SCOPE 53 - Methods to Assess the Effects of Chemicals On Ecosystems

8 Assessment of Effects of Chemicals on Wetlands

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8.1. INTRODUCTION

From the late 1950s, scientific studies have conclusively shown that wetlands contribute substantially to human health and welfare. Specifically, wetlands have been shown to improve water quality, provide food for commercially and recreationally important fish and shellfish living in nearby water bodies, serve as critical habitat for numerous species of wildlife, reduce shoreline erosion, and act as flood control devices (Hemond and Benoit, 1988). As important functions of wetlands were being identified, data were also being assembled that suggested that wetlands were being lost or degraded at alarming rates by either development or chemical pollution (Mitsch and Gosselink, 1986). These data raised questions in both the scientific and environmental management communities about which wetlands are most valuable, current rates of wetland loss or degradation, and the effects of chemical pollutants on wetland characteristics and functions. These and similar questions have been difficult to answer, because wetlands differ enormously in terms of their basic characteristics (i.e., vegetation type, soil type, areal extent, spatial configuration, hydrology), their geographical distribution, and their abundance in different state or provincial or regional watersheds.

Unlike changes resulting from development projects that are easily detected, adverse chemical impacts on wetlands are especially difficult to detect and evaluate. First, wetlands can be damaged by a variety of substances (e.g., high nutrient concentrations, heavy metals, and a vast array of organic compounds). Second, these chemicals may reach wetlands as a result of either point or non-point source discharge, and they may be transported by either water or air from nearby or distant discharge points. Third,
negative impacts of particular chemical pollutants may be limited to the wetland vegetation or to specific faunal species. Fourth, chemical pollutants in wetlands often cause chronic rather than acute problems for the affected organisms, making detection far more difficult. Other factors may further complicate the chemical impact picture. For example, chemical pollutants are often released at very irregular intervals, and, therefore, may move to a wetland site, adversely affect wetland organisms, and not be detected even with a sensitive monitoring program in place.

For these and other reasons, considerable attention over the past few decades has been given to the development and improvement of methodologies to assess the effects of chemicals on both plants and animals and on entire wetland systems. As a result, an array of methodologies is now available. These methodologies are summarized in Table 8.1.

Table 8.1. Methods to assess the effects of chemicals on wetlands

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<td>Fertilization and sewage addition</td>
<td>Assess capacity of wetlands to remove (field experiments) nutrients from sewage, and impact of fertilization on wetland system</td>
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energy/nutrient models
  Track energy flow through or nutrient cycling within wetland ecosystem

Hydrodynamic models
  Simulate pollutant transport through wetlands

spatial ecosystem models
  Combination of ecosystem and hydrodynamic transport models

Process models
  Analyse individual wetlands functions

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Monitoring

Abiotic monitoring
  Track changes in chemical concentrations and provide data for interpreting biomonitoring results

Biomonitoring
  Detect impacts from many sources and integrate intermittent stressor effects

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The discussion of these methodologies begins with a review of those that are routinely employed in bioaccumulation and experimental studies that are undertaken to evaluate the effects of chemicals on wetland plants and animals. Then attention is devoted to the chemical loading and modelling methodologies being developed to determine the effects of chemicals on wetland systems. Finally, wetland monitoring is discussed.

8.2 METHODS FOR STUDYING PLANTS AND ANIMALS

For the past three decades, water quality functions of wetlands have been a major concern of wetland scientists. Among these functions are their capacity to remove or transform excess nutrients, organic compounds, trace metals, sediment, and refractory chemicals from water as it moves downstream (Hemond and Benoit, 1988). In addressing questions related to these water quality functions, wetland scientists have: (1) developed sophisticated protocols for collecting samples, (2) refined methodologies for both field and laboratory experiments as well as methodologies to analyse biological materials, and (3) developed mathematical models for water quality studies. Those specifically applicable to studying wetland plants and animals are discussed in terms of either bioaccumulation determinations or experimental studies.

