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16. WETLAND VALUE IMPACTS

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16.1. Introduction

In some developed countries, there are few natural wetlands and extraordinary efforts are made to preserve those that remain (Klötzli 1982), or numerous regulations have been used to halt or slow the rate of wetland loss (Kusler 1979, 1983; Tiner 1984). Wetlands are not, unfortunately, uniformly accepted as being valuable or appreciated; large wetland areas are being destroyed in both developed and developing countries. In developing countries, wetlands are destroyed because of transmigration projects (Armentano *et al.* 1983) and efforts to increase food production or water supply (Weger and Ellenbrock 1980; Patil and Pathak 1982). In developed countries, wetlands are threatened by economic pressures to convert them into food, timber, or peat producing areas (Clark and Benforado 1981c; Richardson 1981; Sharitz and Gibbons 1982). Are suitable systems available for assessing values of the alternative uses and impacts to these values that result from wetland development? Are adequate models available to assist in the management process? The purpose of this paper is to address these questions by presenting an overview of relevant information. Our discussion is organized into the following sections.

1. *Wetland values.* Overview of wetland values and the value assessment process.
2. *Issues of time and space.* Comparison of how wetland processes and societal values change over time, and description of past and potential large scale wetland impacts.
3. *Wetland evaluation systems.* Overview of three categories of existing evaluation and value impact assessment methods, and critique of each, and
4. *Application of methods in resource allocation decisions.* Evaluation of methodologies as they relate to resource allocation decisions.

16.2. Wetland values

Prior to any consideration of wetland assessment systems or value impacts, it is necessary to determine what wetland functions are important. At one of the first United States meetings designed to evaluate wetland processes (Clark and Clark 1979), the National Wetland Technical Council asked a group of scientists to consider questions generated by several federal agencies. The questions were organized into five categories: (1) food chain values; (2) habitat values; (3) hydrologic and hydraulic values; (4) water quality maintenance values, and (5) harvest and heritage values. More recently, that list has been expanded for use in a method for wetland functional assessment developed by Adamus (1983) and Adamus and Stockwell (1983) for the U.S. Department of Transportation (Sather and Stuber 1984). The Adamus system attempts to formulate a procedure that can be used to provide decision makers with information on all important wetland functions. Adamus included ten functions in his system: (1) groundwater recharge and discharge; (2) flood storage and desynchronization; (3) shoreline anchoring and dissipation of erosive forces; (4) sediment trapping; (5) nutrient retention; (6) food chain support; (7) habitat for fisheries; 8) habitat for wildlife; (9) habitat for active recreation, and (10) habitat for passive recreation and heritage values.

Various other authors have considered the subject of wetland functions, and all basically

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agree with the list used by Adamus (1983). Examples include E.P. Odum (1979b), who considered six wetland functions in a hierarchical framework; Reppert (1981), who discussed wetland function in the context of developing an evaluation system for the U.S. Army Corps of Engineers; and Balco (1981), who summarized functional values used by the U.S. Water Resources Council in a study of existing wetland evaluation methodologies. More recently, cumulative impacts have also been considered (Beanlands *et al.* 1986).

Each of the ten functions used by Adamus (1983) is defined below. A given wetland may be valuable because it performs only one of the functions, but most often wetlands perform many functions whose multiplicity makes wetlands so valuable.

1. *Ground water recharge and discharge.* Ground water recharge is the movement of surface water into the ground water aquifer; discharge is the release of ground water into surface water.
2. *Flood storage and desynchronization.* Flood storage is defined as the storage of peak flows in a wetland basin and resultant delay in delivery downstream. Flood desynchronization is the process of storing peak flows in wetland basins resulting in a gradual release of water in a non-simultaneous manner. Flood storage and desynchronization result in less downstream flooding during high flow periods.
3. *Shoreline anchoring and dissipation of erosive forces.* Shoreline anchoring is the stabilization of substrate by plants, primarily by roots and rhizomes. Dissipation of erosive forces is the reduction of energy associated with waves, currents, ice, water level fluctuations, or groundwater flow.
4. *Sediment trapping.* This is the process by which organic and inorganic particles are retained or deposited in a wetland.
5. *Nutrient retention and removal.* Nutrient retention is the storage of nutrients in the substrate (by burial or sorption) or vegetation (by biomass accumulation) of a wetland. Nutrient removal is accomplished by conversion to gaseous forms that are then lost to the atmosphere. Examples are nitrogen (N_2), sulfur as hydrogen sulfide (H_2S), and carbon as methane (CH_4).
6. *Food chain support.* This is the direct or indirect use of nutrients, in any form, by animals in the wetland or any adjacent aquatic environment.
7. *Habitat for fisheries.* This function includes those physical and chemical factors of wetlands that affect the metabolism, attachment, or predator avoidance of adult or larval finfish and shellfish that are harvested for commercial or sport purposes.
8. *Habitat for wildlife.* This includes those wetland features that affect the food and cover requirements of wetland-dependent birds, mammals, reptiles, invertebrates, and amphibians.
9. *Active recreation.* This refers to activities that depend on water and can occur in wetlands or adjacent aquatic environments. These activities include swimming, boat launching or anchoring, power boating, canoeing, kayaking, and sailing.
10. *Passive recreation and heritage value.* This function includes the use of wetlands for aesthetic enjoyment, nature study, picknicking, education, scientific research, open spaces, preservation of rare or endemic species, maintenance of biotic gene pools, protection of archaeologically or geologically unique features, maintenance of historic sites, and many other intangible uses.

These functions clearly cover all potential values that humans can assign to wetlands (Darnell 1979). The important issue is, however, whether or not values can be assigned to each of these functions for purposes of determining the overall value of a wetland or for use in the process of wetland evaluation.

Wetlands clearly have societal values. Many recent papers (American Society of Civil Engineers 1978; Clark and Clark 1979; Greeson *et al.* 1979; Kennedy 1980; Richardson 1981; Gopal *et al.* 1982) have been devoted to that topic. In the United States, considerable effort has gone into attempts to categorize wetland values. As an example, the ten functions described above could be combined into three categories for purposes of value assessment.

