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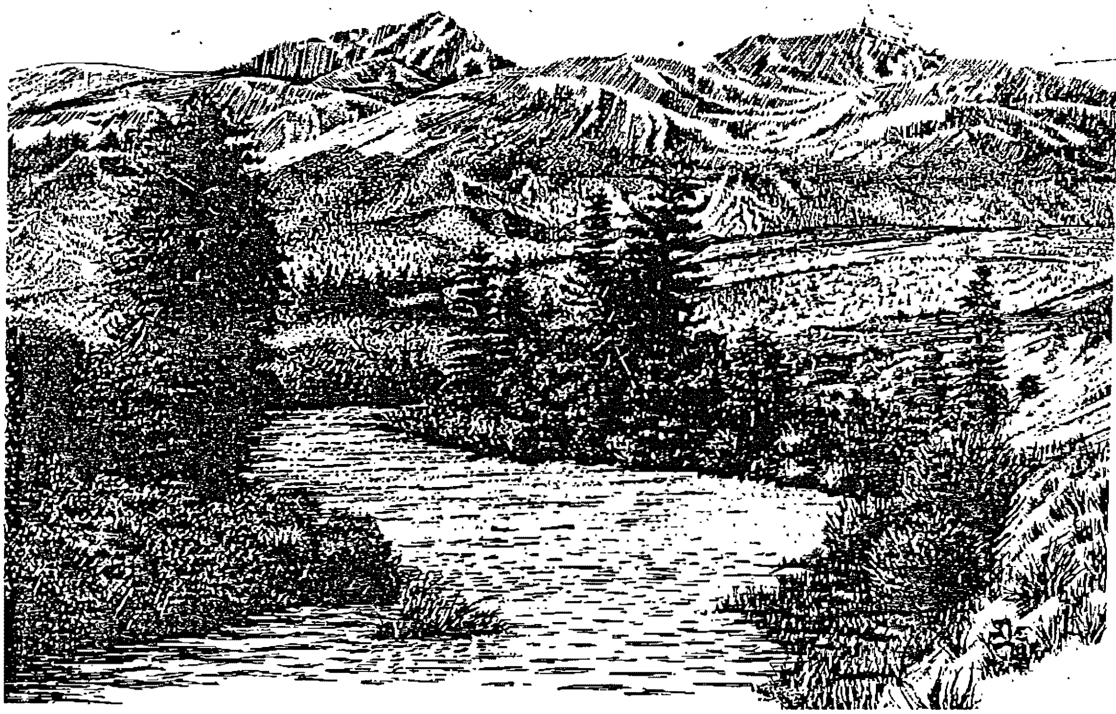
FWS/OBS-81/17
AUGUST 1981

RIPARIAN ECOSYSTEMS: Their Ecology and Status



Fish and Wildlife Service

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UNITED STATES
DEPARTMENT OF THE INTERIOR
FISH AND WILDLIFE SERVICE

EASTERN ENERGY AND LAND USE TEAM
Route 3, Box 44
Kearneysville, West Virginia 25430

September 25, 1981

Dear Colleague:

Enclosed is an Eastern Energy and Land Use Team (EELUT) publication entitled "Riparian Ecosystems: Their Ecology and Status." This report provides a review and synthesis of available information that can be used to document and assess the ecological values of riparian ecosystems. Included are chapters covering: 1) the status of riparian ecosystems; 2) ecological functions and properties of riparian ecosystems; 3) importance of riparian ecosystems to fish and wildlife; and 4) considerations in valuation of riparian ecosystems. This product is intended to serve as a reference document for biologists in Federal and State water resource and fish and wildlife agencies, and for private organizations interested in conservation of riparian resources.

Readers should feel free to contact us for more information and publications pertaining to riparian ecosystems, through:

Information Transfer Center
Eastern Energy and Land Use Team
U.S. Fish and Wildlife Service
Route 3, Box 44
Kearneysville, WV 25430
(FTS) 925-5265 or (304) 725-2061, ext. 5265

Sincerely,

Edgar A. Pash

Edgar A. Pash
Team Leader

Enclosure

FWS/OBS-81/17
September 1981

RIPARIAN ECOSYSTEMS:

THEIR ECOLOGY AND STATUS

by

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National Water Resources Analysis Group
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PREFACE

The purpose of this publication is to document and interpret the information that is available on riparian ecosystems so that the consequences of their alteration and deterioration can be assessed at a national level. The common functional properties of these ecosystems and their attractiveness to wildlife make it possible to address riparian ecosystems as discrete and manageable entities.

The Fish and Wildlife Service has been involved in several efforts that have led to the development of the document. Much of the earlier concern was for the consequences of channelization and other stream alterations on fish and wildlife communities. It was soon recognized that most stream alterations could not be considered separately from changes in floodplain vegetation and animal communities. The growing body of literature on riparian ecosystems suggested a strong interdependency between stream and floodplain processes.

A national symposium held in 1978 on "Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems" was an attempt to focus attention on the research of individuals that were working largely on ecosystems associated with streams.[1] The following year a workshop on riparian ecosystems in Harpers Ferry produced a number of strategies and alternatives for riparian ecosystem protection and enhancement in which the Fish and Wildlife Service could potentially participate.[2] This more recent effort is a "second generation" state-of-the-art whereby we summarize and synthesize what is known about riparian ecosystem function, values, and management.

This publication is intended to provide a geographically balanced treatment of technical information on riparian ecosystems from a nationwide perspective. By focusing on the common properties of these ecosystems, recommendations and decisions that affect their management and protection should be simplified. The manuscript is oriented to provide assistance to decisionmakers involved in ecosystem management who must utilize ecological principles and information.

Any suggestions or questions regarding this report should be directed to:

Eastern Energy and Land Use Team
Route 3, Box 44
Kearneysville, WV 25430

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[1]Johnson, R. R. and J. F. McCormick (tech. coord.). 1978. Strategies for protection and management of floodplain wetlands and other riparian ecosystems. USDA Forest Service, Gen. Tech. Rep. WO-12. Washington, D.C. 410 pp.
[2]Warner, R. E. 1979. Proceedings of a workshop on fish and wildlife resource needs in riparian ecosystems. Eastern Energy and Land Use Team, U.S. Fish and Wildlife Service, Kearneysville, WV. 53 pp.

EXECUTIVE SUMMARY

This report describes the functions, values, and management of riverine floodplain and streambank ecosystems, henceforth called riparian ecosystems. The report is composed of sections on the status of riparian ecosystems, their ecological function and properties, wildlife resources, and valuation considerations. This brief synopsis of the four sections provides an overview of the material covered in each.

STATUS OF RIPARIAN ECOSYSTEMS IN THE UNITED STATES

In the absence of a comprehensive inventory of riparian ecosystems in the U.S.A., existing resource inventories provide only a rough indication of the extent and distribution of these ecosystems. However, when taken together, the data give a great deal of insight on the amount of riparian ecosystem in existence, the quantity of natural area lost to a variety of other uses, and the nature of alterations.

One liberal estimate of the amount of land subjected to flooding (100 year floodplain) and thus potentially supporting riparian ecosystems is 121 million acres, or 6% of the land in the U.S.A. (excluding Alaska). In reality, much less exists in a natural or seminatural forested condition, and a conservative estimate is 23 million acres. From other sources, we estimate that approximately 70% of the original floodplain forest has been converted to urban and cultivated agricultural land uses.

Case histories of riparian ecosystem status and condition show large differences in loss from place to place, but as much as 95% loss of natural vegetation has been reported in some areas. Examples for the lower Mississippi, Colorado, Sacramento, and Missouri Rivers have been particularly well documented, and, in comparison with estimates of loss of natural vegetation in uplands, put riparian lands in the category of the most severely altered ecosystems in the U.S.A.

Along with data on losses in natural floodplain forests, the magnitude of stream alteration provides an independent assessment of changing condition of riparian ecosystems. About 60% of the major stream segments have been judged unsuitable for inclusion in the National Wild and Scenic Rivers System because of water resource or other cultural developments within riparian corridors. Numerous examples exist for losses in stream length due to channel realignment and alteration. Losses in surface area of riparian ecosystem undoubtedly occur in larger proportion than loss in stream length because large amounts of drainage and forest clearing usually accompany relatively small reductions in stream length. Impoundments have also inundated significant areas of riparian vegetation, and the downstream effects of modified streamflow on riparian ecosystem function have seldom been documented.

Alteration and loss of natural riparian ecosystems, as compared with upland ecosystems, are of particular concern because of the greater magnitude of modification required for conversion to other uses. The potential for restoration is lower because drainage precludes most other goods and services to society that flood-dependent riparian ecosystems provide.

FUNCTIONS AND PROPERTIES OF RIPARIAN ECOSYSTEMS

Over geologic time periods, streams undergo phases of erosive downcutting and alluvial deposition. At the same time stream channels migrate back and forth across floodplains, a process which results in a continual replacement and displacement of the plant and animal communities. In this way a stream is responsible for "organizing" the floodplain into a variety of diverse communities, many of which are controlled by the depth, duration, and frequency of inundation.

Flooding and flow water are also responsible for depositing and eroding sediments. Both the suspended material and the water that carries it represent supplies of materials from sources outside the floodplain. Upland ecosystems lack a similar lateral transport system; consequently this is one of the fundamental differences between upland and riparian ecosystems. Both the abundance of water and nutrient supply are partially responsible for maintaining the productivity and vitality of riparian ecosystems.

Primary productivity may be regarded as an indicator of the vitality of an ecosystem. Not only does primary productivity initiate organic energy flow for food webs, but another of its fundamental functions is to maintain the structural integrity of the ecosystem. Studies done on floodplain forests of the Southeast show that they are among the most productive ecosystems in the nation. Riverine wetlands also export a disproportionate amount of organic matter as compared with an equivalent area of upland ecosystem. Thus they augment the amount of energy and structural carbon that downstream aquatic ecosystems, particularly estuaries, receive from continental runoff. Instream communities also are highly dependent on leaf litter from streamside forests for maintaining metabolism and ecosystem structure.

Differences in nutrient cycling between floodplains and upland ecosystems are related to (1) the influence that flooding and an "aquatic" phase has on restricting oxygen availability to soils and sediments, hence altering the metabolic pathways of microbial communities, and (2) the aqueous transport system that provides pathways of exchange through lateral imports, sedimentation, and exports of nutrients. Most nutrient cycling studies conducted in southeastern floodplain forests suggest a high capacity to absorb and recycle nutrients. In arid riparian ecosystems, the quantity of water, rather than its quality, is an overriding factor in ecosystem processes. The potential for floodplains to have an influence on the nutrient status of floodwaters depends partly on the length of time and the quantity of water and nutrients that come in contact with the floodplain.

It should be possible to predict the severity of damage that a particular alteration will have on normal ecosystem processes based on an understanding of natural ecosystem function. Alterations of ecosystems can be categorized as changes in geomorphic processes and water delivery patterns, physiological stress, and biomass removal. Stream channelization, containment of stream flow and channel constriction, impoundments and diversions, introduction of toxins, grazing by livestock, timber harvest, and hunting and fishing correspond with one or more of the three alteration categories.

From this analysis it is possible to predict the consequences of the seemingly diverse sources of intrusions into riparian ecosystems. If goals of mitigation are to restore the multiple services that these ecosystems provide in their natural condition, some alterations can be mitigated and others clearly cannot. If the principal sources of energy and material continue to be supplied to the system, there is a high probability of recovery. If these sources are blocked or diverted, mitigation to reverse the damage can occur only after great investments of time, energy, and money.

FISH AND WILDLIFE RESOURCES IN RIPARIAN ECOSYSTEMS

Many of the attributes of riparian ecosystems that make them attractive to humans are also responsible for the success and maintenance of wildlife populations. These characteristics include the presence of flowing water, moist and nutrient rich soils, relatively high plant productivity, and corridors for migration and travel. The structural complexity of these ecosystems, particularly in comparison with uplands in arid climates, provides many habitat requirements and adds to the landscape diversity of the regional geography.

During the past decade, a large number of studies have documented that riparian ecosystems unquestionably provide essential habitat requirements for a large diversity of vertebrate species. More migratory and nesting species of birds have a higher affinity for riparian ecosystems than they do for upland ecosystems. Although catastrophic flooding may temporarily reduce the abundance of "terrestrial" vertebrates, these species are adapted to rapid recolonization after flood conditions subside. In fact, certain fish populations are augmented by enormous increases in feeding area that floodplain inundation provides, in addition to the seasonal supply of leaf fall into the water surface of the stream channel under non-flooding conditions.

The reasons for dependence on and affinity for riparian ecosystems by such a large and disproportionate number of vertebrates are due to a multiplicity of factors. The presence of flowing water, high plant productivity, and nutrient-rich conditions have already been mentioned as contributing factors. Perhaps of more fundamental importance, riparian and floodplain ecosystems represent a combination of aquatic and terrestrial ecosystems that have somewhat separate spatial and temporal dimensions. Habitat features change dramatically with only small topographic differences, such as the gradient from an open water stream channel to a dense gallery forest. The duration and timing of flooding superimposes a seasonal dimension on these gradients. For these spatial and temporal dimensions to be maintained, it is essential that the changing geomorphic forces that drive riparian ecosystems be allowed to organize and reorganize the plant and animal communities.

THE VALUE OF RIPARIAN ECOSYSTEMS: INSTITUTIONAL AND METHODOLOGICAL CONSIDERATIONS

Allocating land and water in riparian ecosystems among various uses and assessing the relative social values of these competing uses are issues of immediate and major concern. Riparian systems are generally considered quite valuable because of their ecological values and natural service functions. However, institutional mechanisms for allocating resources such as land and water are designed to serve perceived human wants and needs. Therefore, the way in which private and public institutions allocate natural resources will determine whether riparian systems are left relatively undisturbed for wildlife, timber, specific kinds of recreation, natural flood storage, water quality enhancement, and groundwater recharge; or whether they are altered for agricultural production, navigation benefits, flood protection, or commercial development. Central to this process are the forces and incentives which drive resource allocation in one direction or another and the manner in which preferences and values are weighed in decisionmaking processes which directly affect the resources.

The causes of land use patterns in riparian systems appear to be very complex. In some respects they are. Soybean demand, grazing rights on public land, tax laws affecting property and estates, and public flood control projects are but a few factors which appear to affect land and water use in floodplain ecosystems. However, there are broader and, in some respects, more meaningful categories:

1. Market forces affecting private investment patterns (consumer demand for specific goods and services);

2. Political forces affecting private investment (world trade policies, regional economic development, public subsidies); and

3. Institutional factors affecting private and public decisionmaking which include:

a. Market decisionmaking (property rights specifications, failure of markets to capture costs and benefits of private transactions, information problems), and

b. Nonmarket (government) institutions and activities (taxes, subsidies, regulations which affect the incentives of private decisionmakers to engage in particular activities, and publicly conducted and assisted projects).

Having analyzed these categories of factors, one can focus on specific policies, programs, and decisions which determine the fate of riparian systems.

Another distinct aspect of economic analysis of resource allocation in riparian systems concerns valuation. How does one value the various competing uses of riparian systems? This problem arises most frequently in the context of public decisionmaking processes whereby public officials must weigh the value of one land use versus another (i.e., through permitting-licensing activities, zoning decisions, funding of public projects, etc.). Typically, public decisionmakers are confronted with two very different kinds of information regarding values: ecological and economic. The decisionmaker is faced with the dilemma of evaluating noncomparable values before reaching a decision. However, ecological values have economic significance. For example, if riparian system alteration were to result in lost natural flood storage, lower water quality, and fewer wildlife resources, what is the "cost" of these foregone opportunities? Since we do not pay landowners to maintain land for these purposes, it is difficult to assess society's demand for them as expressed through market prices (reflecting aggregate willingness-to-pay). This necessitates use of some surrogate value. We provide a brief review of the approaches to natural resource valuation and a critique of each of the methodologies.

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ACKNOWLEDGEMENTS

The preparation of this report would not have been possible without the cooperation and assistance of a number of dedicated individuals. Discussions and visits with Roy Johnson, Richard Warner, and Charles Wharton were of immense value. Review comments by numerous other persons from Federal agencies have contributed significantly to this effort. East Carolina University provided support for the senior author during the final stages of report preparation.

Much of the impetus for organizing this work on riparian ecosystems was provided by Charles Segelquist. His sustained encouragement to the authors and perseverance in riparian affairs are largely responsible for completion of this report. The various chapters were prepared by the following individuals: Mark Brinson prepared Chapters 1 and 3 and was responsible for final editing and organization; Bryan Swift wrote Chapters 2 and 4 with the assistance of John Barclay; Reuben Plantico prepared the material in Chapter 4 with the assistance of Mark Brinson. Karen Byrd typed several drafts of each of these chapters and the final report.

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CHAPTER ONE

INTRODUCTION

This document addresses the functions, values, and management of riverine floodplain and streambank ecosystems, hereafter called riparian ecosystems. An abundance of water and rich alluvial soils are among the more important attributes that distinguish these ecosystems from uplands. River corridors represent lines of convergence where the energy of flowing water has delivered and concentrated erodible materials from diffuse sources in the landscape. Because of these special attributes and life-supporting features, human society has long perceived their usefulness as sites for urban settlements, as conduits for transportation, and as a source for harvestable products such as timber, crops, and wildlife.

In comparison with average stream flow, catastrophic episodes of stream flooding are more important in molding and shaping the landscape through erosion, sedimentation, alteration of river courses, and rejuvenation of vegetation. A major flood may occur during any given year, and the best we can do is predict the probability of its reaching a particular height and carrying a given quantity of materials. Because of this uncertainty humankind often has found itself poorly adapted to utilizing the resources and benefits of these ecosystems.

Depending on the form of the riparian ecosystem and the particular locality within it, water levels may range from prolonged seasonal inundation of floodplains to periodic rises in the subsurface ground water of streamside forests. When human intrusions alter the natural temporal and spatial pattern of water flow, the essential features upon which riparian ecosystems depend are threatened. By the same token, alteration of these ecosystems may prevent them from providing valuable life support services to society such as maintenance of water quality, flood water storage, and the production of quality timber, fish, and wildlife.

This is not to suggest that riparian ecosystems are immune to management. On the contrary, judicious management may be the preferred alternative, particularly in the context of the numerous alterations that have already occurred in many watersheds. Distinctions need to be made between the types of alterations that can be tolerated and the degree to which alterations can be made without threatening the carrying capacity of riparian ecosystems for providing values and services to society. In order to be in a position to make riparian management decisions, it is essential that we understand the function and importance of the flows of energy and materials within and through riparian ecosystems. This is a necessary prelude to establishing the values of the services that riparian ecosystems provide society.

SCOPE

The riparian ecosystems discussed in this report are those associated with streams and rivers. We include the full continuum from intermittent headwater streams with negligible floodplains to broad meandering rivers, but exclude flooded coastal features such as salt marshes and mangrove swamps. The main focus is on floodplain and streambank plant and animal communities which are affected by the stream through additional water supply, flooding, or lateral transport of nutrients and sediments. It is recognized that riparian ecosystems also may have profound effects on streams. The magnitude of the interaction will be somewhat site specific and depends partly on relative sizes of each. In general, streambank forests will influence to a greater extent the ecological processes in small streams than in large streams. Likewise, streams with high discharge usually will have a greater influence on riparian forests than small ones, particularly in areas of the floodplain that are frequently inundated.

We recognize that "riparian zones" are not restricted to riverine ecosystems, and that the term is frequently applied to the more robust vegetation associated with seeps, springs, meadows, bogs, margins of ponds and lakes, and a number of other "wet" features found in the predominately arid regions of western U.S.A. Although many of these wetter areas have important hydrologic functions and unquestionable wildlife values (e.g., playa lakes), from a functional and management standpoint, they probably have more in common with non-flowing water systems in more humid regions, such as certain bog depressions, lakes, prairie pothole marshes, limestone sinks, and Carolina bays. Because these predominately stillwater systems differ from riverine systems, their management and values should be approached with fundamental hydrologic and geologic differences in mind. The alterations to which riverine and stillwater systems are subjected also differ in many instances.

