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CHAPTER 4

Workgroup II Synopsis: Contaminant Fate and Effects in Freshwater Wetlands

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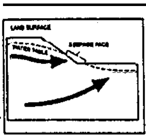
Pollution ecology is one of the few disciplines in biology that grew out of a societal need to fix a problem. The research community was forming questions as well as simultaneously developing methods, both toxicological and analytical, to address the questions in a cultural framework that demanded immediate answers.

Aquatic toxicologists wrestled with pollution issues as they developed. By establishing basic methods and sorting out different responses between ecosystem compartments, an assessment philosophy emerged that enabled us to better investigate contaminant impacts (Mount and Brungs 1967; Mount and Stephan 1967).

CHAPTER PREVIEW



Ecotoxicology and Risk Assessment for Wetlands. Michael A. Lewis et al., editors.

ISBN 1-880611-16-3

More recently developed disciplines, such as sediment evaluation and ecological risk assessment, have benefited from the early investment of scientists who guided the development of aquatic toxicology. Some of the basic philosophical tenets were in place, and as a consequence, these more recent disciplines developed at a faster rate than those established earlier. Even in instances where the foundation was not a good "fit," it provided a starting point from which modifications could be made, increasing the chances that conceptual or methodological mistakes might be few in number or avoided altogether.

Research on wetlands did not originally focus on toxicology. Wetlands research has long been conducted by aquatic ecologists, hydrologists, waterfowl biologists, botanists, limnologists, etc., many of whom were interested in the structure, function, and biota of different types of wetlands. Management values have also figured into the equation. In the U.S., for example, federal and state agencies manage wetlands for migratory birds, endangered species, bait production, and flood control, just to list a few management values driving research. Water quality improvements resulting from implementation of Sections 402 and 404 of the Clean Water Act (CWA) have had a positive effect on wetland management. In addition, the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) also affect many aspects of wetland management. Nongovernment entities such as sportsman groups and conservation organizations also work to protect and manage wetlands.

Unfortunately, there is often a large "disconnect" between the wetlands research community and the risk assessment community regarding wetland data availability and its interpretation. There is a smaller, but nonetheless important disconnect between aquatic toxicology and risk assessment groups. When factors in risk assessments are dealt with as uncertainties, sometimes it is due to a lack of awareness that data exist outside the contaminant realm.

One of the most fundamental oversights in risk assessment is the failure to recognize that most of the remaining freshwater wetlands in the U.S. are altered from their natural state because of changes in hydrology and surrounding land use. 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 (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 function in water storage and supply or as habitat for biota (Frayer et al. 1989; Moore et al. 1990). Dredging and channelization for navigational purposes have 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 et al. 1984; Philips 1989; Richardson 1994; Culotta 1995). These physical alterations constitute a chronic stress that

influences the way wetland ecosystem
regional and national scale, physical a
the integrity of wetlands than are che

In addition to recognizing and unders
already been altered to some degree, it
process into an ecological context. Thi
principal ecosystem attributes (ecolog
to structure these wetlands and deter
speciation, and biological exposure an
based approach for evaluating impacts
chemical, biological, and physical str

Major external factors such as climat
conditions in which wetland ecosyste
influences not only temperature, whic
the ecosystem, but also the amount, f
climatic variables are expressed in the
regional and local geomorphic setting
wetlands is the way in which these ext
processes to determine the risk setting,
contaminants or other stressors (Figure
between uplands and aquatic systems (
connection is across the surface or thro
critical linkage that in part determines
cal filters or transformers buffering fl
addition, it is this critical linkage that
an important component of many tox

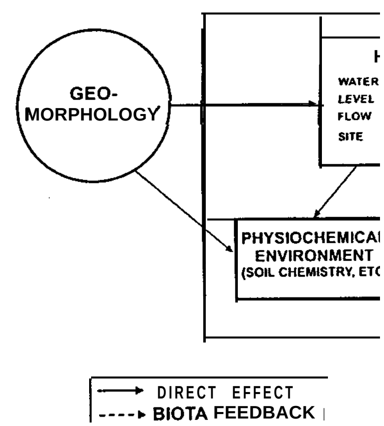


Figure 4-l Major external factors that determi
operate

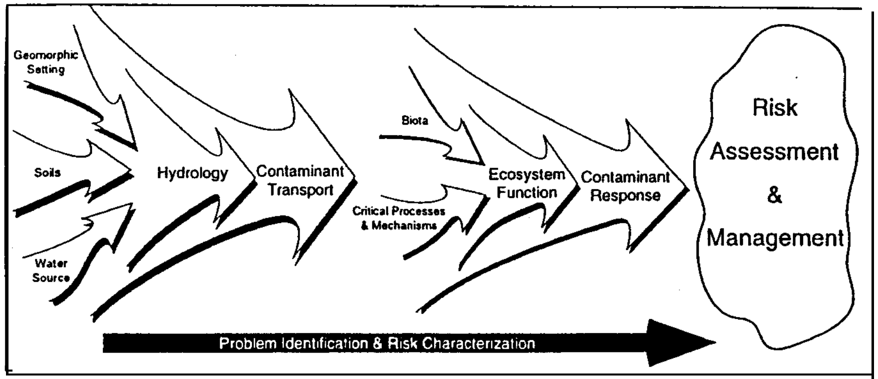


Figure 4-2 Interaction of external factors and internal processes that determine the risk setting (potential for transport of and impacts from stressors) for wetlands

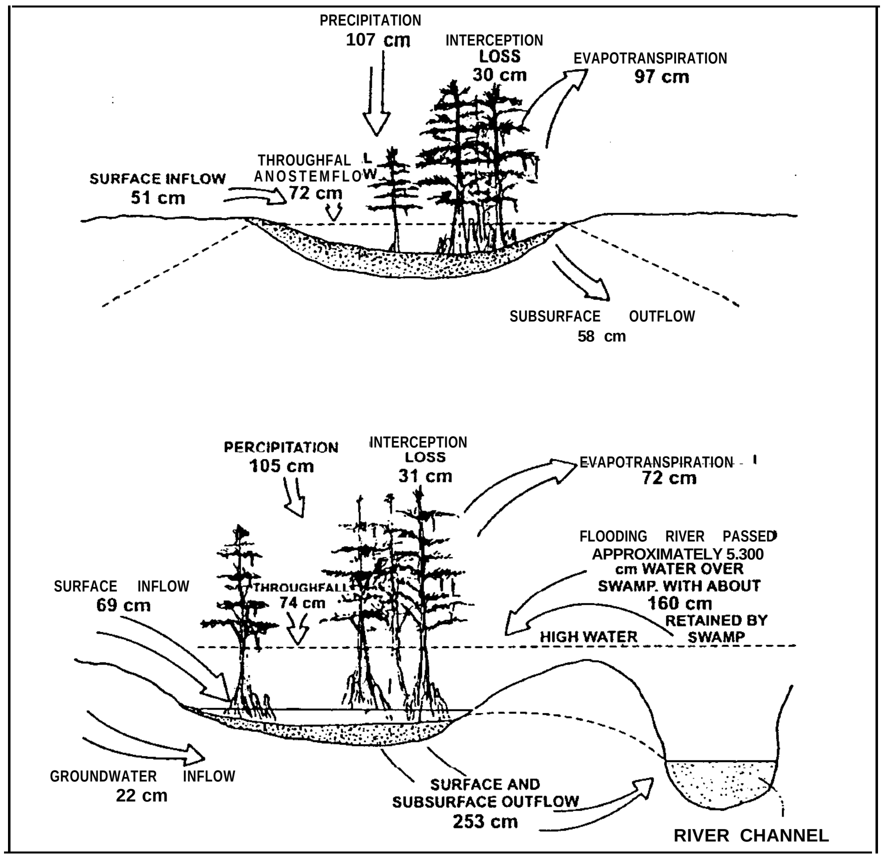


Figure 4-3 Annual water budgets illustrating the critical hydrologic link **that** wetlands provide between uplands and aquatic habitats

The complex interaction of external
when coupled with the diverse array
wetland types throughout North Am
from infrequently inundated, isolate
potholes, **Playa** lakes, and vernal po
systems (such as the Everglades, Ok
The proportion of the landscape dom
Florida and Louisiana, where wetlan
other areas such as the north-central
only a small portion of the landscap

Table 4-l Major types of wetlands in the U

Wetland type	Distribution
Freshwater marsh	Widespread permanent
Tidal salt or brackish marsh	Intertidal z fortnightly
Prairie pothole	Northern pl to permane fluctuating
Fen	Associated water; perm flowing
Bog.	Abundant i regions; pr principal sc
Swamp	Prolonged flooding

Current Practi for Fre

Through various sources of regulato
pursued in either a qualitative or qu
be characterized hydrologically by fo
analysis of physical structure (Brinso
wetlands may be characterized in an
ally yields a focus on soils and v
for Wetland Delineation [FICWD] 1
wetlands, these independent approac
the risk assessment process as it is c

Risk assessment for wetlands may focus on noncontaminant as well as contaminant issues (Leibowitz et al. 1992; Pascoe and DalSoglio 1994). In practice, physical, chemical, and biological stressors generally impact wetlands simultaneously. In evaluating the role of these stressors, various issues must be resolved during the problem formulation and risk characterization phases of the ecological risk assessment process (USEPA 1992, 1998). To ensure that the risk assessment meets the risk manager's goals, management and policy input must be clearly stated prior to the risk analysis activities associated with exposure and ecological effects assessment. Problem formulation includes the resolution of interrelated questions on data interpretation (performance-based versus criteria-based practices) and the distinctions among risk analysis (complete process), risk assessment (determining risk), and risk management (dealing with risk).

Performance-based and criteria-based practices

Performance-based practices are those that specify design-focused evaluation of 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 frequently assess the wetland water quality function by some numeric value developed as a consequence of a regulatory objective; for example, water discharged from a remediation wetland must meet the drinking water standards for heavy metals. Evaluations of wetlands may integrate these concepts to varying degrees (Hammer 1990) with the regulatory context that may be associated with the risk assessment. 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.

Functions versus values and threats versus impacts

The relationships among risk assessment and risk management activities relative to wetlands may be markedly different, especially within the context of a technical characterization of wetland "functions" versus a more risk assessment-like consideration of wetland "values." The roles these potential differences play in evaluating "threats" and "impacts" of anthropogenic activities on wetlands are subsequently dependent upon clear distinctions being given to all these terms.

Wetlands generally are considered to have functions related to hydrology, water quality, and habitat. Hydrologic functions are generally characterized by capacity and input, which may define a wetland as a water source or water sink. Water quality functions are generally focused on physical (e.g., sedimentation and stabilization) or chemical (e.g., denitrification or contaminant removal) characteristics of surface water and ground water within the wetland. 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 v
1983; Adamus et al. 1987; Brinson
critical and plays a major role in mair
functions such as reproduction, feedin
benefits obtained by society from wet
would include flood control and the
function (Kentula et al. 1992; Richar
functions and values is not without t
relatively easy to address within risk a
terized as assessment endpoints wher
critical to their definition.

For wetland risk assessments, these t
assessment disciplines must be clearl
endpoints or measurement endpoints,
to communicate effectively with reso
threats and impacts to wetlands must l
setting. Within a risk assessment con
able disturbance or activities associate
1992), while impacts are anthropogen
that are associated with effects that re
verse." Risk management objectives m
clearly identify measurement endpoin
further analysis) differences between
ments. In order to develop cost-effect
function and value as well as threat ar
wetland scientists and those in the ri
assessors and managers must clearly
their risk assessment needs are suppo
the technical activities of ecosystem

Procedures for assessing and

Technical activities that support wetla
and federal governments (Table 4-2).
designed with wetlands as a chief foc
that make the guidance equally amena
biological assessment methods for eva
Corps of Engineers (USACOE) and U.
have both developed guidance for eva
Resource Conservation Service (forme
developed procedures for identifying
buster" provision of federal wetland
technical approaches developed by sta
when the assessment activities fall un

Table 4-2 A relative comparison of the applicability of technical approaches to risk or risk-related wetland assessment

		Risk analysis		Risk characterization			
Guidance	Problem formulation	Exposure assessment	Effects assessment	Integration	Uncertainty analysis	Risk summary	Ecological significance
Wetland delineation	+		+	+	–	–	+/–
Hydrogeomorphic classification	+		+/–	+			+/–
Wetland evaluation technique	+	–	+	+	–	–	+/–
Avian richness evaluation method	+		+	+			+/–
Synoptic wetland assessment	+		+/-	+		+/-	+/-
CERCLA risk assessment	+	+	+	+	+	+	+
Natural resource damage assessment	+	+	+	+	+	+	+

+: step explicitly included in process
–: step not explicitly included in process
+/–: step can be included depending upon case-specific implementation

(Adamus 1993a, 1993b). While the methods and guidance summarized in the following sections are not exhaustive, they are representative of the technical methods that are currently available.

Wetland delineation

The Federal Manual for Identifying and Delineating Jurisdictional Wetlands (FICWD 1989) was the first effort to bring together the 4 federal agencies that had primary responsibility for oversight of wetland management or enforcement of wetland regulations (USEPA, U.S. Fish and Wildlife Service [USFWS], USACOE, and SCS). Support for this manual was withdrawn in 1991 by Congress and since that time, USEPA, USACOE, and USFWS have agreed to accept the 1987 manual developed by the USACOE for delineating wetlands. The Natural Resources Conservation Service has developed its own manual to deal with lands that fall under the Farm Bill, a specific federally legislated funding and assistance authorization. The 1982 USACOE manual provides technical guidance to establish physical boundaries of wetlands and uses the following definition (USACOE 1982):

> Those areas that are inundated c
> frequency and duration sufficien
> circumstances do support, a prev
> life in saturated soil conditions.
> marshes, bogs, and similar area

Assessment of the status of wetland
They inventory wetlands periodically
make reports directly to Congress. H
precisely that of the jurisdictional de

Currently, wetland delineations are a
monitoring process. Depending upon
associated with any designed wetlan
the extent to which a wetland is char
completed following guidance from
(USACOE 1987) include evaluations
as part of the regulatory process und
methods of evaluation may be pursue
been categorized as screening, interm
screening-level evaluations may be pu
survey may be conducted to collect si
in and around a wetland at risk. Simil
bounds set in the study's design, whi
vegetation data for the wetland or an
identification and mapping of all plar
delineation effort. In completing the
tion, it should be noted that hydrolog
difficult to establish in the field, prin
annual, and seasonal) in water levels
considers wetland hydrology present
inundated sometime during the growi
geomorphic setting influences the int
drainage class of the soils must be cle

Hydrogeomorphic classification

In addition to the wetland delineation
supporting their Waterways Experime
procedure for assessing the functions
assessment process for wetlands (Bri

Wetland ecosystems in the U.S. occur
geomorphic, and hydrologic conditio
of assessing wetland functions diffic
in the same manner or to the same ext

process, USACOE has found it useful to classify wetlands into groups that function similarly. Classification narrows the focus on the functions of a particular type of wetlands and the characteristics of the ecosystem and landscape that influence these functions.

The classification procedure summarized below is intended primarily for evaluating the ability of wetlands to perform specific functions. The benefits of classification are a faster and more accurate assessment procedure, which supports the USACOE regulatory program mandated by Section 404 of the CWA. With this regulatory application in mind, hydrogeomorphic (HGM) classification can be used:

1) to compare project alternatives,
2) to compare pre- and post-project conditions for determining impacts or mitigation success,
3) to provide guidance for avoiding and minimizing project impacts, and
4) *to* determine mitigation requirements.

Hydrogeomorphic classification is modular in its design, and when compared to the risk assessment framework, its hierarchical format should make it easily adaptable to a variety of wetland risk assessment needs, including planning and management of various regulatory situations that involve the assessment of wetland function.

Wetland functions are the actions that are naturally performed by wetlands which result from the interactions among the structural components of a wetland-such as soil, detritus, plants, and animals-and the physical, chemical, and biological processes that occur in wetlands. A process is a sequence of steps leading to a specific end; for example, from a biological perspective, the microbially mediated process of denitrification occurs in many wetlands and leads to a relatively simple wetland function of nitrogen removal. Complex functions resulting from the interaction of structural components and multiple physical processes can also be identified; for example, the physical processes of overbank flooding, reduction of water velocity, and the settling of suspended particulates interact with physical structures and result in the wetland function of particulate retention.

Hydrogeomorphic classification categorizes or groups wetlands on the basis of 3 fundamental characteristics: geomorphic setting, water source, and hydrodynamics. At the highest level of the classification, wetlands fall into 1 of 5 basic HGM classes: depression, slope-flat, riverine, fringe, and extensive peatland.