Those functions that produce direct economic value to society have been called marketplace commodities (Farnsworth *et al.* 1981). Examples of marketplace commodities produced directly or indirectly by wetlands are fish, shellfish, waterfowl, peat, charcoal, and timber. The second and third value categories are described by Farnsworth *et al.* (1981) as non-market attributable and non-market intangible. Non-market attributable values would not normally be part of the marketplace system but have been assigned values through some institutional mechanism. An example would be the purchase of a threatened wetland to assure high quality potable water for a community. Non-market intangible items are those services provided to society that have not yet been incorporated into a marketplace system. An example of this type of value would be the role that undrained wetlands play in the global carbon cycle (Armentano *et al.* 1983). The second and third categories have been variously referred to as heritage values (Clark and Clark 1979; Niering 1979), visual-cultural values (Reimold and Hardisky 1979; Reimold *et al.* (1980) and as benefits (direct, downstream, and non-quantifiable) (Postel 1981).

Assigning wetland processes into functional categories is easier than assigning values to those processes. For example, almost everyone would agree that wetlands provide valuable support for aquatic food chains. There are, however, so few data to evaluate that aspect of wetland ecology that it is almost impossible to definitively conclude, short of complete elimination, what impacts will occur when a wetland is modified. For this purported wetland function there is not yet a suitable body of scientific data, including hypotheses, to assess its value. Evaluation of aesthetic and heritage functions is even more difficult. The presence of an important archaeological site in a wetland may be highly valuable to one segment of society but virtually of no value to another segment (Niering 1979).

16.3. Issues of time and space

Relevant to the specific methods for evaluating wetland functions are the dimensions of time and space within which wetlands function, society perceives these functions, and wetlands are subjected to impacts. These can be clarified by (1) comparing the time scales of wetland ecosystem dynamics with those of societal values and impacts, and (2) examining some differences in impact scales for specific wetland systems.

Societal values change temporally as do wetland processes and ecosystem development (Friedman and DeWitt 1979). However, value time scales are shorter than most processes responsible for wetland formation and persistence of vegetation (Fig. 1). Wetlands develop, for example, within a geological framework while attitude change (reflecting changes in societal values) fluctuates over much shorter time periods. A few years ago in the USA, wetlands were perceived to be of little use or value. As a result of changes in society's perception of their value, there is now voluminous legislation at federal, state and local levels designed to protect wetlands (Costle 1979; Kusler 1979, 1983). Regulations resulting from this legislation do not attempt to distinguish which wetlands are more valuable than others, what impacts are more harmful than others, or specific procedures for evaluation; hence, the need for valuation methodologies. For example, impacts that affect wetland formation would have long-lasting and essentially permanent impacts on wetland processes as compared with impacts that affect a single growing season (Fig. 1). It appears that more sophistication will be required to judge which types of impact are minor, reversible, and perhaps justifiable, and which types are major, irreversible, and unacceptable to society. Brown *et al.* (1978) and Brinson *et al.* (1981) have suggested that three impact categories be recognized: (1) geomorphic/hydrologic; (2) physiological stress, and (3) harvests. In that order, they represent decreasing impact severity, increasing potential for being reversed or mitigated, and, in reference to Fig. 1, decreasing time scales of effect. Concepts such as this should be incorporated into methods that assess wetland values and attempt to project consequences of impacts.

Wetland evaluation, functions, and impacts also need to be addressed within a framework

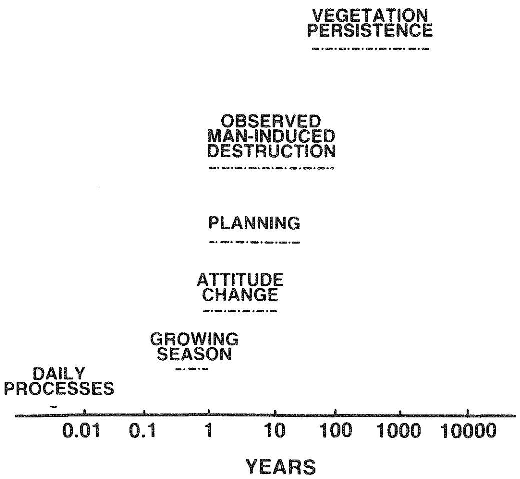

Fig. 1. Time scales of wetland ecosystem dynamics compared with time scales of societal processes and impacts. (From Friedman and DeWitt 1979).

of space or size scale. Such scales range from small, individual wetlands (Whigham 1987; Klötzli 1982; Hull and Whigham 1987) to regional wetlands (Mitsch *et al.* 1979; C. Richardson 1981; Ash *et al.* 1983), and finally to global environmental issues in which wetlands may have a role (Armentano 1980; Armentano *et al.* 1983). Thus, any evaluation system should require that evaluators specify the boundary conditions within which their methodology is to be applied.

There is a tendency in wetland resource management for decisions to be made on a case-by-case basis rather than dealing with problems at regional or global scales. W. Odum (1982) has pointed out how small decisions on resource utilization at lower levels or smaller scales can result in undesirable impacts on a large scale that are actually in conflict with societal values. The alternative would be to establish thresholds of societal values at higher levels and larger scales and provide the mechanisms that prevent these thresholds from being exceeded. As Odum pointed out, "No one chose to reduce the annual surface flow of water to the Everglades National Park, to intensify the effects of droughts, or to encourage unnaturally hot, destructive fires". Thus, any proposed method of evaluating wetland functions should be robust enough that it can be used at any scale.

Below are provided some examples of recent or proposed changes in wetland resources on a regional scale. The purpose is partly to determine how those impacts came about (*i.e.*, through large or small decisions), and partly to illustrate and emphasize the scale at which evaluation methodologies should be effective.

16.3.1. Riparian ecosystems

In the USA almost 70% of the original 23 million acres of riparian forests have been eliminated, most through conversion to urban and cultivated agricultural land uses (Johnson and McCormick 1978; Clark and Benforado 1981; The Nature Conservancy News 1981; Correll 1987). The term riparian forest is used here to include forested floodplains and streamside forests. Also, most areas were converted before there was much scientific and economic data to evaluate their societal importance (Johnson and McCormick 1978; Brinson *et al.* 1981; Clark and Benforado 1981). Thus, decisions on how to manage riparian ecosystems have been made on a case-by-case basis with little regard to regional effects. Although only 30% of the original resource remains, there still does not exist any appropriate mechanism for evaluating the potential impacts of continued loss. Consequently, emphasis has shifted toward preservation of riparian wetlands, and the most successful strategy has been identification and purchase of large land areas containing riparian forests. This strategy has been recently employed by The Nature Conservancy (The Nature Conservancy News 1981) and represents a decision on a large scale. That organization has secured U.S. $15-million to match a grant of a similar sum from a private foundation. The $30-million was used to purchase important wetland areas along six rivers in Florida, Alabama, Mississippi, and Louisiana.