Riverine riparian ecosystems overlap a great deal with some of the ecosystem types in the wetland classification system of the Fish and Wildlife Service (Cowardin et al. 1979). However, we discuss some plant and animal communities that are not included in the wetland classification system. This encompasses areas where streams have the effect of supplying water, sediments, and nutrients that would otherwise not be available under "upland" conditions. Often these lowland areas are clearly not areas that are "flooded or saturated at some time <u>each year</u>" (Cowardin et al. 1979, p. 4) nor do they necessarily have "predominately hydrophytic cover" (ibid., p. 3). In addition to the physiological adaptations to flooding, drought may be an important selective force for plants in floodplains of arid climates. However, the physical aspects of flooding and water flow may be equally important in determining the structure and function of riparian communities. This is especially evident where plant community form and function are influenced by floods that recharge groundwater supplies, initiate new communities by removing vegetation, and provide moist, exposed seedbeds for germination and growth. Whereas, one of the main purposes of the wetland classification system is to "...ensure uniformity throughout the United States..." and one of its principal uses will be "...the inventory and mapping of wetlands..." (Cowardin et al. 1979), the main purpose of the present document is to describe the ecological properties and natural values of riparian systems and their associated streams.

The Marine and Estuarine Systems of the wetland classification system are not included in the riparian category here because our emphasis is on the ecosystems associated with the millions of kilometers of inland streams in the U.S.A. However, the obvious functional influence of exports from certain riparian ecosystems on estuarine and marine systems is discussed.

The Palustrine and Lacustrine Systems, where they occur in floodplains and, in their natural state, become connected to the stream when it floods, are included in this synthesis as part of the riparian system. This normally would include large (>8 ha) and deep (>2 m) oxbow lakes and lakes of levee flank depressions. Palustrine and Lacustrine Systems may be either a large or negligible part of a given sector of floodplain. The wetland classification system does not include wetlands occurring on the river floodplain as part of the Riverine System. For the purposes and uses of the wetland classification system (uniformity, inventory, and mapping), this may be desirable because the number of categories is reduced and the hierarchy simplified by omitting Lacustrine and Palustrine Systems in floodplains from the Riverine System. Cowardin et al. (1979) suggest that "It is the ground water that controls to a great extent the level of lake surfaces, the flow of streams, and the extent of swamps and marshes" (p. 10). However, under arid climatic conditions where evapotranspiration exceeds local precipitation, deprivation of streamflow would cause the disappearance of Lacustrine and Palustrine floodplain features except in anomolous situations where large rock aquifers provide most of the water supply.

Whether the Riverine System of the wetland classification system is included as a part of our functional riparian concept depends on where one chooses to draw boundaries. Although we focus primarily on properties of streambank and floodplain plant and animal communities, the influence of these communities on the stream, and the stream on these communities, makes it impossible to discuss one without the other. The problem with establishing boundaries between the two is the tendency to not consider the movement of water, matter, and organisms that provides the basis for coupling among ecosystems. Thus, the Riverine System is included to the extent that it plays a functional role in maintaining natural properties and attributes of riparian ecosystems.

Another set of "boundary" problems is in the headwater portions of streams where recognizable floodplains cease to exist and, at some point, riparian vegetation disappears. Usually erosion predominates and floodplain area is negligible in headwater streams because the amount of material available for alluvial deposition decreases due to diminishing size of the watershed. There may be a gradual transition from regions of alluvial fill to upstream areas where channels are eroding and the channel is confined by bedrock.[1] Leopold et al. (1964) observed that in humid climates this upper limit of floodplain development in stream systems appears to be the point at which flow in the channel changes from perennial to ephemeral, i.e., where groundwater supply is insufficient to sustain flow through nonstorm periods.

They suggested it is possible that perennial flow promotes rock weathering and subsequent sloughing into the channel, hence initiating lateral deposition and erosion along a small stream. In arid climates where intermittent streams are common because of protracted drought and high evaporative demand, these criteria would not appear to apply. It is possible that the vicinity of headward gully erosion and gully wall collapse (Leopold and Miller 1956) may represent the upper limit of floodplains in arid climates. However, riparian vegetation often continues upstream from that point and thus is not restricted to floodplains.

One of the problems of dealing with riparian ecosystems from a national perspective is the great diversity in vegetation, fauna, and geomorphology that exists. A geographically balanced synthesis of information is difficult to achieve because of the regional differences among research approaches. For example, many nutrient cycling studies have been done on southeastern floodplain forests because of the importance of these systems for water quality. Equivalent nutrient cycling studies are entirely lacking in arid floodplain forests where water, rather than nutrients, limits ecosystem processes. On the other hand, the water regimen of arid riparian floodplains has received considerable attention, yet equivalent studies are lacking in the Southeast. The ecological realities of different controlling factors in the wide diversity of riparian ecosystems in the U.S.A. must be recognized and appreciated.

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[1]Even sectors of large rivers may be confined by bedrock and be undergoing rapid downcutting. Under these conditions, zones of vegetation that are influenced by the stream may be quite narrow relative to broad floodplains where there are abundant alluvial deposits. This is discussed more fully in the section "Diversity Among Riparian Ecosystems."

CHAPTER TWO

STATUS OF RIPARIAN ECOSYSTEMS IN THE UNITED STATES

Throughout history, man has altered, developed, and influenced the extent and condition of riparian ecosystems, and today only a portion of the original floodland area is occupied by natural vegetation. There has been no single comprehensive inventory of riparian ecosystems in the United States to determine the amount of land area originally covered by riparian ecosystems and the proportion of that area presently supporting natural riparian communities. Data needed to provide this information with precision are generally unavailable, due primarily to the historical lack of recognition for the distinct and significant values of riparian ecosystems. However, existing resource inventories provide a rough indication of the extent and distribution of riparian plant communities. We have reviewed documented information from numerous Federal and State agencies and the literature on:

1. the past and present extent (area or length) of major riparian ecosystems in the United States, and

2. the extent and nature of floodplain and stream alterations that are responsible for losses of riparian ecosystems in the United States and the environmental quality of that which remains.

Overall, it appears that more than 70% of riparian ecosystems have been altered, and natural riparian communities now make up less than 2% of the land area in the U.S.A. Although a comprehensive inventory may be required for certain management purposes, there are sufficient data to conclude that these important ecosystems have not received adequate protection.

NATIONWIDE EXTENT OF RIPARIAN ECOSYSTEMS

Two approaches were used to provide insight to the amount and distribution of riparian ecosystems. An analysis of inventories on areas that are naturally prone to periodic flooding provided the best information on the land area of riparian ecosystems. We have also examined inventories on stream length as an independent estimate of riparian ecosystem extent and status.

Inventories of Floodplain Area

Of an estimated 916 million hectares of land in the entire U.S.A. (769 million without Alaska) (Frey 1979), approximately 6 to 9% is subject to flooding. Estimates of the amount of land subject to flooding vary from 49 million hectares (52 with Alaska) for 100 year floodplains (Maddock 1975), to 54 million hectares (without Alaska) subject to floodwater and sediment damage (USDA Conservation Needs Inventory Committee 1971), to 71 million hectares (without Alaska) of non-Federal rural flood-prone land (USDA Soil Conservation Service 1978).

These values probably overestimate the amount of riparian ecosystem once present, because the estimated original area of predominant riparian forest types totals only 27 million hectares (Table 1). Moreover, a portion of this floodplain area can no longer be considered forested because of extensive alteration. For example, only 29% (15

Table 1. Potential and present area of the four predominant riparian vegetation types in the United States. From Klopatek et al. 1979.

Vegetation type[a]	Area (1000 ha)		%
	Potential	Present	decline
Elm-ash forest	2,239	279	88
Northern floodplain forest	7,171	2,227	69
Southern floodplain forest	17,744	6,645	63
Mesquite bosque	71	63	11
Total	27,225	9,214	66

[a]After Kuchler (1964).

million hectares) of the Nation's floodplains were classified as nonurban and nonagricultural land (USDA Conservation Needs Inventory Committee 1971). Similarly, an estimated 30% (21 million hectares) of non-Federal rural flood prone lands are forested (USDA Soil Conservation Service 1978). Several riparian forest types have been cleared extensively in the conterminous U.S.A. (Table 1), with losses ranging from as high as 88% for elm-ash forest to as low as 11% for mesquite bosque (Klopatek et al. 1979). Thus, about 70% of the Nation's floodplain area has been converted from natural forest land to urban and cultivated agricultural areas.

Surveys conducted for purposes other than estimating riparian ecosystem coverage suggest that these lowlands constitute less than 30% of the total floodplain area. Floodplain forest types now account for about 9.3 million hectares of the conterminous 48 States (Table 1) . According to a national wetland inventory in 1954 (Shaw and Fredine 1956), there were 9 million hectares of seasonally flooded basins or flats, and 7 million hectares of wooded swamps, both common forms of riparian wetlands. However, these areas are not synonymous with riparian ecosystems because they include considerable area of wetland that is not riparian, and omit less frequently flooded riparian communities.

Riparian ecosystem inventories at the State level were summed to give a minimum existing area of 23 million hectares, of which 10.5 million are in the lower 48 States (Table 2). In the lower Mississippi Delta, an estimated 2.1 million hectares of bottomland hardwoods were remaining in 1978 (MacDonald et al. 1979a, 1979b). There are about 1.5 million hectares of bottomland hardwoods (12% of State area) in Mississippi (Mississippi Game and Fish Commission 1978), including a substantial amount outside of the Delta region. As of 1963, California had nearly 142,000 hectares of riparian vegetation (0.35% of State area) remaining (California Department of Fish and Game 1966). The total riparian area in Arizona is 113,000 hectares (0.4% of State) (Babcock 1968); the area in New Mexico may be equal or slightly larger (Pase and Layser 1977). Riparian communities on Bureau of Land Management lands constitute 287,495 hectares in western states, 5544 in the East, and 12,029,543 in Alaska (USDI Bureau of Land Management 1980). With the exception of Alaska, riparian ecosystems are clearly most abundant in the southeastern states where an estimated 8.5 million hectares, or 70% of the total documented area, were identified. Such data are generally unavailable for northeastern and northcentral U.S.A.

Table 2. Estimated area of riparian ecosystems in 26 States, or portions thereof.

State	Area (hectares)	Source
Alaska	12,029,500[a]	BLM[b] 1980
Arizona	113,153	Babcock 1968
Arkansas	410,765[a]	MacDonald et al. 1979b
California	140,537	Calif. Dept. Fish & Game 1966
Colorado	24,441[a]	BLM 1980
Idaho	22,909[a]	BLM 1980
Kansas	207,406	Spencer 1979
Kentucky	13,760[a]	MacDonald et al. 1979b
Louisiana	1,214,000[a]	MacDonald et al. 1979b
Mississippi	1,457,000	Miss. Game & Fish Comm. 1978
Missouri	38,851[a]	Korte and Fredrickson 1977
Montana	51,216[a]	BLM 1980
Nebraska	115,824	Spencer 1979
Nevada	36,423[a]	BLM 1980
New Mexico	113,314	Pase and Layser 1977
North Dakota	72,481	Spencer 1979
Oregon	71,135[a]	BLM 1980
South Dakota	53,905[a]	Spencer 1979
Southeast (Fla., Ga. N.C., S.C., Va.)	6,300,000	Langdon et al. 1980
Tennessee	52,610[a]	BLM 1980
Utah	28,934[a]	BLM 1980
Wyoming	18,471[a]	BLM 1980
Total	22,586,635	
Total	10,557,135 (without Alaska)	

[a]Estimates were only available for portions of the State and should be considered an underestimate.
[b]USDI Bureau of Land Management. Values cited as BLM (1980) are for "public land wildlife habitat" only and should be considered underestimates for the respective States.

Certain specialized riparian communities constitute a significant area in some regions of the United States. These areas are of particular interest to resource managers because of specific ecological or functional values associated with them. For example, there were more than 360,000 hectares of saltcedar vegetation in the arid western U.S.A. by 1961, and probably well over 400,000 hectares today (Robinson 1965). (This exotic woody plant has replaced many native floodplain plant communities but has very different and limited value to wildlife.) Beaver ponds occupy about 162,000 hectares of floodplain timber in the southeastern U.S. (Hill 1976, in Hair et al. 1978). In the Uinta basin of Utah alone, 19,733 hectares of vegetation are dependent upon irrigation return flow (Chalk 1979).

Based on these data, it appears that riparian ecosystems comprise between 10 and 15 million hectares in the

48 States, or about 1.5% of the U.S.A. land area. A more precise and comprehensive inventory may be required for certain management purposes, but there are sufficient data to conclude that these important fish and wildlife habitats are quite limited in extent in most regions of the country. That riparian ecosystems cover such a small proportion of the landscape is due to their limited extent originally (except in the Southeast), and to widespread floodplain alterations by man. Brief accounts of some representative riparian ecosystem losses are presented later.

Inventories of Streams and Rivers

Analysis of stream length across the country provides insight on the distribution and abundance of riparian ecosystems. Stream length generally reflects the potential abundance of riparian systems, and provides a common unit for measuring the extent of floodplain alterations.

There are an estimated 5.1-5.6 million kilometers of streams and rivers in the U.S.A., ranging from the smallest first-order tributary to the largest rivers (Leopold et al. 1964, U.S. Army Corps of Engineers 1978). However, only about 1.6 million kilometers were accounted for in a compilation of State stream inventories (Table 3). The latter figure may be more useful for discussion of riparian management potential, because it represents the extent of waterways recognized by respective State water resource agencies.

Riparian ecosystems are most extensive in humid and coastal plain regions, especially where perennial streams are relatively abundant and where warmwater streams and rivers predominate (Figure 1). Stream length per unit of land area (drainage density) is greatest in Louisiana, high throughout the eastern half of the U.S.A., but dramatically lower in the West. Similarly, the average surface area of streams (USDA Soil Conservation Service 1978) relative to length is considerably greater east of the Mississippi River.

There are some 492,000 kilometers of warmwater fishing streams in the U.S.A. (Funk 1970). More than 573,000 kilometers of major stream segments (greater than 40 kilometers long) have been identified in the Nationwide Rivers Inventory (U.S. Heritage Conservation and Recreation Service, pers. comm. 1980) which potentially support extensive riparian communities (Table 4). Some of the most outstanding riparian ecosystems in the country are associated with 141 major rivers (by discharge) totalling 116,000 kilometers in the U.S.A. (USDI Geological Survey 1974; see also Iseri and Langbein 1974). Included in the above list are the Atchafalaya, Brazos, Colorado, Columbia, Connecticut, Gila, Mississippi, Missouri, Rio Grande, Sacramento, Snake, and St. Lawrence Rivers.

While coldwater and intermittent streams are widespread and often support riparian communities with significant value to wildlife, their areal extent is singularly quite limited. Consequently, most available data on riparian areas were derived from large river systems, while the vast extent of small streambank communities has never been quantified.

LOSSES OF RIPARIAN ECOSYSTEMS

Historically, riparian ecosystems have been altered or destroyed to a largely unknown extent, without protection from long-term adverse impacts on their ecological functioning. Causes of riparian ecosystem degradation are numerous, and vary in importance from one region to the next. Available case histories are presented here to illustrate the nature of riparian ecosystem losses across the country.

Alterations of Floodplains

The areal extent of riparian ecosystems has been reduced by a substantial amount in nearly every region of the U.S.A. Losses of bottomland vegetation have been most dramatic in the Mississippi Delta, Midwest, and arid western areas, caused by demand for water and productive farmlands which they normally can provide. It is evident that losses at some locations far exceed the estimated national average of 70% (Table 5). Some examples are described below.

Table 3. Length of streams in the United States.[a]

State	Total stream length (kilometers)[a]	State	Total stream (kilometers)
Alabama	11,839	Montana	27,607
Alaska	1.6 million+	Nebraska	19,904
Arizona	1,287	Nevada	11,908
Arkansas	15,315	New Hampshire	20,241
California	46,959	New Jersey	4,184
Colorado	26,554	New Mexico	5,277
Connecticut	n/a[b]	New York	106,851
Delaware	1,287	North Carolina	6,437
Florida	16,979	North Dakota	n/a
Georgia	62,565	Ohio	70,678
Hawaii	2,364	Oklahoma	37,015
Idaho	25,296	Oregon	43,452
Illinois	21,250	Pennsylvania	40,057
Indiana	145,000	Rhode Island	n/a
Iowa	30,600	South Carolina	n/a
Kansas	16,100	South Dakota	5,544
Kentucky	64,400	Tennessee	30,417
Louisiana	64,887	Texas	128,748
Maine	44,893	Utah	9,864
Maryland	2,736	Vermont	10,461
Massachusetts	17,226	Virginia	n/a
Michigan	44,819	Washington	25,608
Minnesota	40,100	West Virginia	36,194
Mississippi	22,700	Wisconsin	43,713
Missouri	91,068	Wyoming	24,853
Total (without Alaska)			1,525,227

[a]Estimates were obtained by personal communication with State and Federal agencies. Although definitions of streams differ from State to State, most estimates represent perennial streams that potentially support a fishery. Details available from the authors.
[b]Data not available.

Land use changes on the 9.8 million hectare Mississippi Alluvial Plain (mostly riparian) have been documented (MacDonald et al. 1979a, 1979b). Bottomland hardwood forests covered only 4.8 million hectares in 1937, and were reduced to 2.1 million hectares by 1977. Cumulative losses between 1957 and 1977 ranged from 30% to 63% among various States (Table 6). The rate of clearing has averaged around 2% per year over the last 20 years. The majority of bottomland forest clearing in the Mississippi Delta results from conversion to agriculture, particularly for soybeans (Sternitzke and Christopher 1970, Sternitzke 1976, MacDonald et al. 1979a).

Area of bottomland hardwoods in southeastern Missouri declined 96% from an estimated 1.0 million hectares in 1780 to 40,000 hectares in 1975 (Table 5), primarily as a result of lumbering and drainage for agriculture (Korte and Fredrickson 1977). Between 1879 and 1972, the total water surface area of

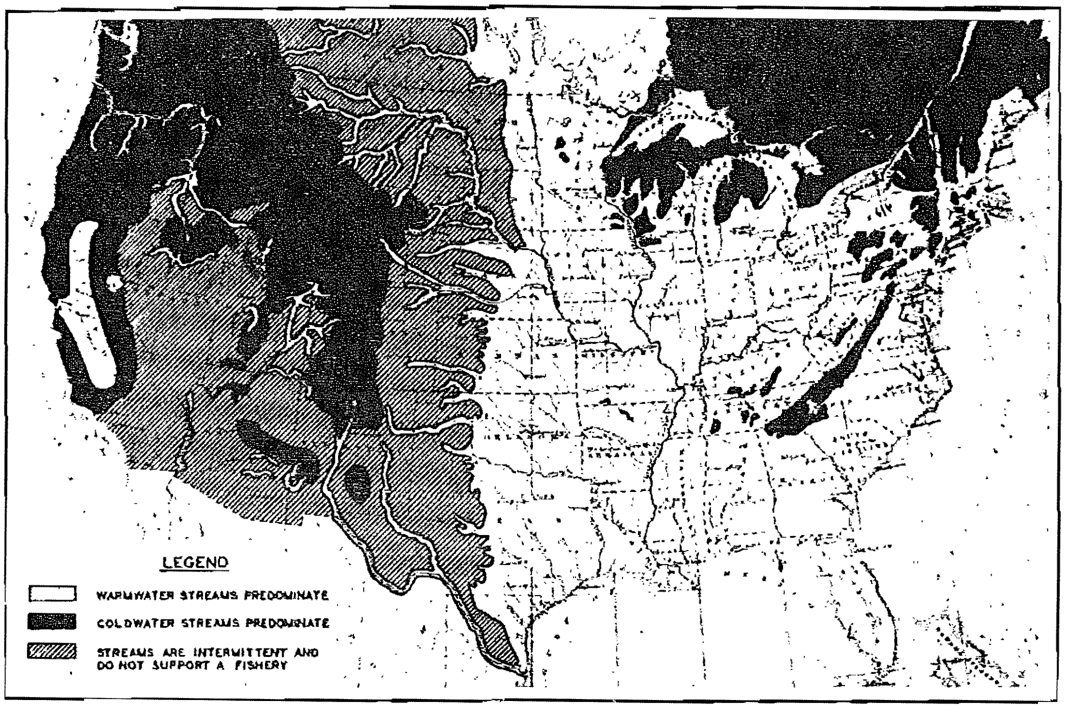


Figure 1. Distribution of warmwater, coldwater, and intermittent streams in the United States. Map used by permission of the American Fisheries Society.

