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WETLAND FUNCTIONAL HEALTH ASSESSMENT USING REMOTE SENSING AND OTHER TECHNIQUES: LITERATURE SEARCH

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WETLAND FUNCTIONAL HEALTH ASSESSMENT USING REMOTE SENSING AND OTHER TECHNIQUES: LITERATURE SEARCH AND OVERVIEW

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WETLAND FUNCTIONAL HEALTH ASSESSMENT USING REMOTE SENSING AND OTHER TECHNIQUES: LITERATURE SEARCH

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Abstract

The objective of this report is to provide a literature search and a short review of wetland functional health determination techniques which are relevant to the NOAA CoastWatch Change Analysis Program and other related programs, e.g., the Environmental Monitoring and Assessment Program (EMAP-Wetlands) of the Environmental Protection Agency. The report also suggests areas where further research is needed. In chapter 2, we review those remote sensing techniques which appear effective for mapping abundance (biomass). We also outline the contributions of remote sensing to early vegetative stress detection, habitat quality, and hydrology. In Chapter 3, we provide an overview of conceptual approaches for the assessment of wetland health, function and value. Then each of the proposed indicators of wetland condition is described in a chapter. Their importance is underlined, the techniques used for indicator sampling and measurement are briefly explained, and the remaining issues that must be resolved are outlined. Complete details of every technique are not discussed in this overview. The reader is encouraged to consult the references in each section for additional information.

Our choice of the health indicators is essentially based on conclusions drawn from various interagency reports and planning meetings attended by us and the EMAP-Wetlands major assessment endpoints: productivity and biodiversity defined by the variety of species inhabiting the wetland, and sustainability defined as the wetland persistence over time. The review includes all the remote sensing techniques that can be substituted for the conventional methods, or that are used in conjunction with them. Some indicators of wetland condition, such as wetland extent and type, habitat structure, and the floral component of wetland productivity, can be studied primarily by means of remote sensing; while others (e.g., vegetation, hydrology, habitat quality) still require the use of more conventional techniques. Satellite and airborne sensors have been used for several decades in wetland mapping, but new remote sensing techniques have recently been developed, that allow researchers to determine wetland biomass production. These new techniques should enhance our ability to determine wetland condition and functional health over large areas and at various repeat intervals.

1 • INTRODUCTION

Only within the past few decades have we begun to recognize the importance of wetlands as productive and valuable ecosystems with numerous functions that benefit society. Wetlands are transitional lands between terrestrial and aquatic systems where the water table is usually at or near the surface, or the land is covered by shallow water (U.S. Fish and Wildlife Service definition). Wetlands are often characterized by high rates of primary production (Gross et al., 1990). They are nursery areas for many commercially and recreationally important species of fish, shellfish, and wildlife. They often act as a protective buffer against storms by water storage and flood abatement, and against erosion damage, by sediment stabilization. They contribute to water quality improvement by immobilizing various pollutants and nutrients and also play a role in geochemical cycling.

Often viewed as wastelands, wetlands have been continually converted for other uses since the 1800s. As wetlands values are better perceived, new legislation is being introduced in many parts of the country to preserve these vital resources. Natural resource managers and scientists are now confronted with a number of questions (Leibowitz et al., 1991): What proportion of wetlands are in good condition; how many are in poor condition? Are conditions improving or degrading over time? In what proportion of the wetland resource are conditions continuing to decline and at what rate? What are the most likely causes of a poor or degrading condition? Which stress factors seem to be most important, adversely affecting the greatest numbers of wetlands? Answers to these questions require the use of standardized and accurate assessment techniques.

Timely quantification of wetland area, location and rate and cause of loss, is needed, now. Management decisions can then be proactive and based on fact rather than supposition of the effects of coastal development on coastal wetlands and wetland dependent fisheries. Current projections for U.S. population growth in the coastal zone suggest accelerating losses of wetlands and adjacent habitats, as waste loads and competition for limited space and resources increase. Agencies responsible for coastal management must be kept current with regard to extent and status of wetlands and adjacent uplands. Changes in wetlands are occurring too fast and too pervasively to be monitored once a decade. Therefore, NOAA within its Coastal Ocean Program, has initiated a cooperative interagency and state/federal effort to map coastal wetlands and adjacent upland cover and change in the coastal region of the U.S. every two to five years and monitor annually, areas of significant change. The program is called the NOAA Coastwatch Change Analysis Program (C-CAP).

NOAA's Change Analysis Program is being designed to determine land/habitat cover and change of seagrass, emergent wetlands and adjacent uplands in the coastal regions of the U.S. on a one to five year repeating basis. The 1-5 year monitoring cycle will provide feedback to habitat

managers on the success or failure of habitat management policies and programs. Frequent feedback to managers will help assure the continued integrity or recovery of coastal ecosystems and the attendant productivity and health of fish and other living marine resources at minimal cost. In addition, the geographical data base developed under the program, will allow both the manager and the researcher to evaluate and ultimately to predict cumulative direct and indirect effects of coastal development on wetland habitats and living marine resources.

Remote sensing (from satellites and aircraft) and other techniques will be used to quantify and map coastal wetlands and adjacent uplands. The first cycle will document status and change (retroactively). The data base, increasing with each subsequent monitoring cycle will be an invaluable resource for research; evaluation of local, state and federal wetland management strategies; and construction of predictive models.

Satellite and airborne remote sensors have been used for several decades to map the acreage of wetlands lost to natural processes and anthropogenic activities. Determining only the area of marsh lost, however, may not give an accurate description of the total degree of environmental degradation. A marsh may have lost only 20% of its area, yet if the hydrology has been seriously disturbed, the productivity of the remaining 80% may be only a small fraction of its previous level. In response to this and other needs, new remote sensing techniques are being developed by NOAA and other researchers to determine not only the change in marsh area but also its biomass production and other indicators of condition and functional health. Therefore in addition to determining losses in wetlands acreage over time, C-CAP is also trying to satisfy the following requirements:

1. To be able to see early functional change before areal change in habitat. We expect this to be a more sensitive approach for "early warning."

2. To be able to assess large areas rapidly and efficiently in order to allow for an early response to stress or impending environmental degradation.

The purpose of this report is to present the results of a literature search and an overview of approaches for the assessment of wetland health, functions and values. In Chapter 2, we review those remote sensing techniques which appear effective for mapping abundance (biomass). We also outline the contributions of remote sensing to early vegetative stress detection, habitat quality, and hydrology. In Chapter 3, we provide an overview of conceptual approaches for the assessment of wetland health, function and value. Then each of the proposed indicators of wetland condition is described in a chapter. Their importance is underlined, the techniques used for indicator sampling and measurement are briefly explained, and the remaining issues that must be resolved are outlined. Complete details of every technique are not discussed in this overview. The reader is encouraged to consult the references in each section for additional information. The indicators order is random and is not a function of their importance. They all are important parameters to be considered eventually for wetland health assessment.

2 • REMOTE SENSING OF WETLAND BIOMASS AND OTHER WETLAND CONDITION INDICATORS

NOAA's Change Analysis Program (C-CAP) is being designed to determine land/habitat cover and change of sea grass, emergent wetlands and adjacent uplands in the coastal regions of the U.S. on a one to five year repeating basis. C-CAP's long term objectives can be summarized as follows:

1. To be able to see early functional change before areal change in habitat. We expect this to be a more sensitive approach for "early warning."

2. To be able to assess large areas rapidly and efficiently in order to allow for an early response to stress or impending environmental degradation.

The Environmental Monitoring and Assessment Program (EMAP) was initiated in 1988 by EPA to provide improved information on the current status and long-term trends in the condition of the nation's ecological resources. The overall goal of EMAP is to provide a quantitative assessment of the current status and long-term trends in ecological condition on regional and national scales. In the short-term, EMAP will provide standardized protocols for measuring and describing ecological condition, provide estimates of condition in several regions, and develop formats for reporting program results. Trend detection will clearly require longer periods of data collection and evaluation, and therefore is an intermediate goal. Diagnostic analyses, to identify or eliminate plausible causes for degraded or improved condition, is considered the long term goal of for EMAP.

NOAA's C-CAP program is designed to determine land/habitat cover and change of seagrass, emergent wetlands, and adjacent uplands in the coastal region of the U.S. using satellite data to obtain continuous coverage of large areas. Thus if a small number of EMAP grid point samples could be used to "calibrate" C-CAP's satellite products, a continuous measurement of biomass and a few other key wetland condition or health indicators becomes possible. In other words, we can use the C-CAP satellite data to interpolate between EMAP grid points or extrapolate to large areas several of the key condition indicators.

To accomplish these objectives cost-effectively over large coastal areas, the C-CAP is cooperating with EPA's Environmental Monitoring and Assessment Program (EMAP), since both programs complement each other. EMAP-Wetlands will use standardized sampling methods and a probability-based sampling design to monitor wetlands over broad geographic areas and for multiple decades. The outputs from this program will be for the estimates of wetland condition for the regional wetland population (i.e., all wetlands of interest, with a given region), not site-specific information. The proposed design strategy is based on a permanent national sampling

framework consisting of systematic triangular point grid placed randomly over the conterminous United States; a similar array is available for Alaska and Hawaii. This grid identifies approximately 12,600 locations at which all ecological resources will be catalogued and classified. Using existing maps, aerial photography, and satellite imagery, the numbers, classes, and sizes of wetlands will be determined for the area included within a 40 km^2 hexagon centered on each grid point. These 40 km^2 hexagons (40-hexes) describe an area sample representing one-sixteenth of the area of the United States, and provide the basis for the Tier 1 estimates of wetland extent and distribution.

The question of which indicators should be sensed remotely can be resolved as follows. The three major assessment endpoints suggested by EMAP Wetlands are:

1. Productivity, including both floral and faunal components.

2. Biodiversity, defined by the variety of floral and faunal species inhabiting the wetland, in terms of both community composition and structure, as well as the functional niches that are represented.

3. Sustainability, defined as the robustness of the wetland; its resistance to changes in structure and function and persistence over long periods of time, as measured by both a wetland's size and hydrology.

Also, at recent interagency meetings, including implementation meetings for the Louisiana Coastal Wetland Planning, Protection and Restoration Act (Mitchell, 1991), it was concluded that vegetative abundance (biomass) and species composition (biodiversity) were the two most practical indicators for monitoring wetlands condition over large areas. It is fortuitous that both of these properties can be detected and mapped from satellites and aircraft with considerable success.

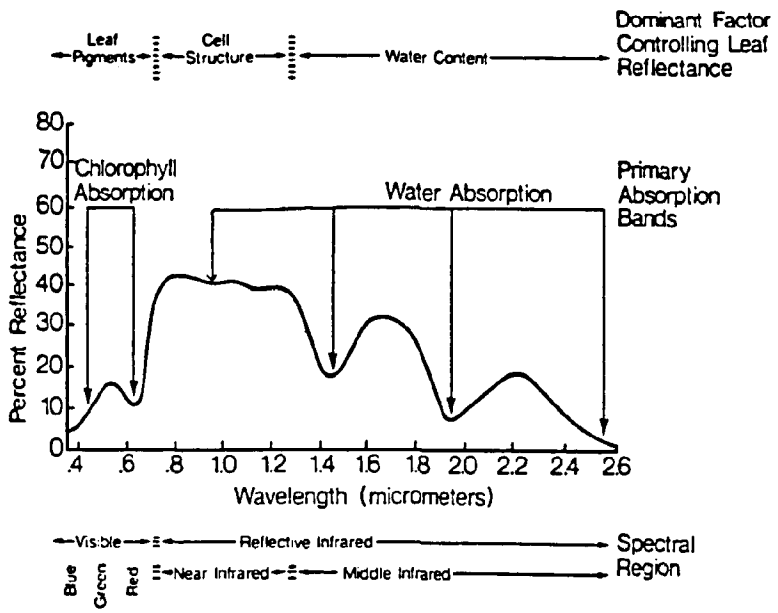
Wetland mapping by remote sensing (areal extent and floral species composition) is amply discussed in chapter 4, and in the Implementation Manual being developed by C-CAP. This chapter will, therefore, turn its attention to remote sensing of wetland biomass, and the contributions of remote sensing to early vegetative stress detection, waterfowl habitat quality, and hydrology. These contributions are also described in the other sections of the report.

In order to understand the principles of remote sensing technology used in wetland monitoring, first let us examine some spectral properties of plants.

2.1. Spectral Properties of Plants

Early reports describing the interaction of leaf tissue with light indicated that changes in the spectral quality of reflected electromagnetic radiation were directly related to the quantity of leaf tissue and pigment concentrations (Allen and Richardson, 1986; Colwell, 1974; Gausman, 1974).

As green aboveground biomass increased, the most significant of the spectral changes were a decrease in red radiation resulting from strong absorption by the chlorophylls, and an increase in near infrared radiation resulting from intra- and interleaf scattering (see the figure below). Although these findings were made in non-wetland environments, wetland plants show the same patterns (Bartlett, 1976; Bartlett and Klemas, 1981; Drake, 1976, Jensen, 1980).



Reflectance characteristic of healthy, green leaves (derived from HOFFER and JOHANNSEN 1969). It applies in the shape of the curve also for trees with full, green foliage and for dense canopies of healthy forest stands.

The low reflectance in the blue and red regions is due to strong absorbance of these wavelengths by the chlorophyll. There is a slight peak in the green region, because plants do not absorb green, but reflect it. The high reflectance in the near infrared region is controlled by plant tissue structure and results from the scattering effects of the mesophyll (Boyer et al., 1988). Beyond 1200 nm, the decrease in infrared absorption is due to the absorption by water (Knipling, 1968).

2.2. Wetland Biomass and Productivity

2.2.1 Definition

The most commonly used and accepted parameter for evaluating an ecosystem condition is biomass and/or net primary productivity of the emergent macrophytes. Both terms refer to the dry weight of plants (expressed as grams dry weight per square meter -gdw/m^2- for the biomass and usually as gdw/m^2 per year for the productivity).

2.2.2 Reflectance and aboveground biomass

Early work by Bartlett (1976, 1979) determined green biomass of wetland grasses to be strongly correlated with the near infrared/red reflectance ratio. Other investigators found good correlations between green biomass and the spectral reflectance of different marsh shrub communities (Hardisky et al., 1986). In most cases, the combination of red and near infrared radiance provided the best correlation with canopy biomass.

Several years ago, a simple linear regression model equating spectral reflectance to biomass was formulated for Delaware *Spartina alterniflora*, one of the most common salt marsh plants in eastern North America (Hardisky, 1982, 1984). Spectral reflectance measurements were made in selected portions of the marsh using a hand-held radiometer that gathered data in three wavebands, spectrally configured to simulate bands 3, 4, and 5 of the Landsat Thematic Mapper: a red band (630-690 nm, TM3) sensitive to chlorophyll concentration, a near infrared (NIR) band (760-900 nm, TM4), sensitive to plant tissue structure, and a middle infrared band (1550-1750 nm, TM5) sensitive to water absorption.

The raw radiance data were transformed and expressed as a normalized difference of two bands as follows:

$$VI = [NIR - red] / [NIR + red]$$

$$II = [NIR - middle\ IR] / [NIR + middle\ IR]$$

where VI is the Vegetation Index and II the Infrared Index. Index values were preferred to raw radiance data because the normalization procedure tended to compensate for both short- and long-term changes in solar irradiance and atmospheric conditions (Tucker et al., 1981). Both indices, VI and II, correlated strongly with the changes in biomass (Hardisky, Smart and Klemas, 1983; Hardisky, Gross and Klemas, 1986). Measurements of biomass were combined over a growing season to yield an estimate of NAPP (net aerial primary productivity). The NAPP estimates were generally within ten percent of harvest estimates (Hardisky et al., 1984).

Biomass evaluation has also been attempted with satellite imagery (Gross et al., 1987). The problem of atmospheric effects on the satellite-measured radiance data was solved by converting the satellite data to the equivalent ground-measurement reflectance. This was done using equations relating the reflectance of certain large, homogeneous sites measured from the ground at the time of the satellite overpass, to their satellite-measured radiance. The satellite-derived estimates were found to be within 13 percent of ground-based biomass estimates. The nature of the relationship linking VI and the aerial biomass was consistent from year to year and between marshes, although there was a difference between northern and southern marshes (Gross et al., 1990).

By comparing satellite data from several years, it was found that in some parts of marshes, the growth of *Spartina alterniflora* was more sensitive to the amount of precipitation than in other

parts of the marsh (Gross et al., 1990). The most sensitive areas seemed to be the closest to the saltiest water. The plants nearest freshwater areas showed the least response to quantity of precipitation. From this work, it will be possible to predict in advance which parts of the marsh will be most affected by drought, by unusually wet weather, or by man-made hydrologic disturbances (Gross et al., 1987; 1990).

2.2.3 Belowground biomass estimation

Light does not penetrate soil, making it impossible to measure root biomass directly by optical remote sensing. However, Gross et al. (1990; 1991) report a strong positive relationship (r2 = 0.86) between the natural logarithm of live aboveground biomass and the natural logarithm of live belowground biomass of *S. alterniflora* ("short" plants only). Therefore, belowground biomass can be indirectly measured using a non-destructive method.

Another promising technique for belowground biomass estimation is the use of ground-penetrating radar (GPR), but it is still under evaluation. Traditionally, GPR has been used to locate things such as archaeological sites, toxic waste drums, and divisions between contrasting soil types like sand and clay. A radar antenna is dragged along the surface of the marsh, emitting electromagnetic waves. These waves penetrate the soil, and are reflected back by objects in the soil. The return signal is recorded and printed in graph form (Gross, 1989). Researchers hope that the characteristics of the return signal will reveal something about root material.