16.3.2. Pocosin wetlands

Pocosins (C. Richardson 1981; Ash *et al.* 1983) and Carolina bays (Sharitz and Gibbons 1982) occupy extensive areas throughout the U.S. Southeastern Coastal Plain from northern Florida to southern Virginia. These wetlands are being eliminated at an alarming rate, and little is known about the ecological and economic impacts of draining large wetland areas in a region. In North Carolina it was estimated that almost 70% of the original 907,933 ha of pocosin habitat had been eliminated (C. Richardson 1981). Most of this land was and continues to be drained for conversion to pine plantations, large corporate farms, or peat production.

All these uses require construction of drainage ditches and resultant lowering of the water table. Clearly, the cost of drainage is worth the short term economic return (C. Richardson 1981), but none of the cost-benefit analyses include consideration of potential negative societal costs. These include water quality degradation (Barber *et al.* 1979; Gilliam and Skaggs 1981), and loss of coastal fisheries (Street and McClees 1981) and hunting and trapping resources (Monschein 1981). Because timber and agricultural industries are excluded from the North Carolina Coastal Area Management Act, little can be done to slow the rate of wetland alteration. This larger scale alteration, resulting from the integrated effects of independent local decisions, may have more than local impacts, as described in the section below on global carbon balance.

16.3.3. Middle Paraná River

An example of hydrologic impacts of great proportion is the proposed modification of the Middle Paraná River, the longest drainage of Rio de la Plata basin in Argentina, for hydroelectric generation (Bonetto 1975). This project draws attention because of the potential impact on one of the world's largest alluvial floodplains, an area in excess of 15,000 km^2. Also, it represents an example of decision for resource use at a higher level rather than chronic, piecemeal wetland loss that is occurring in the riparian and pocosin ecosystems described above.

The hydroelectric project for the Middle Paraná consists of constructing two dams across the floodplain. The first will be located just upriver of Santa Fe at an area known as Capetón where the floodplain narrows from its typical 30–35 km width. The depth of the proposed impoundment will be greatest near the dam and slowly grade to shallow and periodically exposed areas up to 200 km northward near the city of Reconquista. This impoundment alone will flood nearly half of the Middle Paraná floodplain. A second dam is proposed at Goya, a city just upriver from Reconquista. This impoundment will be similar in area and will inundate the floodplain as far north as Corrientes, thus completing permanent alteration of virtually the entire Middle Paraná floodplain system.

The environmental changes on the Middle Paraná as a result of this project will be extreme, as can be somewhat predicted because of the wholesale replacement of an alluvial floodplain ecosystem with a more homogeneous lentic ecosystem. What cannot be predicted, however, are effects of the project on migratory fish by dam obstruction and the extent to which anoxic conditions develop when the impoundment fills and flooded vegetation begins to decompose.

Technical assistance by the Soviets has resulted in designs for fish elevators that will operate six months of the year to address the first concern. Models are being developed to determine an optimum filling rate so as not to cause extreme depression of water column dissolved oxygen concentrations. Diurnal patterns of discharge downstream from the lower impoundment may result from daily energy requirements. A protocol for research in relation to the Middle Parana project is suggested by Lugo *et al.* (1986).

16.3.4. Global carbon balance

Global considerations of the carbon cycle have been focused primarily on terrestrial and marine ecosystems. Little attention has been given to wetlands, which sequester large amounts of carbon in their organic soils. Armentano (1980) suggested that alteration, drainage, and oxidation of peat in wetlands has created an important source of atmospheric CO_2 while potentially removing an important CO_2 sink. He identified the importance of pocosin areas and large agricultural areas in Florida and California that have been developed on organic substrates.

Armentano *et al.* (1983) expanded this analysis to include tropical wetlands. Apparently, large areas of tropical wetlands are being destroyed annually (Table 1) releasing between 0.074 and 0.18 Gmt/y of carbon to the atmosphere. Much of the released carbon may come from an estimated 23×10^4 ha of mangroves that have been recently converted to fish ponds. Although the data base is small, it appears that as many as 2.7×10^6 ha of tropical wetlands have or are being drained for purposes of increasing agricultural production or for farming peat. The impact of these massive changes is not appreciated at the local level and the global impact is clearly unknown. As yet, no international cooperation appears to be evaluating this potential impact at a global scale.

16.4. Wetland evaluation systems

Brinson *et al.* (1981) categorized proposed evaluation systems as either (1) qualitative; (2) economic, or (3) energy based. Much of the discussion that follows is after this paper. More details on qualitative evaluation systems can be obtained from a review by Leonard *et al.* (1981) that was prepared for the U.S. Water Resources Council. Dwyer *et al.* (1977) reviewed economic methods while H. Odum (1971, 1977) and Kemp *et al.* (1977) described energy analysis procedures.

The utilization of any evaluation system requires the compilation of a large amount of relevant information. Table 2, from Brinson *et al.* (1981), is one example of the type of information required to evaluate riparian wetlands.

There is yet no widely accepted system for evaluating wetlands. Conceptually, the qualitative system developed by Adamus and Stockwell (1983) is appropriate because it considers all important wetland functions. The energy analysis method should, conceptually, also provide the same analytical result as the Adamus system because it integrates almost all important wetland functions. There is yet no economic method that considers all wetland functions. All the methods, unfortunately, suffer from the problem that there is seldom enough ecological data available. More importantly, for most evaluation systems there is no clear understanding of what happens ecologically when any wetland is modified. Consequently, it is difficult to assess the impact of proposed modifications. How can this dilemma be dealt with, and can modeling help?

In this section, examples of the three categories of evaluation systems will be described and critiqued. The purpose is to evaluate the strengths and weaknesses of the systems and comment on their suitability for modeling.

Table 1. Areas of wetland drainage in the tropics.

Location	Area (ha)	Purpose
Malaysia	3.3×10^5	Agriculture
Indonesia	2.0×10^6	Agriculture
East Malaysia, Uganda, Ivory Coast, Sierra Leone, Mozambique, Guyana, Surinam, Jamaica, Trinidad, Cuba, Brazil, Venezuela	3.0×10^5	Agriculture and Peat production

Compiled from Armentano *et al.* (1983).

Table 2. Examples of important functions for riparian forested wetlands.