8.2.1 BIOACCUMULATION DETERMINATIONS

Measurement of chemical accumulation in tissues of organisms (bioaccumulation) is a methodology widely used to assess contamination in aquatic ecosystems. Bioaccumulation is assessed in the laboratory by first exposing organisms to a chemical, mixed in water or sediments, and then measuring the concentration of the chemical in the tissues (body burden) as a function of the duration of the exposure. The results may be
expressed as a bioaccumulation rate—the rate at which the body burden increases—or a bioaccumulation factor—the unitless quotient of the equilibrium body burden divided by the average exposure concentration. A fairly large number of bioaccumulation studies have been reported in the literature, including works by Anthony and Kozlowski (1982), Aulio (1980), Behan et al. (1979), Larsen and Schierup (1981), McIntosh et al. (1978), Mouvet (1985), Niethammer et al. (1985), Schierup and Larsen (1981), and Taylor and Crowder (1983).

Laboratory measurements of bioaccumulation permit chemicals to be tested one at a time. Another attractive attribute of this methodology is that it is not as affected by short-term fluctuations in contaminant levels as are measurements of sediment and particularly water-column concentrations of the chemicals. Longer-lived organisms integrate exposure levels over longer periods of time, so selection of a species, or age class, can influence results. Aquatic macrophytes, macroinvertebrates, amphibians, bird eggs, and small mammals may be preferable for detecting short-term exposures, whereas woody plants, fish, turtles, large adult birds, and for-bearing mammals may be better for monitoring long-term exposure (Leibowitz et al., 1991).

The two major drawbacks to bioaccumulation studies are cost and the difficulty of quantitative interpretation of the results. Interpretation is difficult because variability in tissue residues can be high, even within a species, because they are affected by many factors, including age, tissue type, diet, season, and the length of exposure. Therefore, a sufficient number of individuals from each sampling location has to be collected to overcome the limitation imposed by this variability. This requirement is regrettable, because tissue contaminant levels are costly to measure, especially if analyses are made for more than a few of the wide array of contaminants present in some environments (Leibowitz et al., 1991). Even more serious, however, is the difficulty of evaluating the consequence of a measured body burden to the individual organism, the population of which it is a part, or the higher trophic level consumers that feed on it. The problem is that few studies have attempted to correlate body burden with adverse effects on the organism, or to determine the elimination rates of accumulated chemicals from tissues upon cessation of exposure. Therefore, the longevity of the body burden and any associated harmful effects are unknown.

8.2.2 EXPERIMENTAL STUDIES

Experimental studies of the impacts of chemicals on wetlands have been conducted in the field and in laboratories, greenhouses, microcosms, and mesocosms. Most of these studies have emphasized nutrients, but some have been directed at heavy metals. Salt marsh plants, especially Spartina and Juncus, have been used frequently in experimental research projects.

Numerous field experiments to identify the effects of nitrogen and phosphorus fertilization on North American salt marshes (Chalmers, 1982; Morris, 1991) have been conducted. Typically, plots ranging in size from <1.0 m² to 100 m² were established in Spartina or Juncus dominated communities. A few plots were used as controls, while the others were enriched with various levels of inorganic fertilizer materials in a single application at the beginning of the growing season, or at weekly or monthly intervals. Response to the fertilization was often measured by calculating net primary productivity from changes in standing biomass. Examples of studies of this type include those of Tyler (1967), Valiela and Teal (1974), Mendelssohn (1979), de la Cruz et al. (1981), and Morris (1988). This procedure was modified by Patrick and DeLaune (1976) and Buresh et al. (1980) with the use of ¹⁵N-depleted nitrogen to permit differentiation between soil-derived and fertilizer-derived nitrogen in plant shoots. Van Raalte et al. (1976) and Sullivan (1981) investigated the effects of nitrogen enrichment on salt marsh diatom communities. Wetland nutrient addition experiments have also been made in North American arctic tundra (Shaver and Chapin, 1980), in European fens and wet grasslands (Vermer, 1986), and in wet heathlands in the Netherlands (Aerts and Berendse, 1988). Several laboratory studies have been made to test the influence of nitrogen levels on the growth response of Spartina alterniflora (Haines and Dunn, 1976;
Linthurst and Seneca, 1981) and on species of *Juno*us, *Littorella*, and *Sphagnum* (Roelofs *et al.*, 1984). Response was measured by following changes: in plant height, total dry weight, rhizome length and weight, and root weight.

