should be developed within a framework of taxonomic groups rather than for single species. This should make it possible to test representatives from the different wetland compartments and facilitate extrapolation of obtained test results to the wetland of interest. Furthermore, the guidelines and tests should include both acute and subchronic/ chronic toxicity end points.

Most of the impacts on freshwater wetlands will occur in the aquatic environment, i.e., the sediment and overlying water. Even so, the terrestrial environment surrounding or transitional to the freshwater wetland may also be at risk, depending on the type of stressor and the exposure. Species that are dependent on the wetland structure and function, including insects, amphibians, reptiles, small mammals, birds, and transition-zone plants, trees, and shrubs, should be given consideration when evaluating potential effects. Standardized toxicity tests are currently available for many insects, some amphibians, and numerous small mammals and birds but few have been adapted for the species most often associated with freshwater wetlands. Acute and chronic bioassays with rodents and lagomorphs have been used for many years to determine the toxicity of chemicals and other materials that may also pose a risk to humans. Similarly, standard acute and chronic tests with species of waterfowl and upland birds have been widely used in the field of environmental toxicology.

There are, however, few tests that have been developed for nonfood plants, although the tests currently used in regulatory programs for pesticides and herbicides may be useful. Tests for root elongation and shoot development, seed germination, and other methods are known (Table 3) and may be useful in evaluating toxicity of soils in the transition zone. Other soil tests, some using earthworms, might be useful in this context (Table 3). Keep in mind that the primary focus of the assessment is the wetland itself, and it is there that the effort should begin.

Unfortunately, there are few tests that lend themselves easily to determining the potential toxic effects on trees and shrubs that may inhabit the transition zones. In those situations it may be more plausible to determine impacts in situ on those trees and shrubs located adjacent to the wetlands of concern. Methods developed by plant scientists to analyze and interpret vegetation in transitional or upland zones can be utilized for this (Table 3).

Using standardized toxicity tests brings up several important considerations. One of these concerns data interpretation and is driven primarily by the fact that most easily maintained species used in testing are not the same species generally found in freshwater wetlands. Thus, the uncertainty of extrapolating from one species to another within the same genus could be as large as extrapolating from rodents to humans. Therefore, it is important to understand the limitations of surrogate species testing and its application to risk assessment. Other uncertainties arise when extrapolating from acute exposure test data to chronic exposure situations, high concentration-response studies to low-concentration exposures, and many other potential situations.

As mentioned above, selection of biological tests for wetland toxicity evaluation should be driven by the exposure assessments as affected by the hydrogeomorphic and biogeochemical characteristics of the wetland of interest.

Ecological Assessment

Ecological assessment is the determination of harmful impacts at the population, community, or ecosystem level and may entail field studies, laboratory studies, or both. In general, standardized tests do not lend themselves to this type of assessment and few provide useful ecosystem-level information. In addition, there are significant temporal and spatial issues that come into play. Measuring or accurately predicting a significant change in an ecosystem may require years or decades of study, yet the risk assessor and risk manager are faced with much more compressed time lines. Just as important, it is difficult to isolate easily studied areas of a wetland from the surrounding ecosystem that supports it, which may require the risk assessor to include caveats and large uncertainties in the risk assessment.

Given this situation, most ecological assessments have focused on measuring structural components of the ecosystem, including the size and makeup of the habitat, biomass, standing crop of important plants and animals, abundance and diversity of plants and animals, and other measures. There are, however, functional measurements that are useful for understanding the ecological health and sustainability of wetlands. These include hydrologic flux and storage, biological productivity, biogeochemical cycling and storage, decomposition, and wildlife habitat (Richardson 1994). A functional assessment is based on the concept that variables (functions) that integrate key ecosystem-level wetland processes can be used as a metric to compare impacts and quantify functional loss when compared to reference wetlands of the same hydrogeomorphic classification (Brinson 1993b, Richardson 1994). Measured disturbances to wetland functions are scaled to a reference wetland (reference system = 100%), thus creating a comparative ecosystem response surface. The final step is to develop a threshold of acceptability for the wetlands under investigation and to compare the altered functions to that threshold. Practical methods for analyzing wetland functions in the field are developing concurrently with the conceptual framework for this assessment technique (Richardson 1994).