2.2.4 Factors influencing spectral estimates

One of the factors that influence the spectral radiance of the marsh is the solar angle which can easily be corrected (Hardisky, Gross and Klemas 1986). Two other factors are the quantity and orientation of dead biomass, and the amount of soil reflectance (Hardisky et al., 1986). The presence of dead material tends to decrease the vegetation index. Except in marshes with very sparse canopy (<30% cover), soil reflectance is not usually a problem. Richardson and Wiegand (1977) have proposed a perpendicular vegetation index (PVI), which factors out the influence of soil reflectance. The infrared index is less attenuated by dead biomass and soil reflectance than the vegetation index.

2.2.5 Conclusions and research needs

Remote sensing is considered an accurate and effective non-destructive biomass assessment technique in salt marshes despite its limitations: sampling can only be done on sunny days, for four hours, and only during a tidal stage when the marsh surface is not flooded (Hardisky et al., 1984). Hand-held radiometers have been extensively used to assess biomass and NAPP of small wetland tracts, but satellite imagery is more useful for sampling larger areas. The aerial biomass estimation technique is based on the use of simple regression models equating the green biomass with spectral radiance indices. Root biomass can then be estimated using equations linking aboveground and belowground biomass.

Limited remote sensing work has been conducted in other types of wetlands such as brackish marshes and coastal mangrove systems (Hardisky et al., 1986). Salt marshes are often characterized by large monospecific stands of vegetation. In contrast, the physiognomy of brackish marshes is usually more varied because a particular plant community often comprises many species. Different plant morphologies thus coalesce to produce canopy architectures that reflect incident radiation differently from a monospecific canopy (Hardisky et al., 1986). Hardisky and Klemas (1985) analyzed the effects of three canopy types on the vegetation index. Since the quality of reflected radiation (expressed as a vegetation index) differs for each canopy architecture, accurate biomass predictions must rely on separate models describing each type. Studies by Hardisky (1984) suggested that biomass could indeed be predicted for communities of one canopy type using a single model. The work conducted by Hardisky et al. (1986) in the black mangrove, Avicennia germinans, in Costa Rica, describes a positive relationship ($r=0.79$) between the TM vegetation index and live leaf biomass. The more ubiquitous taller mangrove forms will require extensive ground comparisons before an operational biomass estimation procedure can be developed (Hardisky et al., 1986).

2.3. Early Vegetative Stress Detection

When plants are subjected to stressful conditions, certain physiological changes occur, that can be detected by remote sensing, because of the consequent changes in plants spectral reflectance. These physiological changes relate to chlorophyll density, cellular size and arrangement, and moisture content. As a plant is exposed to various stressful conditions (disease, insects, moisture and mineral stress, etc.), two changes in reflectance are observed: 1) visible reflectance increases, because there is less chlorophyll and/or the chlorophyll is less efficient in absorbing red and blue light; 2) near infrared reflectance decreases, because of a deterioration of the mesophyll cells (Campbell, 1987). Reflectance changes can be detected before visible symptoms appear, and thus, are good indicators of plant stress (Knipling, 1968). Moisture stress is usually evidenced by an increased radiant emission from the plant and thus lighter tones in images (Weaver et al., 1968). Nitrogen deficiency will result in increased reflectivity of a single leaf, but in decreased reflectivity of the whole canopy, because of the decrease of leaf surface area per unit ground area (Hardisky, 1984).

2.4. Waterfowl Habitat Quality

Waterfowl habitat quality is a function of both water conditions and terrain characteristics of the surrounding wetland and upland cover types (Colwell et al., 1978). Habitat quality, according to Colwell et al., relates to the potential of the habitat to attract breeding waterfowl and furnish the requirements for survival and successful rearing of broods. They developed a model for the assessment of waterfowl habitat quality based on the various relationships between ponds and the surrounding terrain types.

The model developed by Colwell et al. (1978) evaluates waterfowl habitat quality on the basis of water conditions and terrain characteristics. The specific water conditions are pond number, pond area, and pond size-class distribution. Once these factors were calculated, they integrated them into one single pond factor. The terrain characteristics they evaluated were the presence and spatial arrangement of certain terrain types (hay, grasses, pasture). They incorporated presence and spatial arrangement into a single factor represented by the amount of edge between desirable terrain types. The resulting model for waterfowl habitat quality combined pond and terrain factors, and generated ratings on a section-by-section analysis. This habitat model was preliminary; no detailed analysis of the accuracy of the model ratings has been made. Colwell et al. (1978) used Landsat data in their model. Pond and terrain characteristics were determined from multidate Landsat imagery and aerial photography. Remote sensing data allow monitoring changes in the habitat quality over time. With the advent of satellites with better spatial resolution (e.g., SPOT), it is possible to improve the accuracy of the pond and terrain factors.

2.5. Hydrology

Remote sensing can provide some information on the hydrology regime of the wetland, such as changes in surface level, in open areas, and in soil moisture. A number of studies have used remote sensing as a method for flood analysis and soil moisture assessments (Sollers et al., 1978; Harker and Rouse, 1977; Ragan, 1977; McGinnis and Rango, 1975; Rango and Anderson, 1974; Moore and North, 1974; Rango and Salomonson, 1974; Williamson, 1974; American Water Resources Associations, 1974; Piech and Walker, 1971), (Schmugge, 1983; Cihlar, 1978; Schmugge et al., 1977; Myers et al., 1977; Idso et al., 1975; Blanchard et al., 1974; Waite et al., 1973; Werner et al., 1971). Microwave radiometric sensors are very effective at measuring water content, both in the atmosphere and on the earth's surface. These systems can be used to map areal distribution and variations in rainfall, water absorption rates of surface soils and map flood water distribution and flow patterns on an all-weather synoptic basis (Kennedy 1968). The microwave radiometer functions as a temperature-measuring device. The capability of the radiometer to measure atmospheric hydrology derives from the electromagnetic properties of atmospheric water vapor, oxygen, clouds, rain, and the earth's surface which differ greatly in electromagnetic properties. The dielectric properties of surface materials are strongly dependent on moisture content. Changes in the dielectric constant result in major changes in the emissivity and radiometric brightness temperature (Kennedy 1968).

2.5.1 Flood monitoring

Aircraft and satellite data have been used to perform floodplain mapping by two complementary approaches: static and dynamic (Sollers et al., 1978). The static approach is based on the recognition of geomorphological features formed by historical floods such as terraces, alluvial fans, natural levees, bars, oxbows, marshes, deltas, etc. Floodprone areas tend to have multispectral signatures that are distinctly different from those of surrounding nonfloodprone areas. The dynamic approach uses images of floods as they occur or soon afterward. Visible evidence of inundation in the near infrared region of the spectrum remains for up to two or more

weeks after the flood. The near infrared reflectivity is reduced in the flooded areas because of the presence of increased surface-layer soil moisture, moisture stressed vegetation, and isolated pockets of standing water. The inundated areas are characterized by the water absorption band (700-1100 nm). Visible and near infrared channels are recommended for analysis. The features observed here are the atmospheric conditions (clouds, air mass characteristics, precipitation), flood water levels, and soil and vegetation characteristics after the high waters have receded. Soil moisture and sediment traces in water can indicate the path and extent of flood damage to a plain (Currey, 1977). Vegetation also exhibit patterns related to flood conditions: flood stressed plants reflect more blue and less infrared radiation (Sollers et al., 1978).

Satellites, such as ERTS, NOAA, Landsat, and SPOT could help reduce short- and long-term flood losses and provide regional water resources planning information. Data from these satellites would therefore complement aircraft and conventional surveying methods to ascertain the areal extent of flooding (McGinnis and Rango, 1975). Despite its usefulness in flood monitoring, remote sensing has limitations: 1) some systems don't have the resolution needed to delineate the boundary of flooded areas; 2) the scale of floodplain mapping is not large enough for most legal requirements; and 3) clear weather conditions are necessary with passive sensors. When possible, a combination of sensors should be used. Remote sensing data can serve as a base for assessment of potential flood damage, in identifying areas where further surveys are merited.

2.5.2 Soil moisture assessment

Soil moisture and its spatial and temporal behavior is of critical importance to disciplines such as agriculture, hydrology, and climatology. Specifically, soil moisture assessments are needed to study flood water distribution and flow patterns, distribution and variations in precipitation (especially rainfall), runoff following precipitation, and evapotranspiration (Kennedy 1968; Cihlar, 1978).

Most techniques developed for soil moisture measurement provide point estimates, therefore are not suited for large areas (Cihlar, 1978). The traditional method of soil moisture measurement is to weigh a sample of soil, oven-dry it, and reweigh it. The difference between the wet and dry weights represents the soil moisture, and the percent moisture is then extrapolated to the entire field. This method is time-consuming and representative of only small areas. The status of remote sensing techniques for soil moisture estimation was reviewed in a workshop organized in 1978 in Maryland (Cihlar, 1978). The techniques discussed were: 1) the reflected solar technique; 2) the thermal infrared technique; 3) the active microwave (radar) technique; 4) the passive microwave (radiometer) technique; and 5) the gamma radiation technique. The review indicated the complementary nature of the various techniques. Thus, it is likely that a combination of sensors will be needed to provide accurate soil moisture estimates from satellites. Thermal infrared and both microwave approaches have shown potential for estimating near-surface water contents, but the sensitivity to water at greater depths and under canopy seemed limited to the thermal infrared technique (Cihlar, 1978).

2.6. Conclusions and Recommendations

Remote sensing is a very helpful tool in wetland management. Remote sensors on aircraft and satellites can be used to select sampling sites, establish field transects, identify and delineate the major vegetation types, map wetland changes and to detect early vegetative stress. It also allows to document present conditions for use in future trend analyses. Methodologies and algorithms for the determination of biomass and productivity of coastal wetlands habitat by remote sensing have been recently developed and will significantly enhance our ability to determine wetland condition over time on a regional scale. Remote sensing also provides information on physical alterations to the wetland (flooding, human activities, etc.), soil moisture, and the wetland hydrological regime. By comparing two or more time periods, change in biomass, productivity, wetland extent, type, and patterns, wetland vegetation community composition, or any other factor correlated with spectral reflectance could be used to index functional health. The activity requires ground-based research to relate remotely sensed spectral radiances to these indicators. Various remote sensing methods are available, and the choice of the method will depend on the project objectives and monetary restraints. Low-resolution data may be sufficient for the study of certain parameters, and higher resolution data will be required for detailed studies of selected sample sites. Our management effectiveness in the future will depend on our ability to collect and analyze data on a regional, and eventually, global scale. The advances in instrumentation and in computer analysis techniques will greatly improve the types of data available.

Vegetative abundance (biomass) and species composition (biodiversity) have been identified as two practical indicators for monitoring wetland condition over large areas by remote sensing. Therefore, there is an urgent need for research to test the applicability of remote sensing for operational determination of biomass, productivity and species diversity over a large range of wetland types. Simultaneously, other applications of remote sensing for wetland health evaluation should also be investigated.

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3 • CONCEPTUAL APPROACHES IN WETLAND ASSESSMENT

3.1. Introduction

Development of a monitoring strategy for wetlands depends on the objectives of a study (Brooks 1989). Is the purpose of the study to detect an improvement or decline in a wetland's condition? Is the reason for monitoring related to a specific concern or function, such as water quality, species diversity, etc.? Is the purpose of the study to evaluate the success of a mitigation project? The investigator's objectives will determine the amount of time and funds required for a monitoring program. This program can range from single site visits to assess the condition of a wetland after an anticipated event (e.g., flood occurrence or completion of a mitigation project), to long-term studies of the cumulative impacts on wetlands and their surroundings (Brooks, 1989).

3.2. Assessment of Ecosystem Health

3.2.1. Definition of functional health

The complexity of an ecosystem makes it difficult to define its state of health. The environmental biologist is confronted by an infinite number of parameters which might be measured. But in practice, a healthy ecosystem may be defined only by reference to a few parameters, and the absence of disease is based on comparison to one or more poorly quantified "ideal" ecosystems (Schaeffer et al., 1988). According to Karr et al. (1986), "a biological system ... can be considered healthy when its inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed".

Like physicians, we define health in ecosystems as the absence of disease. The relationships between measures of structure (numbers and kinds of organisms, biomass, etc.) and function (activity, production, decomposition, etc.) help identify some diseases (Schaeffer et al., 1988). Ecosystem impairment has short-term and long-term, major and minor effects. Illness may be defined in relation to short-term shifts in ecosystem elements considered critical for the maintenance of ecosystem function, and may be viewed as acceptable if the degree of degradation is limited by ecosystem resiliency or managerial intervention. Disease, then, would be related to long-term and permanent shifts in critical ecosystem elements; these shifts include falling numbers of native species, overall regressive succession, rapid alteration in the quantity of either living or dead biomass, changes in energy production and flow, changes in mineral macronutrient stocks and in the capacity of the ecosystem to damp undesirable oscillations of contaminant concentrations (Schaeffer et al., 1988). Because an ecological system is comprised of many

interacting species, a functional definition of a healthy ecosystem must be given as a set of ecological requirements (Schaeffer et al., 1988).

3.2.2 Wetland health evaluation

No indices of wetland health currently exist that are widely accepted and have been tested and applied on regional scales. Ecosystem health evaluation involves: 1) the identification of systematic indicators of ecosystem structural and functional integrity, 2) the measurement of ecological sustainability and 3) the detection of potential symptoms of ecosystem disease or stress (Rapport, 1989). Four types of indicators may be distinguished (Leibowitz et al., 1991): 1) response indicators, which provide a metric of biological condition (e.g., vegetation community composition); 2) exposure indicators, which assess the occurrence and magnitude of contact with a physical, chemical, or biological stressor (e.g., nutrient concentrations); 3) habitat indicators, which characterize the natural physical, chemical, and biological conditions necessary to support an organism, a population, or a community (e.g., wetland hydrology); and 4) stressor indicators, which quantify natural processes, environmental hazards, or management actions that result in changes in exposure or habitat (e.g., changes in land use). Indicator selection must be parsimonious, including only those that most effectively define wetland condition. This procedure requires extensive testing and evaluation. To classify ecosystems as healthy or unhealthy, it is important to use objective criteria for a range of parameters that are appropriate to each ecosystem (Schaeffer et al., 1988). Thus, region-based quantitative definitions of ecological health are required (Karr, 1991).

For each wetland class, in each region, wetland condition may be evaluated by comparing the measured indicator values with: 1) expected normal ranges for each response variable, derived from measurements at reference sites, historical records, the available literature and/or expert judgment; 2) information on stress-damage thresholds for each exposure indicator, obtained from the literature and available data (Leibowitz et al., 1991). Reference sites may be monitored for each wetland class and region, representing the least disturbed and most disturbed wetland in the landscape. Wetlands classified as healthy are assumed to perform as expected for a wetland of that type, within that region and for the specific wetland value of interest. Classification of a wetland as healthy or unhealthy should not rely on a single indicator, but on the full set of monitored response, exposure, habitat and stressor indicators (Leibowitz et al., 1991). Additional criteria are needed to distinguish between acceptable and unacceptable degrees of a diseased state because recovery from disease is possible (Schaeffer et al., 1988). Health parameters must be assessed differentially with the age or developmental stage of the ecosystem. Also, the assessment should reflect our knowledge of normal succession or expected sequential changes, which occur naturally in ecosystems. This requires change in target measurements and modification of criteria as ecosystems change (Schaeffer et al., 1988).

Three assessment endpoints are considered in EMAP-Wetlands program (Leibowitz et al., 1991) to reflect the major social and biological values associated with natural wetlands: 1) productivity, including both floral and faunal components; 2) biodiversity, defined by the variety of floral and faunal species existing in the wetland, in terms of community composition and structure, as well

as the functional niches that are represented; and 3) sustainability, defined as the robustness of the wetland, i.e., its resistance to changes in structure and function and persistence over long periods of time, as measured by both wetland's size and hydrology.

The assessment of ecosystem health requires the analysis of selected parameters based on criteria developed from experimentation coupled with some reference to healthy ecosystems (Schaeffer et al., 1988). It requires standard procedures, precision and accuracy in analyses. Unfortunately, these standardized techniques do not yet exist.

3.2.3 Use of biological indexes

Early efforts to develop biological indexes concentrated on detecting a narrow range of variation in biological integrity (Taub, 1987; Ford, 1989; Fausch et al., 1990), resulted in indexes sensitive to only a few types of degradation, or provided only a binary (degraded/not degraded) evaluation (Karr, 1991). Many existing biological indexes may only apply to a narrow geographic area, and do not screen complex cumulative impacts. Such approaches are appropriate when specific narrow impacts are known to be present, but protection of natural resources from a broad range of human impacts requires a more comprehensive approach. The ideal index would be sensitive to all stresses placed on biological systems by human society while also having limited sensitivity to natural variation in physical and biological environments (Karr, 1991). An array of indicators would be combined into one or more simple indexes and could be used to detect degradation and identify its cause, and to determine if improvement results from management actions (Karr, 1991).

Many indicators of the health of biological systems have been tested in recent years (National Academy of Sciences, 1986; Schindler, 1987; Taub, 1987; Ford, 1989; Gray, 1989; Pontasch et al., 1989; Karr, 1990). Each has sensitivity at different levels of degradation and to different kinds of anthropogenic stress. The complexity of biological systems and the diversity of factors responsible for degradation, makes it unlikely that any metric will have sufficient sensitivity to be useful under all circumstances. Karr (1991) suggests to integrate aspects of those promising indicators to create a more robust approach to biological monitoring.