Research on the capacity of wetlands to remove nutrients and other chemicals from wastewater has used nutrient enrichment experiments carried out in the field (Godfrey *et al.*, 1985; Nichols, 1983). The most complete study of the effects of sewage on a salt marsh is the multiyear study by Valiela and Teal in Massachusetts, US, in which they made biweekly additions of a commercial fertilizer made from sewage sludge (Valiela and Teal, 1974; Valiela *et al.*, 1975, 1976, 1985). Similar studies were conducted by Chalmers (1979) and Haines (1979). Sewage treatment capacity has also been studied in several freshwater wetland types (Sloey *et al.*, 1978; Nichols, 1983; Fetter *et al.*, 1978; and Bayley *et al.*, 1985). The retention of nutrients and heavy metals by a tidal freshwater wetland receiving urban stormwater runoff was estimated by comparing inputs with outputs (Simpson *et al.*, 1983). Lan *et al.* (1992) described the use of a pond with cattails (*Typha latifolia*) for treating wastewater from a lead and zinc mine in China.

The uses of laboratory microcosms and mesocosms for marine and freshwater ecosystem research have been described in volumes edited by Giesy (1980) and Grice and Reeve (1982), and a few wetland systems studies have been reported that made use of these enclosures (Kitchens, 1979; Portier and Meyers, 1982). The flume approach was used by Windom (1977) in Georgia to measure the uptake by the marsh of nutrients and heavy metals contained in effluent from dredged material disposal. For the most part, however, wetlands work has emphasized greenhouse-type studies in which various kinds of plants have been grown in different soils under varying conditions. For example, in a greenhouse experiment designed to differentiate between acid and nitrogen effects, mixtures of different wetland plant species were exposed to simulated rain containing various combinations of inorganic sulphur and nitrogen (Schuurkes *et al.*, 1986). Manipulations and additions involving complete portions of wetlands with sediments, many plants and animal species, and water flows have been rare (Nixon and Lee, 1986).

**8.3 METHODS TO STUDY THE EFFECTS OF CHEMICALS ON THE SYSTEM ITSELF**

**8.3.1 INVENTORIES**

Development of methodologies to address the broader issue of the toxicity of chemicals on wetland systems should begin with the realization that all wetland-organisms, processes, and functions cannot be studied, or even monitored, on a regular basis for even a single chemical, much less for the vast array of chemical pollutants found in wetlands. Hence, a major problem for both the wetland scientific and management communities is that of selecting those wetland sites that should be studied (i.e., monitored), and the specific chemical pollutants that should receive special attention in regulatory, monitoring, and research programmes. The resolution of this important problem requires comprehensive inventories of both wetlands and the sources of potential chemical pollutants. Moreover, these inventories need to be prepared on both national and local scales (i.e., city or county jurisdiction, watershed, state or province).

The national perspective is essential for several reasons. First, wetlands are an important natural resource for many nations, and issues concerning them are issues of national interest. Second, funding for either wetlands protection or control of chemical discharges that adversely impact wetlands is generally provided by national governments. Hence, these funds should be used to: (I) protect a nation's most valuable wetlands and (2) control those chemical discharges that are most threatening to their health. The local inventories are essential, because most public decisions involving wetlands focus on specific wetland sites and on particular developmental activities or chemical discharges that threaten the specific sites. The following summarizes the essential features and considerations concerning preparation of the four inventories:
1. National wetlands inventories. A national inventory should show the distribution of all wetlands by location, type, and areal extent; and data on wetland loss over time should be included if available (Office of Technology Assessment; OTA, 1984; Turner et al., 1981; Gosselink and Baumann, 1980; Tiner 1984). Assembly of the national wetland inventories should be based on the local wetland inventories described below.