Hydrogeomorphic classification's hierarchical design can be applied at a regional level to narrow the focus of the classification. For example, ecoregions identified by Omernik (1987), Bailey (1994), or Bailey et al. (1994) may be used as the next filter in the classification scheme. These ecoregions are defined in part by climatic, geologic, physiographic, and other criteria and provide a convenient starting point for applying the classification at a regional level. Within a region, any number of regional HGM subclasses can be based on landscape factors such as geomorphic setting, water source, soil type, and vegetation. While the number of regional

subclasses depends in part on the obj
process, within an ecoregion the num
diversity of condition in a region. Re
the assessment process within the co
Section 404 of the CWA.

The assessment procedure applies the
capacity, reference domain, and refer
the risk assessment framework outlin
the HGM procedure as 3 phases: ch

The characterization phase includes
1) definition of assessment objec
2) characterization of the propos
landscape context;
3) screening for "red flag" feature
4) identification of wetland asses
of HGM classification; and
5) physical separation and potent

Clearly, these elements of the charact
consistent with the elements found in
ecological risk assessment framework
considered in wetlands risk assessme
ment of HGM classification, but clea
more readily available to the risk ass

The assessment phase of the HGM pr
perform in terms of "functional capac
components and physical, chemical, a
Depending upon the function, functio
displayed depend upon interactions b
environment. For example, consider t
some wetlands and the concept of fun
ity, or the theoretical capacity of a riv
floodwater in this example, depends o
that determine the wetland's storage c
actual amount of floodwater stored in
watershed to generate overbank flood
of overbank floods. Watershed charac
intensity and duration of precipitatio
watershed, and the location of contro
wetland will influence the wetland's f
cantly from its inherent capacity.

Functional capacity of a wetland is described by the functional capacity index (FCI), which is a ratio of the functional capacity of a wetland under an existing or predicted condition to the functional capacity of a wetland under "attainable conditions." Attainable conditions are defined as the conditions under which the highest, sustainable level of functional capacity is attained across the suite of functions that wetlands in a reference domain naturally perform. The "reference domain" is simply the group of wetlands for which an FCI is developed. The reference domain normally is a regional HGM subclass, but depending on assessment objectives, it could be composed of a larger or smaller number of subclasses and geographic extent. For example, if the assessment objective is to compare a subclass of wetlands in the watershed, the reference domain would include all wetlands in the subclass in the watershed. Attainable condition, or the highest sustainable level of functional capacity, would ideally occur in wetlands that occur within landscapes that have not been subject to anthropogenic disturbance associated with long-term effects. When undisturbed wetlands and landscapes do not exist or cannot be reconstructed from historical data, attainable condition is assumed to exist in the wetland ecosystems and environments that have been subject to the least amount of anthropogenic disturbance.

Functional capacity indices are based on an assessment model that defines the relationship between the ecosystem- and landscape-scale variables and functional capacity. The condition of a variable is measured directly or indirectly using indicators that correspond to specific variable conditions. Variables are assigned an index ranging from 0.0 to 1.0, based on the relationship between variable condition and functional capacity in the reference domain that is established using reference wetlands. A "reference wetland" is a group of wetlands that represent the range of conditions that exist in wetland ecosystems and their landscapes in the reference domain. The range of conditions include those resulting from natural processes (succession, channel migration, erosion, and sedimentation) and anthropogenic disturbance.

Reference wetlands and their environments serve as the basis for scaling and calibrating variables in assessment models. The relationship between variable condition and functional capacity in the reference domain is established using empirical data, expert opinion, best professional judgment, or a combination of these options. The relationship is formalized by using logical rules or equations to derive an FCI ranging from 0.0 to 1.0. An FCI of 1.0 corresponds to the level of functional capacity that exists under attainable conditions for the reference domain, while an FCI of 0.0 reflects the absence of functional capacity. Functional capacity indices then provide measures of a wetland's capability to perform a function, relative to similar wetlands in the region.

As a result of the wetland assessment process, FCIs can subsequently be applied in various ways during the application phase. Functional capacity units (FCUs) can be calculated by multiplying an FCI by the area of wetland it represents. Once the

functional capacity of a wetland area
comparisons critical to regulatory pe
ample, comparing the same wetland a
project conditions), comparing wetla
time, and comparing wetlands in diff

Wetland evaluation technique

Various monitoring programs have f
evaluation programs have developed
assessing risk. Monitoring activities
may change risk through time (e.g.,
qualitative and quantitative approach
oped, these can be grouped into cate
i.e., the extent of spatial coverage ra
Methods designed for application to i
wetland evaluation technique (WET)
(Adamus et al. 1987), are focused on
wetland assessment for relatively sma

The WET assesses wetland function i
and opportunity and uses predictors
biological processes. These are simila
assessment as it is presented in curre
are generally qualitative but may refl
part of their contribution to wetland
evaluated in WET include

- groundwater recharge,
- nutrient removal,
- sediment retention,
- groundwater discharge,
- nutrient transformation,
- toxicant retention,
- floodflow alteration,
- production export,
- aquatic biodiversity,
- sediment stabilization,
- wildlife biodiversity, and
- recreation and heritage.

From an ecological perspective, WET
nity structure directly but assume co
basis of habitat structure (Adamus et
individual wetlands, it considers larg

topographic, and vegetation features, to develop qualitative estimates of wetland function and condition. These estimates take the form of ratings of high, moderate, or low for each function (except recreation), and in conjunction with a habitat suitability rating for fisheries, wildlife, and waterfowl, yield an evaluation for the wetland at risk. Within a given ecoregion, these qualitative estimates could be compiled to develop thresholds that could discriminate between each of the general categories of risk. While these methods are intended for individual wetlands (of limited spatial coverage), WET or similar methods have been applied to extensive wetlands characterized by many wetland types in a complex landscape. Many state regulatory agencies have applied wetland evaluation methods within their particular ecoregional setting, and as such, these methods may be available for use in wetland risk assessment (Roth et al. 1993).

Habitat evaluation procedures and their applications to wetlands

Evaluation of wetland habitats for wildlife relies on methods developed by the USFWS as habitat evaluation procedures (HEPs) (USDOI 1980). Habitat evaluation procedures use individual species models identified by habitat suitability index (HSI) models to generate a composite of key species within a habitat, but only a limited number of HSI models are available for application to wetland risk assessment. While past criticism has focused on HEP's species-level orientation as opposed to a community-level orientation, its application to wetlands risk assessment should be considered, especially if regulatory drivers fall along single-species lines (e.g., threatened or endangered species in critical wetland habitats). Given criticisms of HEP and similar assessment methods, alternative technical methods are being developed, including community-level metrics focused on bird community structure.

Avian richness evaluation method

The avian richness evaluation method (AREM) is one of the first rapid methods to be developed for assessing biodiversity (Adamus 1993a, 1993b). Without requiring extensive user knowledge of birds, it comprehensively addresses wetlands bird diversity and can be modified to predict diversity of other animal groups. The AREM does the following:

1) assigns a score to each evaluated wetland, which represents the number of bird species that could occur in the wetland multiplied by an estimate of the suitability of the wetland for each species;
2) creates a list of species likely to occur in the evaluated wetland that can be combined with lists predicted for other wetlands to identify minimum combinations of wetlands that will provide habitat for all bird species in an area; and
3) tallies the number of species likely to occur in the evaluated wetland and their particular characteristics, e.g., neotropical migrants, uncommon species, or

game species. If they desire, us
and use them as weights in der

The AREM was developed because sc
scores or ratings to wetland wildlife l
wetland has been rated high or low. K
one wishes to maintain biodiversity a
species composition allows one to avo
many narrow-niched, regionally unco
enhancing a wetland with a perhaps i
are mere generalists.

The AREM is intended to be used in
and can be applied in addition to, or in
scores it assigns are based on assessm
presumed indicator species. Many use
indicator species biases the results, an
validity of assuming that 5 to 10 spec
100 species that are present at a site.
adequately address habitat needs eith
periods, and HEP assessments are of

The AREM can be used to assist and
ways:

1) Performing mitigation calculati
typing" lands that will be alte
has been deemed necessary. Th
habitat before a project and est
among categories as a result of
that are believed to exist both l
coefficients, determined throug
of each category for selected sp
manner, net change in habitat s
predicted. Where wetland and n
expected to change, AREM migh
calculate the habitat suitability
nonwetland cover types are also
scribed above.
2) Diagnosing impaired wetland c
ered by agencies to be "waters
public trust lands (e.g., nationa
exists to determine the degree
Avian richness evaluation mod
assist. For example, they are u
tion problems by defining whic

wetland having a particular habitat structure. If properly designed surveys then fail to find the predicted species, it raises a possibility that nonphysical (e.g., chemical) factors unmeasured by AREM are discouraging wetland use.

3) Selecting appropriate indicator species. By defining which species to expect in particular types of wetlands, AREM can assist resource personnel in selecting indicator species that are the most appropriate for monitoring water quality or physical habitat suitability. Selecting appropriate indicator species is crucial to the proper use of HEP as well as to the development of biocriteria for wetland protection and the accurate monitoring of wetland contamination.
4) Targeting habitat enhancements. Active management of wetlands will usually be most effective when it focuses on improving conditions for species with low species habitat scores, while maintaining conditions suitable for species with high species habitat scores. In combination with other considerations, AREM can be used in this manner to suggest habitat features whose enhancement will support the largest variety of species overall or of species having a particular attribute.
5) Establishing wildlife-based classification of wetland habitats. Wetland types are commonly defined by their vegetative communities. Wildlife communities or individual species also can be useful primary or secondary features in classifying wetlands for scientific or administrative purposes. Avian richness evaluation models can assist such classifications by predicting bird species associated not only with vegetation but also with other environmental factors. Statistically defined, wildlife-based classes of wetlands could be identified by applying AREM to a probabilistic sample of wetlands in a region.
6) Optimizing biodiversity protection. Agencies and conservation groups sometimes have opportunities to purchase or trade properties to enhance regional biodiversity. When biological survey data from the subject properties are lacking, AREM can be applied (during any season) to the properties to predict their avian richness, which is often the largest terrestrial component of a region's vertebrate biodiversity. Richness estimates then can be calculated from the lists of predicted species pooled from multiple wetlands to determine which combination of wetlands is likely to support the greatest species richness. This estimate can be focused further by applying constraints related to land ownership, species characteristics, management costs, or other factors. As such, AREM can provide a complimentary, local refinement of the gap analysis approach currently used for ecosystem management and biodiversity planning at state and regional levels by the National Biological Service.

To date, AREM has been applied to only one ecoregion (the Colorado Plateau), but it was designed for easy adaptation elsewhere. Depending on the situation, the up-

front investment required to adapt AR
order of 0.1 to 0.5 full time equivalent
validation is also desired. Adaptation r
sional field ornithologist or expert bir
cal approaches to building habitat mo
comprehensive review of appropriate
encoding of preliminary models, modi
with local avian experts at several hab
questionnaire. The optional validation
conducting fauna1 surveys, data entry,
adapted for an ecoregion and/or habit
completed in less than 30 minutes an
computer programming expertise is re

Synoptic approach to wetland risk as

As indicated in the previous section, w
geographic scales, ranging from indivi
multiple individual sites may be embe
a single complex wetland may exist ov
Everglades and the bayous of Louisian
consistent, the level of effort required
watershed level-precludes identical m
characterize these wetland features.

A synoptic approach to wetland risk as
spatial coverages, e.g., ecoregions or st
opposed to single occurrence events. F
approach differs from WET in its rout
theless, the synoptic approach may be
one were considering a highly heterog
embedded wetland types or developing
WET, the synoptic approach to wetlan
input, which reflects in part the greate

The synoptic approach is designed for u
intended to relate cumulative impacts t
geographic scales. It is not designed to
spatial scales where WET may be more
Adamus et al. 1987). The synoptic app
et al. 1992) (Table 4-3). but from a tec
indices and the selection of landscape i
assessment completed using the synop

Overall, synoptic indices are those act
of interest, while the landscape indicat

Table 4-3 Steps in conducting a synoptic approach to wetland risk assessment

Steps	Inputs
Define goals and criteria of the assessment	Define assessment objectives Define intended use Assess accuracy needs Identify assessment constraints
Define synoptic indices	Identify wetland types Describe natural setting Define landscape boundary Define wetland functions Define wetland values Identify significant impacts Select landscape subunits Define combination rules
Select landscape indicators	Survey data and existing methbds Assess data adequacy Evaluate costs of better data, Compare and select indicators Describe indicator assumptions Finalize subunit selection Conduct pre-analysis review
Conduct assessment	Plan quality assurance and quality control Perform map measurements Analyze data Produce maps Assess accuracy Conduct post-analysis review
Prepare report of synoptic assessment	Prepare user' s guide Prepare assessment documentation

indices. In general, 4 generic indices are the focus of the synoptic approach—wetland function, wetland value, functional loss, and replacement potential-but each application of the synoptic approach will require that a specific set of functions be identified. Defining wetland functions and values in each synoptic assessment will require an understanding of the interactions among wetlands and the regional landscapes. In practice, each of these elements of the synoptic approach is dependent upon the particular goals and constraints acknowledged in the initial step of the process in which risk assessor and risk manager define goals and criteria of the synoptic assessment. Each step of the synoptic assessment process requires multidisciplinary inputs, which will include technical information such as identification of specific wetland types found in the area of concern and descriptions of natural settings, as well as definitions of wetland values which may be more policy-related than technical.

Methods applicable to National System permit process

For surface waters (including inland f marine waters), an integrated strategy data requirements has been applied b protect water quality beyond the techr method for measuring the biological ef whole effluent testing. The USEPA anc effluent testing to assess compliance w the National Pollution Discharge Elim effluent limitations necessary to attair guidance documents designed to suppc should be consulted as part of the wetl ance for the application of these tools chemical and biological approaches fo and biological testing requirements; us monitoring. For wetlands in particular Water Quality-based Toxics Control (U revised TSD provides an explanation o testing and gives detailed guidance on limitations for toxic pollutants.

In its application to wetlands risk asse control for the protection of aquatic lif to measure the toxicity of wastewaters. standardized, surrogate freshwater or measure the aggregate toxic effect of an typically a test of 96-h or less in durat point. A chronic whole effluent test is effects such as fertilization, growth, ar to lethality. Again, numerous technical that focus on these methods, and their p ment, especially at the organismic level strated (Warren-Hicks et al. 1989; USEI et al. 1994).

Given the policy implications of the NI methods and applications developed as "as is" or in modified form for a wet rity (IBI), index of community integrit ren-Hicks et al. 1989). This section brie risk assessment context, measurement e wetlands. Approaches available for we quantitative-consist of methods comm macroinvertebrates, and fish in a variety of aquatic habitats. Measurement endpoints consist primarily of direct and derived measures of population and community structure, such as relative abundance, species richness, and indices of community organization (e.g., USEPA 1973, 1987; Plafkin et a!. 1988; APHA 1992).

Risk assessment practices associated with CERCLA and similar regulations

Risk assessment activities pursued under CERCLA, or "Superfund," have become increasingly well documented since the Superfund Amendments and Reauthorization Act was promulgated in 1986, and the CERCLA process for conducting ecological risk assessments at contaminated sites has been summarized in numerous publications. When implemented for wetlands, the ecological risk assessment approach completed under CERCLA (Figure 4-4) is clearly rooted in the USEPA framework approach (1992, 1994c, 1997).

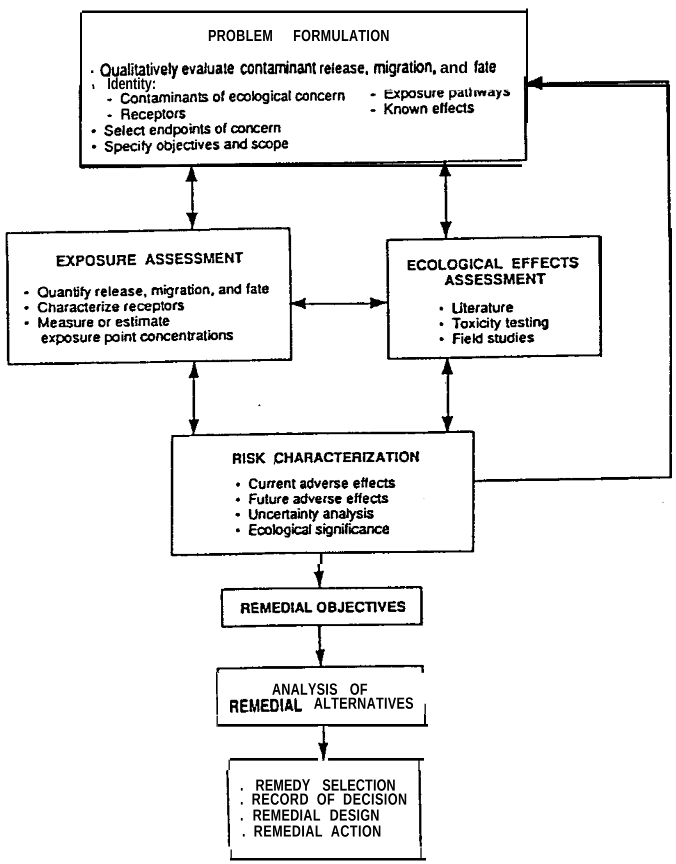


Figure 4-4 Ecological risk assessment approach used in CERCLA or Superfund investigations

In planning ecological risk assessment: scribed to ensure that data sufficient decision-making process. As indicated tion, exposure assessment and ecologi and risk characterization.