Table 4. Total length of major stream segments in the U.S.A. and percentage unsuitable[b] for designation as National Wild and Scenic or Recreational Rivers.

HCRS[a] region	Kilometers of river segments[b] Total	Unsuitable for designation	% unsuitable
Northwest	42,129	7,500	18
Southwest	48,334	5,562	12
Mid-Continent	161,000	129,000	80
South Central	106,911	26,187	24
Lake Central	82,894	71,674	86
Northeast	40,234	31,704	79
Southeast	91,733	73,254	80
Total	573,235	344,881	60

[a]USDI Heritage Conservation and Recreation Service.
[b]Stream segments greater than 40 kilometers in length.
[c]Because of water resource or other cultural developments in the river corridor.

Table 5. Summary of case histories showing losses of riparian ecosystems.

Description	Time period	Estimated change (hectares)	% loss	Source
Bottomland hardwoods in lower Mississippi Delta: Ark., La., Miss., Mo., Tenn., and Ky.	1700's-1977	9.8 million to 2.1 million	79	MacDonald et al. 1979a
	1937-1977	4.8 million to 2.1 million	51	MacDonald et al. 1979a
Cottonwood communities along the Colorado River, Arizona	1600's-1977	2000 to 1133	44	Ohmart et al. 1977
Riparian vegetation along the San Pedro River, Arizona	1935-1978	27,900 to 20,030	28	McNatt 1978
Riparian forests along the Sacramento River, California	1850-1977	313,600 to 7,200	98	McGill 1975, 1979
Bottomland hardwoods in southeastern Missouri	1780-1975	1.0 million to 40,000	96	Korte and Fredrickson 1977
Channel habitats in Missouri River, Mo.	1879-1972	49,000 to 25,000	50	Funk and Robinson 1974
Two riparian forests in southcentral Oklahoma	1871-1969	12,100 to 1,544	87	Barclay 1980

the Missouri River (from Rulo, Missouri to the mouth) was reduced by 50% from 49,000 hectares to 25,000 hectares. Surface area of unconnected islands in the Missouri River was 9900 hectares in 1879, and 170 hectares in 1954, a loss of 98%. Elimination of channel communities in the Missouri River was the direct result of stream channel alterations (Funk and Robinson 1974).

In Oklahoma, 12,100 hectares of riparian forest along two streams experienced an 87% reduction in area between 1871 and 1969; about 81% was gone by 1937 (Table 5). These losses were largely attributable to impacts of channelization (Barclay 1980).

Riparian vegetation along the Colorado River is disappearing at a rate of 1200 hectares per year (Anderson et al. 1978). Pure cottonwood communities have declined from an estimated 2000 hectares to 200 hectares as a result of altered hydrologic regimes, impoundments, and agriculture. There are still some 1133 hectares of willow-cottonwood stands along the river (Table 5), but most are invaded by saltcedar, an exotic tree species of much lower value to wildlife (Ohmart et al. 1977).

Between 1935 and 1978, riparian areas composed of cottonwood, mesquite, saltcedar, and willow along the San Pedro River in Arizona increased from 6900 hectares to 14,200 hectares (Cottonwood and willow were actually declining as a result of eliminating perennial streamflows.) During that same time, other marsh, mesquite shrub, river channel, and streambed thickets of annual or immature plants decreased from 20,600 to

Table 6. Area of bottomland hardwoods in the lower Mississippi Valley, 1957 to 1977.

State	Bottomland hardwood area (1000 hectares) 1957[a]	1967[a]	1977[a]	% loss 1957-1977
Arkansas	843	537	411	52
Kentucky	21	16	14	36
Louisiana	1,743	1,513	1,214	37
Mississippi	613	478	377	39
Missouri	76	43	28	63
Tennessee	84	66	53	38
Total	3,380	2,653	2,097	38
Net loss during previous decade[b]	--	727	556	--
% loss during previous decade	--	21.5	21.0	--

[a]From Tables A1.1-A1.18 in MacDonald et al. (1979b).
[b]From Tables A3.1-A3.18 and A7.1-A7.6 in MacDonald et al. (1979b).

5700 hectares. The net loss of riparian vegetation was 7700 hectares (Table 5). However, along a 35 kilometer stretch of that river, riparian communities have declined from 4300 hectares in 1936 to 2200 hectares in 1972, nearly a 50% reduction (Lacey et al. 1975). Stream channel alteration, irrigation diversion, groundwater pumping, and overgrazing were all contributing factors to the alteration or destruction of those riparian communities (McNatt 1978).

There were nearly 313,600 hectares of riparian forests along the Sacramento River in the 1850's (Sands 1978). By 1952, about 11,000 hectares remained, and in 1972, there were only 7600 hectares (McGill 1975). Native riparian vegetation was further reduced to 7200 hectares by 1977, or about 2% of the original area (Table 5). Most recent losses were the result of converting high terrace forest land to deciduous orchard (McGill 1979).

The U.S. Geological Survey mapped 3700 hectares of phreatophytes in a 74 kilometer reach of the upper Gila River (Gatewood et al. 1950). When examined in 1958, 16% had been cleared for farm use (Horton 1972). Clearing continued, and only 2670 hectares were reported in 1967, a 29% reduction in 23 years (Lacey et al. 1975). About 45,000 hectares of floodplain along the lower Gila River was assumed to have been covered by riparian vegetation in 1860. In 1970 only 6620 hectares (15%) of riparian vegetation were present, and more than one-half was saltcedar communities. When total acreage of this exotic was subtracted, only 2350 hectares of native riparian communities remained, or about

5% of the theoretical riparian base present in 1860 (Haase 1972).

Riparian ecosystems have not been cleared so extensively in some areas of the country. For example, the acreage of bottomland hardwood-cypress forests in five southeastern states (Florida, Georgia, North Carolina, South Carolina, Virginia) remained fairly stable from 1940 to 1980 (Langdon et al. 1980). Cottonwoods, which were scarce along the lower South Platte River in the middle 19th century, increased greatly in number over the next 100 years and may have peaked in the 1950's, after water resource developments reduced the "flashy" flows to more moderate seasonal fluctuations (Crouch 1979). Mountain riparian areas have not changed as distinctly as lowland floodplain areas; there has been some clearing and construction of dams, but in general vegetation along mountain streams has been maintained by near normal ecological processes (Horton 1972).

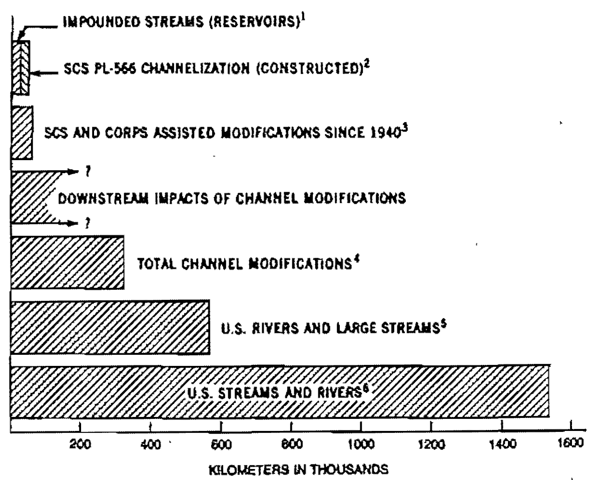


Figure 2. Extent of water resource development on streams in the United States. Sources: (1) estimated by authors; (2) USDA Soil Conserv. Serv. 1980; (3) Little 1973; (4) Little 1973; (5) pers. comm., USDI Heritage Conserv. and Recreation Serv. personnel 1980; (6) estimated by authors.

Alterations of Streams and Rivers

The total nationwide extent of riparian community losses caused by stream alterations has not been determined. However, available data indicate water resource development projects have resulted in substantial disruption of streamside ecosystems (Figure 2).

During the past century and a half, mankind has been responsible for the "development, improvement, or modification of at least 320,000 kilometers of waterways" (Little 1973). This constitutes a direct impact on at least 20% of the stream length recognized by the various States, and would equal over one-half of the total length of warmwater streams where channel alterations are most prevalent. However, actual losses in surface area of riparian ecosystems undoubtedly occur in larger proportion than losses in stream length. This is because large amounts of drainage and forest clearing usually accompany relatively small reductions in stream length.

Extent of recent channel alteration activities by Federal agencies has been documented (USDA Soil Conservation Service 1971, 1975, 1980; Little 1973). Between 1940 and 1971, the Corps of Engineers assisted 889 stream development projects covering a total of 17,827 kilometers of which 9946 kilometers were completed, 6270 were under construction, and 1611 kilometers were planned. As of 1972, SCS channel work in the U.S.A. totalled about 33,800 stream kilometers, of which 13,911 kilometers were constructed or under contract. By 1980, a total of 17,344 kilometers of SCS channel alterations were constructed or under contract, an increase of 483 kilometers per year (Table 7) (USDA Soil Conservation Service 1980).

Among the 1630 projects administered by the Corps and SCS by 1971, 45,614 kilometers were channel alterations and 9490 kilometers involved floodplain alteration by levee work. About 47% was to have been carried out in five States (Louisiana, Mississippi, Arkansas, California, and North Carolina) and an additional 25% in five other States (Texas, Florida, Georgia, Illinois, Indiana) (Little 1973).

Table 7. Summary of Soil Conservation Service channel work through 1972 and 1980 (from USDA Soil Conservation Service 1972, 1980).

	SCS channel work[a] (kilometers)		
	31 December 1972	12 March 1980[b]	
Region	Constructed	Constructed	Planned
Northeast	2,313	2,686	1,310
Midwest	1,651	2,226	1,799
South	9,429	11,830	13,123
West	520	604	367
Total[c]	13,913	17,346	16,599

[a] "Constructed" includes all channel work completed or under contract; "planned" includes all channel work planned and in an approved project but not constructed or under contract.
[b] Includes 1972 figures.
[c] Totals were calculated prior to rounding off of regional figures.

Estimates of stream channel alterations by SCS and Corps activities fall far short of the total carried out by all agencies and private interests (Table 8). In Missouri, for example, 3584 (4%) of the total 91,068 stream kilometers had been channelized, and an additional 4699 kilometers (5%) were inundated by impoundments at flood pool elevation (Missouri Dept. of Conservation, pers. comm. 1980). In a survey of 351 stream kilometers in Kentucky, 144 kilometers (41%) had been recently altered (Russell 1967). Approximately one-third of the total length of streams inventoried (402 of 1236 kilometers in Montana had been altered from their natural condition, of which half (222 kilometers) was by relocation, 103 kilometers were rip-rapped, and 66 kilometers were diked (Peters and Alvord 1964). Among 366 perennial streams in Hawaii, 15% have been channelized, totalling 151 kilometers and including 57% (31 of 54) on the populous island of Oahu (Timbol and Maciolek 1978).

Reduction in stream length is a significant but often unmeasured aspect of channelization projects. Loss of stream mileage from stream alterations may be very high in some stream corridors. One stretch of the Missouri River has been shortened from 875 kilometers in 1870 to 801 kilometers in 1972, a loss of 74 kilometers (Funk and Robinson 1974). Total length of 13 Montana streams and rivers was shortened 109 kilometers (9%) from the original 1236 kilometers by rerouting of 220 kilometers of stream into 111 kilometers of man-made channel (Peters and Alvord 1964). Data from Iowa indicate that stream length across the State has been reduced 1693 kilometers and possibly as much as 4800 kilometers (Bulkley et al. 1976). Other examples of stream length reduction are cited in Table 8.

Impacts of stream alteration clearly extend far beyond the actual development site; consequently data from the Nationwide Rivers Inventory (U.S. Herit-

Table 8. Extent of stream alterations in twelve States.

State	Extent of alteration	Source
Hawaii	Of 366 perennial streams in Hawaii, 15% have been channelized, totalling 151 kilometers (6% of the State total), and including 57% on the populous island of Oahu.	Timbol and and Maciolek 1978
Idaho	In a survey of 1831 stream kilometers, 698 (38%) had been altered.	Irizarry 1969
Illinois	An estimated one-third of the State's natural streams has been channelized.	D. Rogers, Ill. Dept. of Conserv. (pers. comm.)
Iowa	Total stream length in the State has been reduced at least 1693 kilometers and possibly as much as 4800 kilometers.	Bulkley et al. 1976
Kentucky	In a survey of 351 kilometers, 144 (41%) had been recently altered.	Russell 1967
Mississippi	About 3862 kilometers (17%) of the streams in Mississippi have been altered.	B. Freeman, Miss. Game & Fish Comm. (pers. comm.)
Missouri	Across the State, 3584 stream kilometers (4%) have been channelized, and at flood level an additional 4699 kilometers (5%) are inundated by impoundments.	O. Fajen, Mo. Dept. of Conserv. (pers. comm.)
	One stretch of the Missouri River was shortened from 875 to 801 kilometers since 1870, a loss of 74 kilometers (8%).	Funk and Robinson 1974
Montana	Approximately one-third of stream length studied (402 of 1236 kilometers) was altered from the natural condition. Total length had been reduced 109 kilometers (9%) by channelization.	Peters and Alvord 1964
Nebraska	Total stream mileage has been reduced 1341 kilometers (6%) by channelization.	G. Zuerlein, Nebr. Game & Parks Comm. (pers. comm.)
Ohio	An estimated 34,236 kilometers of streams (48% of the State total) have been altered.	A. Spencer, Ohio Div. of Wildl. (pers. comm.)
South Dakota	About 20% of the State stream mileage is altered, including impoundment of 80% (644 of 805 kilometers) of the Missouri River.	R. Hanten, S.D. Dept. of Game, Fish and Parks (pers. comm.)
Tennessee River Basin	Over 5600 kilometers (8%) of the total 67,600 stream kilometers are impounded at normal full pool level. An additional 1770 kilometers (3%) have reservoir-regulated flows.	Tennessee Valley Authority 1971

age Conservation and Recreation Service, pers. comm. 1980) may provide a better indication of stream condition across the Nation. Among the 570,000 kilometers of major stream segments, 60% were judged unsuitable for inclusion in the National Wild and Scenic Rivers System because of water resource or other cultural developments within riparian corridors (Table 4).

Because reservoirs are situated in floodplains and riparian zones, construction of impoundments has resulted in significant losses of riparian ecosystems and their values to wildlife. The total length of streams inundated by reservoirs has not been determined, but probably exceeds 24,000 kilometers. By January 1, 1980, there were 1608 reservoirs with a mean annual pool of 202 hectares or more. This increased the area of these reservoirs by 409,550 hectares since 1970 to a total of 3,989,000 hectares (Ploskey and Jenkins 1980). If an arbitrary 4:1 ratio of length to width and triangular shape were assumed for these reservoirs, they would extend over an estimated 22,000 kilometers of streams. Among the 1562 reservoirs having a storage capacity of 617 hectare-meters or more, 6,002,000 hectares would be covered at maximum controllable water level (Martin and Hanson 1966) and would flood over 27,400 kilometers of stream. In the Tennessee River Basin, an estimated 5734 kilometers (8%) of the total 67,500 stream kilometers are impounded at normal full pool level, and 1814 kilometers (3%) have reservoir-regulated flows (Tennessee Valley Authority 1971). The creation of Lake Oahe on the upper Missouri River in South Dakota inundated 90,650 kilometers of land, including all areas along a 320 kilometer reach of river (Hirsch and Segelquist 1978).

Prosser et al. (1979) state that the "loss of terrestrial habitat from reservoir construction constitutes only 0.6% of all undeveloped lands capable of supporting wildlife." However, direct loss of length in major streams is probably at least 5% nationwide, while extent of downstream impacts cannot be estimated. Further, the land area inundated by large reservoirs alone is equal to 8% of the total 100-year floodplain area, a value which does not include the extent of undocumented loss due to smaller reservoirs.

ASSESSMENT OF RIPARIAN ECOSYSTEM STATUS

Existing inventories of floodplain area and stream length cannot be used alone or without interpretation to provide an overview of the status of riparian ecosystems in the U.S.A. However, when taken together, these data give a great deal of insight to the nationwide amount of riparian ecosystem that was originally present, the quantity lost to other uses, and the nature of alterations. From the foregoing data, we derived some rudimentary estimates of the status of riparian ecosystems in the U.S.A. (Figure 3).

One estimate of the amount of land subjected to riverine flooding (100-year floodplain) and thus potentially supporting riparian ecosystems is 49 million hectares, or 6% of the U.S.A. land area (excluding Alaska). This figure may be considered liberal, because the estimated original area of four predominant riparian forest types totals only 27 million hectares (Table 1). Regardless, much less exists in a natural or seminatural forested condition, and streamside riparian communities now constitute only about one-third of the original area. The extent of bottomland alterations is known to be much greater in Arizona, California, and Missouri, and for certain floodplain forest types. Because at least 10.5 million hectares of riparian communities can be accounted for from State surveys (Table 2), the nationwide total is probably between 10 and 15 million hectares, or about 1.5% of the conterminous U.S.A. land area.

The great difference between potential riparian land area and that now in a woodland condition reflects the extent of alteration that has occurred, and some discrepancies in defining and delineating riparian ecosystem boundaries. Many of our riparian lands have been directly destroyed or converted to urban or agricultural uses that are usually incompatible with natural ecological functions (Chapter 3) and wildlife resources (Chapter 4). These alterations can be considered "acute" because they severely preclude most other goods and services to society that riparian ecosystems provide (Chapter 5). As compared to all other vegetation types in the U.S.A. (Kuchler 1964), conversion of floodplain forests to other land uses puts riparian ecosystems among the most severely altered landforms in the nation.

In addition to these losses of riparian communities that can be quantified, stream alterations, pollution, grazing, and recreation can also reduce the functional quality of remaining areas through more subtle "chronic" impacts. In the northcentral and northeastern states, up to 80% of major stream corridors are interrupted by water resource or cultural developments. In the South and West, existing riparian communities are disturbed by manipulation of streamflows and overflows, and subjected to problems associated with consumptive uses of water and grazing. Numerical estimates of riparian ecosystem area fail to measure these less intensive disturbances.

The significance of riparian ecosystem alterations, whether acute or chronic, lies in the relative irreversibility of man's impacts. Although agricultural and water resource developments can theoretically be reversed, the economic expense and incentives for doing so in floodplains are currently very prohibitive. Most importantly, reclamation of riparian ecosystems requires restoration of complex natural hydrologic regimes. However, because conversion of flood-prone areas to other uses usually involves permanent drainage or impoundment, opportunities for mitigation and recovery by natural succession are practically nonexistent.

Despite the outstanding ecological values of natural riparian ecosystems, natural plant communities on these lands have been reduced in extent by 70% overall, and as much as 95% in some areas. The functioning of remaining areas is threatened by further direct losses and impacts of man's activities in adjacent aquatic and upland ecosystems. The effect of these riparian ecosystem losses to the well-being of society, through the degradation of ecological function, wildlife resources, and production of goods and services, will be apparent in following chapters.