2. Local wetlands inventories. Local wetland inventories should be developed in conformance with national guideline, and should be used to assemble the national inventory of wetlands. Local inventories should contain information in much greater detail than the national inventories, because the local inventories can be used for site-specific decisions on wetland development projects.

3. National inventories of sources of potential chemical pollutants. This inventory should identify both point and non-point sources of chemical pollution (Pait et al., 1992; Quinn et al., 1989). Data on nutrients, heavy metals, and organic compounds should be included. The inventory should be assembled from data contained in the local inventories.

4. Local inventories of sources of potential chemical pollutants. The local inventories should be developed according to national guidelines, and the information collected used to assemble the national inventory on sources of chemical pollution. Like the national inventory, the local inventories should also contain data on nutrients, heavy metals, and organic compounds.

These inventories by themselves will not establish the adverse impacts of chemical pollutants on wetland systems. They can, however, be used to: (1) identify the wetlands within a system that may be threatened by chemical pollutants, (2) identify the threatening pollutant(s), (3) permit generalization to be made about the severity of the threat, and (4) suggest specific wetland sites and particular chemicals for subsequent studies designed to evaluate specific threats.

8.3.2 CHEMICAL LOADINGS AND MASS BALANCE DETERMINATIONS

Once specific wetland sites and chemical pollutants are selected for study, chemical loading estimates can be developed as indicators of the degree to which the wetland is threatened by toxicity or eutrophication. Loading data can also be used to assess the relative importance of various chemical sources (point and non-point) (Stanley, 1992) and to compare loadings among wetlands. By combining loading data with export values, a chemical budget or mass balance for the wetland can be generated. Although difficult and, therefore, costly to obtain in most situations, mass balances provide a measure of the capacity of wetlands to perform their normal water quality functions (e.g., immobilization of toxic pollutants, plant nutrient removal, removal of biological oxygen demand) (Hemond and Benoit, 1988). Anyone interested in the mass balance approach should consult the excellent review by Nixon and Lee (1986). Although this review is limited to the US literature, it provides a critical assessment of all research between 1970 and 1985 on the role of wetlands as sources, sinks, and transformers of nutrients and heavy metals. The authors cite hundreds of examples of loading and mass balance studies, and argue that developing the ability to predict responses to changes in inputs of these materials depends on a knowledge of the various cycles and flows within the system, and that a mass balance can provide the framework for developing this knowledge. Another useful review by Johnston (1991) references numerous studies of internal nutrient fluxes among wetland compartments.

While conceptually straightforward, a complete budget is difficult to develop because, as Brinson et al. (1980) noted, most wetlands are very open ecosystems by virtue of lateral inflows and outflows during flooding. Thus, obtaining accurate water flow data for the computation of fluxes and mass balances is difficult. Consequently, researchers have frequently resorted to estimating fluxes across wetland boundaries indirectly by measuring various internal storages and flows. The result, concluded Nixon and Lee (1986), is a considerable amount of confusion about whether specific wetland systems act as
sources, sinks, or transformers of nutrients, metals, etc., at different times of the year.

8.3.3 MODELLING

Wetland modelling, although relatively new compared to modelling of other types of ecosystems, can also be employed to provide insights into the effects of chemicals on the system itself. Wetland modellers have borrowed from the more developed lake and estuary modelling techniques (Henderson-Sellers, 1984; Straskraba and Gnauck, 1985; Jorgensen, 1988) and from terrestrial models. Reviews of models of freshwater wetlands are summarized by Mitsch et al. (1982, 1988), Mitsch (1983), and Costanza and Sklar (1985).