Hydrologic values
Store flood waters and ameliorate downstream flooding
Serve as areas of aquifer recharge or discharge
Provide year-round source of water in arid climates

Organic productivity values
Have higher primary productivity than surrounding uplands
High secondary productivity supports fisheries, trapping, and hunting
Export organic matter to downstream ecosystems such as lakes and estuaries
Produce high yields of timber and quality lumber

Biotic values
Serve as required habitat for endangered plant and animal species, as refugia for upland species, and as corridors for animal movements
Provide spawning areas for some anadromous and other fish species
Produce organic matter from riparian vegetation for aquatic food chains in small streams

Biogeochemical values
Have high capacity to recycle nutrients; usually accumulate nitrogen and phosphorus
Sequester heavy metals and some poisonous chemicals in anaerobic soil zones, clays and humic matter
Provide buffer zones for maintaining water quality
Accumulate organic matter and thus provide sink for atmospheric CO_2

Geomorphic values
Contribute to landscape diversity
Provide areas of sedimentation for building soils
Have topographic relief that is maintained by stream meandering

Other values
Importance as natural heritage, particularly when scarce
Representative of personal intangible values
Corridors for navigation, highways, and railways
Used as sites for impoundments for recreation, water supply, and electrical generation
Location for recreation and relaxation
Natural laboratories for teaching and research
Locations for construction activities and waste disposal
Rich soils for agriculture and sites for aquaculture

(From Lugo and Brinson 1978 and Brinson *et al.* 1981).

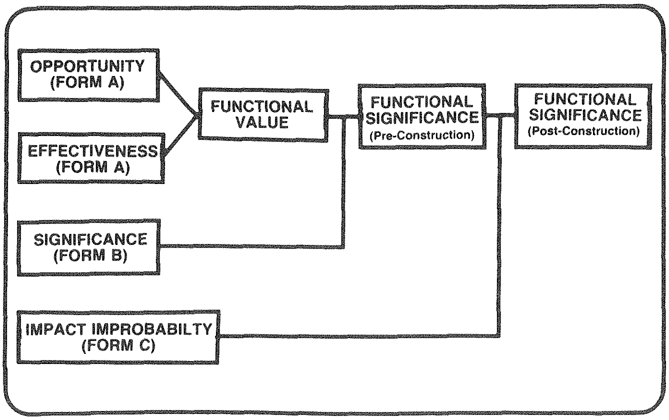

Fig. 2. Flow diagram for the system developed by Adamus (1983) for wetland evaluation.

16.4.1. Qualitative methods

16.4.1.1. Description

This approach requires compilation of as much qualitative information as possible about functional values of a particular wetland and its basin. The information is then ranked and an overall ranking of the wetland made. The rankings of each wetland functional value and overall ranking would normally be described as high, moderate, or low. Advantages of these types of evaluation systems are that they can often be used by inexpert individuals, and the evaluation process is normally not expensive to conduct. Disadvantages are that the data base acquired is so meager that the high, moderate or low rankings are often only guesses. Also, most proposed qualitative methods deal only with limited issues such as habitat functions, especially as they relate to wildlife (Fried 1974 and 1981; Larson 1976; Golet 1979; Schamberger *et al.* 1979; Schamberger and Kumpf 1980; U.S. Army Engineer Division, Lower Mississippi Valley 1980; McCormick and Somes 1982). The system proposed by Larson (1976) also considers hydrologic parameters, as do the systems proposed by Reppert *et al.* (1979) and Schuldinger *et al.* (1979a, b).

The most exhaustive qualitative system yet proposed to evaluate wetlands is that of Adamus (1983). This is large and rather complex, and the two publications that describe the system and hypotheses behind it should be consulted for details (Adamus 1983; Adamus and Stockwell 1983). The proposed method consists of three procedures and is based on the assumption that wetland value has three major components: opportunity, effectiveness, and significance (Fig. 2). In the first two procedures, the ten wetland functions listed earlier (Section 16.2.) are considered. The user determines whether each function can be fulfilled in the wetland being analyzed (Opportunity in Fig. 2). In the second step (Effectiveness in Fig. 2), the user evaluates whether or not the potential opportunity of each of the ten functions is being realized. A third evaluation (Significance in Fig. 2) is then completed. Using the functional and significance information, an evaluator can rate the overall functional significance of each wetland function (Fig. 2). Interpretation keys are provided for this purpose. The third procedure compares mitigation alternatives when the first two procedures have been completed and when the evaluation requires additional cost estimates of alternative management techniques.

16.4.1.2. Critique

The Adamus system was recently reviewed by a group of wetland experts who evaluated not only the procedures but the assumptions behind them (Sather and Stuber 1984). The group

Table 3. Wetland red flag features identified by Larson (1976).

1. Presence of rare, restricted, endemic, or relict flora or fauna.
2. Presence of flora of unusually high visual quality and infrequent occurrence.
3. Presence of flora and fauna at, or very near, the limits of their range.
4. The juxtaposition, in sequence, of several seral stages of hydrarch succession.
5. High production of native waterfowl species.
6. Use by great numbers of migratory waterfowl, shorebirds, marsh birds, or wading birds.
7. Presence of or association with outstanding or uncommon geomorphological features.
8. Availability of reliable scientific information concerning geological, biological, or archaeological history.
9. The known presence of outstanding archaeological evidence.
10. Wetlands that are relatively scarce in a given physiographic region or that provide distinct visual contrast.
11. Wetlands that are integral links in a system of waterways or are so large that they dominate the landscape of a region.

agreed that the system would need to be extensively refined before it would be generally applicable. It also agreed that the effort was worthwhile since it is the most complete system available in terms of considering all important wetland functions in a standard evaluation procedure.

The U.S. Department of Transportation has evaluated the Adamus procedure and trains employees in its use. No other qualitative system reviewed by Leonard *et al.* (1981) is as complete, and most are limited to one or only a few of the Adamus functions.

Schamberger *et al.* (1979) discuss the U.S. Fish and Wildlife Service Habitat Evaluation Procedure (HEP) and give an example of its evaluation. This system, limited to evaluating wetlands as wildlife habitat, seeks to integrate habitat quality and quantity into a single number called the Habitat Suitability Index (HSI). The weakness of this system is that it only evaluates food chain and habitat support for wildlife functions, it does not compare the value of the area as habitat with its value for other uses. In an analysis of 20 evaluation systems, Leonard *et al.* (1981) suggested that the HEP system would be the recommended one when only habitat related functions are being considered.