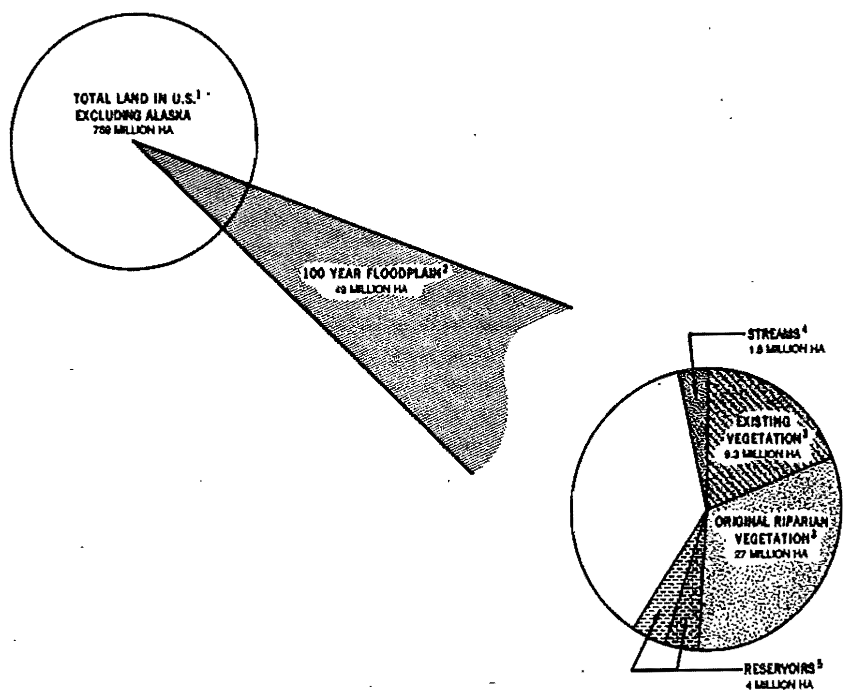


Figure 3. Land area covered by riparian vegetation, streams and reservoirs in the United States. Sources: (1) Frey 1979; (2) Maddock 1975; (3) Klopatek et al. 1979; (4) USDA Soil Conserv. Serv. 1978; (5) Ploskey and Jenkins 1980.

CHAPTER THREE

FUNCTIONS AND PROPERTIES OF RIPARIAN ECOSYSTEMS

All ecosystems have common properties of energy flow, material cycling, and community organization; yet no two ecosystems are organized and function in exactly the same way. However, riparian ecosystems have several unifying properties that set them apart from other ecosystem types.

One of these properties is their linear form, a consequence of being associated with streams. As a result, the abundance of riparian ecosystems depends on drainage density of streams (kilometers of stream length per square kilometer of land area) which, in the northeastern U.S.A. for example, ranges between 1 and 2.5 km/km (Leopold et al. 1964). Thus, there are few places in that region that are very distant from a riparian ecosystem.

Another related property is the function that riparian ecosystems serve in providing corridors for the transport of water and erodible material derived from the landscape. In comparison with upland ecosystems, riparian areas tend to be wetter, to have more nutrients available to them, and to be more frequently subjected to catastrophic water flow. The convergence of energy and material from the landscape on riparian ecosystems is expressed in their nutrient-rich soils and lush growths of vegetation.

Finally, the property of linearity and the function as corridors of material transport combine to assure that riparian ecosystems are profoundly connected to other ecosystems upstream and downstream from them. Few other ecosystem types possess such a large amount of transition zone relative to the area that they occupy. These transition zones are the boundaries at which terrestrial and aquatic ecosystems interface and the sites of important exchanges of material and energy in the landscape.

In spite of these apparent differences between riparian and upland ecosystems, it is difficult to find quantitative data on basic ecological characteristics (energy flow, nutrient cycling, community structure) that clearly distinguish these ecosystem types from one another. A major problem is that riparian ecosystems vary greatly among geographic regions, as do upland ecosystems. One of the purposes of this chapter is to determine the extent to which data on riparian ecosystem structure and function allow us to characterize them as unique ecological entities. Recognition of any unifying characteristics of riparian ecosystems may be useful in assessing the effects of their alteration and in providing guidelines to their management.

FLUVIAL PROCESSES

Plant and animal communities are sensitive to the edaphic conditions under which they develop. In riparian ecosystems, soil moisture is an extremely important variable because small topographic variations in a seemingly level floodplain can mean the difference between a waterlogged, anaerobic environment and a well drained, aerated substrate. Many plant species are intolerant of even brief periods of inundation while fewer species are adapted to survive in constantly waterlogged soil.

As a result, abrupt changes in species composition may occur in floodplains with elevational variations of only a few centimeters.

Natural fluvial processes are responsible for many of the diverse, often subtle, topographic features of floodplains. An understanding of fluvial processes responsible for forming riparian ecosystems is necessary in order to predict consequences of alteration or manipulation of the natural system. Alteration of fluvial processes is likely to create a new set of floodplain features to which plant and animal communities must adapt.

Human activities in riparian ecosystems are frequently oriented toward stabilizing, rather than maintaining the dynamic nature of fluvial processes. The many approaches to stabilizing stream channels and controlling water flow are but a few examples of efforts to counteract dynamic fluvial processes. However, stabilization processes such as these have, in many instances, decreased rather than increased fundamental ecosystem properties such as species diversity and processes such as rates of primary productivity, nutrient cycling, and animal production. Fluvial processes are necessary for the formation and continued maintenance of riparian ecosystems; therefore, we begin with an overview of these processes before examining the more purely biological properties.

Geomorphology

Alluvial portions of valleys where riverine forests normally occur may be undergoing aggradation, degradation, or be in a steady state condition. In the steady state condition, where the supply of alluvium from upstream erosion is balanced by the transport of alluvium downstream, floodplain features do not necessarily remain static. In fact, morphologic features of floodplains continually change as river channels meander laterally and in a downstream direction.

Aggradation and Degradation. Under non-steady state conditions, an alluvial valley and its stream may aggrade or degrade. Over time, these trends of aggradation and degradation may alternate, resulting in complex stratigraphic sequences. Leopold et al. (1964) illustrated the hypothetical development of terraces by means of two sequences of events that lead to the same surface geometry (Figure 4). A large-scale example of these processes has been described for the Mississippi alluvial valley, but the sequence also can occur in smaller streams.

The Mississippi alluvial valley has undergone at least five alternating periods of valley cutting and alluvial deposition that correspond with glacial advance and retreat during the Quaternary period (Fisk 1944, 1952; Fisk and McFarland 1955). Glacial advance and accumulation of water in continental ice masses resulted in a lowering of the sea level by several hundred feet. In an effort to adjust to this lowered base level, erosion of an extensive valley system occurred across the Gulf Coastal Plain. As ice sheets retreated, sea level rose, and the entrenched valley system became alluviated during the interglacial stages. Coarse material was introduced first from steep tributaries which built alluvial cones of gravel and sands. When these materials reached the Mississippi, they were transported seaward and deposited over wide areas by a braided river system as aggradation occurred. As the basal portion of the alluvium thickened, sediments became finer because stream gradients were reduced and did not have the competence to transport coarse sediments. As sea level stabilized, the braided channel was replaced by a single meandering one through a combination of diminishing load, smaller particle size, and deeper scouring action. As a result, the Mississippi River is now in an overall balance between aggradation and degradation.

Smaller streams have been shown to undergo similar but less dramatic phases of downcutting and alluvial filling (Hadley 1960). Factors which cause these shifts can be the result of one or more of the following processes: geologic uplift, change of base level (usually sea level), or change in climate. Particularly for smaller floodplains, colluvium, or material transported from valley sides, can be a source of material for floodplain

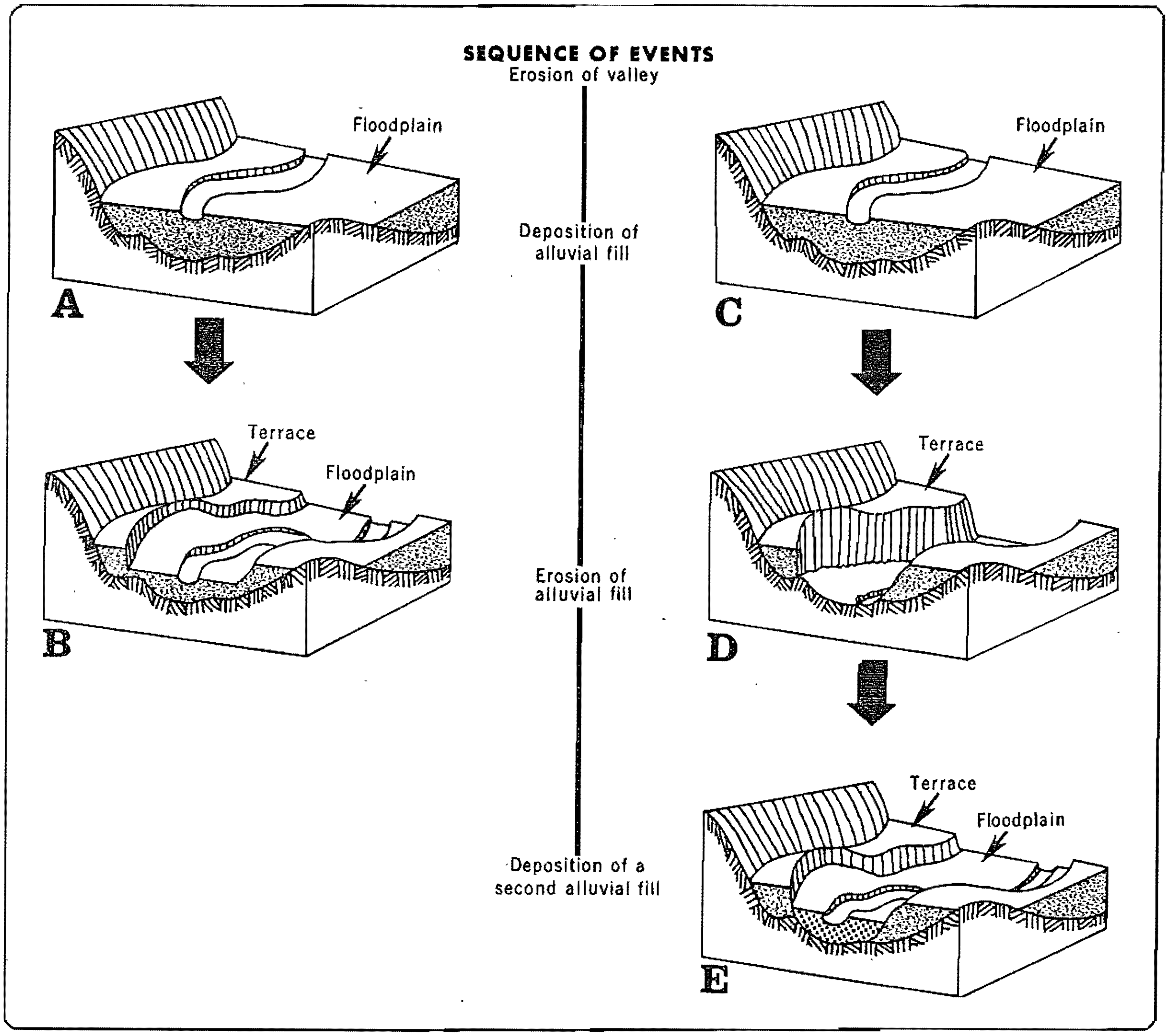


Figure 4. Two sequences of events leading to the development of the same surface geometry in terraces and floodplains. Only example D is confined by bedrock. From Leopold et al. (1964).

deposits. In narrow portions of floodplains this material may predominate as the substrate for floodplain forests. For example, approximately one-fifth of the cross sectional area of the alluvium of Beaverdam Run, Pennsylvania consists of colluvium (Lattman 1960). The remainder consists of channel fill, lag deposits (boulders), lateral accretion, and vertical accretion (including peaty material).

During channel overflow, there is an opportunity for vertical accretion of the floodplain through the deposition of suspended sediment transported from upstream. (Flooding from local precipitation does not result in floodplain accretion.) This deposition is, of course, a feature which contributes to the high fertility of floodplain soils. These deposits represent augmentation of nutrient capital in those areas of the floodplain where they occur. The amount of overbank deposition is proportional to the hydroperiod (duration and depth of flooding) and the amount of suspended-sediment load. While suspended-sediment load varies in proportion to the erodibility of the watershed, hydroperiod depends on local floodplain topography combined with flood frequency of the stream. The recurrence intervals for bankfull flows for 19 streams in the United States summarized by Wolman and Leopold (1957) range from 1.07 to 4.0 years. In the bottomland forests of the White River basin in Arkansas, sites where annual flooding occurs may remain flooded as much as 40% of the year (Bedinger 1979).

Rates of deposition differ greatly among floodplains and within a given floodplain. Observations on the rate of vertical accretion in floodplains range from a few millimeters per year to over a meter during a single flood episode (Table 9). It cannot be determined from these values whether or not the standing stock of alluvium is increasing or decreasing because few studies report rates at which floodplain erosion occurs. The floodplains of Beaverdam Run, which changed to an aggrading regime perhaps 200 years ago due to deforestation of the area, consists of vertical accretion in at least the upper 2 m (Lattman 1960). The floodplain of the Cimmaron River in southwestern Kansas has been undergoing vertical accretion at the rate of 2.1 cm/year since a major flood destroyed the pre-existing floodplain features and replaced them with a valley-wide braided channel (Schumm and Lichty 1963).

Sudden climatic and man-induced changes in discharge and sediment load can reverse trends in aggradation and degradation of stream channels. These altered trends, in turn, can be extrapolated to changes that will occur in floodplain hydrology and geomorphology. Lane (1955) proposed the simple and useful relationship

$$QS \propto Q_s D_{50}$$

in which Q is water discharge, S is the slope of the channel bed, Q_s is the bed-material discharge, and D_{50} is a measure of the size of the channel bed material. For example, if a dam is constructed on a stream, bed material is trapped behind the dam and clear water is discharged downstream. This decreases Q_s on the right-hand side of the equation which would require a reduction in S on the left-hand side, assuming Q and D_{50} remain constant. Consequently, downstream from the dam, channel-bed slope (S) would decrease, a phenomenon which is brought about by degradation or net erosion of the stream channel. This implies an increase in channel capacity and a lower stage height for equivalent discharge volume. Thus, floodplain inundation would occur with less frequency and involve less floodplain area, resulting in dryer conditions in the riparian ecosystem. Even if protective measures were taken to reduce the rate of channel degradation, a reduced sediment supply from upstream and regulated flow below the dam would result in altered floodplain conditions. Several other applications of Lane's equation (Simons et al. 1975) provide examples of man-induced stream changes from which floodplain alterations are implied.

<u>River Meanders and Topographic Features.</u> Riverine forests grow on a number of topographic features that are generally the result of aggradation, degradation, and meandering of the river channel itself (Allen 1965). Some typical floodplain features that are apparent in a section of the Mississippi River, Louisiana (Figure 5) include:

1. Natural levees adjacent to the channel which contain coarser material deposited during flood overflow.

2. Meander scrolls located on the inside curve of bends. These rises and depressions, which are the result of point bar deposits, formed as the channel migrated laterally and downslope.

3. Backswamp deposits and sloughs where finer sediments are deposited in meander scroll depressions or in slack water along the valley wall.

Table 9. Deposition rates in forested floodplains.

River and locality	Deposition rate	Event or period	Source
Missouri R., N.D. Near Bismarck	8 - 10 cm	1952; largest flood on record for river	Johnson et al. 1976
Lowlands between Bismarck and Mandan	180 cm		
Cimmarron R., SW Kans.	5.1 cm/yr	Ca. 12 years of record using tree age since a destructive flood	Schumm and Lichty 1963
Cache River, Ill.	0.8 cm/yr	Of annual total, 0.06 cm from flood of 1.13 yr. recurrence	Mitsch et al. 1979a
Upper Mississippi R.	1.7 cm/yr	Annual deposition in backwater lake on floodplain	Eckblad et al. 1977
Kankakee R., Ill.	590±121 g/m^2	Total sedimentation during spring flood of which 80% was inorganic	Mitsch et al. 1979b
Ohio R., Ohio	0.24 cm	Mean deposition during 100 yr. flood, Jan. - Feb. 1937	Mansfield 1939, in Wolman & Leopold 1957
Connecticut R.	3.47 cm 2.23 cm	March 1936 Sept 1935	Jahns 1947, in Wolman & Leopold 1957
Kansas R.	2.97	July 1951	Carlson & Runnels 1952, in Wolman & Leopold 1957
Rio Grande, N.M.	1.5 cm/yr	Mean aggradation for 16-yr period between Albuquerque and Socorro	Thompson 1955
Alexandra R., Alberta	0.3 cm/yr	Fed by glacial meltwater; average aggradation during past 2,500 yrs.	Smith 1976
MacKenzie R., N.W.T.	1.3-1.9 cm	Mean for sand deposition along point bar for 2 mo. during each of two summers	Gill 1972a

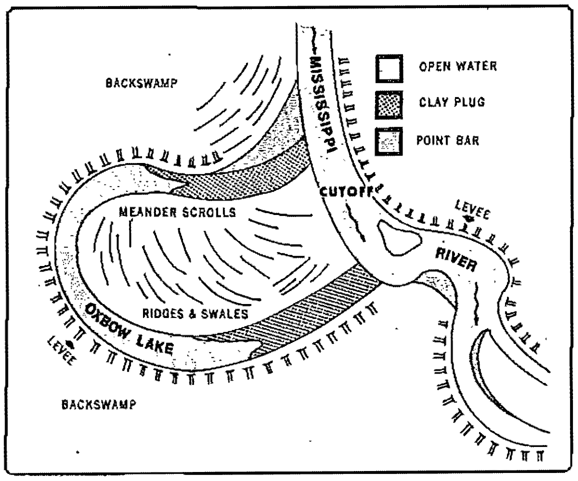


Figure 5. Typical floodplain topographic features, illustrated diagrammatically, of the Mississippi River near False River, Louisiana. Adapted from Fisk (1952).

4. Oxbows or oxbow lakes which are relict meander bends that have been cut off.

5. Point bars on the inside curve of river bends where deposition is rapid.

Streams migrate back and forth across floodplains and move in a downslope direction; consequently all areas in a floodplain, with the exception of those formed by colluvial deposits, have been traversed at one time by the stream channel. If the rate of meander movement occurs on a time scale similar to that of ecosystem succession, younger communities will be encountered on the inside meander curve (Leopold et al. 1964). Wolman and Leopold (1957) report rates of channel migration ranging from 10 ft (3 m) to over 2000 ft (610 m) per year for rivers with drainage basins greater than 100,000 mi^2 (259,000 km^2).

It should be possible in some circumstances to calculate the rate of lateral channel movement from the gradient of tree age in a transect perpendicular to the inside of a meander curve (Everitt 1968). On the basis of successional development in a section of the Missouri River, it has been demonstrated that the youngest communities correspond to the center of the floodplain, while the oldest ones are located at the edge (Table 10). Although any area in the floodplain may be potentially eroded by river meanders, Johnson et al. (1976) showed that the center of the Missouri River floodplain, or the "meander belt", is eroded more frequently. Rivers in the southeastern Atlantic States appear to be migrating southward as indicated by their proximity to bluffs on the south side and by the presence of broad floodplains on the north side. In extremely broad floodplains, such as the lower Mississippi River, large areas of the floodplain have not been occupied by the river channel for thousands of years (Gagliano and van Beek 1975).

Thermo-erosional processes are particularly significant in bank erosion and meander rates in regions of permafrost. Outhet (1974) has classified bank types in the Mackenzie River delta according to their shape and erosional rates (Figure 6). River channels in permafrost environments erode the bank on the outside of meanders as elsewhere; however, the development of thermo-erosional niches (bank undercutting) and the presence of structural weaknesses (ice wedges and other forms of ground ice) result in large-scale sloughing to a somewhat greater extent than occurs in temperate environments. Although nearshore stream current and thermal exchange are usually responsible for niche development, erosion by wave action may be significant where a long open-water fetch is possible on wide rivers. Continuous removal by high current velocities all summer is why type 1 banks have higher rates of erosion than other types (Figure 6). Type 2 banks are a result of intermittent removal of material caused by variations in channel discharge or variation in wind velocity or direction. Type 3 banks are a result of soil flow where ice-rich bank faces retreat continuously through the summer. Destruction of cut bank levees is accompanied by deposition along their backslopes; hence, the levee form is maintained without its total destruction (Gill 1972b). Only where thermo-erosional niches are active (type 1) does the floodplain become destroyed and undergo degradation without compensating alluviation.

Table 10. Dependence of relative stand age on location in a floodplain. Values are percent of stands measured in each age category and floodplain location. After Johnson et al. (1976).