Problem formulation

Early in problem formulation, plans sl requirements for a qualitative ecologic formulation are generally identified as

1) qualitative evaluations of conta
2) identification of contaminants a
3) identification of exposure path
4) selection of ecological endpoint

These steps should be carried out withi landscape setting, habitat can be used a (e.g., habitat provides an ecological se ated with multiple wetlands in a partic

The outcome of the problem formulati evaluation often takes the form of con specific set of objectives designed to a process (e.g., public concerns, natural of the assessment required to answer t

Ecological effects and exposure assess

Following the problem formulation ph resources at risk at any wetland, an eco completed parallel to an exposure asses risk evaluation. Within the framework p plished through a review of existing inf experts familiar with the wetland and s are critical first steps in assessing cont complete. The exposure assessment that effects assessment considers in detail tl nants of potential ecological concern (information, the exposure assessment sl as well as the ecological receptors, bey formulation phase. When available wit formulation, estimates of exposure poi characterized.

To evaluate wetland risks, an ecological effects assessment should include

1) a review and summary of historic data, as well as comparative data gathered from peer-reviewed literature and surveys of local experts;
2) a review and summary of adverse biological and ecological effects associated with chemicals and radionuclides potentially of concern; and
3) a collection of the existing field survey information for the wetland (e.g., monitoring data on wildlife or previous wetland evaluations).

Risk characterization

From some perspectives, an ecological risk assessment may be considered an integrated evaluation of biological effects, derived through measurements of exposure and toxicity. From an ecotoxicological perspective, however, exposure and ecological effects assessments are complex, interrelated functions that yield estimates of risk associated with environmental contaminants in various matrices sampled at a site. Within the risk characterization phase of a qualitative evaluation of ecological risks, the outputs from the exposure and ecological effects assessments are integrated. In screening-level efforts, the integration relies heavily on strength-of-evidence arguments developed on the basis of the existing information for the facility or site. While screening-level efforts and comprehensive studies supporting the more quantitative applications of the ecological or ecotoxicological risk assessment approach differ with respect to levels of effort involved with their development (e.g., time or budget constraints), risk characterizations within any ecological risk assessment should include:

1) an evaluation of current and potential adverse biological or ecological effects,
2) an identification of the uncertainties associated with the risk characterization, and
3) an evaluation of the ecological significance associated with the contaminants or the physical disturbances associated with contaminant-related facility or site management.

In the past, risk assessments for wetlands under CERCLA were often completed as part of groundwater and soil contamination evaluations completed within the risk assessment process for a particular site; such efforts, however, may not capture the characteristics of the wetland within an ecological context. For example, groundwater evaluations completed in lacustrine, palustrine, or riverine wetlands frequently provide data sufficient for the groundwater risk assessment but may inadequately characterize the ecological context within which the ground water occurs.

As one approach to risk assessment for wetlands, guidance under CERCLA was designed to be flexible and implemented with varying degrees of effort, depending upon the landscape setting of the wetland at risk. The ecological risk assessment activities could range from being qualitative yet extensive efforts consistent with the current state of the science to comprehensive projects requiring multidisciplinary

teams of applied ecologists, research
et al. 1989; USEPA 1989, 1991a, 1992,
an ecological risk assessment based u
freshwater habitats will be discussed b
being given to the approach as it relate
throughout the United States.

Regardless of the regulatory and politi
wetland habitats at risk may be evalua
cal risk assessment that has been deve
designed for hazard assessment (Warre
grated or "ecological triad" approach e
physicochemical data in their ecologic
Pascoe et al. 1994; Linder et al. 1994)
tive tools are used to describe these rel
the ecological triad integrate biological
chemical (including contaminants) inf
(Figure 4-5). In addition to risk-driven
related to chemicals or radionuclides, t
and ecological systems also considers
exposures. Noncontaminant-related eff
regarded equally with contaminant effe
proach, especially when remediation,
evaluated within the context of ecologi

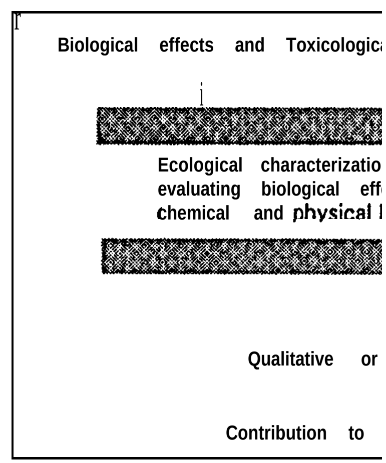


Figure 4-5 Sources of information (biological
ecological) that contribute to ecological risk as

In order to implement any ecological risk assessment, the existing regulatory guidance available from federal and state governments must be considered early in the project's organization (USEPA 1986, 1991a, 1992, 1997, 1998). For ecological risks in wetlands, the CERCLA approach is consistent with the framework document (Figure 4-6; USEPA 1992) and may be considered an integrated evaluation of' ecological effects and exposure (USEPA 1991a). Within an ecological assessment, qualitative risk evaluations should consider physical, chemical, and biological interactions associated with contaminant exposures in various environmental media, e.g., soils and surface water.

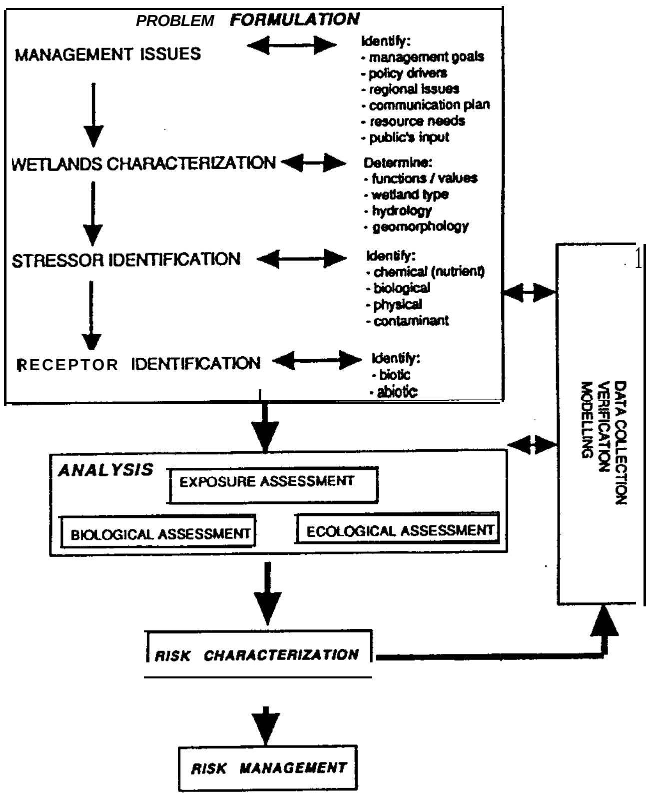


Figure 4-6 Overview of ecological risk assessment as summarized in the USEPA Framework (USEPA 1992)

Such a qualitative evaluation of risk may be approached at various levels of effort and according to various assessment strategies. Integrated ecological risk analyses supporting wetland risk assessments are increasingly being designed under CERCLA, especially if endangered species, critical habitats, or relatively large spatial scales are of concern. Depending upon the level of effort required to satisfy the data

quality objectives for any particular
activities may use

1) a "desktop" analysis of existin
2) a screening-level analysis, or
3) an integrated field and labora

As suggested by their names, these p
different levels of effort that are con
guidance from the interagency manual
HGM assessment process. When a full
completed for wetland risk assessmen
be fully developed from the existing
the risk assessment process.

Wetland risk evaluations then may ra
screening efforts to comprehensive in
The level of effort and implementatior
data quality objectives and scope of th
problem formulation phase of the risk
1997, 1998). Regardless of the level o
approaches to risk evaluations have m
assessment (Parkhurst et al. 1989). Fr
each strategy may be combined to eval
risk assessment setting. For wetlands,
are interrelated functions that will yie
associated with environmental contam
In the process developed for wetlands
tive but extensive screening-level effoi
existing data and include a survey of t
(e.g., historic wetland-specific and co
ecological effects), as well as gatherin
familiar with historic and current statu
are apparent or more detailed informa
tions following initial screening studie
field and laboratory investigations coı
uncertainty (Linder, Bollman et al. 19

Natural resource damage assessı

Wetlands, as highly valued natural res
resource damage assessment (NRDA)
become impaired and their values sub
Pollution Act of 1990 (OPA), the Nat
(NOAA) and other natural resource trı
and tribal governments) have develope

on aspects of the NRDA process that may benefit the wetlands risk assessment process as it develops.

There are 3 phases of an NRDA:

1) The preassessment phase in its simplest characterization requires that the trustees determine whether an incident has occurred (under OPA, the release of oil to the environment) and whether to pursue restoration planning.
2) The restoration planning phase has its central focus on the evaluation of information on potential injuries and the application of that information to an evaluation of the need for and type of restoration.
3) The restoration implementation phase is designed to ensure that the trustees implement the developed restoration plan.

The phases of the NRDA process consider questions that are not unlike those of the risk assessor and risk manager working under CERCLA. At this time in the development history of the NRDA regulatory process, the available guidance documents suggest that the technical support for risk assessment and NRDAs will be very similar. The activities currently included in the NRDA process drive the technical support toward this similarity.

Within the restoration planning phase, the current NRDA practice addresses 2 issues:

1) a primary restoration that evaluates alternative actions proposed to return the injured resources and services to baseline or reference states, including a natural recovery option and
2) a compensatory restoration in which actions are evaluated to compensate the environment and public for the resource or services lost from the date of the incident to the recovery of the injured resources.

The type and scale of compensatory restoration is related to the type and scale of primary restoration,selected, and the scaling of appropriate compensatory-restoration alternatives is primarily achieved on a service-to-service comparison of services lost as a result of the incident. When service-based cost assessment is not feasible or appropriate to the incident, compensatory restoration may also be determined through a cost analysis of lost services and gains from the compensatory restoration (see Federal Register 1995).

As in ecological risk assessment, public participation is integral to the NRDA process, particularly because that public input shapes policy in many instances. The timing and extent of public involvement in the NRDA process, and the type of documents produced at various stages of the process, fit the scope and scale of the incident in a manner distinct from yet analogous to CERCLA. In part, this stems from the past development of technical guidance by the Department of the Interior (USDOI) for assessing natural resource damages resulting from hazardous substance releases under CERCLA and the CWA. The CERCLA regulations originally applied to natural resource damages resulting from oil discharges and hazardous substance

releases. When proposed guidance is fi
currently in place for oil spill and othe
umbrella.

Wetland risk assessments could also b
under the National Comprehensive Pla
similarly, could provide procedures an
could better determine appropriate res
services (FEMA 1992). Natural resourc
services or ecological functions that th
public, and as a result, wetlands would
perspective, such services could be clas

1) ecological services, or the physic
one natural resource provides fo
from predation, nesting habitat,
2) public services, or the functions
(e.g., fishing, hunting, nature p

Value, as proposed for an NRDA action
sents the amount of other goods that an
good or the amount an individual will a
value of a natural resource or service in
als derive from consuming or viewing a
(values not linked to direct use, e.g., th
natural resource exists). In many contex
sented in terms of units of currency, th
However, value also can be measured in
service. In this proposed rule (Federal R
either units of resource services or doll
definitions may yield subtle but signific
an NRDA and those in the HGM, WET,
appear similar upon initial inspection.

From a strictly technical position, the p
and for a CERCLA are very similar. F
include adverse changes in survival, gr
and biological condition; behavior; con
and functions; physical and chemical ha
public, which are not unlike assessmen
assessment process outlined in the fram
injury often is thought of in terms of ad
injury proposed under OPA is broader. I
of oiled sand on a beach) as well as in
associated with a fisheries closure to pr
fish themselves may not be injured) ma

Determining exposure in an NRDA under OPA means determining whether natural resources came into contact with the oil from an incident. Early determination of exposure during the preassessment phase should focus on those natural resources or services that are most likely to be affected by an incident. In a manner similar to the analysis phase in risk assessment, an NRDA for a wetland impacted by an oil spill must determine whether the natural resource came into contact, either directly or indirectly, with the discharged oil. Exposure in an NRDA is broadly defined to include not only direct physical exposure to oil but also indirect exposure (e.g., injury to an organism as a result of a foodweb disruption).

Documenting exposure is a prerequisite to determining injury, except for response-related injuries and injuries from substantial threats of discharges. Evidence of exposure alone may not be sufficient to conclude that injury to a natural resource has occurred (e.g., the presence of petroleum hydrocarbons in oyster tissues may not, in itself, constitute an injury). Exposure can be demonstrated with either quantitative or qualitative methods. As with other elements of the NRDA process, selection of approaches for demonstrating oil exposure will depend on the type and volume of discharged oil, the natural resources at risk, and the nature of the receiving environment. For example, chemical analysis of oil in sediments, alone, may not be adequate to conclude that a benthic organism was otherwise exposed to the oil. Likewise, the presence of petroleum in fish tissue, alone, may not be ad-' equate to link the exposure to the discharge because metabolism of the oil may blur the chemical characterization. The combination of the 2 approaches may, however, demonstrate exposure. As in the ecological triad applied in the risk assessment for the wetlands at Milltown Reservoir (see Chapter 5), exposure analysis should typically include field observations or measurements, laboratory exposure studies, transport and fate modeling, and a search of the literature. As proposed, the NRDA process emphasizes that these procedures may be used alone or in combination, depending on the specific nature of the incident. The trustees must determine the most appropriate approach to evaluating exposure on an incident-specific basis.

As in ecological risk assessment, pathway analysis is a critical component in the injury assessment phase of an NRDA. In a wetland, for example, pathways would include movement and exposure to oil through the water surface, water column, sediments (including bottom, bank, beach, floodplain sediments), ground water, soil, air, direct accumulation, and food-chain uptake. Pathway analysis includes field investigations, laboratory studies, modeling, and the reviewing literature. Again, the current practice emphasizes that these procedures may be used alone, or in combination, depending on the specific nature of the incident. The most appropriate approach to determine whether a plausible pathway exists would vary on an incident-specific basis.

To determine whether an injury resulted from a specific incident, a plausible pathway linking the incident to the injury would have to be identified, but similar to exposure, the existence of a pathway between source and target is not sufficient to

conclude that injury has occurred (e.g
can be used to document that a plausi
such data do not, by themselves, dem
Pathway determination can include e

1) the sequence of events by whi
incident and came into direct
resource (e.g., oil transported
wave action directly oils shellfi

2) the sequence of events by whic
incident and caused an indirect
(e.g., oil transported within a w
reduced populations of bait fish
eating bird; or, oil transported
action causes the closure of a f
being marketed).

Pathway determination does not requi
directly exposed to oil. In the example
as a result of decreases in food availab
the existence of a plausible pathway re
resource or service, even if the injury i

As evidenced by the discussion of exp
the technical methods employed for w
those supporting the NRDA process. C
between NRDA and CERCLA confoun
function. Under NRDA, for example, i
trustees determine the degree and spat
supports the selection of appropriate r
process does not include restoration ac
or more of several different conceptua
may be quantified in terms of

1) the degree and spatial or tempor
2) the degree and spatial or tempor
subsequent translation of that c
the natural resource, or
3) the amount of services lost as a

Within the context of injury quantifica
terms of percent mortality; proportion
habitat affected; extent of oiling; availa
temporal extent of the injury. Quantifi
injured by oil could be obtained by es

1) total number of acres of severel
killed,

2) natural recovery time for severely oiled wetland,
3) total number of acres of moderately oiled wetland in which vegetation is not completely killed but the wetland has lower levels of productivity, and
4) natural recovery time for moderately oiled wetlands.

This information could then be combined to quantify the total number of "acre-years" of wetland injury to scale restoration actions.

An analysis of natural recovery, or the return of injured natural resources and services to baseline in the absence of restoration activities, may include evaluation. of factors such as degree and spatial or temporal extent of injury, the sensitivity of the injured natural resource or service, reproductive potential, stability and resilience of the affected environment, natural variability, and physicochemical processes of the affected environment.

While it is beyond the scope of the present discussion to provide a detailed technical document to support either NRDA or CERCLA ecological risk assessment for wetlands, many of the technical methods applicable to the NRDA process—especially the injury and restoration assessment phases-are currently available and being used in wetland risk assessment (see section "Methods and endpoints for wetlands" and Table 4-4).