3.3. Importance of a Landscape Approach in Wetland Assessment

Traditional assessment procedures are species oriented, assume linear, causal relationships, are focused on individual or segmented processes and sites, and tend to base decisions on a static "snapshot" of the site in its present condition (Gosselink and Lee, 1986). In contrast, cumulative impact assessment requires a landscape orientation in which processes are highly interactive and often nonlinear. Although much has been said about the importance of cumulative impact assessment, there have been a few successful attempts to deal with it within a regulatory framework. Failure to work on appropriate delimited systems continues to be a frequent problem. Besides, cumulative impact assessment is constrained by lack of adequate understanding of the relationships between physical properties of wetlands and their functions.

A larger scale approach that integrates both spatial and temporal dynamics in wetland assessment is needed for several reasons (Bedford and Preston, 1988). First, wetland functions operate at watershed, landscape, regional, or larger scales; they are not primarily the product of particular wetlands but of their relationships to other wetlands, and other land use types; therefore, they need to be studied within these larger contexts. Second, the occurrence and maintenance of wetlands reflect large-scale and long-term characteristics of watersheds, landscapes, and regions, as well as more local processes (Winter, 1988; Siegel, 1988; O'Brien, 1988). Finally, the scale of environmental problems is expanding from the local to the regional and global level. Shifting the scale of impact assessment to larger scales requires that measurable properties or indicators of wetland and landscape condition be defined (Bedford and Preston, 1988). Refer to Gosselink and Lee (1986) and Bedford and Preston (1988) for an overview of cumulative impact assessment principles.

3.4. Evaluation of Wetland Functions and Values

A wide variety of wetland evaluation methods have been developed by federal or state agencies, private consulting firms, and the academic community to ascertain all or selected wetland functions and values (Adamus, 1983; Adamus and Stockwell, 1983; Smith, 1984; Hollands and McGee, 1985; Winchester, 1985; Wencek, 1985; Ammam et al., 1985; McColligan, Jr., 1985; Adamus, 1985). Most of these methods are designed to be fast and easily applied, even by non-specialists. They allow a quick functional evaluation of wetlands and often use a system of weight and scores to rank the wetlands for a variety of functional values (biological, hydrologic, socio-cultural). The Federal Highway Administration (FHWA) method for wetland functional assessment (Adamus and Stockwell, 1983) has been very popular, but still needed improvement and standardization. 25 wetland evaluation methodologies are summarized in Lonard and Clairain, Jr. (1985).

3.5. Conclusions and Research Needs

A number of wetland evaluation methods have been proposed, but they all still need improvement and standardization. The assessment of wetland health requires the analysis of indicators that are appropriate to each ecosystem; the evaluation should take into account the full set of parameters that best indicate the ecosystem condition. As the scale of environmental problems is expanding from the local to the global level, a more comprehensive approach in wetland assessment is fundamental. Further research is needed to: 1) understand the relationships between physical properties of wetlands and their functions; 2) find better health indexes that are sensitive to cumulative impacts; 3) develop standardized and accurate wetland evaluation procedures.

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4 • WETLAND EXTENT AND TYPE

4.1. Introduction

With the rapid disappearance of North American wetlands, there is a great need for long-term data documenting changes in wetland ecosystems. A knowledge of wetland dynamics should be an integral part of wetland management, but detailed studies of this topic are scarce in the scientific literature (Golet and Parkhurst, 1981). Many functional attributes of wetlands are related directly to their size (areal extent) and type. These two parameters are important indicators of wetland sustainability, defined as its resistance to changes in structure and function over long periods of time (Leibowitz et al., 1991). Documenting losses and gains in wetland area and monitoring changes in wetland types are critical for understanding regional trends and identifying the areas that need immediate attention.

4.2. Approach

Remote sensing is ideal for mapping wetlands as large areas can be surveyed relatively quickly without having to conduct extensive and expensive ground work in a difficult environment. The first step in surveying wetlands is to define them. Classification systems have generally been based on water conditions -- permanently flooded, temporarily flooded, saturated soils, etc. -- and vegetation cover -- herbaceous, shrub or tree -- (Cowardin et al., 1979). Once the definition and classification system have been determined, the most suitable remote sensing techniques and equipment can be chosen to meet the objectives of wetlands investigations. Although aerial photography remains the primary data source in many wetlands mapping and inventory programs, new remote sensors are likely to contribute to this effort in the future.

4.2.1 Aerial photography

Inventories of wetlands using primarily color infrared aerial photography began in the late 1960s as legislation protecting coastal wetlands was being formulated (Daiber, 1986). The aerial photography served primarily as a means of delineating plant communities and as a template from which the areal extent of each community could be determined. All investigators reported better tonal contrast for species discrimination with color infrared (CIR) photography. Best results could be obtained using a combination of color and CIR as color was better for detecting submerged vegetation, often an important ecological component of freshwater marshes.

The usual procedure in wetlands inventories is to compare aerial photography from different years and to construct maps that reflect the changes that have occurred (Hardisky and Klemas, 1983). Wetlands are identified on the photographs and classified primarily on the basis of water

regime, soil, and the life form of the dominant vegetation -- such as deep marsh, shallow marsh, meadow, shrub swamp etc. -- (Golet and Parkhurst, 1981). There is an overlap of 60% between consecutive photographs within a flightline to insure complete coverage and allow photography to be viewed in three dimensions using stereoscopic equipment (Hardisky and Klemas, 1983). Such stereo viewing is useful in identification of some plant species. Photointerpretation is generally based on tone, hue, texture, pattern and canopy height (Seher and Tueller, 1973; Howland, 1980). These parameters produce the most information on the color infrared imagery. Infrared wavelengths are superior for wetlands discrimination due to the relative high infrared reflectance of vegetation and the very low infrared reflectance of water and water-logged soils. After photointerpretation is completed, class and subclass boundaries for all wetlands are transferred to mylar topographic base maps, the size of each wetland and its component classes and subclasses is measured with a polar planimeter, and the amount of change calculated (Golet and Parkhurst, 1981). Small or subtle changes can be detected with the aid of the zoom transfer scope (Hardisky and Klemas, 1983). Variability in the characterization of wetlands by aerial photography can occur during the interpretation of the imagery by different observers. A standardized method of classification and ground control are required to produce accurate maps.

Interpretation of aerial imagery can be used in conjunction with a computer-based geographic information system (GIS). Sasser et al. (1986) developed a computerized data base to determine the amount, rate, and location of Louisiana's Barataria Basin marshes change over time. Data sets were assembled into a controlled data base, classified by the size and the spatial distribution (percentage) of water bodies within the marsh. Marshes were classified according to percentage of water. Output of the GIS was numerical and graphic (computer-generated gray maps and color images). The sequence and spatial patterns of water body development (indicative of marsh deterioration) were monitored to determine the phases and causal mechanisms of marsh loss.

4.2.2. Satellite imagery

Multispectral scanners have two main advantages over aerial photography when a high degree of resolution is not required; data can be obtained in a digital form and can be automatically classified by computer, thus speeding the interpretation process considerably and allowing direct quantitative treatment of the radiance data; large areas can be rapidly covered where photographic mapping is either cost prohibitive or too slow. The disadvantages are that the equipment is very expensive and resolution and accuracy of boundaries are much lower than with photography. Automatic classification techniques are based on either reflectance values or pattern recognition. The best discrimination of marsh vegetation occurs in the infrared bands.

A statewide computerized Landsat land cover and classification system has been developed in Florida in 1985 (Groce and Craig-Ayotte) to monitor changes and generate numeric acreage data, as well as color graphic displays of various wetland systems and other land use features. Computer tapes are obtained from the NASA for the areas needed and the time period desired. The computer provides 22 different classifications of land cover, including seven different classes of wetlands. It also gives acres of each classification, on a county by county basis. By obtaining the data for different time periods, it is possible to locate each wetland class and to determine

if there is a net gain or loss of each classification. This Landsat system has two major advantages: firstly, it provides its users with accurate, up-to-date information on the status and trends of land use within the state; secondly, land use planners have much more complete data to use in their planning processes.

As a result of the coarse resolution, multispectral (MSS) data have been generally supplemented with high-resolution aerial photography. Usually, the aerial photography provides a familiar data source for plant community identification, and the MSS data serve as a template for extrapolation to large areas. The combination of MSS data and aerial photography has been used to develop vegetation maps of forested wetlands, such as the Great Dismal Swamp (Garret and Carter, 1977; Carter et al., 1977). The upgraded multispectral scanner, the Thematic Mapper (TM), has a better ground resolution (30 m) and has three more spectral bands than the original MSS. This has improved species discrimination capabilities in wetland systems. The spectral and spatial improvements of the TM are likely to enhance our ability to discriminate species as well as to estimate percentage cover in wetlands. Digitized data from Landsat or SPOT constitute important data bases that may be used to provide temporal verification of selected wetland sites and updated information on land use.

4.3. Important Parameters in Remote Sensing of Wetlands

4.3.1 Scale

An important parameter in wetland investigations by remote sensing is image scale. Scale is generally related to the accuracy in determining the wetland boundary and the minimum size mapping unit obtained. Regulatory maps require as precise a location of wetland boundaries as possible, so large scale images are usually desirable. A high degree of accuracy is possible for tidal wetlands since they are generally large tracts of homogeneous vegetation communities that are closely related to tidal boundaries. Freshwater wetlands have proved to be more difficult to map. The majority of freshwater wetlands are relatively small, often less than an acre, with highly diverse, heterogeneous vegetation communities, making them difficult to discern and separate on all but low altitude images. A series of investigations conducted on the Georgia coast identified color infrared photography at scales ranging from 1:2,500 to 1:40,000 as the best photographic product for discrimination of salt marsh plant communities (Gallagher et al., 1972a, 1972b; Reimold et al., 1972). Brackish tidal marshes required larger-scale photography -- 1:5,000 -- (Gallagher and Reimold, 1973). Larger-scale imagery in brackish or salt marshes provided more vegetation detail, yielding better discrimination. However, the cost of photography rises with increases in scale. Small-scale (1:12,000 to 1:24,000) color infrared photography usually acquired in tandem with natural color photography became the standard for state wetland inventories (Daiber, 1986).

4.3.2 Time of the year and water level

The time of the year the imagery is taken is extremely important in investigating wetlands. The size of many wetland areas changes significantly with seasonal and annual variations in precipitation. More than one season is generally required unless the wetlands involved contain only emergent vegetation. In this case, imagery taken from late August through mid-October at the end of the growing season is sufficient. If the only objective is to delineate wetlands, regardless of vegetation cover, late winter-early spring imagery alone is sufficient. Wetlands are best delineated in this period as ground water is most likely to be at or above the surface. If wooded wetlands or a mixture of wetland types are to be surveyed, better accuracy can be achieved through multiseasonal analysis. Freshwater wetlands do not maintain the relatively constant boundaries of tidal wetlands from year to year or even from season to season; hydrologic characteristics and the resulting vegetation change through the year altering the appearance and reflective patterns of these wetlands on the images. The extent of forested wetlands can be difficult to assess through a leaf-covered canopy; thus early spring photography is usually best (Leibowitz et al., 1991). Water level must be taken into consideration in the analyses; if water level is too high, or too low, then the comparative interpretations are compromised. Submerged vegetation appears emergent, mudflats are extensive (or absent), etc. (R.E. Turner, personal communication).

4.4. Interpretation of Wetland Changes

Wetlands are dynamic ecosystems deriving much of their unique qualities from the fluctuating and oftentimes catastrophic environmental disturbances that mold them. These natural changes, however, are dwarfed by the potential manipulative power of humans (Hardisky and Klemas, 1983). Once comparative data are available, the factor(s) responsible for observed changes in wetland area and type will be identified whenever possible from the aerial photos and/or the satellite imagery or the site will be visited on the ground so that a positive determination as to the nature of the alteration could be made. Aerial photography is probably more reliable than satellite imagery for identifying causes of observed changes. Distinctions between major impacts, such as drought, manmade disturbances, and natural succession can be made (Golet and Parkhurst, 1981); but according to Keddy (1983), defining human-induced changes is difficult since there is a continuum of effects. Keddy gives examples to illustrate how numerous and subtle indirect human impacts could be. At one extreme, there are direct and obvious human-induced effects, such as drainage and infilling. The observed changes are rapid. and major changes in species composition occur. There are the more subtle human-induced effects such as rising atmospheric CO2 levels, acid rain, or low-level radioactive fallout. The resulting changes in wetlands might be slow. In between these extremes lie all other human impacts on wetlands. Keddy suggests that it is virtually impossible to recognize many human-induced effects, even with careful field observation (such as atmospheric CO2 or acid rain). Studies that attempt to recognize human-induced effects on ecosystems must consider indirect human-induced effects, but the difficulty is to define which human-induced effects to include and to recognize them in the field.

4.5. Conclusions

Remote sensing contributions to wetland management have essentially been in the areas of mapping and vegetal classification. A variety of remote sensing methods are available to locate and map wetlands. The choice of the method is dependent on project objectives plus time and monetary restraints. Infrared wavelengths appear to offer the greatest amount of information for discriminating and identifying wetlands whether interpretation is done manually or by computer. Large and medium scale color infrared photography is preferred where high resolution and accuracy are required. However, automatic techniques using digital data are constantly being improved and offer the advantages of speed and objectivity over human interpreters. Satellite imagery provides frequent coverage and is adequate for identifying landscape patterns of all kinds of ecosystems, but aerial photography can provide the resolution necessary to evaluate individual wetlands and the impacts that affect them. The advances in instrumentation and in computer analysis techniques will greatly enhance the types of data available for use by wetland managers. The imaging spectrometer is a major advance in remote sensing, since it allows to collect images in many narrow, contiguous spectral bands spanning the visible, near-infrared, and middle-infrared regions (Hardisky et al., 1986). As a result, a complete reflectance spectrum can be derived for each pixel in the image. Imaging spectrometry can reveal reflectance features unique to a particular vegetation type or environmental condition. Such features could either be hidden within a broad spectral band or, by residing outside of the wavelengths sensed, be missed entirely by MSS, TM or SPOT sensors (Hardisky et al.,1986). Our management effectiveness in the future will be directly related to our ability to collect and analyze data on a regional, ecosystem, and, eventually global scale.

4.6. References

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5 • LANDSCAPE AND WETLAND PATTERNS

5.1. Introduction

Landscape pattern or structure is now recognized as an important factor that affects ecological functions. The architectural qualities of structure and the arrangement or separateness of structural elements in the landscape particularly influence faunal diversity and abundance. The impact of separateness is related to dispersal ability of the species. The topic of habitat structure is poorly represented in historical reviews of ecology. Bell et al. (1991) discuss this concept and synthesize a number of ideas concerning habitat structure from different ecological perspectives. The importance of this parameter is well documented for birds (Roth, 1976; Keller et al., 1983; Robbins et al., 1989; Gosselink et al., 1990). The term "patchiness" has been commonly used to refer to habitat heterogeneity. It is the spatial variability within a habitat of any resource or feature critical to a taxon's existence in that habitat (Roth, 1976). Numerous indicators of spatial patterns have been suggested (see O'Neill et al., 1988 and Turner, 1989 for reviews), although relatively few have received rigorous empirical scrutiny. Some indicators describe landscape heterogeneity as a function of patch characteristics; others emphasize the arrangement of patches. In all instances, the choice of scale is critical to the measurement and interpretation of pattern indicators (Leibowitz et al., 1991).

5.2. Importance of Scale in Pattern Analysis

Because landscapes are spatially heterogeneous areas, their structure, function and change are themselves scale-dependent (Turner, 1989). The measurement of spatial pattern and heterogeneity is dependent upon the scale at which the measurements are made. Gardner et al. (1987) demonstrated that the number, sizes, and shapes of patches in a landscape were dependent on the linear dimension of the map. Observations of landscape function, such as the flow of organisms, also depend on scale. The scale at which humans perceive boundaries and patches in the landscape may have little relevance for numerous flows and fluxes (Turner, 1989). If we are interested in a particular organism, we are unlikely to discern the important elements of patch structure or dynamics, unless we adopt an organism-centered view of the environment (Whittaker, 1975). We must possess sufficient information about the organism's behavior to understand its responses to structure, that is how it perceives and uses structure. But despite such data on behavioral traits, it is not always obvious exactly at what spatial scale a particular interaction should be judged (Bell et al., 1991). Finally, changes in landscape structure or function are scale-dependent. A dynamic landscape may exhibit a stable mosaic at one spatial scale but not at another (Turner, 1989).

The scale at which studies are conducted may profoundly influence the conclusions; processes and parameters important at one scale may not be as important at another scale. Thus, conclusions regarding landscape patterns and processes must be drawn with an acute awareness of scale (Turner, 1989).

5.3. Quantifying Landscape Structure

Landscape structure must be identified and quantified in meaningful ways in order to understand the interactions between landscape patterns and ecological processes. The spatial patterns observed in a landscape result from complex interactions between physical, biological, and social forces (Turner, 1989). Quantitative methods are required to compare different landscapes, identify significant changes through time, and relate landscape patterns to ecological function. Considerable progress in analyzing and interpreting changes in landscape structure has been made (for detailed methods and applications, see Turner and Garner, 1990; statistical approaches are reviewed in Turner et al., 1990).