Functional assessment can be a powerful tool for evaluating ecosystem-level impacts and risks within a time frame necessary for risk assessment. One of the main strengths of this approach is that it causes the risk assessor and risk manager to identify and resolve problems in the context of what a wetland does on the landscape. Even though human values may ultimately drive the risk assessment, the inseparable linkage of values to ecological functions, particularly for constructed wetlands, becomes evident as this approach is planned and implemented. Moreover, it provides the risk manager with information on what functions must be maintained or restored in order to provide the desired outcome (values) of the wetland.

In cases where sufficient supporting data already exist, it may be possible to evaluate chemical stressors by developing contaminant-specific assessment protocols that can be applied at an ecosystem level. For example, using published information on toxicity, bioaccumulation, and environmental cycling, Lemly (1995a) developed a protocol for aquatic hazard assessment of selenium. This protocol integrates data for abiotic (water, sediments) and biotic (plankton, macroinvertebrates, fish, aquatic birds) components and yields information that can be directly applied to an ecosystem approach to risk assessment. This technique can save time and money in the assessment process by targeting data collection toward specific parameters and utilizing existing toxicity information in the evaluation phase. It also has the advantage of being applicable regardless of wetland type, size, or location and can be used to evaluate the success of risk management activities. Thus, in certain situations, ecological assessment in the risk analysis context can be simplified to a linkage of contaminant monitoring data with existing toxicity and hazard profiles.

An Example of Ecosystem-Based Assessment

The ecosystem approach outlined in this paper was used to evaluate ecological risk at wetlands in Utah (Waddell and Stanger 1992, Stephens and others 1993, Hamilton and Waddell 1994, Finger and others 1995, Lemly 1995b, Waddell and May 1995, Buhl and Hamilton 1996) (Table 5). A team effort involving risk assessors, wetland managers (which were, in effect, the risk managers), biologists, ecotoxicologists, geologists, hydrologists, and wetland scientists was used from the outset of the assessment. Problem formulation began by comparing information on historical conditions in the wetlands to more recent data, which suggested that

Table 5. An example of ecosystem-based risk assessment for wetlands[a]

	Approximate timeframe
Problem formulation	
Review historic conditions, water quality issues, wildlife health issues, land-use issues	6 months
Problem identification	
Possible contamination from agricultural irrigation drainage	3 months
Operational strategy	
Consensus by risk assessor and risk manager on scope, timing, effort, and constraints for the assessment	3 months
Risk characterization	
Engage expertise of necessary disciplines and evaluate sources, transport, fate, and effects of stressor on wetlands with focus on health and reproduction of wildlife	3 years
Risk statement	
Probabilistic estimates from toxicity studies, site-specific hazard ratings	6 months
Risk management	
Divert contaminated drainage, secure water through cooperative agreement	6 months
with landowners	3 years
Total	8 years

[a]This example involved evaluating risks at wetlands managed for wildlife conservation in Utah. Application of this operation framework resulted in proper formulation and identification of the problem, accurate risk characterization, and effective risk management,

changes in land use had led to degradation of water quality. The key problem identified, i.e., the risk factor to be evaluated, was that drainage water from agricultural irrigation could be contaminating the wetland and threatening wildlife. With this operational guidance, the risk assessors and risk managers held discussions to define the scope, timing, and level of effort for the assessment. This resulted in a well-coordinated effort that utilized the expertise of several disciplines. For example, wildlife biologists studying the health of waterfowl observed teratogenic deformities in embryos and identified selenium (a naturally occurring soil trace element) as a major concern to wetland biota because of its potential to impair reproduction. Hydrologists and geologists examining water budgets, geomorphology, soils, and water chemistry identified sources and transport pathways for the contamination to reach the wetlands. They determined that agricultural irrigation was a primary source of selenium because it was found to leach the trace element from soil and produce seleniferous drainage water that ultimately reached the wetlands. Ecotoxicologists determined the severity of threats to fish and migratory birds using a combination of toxicity studies and ecosystem modeling of selenium fate and effects. Wetland scientists and biologists aided in exposure assessment by identifying major ecosystem processes that influenced cycling and bioaccumulation of selenium in food pathways leading to wildlife. They also identified biota of special concern, i.e., rare or endangered species, and provided assistance interpreting the potential population and community level impacts of the mortality indicated by the toxicity studies.