Table 4-4 Representative technical references for aquatic and sediment biological test methods for evaluating risks in wetland habitats

Test matrix	Target biota	Reference
Freshwater	Vascular plants	Wang 1991; ASTM 1997a
Freshwater/marine/estuarine	Algae and vascular plants	Swanson et al. 1991; ASTM 1997a
Freshwater	Aquatic vertebrates and invertebrates	USEPA 1990; Weber 1993; ASTM 1997b
Marine	Marine or estuarine invertebrates and vertebrates	Weber 1993; Klemm et al. 1994; Chapman et al. 1995; ASTM 1997b
Freshwater sediments	Epifauna, infauna, and vertebrates	USEPA 1994b; ASTM 1997b
Marine/estuarine sediments	Epifauna. infauna, and vertebrates	ASTM 1997b

For injury assessments for wetlands, whenever practicable, procedures should be chosen that provide information of use in determining the restoration appropriate for that injury, and frequently a range of assessment approaches, from simplified to more detailed, should be considered. In general, more detailed assessment procedures may include, alone or in any combination,

1) field investigations,

2) laboratory methods,
3) model-based methods, and
4) literature-based methods.

Technical support for evaluating prim
consistent with many of the technical
risk assessment. Within a NRDA, trust
satory-restoration action as well as a
alternatives. Here, a scaling of compe
ate. For example, in a wetland restorat
service approach may be appropriate a
mitigation analysis. Here, under a serv
appropriate quantity of replacement se
lency between lost and replacement se
differences in habitat value. As current
service approach for evaluating alterna
same type and quality and are subject
conditions as those lost. This proposed
developed by NOAA in response to OP
process when lost resource services are
habitat or biological resources like we
may be used to scale restoration projec
that support multiple species or that re
of resource services. To ensure that th
project does not over- or undercompen
trustees must establish an equivalency
lost services and the present value of th
compensatory-restoration project over

Trustees may use any reliable method f
site-specific application of one of these
reasonable cost criterion, the trustees m
using benefits transfer. The choice of a
upon the types of injuries and the type
restoration alternative. Trustees should
ing the value of the lost services and th
compensatory-restoration alternatives.
trustees should take steps to ensure tha
bias.

To evaluate restoration, monitoring act
process. As in the monitoring tasks that
cal risk assessments, monitoring plans
study design elements such as duration
evaluate progress and success, the inten

or the need for corrective action, and monitoring of a control or reference site to determine progress and success. To evaluate success of restoration actions, performance criteria may be developed which evaluate structural, functional, temporal, and other goals. For example, an agreement to create new marsh habitat as compensation for marsh impacted by oil could be described by performance criteria including the number of acres to be created, the location, the elevation of new habitat, the species to be planted and details for planting, such as density, and the time frame in which identifiable stages of the project should be completed.

Strengths and limitations of current risk assessment approaches for wetlands

From a technical perspective, each of the regulatory-associated practices considered above may be compared relative to the steps outlined in the USEPA framework approach (Figure 4-6). In a strict sense, no one method is best nor was any originally developed for wetlands risk assessment. Each has been molded, however, to assure their implementation for risk assessments mandated by law and regulation. In many respects, each approach summarized in this section, as well as those not included in this discussion (but available from many states and other federal agencies), requires technical support from wetland scientists, ecotoxicologists, and applied ecologists. Each approach identified in Table 4-2, for example, includes guidance for reviewing existing information for the risk assessment process or, alternatively, for designing and completing studies or surveys to address questions identified in the early phases of the risk assessment process. Similarly, each approach recognizes the importance of evaluating ecological effects, although the linkages between stressors (especially chemical stressors) and ecological effects are more thoroughly explored in some implementation plans than others. For example, explicit guidance for evaluating exposure is poorly described in some strategies for evaluating wetlands, but these guidance documents are also better developed for an analysis of physical stressors that may have impacted a wetland as a consequence of changes in land-use practice, e.g., synoptic wetland assessment versus CERCLA risk assessments. Shared limitations among all approaches include problems associated with interpreting existing information within a risk context, especially in comprehensive risk assessments that rely on statistical methods. Here, for example, data quality issues cut across all approaches, and regardless of the risk strategy employed, each shares problems related to inter-study comparisons and their interpretations, data pooling, and statistical issues related to encountered data.

Overall, the strengths and limitations of each approach considered here, as well as other approaches addressing similar risk-related questions, reflect the policy and management issues that are critical to the process, as noted in the USEPA Framework (1992, 1998). The technical support tools available for ecological risk assessment are numerous (see, e.g., "Methods and endpoints for wetlands," this chapter). But to ensure that the best available state-of-the-science is implemented to support wetlands policy and management, clear lines of communication must exist among

the policy, management, and scientifi
process, and the risk assessment must
the site.

The Ecosystem Approach: Geomorphology,

Abiotic characteristics of fresh

Freshwater wetlands represent a host
surface water some time during most
thread, they vary greatly in characteris
ogy, and hydrodynamics have acted ov
diverse and dynamic nature of these

The purpose of this general overview o
wetland characteristics is to point out
generic functional traits that should be
ment. However, these are general guid
to determine whether expected conditi
in this context, it should prove helpful
studies on relevant issues.

Climate

For wetlands to occur, there must be ex
upland drainage areas. A simple form

d

where dS = storage, P = precipitation,
wetlands, ET tends to dominate this e
ET may exceed P so that no water is av
occurs either by overland flow and/or s
sions, thereby creating wetlands. Such
state and federal agencies, and they pr
of wetlands within a region (Figures 4
exceeds 0 from August through April (
wetlands to be highly evident during d
growing season. In contrast, at Fort La
March through November; thus, wetlan
during the summer months or growing

Geomorphology

Geomorphology is the landscape posit
the runoff and storage of water (Brins

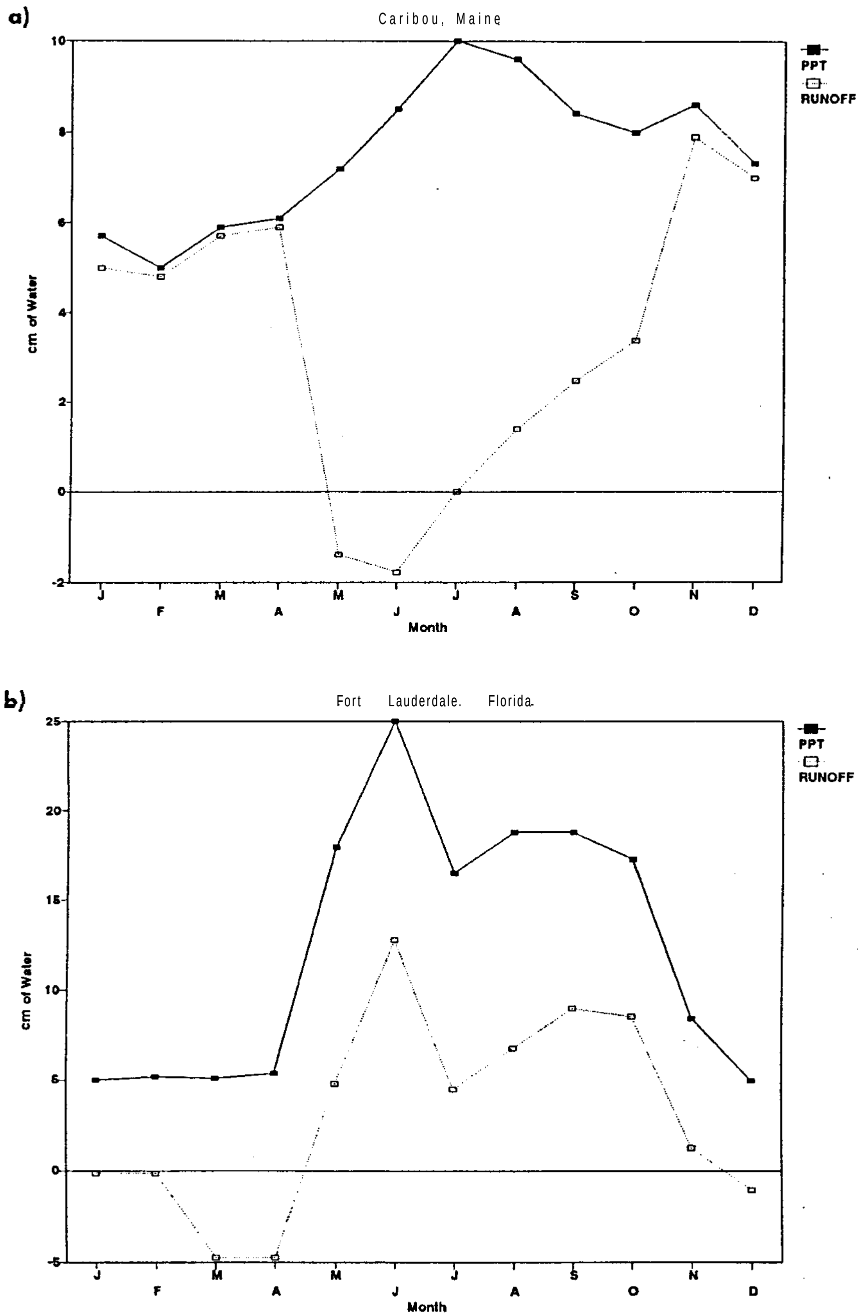


Figure 4-7 Relationship of total rainfall in cm to runoff in cm for a) Caribou, ME and **b)** Fort Lauderdale, FL

is generally linked to runoff and wet
and fringe categories of geomorphol

Depressional wetlands include such l
Carolina bays. They frequently occur
depend heavily upon local precipitat
settings. In climates where ET exceed
dry much of the time, or they depen
1993). In climatic regions where R >
sions may accumulate sufficient peat
types of wetlands receive their water
overbank flooding.

Extensive peatlands are usually the
depressions, followed by radiating p
create domed landscapes where the h
sole source of water. These are gene
cover large areas of land such that t
storage of water, the mineral nutritio
landscape (Moore and Bellamy 1974)
cation across the landscape may dev
the underlying topography. As a cons
ter ombrotrophic wetlands with diffu
like characteristics (Siegel and Glase

Riverine wetlands form as linear strip
rated from the stream channel by nat
most of the floodplain in large rivers
but may be very small or nonexistent
of the South (**Theriot** 1988; Hook et
flashy in low-order streams to long a
the stream channel determines wheth
nately erosional or depositional.

Freshwater fringe wetlands are restri
estuaries. These types of wetlands are
some may be headwaters (nonalluvial
into rivers near estuaries.

Hydrodynamics

The source of water for freshwater v
discharge, surface or near-surface inf
depressional wetlands receive their w
wetlands occupy depressions in the l
level (Figure **4-8a).** They are generall
relatively impermeable soil that rest

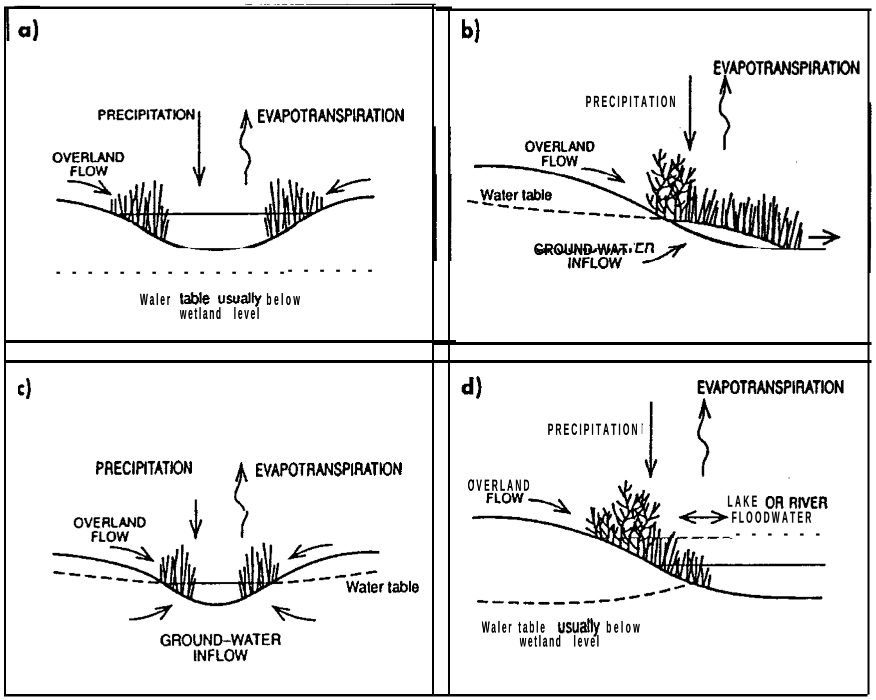


Figure 4-8 Four major hydrologic types of wetlands in Wisconsin: a) surfacewater depression, b) groundwater depression, c) groundwater slope, and d) surfacewater slope (after Brinson 1993)

through the soil. Therefore, the dynamics of the water table are vertical. It moves up when it receives runoff and down primarily due to ET. 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. Research in Florida has shown that the cypress domes may be more interconnected than originally thought (Riekerk 1993). Depending on size, geomorphology, and regional location, they 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 productivity. However, productivity may vary with the geology, climatic conditions, and types of soils and vegetation that develop.

Some depressional wetlands receive ground water in addition to runoff from precipitation (Figures 4-8b, 4-8c, 4-8d). If the groundwater table intersects the slope

at or within the depression, water en
water may enter wetlands or create
intersects the soil surface (Figure 4-
springs. However, relatively large w

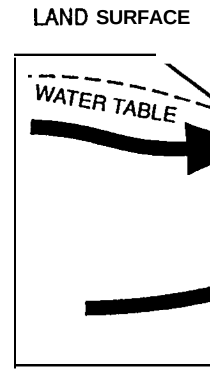


Figure 4-9 Relationship of land surface and

If ground water enters a wetland, it l
an aquifer or soil. Depending on the
lithology, such water normally has l
precipitation. Consequently, plant cc
ter discharge tend to be more produ
Furthermore, the hydrodynamics of t
precipitation-driven wetlands (i.e., dı
The dynamics of the water table in th
relation to water inputs and outputs (

The source of water in riverine wetla
water, and precipitation. The domina
extensive exploration. A study in the
fourth-order stream received periodic
during the dormant season. However
driven entirely by precipitation (Hool
stream in coastal Georgia, water cam
season as well as the dormant season
watershed, precipitation and ground
degrees depending on topographic re
topography had important influences
between flood events (Saul 1995).

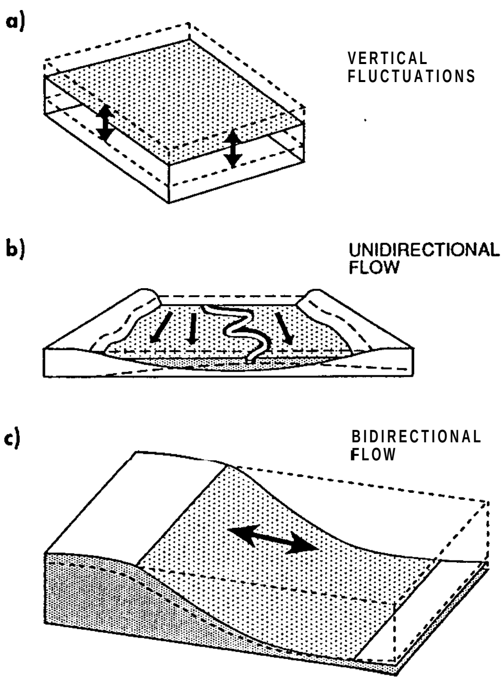


Figure 4-10 Categories of hydrodynamics based on dominant flow pattern: a) vertical fluctuations normally are caused by evapotranspiration and precipitation, b) unidirectional flows are horizontal surface and subsurface, and c) bidirectional flows are horizontal across the surface (after Brinson 1993)

The water in a floodplain tends to flow unidirectional down stream (Figure 4-10b), but depending on topography 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 flows similar to those in estuaries (Figure 4-10c). The variation in hydrodynamics among wetlands and within localities of a wetland must be carefully considered if contaminant studies are to successfully identify key transport and exposure pathways to biota.

Biogenic and fluvial deposition in wetlands tend to be causally related to water flow rate (energy; Figure 4-11a). Hydrologic energy, hydrodynamics, nutrient availability, temperature, salinity, fire frequency, and herbivory are also related in a general manner to wetland type and core factors (Figure 4-11b).