5.3.1 Important aspects of landscape structure

Most landscapes are a mixture of natural and human-managed patches that vary in size, shape, and arrangement (Forman and Godron, 1981; Krummel et al., 1987; Turner and Ruscher, 1988). The size and distribution of patches in the landscape may be of particular importance for species that require habitat patches of a minimum size or specific arrangement (Turner, 1989), and thus are critical to the maintenance of biodiversity. Franklin and Forman (1987) analyzed the potential effects of changes in patch structure on the persistence of interior and edge species. Patch size and arrangement may also reflect environmental factors, such as topography or soil type (Sharpe et al., 1987). The amount of edge between different landscape elements is another parameter of critical importance for the movement of certain organisms or materials across boundaries (Hansen et al., 1988; McCoy et al., 1986; Turner and Bratton, 1987; Wiens et al., 1985); and the importance of edge habitat for various species is well know. Thus, it may be essential to monitor changes in edges when one quantifies spatial patterns and integrates pattern with function (Turner, 1989). It is important to distinguish between edges formed along expected environmental gradients, such as bands of vegetation along a moisture gradient, versus edges considered to be a negative result of human activities -- utility corridors, agricultural fields, etc. -- (Leibowitz et al., 1991).

5.3.2 Spatial pattern indices

Three complementary pattern indices -- dominance, contagion, and fractal dimension -- allow the discrimination of major landscape types, such as urban coastal landscapes, mountain forests, and agricultural areas (O'Neill et al., 1988). They appear to provide information at different scales, with the fractal dimension and dominance indices reflecting gross features of landscape pattern, and the contagion index reflecting the fine-scale attributes of the landscape. This type of scale sensitivity might be useful in selecting measures of pattern that can be easily monitored through time (e.g., by remote sensing) and related to different processes (Turner, 1989). It is important

to note that the value of any measurement is a function of how the landscape units were classified (e.g., land use categories vs. successional stages) and of the spatial scale of the analysis (e.g., grain and extent). "Grain" refers to the level of spatial or temporal resolution within a data set, and "extent" refers to the area of study (Turner, 1989).

Dominance index. The dominance index measures the extent to which one or a few land uses dominate the landscape (O'Neill et al., 1988):

$$\textbf{Dominance} = \ln(n) + \sum_{i=1}^{m} P_i \ln(P_i)$$

where Pi is the proportion of the landscape in land use i, and n is the total number of land use categories in a particular scene. The term, ln(n), represents the maximum diversity with all land use types present in equal proportions. Large values of the index indicate that the landscape is dominated by one or a few land uses. Small values suggest that many land use types are present in approximately equal proportions.

Contagion index. The contagion index is a measure of the probability of patches to be adjacent to each other (O'Neill et al., 1988):

$$\textbf{Contagion} = 2n\ln(n) + \sum_{i=1}^{m} \sum_{j=1}^{m} P_{ij} \ln(P_{ij}).$$

where Pij is the probability of a grid point of land use i being adjacent to a grid point of land use j. The term, 2n ln(n), represents a maximum in which all adjacency probabilities are equal. High values of contagion are obtained when large, contiguous patches are found on the landscape; whereas, low values indicate that the landscape is fragmented into many small patches (O'Neill et al., 1988).

Fractal dimension index. Fractal geometry (Mandelbrot 1983; Bell et al., 1991) has been introduced as a method to study shapes that are partially correlated over many scales. It has been used to compare simulated and actual landscapes (Gardner et al., 1987; Turner, 1987), and to compare the geometry of different landscapes (Krummel et al., 1987; Milne, 1988; O'Neill et al., 1988; Turner and Ruscher, 1988). The fractal dimension is an index of the complexity of shapes on the landscape. It is estimated by regressing polygon area against perimeter for each patch on a digitized map. The fractal dimension is related to the slope of the regression, S, by the relationship (Lovejoy, 1982):

$$\text{Fractal dimension} = 2\ S$$

If the landscape is composed of simple geometric shapes like squares and rectangles, the fractal dimension will be small, approaching 1.0 (Krummel et al., 1987). If the landscape contains many patches with complex and convoluted shapes, the fractal dimension will be large. Krummel et al. (1987) suggest that human-influenced landscapes exhibit simpler patterns than natural ones,

as measured by the fractal dimension. An increase or decrease in the fractal dimension through time indicates the degree to which human activities disturb and simplify the landscape patterns, regardless of the specific land uses (O'Neill et al., 1988).

5.3.3 Keller method

Keller (1985) developed a method for the quantification of edge and the spatial arrangement of habitat. His aim was to identify the characteristics of patches associated with particular groups of birds. A grid of hexagonal cells is superimposed over the imagery of the studied site (air photos or other remotely sensed imagery), and each cell is classified as to habitat component type. Then, the cells are numbered on a cartesian (X,Y) coordinate system so that each cell can be located in two-dimensional space in relation to every other cell (spatial arrangement). In order to apply this technique, it is important to:

1) identify the species of interest;
2) use a habitat classification appropriate to the taxonomic group of interest;
3) choose a biologically meaningful grid-cell size (i.e., equivalent to the smallest resolvable habitat component of interest). Then, we can abstract the remotely sensed habitat information to the grid and store the data base on a computer for later analyses. A series of algorithms that provide a variety of measures of horizontal habitat structure is available. The collective name of this system is SPADIST (Spatial Distribution) (Keller, 1979, 1980, 1985).

5.4. Relating Landscape Patterns and Ecological Processes

Identifying the relationship between landscape pattern and ecological processes is a difficult goal for landscape research. The broad spatial-temporal scales involved make experimentation more challenging, and we may have to extrapolate the results obtained from small-scale experiments to broad-scales (Turner, 1989).

5.4.1 Landscape heterogeneity and disturbance

The spread of disturbance across a landscape is an important ecological process that is influenced by spatial heterogeneity (Risser et al., 1984; Romme, 1982; Turner 1987). Disturbance can be defined as "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (Pickett and White, 1985). Disturbances operate in a heterogeneous manner in the landscape; gradients of frequency, severity, and type are often controlled by physical and vegetational features. The differential exposure to disturbance, in concert with previous history and edaphic conditions, leads to the vegetation mosaic observed in the landscape. The spatial spread of the disturbance may be enhanced or retarded by landscape heterogeneity, depending on its mode of propagation. If the disturbance is likely to propagate within a community, high landscape heterogeneity should retard its spread. If the disturbance is likely to move between communities,

increased landscape heterogeneity should enhance it (Turner, 1989). The relationship between landscape pattern and disturbance regimes must be studied further, particularly in light of potential global climatic change (Turner, 1989). Disturbances operate at many scales simultaneously, but their interactive effects are not well known, partly because we often tend to study single disturbances in small areas rather than multiple disturbances in whole landscapes.

5.4.2 Movement and persistence of organisms

Landscape connectivity may be quite important for species persistence. Modifications of habitat connectivity can have strong influences on species abundance and movement patterns. The size, shape, and diversity of patches also influence species abundance. Woodlot size was found to be the best single predictor of bird species richness in the Netherlands (Van Dorp and Opdam, 1987). Theoretical approaches are being developed to identify scale-dependent patterns of resource utilization by organisms on a landscape (Turner, 1989). The interaction between dispersal processes and landscape pattern influences the temporal dynamics of populations. Results suggested that if an organism disperses along corridors, then the spatial relationships between habitat patches are important. If, however, the organism disperses large distances in random directions from patches, then the spatial arrangement of habitat patches will have less effect on population dynamics (Turner, 1989).

5.4.3 Redistribution of matter and nutrients

Few studies have addressed the influence that spatial pattern may have upon the flow of matter and nutrients, although there is increasing recognition that such an influence is important (Gosz, 1986).

5.5. Conclusions and Research Needs

Spatial pattern has been shown to influence many ecologically important processes. Therefore, the effects of pattern on process must be considered in future ecological studies, particularly at broad scales, and in resource management decisions. Habitat fragmentation may progress with little effect on a population until the critical pathways of connectivity are disrupted; then, a slight change near a critical threshold can have dramatic consequences for the persistence of the population. Methods for characterizing landscape structure and predicting changes are now available. Current research suggests that different landscape indexes may reflect processes operating at different scales. The broad-scale indexes of landscape structure may provide an appropriate metric for monitoring regional ecological changes. Such an application is of particular importance because changes in broad-scale patterns can be measured with remote sensing technology (Turner, 1989).

Important questions remain about landscape patterns and their changes (Turner, 1989). What constitutes a significant change in landscape structure? Which measures best relate to ecological processes? How do the measurements of pattern relate to the scale of the underlying processes?

Which measures of structure give the best indications of landscape change; that is can any serve as "early warning" signals? Answers to these and other questions are necessary for the development of broad-scale experiments and for the design of strategies to monitor landscape responses to global change.

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6 • WETLAND BIOMASS AND PRODUCTIVITY

6.1. Introduction

The most commonly used and accepted parameter for evaluating an ecosystem condition is biomass and/or net primary productivity of the emergent macrophytes. Both terms refer to the dry weight of plants (expressed as grams dry weight per square meter -- gdw/m2 -- for the biomass and usually as gdw/m2 per year for the productivity). Biomass represents the amount of fixed energy available to consumer organisms. Net aerial primary productivity (NAPP) is defined by Odum (1983) as the rate of storage of organic matter in aboveground plant tissues exceeding the respiratory use by the plants during the period of measurement. Changes in live and dead biomass over the growing season can be combined in a variety of ways to yield an estimate of NAPP.

Rapid changes in biomass and primary production are the signs of illness or profound disease in the ecosystem (Schaeffer et al., 1988). Change can include both accumulation and loss of biomass. The accumulation of biomass reflects the inability of the ecosystem to reach a new stable state. Ecosystems can be viewed as systems which maintain their structural integrity by degrading energy while avoiding entropy. If the amount of energy available changes, the system changes. The onset of illness occurs when subtle shifts in production occur; profound disease is indicated when energy is lost from the ecosystem in an uncontrollable manner (Schaeffer et al., 1988).

The high productivity of wetlands makes them an important part of local nutrient cycles, such that biogeochemical cycling in wetlands is intricately linked to vegetation biomass production. (Gross et al., 1990). In recent years, the role of soil microorganisms in wetland nutrient cycles has been an extensively researched topic. Soil microbes feed on senescing roots and their exudates, producing reduced gases, such as hydrogen sulfide and methane, that are important in biogeochemical cycles. The amount of plant biomass can serve as an indicator of potential production of these microbially-generated gases. Methane, is involved in the greenhouse effect, the unnatural warming of the earth's atmosphere, caused by the buildup of certain gases.

6.2. Traditional Biomass/Productivity Assessment Techniques

6.2.1 Destructive harvesting technique

The traditional method of measuring aboveground biomass is to clip the plants at the soil surface, sort the materials into live and dead components and dry them in an oven, to obtain a constant dry weight. This method is both destructive and time consuming. Determining belowground

biomass (root) by traditional techniques is even more tedious and damaging. A stainless steel or plastic tube is used to extract a cylinder of the soil; the soil is washed away and the live and dead roots sorted before the materials are dried in an oven and weighed. Each time a core is removed, a hole is left in the marsh.

Because of the severe damage to vegetation and soil, these techniques cannot be repeated in one area longer than a year, making year-to-year biomass comparisons for the same site impossible. Such comparisons are needed in order to detect changes and determine the sensitivity of wetlands to phenomena such as pollution, drought or manmade disturbances (e.g. highway construction).

6.2.2 Tagging technique

Another technique of monitoring plant production is to tag individual stalks of marsh plants to determine changes in live and dead tissues. Each site is visited periodically and each live and dead culm is measured for height, and the number of live and dead leaves counted. The differences in the measurements between successive intervals are analyzed to determine leaf production, senescence and abcission which approximates the rate of tissue transition from the live to the dead component. Recruitment is assessed by counting and measuring all new sprouts too short to be tagged. Culms which died during a sampling period are used as an estimate of mortality. This method estimates live culm production in the form of culm mortality as it passes to the dead component. A reasonable estimation of net aerial primary productivity (NAPP) can be determined by deriving a simple model equating live culm height and dry weight biomass, and estimating leaf abcission, culm mortality and a mean weight for dead leaves. For more details on the destructive and tagging procedures, refer to Hardisky 1980.

The frequent inaccessibility of wetlands makes biomass assessment by traditional harvesting or tagging techniques difficult. In addition, these techniques are tedious and very labor-intensive, making them impractical for studying large areas.

6.3. Remote Sensing Technique

Remote sensing is a rapid and non-destructive technique that allows repetitive studies over time. One of the instruments commonly used, a hand-held radiometer, is very easy to work with and saves a lot of time compared to a typical harvest study. With this more efficient method, more stations can be sampled in less time. Although it is useful for assessing biomass over small areas, measurements for entire marshes are more practical if satellite imagery is used.

6.3.1 Spectral properties of plants

In order to understand the principles of remote sensing techniques used to monitor wetland health, let us examine some spectral properties of plants. Early reports describing the interaction of leaf tissue with light indicated that changes in the spectral quality of reflected electromagnetic radiation were directly related to the quantity of leaf tissue and pigment concentrations (Allen

and Richardson 1968, Colwell 1974, Gausman 1974). As green aboveground biomass increased, the most significant of the spectral changes were a decrease in red radiation resulting from strong absorption by the chlorophylls, and an increase in near infrared radiation resulting from intra- and interleaf scattering. Although these findings were made in non-wetland environments, wetland plants show the same patterns (Bartlett 1976, 1979, Bartlett and Klemas 1981, Drake 1976, Jensen 1980).

6.3.2 Reflectance and aboveground biomass

Early work by Bartlett (1976, 1979) determined green biomass of wetland grasses to be strongly correlated with the near infrared/ red reflectance ratio. Other investigators found good correlations between green biomass and the spectral reflectance of different marsh shrub communities (Hardisky et al., 1986). In most cases, the combination of red and near infrared radiance provided the best correlation with canopy biomass.

Several years ago, a simple linear regression model equating spectral reflectance to biomass was formulated for Delaware *Spartina alterniflora*, one of the most common salt marsh plants in eastern North America (Hardisky 1982, 1984). Spectral reflectance measurements were made in selected portions of the marsh using a hand-held radiometer that gathered data in three wave-bands, spectrally configured to simulate bands 3, 4, and 5 of the Landsat Thematic Mapper: a red band (630-690 nm, TM3) sensitive to chlorophyll concentration, a near infrared (NIR) band (760-900 nm, TM4), sensitive to plant tissue structure, and a middle infrared band (1550-1750 nm, TM5) sensitive to water absorption.

The raw radiance data were transformed and expressed as a normalized difference of two bands as follows:

$$VI = [NIR - red] / [NIR + red]$$

$$II = [NIR - middle\ IR] / [NIR + middle\ IR]$$

where VI is the Vegetation Index and II the Infrared Index. Index values were preferred to raw radiance data because the normalization procedure tended to compensate for both short- and long-term changes in solar irradiance and atmospheric conditions (Tucker et al., 1981). Both indices, VI and II, correlated strongly with the changes in biomass (Hardisky et al., 1983; Hardisky et al., 1986). Measurements of biomass were combined over a growing season to yield an estimate of NAPP. The NAPP estimates were generally within ten percent of harvest estimates (Hardisky et al., 1984).

Biomass evaluation has also been attempted with satellite imagery (Gross et al., 1987). The problem of atmospheric effects on the satellite-measured radiance data was solved by converting the satellite data to the equivalent ground-measurement reflectance. This was done using equations relating the reflectance of certain large, homogeneous sites measured from the ground at the time of the satellite overpass, to their satellite-measured radiance. The satellite-derived

estimates were found to be within 13 percent of ground-based biomass estimates. The nature of the relationship linking VI and the aerial biomass was consistent from year to year and between marshes, although there was a difference between northern and southern marshes (Gross et al., 1990).

6.3.3 Belowground biomass estimation

Light does not penetrate soil, making it impossible to measure root biomass directly by optical remote sensing. However, Gross et al. (1990) report a strong positive relationship ($r^2 = 0.86$) between the natural logarithm of live aboveground biomass and the natural logarithm of live belowground biomass of *S. alterniflora* ("short" plants only). Therefore, belowground biomass can be indirectly measured using a non-destructive method.

Another promising technique for belowground biomass estimation is the use of ground-penetrating radar (GPR), but it is still under evaluation. Traditionally, GPR has been used to locate things such as archaeological sites, toxic waste drums, and divisions between contrasting soil types like sand and clay. A radar antenna is dragged along the surface of the marsh, emitting electromagnetic waves. These waves penetrate the soil, and are reflected back by objects in the soil. The return signal is recorded and printed in graph form (Gross, 1989). Researchers hope the characteristics of the return signal will reveal something about root material.

6.3.4 Factors influencing spectral estimates

One of the factors that influence the spectral radiance of the marsh is the solar angle which can easily be corrected (Hardisky et al., 1986). Two other factors are the quantity and orientation of dead biomass and the amount of soil reflectance. The presence of dead material tends to decrease the vegetation index VI. Except in marshes with very sparse canopy (<30% cover), soil reflectance is not usually a problem. Richardson and Wiegand (1977) have proposed a perpendicular vegetation index (PVI), which factors out the influence of soil reflectance. The infrared index is less attenuated by dead biomass and soil reflectance than the vegetation index.

6.4. Conclusions and Research Needs

Remote sensing is considered an accurate and effective non-destructive biomass assessment technique in salt marshes despite its limitations: sampling can only be done on sunny days, for four hours, and only during a tidal stage when the marsh surface is not flooded (Hardisky et al., 1984). Hand-held radiometers have been extensively used to assess biomass and NAPP of small wetland tracts, but satellite imagery is more useful for sampling larger areas. The aerial biomass estimation technique is based on the use of simple regression models equating the green biomass with spectral radiance indices. Root biomass can then be estimated using equations linking aboveground and belowground biomass.