Several types of models have been developed that incorporate the effects of chemicals on wetland biological structure and processes. Mitsch et al. (1983) summarized this modelling in the following way:

1. Energy/nutrient ecosystem models. Energy, nutrients, or other materials flow through or cycle among biotic and abiotic components and exchange with the surroundings. These models are often an outgrowth of chemical loadings and mass balance studies, or studies of the ability of wetlands to process wastewaters (Jorgensen, 1988). Examples include a model of energy flow and nutrients in an Everglades marsh system in Florida (US) (Bayley and Odum, 1976), a salt marsh carbon flow model (Wiegert et al., 1981), an energy-nutrient model of water hyacinth (Mitsch, 1976), models for cypress swamps in Florida (Ewel, 1976), a model of hydrology and nutrient cycling in forested wetlands (Mitsch, 1988a, 1988b), models of nutrient retention in a freshwater wetland (Mitsch and Reeder, 1992; Kadlec and Hammer, 1988), and a model dealing with acid mine drainage in a freshwater wetland (Mitsch et al., 1983).

2. Hydrodynamic transport models. These models, frequently used by engineers for stream flow and runoff calculations, have been applied to simulate wetland hydrology and pollutant transport. Hopkinson and Day (1980) described the application of one of these models (the Storm Water Management Model, SWMM, developed by the USEPA) to predict nutrient dynamics and eutrophication effects in a Louisiana (US) swamp forest resulting from anticipated increases in nutrient loading over a twenty-year period.

3. Spatial ecosystem models. These models combine features of ecosystem models with hydrodynamic transport models. The entire ecosystem is separated into discrete blocks or nodes, with each node having the characteristics of an ecosystem model. The nodes are connected by hydrodynamic processes. A suite of spatial models has been used to investigate various consequences associated with using freshwater wetlands for disposal of treated sewage (Kadlec and Tilton, 1979).

4. Process models. This type of model emphasizes individual processes, such as photosynthesis and nutrient cycling processes, rather than emphasizing interactions among compartments as in ecosystem models. Among the relatively few wetland examples of this model type are process models of primary production and vascular plant growth for wet arctic tundra (Miller et al., 1976, 1978a; 1978b; Stanley 1976).

8.4 MONITORING

8.4.1 ABIOTIC MONITORING

Programs designed to monitor chemicals in aquatic ecosystems have as their primary goal the identification of long-term trends in the quality of water and sediments of these systems. These programs also have secondary goals that include the provision of data for interpreting the results of other
approaches, such as biomonitoring studies. In addition, monitoring is required to estimate chemical loadings and mass balances, and to validate wetland process models. However, field monitoring is inherently retrospective, allowing effects of an activity to be observed, but not predicted. Many works provide information on strategies and practices for environmental monitoring. One of the best is by the National Research Council (NRC, 1990). In addition, a USEPA Environmental Monitoring and Assessment Program report is entitled *Research Plan for Monitoring Wetland Ecosystems* (Leibowitz et al., 1991).

Wetlands pose unusual challenges for monitoring programs. As transitional areas, they are located between uplands and deep water areas. Thus, they exhibit extreme spatial and temporal variability, which requires the collection of many samples for adequate characterization of the wetland. Only a minority of wetlands have permanent surface water, so access may be difficult, and sampling techniques developed for other surface waters are not always applicable. In some cases, downstream sampling may provide appropriate spatial averaging. Unless a relatively frequent sampling protocol can be followed, sampling for chemical contaminants in sediments should provide more consistent results than samples from the water column.

Several standard chemical measurement techniques developed for water and upland soil samples (American Public Health Association, 1989; USEPA, 1979; Parsons et al., 1984; Black, 1965) have been adapted for use in wetlands. Wetland pore waters are often anoxic, so that substantial chemical changes may occur between sampling and analysis if water samples are not rigorously protected. For example, oxidation can cause iron to precipitate and adsorb transition metals, while sulphide may oxidize and release metals. Also, high concentrations of humic substances in wetland surface and pore waters interfere with the chemistry of many standard analyses; nitrate and sulphate are prime examples in this regard. Apparently, no compilation or critical evaluation of chemical analytical methods used in wetlands has been produced (Hemond and Benoit, 1988).