When a wetland has 2 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 contaminants may be delivered, retained, and transported. Some rivers have stream gauges

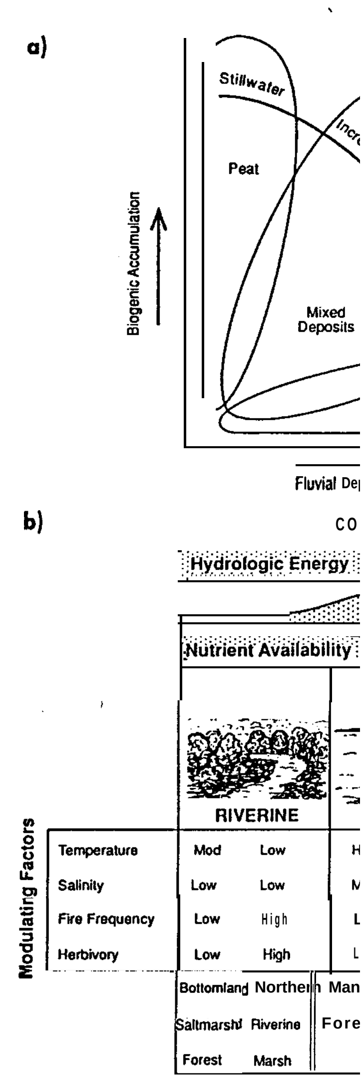


Figure 4-11 a) Relation of water turnover to l wetlands b) The use of core factors and modı (after Brinson 1993)

maintained by the U. S. Geological Su records are invaluable for ecological a various wetland functions. In the absen ric (soil saturation) studies are necessa wetland. Problems arise in determinin be useful. For example, a 38-y record f demonstrated that, depending on which 3-or 5-y period was selected for measurement, the site could be classified either as a wetland or nonwetland using jurisdictional criteria (W. Skaggs, personal communication).

Use *of soil surveys*

Many, if not most, counties in the United States have surveys of the soils. The surveys contain more 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 (OM), 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 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. The U.S. Department of Agriculture Natural Resources Conservation Service generally has offices in each county, with trained personnel who can help interpret soil survey information and sometimes assist with actual field checks. In addition, county agents and university extension personnel may be available to help interpret the data or provide guidance on where to seek help.

Integration of abiotic factors

The salient characteristics of wetland ecosystems are embodied in the integration of local climatic conditions 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. 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 or to actively transport pollutants or nutrients out of the system. Furthermore, the system would not be expected to be highly productive, but caution is needed for the latter. If the depression has a groundwater source that is rich in nutrients, its productivity may be high; thus, it could act as an efficient buffer or transformer. Additionally, the amount of OM in the soil will influence its potential to support microorganisms for decomposition and other soil reactions.

Geomorphic settings and traits can be a practical starting point for identifying the basic type of wetland as well is its principal ecosystem functions and associated ecological significance (Table 4-5). Moreover, the relationship of abiotic factors to wetland characteristics is useful for identifying those generic functional traits that should be addressed in a wetland-specific risk assessment. The approach can be simplified to a protocol that incorporates 7 steps:

1) Determine the geomorphic setting. Is it a depression or basin, a riverine system, or a fringe wetland?
2) Determine the dominant source of water. Is it rain water, ground water, or overbank flooding?
3) Determine the dynamics of the hydrological mechanisms. Does the water table fluctuate vertically? Is it primarily unidirectional or bidirectional? Do the dynamics change with water-table level or season? Use the water balance equation and determine when R exceeds 0. This will identify the seasons or times that the water table is apt to be the highest or lowest.
4) Use all available resources, i.e., aerial photographs, maps, interviews with local people, field reconnaissance in and around the wetland, to determine if the hydrology has been significantly altered. If it has, try to determine how the alterations may have affected the hydroperiods, timing, frequency, and flow patterns that would be expected to be associated with the existing geomorphic setting.
5) Use soil surveys to determine soil series, texture, drainage class, vegetation, hydroperiods and hydrodynamics, and the relative productivity based on site index or other site productivity documentation in the survey.
6) Scout the entire area to determine the patterns of inundation, vegetation types, and vegetation densities to identify any zones or patterns that may affect how toxins may enter the wetland and how they could be influenced by open water, vegetation traits, and seasonality of hydrodynamics.
7) Determine where and how the wetland is positioned in the watershed and whether it may have been impacted by long-term chronic conditions (disturbance) of any type. Look for differences in vegetation. Does the regeneration match what is expected for the site? If not, is the regenerating vegetation more hydric or more mesophytic than is characteristic for the wetland type?

Analyzing these abiotic factors is the fist step in an ecosystem approach to wetland risk assessment. Although abiotic traits alone can provide valuable clues for targeting ecotoxicological investigations or other studies, one must also overlay information on the biology and ecology of the system in order to conclusively identify and evaluate the full range of potential issues or problems for a given assessment.

Knowledge of wetland science is necessary in order to effectively address the biotic components ofwetland ecosystems in the context of risk assessment. A discussion of some of the key principles is given here to point out important factors that must

Table 4-5 Relationships between geomorphic setting and ecosystem attributes of freshwater wetlands

Geomorphic setting	Qualitative evidence	Quantitative evidence	Hydrologic functions	Ecological significance
No apparent inlet or outlet	Topographically isolated from other surface water	Drydowns frequent: water table frequently below the wetland	Retains inflow; losses mostly by infiltration or evapotranspiration (ET)	Inaccessible to aquatic life dependent on streams; endemics likely
Positioned on local topographic high; surface output only	Outlet may be defined by contours or intermittent streams	Drydowns frequent: water table frequently below the wetland	Temporary flood storage: outlet may overflow during high surface water or flow continuously during high ground water: outlet controls maximum depth	Wetland open to immigration and emigration of aquatic life: potential for recolonization if drydowns cause local extinctions
Located in marginally dry climate; variable inlets and outlets	Inlets and outlets may be defined by contours or intermittent streams	Water conductivity high = wetland is recharging underlying aquifer: if low = aquifer is supplying the wetland	Retains inflow; loss primarily by ET or infiltration; may be subject too wide for migrating fluctuations in water depth	Import and export detritus: critical habitat for migrating waterfowl: vulnerable to eutrophication and toxic accumulation due to long retention time
Both surface inlet and outlet: large catchment sustains marginal riverine features	Inlets and outlets may be defined by contours or intermittent streams	Water budget dominated by lateral surface flows or strong groundwater discharge	Temporary flood storage; drainage back to stream or continuously saturated	Import and export detritus: provides fish and wildlife habitat
Located on break in slope	Soil saturated most of the time	Chemically indicative of ground water. discharge from slope base or face	Inflow steady and continous; seasonal loss by ET; low surface storage capacity	Provides stable source of moisture: contributes to biodiversity
Ombrotrophic bog	Peat substrate saturated most of the time: plants indicate ombrotrophic bog: surface flows are negligible	Peat confirmed by organic content and thickness: ombrotrophy evident from low pH and ion content	Some storage of storm runoff; groundwater conservation when water table is below surface	Upland habitat scarce: species composition is unique to bog conditions
Rich fen	Peat' substrate saturated most of the time: graminoid species indicative of groundwater supply	Peat confirmed by organic content and thickness: minerotrophy evident from circumneutral pH and high ion content	Subsurface water supply maintains saturation to surface and hydraulic gradient to maintain flow	Allows lateral movement of water without channelized flow; exhibits moderate level of primary productivity and detritus export
Streamside zones of intermittent streams	Headwater position: first-order stream	Flows not continous; no headwater flooding or overbank flows	Interface of landscape where ground water and surface water change to fluvial environment	Riparian zone critical to maintain buffer between the stream and uplands
High-gradient downcutting portions	Bedrock-controlled channel	Substrate lacks alluvium; flow may be continuous but flashy	Downslope transport is dominant feature	Scour prevents extensive wetland development
High-gradient aggrading portions	Substrate controlled by fluvial processes	Stratigraphy shows imbeddlng, of coarse particles within fines	Wetland on coarse substrate maintained by upslope groundwater source	Unstable substrate in a scour-prone environment colonized by pioneer species Allochthonous organic supply
Middle-gradient landform	Channelized flow: evidence of oxbows and meanders consistent with fluvial processes	Flow likely continous with moderate to high base flows	Channel process establishes variation in topography, hydroperiod, and habitat interspersion on a floodplain	Interspersion of plant communities increases biodiversity
Low-gradient alluvial; floodplain of bottomland hardwood	As above, but in low-gradient landform	Flow continuous with cool season flooding: high suspended sediments in stream	Flood storage: conserves groundwater discharge	Major habitat for wildlife; biogeochemical activity and nutrient
Shoreline of large lakes	Subject to seiches: lake level controls position	Year-to-year trends in zonation follow climatic cycles: wind-generated fluctuations possible	Lake Is water supply for wetland and establishes hydroperiod gradient for wetland zonation	Stabilizes shoreline: transition habitat used by aquatic and terrestrial biota
Coastal sea-level location	Subject to tides; sea-level controlled	Elevation relative to tides and changing sea level	Wetland is responsive to tides and sea level	Barrier to saltwater encroachment; retains sediment: nursery habitat for estuarine organisms

be considered when identifying biological characteristics of a wetland. These characteristics may ultimately affect the direction of the risk assessment as well as the effectiveness of subsequent risk management.

Biological processes and ecosystem functioning

in addition to the complexity introduced by the myriad of interactions of external factors, differential biotic responses to these external factors also yield a complex set of interactions among 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 are adapted to a greater or lesser extent to flooded conditions, but that are, in most respects, structurally and physiologically similar to their terrestrial ancestors. Yet, wetlands may also have features similar to deepwater 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 2 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. We provide here only a brief discussion of certain unique aspects of wetland ecosystems. The reader is encouraged to review relevant published literature for a more complete foundation in wetland ecology. Recommended readings include Ethrington (1983), Mitsch and Gosselink (1993), and NRC (1995).

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 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 2 very divergent wetland types (Figures 4-12 and 4-13).

Wetlands may display dramatic temporal shifts in zonation patterns in response to changing hydrology. Entire systems may even shift, for example, between predominantly emergent and open water zones. In periods of little or no water, some

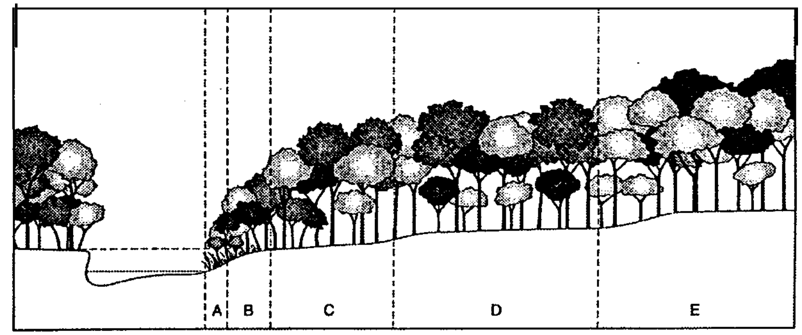


Figure 4-12 Vegetation zones along South Skunk River, IA (and similar-sized rivers in central hardwoods forest region). A-B) Deposition bank with A) herbaceous plants and tree seedlings grading to B) dominance by *Salix interior* and young *Salix nigra* and *Populus deltoides*. C) Floodplain with maturing *Salix nigra. Populus deltoides* and *Acer saccharinum*. D) First terrace dominated by *Celtis occidentalis. Junglans nigra.* and *Fraxinuspennsylvanica.* E) Second terrace dominated by *Quercus macrocarpa* and/or *Acer nigrum* depending on soil type and aspect. In larger river bottoms, area C is much expanded with relatively less of areas D and E.

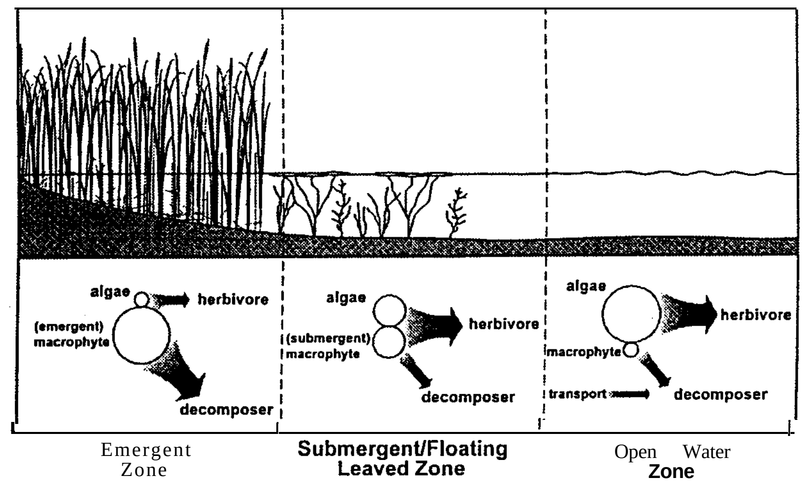


Figure 4-13 Spatial pattern in vegetation and energy flow in prairie pothole wetlands

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, largely 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. These systems are flooded during winter and spring periods of high stream flow and bankfull discharge but are typically dry by mid to late summer due to drainage and ET. Longer-term cycles are a major feature of prairie pothole wetlands, which 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 (Figure 4-14).

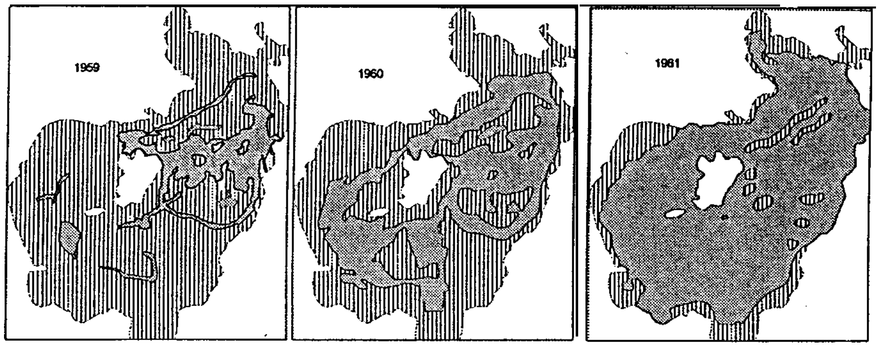


Figure 4-14 Annual changes in open water (shaded) and emergent vegetation (hatched) in a prairie pothole wetland (reprinted with permission from University of Notre Dame, Weller and Spatcher 1965)

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. 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 (Carpenter and Lodge 1986; Rose and Crumpton 1996). 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 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 (see Chapter 3), vegetation structure is one of the most important factors affecting foodweb structure and bioenergetics in wetland ecosystems. Despite the obvious oversimplification, it is useful to distinguish 3 broad classes of primary producers in wetlands with regard to foodweb 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 be extensive at times. 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 from external sources such as chemical fertilizers.

In contrast to emergent macrophytes, submergent and floating leaved macrophytes have substantially less structural material. Their tissues generally have higher nutrient content 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, much of the algae produced in wetlands is consumed by herbivores rather than being transferred directly to the detrital pool, significantly more than in the case of emergent or submergent macrophytes. Detritus derived from algae is very labile and easily digested or decomposed.

Most freshwater wetlands are assumed to be dominated to a lesser or greater extent by a food chain that is weblike and detritus-based (Mitsch and Gosselink 1993). However, based on the preceding discussion, it is clear that spatial heterogeneity in vegetation structure can result in a mixture of detritus-based and producer–herbivore-based food webs (Figure 4-13). 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 understand 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 (Figure 4-14).

Applying the Ecological Factors to a Wetlands-specific Risk Assessment

As part of the data collection for the risk assessment, keep in mind that, as a general rule, ecotoxicological or other types of tests that might be applicable for coastal or marine wetlands may not be suitable for freshwater wetlands and vice versa (Kent et al. 1994). It is incumbent on those using any of the tests or undertaking the laboratory or field studies to fully understand their applicability, limitations, and interpretation.

The ecosystem approach given here was constructed to maximize flexibility in approaching the risk assessment, made necessary by the diversity of freshwater wetlands that may be encountered, in addition to the multitude of factors or stressors that may be at work in the particular wetland under study (Kusler and Kentula 1990; Zentner 1994). Figure 4-15 provides a simple hypothetical illustration of the stressors or factors at work in a wetlands at 2 different times to explain that the magnitude of these stressors is highly dynamic. This figure further emphasizes that all forms of stressors, biological, chemical, and physical, are integrated within the overall risk faced by ecological receptors, such as wetlands, and that the interlinkage of these stressors must be understood and recognized when conducting a risk assessment (Kentula et al. 1993).

An ecosystem approach stresses the key concept of interlinkage of the wetland components (NRC 1992, 1995). An additional overarching provision is that the approach to data collection and evaluation should 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 (USEPA 1994a, 1997).

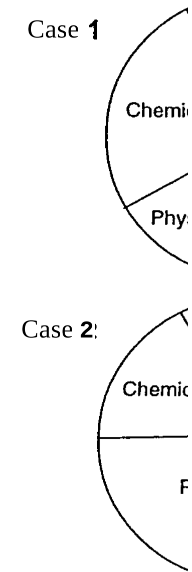


Figure **4-15** Main groups of stressors in th
stressors to the whole stress are dynamic, t

Problem formulation

There are several main points to consic
ment. First is to gather and review prev
photographs, historical maps, land use
studies, etc. Also important is to gain
geology driving the wetlands under stu
prairie pothole, or another type? As no
primarily due to hydrological and geolc
ence the focus of the risk assessment (N
process. Another key aspect is to detern
under study. For some wetlands, this will
may encompass an entire watershed of

An early step in problem formulation, v
evaluation models (e.g., Brinson 1993;
establish the important characteristics
tantly, however, these and other model
and measurement endpoints.