The results of this assessment in Utah were expressed both in terms of probabilistic risk to specific biota (by way of the toxicity studies) and as site-specific hazard ratings derived from the ecotoxicology data base for selenium. With this information in hand, the risk managers developed a two-step wetland management plan to protect wildlife. First, water management actions were implemented to reduce contamination by diverting agricultural irrigation drainage water that previously had been used to flood some of the wetlands. Second, a long-term strategy for securing adequate freshwater supplies and reducing inputs of selenium from upgradient sources (municipal and agricultural) was developed. This required cooperative efforts between the risk managers and several other parties with jurisdiction over part of the hydrologic watershed feeding the wetland, i.e., federal, state, municipal, and private landowners.

This example illustrates some important points for wetland risk assessment. It is crucial that the risk assessor and risk manager interact early on in the process to define the scope, timing, level of effort, and constraints involved with the risk assessment. Having this agreed to up front allows the risk assessor to focus on those issues most important to reaching an informed risk management decision. It also aids the risk assessor in engaging those experts and scientific disciplines necessary for the assessment in an efficient and timely manner. There will be issues specific to freshwater wetlands, and the particular type of wetland, that will need discussion and clarification between the risk manager and risk assessor. In the case of freshwater wetlands, this discussion may have several important outcomes. For example, small, easily managed wetlands may require a reduced level of effort and only a screening-level assessment to satisfy the requirements of the risk manager. On the other hand, wetlands that are tens or hundreds of acres, are in the midst of major industrial activities, or are complex in terms of their hydrology, soils, geomorphology, etc., may require a much greater level of effort on the part of the risk assessor. In the example used here, the wetlands were partially dependent upon water that originated outside the study area, which made resolution of the problem more complex because it involved multiple landowners. The risk manager must convey a clear understanding of the societal values and associated wetland functions that are driving the risk management decision, as well as the regulatory and jurisdictional issues involved. With this information in hand, the risk assessor can build a sound risk assessment. Using a team approach allows the various ecosystem components and diagnostic tools discussed in earlier sections of this paper to be expertly examined and effectively applied in the risk assessment. An ecosystem-team approach can target efforts to priorities and reduce the level of uncertainty in the risk assessment by providing multiple lines of evidence. This, in turn, will lead to more effective risk management.

Information Needs and Conclusions

There is a clear need to establish and implement a consistent operational framework in order to make full use of ecosystem-based risk assessment. Several concerns are evident. Determining the effects of multiple stressors (chemical, physical, biological; of anthropogenic or natural origin) must be an integral component of the assessment process. Development of reliable acute, subchronic, and chronic tests specific to actual wetland biota rather than standard or surrogate test species is necessary. Alternative exposure/effects scenarios must be evaluated. Understanding the fate and transport of chemicals and their interaction with physical, chemical, and biological toxicity-modifying factors is critical. The parameters that must be measured on site to determine potential pathways and the fate of toxins need to be better quantified. Tools for integrating and analyzing complex ecosystem interactions need to be refined or, in some cases, are yet to be developed. Approaches for evaluating the influence of seasonal and spatial variability are especially needed.

The ecosystem approach presented here uses hydrogeomorphic characterization together with wetland functions as the criteria for establishing the transport, fate, and effects of both chemical and nonchemical stressors. Coupled with toxicity assessments at three organizational levels-organismic, population and community, and ecosystem-this approach may be used as a predictive tool to describe the exposure and effects of stressors in freshwater wetlands and to describe existing conditions. Toxicity assessments involve tests of varying complexity (single species, mesocosm, ecosystem assessments, etc.). As a rule of thumb, costs escalate with increasing complexity, single-species laboratory bioassays being the least expensive. From a cost-benefit perspective, the least complex test that can adequately predict ecosystem effects should be the method of choice, providing proper validation has been carried out. An ecosystem approach may reduce the overall cost of risk assessment by identifying key biological, chemical, and physical parameters that must be evaluated early on in the assessment process.

Wetland risk assessment forms the information base that drives important environmental management decisions on a local, state, and federal level in the United States. The quality of these assessments and the effectiveness of the resultant risk management actions can be significantly improved if an ecosystem-based approach that integrates information on geomorphology, hydrology, soils, and ecology is used. Practical application of the approach will result in a better understanding of how physical, chemical, and biological stressors impinge on wetlands and provide a foundation for prudent wetland management.