Relative stand age class	Percent of stands measured in each age category and floodplain location		
	Meander belt	Intermediate	Edge of floodplain
Young	64	36	0
Medium	18	73	9
Old	8	25	67

Thus fluvial processes have at some time been responsible for shaping nearly all floodplain features. These processes produce topographically diverse and spatially heterogeneous conditions that result in a mosaic of diverse habitats for plant and animal communities.

Hydrology and Hydroperiod

Riparian ecosystems vary considerably from stream to stream and even in sectors along a single stream. However, differences in hydrologic properties are mainly those of magnitude since all riparian ecosystems are influenced by flooding, possess topographic features of fluvial origin, and are dominated to various degrees by the streams that flow through them. Surface water hydrology is the most visible feature of floodplain hydrology, but it cannot be fully understood without considering its interaction with groundwater.

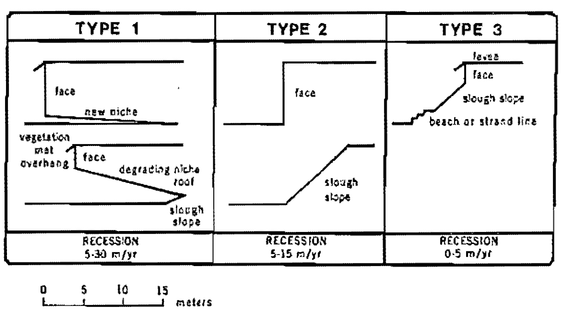


Figure 6. Rates of erosion and diagrammatic side views of stream banks in permafrost environments. Modified from Outhet (1974).

Surface Water. The flooding regime of riparian ecosystems may differ in depth, frequency, duration, and time of the year. Some of the factors that may influence the depth of flooding (defined as the difference in stage of a stream at median discharge and a given flood recurrence interval) include climate, topography, channel slope, soils, and geology (Coble 1979). If all these factors remain constant, then the depth of flooding depends largely on size of the drainage basin and storage capacity of the floodplain surface. Topographic features of floodplains may also impound water and cause flooding as a result of local precipitation independent of stream discharge. This flooding is particularly common in oxbows, depressions between parallel levees, and in back swamp depressions where drainage patterns to the stream channel are poorly developed (Figure 5). More commonly, where floodplains slope gently from the river channel to uplands, both flooding frequency and depth from overbank flow are inversely proportional to floodplain elevation. Typically, annual floods occupy a greater area of floodplain than do less frequent floods. Successively higher levels of the floodplain occupy less of the total area.

Duration of flooding is directly related to the drainage area of the stream basin upstream from the site in question. For floodplain areas with annual flooding on the Ouachita and White River basins in Arkansas, flood duration ranges from 10 to 18% of the year for sites having drainage areas from 13,000 to 18,000 km , and from 5 to 7% of the year for sites having drainage areas less than 780 km (Bedinger 1979). This is a consequence of broader storm hydrograph peaks for streams with large drainage basins than those with smaller ones. Sites on streams having drainage areas of several 10's of thousands of square kilometers in these river basins typically flood for as much as 40% of the year. In doing so they hold flooding waters from the trunk stream which serves to ameliorate downstream flooding.

An example of where many factors that regulate flooding come into play is the gradient beginning in the eastern slope of the Appalachian Mountains, continuing through the Piedmont province, and terminating along the south Atlantic seacoast. Mountainous headwater streams are characterized by small watersheds, steep slopes, and constricted V-shaped valleys. The typically shallow soils have limited storage capacity for water. Orographic rains result in greater precipitation than occurs at lower altitudes. Consequently hydrographic peaks are sharp and frequent, particularly toward the end of the winter season and into the spring when evapotranspiration is low and soil water storage reaches annual highs. In the rolling topography of the Piedmont, flood peaks are the highest among the three physiographic provinces (Coble 1979) and tend to occur when frontal weather systems stabilize over the region and provide abundant precipitation. Flash floods are less likely than in the mountains partly because of larger watershed size and greater storage capacity of the deeply weathered soils. Coastal plain rivers that have their origin in the Piedmont and mountains tend toward a prolonged winter hydrographic pulse as a result of integrating the upstream peaks. Floodplains in low elevations of broad alluvial valleys may remain flooded for months at a time.

Generalizations on surface water hydrology can seldom be made for large geographic regions. For example, the physiographic and climatic diversity of Alaska results in a variety of flooding regimes. The largest floods occur along the Pacific Ocean, where the Pacific Mountains System forms a barrier to moist air from the ocean, resulting in high precipitation and rapid runoff in the fall and winter from the rugged slopes (Childers 1970). North of this mountain system precipitation is less, flood discharge rates are much lower, and floods are confined to spring and summer. In interior Alaska and the north slope drainage, extensive freezing and rapid warming in the spring may cause spectacular spring breakup floods when snowmelt flows into ice-jammed channels.

Where glaciers flow across the mouths of valleys, water flow may become blocked and form a lake (Post and Mayo 1971). Catastrophic floods may occur when glacier dams fail. These events are especially prevalent in the Pacific Mountain System of Alaska where outburst flooding from glacier-dammed lakes may be annual, once each 2 to 4 years, or only after several years. Wide floodplains may be inundated to unusual depths, and rapid erosion, deposition, and stream channel changes may occur.

In the annual cycle of interior and north Alaskan rivers, five hydrologic periods can be recognized (MacDonald and Lewis 1973). The longest period is when the river is frozen beginning as early as October and lasting into May. During this prolonged period the availability of unfrozen water under ice is critically important to aquatic invertebrates and fish and also to several species of mammals and birds (Wilson et al. 1977). Rising temperatures in May melt snow and flow is initially on top of the winter ice cover during the pre-breakup phase. The breakup phase may last only several days and may be accompanied by ice jamming, depending on local conditions such as river level when freezing initially occurred and whether the stage rises sufficiently to cause ice to float freely downstream. A post-breakup flood normally coincides with peak snowmelt. The summer flow phase may be established by mid-June when the general trend is of decreasing discharge except for occasional summer storms that may cause rapid rises in river stage.

Due to the variety of factors that control flooding regimes, surface water hydrology in riparian ecosystems is highly site specific. To understand the hydrology of a given area of floodplain, both the hydrologic characteristics of the watershed and local groundwater hydrology must be taken into consideration.

Ground Water. Ground water in the alluvial aquifer is in intimate connection with surface water in streams and floodplain depressions (e.g., oxbow lakes). The normal gradient and direction of ground water movement is toward these surface water features through ground water discharge. During periods of high river stages the gradient is reversed and water moves from the stream to the aquifer. The extent to which the alluvial aquifer is an important area for discharge and recharge of ground water depends upon its size. Two extremes were illustrated in Figure 4. In example D the floodplain is narrow and alluvium mostly lacking; under these conditions the floodplain will have little groundwater storage and a small alluvial aquifer. In example A, the groundwater storage of the alluvium is potentially large and may greatly influence the surface water hydrology either by serving as a source of water for the channel at low river stage or as a recipient of water from the channel at high river stage.

For the lower Missouri River floodplain, Grannemann and Sharp (1979) have shown that the river itself has the most important influence on groundwater levels. During sustained high river stages, which normally occur between spring and early autumn, inflow of lateral seepage keeps groundwater levels high. The hydraulic gradient is reversed as river stage falls from late autumn through the winter when floodplain groundwater supplies base flow to the river. Grannemann and Sharp (1979) discuss several other factors that control groundwater flows and levels in the floodplain. These include:

1. Distance from the river channel. Equalization of differences in water head change more slowly farther from the river than close to it.

2. Time elapsed since the river has risen or fallen. Provided the river stage does not overtop the levee system, a sustained flood peak will contribute more water to the groundwater system than a higher flood of shorter duration.

3. Geometry of the river meanders and valley walls. Where an area of floodplain is partially encircled by a sharp river meander or where floodplain segments are narrow due to proximity of stream channel and valley wall, river stage and groundwater levels will respond to each other more quickly.

4. Variations in the composition of alluvium. Thick clay strata and clay plugs will create a longer time lag than sand or silt in groundwater head response to river stage changes due to the lower transmissivity of clay sediments.

5. Tributary creeks flowing into the floodplain. These may cause permanent groundwater highs and promote downvalley flow where they are oriented parallel to the major river.

Water table fluctuations in the floodplain of the upper Sangamon River, Illinois, are strongly controlled by the water level in the stream channel (Bell and Johnson 1974). At middle elevations between the stream and uplands, groundwater loss to evapotranspiration during certain summer periods may exceed the combined sources of water by infiltration of groundwater from the river and drainage from higher elevations. Thus, even in the absence of overbank flooding, groundwater levels in floodplains may fluctuate in response to other factors.

Attempts to quantitatively determine water budgets from inflow/outflow measurements are restricted to streams in arid regions where floodplain or bottomland groundwater deposits are subjected to competitive demand by phreatophyte vegetation and by withdrawals for consumptive human use and irrigation. Figure 7 is a generalized model for a water budget of the alluvial fill of a floodplain. Results of a study for the Gila River floodplain (Gatewood et al. 1950) are superimposed on this figure to show the magnitude of water movement. The predominant flows of water for the various reaches studied were inflows from upstream and downstream outflows. Among the total outflow from the lowermost reach, only 2.5% was due to evaporation from the river surface and wet sand bars and 12.3% to evapotranspiration by the bottomland vegetation. While the value for evapotranspiration may be an overestimate according to more recent studies (van Hylckama 1980; R. M. Turner, pers. comm.), the magnitudes of flow suggest that groundwater storage and flow is extremely important to the maintenance of surface flows. Greatest groundwater use by evapotranspiration occurred during the warm months when flows through the stream sector were lowest. During the early winter months groundwater recharge coincided with increasing throughflows.

Significance of Fluvial Processes

The kinetic energy of flowing water and its capacity to erode, transport, and deposit materials are responsible for the origin and necessary for the maintenance of riparian ecosystems. Fluvial processes are essential for producing and maintaining topographic features. If stabilization of water flows and stream banks interferes with natural fluvial processes, much environmental diversity normally present will disappear. Floodplains should be considered the part of the stream channel that is utilized to accomodate high flows. Flooding opens up the riparian ecosystem to inflows of material from upstream that would not be available if flooding were controlled.

The water storage capacity of alluvial deposits is particularly critical for maintaining riparian vegetation during the warm season in arid climates when upstream supplies of water are low. Where base flow of streams is dependent on groundwater storage, it may be advantageous to maximize groundwater recharge through overbank flooding. Plant and animal communities are adapted to or even dependent on these pulses of flow because they evolved under the natural conditions of flooding.

ENERGY FLOW AND BIOMASS DISTRIBUTION

Energy flow is often regarded as an indicator of the vitality of an ecosystem. It does not necessarily follow that ecosystems with high primary productivity are inherently more valuable or in better condition than those with lower productivity. For example, a northern bog swamp would undoubtedly have lower primary productivity than a southeastern river swamp. However, both are responsible for contributions to productivity of the regional landscape and must be evaluated in the context of their location. For a given ecosystem, primary productivity will vary widely depending on weather conditions, time of year, water availability, and other environmental variables. However, indicators of primary productivity such as litterfall and biomass accumulation provide insight to the magnitude of energy flow so that ecosystems can be compared and factors that control the energy flow can be evaluated and identified. Environmental manipulations that either severely diminish or abruptly augment energy flow, particularly if the change is irreversible, may be considered disruptive to plant and animal communities as well as other goods and services derived from ecosystems.

One of the fundamental functions of primary productivity, in addition to providing energy flow to food webs, is that of maintaining the structure and integrity of ecosystems. During ecological succession in forested ecosystems, large amounts of energy flow initially are diverted toward the accumulation of new plant and animal biomass and the formation of more complex

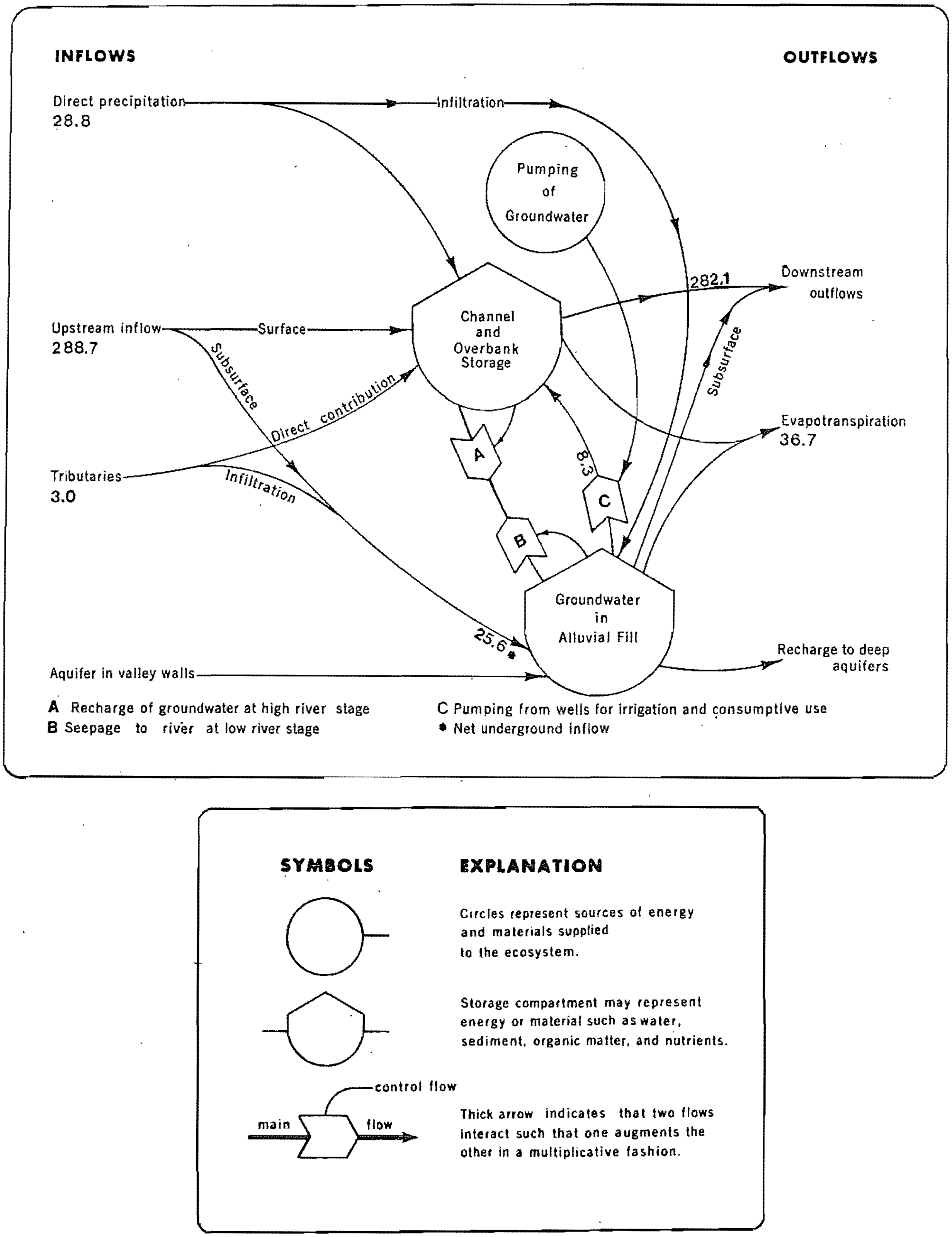


Figure 7. General model of floodplain hydrology. Numbers represent fluxes of water in centimeters per year for a reach of the Gila River and its floodplain. Values from Gatewood et al. (195). Symbols after Odum (1971).

ecosystem structure. When the quantity of biomass stabilizes, energy flow continues to be utilized for the maintenance of existing biomass levels through replacement of organisms that have died and are undergoing decay.

Therefore, food production and maintenance of ecosystem structure are the two basic ways in which primary productivity is important to consumer organisms. Numerous studies in riparian ecosystems have documented the capacity of these ecosystems to maintain high vertebrate population densities, particularly in comparison with upland ecosystems. The extent to which these higher standing crops of vertebrates respond to the production of food and the maintenance of structure will be discussed in later sections. Here we examine the nature of biomass distribution in riparian ecosystems and annual rates of biomass accumulation.

Biomass Distribution and Accumulation

Aboveground biomass in riparian ecosystems varies widely ranging from 10 kg/m^2 to 119 kg/m^2 (Table 11). There is insufficient information to determine the basis of this variation, but differences in stand maturity or age probably obscure regional trends. However, basal area, a rough index of the amount of woody biomass, is available for a larger number of ecosystems. When the basal areas of both riparian and upland ecosystems are compared (Figure 8), it is apparent that basal area for uplands follows a more regular pattern and is under some control by annual precipitation. The curve shown in Figure 8 delineates a hypothetical maximum set of values for basal areas in upland forests and shows a decline below about 50 cm annual precipitation where grasslands begin to replace forests. In contrast, the basal areas of riparian forests appear to be independent of precipitation, resulting in the presence of floodplain forests in climates where upland ecosystems support only grassland or desert vegetation. The more moist conditions of riparian ecosystems, as compared with uplands, are a result of the convergence of runoff along river corridors.

Annual aboveground biomass production of riparian forests varies between 339-650 g dry wt/m^2 for litterfall (leaves, fruits, and flowers) and between 311-1100 g dry wt/m^2 for stem wood production (Table 11). Since litterfall varies in a predictable fashion with climatic and edaphic factors (Bray and Gorham 1964) and leaves are the photosynthetic structure, litterfall values are probably highly correlated with primary productivity. Lowest and highest values for litterfall roughly correspond at respective sites with lowest and highest values for stem wood production, but the correlation between the two is rather low (r=0.38). The production of stem wood biomass accounts for about 54% of aboveground biomass production. The remainder is litterfall (mostly leaves) which is available to different groups of consumers depending on the season.

Production of belowground biomass and subsequent mortality of roots may be essential in maintaining levels of organic matter in soils. No conclusions can be drawn for belowground biomass standing crop and production since only incomplete estimates of the total are available (Table 11). However, Burns (1978) reported higher standing stocks of fine root biomass and greater seasonal differences at undrained, as compared with drained, cypress strands in Florida. This suggests that the drier site (with less aboveground production; Table 11) had slower root turnover rates than the wetter site with natural flows. All of the reported belowground root values exclude stump biomass which may account for approximately one-half of the total belowground biomass (Harris et al. 1975). More information is needed to determine if root biomass distribution and production respond to other factors such as hydroperiod, water table depth or sediment composition and to evaluate the influence of these variables on species composition of riparian ecosystems.

Ecosystem Metabolism

Biomass distribution and annual rates of biomass production are relatively static measurements that tend to obscure seasonal differences in energy flow. Both temperature and hydroperiod have a profound influence on the carbon balance of floodplain soils. Mulholland

Table 11. Structural characteristics and biomass production of riparian forests.

Forest type	Stem density (No./ha)	Basal area (m^2/ha)	Biomass (kg/m^2) Above-ground	Biomass (kg/m^2) Below-ground[a]	Leaf & fruit litterfall ($g/m^2 \cdot yr$)	Stem wood production ($g/m^2 \cdot yr$)	Total biomass production ($g/m^2 \cdot yr$)	Source
Cypress floodplain, Fla.	1644	32.5	28.4	--	521	1086	1607	Brown 1978
Bottomland hardwood, La.	1710	24.3	16.5	--	574	800	1374	Conner & Day 1976 and pers. comm.
Cypress-tupelo, La.	1235	56.2	37.2	--	620	500	1120	Conner & Day 1976 and pers. comm.
Cypress-tupelo, Ill.	--	--	45.2	--	348	330	648	Mitsch et al. 1977, Mitsch 1978
Cypress strand, Fla.	--	--	19.2	0.80(to 30 cm)	339	772	1111	Burns 1978
Cypress strand, drained, Fla.	--	--	10.3	0.31 (to 30 cm)	311	370	681	Burns 1978
Cypress strand, sewage-enriched, Fla.	--	--	28.6	2.34 (to 40 cm)	650	640	1290	Nessel 1978
Floodplain swamp, N.C.	705	47.8	27.6	2.70(to 40 cm)	524	585	1384	Mulholland 1979, Brinson et al. 1981b
Fenn, Minn.	3348	25.1	10.0	--	412	334	746	Reiners 1972
Riverine forest, Panama	3792	59.6	118.9	1.22	--	--	--	Golley et al. 1975
Floodplain forest, Ill.	--	--	29.0	--	--	--	1250	Johnson & Bell 1976
Transition forest, Ill.	--	--	14.2	--	--	--	800	Johnson & Bell 1976
Alluvial swamp, N.C.	2730	69.0	--	2.35(to 40 cm)	522	--	--	Brinson et al. 1980; Brinson et al. 1981b

[a]Root biomass, not including stump roots.