Development of assessment and measurement endpoints

One of the most important steps in the problem formulation phase is establishing clear assessment endpoints because they set the stage for all of the forthcoming effort. Assessment endpoints specific to freshwater wetlands can vary tremendously due to the diversity of potential wetland types that may be encountered and due to the myriad functions the wetland may serve. In a diversion from the general practice in ecological risk assessment, under the proposed USEPA framework (USEPA 1992) the assessment endpoints may or may not be biologically or ecologically based. For example, the hydrology, geomorphology, soils, and other aspects of the wetlands may be far more important a focus than some of the biological resources (Brinson 1993). This is not to suggest that ecologically based endpoints are not important, but that they entail abiotic as well as biotic considerations. In fact, directing the risk assessment at the ecosystem or landscape level requires recognition of the abiotic and biotic components and their linkage.

Some important values and functions of freshwater wetlands, from which assessment endpoints can be derived, are shown in Table 4-6 (see also Brinson 1993; Bartoldus et al. 1994; Richardson 1994). These are not exhaustive but can be used as starting points in the risk assessment. Examples of possible assessment endpoints specific to freshwater wetlands are shown in Table 4-7.

Table 4-6 Important values and functions of freshwater wetlands

Value or function	Mechanism or activity
Hydrological	
Flood protection	Water storage and control
Water quality	Sediment control; nutrient production or export
Ecological	
Habitat	Vegetative growth and maintenance
Human	
Recreation	Fishing, hunting, wildlife watching
Commercial	Fishing, timber harvesting

Table 4-7 Possible assessment endpoints for freshwater wetlands

Assessment endpoint	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 stabilitv	Reduces erosion of stream and river banks
Ecological	
Maintain level of primary productivity	Underpins food web stability

A hypothetical case helps illustrate
the shift in focus of the risk assessm
study is one that is dependent on a c
this example, one assessment endpoi
quality ground water to the wetlands
nonchemical stressors (physical dive
Albeit an oversimplification of an ac
ecologically driven assessment endp
However, as stressed in the ecosyster
and geomorphology are inseparable a
protected. In this situation, protectio
the sustainability of the wetland: loss
ceases to exist.

In one of the most extensive ecologic
environment, the Clark Fork River (C
DalSoglio 1994; Linder et al. 1994; se
endpoints for the riparian wetlands o
water supply per se. This does not im
rectly, but that the primary focus was
plants and animals) versus the one k
themselves: water. Further, this does
biologically driven assessment endpo
wetlands should include other parame
ability of the wetland itself.

Under the hypothetical case describe
could be ensuring that the ground wa
Quality could be defined as a particula
tance, or as an absence of chemical st
dissolved Se concentrations below 5 g
to establish the assessment endpoints
sustaining or improving the health of
inseparability of the ecological, hydr

Likewise, measurement endpoints may
basis. Nevertheless, they must be relev
endpoints. In the hypothetical case, m
determinations of contaminant concen
the specific conductance or suspended
the wetlands, and others. Implicit, too
assessment and measurement endpoint

Numerous endpoints can be used to assess impacts to biological functions. Following is a synopsis of some key biological measurement endpoints for wetland risk assessment (Table 4-S).

Table 4-8 Important hydrogeomorphic, biogeochemical, ecological. and compound-specific parameters for assessing exposure in freshwater wetlands (field and/or laboratory measurements)

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

Methods and endpoints for wetlands

While numerous field and laboratory methods are available for evaluating aquatic habitats and sediments within wetlands, relatively few are available for testing wetland soils. Sources of information regarding aquatic and sediment contamination evaluation are listed below, and only more recently developed soil test methods will be summarized here for use in wetlands risk assessment.

Whether qualitative and reliant on published information or quantitative and implemented as part of a designed study, aquatic field surveys and biological tests for evaluating wetland risks can be achieved by evaluating biological effects associated with chemical, physical, or biological stressors. Frequently, these tools are used in the measurement or monitoring of wetland populations and community structure through structural endpoints such as relative abundance, species richness, community organization (diversity, evenness, similarity, guild structure, and presence or absence of indicator species), and biomass. Functional endpoints, 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. While functional measurements are important in interpreting the significance of an observed change in population or community structure, 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.

Species richness and relative abundance

Species richness (the number of species in a community) and relative abundances (the number of individuals in any given species compared to the total number of individuals in the community) are structural endpoints commonly measured in field surveys of periphyton, plankton, macroinvertebrates, and fish regardless of whether the habitat is a wetland or flowing surfacewater feature. Estimates of relative abundance or species richness can yield readily interpretable information on the degree of contamination of wetland habitat (Pascoe et al. 1994). Loss of a particular species can be critical when that species plays an important role in a community or ecosystem (Karr et al. 1986).

Biomass

Biomass measurements, defined as the mass of tissue present in an individual, population, or community at a given time, are another potential structural endpoint critical to wetland risk assessment. As summarized by LaPoint and Fairchild (1989), biomass can be directly measured gravimetrically on wet or dry tissue. For example, biomass may be estimated gravimetrically by using pooled samples of individuals or by an indirect method, e.g., invertebrate or fish biomass can be indirectly estimated by using empirical or published length:weight regressions. Biomass of periphyton communities is also commonly measured. Measurements of phytoplankton or periphyton biomass can be estimated on the basis of ash-free dry mass (AFDM) or chlorophyll a content (APHA 1992). Chlorophyll measurements are performed by solvent extraction, followed by spectrophotometry or fluorometry (APHA 1992).

Indicator species

The presence or absence of indicator species is commonly used to assess adverse effects to ecological communities (Karr et al. 1986; Hilsenhoff 1988; Plafkin et al. 1988). While originally derived from the saprobian system in which certain species and groups were found to generally characterize stream and river reaches subject to organic wastewaters (Kolkwitz and Marsson 1902; Gaufin 1958; Sheehan 1984), the application of indicator species to wetlands is clearly practiced, e.g., within the delineation process. History has shown that the indicator species concept lacks broad applicability to all types of contaminant stress, however. Furthermore, species selection may occur in aquatic habitats that are chronically polluted with low levels of contaminants over sufficiently long periods. In some wetlands, as well as flowing surface water, the IBI may be pertinent to the risk assessment process.

Indices

Biological indices in wetland risk assessments, as in other ecological risk assessment applications, can be used to mathematically reduce taxonomic information to a single number, or index, to simplify data for interpretation or presentation. Indices can be classified among several types:

1) evenness (measuring how equitably individuals in a community are distributed among the taxa present),
2) diversity (calculating the abundance of individuals in 1 taxon relative to the total abundance of individuals in all other taxa),
3) similarity (comparing likeness of community composition between 2 sites), and
4) biotic indices (examining the environmental tolerances or requirements of individual species or groups).

Although indices may aid in data reduction, they should never be divorced from the actual data on species richness and abundance. Relying on a single index such as the Shannon-Weiner may be misleading for any system at risk, including wetlands. For example, a few individuals evenly distributed among several species could give a relatively high index of diversity, even though a habitat is grossly polluted. In addition, statistical assumptions of independence, normality, and homogeneity of variance are frequently invalid for these derived, proportional measures. Hence, when indices are used, statistical transformations (e.g., arc sine) or rank-order statistics are recommended (Siegel 1956; Green 1979; Hoaglin et al. 1985).

Guild structure

For wetland communities, data generated at the species level can be analyzed according to guild structure. Guilds, or functional feeding groups, are classifications based on the manner in which organisms obtain their food and energy. Invertebrates can be classified among such functional groups as collector-gatherers, piercers, predators, scrapers, and shredders (Merritt and Cummins 1984; Cummins and Wilzbach 1985); and fish can be classified as omnivores, insectivores, and piscivores (Fausch et al. 1984; Karr et al. 1986). Avian communities in wetlands are increasingly being analyzed within the context of guild structure (Adamus 1993a, 1993b). Shifts in community guild structure may reflect changes in the trophic-dynamic status of a wetland. For example, contaminant impacts on a wetland may eliminate or reduce periphyton and thus concomitantly reduce the relative abundance of scrapers (herbivores) in relation to other invertebrate guilds such as collector-gatherers. Effects must be fairly strong to assess changes in guild structure. For contaminant studies in wetlands, community and guild analysis should also be supported by physical habitat and chemical information, since these may alter production and dynamics of biological populations and, consequently, confound the interpretation of wetland community data. Needless to say, the selection of appropriate reference locations is critical to wetland assessments that incorporate community and guild analysis.

Plankton

Many devices are available for sampl analysis. Sampling techniques for ph various surfacewater habitats. The ch sample size, and sample numbers, wh depend upon the characteristics of th organisms, and spatial variation).

Macroinvertebrates

Benthic invertebrates are the most co contaminants, whether sediments are Numerous excellent references deal w of benthic invertebrate populations (e measurement endpoints include relativ Trophic guild structure can be determ (Merritt and Cummins 1984; Cummins evenness, and community similarity ca nant effects study, careful consideratio samples among stations.

Fish

In biological monitoring and evaluatio may be recommended for use because

1) regulators and the public can ea of pollution on fish;
2) fisheries have economic, recreat
3) the identification of fishes is rel macroinvertebrates);
4) the environmental requirements
5) fish are perceived as "integrators et al. 1980).

However, the size, distribution, and re difficult to quantify because variations large (Lagler 1978). Additional difficul are caused by the selectivity and efficie al. 1980). However, consideration of t of different wetland habitats that supp wetland risk assessment process.

The types of analyses performed on dat abundance, species richness, and size s assessment is the IBI (Karr 1981; Karr et of individual species tolerances for wat was developed to determine the effects of decreased habitat quality on fish communities of midwestern streams, but for some wetlands it may be quite applicable to the risk-assessment process. The index is composed of 12 individual metrics divided into the fields of species composition and richness, trophic composition, abundance, and condition. Scores of each metric are classified as "best," "average," or "worst" (each class having a numerical weighting) in relation to reference data (Fausch et al. 1984).

Sediment and soil methods and endpoints for wetlands risk assessment

While not as readily available as aquatic or sediment toxicity test methods (e.g., Peltier and Weber 1985; Weber et al. 1988), methods have been identified for testing soil biota (e.g., USEPA 1989). For wetlands, the application of biological tests should provide a comparative toxicity database upon which wetland-specific soil evaluations can be completed. Screening (unamended wetland soils yielding percent effect) and definitive tests (amended soils potentially yielding median effective concentrations) may be completed with standardized test species to evaluate toxicity within a biological assessment. Additionally, to assure adequate information for ecological evaluations of soil contamination, species having site-specific relevance may also be tested (Parkhurst et al. 1989). When performed in parallel with standard test methods, these site-specific tests (e.g., using resident plant species) may be diagnostic and indicate biological responses (e.g., development of metal resistance) that are associated with soil exposures. Presently, the application of laboratory bioassays to wetland risk assessment is increasing, particularly in developing biological databases that contribute to the ecological risk assessment process. To enhance the ecological relevance of site-specific biological tests and to reduce the potential extrapolation error associated with interspecific comparisons, use of standard and site-specific test species in ecological assessment should be considered in soil testing (see Linder et al. 1993).

Plant test methods

Plants associated with wetlands have been used extensively to assess water and sediment quality. The wide variety of tests developed has targeted the effects of both water column and sediment-borne toxic materials. The types of aquatic vegetation used for these purposes range from microscopic unicellular algae to relatively large flowering plants. The 3 most commonly applied test methods include chlorophyll *a* concentration, growth, and contaminant uptake.

Growth measurements (biomass accumulation per unit of time) have been widely applied as an assessment method for a variety of freshwater estuarine and marine species. Much of the testing has been conducted on sediments in the laboratory, using unicellular phytoplankton such as *Selenastrum capricornutum* (freshwater) and *Skeletonemu costatum* (marine) (e.g., Thomas et al. 1990; Ankley et al. 1993). Until recently, use of rooted wetland macrophyte growth has been limited. Growth is perhaps the least specific measurement endpoint. A response such as reduced growth rate is not tied to specific sites within the plant where reactions or processes are altered by specific chemicals. This is especially true for rooted macrophytes. The advantage of measuring growth is that it is an integrator of all effects of toxicants on plants, it is relatively easy to measure, there is a wide range of past use, and it can be done with acceptable precision in both the field and laboratory.

The physiology of chlorophyll production and maintenance is quite well known. Chlorophyll occurs in virtually all plants and is the primary pigment involved in the important ecological process of photosynthesis. The correlation between chlorophyll concentration and photosynthetic rate commonly is strong. Chlorophyll concentration relative to contamination of water or soils has been measured in unicellular algae, macrophytes, and periphyton communities (e.g., Bassi et al. 1990). Chlorophyll concentration generally reflects the mass of plant material present, as well as being an indication of the health of the material. Toxicants can affect the chlorophyll molecule directly or through the process of energy transfer during photosynthesis. A method recently applied for determining the effects of toxicants on chlorophyll (and photosynthesis) involves the measurement of delayed fluorescence. The technique appears to be highly sensitive and relatively easy to conduct.

Contaminant uptake by plants has been applied primarily to rooted macrophytes. It is assumed that most of the uptake occurs through the roots and that the concentration of the contaminant compounds in leaf tissues is directly related to the concentration in the soil or sediment. Uptake has received wide application in fresh and marine systems and has been carried out under both laboratory and field conditions (e.g., Kovacs 1978; Lee et al. 1981). Uptake of contaminants relies on several assumptions that must be taken into account for interpretation of results. Chemicals may be modified to form nontoxic compounds by the plant. Certain chemicals are not concentrated, while others are, which may bias the interpretation of what chemicals are present in the test medium. However, these uptake measurements are more relevant for evaluating risks to herbivores (and bioavailability of chemicals in sediment) than for deciding what is there per se. Finally, uptake rates may be inhibited by the toxicity of other materials in the medium, and the test organism may be inhibited in its ability to accumulate the contaminants.

While measurements of plant growth, chlorophyll content, and contaminant uptake are the most commonly used methods, several other are in various stages of development and implementation. These methods include measurements of photosynthetic rate, chloroplast morphology, peroxidase activity, root growth, seed germination, seedling growth, and reproduction.

The strongest approach to the assessment of wetland subsystems may be to use a combination of several methods to evaluate contamination of water and sediments. This combination would indicate both ecological and physiological responses of the plants to the media and would increase the power of the analysis through verification of responses using several endpoints.

Seedgermination and root elongation

Techniques modified from methods originally developed in the plant and weed science disciplines have yielded short-term tests that assess toxic chemical effects on plants. The seed germination and root elongation bioassays are laboratory toxicity tests that directly and indirectly assess toxicity of soils and evaluate toxicity end-points (seed germination and root elongation) pertinent to ecological assessments for terrestrial and wetland habitats. Seed germination tests measure toxicity associated with soils directly, while root elongation tests consider the indirect effects of water-soluble constituents which may be present in site samples. These methods have been used extensively in soil contamination evaluation, including a comprehensive wetlands risk assessment (Linder et al. 1994; Pascoe and DalSoglio 1994; Pascoe et al. 1994).

Rooted aquatic plants

Wetland soils frequently complicate standard methods for phytotoxicity assessment, owing to the saturated character of their soils. Wetland soils may resemble sediments in many respects, particularly when seasonal or ephemeral climatic conditions alter soil water-holding capacity, which may confound interpretations of germination and growth responses in standard plant testing species (e.g., buttercrunch lettuce, *Lactuca sativa*). Standardized rooted aquatic plant toxicity tests, however, have been developed and should be considered on a site-specific basis for hydric soils and freshwater or estuarine sediment evaluations. The most well-, developed method uses *Hydrilla verticillata*, but additional test methods using sago pondweed (*Potamogeton pectinatus*) may also be valuable in evaluating wetland soils or sediments (Byl and Klaine 1991; Fleming et al. 1992).

Laboratory evaluations with wetland and uplandplants

Freshwater marsh plants may be used to evaluate sediments or hydric wetland soils as outlined by Walsh et al. (1991). The method was originally designed to test single toxicants or defined chemical mixtures in defined media, but it can be modified to test field-collected sediments or wetland soils that may be appropriate to wetland risk assessment. In general, the method utilizes rooted marsh plants and evaluates the effects of contaminated soils and sediments on early seedling growth and survival. For example, *Echinochloa crusgalli* is one species of marsh plant specifically identified in the test procedure, but alternative marsh plants (e.g., *Spartina alterniflora*) may be identified on a site-specific basis and tested, provided the selected plants are amenable to the test format outlined.