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Literature Cited

Adamus, P. R. 1993a. Users manual: Avian richness evaluation method (AREM) for lowland wetlands of the Colorado Plateau. EPA/600/R-93/240. US Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

Adamus, P. R. 1993b. Computer program: Avian richness evaluation method (AREM) for lowland wetlands of the Colorado Plateau. US Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

ASTM (American Society for Testing and Materials). 1995. Annual book of ASTM standards: Water and environmental technology, vol. 11.04. American Society for Testing and Materials, Philadelphia, Pennsylvania.

Brinson, M. M. 1993a. A hydrogeomorphic classification for wetlands. Technical report WRP-DE-4. US Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi.

Brinson, M. M. 1993b. Changes in the functioning of wetlands along environmental gradients, *Wetlands* 13:65–74.

Buhl, K. J., and S. J. Hamilton. 1996. Toxicity of inorganic contaminants, individually and in environmental mixtures, to three endangered fishes (Colorado squawfish, bonytail, and razorback sucker). *Archives of Environmental Contamination and Toxicology* 30:84–92.

Culotta, E. 1995. Bringing back the Everglades. *Science* 268: 1688-1689.

Finger, S. E., A. C. Allert, S. J. Olsen, and E. V. Callahan. 1995. Toxicity of irrigation drainage and associated waters in the Middle Green River Basin, Utah. Technical report. National Biological Service, Midwest Science Center, Columbia, Missouri.

Frayer, W. E., D. D. Peters, and H. R. Pywell. 1989. Wetlands of the California Central Valley: Status and trends-1939 to mid-1980's. US Fish and Wildlife Service, Portland, Oregon.

Gorsuch, J. W., W. R. Lower, W. Wang, and M. A. Lewis. 1992. Plants for toxicity assessment. STP 1115. American Society for Testing and Materials, Philadelphia, Pennsylvania.

Hamilton, S. J., and B. Waddell. 1994. Selenium in eggs and milt of razorback sucker (*Xyrauchen texanus*) in the Middle Green River, Utah. Archives of *Environmental Contamination and Toxicology* 27:195–201.

Hook, D. D., W. H. McKee, Jr., T. M. Williams, S. Jones, D. Van Blaricom, and J. Parsons. 1994. Hydrologic and wetland characteristics of a Piedmont bottom in South Carolina. *Water, Air; und Soil Pollution* 77:293–320.

Johnston, C. A., N. E. Detenbeck and G. J. Niemi. 1990. The cumulative effect of wetlands on stream water quality and quantity: A landscape approach. *Biogeochemistry* 10:105–141.

Lemly, A. D. 1994. Irrigated agriculture and freshwater wetlands: A struggle for coexistence in the western United States. *Wetlands Ecology and Munagemmt* 3:3-15.

Lemly, A. D. 1995a. A protocol for aquatic hazard assessment of selenium. *Ecotoxicology and Environmental Safety* 32:280–288.

Lemly, A. D. 1995b. Hazard of selenium to fish and migratory birds at Ouray National Wildlife Refuge, Utah. Technical report. Department of Fisheries and Wildlife Sciences, Virginia Tech University, Blacksburg, Virginia.

Linder, G., E. Ingham, G. Henderson, and C. J. Brandt. 1993 Evaluation of terrestrial indicators for use in ecological assessments at hazardous waste sites. EPA 600/R-92-183. US Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

Linder, G., R. Hazelwood, D. Palawski, M. Bollman, D. Wilborn, J. Malloy, K. DuBois, S. Ott, G. Pascoe, and J. DalSoglio. 1994. Ecological assessment for the wetlands at Milltown Reservoir, Missoula, Montana: Characterization of emergent and upland habitats. *Environmental Toxicology and Chemistry* 13:1957–1970.

Lowrance, R. R., R. L. Todd, J. Fail, 0. Hendrickson, R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. *Bioscience* 34:374–377.

Mitsch, W. J., and J. G. Gosselink. 1993. Wetlands, 2nd ed. Van Nostrand Reinhold, New York.

Moore, P. D., and D. J. Bellamy. 1974. Peatlands. Springer-Verlag, New York.

Moore, S. B., J. Winckel, S. J. Detwiler, S. A. Klasing, P. A. Gaul, A. R. Kanim, B. E. Kesser, A. B. Debevac, A. Beardsley, and L. A. Puckett. 1990. Fish and wildlife resources and agricultural irrigation drainage in the San Joaquin Valley, California. San Joaquin Valley Drainage Program, Sacramento, California.