Of the 20 systems evaluated by Leonard *et al.* (1981), only those by Reppert *et al.* (1979), U.S. Army Engineers Division, New England (1972), and U.S. Department of Agriculture (1978) addressed the five functions Leonard *et al.* considered to be important (habitat, hydrology, recreation, agriculture/silviculture, heritage). All three systems require a large amount of information, and the systems by Reppert *et al.* and the U.S. Army Engineers' New England Division (1972) require that decisions on the values of most functions be made either by an interdisciplinary team or by an evaluation based on professional judgment.

Unlike the HEP system, these three systems provide no opportunity for judging value impacts. The HEP system does provide a mechanism for evaluating impacts on wildlife. None of these systems is suitable for model development, although most can be computerized for easier application.

There is another factor in most qualitative systems that the majority of wetland scientists believe is necessary. This is referred to as the 'red flag feature' (Larson 1976; Leonard *et al.* 1981). Red flag features are criteria that identify wetlands that should be preserved because they have some outstanding value. Examples of red flag features would be the presence of rare or endangered species, or of unique geological, biological, or archaeological features. Table 3 lists some red flag features considered to be important by Larson (1976). This system will be further discussed in the next section because it utilizes qualitative information to generate economic values.

16.4.2. Economic methods

16.4.2.1. Description

It is reasonable to attempt to evaluate wetlands using economic models because many social decisions are based on some type of cost–benefit analysis. In the USA, cost–benefit analyses are often required by law and are used to determine whether projects should or should not be performed. Cost–benefit analyses can be based on both qualitative and quantitative information, but most often are based on a monetary analysis of all costs and benefits (Rosen 1977; Diedrick 1981).

While it is almost universally agreed that many wetland ecosystems have economic values and that ecosystems provide goods and services to man, there is very little agreement on how non-market intangible values should be incorporated into evaluation systems. The principal problem is that goods and services that do not have readily discernible market value are difficult to accommodate with economic techniques. Attempts have been made, however, to develop economic models that include the non-market goods and services (Gosselink *et al.* 1974; Gupta and Foster 1975; Kemp *et al.* 1977; Shabman *et al.* 1979). The paper by Gosselink *et al.* (1974) started an ongoing debate between ecologists and economists that continues today. The interested reader is referred to a series of articles on the subject (Gosselink *et al.* 1974; Shabman and Batie 1978, 1979, 1980, 1981; E. Odum 1979a; H. Odum 1979, Batie and Shabman 1982). That no suitable economic method yet exists is demonstrated by the fact that Leonard *et al.* (1981) excluded all economically based models from their review of wetland assessment methodologies. Similarly, economic models were not incorporated into the system of Adamus (1983). Nevertheless, economic models are being developed and have been used and tested (Gupta 1972; Gupta and Foster 1975; Geis and Kee 1977; Batie and Shabman 1982).

As an example, the system developed by Gupta and Foster (1975) for evaluating freshwater wetlands in Massachusetts will be described. The method is a type of cost–benefit analysis that compares social opportunity costs of permit denial with benefits of wildlife production, visual–cultural values, water supply, and flood control. Costs of wildlife and visual-cultural benefits are calculated according to the costs of wetland acquisition. Water supply costs are based on comparing the cost of supplying water from a wetland to the cost of water from alternative sources. Flood control benefits were based on a study done by the U.S. Army Corps of Engineers. Table 4 is an example from Larson (1976) that shows the high, medium, and low per acre values of these four benefits for Massachusetts wetlands. Table 5, from the same publication, contains a matrix of benefits for the four functions and the two estimates of the total per acre values for each combination. Clearly, having a medium to high water supply benefit makes a wetland valuable in the Gupta and Foster system.

16.4.2.2. Critique

The most widely applied economic model is the one proposed by Gosselink *et al.* (1974). As discussed earlier, that model has been widely used, but it has also been widely criticized. The most ardent critics have been Shabman and Batie, who published a conceptual model for estimating economic values. The model (Batie and Shabman 1982), which includes non-market values, is unfortunately still only conceptual and apparently has not yet been applied to any wetland system. Other economic models are based on replacement costs (Wharton 1970; Mitsch *et al.* 1979), estimates of economic values of fish, wildlife, and recreation (Raphael and Jaworski 1979), trade-off prices (Gupta and Foster 1975; Shabman *et al.* 1979), the cost of converting wetlands to agricultural areas (Brown 1976; Leitch 1981), or the cost of purchasing and constructing wetlands (Tilton *et al.* 1978). Harwood and McMullen (1981); Lindenmuth and Vasievich (1981); Monschein (1981), and Postel (1981) contain examples of economic analyses related to development of pocosin wetlands.

The system developed by Larson (1976) is interesting because it is a three level model that combines a number of wetland functions to estimate an economic value (Fig. 3 and Tables 4