Primarily in response to the assessment needs associated with land disposal of dredging materials, the U.S. Army Corps of Engineers Waterways Experiment Station (WES) has developed a test method for evaluating phytotoxicity and bioaccumulation potential in a freshwater plant, the yellow nutsedge (*Cyperus esculentus*). The method is applicable to wetland risk assessments and can be used in either flooded wetland or upland habitats. From an ecological perspective, the test

evaluates toxicity endpoints (e.g., growth) that may directly relate to field observations regarding plant cover or vegetative vigor (WES 1989; Folsom and Price 1992). It is also useful for evaluating bioaccumulation of contaminants in the diet of herbivores.

Alternative test species in seedgermination, root elongation, and early seedling survival and vegetative vigor tests

In these tests, measurement endpoints are frequently similar (e.g., growth, germination), but the species being tested differ. In part, these differences reflect soil matrix characteristics that might limit the success of any given test system, especially in wetland soils. For example, lettuce seed is frequently used in seed germination tests, but some soils may not be amenable to testing with a domesticated species selected for optimal growth in a particular soil matrix. Contaminant effects and matrix effects may potentially be confounded when the life history characteristics of a test species preclude or potentially limit its usefulness in any given phytotoxicity test method. Additionally, for interpretation of wetland-specific ecological effects, the support of a comparative toxicity database may be insufficient within a risk assessment context. Thus, more relevant test species may be beneficial to evaluate ecological effects with a wetlands risk assessment, and measurement endpoints (e.g., survival and growth) used to evaluate relationships between ecological indicators and soil toxicity may be considered using methods modified for tests with alternative species. For example, methods to evaluate seed germination using various species of plant seeds (agricultural crops, vegetables and herbs, flowers, and trees and shrubs) are briefly summarized by the Association of Official Seed Analysts (AOSA) in their Rules for Testing Seeds (1990). Here, exposure conditions specific to various species are tabulated, including suggested substrates and optimum incubation temperatures for germination testing as well as test duration specifications. Furthermore, special pretreatment of native seeds, e.g., prechilling or scarification, is also specified, and methods for distinguishing between nongerminated seeds and nonviable seeds are identified (e.g., tetrazolium and embryo excision tests). On a wetland-specific basis, these alternative test species may be more conducive to ecological interpretation, especially when soil matrix effects unique to wetlands can potentially confound contaminant effects on seed germination and emergence.

Soil biota biomass and diversity

Without question, wetlands are complex biological systems, and wetland soils are critical components in the characterization process. A thorough consideration of the methods applicable to wetland soils characterization with a risk assessment setting is beyond our present scope. However, wetlands functions and processes are clearly dependent upon a healthy soil. For example, nutrient cycling would not occur without organisms to perform the majority of the critical processes. Soil organisms perform many wetland processes, and in unimpacted soil, there usually (but not always) are several organism groups that perform any particular process. For example, the dependency of vegetation on the presence of mycorrhizal fungi and on a functional soil-organism nutrient cycling system may be quantified within a wetlands risk assessment, and evidence is accumulating that at least some plants are dependent on symbiotic organisms for establishment or survival (Reeves 1985; Janos 1987). Clearly, other measurement endpoints could be identified (Linder et al. 1992), and while not exhaustive, methods are available to evaluate these within the context of wetland risk assessment:

1) bacterial biomass and community structure,
2) fungal biomass and community structure,
3) protozoan diversity, and
4) nematode diversity and community structure.

Solid-phase and aqueous-phase Microtox

While aqueous-phase testing with Microtox has been readily available for 10 to 15 years, solid-phase testing has only recently been commercially available (Microbics 1992). As previously summarized (Warren-Hicks et al. 1989), Microtox relies upon measurements of bioluminescence for an evaluation of a sample's toxicity. The test, whether aqueous- or solid-phase, utilizes freeze-dried cultures of the marine , bacterium *Photobacterium phosphoreum* and is based on the inhibition of bioluminescence by toxicants (Bulich 1979, 1982, 1986). The results of several studies of pure compounds and complex chemical mixtures suggest that aqueous-phase testing with Microtox generally agrees with standard fish and invertebrate toxicity tests (Curtis et al. 1982). Solid-phase testing with Microtox, however, does not have a comparable database established for developing statements regarding its correspondence with standard soil tests using, for example, earthworms.

Earthworms tests

While not applicable to all wetland soils, earthworms have become a primary test organism for soil contamination evaluations. From an ecological perspective, earthworms are significant in improving soil aeration, drainage, and fertility (Edwards and Lofty 1972), although the comparative database does not unequivocally suggest that earthworm toxicity measurements are reflective of soil health. To enhance the ecological relevance of site-specific biological tests and to reduce the potential extrapolation error associated with interspecies comparisons, testing with site-specific species should be considered in soil evaluations. The earthworm bioassay most frequently used is a modification of a method described by Goats and Edwards (1982) and Edwards (1984) and uses lumbricoid earthworms as the test species. *Eisenia foetida* may be used in these tests because it is easily cultured in the laboratory and reaches maturity in *7* to *8* weeks at *25 "C. E. foetida* is responsive to a wide range of toxicants, and the comparative database suggests that similar toxicity responses can be anticipated regardless of the subspecies being tested (Neuhauser et al. 1986).

Nematodes tests

Soil-inhabiting nematodes represent on
brates that should be studied during s
ecological effects assessment for a wet
their role in soil decomposition proces
nutrient dynamics (e.g., dispersion an
of bacterial activity, and promotion o
directly as well as indirectly reflect the
redivivus has a relatively well-develope
(Samoiloff et al. 1980) and has been us
complex chemical mixtures (Samoiloff
sediment evaluations. Most frequently,
with other biological assessments (*e.g.*,
testing) for evaluations of water quality
sediment toxicity testing (Samoiloff et al
described in the comparative toxicity l
oped nematode test using *Caenorhabditis*
Williams and Dusenbery 1990) may be
f? redivivus and *C. elegans* tests meas
effects related to growth, reproduction,
term tests and generally require less th
term tests that measure reproductive ef
7-d exposures.

Unlike *P. redivivus, C. elegans* is a nat
by van Kessel et al. 1989), and tests
contaminant effects in terrestrial habit
the toxic effects of metals in aqueous
their comparative analysis, *C. elegans* a
metal exposures complemented and we
Daphnia magna and sediment macroinv
(e.g., Popham and Webster 1979; Haigl
et al. 1989), for some toxicants like h
nematodes was developed, and extendin
ered within ecological effects assessmen
P. redivius or C. *elegans* was originally d
pore waters, nematode tests are directly
interstitial waters.

Arthropods (insects) tests

Various methods have been developed f
insects, especially pesticide effects on n
these methods are directly applicable to
indicators of soil contamination, terrest

are potentially critical targets within an ecological effects assessment. Within ecological contexts, terrestrial invertebrates play a role in communities and ecosystems that involves integrated functions such as decomposition, grazing, predation, and pollination (Croft 1990). While methods that evaluate adverse biological effects in terrestrial invertebrates exposed to soil contaminants are not widely considered in the ecological effects assessment process at present, their contributions have increased and should continue to increase in the near future, especially for wetlands risk assessment. Through strategies similar to those used with aquatic invertebrates (e.g., Plafkin et al. 1989; Klemm et al. 1990), terrestrial insects would be amenable to soil contaminant evaluations for wetlands, particularly given field survey information regarding insect community structure and population numbers in wetlands at risk. For example, to evaluate soil microarthropods quantitatively and qualitatively, techniques are readily available to extract, enumerate, and identify these organisms in reference and impacted soil samples. Soil microarthropods are easily extracted from the soil using Tulgren high-efficiency extractors (e.g., Seastedt and Crossley 1980; Anderson 1988). The extracted organisms can then be counted using dissecting microscopes and identified to genus, or form-group. Recent innovations in computer-assisted identification (HyperCard) have also reduced the time required to identify these organisms (Moldenke et al. 1991).

Terrestrial arthropod (non-insect) and isopod tests

Outside of North America, terrestrial arthropods other than insects have been considered from the perspective of accidental or coincidental exposure to potentially harmful chemicals (Croft 1990). While not exclusively focused on wetlands, these methods are directly applicable to the risk assessment process for wetlands. For example, to evaluate effects of agrichemical pesticides or biological control agents on nontarget invertebrates, laboratory methods have been standardized for evaluating chemical effects on mites (e.g., Sewell and Lighthart 1988). While terrestrial arthropod tests methods are few and present a limited history in ecological effects assessments for wetlands, their role in the environment (Croft 1990) requires that these organisms should receive consideration as ecological receptors during the risk assessment process. The methods developed for pesticide evaluations could be directly applied to wetland soils contamination evaluation. Alternatively, soil-derived eluates could be used in the testing process, if the study design indicated that indirect routes of exposure were likely to occur, e.g., nonpoint source runoff into wetlands from agricultural lands. While a variety of test species have been used in the standard tests developed in Europe and the United States (Hassan 1985; Hassan et al. 1987; Croft 1990), the laboratory test methods using non-insect arthropods are relatively straightforward and easily could be modified to directly meet the requirements of a soil contaminant evaluation for wetlands.

Similarly, biological assessments using terrestrial isopods have historically been considered in soil contamination evaluations, although standardization, e.g., through the American Society for Testing and Materials (ASTM) or the Organiza-

tion for Economic Cooperation and Dev
indicators of contaminant exposure, the
and organ-specific contaminant bioacc
animals, particularly for some environm
1984; Beyer and Anderson 1985; Hopk

Mollusk tests

Wetlands are habitats that are frequentl
sites, and mollusks are often regarded
of these habitats (Pennak 1978). Coinci
some families of freshwater mussels (Un
species for ecological risk assessments f
chemicals) (USDOI 1989). Accordingly,
effects assessments at Superfund sites h
effects and acute toxicity for sensitive l
1990). In contrast to concerns regarding
freshwater mollusks, efforts to develop e
methods (e.g., Getzin and Cole 1964; Cr
ecological assessment needs for wetland
mollusks have been used in toxicity and
Office of Pesticides Program (USEPA 19
applicable to contaminant-related questi
gous tests with freshwater mollusks have
Unionidae mollusks are characteristic fr
could be considered within a toxicity ass
mussel test, *Anodonta imbecilis* was init
mollusk; however, the techniques descri
for testing mussels with similar reproduc
involve the early developmental stages o
mussels, depending upon endpoints bein
test with freshwater mussels followed AS
used at this time, toxicity assessments wi
within an ecological effects assessment f

In contrast to the freshwater mussel test t
ecological risk assessment questions relat
evaluate terrestrial snails and slugs were
molluscides (e.g., Getzin and Cole 1964;
are readily adapted for wetland risk asse

Amphibian test methods

Wetlands are habitats that are frequently
evaluating and monitoring these transitic
areas will require a variety of field and la

and Brandt 1990). Amphibians-frogs and salamanders-may be representative of the fauna potentially critical to ecological effects assessments for wetlands. Amphibian test systems are standardized through ASTM (T29 1997b, E1439 1997c). Early embryos of the African clawed-frog (*Xenopus laevis*) are used in the standardized test; however, much work has been completed with alternative test species and should be considered on a site-specific basis (e.g., Linder et al. 1990; ASTM E1439 1997c; Linder, Wyant et al. 1991).

Interplay of risk management and risk assessment

Important to all risk assessments, whether for wetlands or terrestrial environments, are the early discussions held between the risk assessor and the risk manager. These should define the scope, timing, level of effort, and constraints involved with the risk assessment. There will need to be resolution of issues specific to freshwater wetlands, and the particular type of wetland, between the risk manager and risk assessor before any work is begun.

This discussion may have several important outcomes. First is agreement on the spatial extent or magnitude of the wetland. Small, easily managed wetlands may require a reduced or screening-level assessment to satisfy the requirements of the risk manager. On the other hand, wetlands that are tens or hundreds of acres, that reside in the midst of major industrial activities, or that 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 this latter situation, landscape and ecosystem issues arise and can readily complicate the effort. For example, some wetlands may be dependent on source water outside of the study area, or for that matter, in another state, region, or watershed. Like a number of stressed wetlands in North America, the wetland may be vitally important in controlling floods in a particular area but may not represent a highly valuable habitat (e.g., a *Phragmites* sp.-dominated wetlands) (Bartoldus et al. 1994).

It is also important for the risk manager and the risk assessor to decide on the important stressors and receptors that will be the focus of the assessment. As data are collected and evaluated, additional stressors and receptors may become evident and may justify a realignment of the focus. A confounding issue that often arises at this time is whether the risk assessment will take a multi-stressor or single-stressor approach. It is rare that only a single stressor will be present, yet to approach the risk assessment using multiple stressors requires advancement beyond current science. Today there is inadequate understanding of how to deal with multiple stressors only qualitatively because there is no recognized, validated method for integrating impacts from multiple stressors. Thus, without a clear understanding of what is driving the risk management decision and of the regulatory and jurisdictional issues, the risk assessor may be left with insufficient or at least unclear guidance.

Exposure assessment

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, although poorly understood, may be an important source of exposure in some wetlands. Biological inputs include photosynthetic uptake of C, N fixation, and biotic transport of materials by animals. Except for gaseous exchanges such as C and N fixation or aerial deposition, however, inputs to wetlands are generally dominated by hydrology. Hydrologic transport to freshwater wetlands may occur through precipitation, surfacewater flow, 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, as it will to a large extent determine the soil-sediment chemistry by producing anaerobic conditions, importing and removing OM, and replenishing nutrients. Exposure can occur in transition zones between the wetland and surrounding upland areas. It is important to consider this area as well when examining potential exposure scenarios.

Ideally, exposure in the wetland ecosystem is assessed based on representative monitoring data. In the absence of measured data, exposure can be predicted in the context of a wetland-specific hydrogeomorphic, biogeochemical, and ecological setting. In the case of a chemical exposure assessment, information on the inherent properties of substances 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 and its changes with time (e.g., in concentration or 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 measurements or parameters that can be important when evaluating or predicting exposure of chemical and/or nonchemical stressors in freshwater wetlands are listed in Table 4-8.

Compound-specific information and biogeochemical processes affecting exposure in the different compartments are 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.

Biological assessment

Defined earlier, biological assessments are primarily ecotoxicological tests performed in either a field or laboratory setting. While there are many issues related to the conduct and application of ecotoxicological tests (Levin et al. 1989), they represent one of the main sources of effects information available to the risk assessor. It is beyond the scope of this section to detail the methods or protocols for these tests. However, the publications cited in Table 4-4 include standard testing protocols as well as those developed through the auspices of the OECD.

Once the key stressors and receptors have been identified, the biological assessment should consider toxicity to wetland organisms or 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 need to extend to the transition zones surrounding the wetlands because some stressors will impact adjacent terrestrial environments. These areas should be evaluated only if there are clear, potential pathways for exposure of receptors. Because the primary focus of the biological assessment should be at higher levels of organization, the risk assessor should be cognizant of which tests or series of tests are designed to measure population-, community-, or ecosystem-level effects. Furthermore, the endpoints of the test, whether lethality, reproductive impairment, growth, etc., should be understood and their linkage to the assessment endpoints clearly defined before any work is begun.

Depending on their scope, biological assessments in the aquatic environment could include representative, and ideally sensitive, species of

1) primary producers,
2) primary consumers,
3) microbial community,
4) saprophages or detrivores, 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 amphibia. Inhibition of microbial activity, important in wetland's nutrient recycling and transport, 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 saphrophages/detrivores community. Finally, standard acute and chronic tests are available to assess effects on fish.

Biological assessments of the benthic communities should take into account pathways of exposure. In addition, observed effects will be strongly influenced by sediment-soil biogeochemical conditions such as organic carbon content, particle size distribution, sulfide content, redox potential (RP), and time period allowed for equilibration to occur between dissolved and sorbed fractions of chemical stressors (USEPA 1990). Available test methods concern detrivores or mixed detrivores/herbivores/carnivores and include insect, annelida, and crustacea species with both acute and chronic endpoints (USEPA 1990).

Recently, the OECD reviewed aquat
chemicals (OECD 1995). The review
ods. An overview of the recommen
wetlands-is shown in a foodweb fr

It was recommended by OECD that t
framework for taxonomic groups rat
This should make it possible to test
ments and facilitate extrapolation of
Furthermore, the guidelines and test
chronic toxicity endpoints, dependi

Most of the impacts on freshwater w
i.e., the sediment and overlying wate
rounding or transitioning to the fres
on the type of stressor and the expo
structure and function (e.g., insects,
birds, and transition-zone plants, tre
potential effects are evaluated. Stand
many insects, some amphibians, and
have been adapted for the species m
Acute and chronic bioassays with ro
years to determine the toxicity of che
risk to humans. Similarly, standard a
and upland birds have been widely u

There are, however, few tests that ha
the tests currently used in regulatory
useful. For example, tests for root e
tion, and other methods are known ar
in the transition zone. Other soil test
this context. Keep in mind that the p
itself, and it is there that the effort

Unfortunately, few tests lend themse
effects on trees and shrubs that may
it may be more plausible to determin
located adjacent to the wetlands of c
scientists (e.g., measuring growth rat
utilized for this.