### 8.4.2 BIOMONITORING

A growing body of literature exists which suggests that monitoring of biological - community structures provides cost-effective diagnostic information about the health of ecosystems. Biological monitoring (or biomonitoring) (1) directly addresses the result of pollution, not its cause, (2) integrates intermittent stressor conditions, and (3) can detect impacts from many sources for which chemical criteria are poorly developed. If biomonitoring suggests that a chemical stress is occurring, then traditional methods (e.g., field monitoring of chemicals, toxicity testing) can be used to help determine the cause(s).

Adamas and Brandt (1990) have prepared a valuable synopsis of the uses of biomonitoring to measure responses to chemicals and other anthropogenic stresses in wetlands. This USEPA-sponsored report first describes general considerations in the design of wetlands biomonitoring studies. Then, for each major taxonomic group (e.g., algae, herbaceous vegetation, birds, fishes), the authors discuss the group's responses to various stressors, and the sampling protocols and equipment appropriate for monitoring those responses. The following general considerations are discussed in the report:

1. Monitoring of multiple indicators¾having both short and long lifespans, and both localized and broad ranges¾is necessary, because some chemical effects may be brief (e.g., biodegradable pesticides), while others occur over long time periods (e.g., bioaccumulation of metals).

2. The choice of species to be used as indicators must take into account both policy and scientific considerations. Many of the scientific criteria point to the use of so-called keystone species, which physically alter the landscape so profoundly that they create or destroy habitat for a much larger group of species over a wide area. Examples of wetland keystone animals include beavers, muskrats, alligators, and some herbivorous birds. However, the choice of indicator species is an
unsettled issue; some researchers point out that changes in community-level measures give a clearer indication of chemical stress than does the presence or absence of a single indicator species, regardless of its reputation as a keystone (Browder, 1988; Karr, 1987; Kelley and Harwell, 1989).

3. A wide range of qualitative and quantitative biological sampling methods is available. Qualitative methods include the use of chronological vegetation maps to identify changes in plant composition, species lists and estimates of the presence or absence of indicator species, and the Habitat Evaluation Procedures (HEP) and the Wetland Evaluation Technique (WET), that do not directly measure biological communities but assume biological community structure or wetland function using information on habitat structure (Schroeder, 1987). Quantitative methods are detailed in the Adamus and Brandt (1990) report and in Erwin (1989), Hellawell (1986), Welcomme (1979), and Woods (1975).

4. Data analysis and interpretation involve the selection of appropriate measures (e.g., abundance, biomass, species richness, and similarity indices). To optimize detection of impacts, several procedures should be used in combination (Schindler, 1987). The utility and sensitivity of some procedures may vary by wetland type. For example, those that depend on species-level data (richness, ordination, and similarity indices) may be ineffective at describing the condition of wetlands that characteristically have low species richness (e.g., breeding bird richness in salt marshes and fish richness in montane wetlands).

8.5 CONCLUSIONS AND RECOMMENDATIONS

For the past three decades, wetland scientists have developed a wide array of methodologies to study the effects of chemicals on wetland organisms, processes, and functions. These methodologies include sampling protocols, field and laboratory experimental methods, and mathematical models. Use of these methodologies has greatly enhanced understanding of impacts of particular chemicals on wetland plants and animals. These methodologies to study plants and animals are now being coupled with information collected in inventories of wetlands and of chemical pollutant sources to characterize the effects of chemicals on wetland systems. Inventories are generally of national and local wetlands, and of national and local sources of chemical pollutants. They are undertaken to identify critical wetlands being threatened by either development projects or chemical pollutants, and to pinpoint pollution sources. The national inventories can also be used to either develop or justify: (1) regulatory restrictions on chemical pollutant discharges that may adversely affect wetlands, (2) the structure and funding of monitoring programs, (3) the allocation of funds for mitigation or restoration activities, and (4) the establishment of chemical pollutant and wetland ecology research priorities. Because most public decisions involving wetlands focus on specific wetland sites, the local inventories can also be used in public policy decisions concerning development projects or chemical discharges that threaten specific wetland sites.

8.6 REFERENCES


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