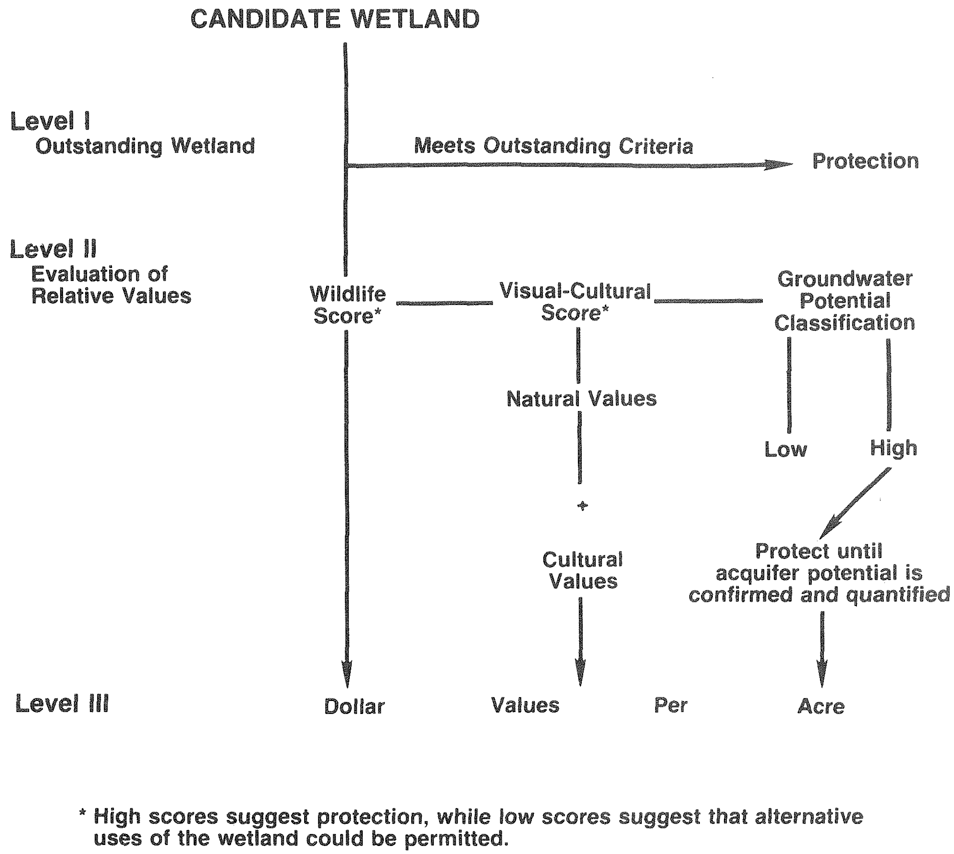

Fig. 3. General outline of the three level model for freshwater wetland evaluation. (From Larson 1976).

Table 4. High, medium, and low U.S. dollar values for four wetland benefits (Larson 1976).

Type of benefit	Annual dollar values of benefits per acre of wetland		
	High	Medium	Low
Wildlife	70	35	10
Visual–cultural	270	135	20
Water supply	2,800	1,400	400
Flood control	80	40	10

1 acre = 0.4 hectares.

and 5). Initially, wetlands are evaluated for red flag features (Fig. 3). Those with outstanding features are recommended for protection. In the second phase, wildlife, visual–cultural, and ground water functions are evaluated. Finally, economic values are assessed for high, medium, and low benefits (Table 4). The model was used to generate a matrix of capitalized per acre values for Massachusetts wetlands (Table 5). As such, it was designed primarily for use in evaluating wetlands in existing conditions, and is not totally appropriate for evaluating potential impacts.

Table 5. Matrix of capitalized values for Massachusetts wetlands that have various combinations of high, medium, and low values for four wetland benefits (Larson 1976).

Nature of benefits				Capitalized values (U.S. dollars) of total wetland benefits per acre at:	
Wildlife	Visual–cultural	Water supply	Flood control	5.375%	7%
High	High	High	High	$59,900	$46,000
High	High	Medium	High	33,800	26,000
High	High	Low	High	15,200	11,700
High	High	None	High	7,800	6,000
Medium	Medium	None	Medium	3,900	3,000
Low	Low	None	None	700	500
High	Low	None	None	1,800	1,400
Low	High	None	Low	5,300	4,100
Low	Low	None	High	2,200	1,700
Low	Low	High	Low	53,000	40,700
Low	Low	Low	Low	8,300	6,400

16.4.3. Energy analysis

16.4.3.1. Description

Energy analysis for evaluating wetlands has undergone considerable development and modification during the past decade (Gilliland 1975; Farber and Costanza 1987; H. Odum, 1971, 1978, 1982; H. Odum *et al.* 1981a, b).

Basic to the energy analysis method is that values in both the natural environment and human systems can be evaluated by the common denominator of energy units (Calories, kcal). Because energy flows are governed by the laws of nature, the approach has the conceptual appeal of being less subjective than those that are based on unpredictable consumer demands or on resources only when they are scarce or limiting. In any case, flows of energy at different levels of a system have different qualities associated with them. It is accounting for this quality that allows one to calculate 'embodied energy' or the total value of the system.

In order to arrive at a computation of energy quality, energy transformations must be employed that adjust a given flow of energy into its equivalent value of solar energy (as kcal), the standard used by convention. Because solar energy is a dilute form from which all other energy flows emanate, it tends to have the lowest quality. For example, 1 kcal of energy photochemically transformed into gross primary production represents the manifestation of 200 kcal of solar energy (a global average that assumes 0.5% conversion efficiency of incident radiation into gross primary production). Thus, phytoplankton metabolites are 200 times more valuable in performing work in the ecosystem than the original solar radiation. Efficiency transformations in food chains can be used to calculate even higher solar equivalents for higher trophic levels, although alternate approaches may also take into account the embodied energy required to evolve a species. Certain limiting nutrients such as phosphorus and nitrogen, which can amplify lower quality energy flows, may in fact have enormously high energy qualities. Procedures have now been outlined for calculating these transformation ratios (Odum *et al.* 1981a, 1981b; H. Odum 1982).

Likewise, products of human society have embodied energy, and transformation ratios can similarly be used to account for their quality. In this way, the human and natural environmental sectors of a system with finite boundaries (across which inflows and outflows are taken into account) can be assigned a total embodied energy flow simply by summing the qualities of its parts that are all expressed in equivalent (solar) quality terms. If there is a need to have this expressed in economic terms, monetary values can be assigned. This requires an additional calculation using a ratio of embodied energy of a country to total money flow (Gross National Product).

For evaluating the impact effects, the methodology can be applied by comparing various alternatives to a given project (including no action). Thus, an energy analysis could be done on any number of project alternatives to give decision makers an array of choices and information.

Energy analysis has been used to consider issues ranging from regional hydrologic problems (Littlejohn 1977) and assessment of the regional implications of converting wetlands into areas for cattle production (Day *et al.* 1977), to cost–benefit analysis of site development for power plants (Kemp *et al.* 1977). Although this approach has been useful in many instances, its tenets are not yet widely accepted.