Limited remote sensing work has been conducted in other types of wetlands such as brackish marshes and coastal mangrove systems (Hardisky et al., 1986). Salt marshes are often characterized by large monospecific stands of vegetation. In contrast, the physionomy of brackish marshes is usually more varied because a particular plant community often comprises many species. Different plant morphologies thus coalesce to produce canopy architectures that reflect incident radiation differently than monospecific canopy (Hardisky et al., 1986). Hardisky and Klemas (1985) analyzed the effects of the three canopy types on the vegetation index. Since the quality of reflected radiation (expressed as a vegetation index) differs for each canopy architecture, accurate biomass predictions must rely on separate models describing each type. Studies by Hardisky (1984) suggested that biomass could indeed be predicted for communities of one canopy type using a single model. The work conducted by Hardisky et al. (1986) in the black mangrove, Avicennia germinans, in Costa Rica, describes a positive relationship ($r = 0.79$) between the TM vegetation index and live leaf biomass. The more ubiquitous taller mangrove forms will require extensive ground comparisons before an operational biomass estimation procedure can be developed (Hardisky et al., 1986).

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7 • WETLAND VEGETATION

7.1. Introduction

Wetlands are primarily described and classified according to their vegetation (Cowardin et al., 1979). The vegetation is intimately associated with the hydrologic and edaphic characteristics of the ecosystem (D'Avanzo, 1986), so it plays a crucial role in the various wetland processes (Nickerson et al., 1989). Both the community level and the organism level should be considered in the vegetation assessment.

Vegetation community composition and abundance are important parameters in wetland health assessment because they reflect habitat suitability for wildlife and fisheries, ecological productivity, water chemistry, and landscape aesthetics (Brooks and Hughes, 1986). Plant communities and their characteristics have been extensively studied and sampling methods are well developed (e.g., Frederickson and Reid, 1988). Because of their immobility, plants are reliable indicators of certain types of stress, such as changes in hydrology and nutrient/pollutant loadings (Leibowitz and Brown, 1990). The composition and density of herbaceous communities and the forest understory will respond readily to short-term impacts; whereas, trees and shrubs are better indicators of long-term disturbance. Changes in the community composition or density should coincide with the coarse spatial pattern indicators (Leibowitz et al., 1991).

At the organism level, eco-physiological indicators of plant stress have shown promise in environmental impact assessment: proline concentration in the leaf, root alcohol dehydrogenase activity, activities of nitrogen metabolizing enzymes, and adenylate compounds (Mendelssohn and McKee, in press). These indicators respond, respectively, to salt stress, soil oxygen and nitrogen deficiencies, and to cumulative impacts. Physiological stress can be detected by remote sensing techniques, even before visible symptoms appear in the field.

7.2. Vegetation Assessment Techniques

7.2.1 Community level

Plant communities are commonly described by their floristics (species list), vertical structure (life form, layers), and horizontal arrangement (coverage, density) (Leibowitz et al., 1991).

<u>Sampling methods</u>. The sampling methods outlined below are essentially those suggested by Leibowitz et al. (1991) for their wetland monitoring program.

Permanent transects are established in each major plant community of the wetland, allowing change detection over time. Aerial photography can be used to identify and delineate these major vegetation types. Transects are usually oriented parallel to certain ecological gradients within each community. The number and length of transects will depend on the shape, orientation, hydrologic gradients, and interspersion of plant communities. Sampling points are marked on the ground with iron rods or other suitable markers and located on the map. Different kinds of sampling methods for study site selection are discussed in Shimwell (1971): subjective samples, partially random samples, regularly spaced samples in a checkerboard arrangement, contiguous study sites in a belt sample arrangement etc. The size and number of sites to select can be based on the added species approach (Whittaker, 1975).

For herbaceous vegetation, 1 m^2 square plots or rectangular (2:1) quadrants are the standard, although microplots also may be used (Federal Interagency Committee for Wetland Delineation, 1989). Herbaceous communities must be sampled during the growing season. A multiseasonal analysis is recommended to account for non-persistent species (Brooks and Hughes, 1986). For shrubs, saplings, and vines, sampling should be done using a circular plot with a 9-meter (30-foot) radius centered on the transect as recommended in the Federal Wetland Delineation manual (Federal Interagency Committee for Wetland Delineation, 1989). This allows some standardization in the method throughout the United States. Coverage for each plant or multistemmed clump is estimated by measuring the diameter of the maximum extent of foliage and assuming a circular outline. Trees may be sampled by determining their basal area with a prism or angle gauge, within the 9-meter circular plot and beyond its perimeter (FICWD, 1989). The basal area factor appropriate for each region or forest type is used. Sampling for shrubs and trees may be done in the dormant season, but the growing season is preferred. Alternative sampling methods used include nested 2:1 rectangles for herbs (1 m^2), shrubs (10 m^2), and trees (100 m^2) (Brower and Zar, 1984; Brooks and Hughes, 1986); 20 x 50 cm microplots for herbs, a 2:1 rectangle of 50 m^2 for riparian shrubs, and 375 or 500 m^2 plot for trees in the western United States (Platts et al., 1987).

The total area occupied by each vegetation type can be estimated by planimetry (Roman et al., 1984). Their dominant and associated plant species are identified, and percent cover of each species estimated. Plant identification is usually done in the field. For more consistency nationwide, it is recommended to follow the National Wetland Inventory plant nomenclature (Reed, 1988). Photographs can be taken to document the general appearance and extent of the vegetation.

Other characteristics that could be useful in vegetation analysis include (Leibowitz et al., 1991): the occurrence of wetland indicator species, the ratio of wetland obligate species to facultative species, the occurrence of species considered intolerant or tolerant of wetland stressors of concern, the ratio of exotic species to native species, the occurrence of competitive species such as Phragmites australis, indicators of a disturbation in the marsh (Roman et al., 1984), vegetation height, and vegetation age.

Measures of community composition. Five measures of community composition are commonly used to compare vegetation communities (Nickerson et al., 1989): number of species, and total stem count per plot, calculated directly from the raw data; plant diversity, species richness, and species evenness, that derive from the two first measures. Plant diversity is based on the heterogeneity index of Shannon and Weaver (1949) and is calculated using the following equation:

$$H = \sum_{i=1}^{s} \left[(n_i/N)\ln(n_i/N)\right]$$

where H is the general diversity, n_i is the stem count per species per plot, and N the total stem count per plot. General diversity is influenced by both the total number of the species and their distribution (Krebs, 1978). To distinguish between the contributions of these two factors, species richness and species evenness are also calculated.

Species richness, r, is computed as follows (Margalef, 1957):

$$r = (S - 1)/\ln(N)$$

where S is the total number of species in a plot.

Species evenness, e, (developed by Shannon and Weaver, 1949) is calculated as follows:

$$e = H/\ln(S)$$

Nickerson and Thibodeau (1984) and Thibodeau and Nickerson (1986) discuss in more detail the use of these standard ecological descriptive measures.

7.2.2 Individual level

Salinity, soil anaerobiosis, and nitrogen deficiency are environmental factors that affect plants vigor and productivity. Some indicators of these types of stress have been tested (Mendelssohn and McKee, in press): leaf proline concentration, root alcohol dehydrogenase activity, nitrogen metabolizing enzymes activities, and adenylate energy charge ratio and related adenine nucleotide concentrations. The first indicators are "stressor-specific", whereas the adenylate parameters are considered "integrative" that respond to the cumulative impact of all stressors to which vegetation is exposed. Stress-related physiological changes that occur in the plant result in changes in their reflectance properties; thus, remote sensing allows early detection of plant disturbance.

Proline concentration: an indicator of salt stress. Salinity is a major factor that can affect species composition and productivity of coastal wetland vegetation (Mitsch and Gosselink, 1986). Many species of plants accumulate specific low molecular weight solutes in response to elevated salinities (Jeffries et al., 1979). Some species (e.g., salt marsh plants) will accumulate amino acids such as proline, while others will accumulate carbohydrates such as sugars and polyols

(Flowers et al., 1977; Jeffries, 1981). These compounds are considered to have an osmoregulatory function: they act as compatible solutes to reduce plant osmotic stress due to the increased salinity (Mendelssohn and McKee, in press). The concentration of proline in salt marsh plants increases significantly as the salt concentration increases above a threshold level, which is species specific (Cavalieri and Huang, 1979; Mendelssohn and McKee, 1987b). The greater the salt tolerance of the plant, the higher the threshold level at which there is a proline increase (Jain et al., 1987). Leaf proline concentration seems to be a good salt stress indicator for the plants that accumulate this compound. The proline analysis technique is described by Bates et al. (1973).

Alcohol dehydrogenase: an indicator of soil anaerobiosis. Wetland plants are often subjected to increased submergence due to sea level rise or to man-induced changes in hydrology. Excessive soil and plant submergence may result in root oxygen deficiencies, even in relatively flood-tolerant species, affecting their growth (Kozlowski, 1984). When soil oxygen is lacking, the enzyme alcohol dehydrogenase (ADH) becomes very active, catalyzing the reduction of acetaldehyde to ethanol. ADH activity has been used extensively in the literature as an index of waterlogging response in vegetation (Mendelssohn and McKee, 1987a; Smith et al., 1986; Mendelssohn and Burdick, 1987). However, this indicator is limited to those wetland conditions where sulfide and possibly other unidentified soil toxins do not accumulate to concentrations that inhibit this enzyme (Mendelssohn and McKee, in press).

Nitrogen metabolizing enzymes: indicators of nitrogen deficiency. Nitrogen is a determining factor in the growth of salt marsh vegetation (Mendelssohn et al., 1982; Howes et al., 1986) and is often the primary growth-limiting factor for emergent wetland macrophytes in general (Mendelssohn and McKee, in press). Two enzymes, glutamate dehydrogenase (GDH) and glutamate synthetase (GS), which catalyze the initial incorporation of ammonium into amino acids, have been used to measure nitrogen metabolism and quantify nitrogen deficiencies (Mendelssohn and McKee, in press). GS is considered the primary nitrogen assimilatory enzyme in plants (Rhodes et al., 1976; Miflin and Lea, 1976). Mendelssohn (1979) used GDH activity to show differences in nitrogen utilization among the height forms of Spartina alterniflora in a North Carolina marsh.

Adenylate energy charge ratio: an integrative stress indicator. The adenylate energy charge ratio (AEC, Bromsel and Pradet, 1968) is considered to provide an integrated measure of the physiological "health" of a particular community, population, or organism (Stewart and Guinn, 1969; Jones, 1979; Wiebe and Bankcroft, 1975; Ivanovici, 1979; Mendelssohn and McKee, 1981; McKee and Mendelssohn, 1984; Sklar and McKee, 1984; Mendelssohn and McKee, 1985). An organism maintains a particular cellular ratio of ATP (adenosine triphosphate), which depends on the organism's physiological vigor and state of growth (Mendelssohn and McKee, in press). AEC ratio (ATP + 0.5 ADP)/(ATP + ADP + AMP) is a measure of the "energy rich" adenylate compounds present in the organism. Adenylate analyses are described by Mendelssohn and McKee (1981). AEC index is superior to growth measurements, because it is more sensitive to subtle environmental alterations and thus can indicate plant stress before symptoms are visually apparent. The adenylate parameters are presently being compared with other integrative stress

indicators (e.g., photosynthesis, leaf reflectance and fluorescence) to determine which are most efficient in quantifying plant stress (Mendelssohn and McKee, in press). Seasonal fluctuations of AEC and adenylate concentrations must be considered when evaluating the effect of a natural or man-induced stressor in the field (Mendelssohn and McKee, in press).

Use of remote sensing for plant stress detection. When plants are subjected to stressful conditions, certain physiological changes occur, that can be detected by remote sensing, because of the consequent changes in plants' spectral reflectance. These physiological changes relate to chlorophyll density, cellular size and arrangement, and moisture content. The figure below shows the typical reflectance curve of a leaf. The low reflectance in the blue and red regions is due to strong absorbance of these wavelengths by chlorophyll. There is a slight peak in the green region, because plants do not absorb green, but reflect it. The high reflectance in the near infrared region is controlled by plant tissue structure and results from the scattering effects of the mesophyll (Boyer et al., 1988). Beyond 1200 nm, the decrease in infrared absorption is due to the absorption by water (Knipling, 1969).

As a plant is exposed to various stressful conditions (disease, insects, moisture and mineral stress, etc.), two changes in reflectance are observed: 1) visible reflectance increases because there is less chlorophyll and/or the chlorophyll is less efficient in absorbing red and blue light; and 2) near infrared reflectance decreases because of a deterioration of the mesophyll cells (Campbell, 1987). Reflectance changes can be detected before visible symptoms appear, and thus, are good indicators of plants stress (Knipling, 1969). Moisture stress is usually evidenced by an increased radiant emission from the plant and thus lighter tones in images (Weaver et al., 1968). Nitrogen deficiency will result in increased reflectivity of a single leaf, but in decreased reflectivity of the whole canopy, because of the decrease of leaf surface area per unit ground area (Hardisky, 1984).

7.3. Uncertainties Associated With the Assessment of Vegetation

Ecosystem are dynamic and in a constant state of change. Consequently, data representing a single year reflect conditions only for that year and can be misleading if compared to any other year (Treshow and Allan, 1985). In addition to this temporal variation, considerable spatial variation exists in species composition and cover. Treshow and Allan (1985) discuss the uncertainties associated with vegetation assessment techniques.

7.3.1 Sources of uncertainties

Considerable spatial variation exists in the relative abundance of the dominant species, as well as vegetal cover, depending on the aspect, elevation, and latitude (Tueller et al., 1979). On a more local basis, the patchiness, that is the biotic heterogeneity of the plant cover, is governed by both biotic and abiotic disturbance, climatic and soil effects, the reproductive patterns of the vegetation, and effects such as grazing preferences, seed dispersal, and nutrient deposition (Piemeisel, 1951; Wells, 1960; West, 1969; Wiens, 1976).

The temporal variation in a plant community is even more critical than the spatial variation. The year the study is conducted may probably not be representative over time, because of climatic fluctuations. Data necessary to incorporate this variability are rarely obtained and little is known about the probability that baseline conditions established one year will remain the same in subsequent years, even in the absence of perturbation (Treshow and Allan, 1985).

In addition to these sources of uncertainties, "noise" can be caused by chance distribution and establishment of individuals, by statistical limitations of finite samples, and from limitations in estimating the species data (Gauch, 1980 and 1982).

7.3.2 Minimizing the uncertainties

To compensate for the spatial heterogeneity, ample sampling is required. Sampling methods should portray a realistic and accurate picture of the community composition. Teshow and Allan (1985) suggest a random sampling plan coupled with a stratification based on proportional representation of the major components of the community.

Yearly variation can be minimized by adequate sampling time frames of at least two years (Treshow and Allan, 1985). Another approach is considered in which the similarity of the plant cover among the study sites is examined over time. A similarity index (SI) compares the variation of each stand every year (Treshow and Allan, 1985). The minimum and maximum frequency of occurrence and percent cover of each species in each stand is compared using an equation modified from Ruzicka (1958):

$$SI = \sum \text{minimum value} / \sum \text{maximum value} \times 100$$

The similarity is analyzed by a weighted pair group method (Sneath and Sokal, 1973). The plots are arranged according to their highest similarities. The ordered plots are then graphed into dendograms.

Similarity matrices serve to delineate plant communities that are floristically distinctive and to determine those which are most similar. Selection of stands with some initial similarity would allow the use of fewer replicated stands and reduce the uncertainty due to the differences among them. This could be done by placing the highest value on the sites that are closest to the center of the gradient. Although the use of similarity indexes reduces the yearly variation, it does not eliminate it (Treshow and Allan, 1985).

7.4. Some Observations About Recovery Rates of Wetlands

The recovery rate of an ecosystem is a consequence of its productivity, regrowth ability of climax species, and harshness of the environment (reviewed in Thorhaug, 1980). Results of a ten-year analysis (Nickerson et al., 1989) demonstrate that once disturbed, different wetlands display distinctive recovery rates. While some wetland ecotypes recuperate within one year (e.g., cattail

marsh), other areas (e.g., bogs) are much more sensitive to perturbation and consequently require longer recovery time periods. For adaptable protection and management of natural resources, it is critically important to address and, whenever possible, to capitalize on differential resilience and stability attributes of ecological systems (Holling 1973). It should be emphasized that the vegetation recovery may or may not be indicative of functional recovery. Unfortunately, the scientific understanding of the relationships between vegetation and wetland functions is limited and fragmented (Nickerson et al., 1989).

7.5. Conclusions and Research Needs

Vegetation is a fundamental indicator of wetland health. It is the primary short-term parameter that can be measured in disturbed ecosystems. Specific and integrative indicators of plant stress have been successfully tested in a variety of environmental conditions; some of them may allow the early detection of disturbance. Remote sensing is also an efficient tool to detect early vegetative stress. These indicators should be used in conjunction with community level response indicators to provide a comprehensive evaluation of ecosystem health. A number of measures are available to describe vegetation communities, but further investigation is required to analyze the sensitivity of these measures for wetland health assessment. Additional research is also needed to evaluate the role of vegetation in wetland functions and the cause and effect relationships between them. Yet, there is no single analytical method that can be used to predict how wetland plants will respond to changing environmental conditions. Vegetation systems are dynamic and vary significantly in space and time. If comparisons, predictions, or estimates of potential impact are to be made, interpretations of data must consider yearly variations. Sampling methods should be sought and tested to minimize the natural background "noise" inherent in any ecological system. Variation can be reduced by sampling over a period of at least two years and by the use of similarity indexes.