Nielsen, L. A., and D. L. Johnson. 1983. Fisheries techniques. American Fisheries Society, Bethesda, Maryland.

Pascoe, G. A., and J. A. DalSoglio. 1994. Planning and implementation of a comprehensive ecological risk assessment at the Milltown Reservoir-Clark Fork River Superfund Site, Montana. *Environmental Toxicology and Chemistry 13:1943*-1956.

Philips, J. D. 1989. Nonpoint source pollution control effectiveness of riparian forests along a coastal plain river. Journal of Hydrology 110:221–237.

Presser, T. S. 1994. The Kesterson Effect. *Environmental Management* 18:437–454.

Presser, T. S., M. A. Sylvester, and W. H. Low. 1994. Bioaccumulation of selenium from natural geologic sources in western states and its potential consequences. *Environmental Management* 18:423–436.

Rand, G. M., and S. R. Petrocelli. 1985. Fundamentals of aquatic toxicology. Hemisphere, New York.

Ratsch, H. C. 1983. Interlaboratory root elongation testing of toxic substances on selected plant species. EPA 600/S3-83-051. US Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

Richardson, C. J. 1994. Ecological functions and human values in wetlands: A framework for assessing forestry impacts. *Wetlands* 14:1–9.

Riekerk, H. 1993. Groundwater flow in pine-cypress flatwoods. In General technical report SO-93, Proceedings of the seventh biennial southern silvicultural research conference. USDA, Forest Service, Southern Forest Experiment Station, New Orleans, Louisiana.

Saul, B. 1995. Nutrient exchange between floodwater and groundwater in a forested riparian zone in the southeastern coastal plain. MS thesis. Clemson University, Department of Forest Resources, Clemson, South Carolina.

Siegel, J. R., and P. H. Glaser. 1987. Groundwater flow in a bog-fen complex, Lost River Peatland, northern Minnesota. Journal *of Ecology* 75:743–754.

Stephens, D. W., B. Waddell, L. A. Peltz, and J. B. Miller. 1992. Detailed study of selenium and selected elements in water, bottom sediment, and biota associated with irrigation drainage in the Middle Green River Basin, Utah, 1988-90. Water-resources investigations report 92-4048. US Geological Survey, Salt Lake City, Utah, 164 pages.

Suter, G. W. 1993. Ecological risk assessment. Lewis Publishers, Chelsea, Michigan.

Theriot, R. 1988. Relationship of bottomland hardwood species to natural water regimes. Pages 344-351 *in* The ecology and management of wetlands. vol. 1, ecology. Croom-Helm, London.

Thompson, S. P., and K. L. Merritt. 1988. Western Nevada wetlands-history and current status. *Nevada Public Affairs Review* 1:40–45.

US EPA (US Environmental Protection Agency). 1989. Protocols for short term toxicity screening of hazardous waste sites. EPA/600/3-88/029. US Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

US EPA (US Environmental Protection Agency). 1990. Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface waters. EPA/600/4-90/030. US Environmental Protection Agency, Office of Research and Development, Washington, DC.

US EPA (US Environmental Protection Agency). 1991. Ecological assessment of Superfund sites. Publication 9345.0-051. US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Hazardous Site Evaluation Division (OS-230), Washington, DC.

US EPA (US Environmental Protection Agency), 1992. Framework for ecological risk assessment. EPA 630/R-92-001. Risk Assessment Forum, Washington, DC.

van der Valk, A. G. 1989. Northern prairie wetlands. Iowa State University Press, Ames.

Waddell, B., and T. May. 1995. Selenium concentrations in the razorback sucker (*Xyrauchen texanus*): Substitution of non-lethal muscle plugs for muscle tissue in contaminant assessment. *Archives of Environmental Contamination and Toxicology* 28:321–326.

Waddell, B. H., and M. C. Stanger. 1992. The influence of selenium on incubation patterns and nesting success of waterbirds at Ouray National Wildlife Refuge, Utah. Contaminant report number R6/400S/92. US Fish and Wildlife Service, Salt Lake City, Utah.

Wetzel, R. G., and G. E. Likens. 1995. Limnological analyses. Springer-Verlag, New York.

Zedler, P. 1987. The ecology of southern California vernal pools: A community profile. Biological report 85(7.11). US Fish and Wildlife Service, Washington, DC.