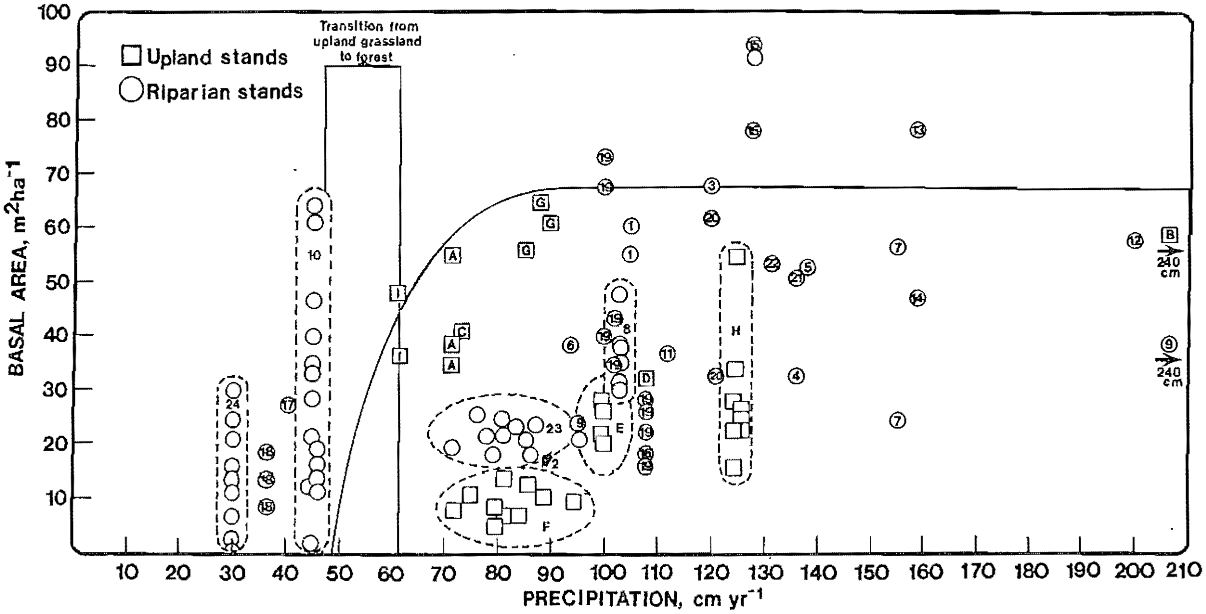


Figure 8. Effect of increasing annual precipitation on the basal area of vegetation for upland forests (curve) in comparison to riparian forests (no pattern). Sources for upland forest: (A) Eggler (1938); (B) Fonda (1974); (C) Gilman (1976); (D) Hough (1936); (E) McEvoy et al. (1980); (F) Rice (1965); (G) Stearns (1951); and (H) Whittaker et al. (1974). Source for riparian forests: (1) Anderson and White (1970); (2) Barclay (1980); (3) Brinson et al. (1980); (4) Brown (1978); (5) Burns (1978); (6) Conard et al. (1977); (7) Conner and Day (1976); (8) Crites and Ebinger (1969); (9) Fonda (1974); (10) Freeman and Dick-Peddie (1970); (11) Fredrickson (1979); (12) Golley et al. (1975); (13) Hall and Penfound (1939a); (14) Hall and Penfound (1939b); (15) Hall and Penfound (1943); (16) Hosner and Minckler (1963); (17) Johnson et al. (1976); (18) Lindauer (1978); (19) Lindsey et al. (1961); (20) Mulholland (1979); (21) Nessel (1978); (22) Penfound and Hall (1939); (23) Rice (1965); and (24) Zimmerman (1969).

(1979) obtained detailed measurements of floodplain forest floor respiration for 2 years under flooded and unflooded conditions in North Carolina. Highest respiration rates corresponded with highest temperatures and greater respiration rates were observed for unflooded conditions. Since unflooded conditions and high temperatures coincided during the growing season, a large proportion of the respiration and carbon dioxide loss from the forest floor occurred at this time. Total forest floor respiration averaged 0.95 g C/m^2·day taking into account changes in flooded and unflooded portions of the swamp throughout the year. Of this, 74% was due to the respiration of unflooded portions, 17% to flooded portions, 5% to respiration of the water column, and the remaining 4% to anaerobic respiration. This suggests that alternate wetting and drying of wetland soils augments losses of carbon dioxide and prevents organic matter from accumulating. Where water level fluctuations are absent, organic matter frequently accumulates as peat deposits.

Primary productivity and respiration measurements for riparian ecosystems are available only for a cypress forest in Florida studied by Brown (1978). Gross productivity averaged 26 g C/m^2·day, highest of all other cypress ecosystems that she studied, especially when compared to the non-riparian cypress domes of Florida. Community respiration was also higher (25 g C/m^2·day) than other cypress ecosystems. These values are among the highest reported for any ecosystem. The abundance of phosphorus and other nutrient supplies from stream flooding provide resources necessary to sustain these high rates of primary productivity.

Factors Affecting Primary Productivity and Growth

The hydrology of riparian ecosystems can have an effect on the metabolism and growth of vegetation in three basic ways. First is water supply, whereby water storage is recharged through seepage and channel overflow to floodplains. This is of great importance for plants in arid climates since it has been shown that riparian forest communities are maintained in regions too dry to support upland forests (Figure 8). Second, nutrient supply in riparian ecosystems depends partly on sedimentation of particulate matter transported by overbank flow and partly on the availability of dissolved nutrients in the water in contact with floodplain soils. Finally, in comparison with stagnant water in non-riverine wetlands, flowing water in floodplain swamps ventilates soils and roots so that gases are exchanged more rapidly. Oxygen is supplied to roots and soil microbes; at the same time the release of gaseous products of metabolism such as carbon dioxide and methane is enhanced. Water flow provides the medium for the export of dissolved organic compounds, some of which are metabolic wastes.

Flood frequency and groundwater supply are major environmental factors controlling the growth of floodplain trees. To determine if reduced flooding would affect tree growth on the Missouri River floodplain, Johnson et al. (1976) measured radial wood growth representing 15-year periods prior to and following flood control by reservoirs. Significant decreases in growth of older, established trees downstream from the reservoir occurred after flood control in three species that germinate under normal floodplain forest conditions. Simulation of actual evapotranspiration rates showed that when water surpluses from flooding were absent, low autumn and winter precipitation in the region was insufficient to bring moisture in the surface soil to field capacity by the initiation of the growing season. Even in the more humid climate of Mississippi, bottomland hardwood vegetation shows accelerated growth when artificially impounded water is available until about June (Broadfoot 1967). Perhaps three-fourths of the annual radial growth occurs between late April and late June in southeastern Arkansas (Phipps 1979). Abundant water supplies at that time may be critical to support maximum growth.

Either too much or too little water can have detrimental effects on growth of vegetation that is already adapted to an existing water regime. For cypress trees in the Cache River floodplain in Illinois, Mitsch et al. (1979a) reported an increase in basal area growth rate as a function of average river discharge (Figure 9). The slower growth rates prior to 1937 and following 1966 are believed to be a result of water levels raised and maintained by beaver dams. In the drained portion of a cypress strand in Florida, Burns (1978) reported reduced litterfall and root biomass as compared with a similar site with natural water flows.

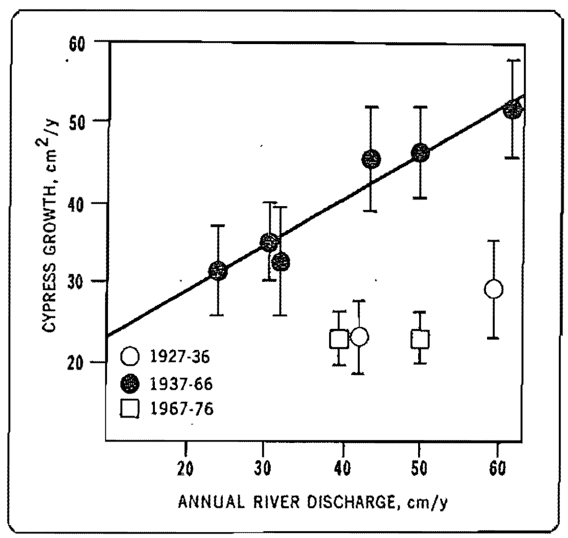


Figure 9. Relationship between annual river discharge and cypress tree growth on the Cache River, Illinois. (Mitsch et al. 1979a).

Reductions in growth attributed to lower nutrient and sediment supply rates in the absence of river overflow have not been documented. The higher fertility of many floodplain soils would likely have sufficiently large stocks of nutrients so that effects would be noticeable only after protracted periods of nutrient deprivation. Moreover it is difficult to separate effects of water and nutrient supply. However, comparisons among riverine and stillwater forested wetlands suggest that sustained nutrient supply from river overflow is responsible for higher nutrient cycling rates (Brinson et al. 1980) and higher rates of primary productivity (Brown 1978) in riverine forests. These examples provide indirect evidence that the nutrient supply to riparian ecosystems can control tree growth and affect soil fertility. Stimulation of tree growth due to artificially augmented nutrient supply has been demonstrated and will be discussed later.

Energy Transfer from Producers to Consumers

The preceding discussion on energy flow illustrates how plant biomass production is allocated between the building of riparian ecosystem structure and its continued maintenance. These processes are similar to those that occur in upland ecosystems, except to the extent that they are affected by additional water supply and flooding. However, riparian ecosystems are unique in the manner in which some of the energy as organic matter or organic carbon is transferred from producer to consumer organisms. This uniqueness derives from the fact that litterfall produced within the riparian ecosystem may be transported laterally (Bell and Sipp 1975) and made available to instream animal communities as well as those downstream from the source of organic matter production. As compared with purely aquatic or terrestrial ecosystems, organic matter produced in riparian ecosystems has the potential of supporting a diversity of food webs within both habitat types.

There appears to be a useful distinction between swamp-draining and upland-draining streams in the manner in which organic matter is transferred from the riparian to the aquatic ecosystem. Upland-draining streams are those that have negligible or narrow floodplains that receive organic matter from the riparian zone principally by litter falling directly from streamside vegetation to the surface of the stream. Flood events may transport litter from stream banks into channel and downstream. In comparison, swamp-draining streams are in watersheds that have a higher proportion of floodplain to upland surface area than do upland-draining streams. Not only do swamp-draining streams receive litter falling directly to their channel, but inundation of broad floodplains provides the opportunity for additional transport of organic matter from the floodplain forest.

Export of Organic Matter from Swamp-Draining Streams. Runoff is the principal forcing function that influences export of organic carbon from upland watersheds (Brinson 1976). The extent to which export is augmented by floodplains and wetlands associated with a river system probably depends on the proportion of wetland to upland surface area. Peculiarities of flow and inundation patterns in floodplains may directly influence the export of litter (Bell and Sipp 1975). However, particulate forms of organic carbon usually make up only a small portion of the total organic carbon supply in rivers, although the value of the particulate fraction in providing food for certain organisms is quite high.

Organic matter export from both upland-draining and swamp-draining watersheds (Table 12) shows a pattern of both higher concentration and higher export rate from watersheds that have extensive wetland coverage. Mulholland and Kuenzler (1979) demonstrated that there was a linear relationship between annual organic carbon export and runoff for both watershed types, but that swamp-draining watersheds export significantly more organic carbon than upland-draining watersheds. Rapid leaching of organic carbon has been demonstrated from newly fallen leaves of water tupelo (*Nyssa aquatica*), a species common in some southeastern swamps, which would contribute to the organic carbon supply of

Table 12. Concentration and export of organic carbon in drainage waters for upland- and swamp-draining watersheds. Values are dissolved organic carbon except as indicated for total organic carbon (TOC). Values originally reported as organic matter were multiplied by 0.5 to estimate organic carbon.

Locality	Area (km^2)	Annual runoff (cm)	Concentration mean or range (mgC/liter)	Export ($gC/m^2 \cdot yr$)	Source
Swamp-Draining Watersheds					
Neuse River, N.C.			7.1		Malcolm & Durum 1976
Sopchoppy River, Fla.			27.0		" "
Oscuro, Guatemala			1.5-18.4		Brinson 1976
Amatillo, Guatemala			2.8-18.0		" "
Mississippi River Delta, La.	770	89	11.2-12.2	10.4(TOC)	Day et al. 1977
Fahkahatchee River, Fla.			5-27		Carter et al. 1973
Barron River, Fla.			9-26		" "
Lower Satilla River, Ga.			12.7-36.2		Beck et al. 1974
Creeping Swamp, N.C.					
CP-10 (1976)	80	22.3	15.1	3.37	Mulholland & Kuenzler 1979
CP-10 (1977)	80	40.3	20.8	8.37	
CP-20 (1976)	32	17.9	10.6	1.89	" "
CP-20 (1977)	32	38.7	17.6	6.81	" "
Palmetto Swamp, N.C.	54	22.2	11.2	2.49	Mulholland & Kuenzler 1979
Tracey Swamp, N.C.	141	22.3	12.2	2.72	" "
Chicod Swamp, N.C.	132	22.3	15.2	3.39	" "
Clayroot Swamp, N.C.	110	22.3	14.5	3.23	" "
Upland-Draining Watersheds					
Arctic					
Char Lake, N.W.T.	43.5	15.8	1.9(TOC)	0.30(TOC)	deMarch 1975
Temperate					
Brazos River, Tex.			3.3		Malcolm & Durum 1976
Mississippi R. above delta, La.			3.4		" "
Missouri River, Neb.			4.6		" "
Ohio River, Ill.			3.1		" "
Hubbard Brook, N.H.					
watershed No. 2 (defor.)	0.16	122.1	2.2	2.71	Hobbie & Likens 1973,
watershed No. 6 (forest)	0.13	96.2	1.6	1.51	Bormann et al. 1974
Bear Brook	1.30	72.0	2.7	1.95	Fisher & Likens 1973
Mirror Lake	0.85	64.7	2.9	1.89	Jordan & Likens 1975.
Fort River, Mass.	107.3	79.8	4.1	3.29	Fisher 1977
Marion Lake, B.C.	13	204.8	2.5	5.17	Efford 1972
Nanaimo River, B.C.	894	168	8.7	14.6	Naiman & Sibert 1978
Humid tropics					
Polochic, Guatemala	5,247	194	1.1-3.7	4.8(TOC)	Brinson 1973 & 1976
Sauce, Guatemala	300	86	1.7-6.2	3.2(TOC)	" "
San Marcos, Guatemala	170	85	0.9-1.9	2.2(TOC)	" "

waters flowing through forested wetlands (Brinson 1977). In contrast, leaching of soluble organic carbon through well-drained or upland mineral soil horizons is relatively slow and inefficient since residence times for absorbed organic carbon may be several centuries (Scharpenseel et al. 1968). Higher organic matter export from swamp-draining streams appears to be related to long retention times of water in contact with the litter, detritus, and organic soils of the forest floor.

The significance of particulate organic detritus to filter feeding crustaceans in lacustrine and marine ecosystems is well established. This evidence suggests that detritus exported to downstream ecosystems is an important source of energy for lakes and estuaries (Seki et al. 1969, Brinson 1973, Livingston et al. 1974, Livingston and Duncan 1979). The correlation between intertidal vegetation surface area and commercial yields of penaeid shrimp (Turner 1977) as well as the influence of estuaries on the plankton of the continental shelf (Turner et al. 1979) extend the concept of ecosystem coupling to near-shore waters of the ocean.

The significance of dissolved organic carbon exports is less apparent, but concentrations and biological demand for oxygen are high in surface waters of many wetlands. This suggests that at least a portion of the dissolved organic carbon is readily available for microbial metabolism and thus conversion into particulate forms for filter feeders (Correll 1978). Other fractions, particularly low molecular weight humic and fulvic acids, have been shown to have stimulating effects on marine phytoplankton (Prakesh et al. 1973) presumably owing to their capacity to make certain micronutrients available for uptake by algae. Flocculation of dissolved organic matter induced by the brackish waters of estuaries may serve as a mechanism for generating particulate forms that would be available for filter feeders.

It has been demonstrated in several estuaries that a large proportion of the organic carbon in estuarine sediments is derived from terrestrial sources (Rashid and Reinson 1979, Tan and Strain 1979). Since it has been shown that forested wetlands export disproportionately high amounts of organic carbon in relation to their surface area, as compared with upland regions, sources of organic carbon from wetlands may be vital in maintaining organic carbon supplies to the sediments in some estuaries. The amount of organic matter in estuarine sediments can in turn affect a number of other variables including chemical oxidation/reduction gradients, microbial processes that convert nitrate to gaseous nitrogen, sediment/water exchanges of ammonium and phosphate, and benthic community species composition. Thus where wetlands contribute to watershed exports, stream alteration and wetland drainage could reduce the concentration and alter the distribution of organic carbon in estuarine sediments.

Energy Flow and Community Structure in Upland-Draining Streams. In the headwaters of upland-draining streams, organic matter contributions to flowing water (beyond those from groundwater sources) derive principally from leaves falling directly to the water surface from streamside vegetation (Figure 10).

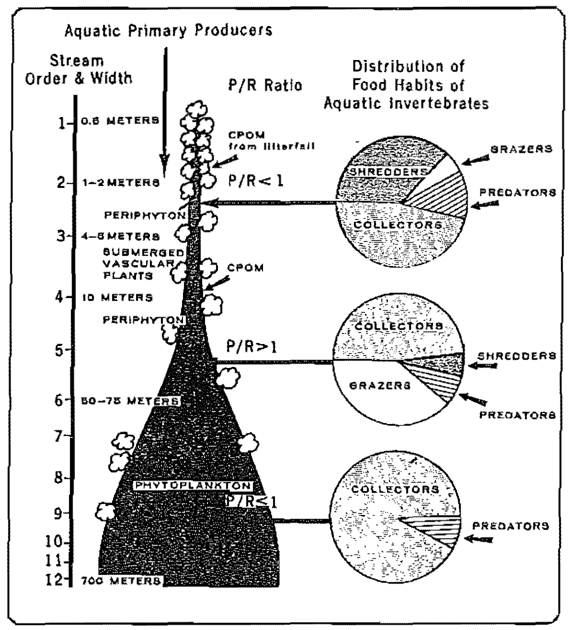


Figure 10. Changes in structure and function of upland-draining streams from headwater to mouth. Modified from Marzolf (1978).

In these situations, water flow is confined to rather discrete channels in relatively narrow valleys, as compared with swamp-draining streams in floodplains where the water flow is usually sluggish, and small increases in discharge may increase the water surface area by severalfold. It is obvious that streamside or riparian vegetation will have greater influence on instream energy flow near the headwaters where the forest canopy is continuous, and have less influence in higher order streams where the ratio of stream margin to surface area decreases.

Inflows and outflows of organic matter are frequently segregated into size classes as coarse particulate (>1 mm), fine particulate (<1 mm), and dissolved organic matter (Figure 10). Whole leaf detritus is consumed by groups of invertebrates that "shred" or fragment leaves into smaller particles. These particles are further fragmented by even smaller organisms while others act as collectors or macrogatherers (Cummins 1974). The coarsest fractions have the greatest probability of being processed (either fragmented to smaller fractions, metabolized by micro-organisms, eaten by invertebrates or leached) Fisher (1977) reports that 61% of the gross input of coarse particulate organic matter to Fort River, Massachusetts was metabolized, retained or converted to smaller particles. For the fine particulate organic matter, only 9% was processed and the rest exported. Dissolved organic matter actually showed a net gain which means it was being added at a greater rate than the stream ecosystem could process it. Consumption of dissolved and particulate organic matter is largely through microbial respiration (McDowell and Fisher 1976); however, immature aquatic insect populations are largely particle feeders and are dependent on particulate organic detritus and the associated microbial community for energy and nutrition. Experimental removal of leaf packs in streams reduces the consumption of dissolved and particulate organic matter in the water (Bilby and Likens 1980). Leaf packs derived from riparian vegetation thus provide sites for utilization of organic matter which would otherwise be exported downstream.