16.4.3.2. Critique

The foregoing described a few of the methodological steps in energy analysis, but gave little information on assumptions, premises, and uncertainties. The method is not without its critics, as exemplified by the debate discussed earlier (*cf.* E. Odum 1979a *vs.* Shabman and Batie 1978). However, Lavine and Butler (1981) provide a critique of the energy analysis approach from a refreshingly neutral perspective. The major points of their analysis give further insight into the method and they fall into the following categories:

1. Using power as an indicator of value. Here they examine the basic premise that "systems with the most energy resources can use them to meet all other contingencies so as to survive competition and maximize vitality of the combined economies of man and nature". This premise holds that maximum power theory (Odum and Pinkerton 1955) can explain system survival. This may present a problem when deciding what subsystem should survive at the expense of others, *i.e.*, the boundary determination mentioned earlier.
2. Accuracy of energy transformation ratios. They state that embodied energy, as an indicator of the amount of energy used to generate a specific energy flow, is generally accepted by all practitioners of energy analysis. Because the discipline of embodied energy analysis is relatively new, there remains ample opportunity for improving the resolution and accuracy of these ratios.
3. What should and should not be counted as energy contributions. There is some controversy among energy analysts as to whether to include renewable energy flows, some nonrenewable energy flows embodied in non-fuel minerals, and energy flows used in the support of human labor.
4. Double counting possibilities. Unknown interdependencies of energy flows are likely to overestimate embodied energy flow.
5. Comparison of monetary and embodied energy indicators. By using GNP (gross national product) as an indicator at final monetary demand levels in economic systems (generally valid), certain projects that focus principally on intermediate or primary demand levels may be misrepresented by the GNP: embodied energy ratio.
6. Assumptions concerning discounting in energy analysis. There is no general consensus on whether energy analysis implicity addresses uncertainties surrounding the discount issue that commonly surfaces in benefit–cost economic analyses. However, energy analysis appears to be more robust in that it can accommodate what is equivalent to a negative discount rate (*i.e.*, decreasing energy values).

One of the clear advantages of energy analysis over other purely economic methods is its ability to evaluate those resources that are considered external in economic approaches. While it has been argued that externalities can be dealt with by other techniques (Batie and Shabman 1982), shadow pricing systems such as willingness-to-pay are somewhat contrived and are dependent on present-day consumer perceptions of resource scarcity.

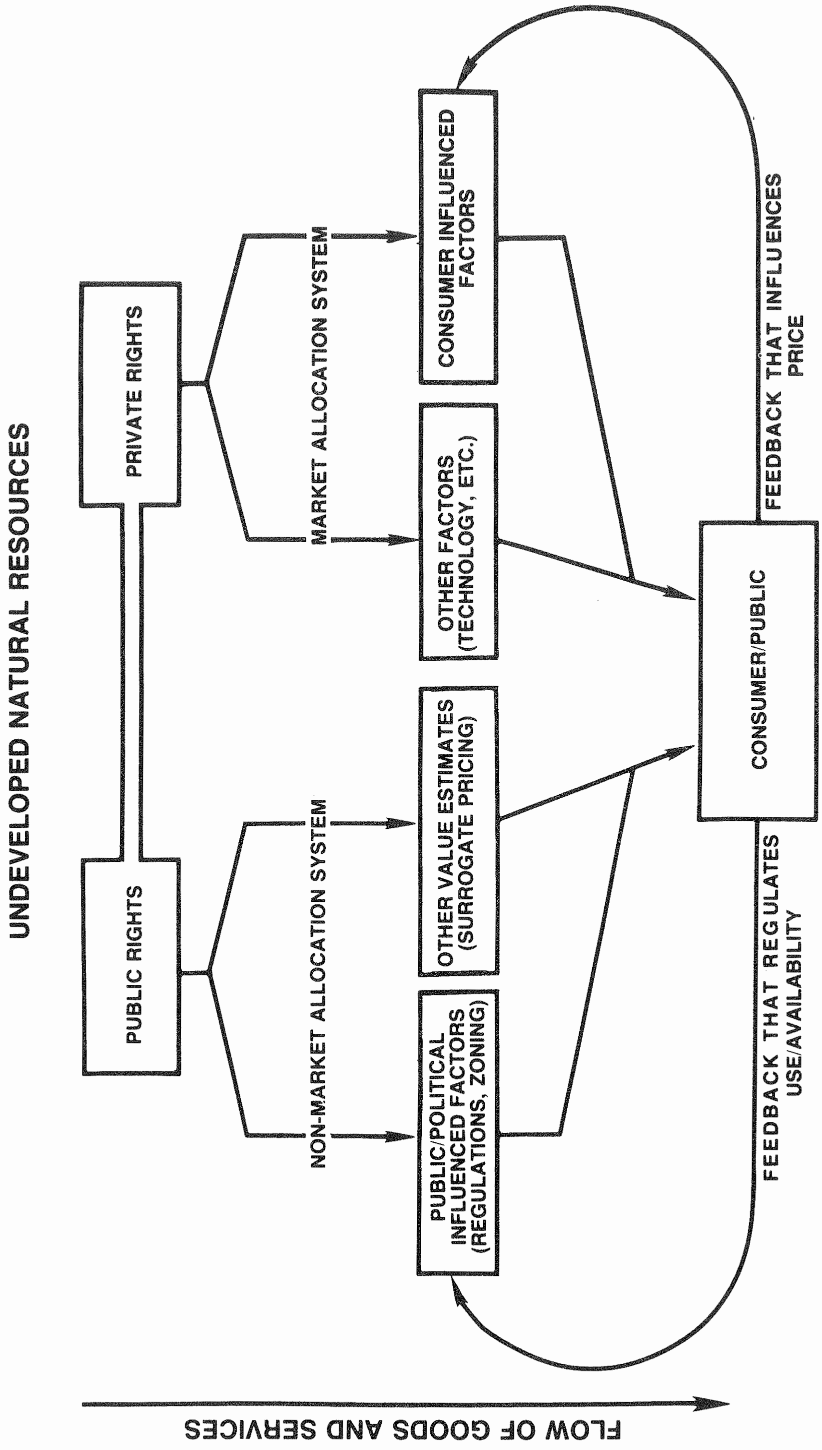

Fig. 4. Interaction between public and private rights in regulating the flows of goods and services from undeveloped natural resources.

16.5. Applications of methods in resource allocation decisions

The basic value of the methodologies described above is their application in resource allocation decisions. In the case of wetlands, options may include whether to drain for agricultural production (Day *et al.* 1977); impound for hydroelectric generation, flood control, irrigation, or recreation; or simply allow the wetland to function naturally and provide services without alteration. If the resource were small and not scarce, there would be no problem in meeting the needs of all consumers and alternate users, and thus no need to consider the question further. However, usually situations are confronted that require large-scale modifications to make resources available for some purposes (and people) while at the same time making them less available for others.

Resource ownership must be considered first as it influences the institutional approach to resource allocation. Scarce resources may be held by private owners or the public, the former operating by the market system through exchanges of goods and services between the owner and the demands of consumers (which establish prices). Public ownership requires institutions such as government agencies to make decisions about how, when, and to whom public resources are allocated.