Using standardized toxicity tests bri
of which are mentioned in Chapter 9
is driven primarily by the fact that m
not the same species generally found
of extrapolating from one species to

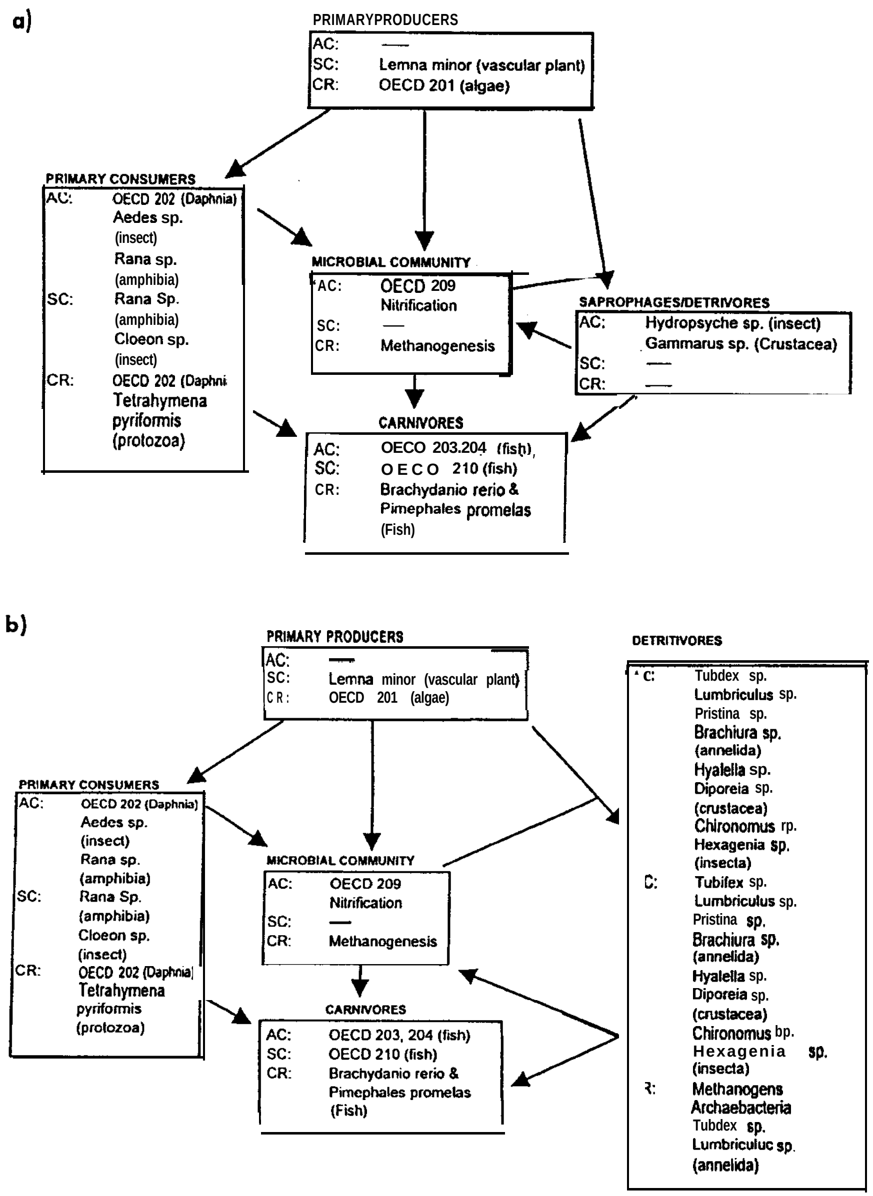


Figure 4-16 Taxonomic grouping of test organisms recommended for freshwater wetland risk assessment by the OECD (1995): a) Primary producer-herbivore-carnivore food web, b) **Detritus-**based food web. AC= acute tests, SC= subchronic tests, CR= chronic tests

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 acute exposure test data are extrapolated to chronic exposure situations, high concentration-response studies to low-concentration exposures, laboratory to field results, and others. All of the results from the biological assessment should be taken in context with other data that will be developed as part of the risk assessment.

Selection of biological tests for wetland ecotoxicity evaluation should be driven by the exposure assessments affected by the hydrogeomorphic and biogeochemical characteristics of the wetland of interest.

Ecological assessment

Ecological assessment primarily determines the impacts of stressors at the population, community, or ecosystem level. In general, standardized ecotoxicology tests do not lend themselves to this type of assessment, and few provide useful ecosystem-level information (Kelly and Harwell1989; Cairns and Niederlehner 1992). In addition, there are significant temporal and spatial issues that come into play. Measuring a significant change in an ecosystem or at the landscape level may require years or decades of study, yet the risk assessor and risk manager are faced with a much more compressed time line. Just as important, it is difficult to isolate easily studied areas of the wetlands 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 are field studies that measure structural components of the ecosystem, including the size and make-up of the habitat, the biomass or standing crop of important plants and animals, and the abundance and diversity of plants and animals. There are, however, functional measurements (Bartoldus et al. 1994; Richardson 1994) that might be useful in understanding the ecological integrity of the wetland. For example, wetlands are extremely important to biogeochemical processing and nutrient cycling (e.g., N and P) (NRC 1995; Chapter 3, this volume) as well as in primary productivity and C, N, P export (Chapter 2, this volume). These functional aspects of wetlands, often considered to be indicative of ecosystem-level processes, depend heavily on microbial communities, water flow, benthic macroorganisms, and other parameters (Brinson 1993). As a result, these functions may be important areas for the risk assessor to consider when designing and conducting the ecological assessment, especially when the assessment focuses on effects at the ecosystem level. Similarly, population- or community-based measures may be useful, provided they have a direct relationship to the assessment endpoints and have been validated scientifically.

Net primary productivity and carbon or energy flow also offer wetland processes that may be measured to assess ecosystem-level effects, provided the measures are integrated across the entire wetland. In this situation, the measure is made of a wetland's net product, resulting from an integrated, interconnected process.

Often, results of the biological and ecological assessments can become inputs to various trophic-level or foodweb models. Such models can give the risk assessor a useful tool to develop a refined conceptual model of how stressors could impact the various processes in the wetlands. The problem with some of the trophic-level or foodweb models is that they require a substantial amount of data, preferably site-specific in nature, lest the uncertainty remain high. Given that fact, the risk assessor and risk manager should decide early on whether the size of the wetland or the complexity of the problem warrants such data-intensive assessments.

Evaluation of Case Studies using the Ecosystem Framework

In retrospect: Would ecosystem-based wetland planning have altered the outcome of the Kesterson episode?

Kesterson Reservoir (see Chapter 6) provides a case history that can be used to assess how well the ecosystem approach performs in evaluating risks associated with proposed wetlands. Limited availability of water was the key issue driving the development of Kesterson's wetlands. Since the 1890s, diversion of water for agricultural use had taken a tremendous toll on the quantity of wetlands remaining in the San Joaquin Valley of California. By the 1970s, when Kesterson was developed, the view generally held by wetland managers in the valley was that any water was better than no water. Viewed in hindsight, the rationale for this thinking is clearly flawed because of water quality issues such as selenium contamination, but at the time, there was no equivalent wetland from which to draw information. However, had an environmental planner been present using the ecosystem approach, would the resultant risk assessment have effectively identified and predicted the problems that eventually occurred?

In order to answer this question, we must look at the basic components of the ecological framework (Figure 4-2). A key factor indicated in the assessment process for Kesterson would have been to thoroughly characterize the water sources and hydrologic regime, i.e., quantity and quality of irrigation drainage, in the context of the arid climate present at the site. Had this step been performed adequately, several key pieces of information should have emerged to guide the decision process. First, it should have been apparent that the evaporative nature of the climate would maximize the likelihood that salts and chemical contaminants in the water source could become concentrated in the wetlands. Second, knowing that the intended water source was subsurface irrigation drainage and not fresh water, adequate chemical characterization would have been indicated. A water quality analysis would have revealed the presence of elevated concentrations of Se, B, and, in some instances, As or other elements. Even though much of the toxicity database that now

exists for these trace elements would not have been available then, it should still have been clear that the water source contained an atypical concentration of salts and trace elements. This, in turn, would have signaled a risk factor that required further investigation. The ecological framework would have indicated to the planner that thorough biological effects testing was necessary to determine whether the water source was acceptable for developing the wetland to meet its primary goal, i.e., as habitat for migratory waterfowl and shorebirds. Carrying out these effects studies would have quickly revealed the toxic hazards from trace elements and indicated that irrigation drain water should not be used to develop Kesterson.

The critical failure in the Kesterson episode was lack of recognition that water quality is a primary consideration in wetland development. Kesterson also illustrates the difficulty of using 1 wetland to achieve 2 objectives. In the case of Kesterson, these were wildlife habitat and disposal of irrigation drainage. Clearly, these were not compatible objectives from the standpoint of water quality. The ecological framework to risk assessment could have identified this problem early in the planning stage and recommended steps to avoid the wildlife toxicity problems that eventually developed.

Current evaluation: Application of the ecosystem framework to risk assessment at Milltown Reservoir Wetlands

The work at Milltown Reservoir Wetlands (MRW) (see Chapter 5) illustrates the strengths and limitations of an integrated ecosystem-based approach to ecological risk assessment. This work at MRW also illustrates how the approach, when applied within a risk assessment context, provides resource managers with tools that would enhance their decision-making process and minimize or at least clearly identify sources of uncertainty. At MRW, the ecosystem approach outlined in this chapter clearly provided a framework for minimizing the heavy-metal-related problems that have developed and are being evaluated throughout the MRW-CFR watershed today. For example, at MRW, land-use and water-use planning was poorly implemented in the up-front siting of the construction project for the hydroelectric facility located at the confluence of the Blackfoot and Clark Fork Rivers of western Montana. This historic, and in many instances current, practice of pursuing widespread land-use and water-use practices with only limited forethought for the interconnectedness within ecological systems is a serious flaw that quickly becomes apparent when the ecosystem-based approach is applied. Whether these resource-use practices are mining, agriculture, forestry, or recreation oriented, various environmental problems have arisen throughout the western U.S. in the absence of an ecosystem-based approach to risk assessment.

Using MRW as our example, the initial decision to site a hydroelectric facility at the Hellsgate of the Clark Fork just east of Missoula, MT might have been reconsidered, especially if the watershed had been more fully characterized and appreciated. For example, the relationships between the upstream source areas near Anaconda and

Butte were clearly not understood at the turn of the century when the hydroelectric facility was constructed at Milltown. If an analysis of the hydrology (surface and subsurface) as well as the geomorphology had been completed as part of the current problem formulation phase of the risk assessment process, the facility might have been constructed at an alternative location, or other measures to reduce sedimentation behind the dam would have been considered.

The current problems from metals and arsenic associated with the soils and sediments are a direct consequence of an incomplete analysis of the surface and subsurface hydrology within the CFR watershed. While this criticism is retrospective, the history of the MRW nonetheless reinforces the value that the ecological risk assessment framework offers to resource managers today. Again, using MRW as it looks today, the available risk analysis for the wetland clearly indicates that the present and near-term risks are low relative to metal- and As-related questions in the wetland, and the focus of attention upstream from the reservoir is well deserved from a management perspective. Here again the ecosystem-based approach has served decision-makers well, and while more subtle issues remain regarding incompletely answered questions (e.g., regarding rhizosphere exposures in the wetland), within a risk assessment context, sufficient information was available to address the current and near-term issues related to the wetland. More importantly, the uncertainty associated with these decisions was more clearly understood and characterized in the ecological risk assessment for the wetlands at Milltown Reservoir, primarily because of the risk analysis activities indicated by the framework. Even in the comprehensive ecological risk assessment for MRW that is currently available, incomplete knowledge is apparent. However, when pursued within an ecosystem context, the uncertainties associated with those data gaps were manageable within the near-term and long-term plans for the wetland and the CFR watershed.

As the work at MRW illustrates, environmental contaminant problems in wetlands often are not a simple problem of chemicals alone, but instead are a complex set of interconnected issues that involve a large noncontaminant component. More often than not, habitat alteration has provided an equal, if not greater, contribution to a multiple stressor setting for resources at risk like those at MRW. Within the ecosystem-based approach, the ability to distinguish between and among various stressors will be required more frequently in resource management decisions that are focused on low-concentration exposures to environmental contaminants and the potential subacute effects that may result. While our present state-of-the-science achieves varying degrees of completeness for any particular risk assessment, the ecosystem approach clearly supports a decision-making process that will minimize uncertainty and potentially yield resource management decisions that are dynamic and achievable in the near and distant future.

In the future: Will the Evergl
A challenge of hydrological, c

Restoration of the Everglades involv
challenges. The policy and partnersh
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Successful Everglades restoration will
the concepts discussed in this chapte
makers. If this integrative approach i
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From the time the earliest explorers
drain the region so that productive u
began in earnest during the 1880s wi
projects to connect Lake Okeechobee
the state had completed the main no
agriculture became an important part
USACOE completed the major comp
Project, which linked all the drainag
a comprehensive water management
successful in meeting its major objec
and protection of urban well fields

However, the project, and the 5 millio
producing unexpected side effects. T
diminished to less than 10% its level
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populations living there. Nutrient ru
Lake Okeechobee is transforming the
Citizens and government are looking
function in the Everglades.

For the past 20 years, scientists and
from within their areas of expertise.
management problems. The land-use
chemists and toxicologists studied th
using their established protocols and
biological problems, but usually in a
ecosystem perspective. The only clea
that the altered (drained) system is c
stands very well.

The progress that is needed will depe
the section entitled "The ecosystem
geomorphology, and soils of wetland
on sawgrass cannot be complete wit

hydrological alternatives. The source of elevated mercury levels in the food chain is a vexing dilemma. Altered hydrology is cited as one of many possible causes, with a restored hydrology proposed as the solution. However, the changes in water quality parameters other than Hg, as well as shifts in plant and animal life that would likely accompany these hydrological modifications, must also be considered. The soils in the region vary greatly, and engineers routinely design water-control structures based on their compatibility with the soil conditions. However, less studied and understood are the possible influences of changing water regimes on biology and groundwater hydrology. Most importantly, land-use assumptions and decisions will continue to have a decisive impact on all of these analyses and outcomes.

The critical challenges in south Florida will be to develop an ecosystem approach and a landscape view to our science. Both of these areas represent critical gaps in our knowledge, but both are the focus of current initiatives to adjust our approach. Without an ecosystem approach, the information is incomplete and consensus is impossible. Without a landscape view, the issues become intractable and solutions impossible. The ecological framework to risk assessment allows scientists to examine the issues in a context that can provide the consensus necessary for success.

Research Needs and Recommendations

Previous ways of assessing wetlands have been expanded into the ecosystem approach outlined in this chapter. This approach integrates ecology, hydrology, geomorphology, and soils of wetlands for the evaluation of impacts and risks from chemical, biological, and physical stressors. When the ecosystem method to wetlands-specific risk assessment was applied, it became apparent that there is a need to establish and implement a consistent operational framework in order to make full use of this approach. Several concerns are evident. The effect of multiple stressors (chemical, physical, and biological, of anthropogenic or natural origin) must be an integral component of the assessment process. Standardization of reliable acute, subchronic, and chronic tests is necessary. Alternative exposure-effects scenarios must be evaluated. Understanding 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 fate of toxins need to be better quantified. There are also specific information needs for organismic, population and community, and ecosystem levels of organization.

Organismic

The levels of uncertainty resulting from presently used, standardized toxicity tests have not been carefully scrutinized in the context of freshwater wetland ecosystems. For example, plant toxicity data are generally based on one green alga *(Selenastrum capricornutum, Scenedesmus* sp. or *Chlorella* sp.) and one vascular aquatic plant

(duckweed species, *Lemna minor* or *L*
represent different groups of photos
organisms. Using a species battery ap
associated with interspecies extrapolat
vertebrates. Laboratory-to-field extrap
be improved by using ecologically rel
validation.

Population and community

As with interspecies comparisons, err
organizational extrapolation need to b
associated with transitions between d

Ecosystem

The ecosystem approach proposed he
wetland functions as the criteria for es
chemical and nonchemical stressors. (
zational levels-organismic, populati
approach may be used to describe exp
wetlands, both as a predictive tool anc
application of the approach will prov
chemical, and biological factors modif
integrating and analyzing these comp
or, in some cases, still need to be deve
of seasonal and spatial variability are

Toxicity assessments involve tests of
ecosystem assessments, etc.). As a rul
complexity and single-species laborat
cost-benefit perspective, the least com
tem effects should be the method of c
carried out. The ecosystem approach r
by identifying key biological, chemica
evaluated early in the assessment pro

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