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8 • WETLAND HABITAT QUALITY

8.1. Introduction

Monitoring vertebrate communities can be an effective way to evaluate changing environments (Karr, 1987; Brooks et al., 1990; Root, 1990). Vertebrates are readily observable, and their presence represents an integration of the environmental features over relatively large areas (Brooks and Hughes, 1988). Thus, they are good candidates for response indicators of cumulative impacts (Leibowitz et al., 1991). Many investigators used birds as indicators of wildlife habitat quality (Roth, 1976; Adamus, 1983; Harris et al., 1983; Keller, 1985; Cable et al., 1989). Karr's (1981) "Index of Biotic Integrity" (IBI), based on fish assemblages, has been a widely applied method in North America to biologically assess aquatic habitats. Invertebrates have been also used as indicators of water quality and biological integrity of the ecosystem (Ohio EPA, 1988; Plafkin et al., 1989; Berkman et al., 1986; Lenat, 1988). Biological integrity is defined by Karr and Dudley (1981) as the ability to support and maintain "a balanced, integrated, adaptive community of organisms having species composition, diversity, and functional organization comparable to that of natural habitat of the region".

Biological criteria are valuable for assessing the alterations caused by human activities, because they directly measure the condition of the resource at risk, and detect problems that other methods may miss or underestimate (EPA, 1990). They do not replace chemical and toxicological methods, but they do increase the probability that an assessment program will detect degradation due to anthropogenic influences (Karr, 1991).

8.2. Habitat Quality Assessment Techniques

8.2.1 Habitat Evaluation Procedures (HEP)

One widely used assessment method is the U.S. Fish and Wildlife Service Habitat Evaluation Procedures (USFWS 1976, 1980). This method is based on models that allow the assessment of habitat quality and quantity for selected wildlife species. The habitat is the target of assessment; the species themselves are not studied and need not to be present. The HEP methodology has been primarily developed for application to terrestrial and inland aquatic habitats. However, the concepts of habitat evaluation may be applicable to other systems (USFWS, 1980).

The procedures provide a quantification of wildlife habitat that is based on two primary variables: the Habitat Suitability Index (HSI) and the total area of available habitat. Two types of wildlife habitat comparisons are combined: the relative value of different areas at the same point in time

and the relative value of the same area at future points in time. The HEP is based on the assumption that habitat for selected wildlife species can be described by an HSI. This index value is multiplied by the area of available habitat to obtain Habitat Units which are used in the above comparisons.

The first step generally involves delineating the study area, determining cover types (by remote sensing), and selecting evaluation species. The next step is to describe baseline conditions in terms of Habitat Units (HU's). The third step is the projection of future habitat conditions. So, a HEP analysis is structured around the calculation of HU's for each evaluation species in the study area.

Total area of available habitat. It is calculated by summing the areas of all cover types likely to be used by the evaluation species. If the study area is not subdivided into different cover types, then the total available habitat area is identical to the entire study area.

Habitat Suitability Index for available habitat. HSI values are obtained from models. We can use existing habitat models and convert them to HSI format, or develop an HSI model. The models must be in index form and linear:

HSI = study area habitat conditions / optimum habitat conditions

The HSI ranges between 0.0 and 1.0. The ideal HSI model is an index with a proven, quantified, positive relationship to carrying capacity (i.e., units of biomass/unit area or units of biomass production/unit area).

The HSI for available habitat is a function of the suitability of all cover types used by the species. Each cover type is assigned a suitability index. When all habitat needs are met by one cover type, the HSI is equivalent to the cover type suitability index. But if the available habitat consists of two or more cover types, then methods are required to aggregate cover type indices into an HSI for available habitat. Interspersion between cover types is important when all habitat needs are not provided by each cover type. In this case, the model should aggregate cover type HSI's into one HSI value. If interspersion is not important, then a different aggregation method is required, that is a simple weighted mean of the suitability indices for the cover types (weighted by the area of each cover type).

Habitat assessments using habitat units. Habitat assessments involve measurement and description of habitat conditions for baseline (present) assessments and impact (future) assessments. Baseline assessments describe existing ecological conditions. They identify wildlife resource capabilities at one point in time. For each evaluation species, the number of HU's at one point in time is computed, i.e., the area of available habitat is multiplied by the mean HSI. The HU's are evaluated and compared directly, if the objective of the assessment is to compare existing conditions in two or more areas. Additional calculations are required to quantify habitat conditions at several points in time (impact assessments).

For further details concerning the HEP methodology, refer to U.S. Fish and Wildlife Service (1976, 1980). The procedures may have been improved since then, because of certain inconsistencies (Keller, 1985).

8.2.2 Methods using bird communities

Birds are easily identified and occur in nearly every habitat type. Their responses to various types of stressors are fairly well known. They are sensitive to cumulative negative effects on the environment, that may be detected by the absence or reduction of certain specific species. The availability of historical data bases throughout the U.S., such as the Breeding Bird Survey (BBS), Breeding Bird Censuses (BBC), Christmas Bird Counts (CBC), and state Breeding Bird Atlases (BBA), provides a benchmark for future monitoring (Leibowitz et al., 1991).

The standard protocol for an avian census is a five-minute visual and auditory observation at selected vegetation plots during the early morning hours and ideally during the breeding season (Leibowitz et al., 1991). If the avian community is especially important, taped songs can be played and responses noted (Brooks and Hughes, 1988). The occurrence and relative abundance of bird species are recorded, and diversity indices (Krebs, 1989) and guild analyses (e.g., Short, 1984; DeGraff et al., 1985; Brooks and Croonquist, 1990) can then be extracted. Additional sampling during spring or fall migrations and/or during winter residency may be conducted for subpopulations of particular concern (Leibowitz et al., 1991).

The community composition and abundance of birds are subject to considerable spatial and temporal variability. However, long-term monitoring can help identify trends. It is not necessary to detect all the individuals present in a given area. By using only data on species presence or absence, we can ascertain the functional composition of the community. It is important to determine which avian taxa and guild combinations provide the most information about changes in the wetland resource (Leibowitz et al., 1991).

Habitat Assessment Technique (HAT). Cable et al. (1989) present a wetland habitat assessment technique using birds as indicators of habitat quality, but in theory, any taxa could be used. The technique is a refinement of the Graber and Graber method (1976) and its modified versions (Holmes et al., 1986; Brack et al., 1987). The technique is quick, simple, inexpensive, and can be used to screen large numbers of wetlands. Measures of species diversity and rarity are used to assess the quality of the wetland. The presence of more species and uncommon species makes an area more valuable. Generally, a habitat of greater quality retains a greater number of species (Cable et al., 1989).

Faunal index. A faunal index is used as a basis for assessing the quality of the wetland. It is calculated by dividing a measure of species diversity and uniqueness (total species points) by a factor that accounts for wetland size (area factor).

Species index. To calculate the species index, wetland-dependent species are assigned base values (species points) on the state's (or other appropriate geographic area) breeding

population. Wetland-dependent species are those that require wetlands for a major life function, such as reproduction or feeding. For example, a wetland-dependent species with a state breeding population of 5000 or more individuals may have an assigned base value of 10 points, whereas a species with 2500-4999 individuals may have an assigned value of 20 points. Extremely rare species (e.g., less than 50 individuals or any species on a state or federal endangered species list) are not assigned points. Instead, they are considered "red flag" species. Accidentals and exotic species are excluded from the calculations.

A field survey is required to assess the avifauna composition of each area if reliable information for the site is not available. For each area, the species are enumerated and the species points totaled. The point totals are then averaged, thereby providing a species index for the site. This index, by virtue of the species composition, reflects habitat quality. To adjust for habitat size when calculating the faunal index, the total species points are divided by an area factor.

Area factor. The importance of wetland size in the assessment is dictated by both the theory of island biogeography and a concern for the efficient use and allocation of resources (Cable et al., 1989). The theory of island biogeography addresses the relationship between island size and species richness (e.g., MacArthur and Wilson, 1967; Simberloff, 1974). Specifically, isolated pieces of habitat, such as islands, will not retain a high species complement over time. The extensive body of literature on this subject indicates that species richness is associated most with tract size. HAT incorporates the notion of "optimum size" and penalizes wetlands that are too small or excessively large. Optimum is based upon the area required to maintain species diversity, especially of rarer species. To calculate the area factor, wetland habitat type and area size must first be determined from aerial photographs, maps, or field measurements.

Discussion. Besides focusing on wetland size, HAT is sensitive to other biogeographical factors. For example, the proximity of smaller tracts to one another and their configuration can affect the number of species they can support (Wilson and Willis, 1975). Small tracts connected by corridors hold species better than separate tracts. Tracts in a small cluster hold species better than tracts in a linear association (Cable et al., 1989). HAT requires minimal field time, since the only environmental variable of interest is wetland-dependent bird species. A field visit is not required at all, if site-specific bird records can be obtained. When field surveys are necessary, they are best conducted when the likelihood of migrants is low and breeding birds are vocal.

For wetlands, HAT has a variety of uses that supplement existing habitat assessment techniques. The technique is fast and provides a comparison of sites as part of the output. The raw data or output of HAT can frequently be used as input to some of the more extensive evaluation techniques. HAT avifauna data may be helpful in selecting appropriate birds for a HEP analysis. The wetland values derived from HAT are directly related to species diversity, an important component of habitat value. The rarity of species and the importance of habitat patch size are taken into account in the assessment.

Edge diversity and cover type diversity indices. Harris et al. (1983) tested two habitat indices -- edge diversity (ED) and cover type diversity (CTD) -- to predict bird species diversity (BSD)

in freshwater coastal marshes. Edge and unit size are important aspects of habitat structural diversity. They are important in the habitat selection of marsh-nesting birds (Kiel, 1955; Willson, 1966; Burt, 1970). BSD is directly associated with the structure of the vegetation (see Hilden, 1965, for review) and is often used as an indicator of ecological quality (Harris et al., 1983).

Harris et al. (1983) conducted breeding bird censuses and calculated BSD from the census data using the Shannon-Wiener information theory formula (Shannon and Weaver, 1949):

$$H = \sum_{i=1}^{s} p_i \ln p_i$$

where, s is the number of categories (species) and p_i is the proportion observed in the ith category.

They mapped major cover types found along the transects, using aspect dominance at the time bird nest searching occurred. Estimates of cover type proportions were derived from the cover maps and area was measured with a polar planimeter. CTD was then calculated with the Shannon-Wiener formula.

Cover maps were also used to estimate edge diversity. ED was calculated as follows:

$$ED = \frac{TE}{2} \sqrt{A(\pi)}$$

where, TE is the total linear edge between cover-cover and cover-water, and A is the area of the study plot.

The relationship between CTD, ED, and BSD were determined through multiple regression analysis. They found a linear relationship between BSD and CTD and a curvilinear relationship between BSD and ED. This means that BSD is related not only to the diversity of vegetation types, but also to the arrangement of those cover types within the marsh, as measured by edge diversity.

Harris et al. (1983) suggest that, for freshwater marshes, CTD and ED are useful measures of community change, and further, that they may be used as a "sign stimuli" (Hilden, 1965) by breeding birds in their selection among available marshes (Milligan, 1981). Habitat quality for marsh-nesting birds is thus equated to habitat diversity as measured by CTD and ED. BSD is an indicator of ecological quality because habitat heterogeneity is a function of the number of spatial niches in the habitat and it reflects the number of species populations present in a particular habitat (MacArthur and MacArthur, 1961; Anderson, 1979; Asherin et al., 1979; Milligan, 1981).

Harris et al. (1983) believe that the quantitative relationships they described can provide a rapid evaluation technique for use in assessing ecological quality in marshes. The potential of aerial

small-scale imagery coupled with computer-assisted digital analysis presents considerable promise in ecological evaluation.

Roth's index of heterogeneity. Pianka (1966), Murdoch et al. (1972), Blondel et al. (1973) and Wiens (1973, 1974) measured the horizontal component of habitat diversity with various techniques, and related it to the diversity of the animal community being studied. MacArthur et al. (1962) concluded that patchiness -- horizontal variability in the types of profiles in a habitat -- was the principle factor affecting bird species diversity, and that its effect was much more important than that of additional vegetation layers to support ground species, canopy species, etc. (Roth, 1976). Since vertical measures, such as foliage height diversity (MacArthur and MacArthur, 1961) and percent vegetation cover (Karr and Roth, 1971), do not measure horizontal patchiness, a measure that combined both horizontal and vertical variability of the vegetation seemed desirable. Roth (1976) developed an index of heterogeneity that is a measure of patchiness on a scale important to birds.

Index of heterogeneity (D). In searching for a measure of heterogeneity of ecological relevance to birds, Roth (1976) considered habitat features which seemed to contribute significantly to habitat patchiness. A view of Texas brush-grasslands suggested the shrubs and their spacing as a logical choice. A regular distribution of shrubs or trees of uniform size and shape is the ultimate in homogeneity for a habitat (Roth, 1976). We could expect only a few bird species to be able to partition this habitat spatially because of the difficulty of discriminating specific patch types. A change in dispersion of the plants in either direction should create patches with shrubs and trees of different densities, and consequently, patches preferred and recognizable by several different bird species (Roth, 1976). Hence, more species can live in the area. Therefore, variation in spacing of vegetation dominants, coupled with variations in their height and shape and in associated plant cover, is considered a major cause of spatial heterogeneity.

The heterogeneity index is derived from the point-centered quarter technique that involves measuring the distances from a central point to the nearest plant in each quadrant of a circle. Since these distances give information about dispersion and density, they should measure heterogeneity. Distances collected from a habitat with regularly distributed vegetation should have less variation than those collected from habitat with random or clumped distributions. Thus, the coefficient of variation (CV) of the distances from a homogeneous distribution should be lowest. The index of heterogeneity (D), or coefficient of variation of distance (CV) is calculated as follows:

$$D = 100 \frac{SD}{\bar{x}}$$

where, SD is the standard deviation for CV and $\bar{x}$ is the mean of the point-to-plant distances. Roth used D to emphasize the distance concept. Where D is used without subscript, it refers to the general index without specifying the plant life form being sampled. An appropriate sampling is added when discussing D calculated for a particular life form, e.g., D_s for point-to-shrub

distances. Difficulty of identification of a common plant form which is important in all habitats may limit the usefulness of the index.

Correlation between D and BSD. The heterogeneity index was significantly correlated with BSD for several shrub and forest areas (Roth, 1976). D predicted BSD for a series of similar brushlands where other indices had failed (Roth, 1976). Roth gives a biological explanation of the positive correlation between vegetation patchiness and BSD. Individuals of a species select sites of certain vegetational configuration for breeding purposes that are more or less unique to that species. A species is likely to be present in a habitat where its patch type(s) is present, assuming that other resources in that habitat are adequate, and competitive pressure from other species is low. For additional species to be accommodated in the same habitat, there must be either an increase in the number of kinds of patches available to permit spatial segregation, or more spatial overlap by species in their utilization of the patches available. To find out which hypothesis is correct, we can examine changes in species overlap with increased species packing. Roth's explanation of heterogeneity and species packing agrees with results and suggestions of other work (MacArthur et al., 1962; Karr and Roth, 1971; Willson, 1974).

Conclusion. Earlier work has failed to find significant BSD/heterogeneity correlations especially when the common index of foliage height diversity (FHD) was used. FHD predicts BSD on a coarse scale because vertical layers of vegetation have only a coarse effect on BSD. Subtle differences in BSD among similar habitats are due to subtle differences in habitat structure to which FHD is insensitive. Thus, as Willson (1974) has noted, there is a need for an efficient, simple, and biologically reasonable heterogeneity index which will take both horizontal variation and layering into account. While D holds promise, it is only an indirect measure and it requires sampling of a common plant life form to be universally applicable (Roth, 1976).

Use of remote sensing to assess waterfowl habitat quality. Waterfowl habitat quality is a function of both water conditions and terrain characteristics of the surrounding wetland and upland cover types (Colwell et al., 1978). Habitat quality, according to Colwell et al., relates to the potential of the habitat to attract breeding waterfowl and furnish the requirements for survival and successful rearing of broods. They developed a model for the assessment of waterfowl habitat quality based on the various relationships between ponds and the surrounding terrain types. They also tried to estimate annual duck production by monitoring the number of breeding and brood ponds present in the habitat. Numerous investigators have indicated a relationship between amount and timing of water bodies and current year's duck production (Colwell et al, 1978). Early-season ponds are of some importance in attracting the migrating ducks; in the absence of adequate ponds, some of the potential breeding population may overfly the area. Later-season ponds are also important, for breeding pairs and brood rearing.

The model. The model developed by Colwell et al. (1978) evaluates waterfowl habitat quality on the basis of water conditions and terrain characteristics. The specific water conditions are pond number, pond area, and pond size-class distribution. The literature suggests that 10 or more ponds per section are optimal for duck production, depending on the species of duck and the region. Both large and small ponds are important for good waterfowl habitat, although there

is some disagreement about their relative importance (Colwell et al., 1978). They suggest that 17.5 ha of water area is an optimal habitat. Once these factors were calculated, they integrated them into one single pond factor. The terrain characteristics they evaluated were the presence and spatial arrangement of certain terrain types (hay, grasses, pasture). They incorporated presence and spatial arrangement into a single factor represented by the amount of edge between desirable terrain types. The resulting model for waterfowl habitat quality combined pond and terrain factors, and generated ratings on a section-by-section basis.

This habitat model was preliminary; no detailed analysis of the accuracy of the model ratings has been made. The available knowledge of the relative importance of habitat characteristics and their relationships with each other was limited at that time.

Use of remote sensing. Colwell et al. (1978) used Landsat data in their model for assessing waterfowl habitat quality. Pond and terrain characteristics were determined from multidate Landsat imagery and aerial photography. Remote sensing data allow monitoring changes in the habitat quality over time. With the advent of satellites with a better spatial resolution (e.g., SPOT), it is possible to improve the accuracy of the pond and terrain factors.