The energy flow or metabolism in any stream sector can be quantified on a unit area basis and consists of inputs from upstream and tributary flows, direct litterfall, and aquatic primary productivity (aquatic macrophytes and benthic and planktonic algae), and of outputs from respiration and downstream export (Fisher 1977). It is possible that natural stream communities adjust their structure and activities in undisturbed watersheds to maintain an idealized stream metabolism such that accretion of all newly derived organic matter (from leaf fall or autotrophic production) will be consumed within a given stream segment. For headwater streams where autotrophic contributions are negligible and leaf litter inputs (coarse particulate organic matter) predominate, invertebrates such as collectors and shredders are in greatest abundance (Figure 10). The rapid leaching of soluble organic matter from newly fallen leaves also supplies microbial communities with an energy source, which then becomes available to certain collectors as fine particulate organic matter. Further downstream where the canopy opens, or in headwater streams with little shading (Minshall 1978), grazers assume greater relative abundance. Thus it appears that the presence of riparian vegetation plays a profound role in the structure of invertebrate communities. Since many fish species are dependent on these invertebrates as their sole source of food, the riparian vegetation indirectly plays a role in fish community structure.

Ecosystem metabolism is an indicator of similar trends in the significance of riparian vegetation. The ratio of gross photosynthesis to ecosystem respiration (P/R) in the aquatic ecosystem increases from a value of less than 1.0 in shaded headwater streams with a continuous canopy to greater than 1.0 where autotrophic activity dominates (Odum 1956). Minshall (1978) points out that using this ratio alone to characterize stream metabolism may obscure the significance of the primary producer-grazer food chain since a ratio of less than 1.0 (predominantly heterotrophic) does not imply that primary productivity is negligible. He warns against application of ecosystem generalizations (as in Figure 10) since there are substantial geographic areas in arid regions where streamside woody vegetation is water limited and within-stream autotrophic processes predominate on an annual basis. Litterfall inputs, primary productivity, and respiration vary widely in streams of differing size and degrees of shading (Table 13). Primary productivity varies inversely with litterfall and thus decreases with decreasing stream size in humid climates. Although Fort River, Massachusetts has abundant aquatic macrophyte production, little is grazed, and it enters the food web as detritus (Fisher and Carpenter 1976). However, in arid regions where streamside woody vegetation is sparse, small streams receive most of their organic matter from instream primary production of algae and aquatic plants. Following a late summer flash flood in a Sonoran Desert stream, 90% of the preflood algal standing crop and primary productivity was attained in 2 weeks (Fisher et al. 1980). This demonstrates the rapid recovery of a probable food base for consumers after disruptive floods.

Implications for removal of riparian vegetation extend beyond those of disrupting coarse particulate matter inputs and shifting energy flows toward a more autotrophically based food chain. Removal of vegetation is usually accom-

Table 13. Comparison of litterfall, primary productivity, and respiration for several sizes of streams.

	Stream width(m)	kg organic matter/$m^2 \cdot yr$			P/R ratio
		Litterfall	Gross primary productivity	Ecosystem respiration	
Humid climate					
Headwater stream, canopy continuous[a]	2.2-4.0	0.56	0.002	0.46	0.004
4th order stream, canopy discontinuous[b]	14	0.38	0.61	1.25	0.49
Broad stream, canopy negligible[c]	90	0.028	1.2	--	--
Arid climate					
Small stream, trees lacking[d]	1-6	0.006	1.78	1.76	1.01

[a]Bear Brook, N.H. (Fisher and Likens 1973).
[b]Fort River, Mass. (Fisher 1977).
[c]Thames River, England. Only net productivity of plankton is available (Mann et al. 1972).
Litterfall from Mathews and Kowalczewski (1969).
[d]Deep Creek, Id. (Minshall 1978).

panied by changes in hydrology, sediment and nutrient loading, and temperature so that results are neither orderly nor predictable. If the consequences were merely an increase in P/R ratio, invertebrate communities would probably rapidly adjust their species composition and community organizations to new conditions.

The structural integrity of stream beds in lower order streams is dependent in part on stabilization by roots and the presence of snags, logs, and other obstructions for creating stable surface area and a varied and complex substrate (Marzolf 1978, Benke et al. 1979, Bilby and Likens 1980). Moreover, if riparian vegetation removal is accompanied by clearcutting of the watershed, consequences may include greater pulses in discharge, higher amounts of annual runoff, and increased concentrations of nutrients and sediments (Likens et al. 1977, Bormann and Likens 1979). The shift to a higher energy, more eutrophic environment will produce conditions to which only a few of the existing species of aquatic invertebrates and fishes are adapted. Geomorphic changes in erosion and sedimentation may accelerate severalfold with these disruptions. Where alternatives to deforestation and land use changes are not possible, a protective buffer of riparian vegetation should remain intact to maintain the integrity of at least some of the energy sources and organic matter processing mechanisms.

NUTRIENT CYCLING

Nutrient cycling in riparian ecosystems can influence the water quality of streams and rivers. Riparian ecosystems along small, low order streams are buffer zones where excessive nutrients and sediments from upland disturbances may be trapped and assimilated. For larger streams and rivers, overbank flow of water during flood events provides an opportunity for upstream flows to come in contact with the riparian ecosystem. In the absence of a vegetated riparian zone, water is exported downstream with little opportunity for nutrient assimilation and transformation, except that provided in stream channels. In comparison with most stream channels, floodplain forests have greater structural complexity due to the presence of more stable sediments, anastomosing roots, a layer of decomposing leaves and woody material on the forest floor, and complex topographic features.

Many of the mechanisms of nutrient conservation by riparian ecosystems are universal and differ little from those found in upland ecosystems. Where fundamental differences exist, they are related to (1) the influence that flooding and an "aquatic" phase has on restricting oxygen availability to soils and sediments, thus altering metabolic pathways of microbial communities, and (2) the aqueous transport system that provides pathways of exchange between stream channel and floodplain through lateral imports, sedimentation, and exports of elements. These mechanisms of nutrient assimilation and transformation are examined in detail.

Most nutrient cycling studies focus on nitrogen and phosphorus, and have been conducted in southeastern floodplain forests where the presence of relatively long hydroperiods and broad floodplains has considerable influence on water quality of streams and rivers. In arid riparian ecosystems, water quantity, rather than its quality, may be the overriding controlling factor in ecosystem processes.

Distribution of Nutrients

In forested ecosystems, the distribution of nutrients among ecosystem components and annual changes in nutrient content of these compartments tend to be proportional to the distribution and changes in biomass. High or low standing stocks of nutrients generally correspond with high or low standing stocks of organic matter in both wetland and upland forests. For example, data on phosphorus distribution in riverine forests show that the rank, from highest to lowest standing stocks of phosphorus, is usually (1) soil (total P to approximately 25 cm depth), (2) aboveground wood, (3) belowground wood, (4) canopy leaves, (5) litter layer, and (6) surface water (Table 14). Canopy leaves and other non-perennial structures such as flowers and fruits tend to be highly enriched in phosphorus concentration

Table 14. Distribution of phosphorus in riverine forests.

Component	g P/m^2			
	Prairie Cr., Fla.[a]	Cypress Strand Fla.[b]	Cache R., Ill.[c]	Creeping Swamp, N.C.[d]
Leaves	1.26	0.4[e]	1.22	1.2
Aboveground wood	3.52	3.6	5.09	5.45
Belowground (lateral roots)	---	6.2[f]	2.82	1.52[g]
Surface water	0.19	0.8	0.176	0.0095
Litter layer	---	2.1	---	0.45
Soil	46.6[h]	90.2[h]	119[i]	33.65

[a]Brown (1978); [b]Nessel (1978); [c]Mitsch (1978); [d]Yarbro (1979); [e]Annual litterfall; [f]3.2 to 23 cm depth; [g]Brinson et al. (1981b); [h]To 20 cm depth; [i]To 24 cm depth.

relative to other biomass components, particularly woody ones, but the total quantity per unit area is lower. Sediments represent a large proportion of the phosphorus capital of the ecosystem although only a small proportion of this is available for plant uptake at one time.

Major Flows in the Nutrient Cycle

Major nutrient flows that are most frequently studied are nutrient return from the canopy (as litterfall and stemflow), decomposition of the litter layer, increment in wood accumulation, and sedimentation. Shorter term flows, such as sediment-water exchanges, are discussed later. Taken alone, each of these pathways would give an incomplete picture of nutrient cycling. However, when similar pathways are compared for different ecosystems, patterns may emerge which provide information on overall ecosystem fertility. For example, phosphorus flows for riverine forests (Table 15) are higher than those for upland ecosystems and stillwater wetlands of similar latitudes (Brinson et al. 1980). There is a similar trend for nitrogen which tends to substantiate the importance of fluvial processes in maintaining the relatively high fertility of riverine forests.

Annual phosphorus uptake by stem wood also appears to correspond to phosphorus supply. For a cypress strand in Florida, phosphorus uptake in stem wood increased approximately threefold when nutrient rich sewage effluent was released into the ecosystem (Nessel 1978). As compared with other cypress-containing ecosystems that had lower fluvial inputs, a floodplain forest in Florida had greater stem wood production as measured by annual basal area increment (Brown 1978). However, because of the extremely low concentrations of phosphorus in stem wood, annual increments in phosphorus accumulation by this process tend to be quite low when compared to other major flows (Brown 1978, Nessel 1978, Yarbro 1979).

Table 15. Litterfall and aqueous flows of phosphorus from the canopy to the forest floor in riverine swamps.

Locality	Annual precip-itation (cm)	Litterfall (kg dry wt/ha)	kg P/ha·yr Litter-fall	Aqueous	Total return	Source
Tar River Swamp, N.C.	104.7	6428	5.38	1.55	6.93	Brinson et al. 1980
Creeping Swamp, N.C.	124	6010	3.29	1.6	4.9	Yarbro 1979
Prairie Creek, Fla.	--	5970	9.1	---	9.1	Brown 1978
Cache River, Ill.	105	3480	7.7	1.4	9.1	Mitsch et al. 1979a
Cypress strand, Fla.	105.3	8150	6.86	--	6.86	Nessel 1978

Release of nutrients by decomposition of leaf litter in riverine forests is usually sufficiently rapid that there is little or no accumulation from year to year. The "half time" of loss is the time, in years, that would be required for one-half of the dry weight to disappear by decomposition. Half times for deciduous leaves range from less than 0.5 year to greater than 1.5 years (Table 16), while woody material and Pinus spp. leaves decompose more slowly and have longer half times. Stagnant backwater areas and depressions of floodplains tend to accumulate litter and sometimes peat. In spite of these exceptions, most of the nutrients of the litter layer appear to be recycled on an annual time scale. However, some studies have shown immobilization of nitrogen and phosphorus that may continue for several months (Figure 11), particularly under flooded conditions during the cool season that follows autumn leaf fall in temperate zones (Brinson 1977). This suggests a capacity for short-term accumulation of nutrients from the water, and thus an influence on water quality, even during the dormant season when losses of dissolved nutrients due to flooding might be greatest.

Sedimentation of particulate material on floodplains has been documented in a number of studies (Table 17). Although these data tend to be biased by not considering erosion and scouring as well, considerable quantities of sediment may accumulate over large areas, particularly during large flood events of low recurrence intervals. Estimates of annual phosphorus deposition by sedimentation range between 1.72 kg P/ha for a clear stream floodplain in North Carolina (Yarbro 1979) to 30 kg P/ha for a floodplain swamp in Florida (Brown 1978). These sedimentation rates approach or exceed some of the fluxes first described, although not all of the sediment is immediately available in ionic forms for plant uptake. Nevertheless, sedimentation represents a nutrient source that would otherwise be transported down-

Table 16. Summary of decomposition rates of litter in riverine forests.

Forest type	Duration of measurement (weeks)	Litter type	Site	mm mesh	Half times of loss, years[a]	Reference
Cypress strand, Fla.	52	Site litter, leafy	Forest floor	0.8	0.81	Burns 1978
				1.6	0.50	" "
			Debris pile	0.8	0.92	" "
				1.6	1.00	" "
	52	Site litter, woody	Forest floor	0.8	1.54	" "
				1.6	1.33	" "
			Debris pile	0.8	0.80	" "
				1.6	1.78	" "
Cypress strand, Fla.	51	Site litter	Flooded 0% time	1.6	1.47	Duever et al.
			Flooded 50% time	1.6	3.01	1975
			Flooded 61% time	1.6	2.31	" "
Cypress strand, Fla.	52	*Taxodium ascendens* lvs	Wet site	1.6	1.26	Nessel 1978
			Dry site	1.6	1.51	" "
	52	*Nyssa sylvatica* lvs	Wet site	1.6	0.82	" "
			Dry site	1.6	0.91	" "
	52	*Acer rubrum* lvs	Wet site	1.6	1.36	" "
			Dry site	1.6	0.95	" "
Alluvial swamp, N.C.	48	*Nyssa aquatica* lvs		1.6	0.37	Brinson 1977
	48	*Nyssa aquatica* twigs		1.6	2.48	" "
Beaver pond, Alberta	75	*Salix* sp. lvs		3.5	0.71	Hodkinson 1975
	75	*Juncus tracyi* lvs		3.5	1.69	" "
	75	*Pinus contorta* lvs		3.5	3.30	" "
	75	*Deschampsia cespitosa* lvs		3.5	1.03	" "
Mixed floodplain forest, Mich.	50	*Fraxinus nigra* lvs		0.05	0.64	Merritt &
				0.5	0.41	Lawson 1978
				8.0	0.14	" "

[a] Half time is the time required for disappearance of one half of the dry weight, according to the exponential decay formula $X/X_0 = e^{-kt}$ where X_0 is the dry weight initially present and X the dry weight remaining at the end of the measurement period, t, in years. Half time is calculated as 0.693/k.

Table 17. Sedimentation rates of phosphorus in the floodplains of riverine forests.

Locality	Sedimentation rate	Kg/ha	Source
Cache River, Ill.	3.6 g P/m^2 contributed by flood as sedimentation for flood of 1.13 yr recurrence interval	36	Mitsch et al. 1979a
Prairie Creek, Fla.	3.25 g P/m^2 yr as sedimentation from river overflow	32.5	Brown 1978
Creeping Swamp, N.C.	0.17 g P/m^2 yr sedimentation on floodplain floor from river overflow	1.72	Yarbro 1979
Creeping Swamp, N.C.	0.315-0.730 g P/m^2 yr based input-output budget of floodplain (most was filterable reactive phosphorus)	3.15-7.30	Yarbro 1979
Kankakee R.,	1.357 g P/m^2 contributed by unusually large spring flood lasting 62-80 days	13.6	Mitsch et al. 1979b

stream if the floodplain did not function as an area of deposition.

The magnitude and rate of nutrient uptake by vegetation, return to the forest floor as litterfall, and nutrient release by decomposition in southeastern floodplain forests suggest that they are capable of retaining nutrients by recycling them as fast or faster than most other forest types. Possession of a strong recycling component reduces the probability that nutrients entering the system will be lost by leaching from the soil and by export in throughflowing water. Sedimentation of phosphorus in the system provides evidence for sustained supplies of new material for recycling as long as inflow pathways are maintained (by channel overflow and flooding).

Soil-Water Nutrient Exchanges

When floodwaters come in contact with the soils of riverine forests or when runoff from uplands passes through the riparian zone to headwater streams, the relatively slow movement of these water masses provides an opportunity for mechanisms to function that may alter the nutrient constituents of the water. Nitrate (NO_3) is often the most abundant form of nitrogen in stream waters and, when present in high concentrations, contributes to water quality problems. When an anaerobic zone is present near the surface of poorly drained sediments,

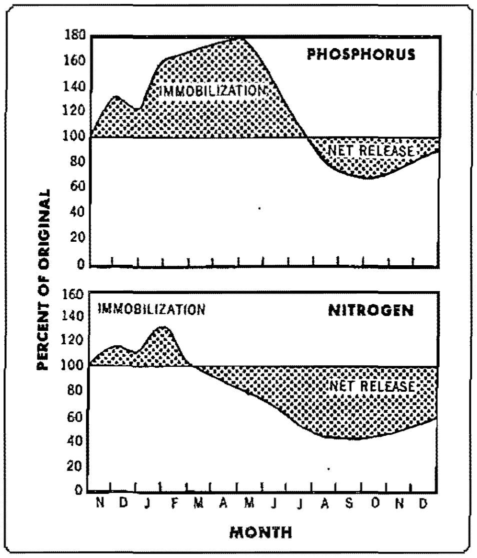


Figure 11. Immobilization of phosphorus and nitrogen by decaying leaf litter in an alluvial swamp. After Brinson (1977).

it profoundly affects the pathways of nitrogen. Denitrification ($NO_3 \longrightarrow N_2$) in anaerobic layers depends largely on the rate of nitrate supply. In the absence of external inputs of nitrate, it can be supplied internally by nitrification of ammonium ($NH_4 \longrightarrow NO_3$) under aerobic conditions. Patrick and Tusneem (1972) have proposed a scheme whereby ammonification (organic $N \longrightarrow NH_4$) in an anaerobic zone supplies, through diffusion, the substrate for nitrification in the aerobic surface layer. Diffusion of nitrate back to the reduced zone results in denitrification, so that the nitrogen gas (N_2) produced is not in a form that can contribute to water quality and eutrophication problems. These pathways are illustrated in Figure 12.

Evidence for denitrification is reported for the Santee River swamp in South Carolina (Kitchens et al. 1975). Concentration of nitrate progressively decreased from the river channel to the interior of the swamp backwaters, suggesting that increased contact time of overflow waters with the forest floor resulted in decreases in nitrate concentration, presumably by denitrification. More direct evidence is available from a cypress-tupelo swamp where amended nitrate concentrations decreased rather rapidly from surface water in contact with organic sediment (Brinson et al. 1981a). The sediments are a permanent sink for nitrate because it is denitrified when it diffuses to the anaerobic sediments.

Although natural rates of denitrification are difficult to determine, the potential for this process is high and can be sustained over protracted periods as long as anaerobic conditions are maintained and an energy source is available to drive the process. Consequently, poorly drained areas of riparian ecosystems can assimilate nitrate at rates well in excess of natural supplies, whether the source is from nitrogen-rich stream water in overbank flooding or is from nitrogen-rich runoff from adjacent agricultural land. In either situation, less nitrate would be exported to downstream aquatic ecosystems for possible eutrophication if the riparian zone is protected and natural hydrologic processes are allowed to operate.

Analysis of exports from watersheds containing riverine wetlands support these observations. For small coastal plain swamp streams in North Carolina, Kuenzler et al. (1977) showed that concentrations and exports of nitrate were considerably higher for channelized streams in which the forested wetlands had been circumvented than for natural streams in which considerable flooding occurred during high discharge.