Ownership boundaries are seldom fixed (or even clear in many environmental issues), and ownership may be shifted from private to public when markets fail to allocate resources efficiently. Government may also affect resource allocation more indirectly through taxes, subsidies, and other incentives or disincentives. However, market failure is not by itself justification for public intervention because non-market systems also fail (Wolf 1979). The issue is not whether public intervention (non-market) is appropriate or not, but rather what appropriate form of action will lead to the best use of the resources in question.

The procedures for wetland resource analysis described above are designed to provide the critical information needed to assess value impacts and make decisions about optimum resource allocation. Figure 4 illustrates where in the economic process this information can be used in controlling flows of goods and services to consumers. In capitalistic economies, natural resources are predominately in the domain of private rights, and thus controlled by market allocation. In socialistic economies, public ownership predominates, and thus allocations are made by governmental institutions.

There is a tendency, however, for wetland functions to be perceived as within the jurisdiction of the public in otherwise predominately capitalistic economies, although exceptions exist. The justification for this is that private wetland owners are seldom given incentives to maintain wetland benefits (*e.g.*, waste processing, wildlife habitat, fish production, etc.). When the private owner manages for agriculture, this may be perceived as conflicting with the foregoing benefits to the public. Many examples of market failure such as this have precipitated the intervention of public agencies. Until decision makers are given effective tools and methodologies to determine the worth of wetland resources, both capitalistic and socialistic economies are unlikely to recognize their true value, and will continue to misjudge consequences of wetland devastation.

16.6. Conclusions

In North America, several methodologies are available for assessing wetland values, and some of these can also be used for evaluating impacts. There is obviously considerable disagreement about the utility of different types of methodologies, and many public agencies seem primarily interested in qualitative systems. Economic and energy analysis methodologies have, however, been developed to the point where they have been useful in specific instances. Some simulation models have been useful in evaluating development impacts, but most have been developed to assess wetlands under natural conditions (Mitsch *et al.* 1982).

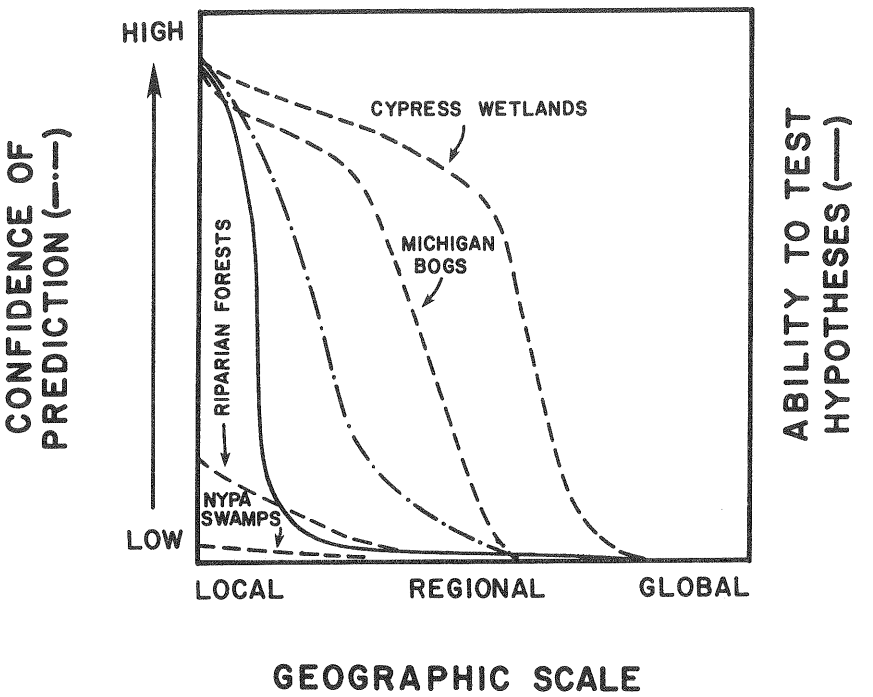

Fig. 5. Conceptualization of the relationship between geographic scale and ability to test hypotheses about wetland functions as well as confidence in predictions about the changes in wetland functions. Four specific types of wetlands are identified to demonstrate situations where confidence in the prediction of consequences of impacts varies from very low to slightly above the ability to actually test hypotheses.

How far advanced is man's ability to perform impact assessments on wetland values? Clearly there does not exist a uniform answer to this question. It seems that progress is less developed than the process of understanding how wetlands function. Even so, knowledge is slight about how wetlands will function in response to proposed modifications. There may in some instances, be suitable information to evaluate a particular wetland, especially if it is a type that has been well studied ecologically. Resolution is lost beyond the scale of the local wetland, however, because of lack in the ability and resources to conduct important large scale research projects. This situation is described in Fig. 5. Ability to test hypotheses beyond the local scale is very limited. Consequently, confidence in model predictions also falls sharply beyond the scale of the local wetland. Little is known, for example, about the ecology of Nypa swamps in Southeast Asia (Gopal *et al.* 1982), and ability to predict development consequences is very limited. Perhaps the only exceptions to this situation are for wetlands in Florida and Louisiana, USA, where large scale landscape changes have provided researchers with opportunities to develop and test large scale models.

In most situations, the reliability of any prediction is primarily a function of how much is known about a particular wetland or type of wetland. Thus, for Florida cypress (*Taxodium*) wetlands and Michigan peat bogs, predictions about responses to wastewater additions are reliable. In other instances little is known about wetland function, and consequently little can be said about what will happen at any geographic scale. Perhaps the best example of this situation exists in the pocosin region of North and South Carolina (C. Richardson 1981; Sharitz and Gibbons 1982; Ash *et al.* 1983). Many local decisions are being made that affect small geographic areas. Collectively, the sum of these small decisions is affecting the region and perhaps even having a significant impact on the global carbon dioxide balance. There are, unfortunately, no models available to adequately assess many questions related to the impact of numerous small scale decisions on pocosins. This example probably represents the state-of-the-art for most types of wetlands. There is a clear need to continue to develop suitable assessment methodologies, especially impacts that accumulate their effects in time and space. In developed countries, red flag features should be the first consideration in evaluating any particular wetland or group of wetlands. In developing countries, models are desperately needed because development decisions are not made at the local level and models may be used to demonstrate the importance of certain wetland areas and provide guidance to assure minimal disruption (Weger and Ellenbrock 1980).

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