8.2.3 Assessing fish communities

Fish are not particularly suited for monitoring many wetland types because of the high variability of water depths; however, they should be included in the biological sampling of wetlands associated with deepwater habitats and rivers. Sampling methodologies vary with the habitat type and species expected (Brooks and Hughes, 1988). Seines suffice in shallow waters with little woody debris. Backpack electrofishers are often more effective where woody debris make seining impractical. In waters too deep for wading, or in those with dense macrophyte beds, experimental gill nets (multiple mesh sizes) and minnow traps or grabs should be used. All habitat types should be sampled. Brooks and Hughes (1988) recommend sampling in early spring (adults) and late summer (juveniles). See Pardue and Huish (1981) for applications in wetlands.

Many individual studies demonstrate correlations between degradation and some biological indicator (e.g., species richness, abundance of an indicator species, production/respiration ratio). Karr (1981) developed an index -- the Index of Biotic Integrity -- that integrated several of these indicators into a single index. He used a set of attributes that measure fish community organization and structure. This index has now been widely applied in North America.

Karr's Index of Biotic Integrity (IBI). The IBI provides a broadly based and ecologically sound tool to evaluate biological conditions in streams (Karr, 1981). Each metric is compared to a regional reference site with little or no influence from human society (Fausch et al., 1984). For each metric, an index score of 5 is assigned if the study site deviates only slightly from the reference site, 3 if it deviates moderately, and 1 if it deviates strongly from the undisturbed conditions. Twelve attributes of a fish community are rated. The sum of those ratings (5, 3, or 1) provides an IBI value, that reflects the local biological integrity. IBI uses three groups of metrics: species richness and composition, trophic composition, and fish abundance and condition.

Species richness and composition metrics. Because richness varies as a function of region, stream size, elevation, and stream gradient, all sites must be compared to the expected richness from a similar undisturbed site (Karr, 1991). This group of metrics includes the total number of fish species, the number of benthic species, water-column species, long-lived species, intolerant species, and the percentage of tolerant species.

Trophic composition metrics. This group of three metrics -- respective proportions of omnivores, insectivorous cyprinids and top carnivores -- evaluates the trophic composition of the fish community to assess the energy base and trophic dynamics of the resident biota. Instead of measuring the productivity at several trophic levels directly, Karr (1991) suggests to measure divergence from expectation as a way to assess energy flow through the community.

Fish abundance and condition metrics. They include the total number of individuals in the sample, the frequency of hybridization, and the proportion of individuals with disease, tumors, fin damage, and major skeletal anomalies that can be discovered by external examination.

IBI scores can be used to 1) evaluate current conditions at a site; 2) determine trends over time in a given area; 3) compare sites; and 4) to some extent, identify the cause of local degradation (Karr et al., 1986). This index is quantitative, and there is no loss of information from constituent metrics when the total index is determined (Karr, 1991). IBI does not serve all needs of biological monitoring (Karr et al., 1986), and certainly cannot replace physical and chemical monitoring or toxicity testing (Karr, 1991).

Successful use of IBI in a variety of contexts and in a diversity of geographic areas prove the utility of its concept (Karr et al., 1986; Miller et al., 1988; Steedman, 1988; Fausch et al., 1990). IBI can be modified to incorporate other aspects of the fish community, such as growth rates, population structure, etc. Adaptation of IBI to geographic regions outside the midwestern United States requires modification, deletion, or replacement of selected IBI metrics. Miller et al. (1988) provide the most up-to-date review of changes needed to reflect regional differences in biological communities and fish distribution. Efforts should be made to develop IBI-type indexes for use in other environments, such as wetlands, lakes, and terrestrial ecosystems. Brooks and Hughes (1988) have advanced the evaluation of wetlands.

8.2.4 Assessment of other taxa

Although initially developed for use with fish communities, the ecological foundation of IBI can be used to develop analogous indexes that apply to other taxa, or even to combine taxa into a more comprehensive assessment of biotic integrity (Karr, 1991).

Invertebrates. The framework of the fish IBI has been adopted by invertebrate biologists to develop assessment methods that use benthic invertebrates. The most extensively tested, integrative effort is the Invertebrate Community Index (ICI) developed by Ohio EPA (1988). ICI is a 10-metric index that emphasizes structural attributes of invertebrate communities. Another method, The Rapid Bioassessment Protocol III (Plafkin et al., 1989) is similar to the ICI but has

only eight metrics. The RPB III has been less extensively tested, and many validation studies remain to be done (Karr, 1991). These and other approaches that use invertebrates for the assessment of biotic integrity (Berkman et al., 1986; Lenat, 1988) are not as widely validated as is IBI, but they show considerable promise as additional water resource tools.

Brooks and Hughes (1988), in their guidelines for assessing the biotic communities of freshwater wetlands, describe some sampling methodologies for invertebrates. Terrestrial flying insects and emerging aquatic insects can be sampled either by clear, plastic funnel traps or sweep nets. Insects are captured in a removable jar at the top of the cone or pyramid-shaped funnel (McCauley, 1976) and placed in 70% ethanol for later identification. Aquatic macroinvertebrates can be sampled with activity traps (Murkin et al., 1983) placed underwater where water depth is sufficient. Traps are checked daily. Benthic invertebrates are sampled from hydric sediments or soils using a lightweight coring device.

Mammals. Mammals tend to be more sedentary than birds, but they are often more difficult to detect. Mammals sampling requires the use of several trapping techniques. Small ground-dwelling mammals are sampled using pairs of baited live and snap traps placed along the transects (Brooks and Hughes, 1988). The presence of larger mammals is ascertained by looking for signs, such as tracks or droppings. If presence of large mammals is of particular concern for a given wetland, then additional methods, such as scent stations (Linscombe et al., 1983) can be used. However, the attraction of mammals with large ranges to a scent station in a small wetland may bias the results (Brooks and Hughes, 1988). Brooks et al. (1990) and Croonquist (1990) found only weak correlations between the occurrence of mammals and the variable levels of disturbance affecting wetland and riparian areas. Many mammalian species are adaptable to habitat alterations, and only a few species are sensitive to the negative impacts that threaten aquatic systems.

Herpetofauna (reptiles and amphibians). Herpetofauna are common in wetlands. Reptiles, such as turtles, with their long life span and amphibians sensitive to water pollution are good indicators of cumulative impacts, thus good potential candidates in a monitoring program (Phillips, 1990). Arrays of pitfall traps and drift fences have been recommended as the best means of sampling herpetofauna (Vogt and Hines, 1982; Campbell and Christman, 1982). Less labor intensive techniques include timed searches of concealed herpetiles under logs, slash, and rocks, and recordings of vocalizations during the breeding season (Brooks and Hughes, 1988). For most regions of the United States, herpetofaunal communities may be too small and elusive to give comparable results across a broad spectrum of wetlands (Leibowitz et al., 1991).

8.3. Conclusion and Research Needs

Diversity is one of the most frequently used criterion to assess conservation potential and ecological value (Margules and Usher, 1981). Though diversity has been studied in many community or habitat types (Asherin et al., 1979; Fuller, 1980), it has not been validated for marsh communities. Current wetland evaluation procedures employ concepts of vegetation

structure and complexity but have not quantitatively demonstrated the relationship between vegetation structure and habitat quality (Harris et al., 1983). Birds are broadly used to assess biological integrity of the habitat, because relative to other taxa, they are conspicuous, easy to identify, and they are present in almost all habitats, thereby minimizing field time. Karr's Index of Biotic Integrity is a widely applied method to assess water resource quality. The ecological foundation of IBI can serve as a basis to other indexes that apply to other taxa. As suggested by Karr et al. (1986), biological assessments require an integrative approach relying on several taxa and variables.

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9 • WETLAND HYDROLOGY

9.1. Introduction

Hydrology is probably the single most important determinant for the establishment and maintenance of specific types of wetlands and wetland processes (Mitsch and Gosselink, 1986). The presence or absence of ground and surface waters and the frequency of inundation determines the existence of a wetland. Disturbances of wetland hydrology are commonly associated with human activities on the landscape. When the hydrologic pattern remains similar from year to year, a wetland's structural and functional integrity may persist for many years. But if hydrologic conditions change even slightly, the biota may respond with massive changes in species richness and ecosystem productivity (Mitsch and Gosselink, 1986). Wetland functions, such as flood storage, and sediment and contaminant retention, as well as nutrient fluxes may also be altered. In the longer term, hydrology determines, through erosion and deposition, the shape, size, depth, and even the location of a wetland (Kusler, 1987). Some types of wetlands (e.g., bogs and mires) exhibit little variation in water levels from year to year, whereas other types (e.g., floodplains) may depend on frequent and drastic changes in the amount of water entering and leaving the system (Leibowitz and Brown, 1990).

The literature dealing with wetland hydrology is meager and topically specific (Mitsch and Gosselink, 1986). Emphasis is often placed upon narrow water budget calculations and depth of water without taking into account other factors. The role of long-term fluctuations in precipitation and water level and interactions of wetlands with hydrology are rarely considered (Kusler, 1987).

9.2. Critical Hydrologic Features of a Wetland

Critical hydrologic features of a wetland may be broadly grouped into two categories: those relating directly to the water in the wetland, and those which interact with the water to produce or affect certain characteristics or functions (Kusler, 1987).

9.2.1 Water

<u>Sources of wetland water and path of water discharge</u>. Sources of wetland water (direct precipitation, groundwater, surface water) determine specific wetland characteristics. For example, water entering wetlands from the ocean generally has some salinity and a high energy profile due to tidal action, determining vegetation and fauna. The way the water is discharged from the wetland is also important: discharge primarily through evaporation or transpiration indicates that the wetland is probably an effective sink for sediment and very sensitive to

nutrients, sediment, or dissolved/suspended substances; whereas, if discharge is through a surface outlet and active flow, the wetland is less likely to be a sink (Kusler, 1987).

Number of water exchanges. The number of water exchanges into and out of the wetlands is very important to sediment exchanges, and is a good indicator of sediment accumulation and fisheries movement (Turner, personal communication).

Water depth. The depth of water in various portions of a wetland during "normal" periods, during long-term fluctuations in precipitation, and during rare but extreme events such as floods is critical to many wetland characteristics and functions. Generally, depth determines vegetated vs. open water areas of a wetland and its vegetation type. Many plant species, particularly forested wetland species, are very sensitive to depth. Depth is also critical to use of particular portions of wetlands by muskrats, fish, wading birds, turtles, and other species. Average depth at a particular point over a period of months or years may be critical for certain plant and animal use. Water depth reflects the topographic contours of the wetland and the amounts and levels of incoming and outflowing waters including direct precipitation and surface and subsurface flows. Depth varies with precipitation and tide levels and also varies over time as deposition and erosion occur within a wetland (Kusler, 1987).

Water velocity. The velocity of the water entering and passing through a wetland determines some wetland functions such as: 1) flood conveyance (i.e., its ability to convey a given amount of water from upstream to downstream within a certain period of time); 2) flood storage; 3) sediment transport and trapping; 4) potential for short and long term flushing of sediments out of the wetland; and 5) pollution control. In general, the higher the velocity, the greater the flood conveyance capability. The higher the velocity of the water entering the wetland and the lower the velocity of the water exiting, the greater the sediment trapping and pollution control functions (Kusler, 1987).

Wave action. Wave action is important where the wetland contains substantial open water or is adjacent to open water with at least moderate depth (over four feet). Wave height and the erosive force of waves depend on water depth, the "fetch" (width) of open water, the presence or absence of particular types of vegetation, and the substrate material. Wave action determines, in part, the type and condition of wetland vegetation and soils. Most wetland plants cannot germinate and grow in moderate to high energy zones. However, some plants may survive in a less moderate energy area if protected until maturity (Kusler, 1987).

Fluctuations in water sources, velocities, sediment loadings etc.. Fluctuations in the characteristics of the water flowing into and out of a wetland may be as important as "normal" or mean conditions. The timing and duration of maxima and minima for water depths and velocities determine many wetland characteristics and functions. Relatively short-term inundation by flood flows, which is common for coastal and riverine wetlands, may not affect vegetation much, unless it changes salinity or causes erosion; whereas, long-term inundation in isolated wetlands will often kill trees and other plants (Kusler, 1987; Weller, 1987).

Dissolved/suspended materials content in water, turbidity, and temperature. Although dissolved and suspended substances in waters (nutrients, sediment, detritus) flowing into, through, and out of a wetland may not be considered a "hydrologic characteristic" per se, such materials play major roles in determining wetland habitat, food chain support, pollution control, and other functions (Kusler, 1987). Suspended solids also affect flow rates and the erosive force of water. Waters with a very high sediment load flow more slowly through wetlands than clear waters, and have less erosive force. Suspended and dissolved substances determine the long-term shape, size, depth, and even location of the wetland and its long-term functions. For example, high sediment loadings may fill a wetland, reducing or destroying its flood conveyance and storage potential, and virtually all other functions. Similarly, high loadings of nutrients and organics may lead to rapid filling of the wetland by organic matter, altering its hydrology. Water temperature and turbidity influence vegetation and fauna, affecting habitat values and indirectly affecting flood conveyance, pollution control, and other functions.

9.2.2 Wetland features that interact with water flow

A wetland develops certain features in response to its hydrology, which, in turn, affect wetland hydrology.

The size, shape, contours, and depth of most naturally occurring wetlands are formed by the forces of ice or flowing water. The shape of the wetland determines, to some extent, its hydrology and its functions, especially its flood conveyance capability. However, the same erosional and depositional forces that create the wetland continue to modify its shape over time. These changes may be accelerated by impacts to the watershed such as drainage, urbanization, tree-cutting, which increase peak flood flows and sediment loadings (Kusler, 1987).

Wetland hydrology determines its vegetation; but also, the vegetation modifies the hydrologic characteristics of the wetland. Vegetation produces detritus and organic soils which may gradually decrease wetland depths. The type and density of vegetation affect water velocities and flood storage and conveyance potential. It determines, in part, sediment and detritus trapping potential, affects erosion rates, and in some cases (forested wetlands) groundwater levels through evapotranspiration (Kusler, 1987).

The suspended and dissolved substances in the inflowing waters influence wetland soil characteristics, which, in turn, determine, to some extent, vegetation, substance retention potential of the wetland, and erosion rates during periods of high flows (Kusler, 1987).

Animal life also interacts with wetland hydrology. Microorganisms and small organisms feed and break down detritus, thereby influencing wetland soils and wetland depth; beavers alter drainage; muskrats and alligators deepen some wetland areas or reduce vegetation in others; waterfowl may denude some areas of a wetland; carp destroy aquatic vegetation and increase water turbidity (Kusler, 1987).

9.3. Assessment of Wetland Hydrology

Wetland hydrology assessment includes the study of water flow (precipitation, ground water, surface water) into, through, and out of the wetland, the characteristics of this flow, and its interaction with the wetland. It would be ideal to have site-specific and quantitative hydrologic data for every wetland, but this is quite unrealistic, because of the high costs and manpower and the long time frame required for evaluation. Besides, the calculations of water budgets are complex, and there are limits to short-term quantified measurements. For example, it may take, at minimum, several years of monitoring with a series of wells to determine groundwater relationships (Kusler, 1987). Wetland hydrology can be approached with varying levels of generality and quantification (Kusler, 1987):

- General, unquantified presumptions based on wetland origin and type, location in the watershed, or other factors. Generalized scientific information can usefully be used for some purposes, although generalizations must be made with care.

- More specific unquantified evaluation of the functions of a particular wetland based on its location, its shape and size, the topography of the surrounding land, and other site-specific and readily observable factors.

- Site-specific quantitative evaluations based on flood and stream flow data, topographic maps, aerial photography, superficial field surveys, and hydrologic monitoring (time series).

9.3.1 Water depth

Depth can be directly measured in a field survey through the use of a stadia rod or similar device. However, it is often difficult to decide what is "bottom" when the substrate includes many feet of unconsolidated organics. Depth during long-term precipitation cycles and flood events is harder to predict. Flood maps, stream flow records, and various modeling approaches based on precipitation may be used (Kusler, 1987).

9.3.2 Water velocity

"Guesstimates" of water velocity in a wetland can be made based on the overall characteristics of the wetland and the adjacent water body. In general, low velocities can be expected in isolated wetlands and along very low gradient streams. Whereas, relatively high velocities may be expected for coastal wetlands impacted by hurricane waves and for riverine wetlands along high gradient rivers and streams. "Guesstimates" are also sometimes possible based on examination of soils including the organic content and size of materials. Deep organic soils imply low velocities, whereas mineral soils, particularly those containing small rocks, imply higher velocities (Kusler, 1987).

More accurate values of water velocity can be obtained using continuous recorders attached to weirs or flumes (Leibowitz et al., 1991). Weirs are devices used to determine the quantity of water flowing over it, based on measurements of water depth over the crest or sill and known dimensions of the device. A flume is a channel placed in a stream of water to measure the volume or rate of flow. The discharge of channelized surface water can be measured very accurately (Rantz et al., 1982). Groundwater fluxes determination is more complex, because of the difficulty of defining flow-system boundaries, dynamics of recharge and discharge, hydraulic gradients, and permeability distribution (Winter, 1981). The groundwater flow-system modeling studies by Toth (1963), Freeze (1969), and Winter (1976, 1978) are mostly of steady-state, generalized systems and are most appropriate for understanding regional flow systems (Winter, 1988).

9.3.3 Wave energies

"Guesstimates" of wave energies may be made based on air photos or topographic maps indicating open water areas and the depths of such areas. The lack of vegetation or wetland soil along the shores of lakes, larger rivers, or the ocean may also infer moderate to high wave energies. Various models and other predictive approaches can be used to calculate potential wave energies at particular points (Kusler, 1987).

9.3.4 Changes in surface level

The land surface can gain water from the atmosphere by precipitation and lose water by evaporation and transpiration. Precipitation that falls on the land surface will flow over it, remain ponded in depressions, and/or infiltrate into the subsurface. The relative distribution of water involved in each of these processes depends on the slope and permeability of the land surface (Winter, 1988).

Field observations and aerial photography can be used in conjunction to define the maximum and minimum reaches of surface waters in and around wetlands. Continuous water-level recorders are desirable, but are more expensive to install and maintain than staff gauges that are read at periodic intervals (Leibowitz et al., 1991). Other similar devices can be used, such as iron rods or PVC wells. The wells must be placed at strategic stations along transects established for plant and animal sampling. Slotted PVC pipes allow measurements of water level above the surface during wet periods, and below the surface during dry periods. All types of gauges must be anchored below the frost line or tied into a surveying grid to maintain accurate depths over a period of years (Leibowitz et al., 1991).

Research on evaporation and transpiration from wetlands has been minimal. Evapotranspiration is determined by 1) differences in the water budget; 2) evaporation pan data; 3) evaporimeters and lysimeters; or 4) any of several empirical formulas (Winter, 1988). If pans are used, large errors in the calculated evapotranspiration are common because the openwater evaporation from a pan is not a surrogate for transpiration by plants. Studies of evapotranspiration using lysimeters and evaporimeters containing wetland vegetation are not common (Carter, 1986). When

empirical formulas are used to estimate evapotranspiration, the sensors that provide the data are often located at the nearest weather station, not at the site of interest, leading to large errors.

Long-term fluctuations in water levels and hydroperiod (duration) are often difficult to predict. Direct gauging of water levels is, of course, desirable. But long-term records are rarely available (Kusler, 1987). Where hydrologic gauging stations and other relevant records are not available, there may be natural indicators of hydrologic history. Sediments often contain information on their source, rate of sedimentation, and distribution within the wetland (Cooper et al., 1987). Dendrochronology and aberrations in the form of tree growth (Sigafoos, 1964; Everitt, 1968) may help interpret the magnitude of flood events where no other records are available. However, the existing plant community species composition may not be a reliable indicator of hydrologic conditions (Brinson, 1988).

9.4. Use of Remote Sensing in Hydrology

Remote sensing can provide some information on the hydrologic regime of the wetland, such as changes in open areas, in surface level, and in soil moisture. A number of studies have used remote sensing as a method for flood analysis and soil moisture assessments (Sollers et al., 1978; Harker and Rouse, 1977; Ragan, 1977; McGinnis and Rango, 1975; Rango and Anderson, 1974; Moore and North, 1974; Rango and Salomonson, 1974; Williamson 1974; American Water Resources Associations, 1974; Piech and Walker, 1971), (Schmugge, 1983; Cihlar, 1978; Schmugge et al., 1977; Myers et al., 1977; Idso et al., 1975; Blanchard et al., 1974; Waite et al., 1973; Werner et al., 1971). Microwave radiometric sensors are very effective at measuring water content, both in the atmosphere and on the earth's surface. These systems can be used to map areal distribution and variations in rainfall, water absorption rates of surface soils and map flood water distribution and flow patterns on an all-weather synoptic basis (Kennedy). The microwave radiometer functions as a temperature-measuring device. The capability of the radiometer to measure atmospheric hydrology derives from the electromagnetic properties of atmospheric water vapor, oxygen, clouds, rain, and the earth's surface which differ greatly in electromagnetic properties. The dielectric properties of surface materials are strongly dependent on moisture content. Changes in the dielectric constant result in major changes in the emissivity and radiometric brightness temperature (Kennedy).

9.4.1 Flood monitoring

Aircraft and satellite data have been used to perform floodplain mapping by two complementary approaches: static and dynamic (Sollers et al., 1978). The static approach is based on the recognition of geomorphological features formed by historical floods such as terraces, alluvial fans, natural levees, bars, oxbows, marshes, deltas, etc. Floodprone areas tend to have multispectral signatures that are distinctly different from those of surrounding nonfloodprone areas. The dynamic approach uses images of floods as they occur or soon afterward. Visible evidence of inundation in the near infrared region of the spectrum remains for up to two or more weeks after the flood. The near infrared reflectivity is reduced in the flooded areas because of

the presence of increased surface-layer soil moisture, moisture stressed vegetation, and isolated pockets of standing water. The inundated areas are characterized by the water absorption band (700-1100 nm). Visible and near infrared channels are recommended for analysis. The features observed here are the atmospheric conditions (clouds, air mass characteristics, precipitation..), flood water levels, and soil and vegetation characteristics after the high waters have receded. Soil moisture and sediment traces in water can indicate the path and extent of flood damage to a plain (Currey, 1977). Vegetation also exhibit patterns related to flood conditions: flood stressed plants reflect more blue and less infrared radiation (Sollers et al., 1978). Satellites, such as ERTS, NOAA, Landsat, SPOT could help reduce short and long term flood losses and provide regional water resources planning information. Data from these satellites would therefore complement aircraft and conventional surveying methods to ascertain the areal extent of flooding (McGinnis and Rango, 1975). Despite its usefulness in flood monitoring, remote sensing has limitations: 1) some systems don't have the resolution needed to delineate the boundary of flooded areas; 2) the scale of floodplain mapping is not large enough for most legal requirements; and 3) clear weather conditions are necessary with passive sensors. When possible, a combination of sensors should be used. Remote sensing data can serve as a base for assessment of potential flood damage, in identifying areas where further surveys are merited.

9.4.2 Soil moisture assessment

Soil moisture and its spatial and temporal behavior is of critical importance to disciplines such as agriculture, hydrology, and climatology. Specifically, soil moisture assessments are needed to study flood water distribution and flow patterns, distribution and variations in precipitation (especially rainfall), runoff following precipitation, and evapotranspiration (Kennedy; Cihlar, 1978).

Most techniques developed for soil moisture measurement provide point estimates, therefore are not suited for large areas (Cihlar, 1978). The traditional method of soil moisture measurement is to weigh a sample of soil, oven-dry it, and reweigh it. The difference between the wet and dry weights represents the soil moisture, and the percent moisture is then extrapolated to the entire field. This method is time-consuming and representative of only small areas. The status of remote sensing techniques for soil moisture estimation was reviewed in a workshop organized in 1978 in Maryland (Cihlar, 1978). The techniques discussed were: 1) the reflected solar technique; 2) the thermal infrared technique; 3) the active microwave (radar) technique; 4) the passive microwave (radiometer) technique; and 5) the gamma radiation technique. The review indicated the complementary nature of the various techniques. Thus, it is likely that a combination of sensors will be needed to provide accurate soil moisture estimates from satellites. Thermal infrared and both microwave approaches have shown potential for estimating near-surface water contents but the sensitivity to water at greater depths and under canopy seemed limited to the thermal infrared technique (Cihlar, 1978).

9.5. Disturbances of Wetland Hydrology

Because the hydrologic system is a continuum, any modification of one component will have an effect on contiguous components. Disturbances commonly affecting the hydrology of wetlands include weather modification, alteration of plant communities, storage of surface water in reservoirs, road construction, drainage of surface water and soil water, and alteration of groundwater recharge and discharge areas (Winter, 1988).

Weather modification refers principally to inducing precipitation through cloud seeding (Winter, 1988). Alteration of plant communities refers to the removal of biomass through harvests. If plants are removed from a wetland, the loss of water by evapotranspiration may change, thereby changing the quantity of water available for surface water flow or groundwater recharge. Complete removal of the transpiration process, which usually accounts for the greatest loss of water from wetlands, could result in considerably more water available for surface runoff, and/or groundwater recharge (Winter, 1988). The effects of drainage on nutrient concentrations flowing from wetlands (Kuenzler et al., 1977) and aboveground biomass production (Bayley et al., 1985; Carter et al., 1973) can be detected soon after alteration of hydrology. In contrast, the species composition of forested wetlands may persist for decades. Therefore, wetland inventories based only on aerial photographs may not detect changes resulting from altered hydrology (Brinson, 1988). Impacts that reverse depositional tendencies may cause wetlands to be large exporters rather than importers. Increases in rates as well as direction can cause stocks of materials, accumulated over centuries in wetland sediments, to be lost within decades, resulting in nutrient loading to downstream aquatic systems (Brinson, 1988). Wetlands should not only be assessed for their capacity to accumulate sediments, which may be slow, but for their vulnerability to export when hydrologically altered, which is potentially rapid.

The consequences of these alterations are fairly predictable on a local scale and are described by Brinson et al. (1981). In contrast to these single impacts on individual wetlands, cumulative impacts present a scale of complexity in time and space that is much more difficult to describe and predict (Brinson, 1988). In dealing with the spatial scale, we are confronted with complex watersheds units, often containing wetlands of several types.

The sensitivity of wetlands to various impacts is related to hydrologic characteristics. For example, salt marshes and marshes along rivers with continuous water supply and periodic flushing by flood flows are quite "self healing", provided the wetland topographic contours are maintained (Kusler, 1987). Pothole wetlands and other closed systems with perched water may be very sensitive to impacts and have a very slow recovery rate.

9.6. Relationships Between Hydrologic Functions and Wetlands

Wetlands are determined by their physiographic setting and the water balance that favor the accumulation or retention of soil water and/or surface water for a period of time (Winter, 1988). To determine the hydrologic support function of a wetland, the wetland must be placed in a

regional hydrogeologic context (Winter, 1976). Hydrogeologic classification of wetlands is important to understand a wetland's water balance and the effect of hydrology on other wetland functions (Hollands, 1985). The classification that appears to be most used by wetland regulators is that of Novitzki (1978). This classification combines topography, surface water, and groundwater parameters.

Flood storage and sediment retention processes are reasonably well defined (Novitzki, 1986). Flood desynchronization and nutrient retention are defined for some circumstances but not all. But the relations between wetlands and seasonal streamflow patterns, and between wetlands and groundwater are poorly defined (Novitzki, 1986). O'Brien (1988) explores the relationships between wetlands and flood reduction, wetlands and streamflow, wetlands and groundwater. A year or more of groundwater elevation observations (using piezometers) are required before recharge/discharge functions can be understood (Hollands, 1985).

Many wetlands have a capacity for maintaining water quality by buffering surface and ground waters from potentially damaging compounds (Brinson, 1988). The biological and non-biological mechanisms by which wetlands retain and transform high concentrations of nutrients and toxic compounds are covered in a review by Hemond and Benoit (1988). Sampling of water constituents is essential; however, it is too expensive to analyze them all and sometimes difficult to select those that deserve more interest. Besides, their concentrations fluctuate considerably, especially after climatic events and anthropogenic disturbances. Water quality data, by themselves, cannot be used to illustrate trends that imply deterioration due to wetland degradation. Data are not available for all surface waters of interest, nor are they generally available over a long enough period to detect changes in water quality. Brinson (1988) presents a geomorphological classification of wetlands that has relevance for water quality. He also shows how the position of these wetland types in the landscape may influence strategies for protecting water quality in different areas of a watershed.

Wetlands are intrinsically depositional landforms (Brinson, 1988). They tend to import many elements. But when a wetland is filled, drained, or deprived of its sedimentary function, it exports rather than imports elements. Data are required on both sedimentation and erosion to place the sedimentation process within a meaningful context at a cumulative impact scale (Brinson, 1988). A common technique to measure sediment flux is to place artificial surfaces into the substrate, and quantify the accumulation of sediment during the sampling period. Measurement of erosion rates is more difficult. The sediment inventory may best reflect the integrated history of a wetland (Hemond and Benoit, 1988). This is particularly useful where the manager must assess previous impacts on a particular site. Many pollutants are more concentrated in sediments than in water, thus simplifying the problem of making accurate measurements.

9.7. Conclusions and Research Needs

Hydrology is the primary and critical force that creates and modifies wetlands, thus it is a good indicator of wetland condition. Unfortunately, wetland hydrology and its changes over time (years, decades and longer) are not adequately understood. Many of the studies are empirical and focus on particular aspects of hydrology, and a few articles deal with practical assessment needs. Site-specific and quantitative data are essential for the assessment, but they are missing because of the high costs and the long time frame required for evaluation. Detailed hydrologic budgets for different wetland types must be calculated for a period long enough to include a range from relatively wet to relatively dry years. Studies on both reference sites and disturbed wetlands are needed to calculate flux rates under a variety of environmental conditions before threshold values associated with disturbance can be determined. Sediment analysis is a potential tool for the wetland manager, and is a fruitful area for future research (Hemond and Benoit, 1988). Remote sensing has been found to be very helpful in soil moisture and flood damage assessments. A combination of sensors seems to provide more accurate estimates.

It is important to understand the regional differences that exist in wetlands and in their potential hydrologic function. Considerable care should be exercised in extrapolating results from one region to another. Additional research is needed before hydrologic function can be reliably correlated with physical properties of wetlands and landscapes. Special effort should be directed toward understanding surface and near-surface flow processes within the wetland, the relation of the wetland to the local groundwater system, and the relation of the hydrologic regime to the wetland plant community. More research is also required to evaluate the relationships between the hydrologic regime and the cycling and transport of nutrients and pollutants. The ambitious and time-consuming studies required to answer all these questions would be facilitated by open cooperation between researchers, wetland managers, and funding agencies.

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10 • CONCLUSIONS AND RECOMMENDATIONS

Currently, there are no standard methods for monitoring wetlands, but a consensus is beginning to form among the scientific and regulatory communities. The combination of biological monitoring and evaluation of physical and chemical data may improve our understanding of processes that operate at higher levels of biological organization, such as communities and ecosystems (Brooks, 1989). Vertebrates, invertebrates, and vascular plant communities, when analyzed in conjunction with selected abiotic parameters, serve as ecological indicators of change. Biotic communities are likely to integrate the incremental changes that occur in ecosystems over time, and therefore, reflect long-term, cumulative effects (Brooks, 1990).

As the scale of environmental problems is expanding quickly and spatially, it becomes urgent to develop standardized and fast procedures to manage our natural resources and stop or prevent degradation. Remote sensing technology (satellite imagery and aerial photography) is a very promising tool in environmental monitoring. It allows large areas of wetlands to be surveyed rapidly and easily. Some indicators of wetland condition, such as wetland extent and type, habitat structure, and the floral component of wetland productivity, can be determined primarily by means of remote sensing; while others (e.g., vegetation, hydrology, habitat quality) still require the use of more conventional techniques. In most cases, a combination of remote sensing and other techniques is used to collect data and assess the environmental condition of wetlands.

Aerial photography can be used to select sampling sites, establish field transects, identify and delineate the major vegetation types and to detect early vegetative stress. It also allows us to document present conditions for use in future trend analyses. Methodologies and algorithms for the determination of biomass and productivity of coastal wetlands habitat by remote sensing have been recently developed and will significantly enhance our ability to determine wetland condition over time on a regional scale. Remote sensing also provides information on physical alterations to the wetlands (flooding, human activities, etc.), soil moisture, and the wetland hydrological regime. By comparing two or more time periods, change in biomass, productivity, wetland extent, type, and patterns, wetland vegetation community composition, or some other factor correlated with spectral reflectance (e.g., greenness) could be used to index functional health. The activity requires ground based research to relate remotely sensed spectral radiances to these indicators. Various remote sensing methods are available and the choice of the method will depend on the project objectives and monetary constraints. Low-resolution data may be sufficient for the study of certain parameters, and higher resolution data will be required for detailed studies of selected sample sites. Our effectiveness for managing wetland resources in the future will depend on our ability to collect and analyze data on a regional and eventually global scale. The advances in instrumentation and in computer analysis techniques will greatly improve the types of data available.

Wetland health assessment procedures still require improvement through further research and testing; this would be facilitated by more cooperation between researchers, wetland managers and funding agencies. The importance of a more comprehensive approach in the protection of ecological resources is now being recognized. A large set of parameters must be taken into account to accurately assess the overall condition of wetlands. Sampling methods should minimize the natural variation inherent in any ecological system, and should be standardized throughout the country to provide a national assessment of wetlands status, changes, and long-term trends.

Further research is needed to better understand the physical and biological processes that contribute to the ecological integrity of wetlands, i.e., understand the relationships between physical properties of wetland and their functions. Some of the questions that still need to be answered are: which measures of structure best indicate landscape changes? What are the ecological processes that are related to landscape pattern? How does pattern affect disturbance regime, movement and persistence of organisms, redistribution of matter and nutrients? What is the role of vegetation in wetland functions? How much does vegetation structure influence habitat quality? How important are habitat characteristics for the maintenance of biodiversity? Is it possible to predict how wetland plants will respond to changing environmental conditions? What are the relationships between the wetland hydrologic regime and the cycling and transport of nutrients and pollutants? Hydrology is the driving force of wetlands and needs to be better understood.

Particular emphasis should be placed on conducting the following specific research:

- Test the applicability of remote sensing techniques for biomass and productivity determination in a large range of wetland types.

- Find other applications of remote sensing technology in wetland health evaluation.

- Analyze the sensitivity of the proposed indicators for wetland health assessment; a number of methods described here need more testing and validation in wetland environments.

- Find early warning indicators that would detect initial impairment in wetlands.

- Test other indicators of wetland condition, such as sediment characteristics, chemical contamination, bioaccumulation in tissues (Leibowitz et al., 1991).

- Develop health indexes sensitive to cumulative impacts (Karr, 1991).

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