Floodplain forests also show a high capacity for phosphorus retention and cycling. Yarbro (1979) developed a rather complete phosphorus budget for a swamp floodplain ecosystem in North Carolina (Figure 13). Inputs to the ecosystem, mostly from upstream inflows, exceeded outputs by 3.15 and 7.30 kg P/ha·yr for each of the 2 years of study which characterizes the floodplain as a phosphorus sink. Although most of the loss appeared to be from soluble reactive phosphorus in the water, there was a substantial amount of sedimentation (1.7 kg/ha·yr) of particulate forms. High forest floor/surface water exchanges substantiate the idea that the sediments are the principal site for transformation of various forms of phosphorus fractions. Transfers between the forest floor, the deeper soil, and woody

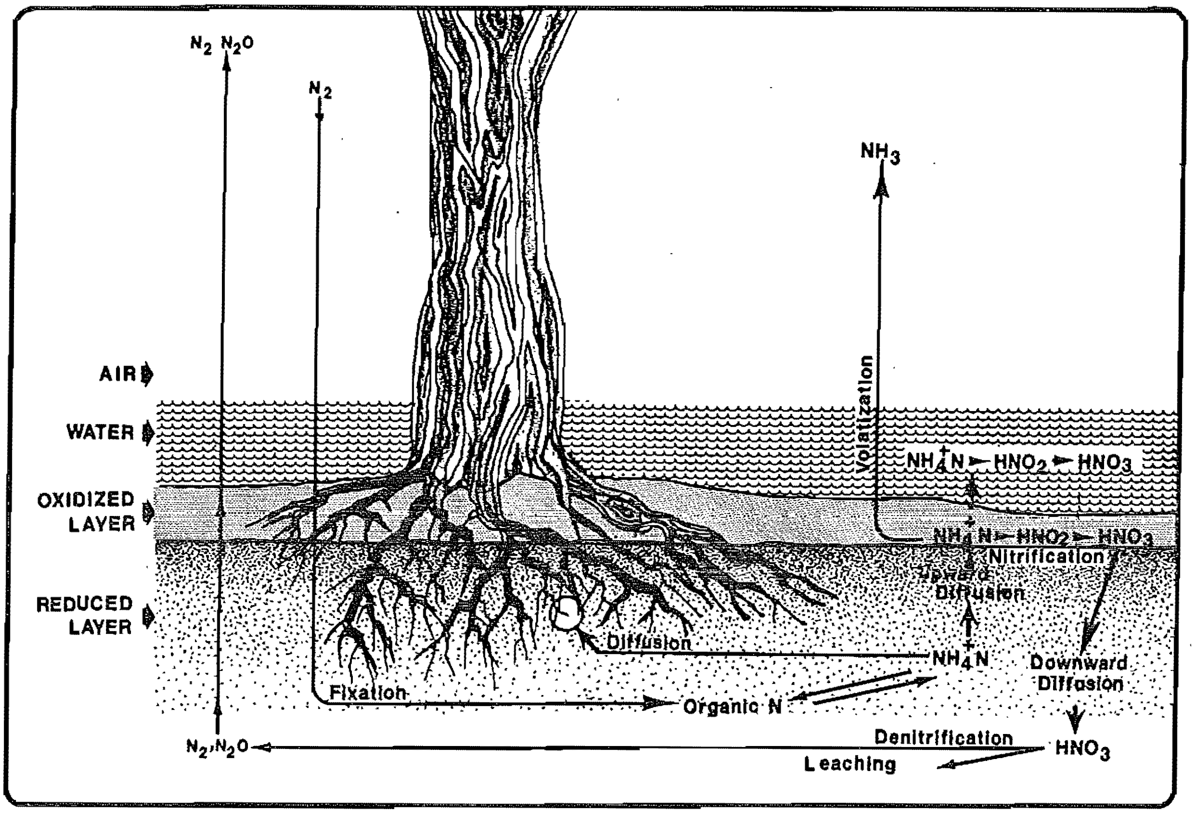
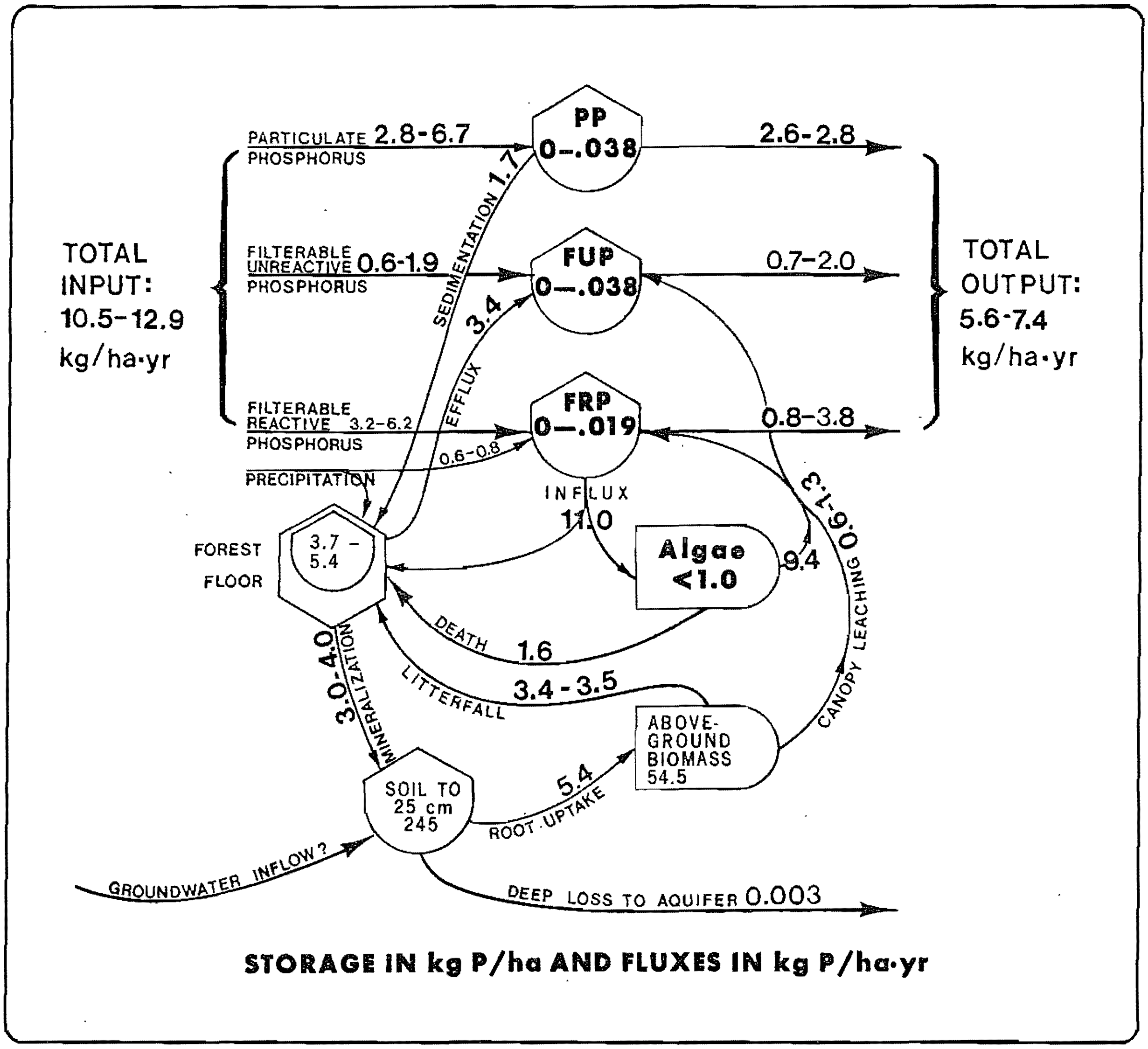


Figure 12. Pathways of nitrogen transformations in an oxidized and reduced sediment-water system. Modified from Gambrell and Patrick (1978).

biomass were shown to be approximately half those of surface water/forest floor exchanges. Estimates of tree wood increment (0.6-1.2 kg P/ha•yr) suggest that the vegetation would serve as a sink for phosphorus only if the forest were accumulating biomass. This is relatively small compared to the rate that phosphorus is recycled by the vegetation, which suggests rather tight coupling between litterfall from the canopy, decomposition of litter, and phosphorus uptake by roots. In the absence of a complex floodplain ecosystem, such as that which would result from stream channelization, there would be little opportunity for phosphorus recycling and sedimentation. Under channelized conditions, downstream exports would increase and the phosphorus would likely be made available to an aquatic ecosystem such as a lake or estuary.

The Significance of Hydroperiod and Nutrient Cycling

The importance of seasonal changes in water level and flow to nitrogen cycling can be illustrated by considering the annual cycle of an idealized stream-floodplain complex. The scenario begins with a major flood in the winter of a southeastern swamp forest (Figure 14). Suspended sediments and dissolved nutrients are transported from the stream into the floodplain where water velocity diminishes. Suspended sediments and the particulate forms of nitrogen that they contain settle to the forest floor and the dissolved nitrogen forms in the water diffuse to the soil to interact with detritus and sediment on the forest floor. Deciduous trees of the floodplain are dormant in the winter; consequently they are not then



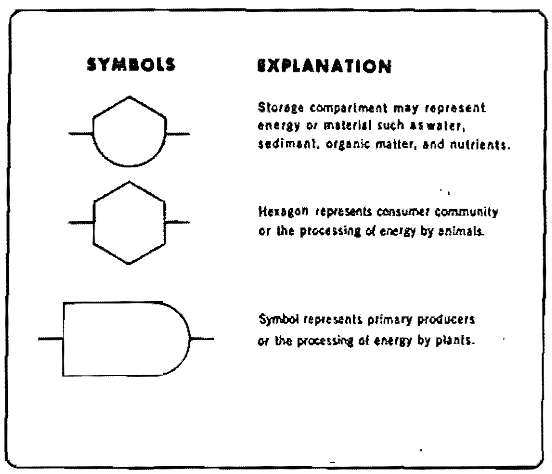


Figure 13. Phosphorus storages and fluxes in the Creeping Swamp floodplain ecosystem, North Carolina. Modified from Yarbro (1979). See text for explanation.

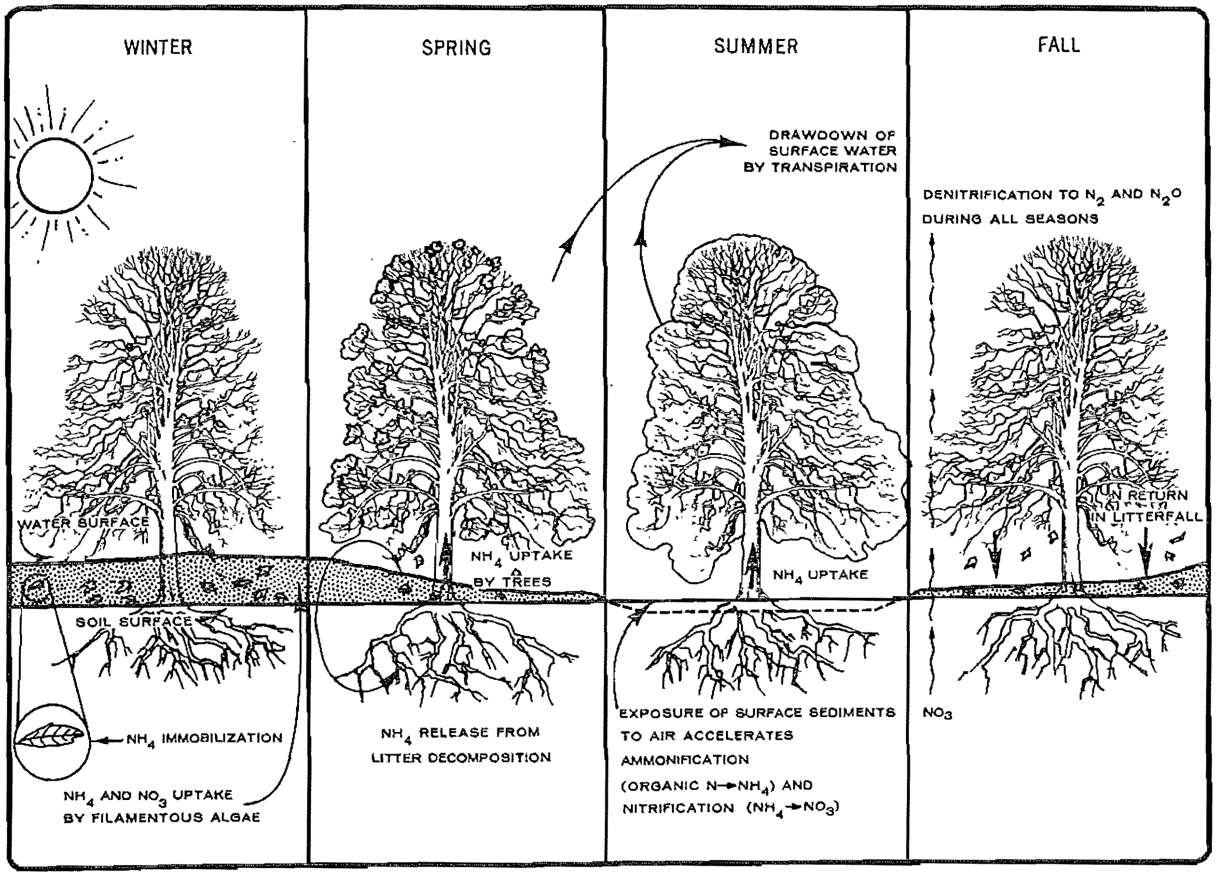


Figure 14. Seasonal phenology of a tupelo-cypress swamp showing mechanisms of nitrogen conservation and recycling.

capable of nutrient accumulation. Mechanisms of nutrient removal under these conditions may include (1) uptake by a community of filamentous algae that receives sufficient light for maintenance only when the forest canopy is leafless and (2) immobilization by decomposing microbes that are utilizing the carbon rich but nutrient poor leaf litter that fell during the previous autumn.

When the floodwaters warm in the spring, decomposition of detritus is enhanced, thereby releasing nutrients for plant uptake and growth. Appearance of leaves in the forest canopy shades the forest floor, resulting in death of the filamentous algae. Decomposition of the algae augments the nutrient supply for plant uptake. Evapotranspiration by the forest depresses the water level and eventually depletes most standing water. The seasonal events turn full cycle with leaf fall in autumn and resumption of flooding in the winter.

The timing of these seasonal events and the magnitude and mechanisms of nutrient cycling described more fully in the sections above illustrate two important features: (1) the high capacity of certain riparian forests to recycle nutrients such as nitrogen and phosphorus as compared with the generally lower rates at which they are imported from outside the system, and (2) the influence that contact with the forest floor has on the nutrients in flood water. The mechanisms just discussed describe how floodplain forests can capitalize on and utilize these inputs.

Of course the potential for these interactions to occur depends on the hydroperiod or the length of time and the quantity of water and nutrients that come in contact with the floodplain. Many southeastern river swamps tend to have geomorphic, hydrologic, and climatic characteristics that are optimal for strong coupling between streams and floodplains.

Measures to control flooding or speed the conveyance of water downstream tend to deprive riparian ecosystems of the influx of materials that sustain their nutrient-rich properties. When drained and deprived of flooding by streams, it is likely that disrupted riparian ecosystems will become sources, rather than sinks, of nutrients and sediments for ecosystems downstream due to the elimination of specialized nutrient tranformations that depend on an "aquatic" phase. Drainage will convert them from systems characterized by lateral inputs and outputs to ones of vertical movement and downward leaching. Downstream ecosystems must then adapt to receiving altered rates of organic matter and inorganic nutrient supply. Changes in nutrient regimes represent only one example of a host of other effects on riparian ecosystems when they are altered.

DIVERSITY AMONG FLOODPLAIN ECOSYSTEMS

A great deal of emphasis has been placed on the similarity among riverine ecosystems in the material above. The underlying theme is that ecosystem structure and organization is the result of the energy and pattern of delivery of flowing water. Hydrologic and geomorphic factors, both in the riparian zone and in the watershed, appear to have a fundamental influence on differences observed among riparian ecosystems. It is the differences, particularly in vegetation and factors that influence vegetation, that will be discussed below.

Although no attempt is made to present a formal classification for riparian ecosystems, broad distinctions exist among them that fall into useful categories. Differences in climate, in spite of the local edaphic properties of floodplains, have an influence on species composition of the plant community. Whether a stream channel is composed of bedrock or passes over alluvial fill will greatly influence the dimensions of the riparian zone. Within a given floodplain plant community, abrupt changes in stream channel adjustment and catastrophic flood events can be so prevalent and recurrent that the community is maintained in an early stage of succession.

Climate

The transition from humid to arid climates does not have nearly the control on the structure of riparian forests as it does on that of upland ecosystems (Figure 8). Presumably this is due to the fact that floodplains capture runoff water that is exported from upland regions and at least part of that water is available for riparian vegetation. The line or isopleth separating areas of less and greater than 2.5 cm runoff annually in the central U.S.A. shows good agreement with the separation between wet and dry climatic zones (Figure 15). Where runoff is less than 2.5 cm/year there is a greater probability of encountering intermittent streams in relatively large drainage basins than is true in more humid climates. As a result, riparian vegetation may be subjected to water deficiency as well as water excess resulting in corresponding changes in species composition. In floodplains of arid regions, plants that can tolerate periods of drought by ex-

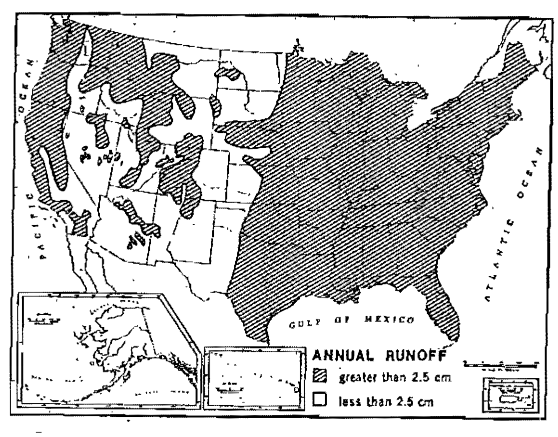


Figure 15. Map showing 2.5 cm isopleth of annual runoff. From Langbein et al. (1949).

tending roots to the water table (phreatophytes) and also withstand flooding are the most likely to survive. Under humid climates, water is much more readily available to plant communities in floodplains and species composition will correspond accordingly.

In the western U.S.A. where the temperature at higher elevations has a large influence on water balance, the 2.5 cm isopleth circumscribes many of the mountainous areas. The extent to which this water supply is available to floodplain vegetation at lower elevations depends largely on the amount of discharge relative to the volume of alluvial fill. The ratio of evapotranspiration to precipitation increases with decreasing altitude under most circumstances. It is possible for runoff from mountainous areas to be lost as evapotranspiration or in groundwater flow by the time it reaches lower altitudes (Thomsen and Schumann 1968). In areas where there is little alluvial fill for water storage (e.g., steep rocky ravines), xeric conditions prevail and vegetation may differ little from the surrounding uplands (Zimmermann 1969).

One of the major features that distinguishes certain arctic drainage basins from those in warmer climates is the impermeable layer of frozen ground (permafrost). As a result, runoff is from the soil surface so that groundwater infiltration and storage play an insignificant role in hydrologic patterns. Permafrost also affects channel stability and morphology. For example downcutting of the river channel may be retarded because the streambed remains frozen during much of the ice-free period. Although it has been established that rooted vegetation along stream banks retards erosion due to the binding capacity of roots (Smith 1976), the insulating effects of vegetation in permafrost regions may be more significant in maintaining stream banks in a consolidated, frozen state. On the Porcupine River removal of river bank vegetation increased the depth to summer thaw from a maximum of 0.3 m with vegetation cover to as much as 1 m in cleared areas (Cooper and Hollingshead 1973).

Underfit Streams and Downcutting Channels

Within the context of differences imposed by climate on riparian ecosystems, a further distinction can be made between stream systems with bedrock controlled channels and those with alluvial channels. The latter, referred to as "underfit" streams (Dury 1964a, 1964b, 1965), may have extensive floodplains, and are free to adjust their dimensions, shape, and gradient in response to hydraulic changes. Their channel bed and banks are composed of material transported by the river under present flow conditions. By comparison, bedrock controlled channels are confined between rock outcrops, and in extreme cases, have virtually no floodplain so that only a very narrow margin can be considered riparian. Of course a given stream may have alternating sectors of both conditions which makes generalizations difficult. However, the distinction is important when considering the values and attributes of riparian ecosystems and their plant and animal communities.

Most of the ecosystems described in the previous sections are those with clearly distinguishable floodplains and can be categorized in the underfit stream type. However, even in the absence of distinct floodplains, streamside plant communities are usually distinguishable from upland communities in species composition, moisture availability, and physiognomy. They represent the riparian zone, although usually quite narrow compared with floodplains, that has an abundant water supply, is characterized by fluctuating water levels, and is exposed to the abrasive force of flowing water during floods. In the situation of lower order streams that have considerable canopy cover, the importance of leaf fall has been described as essential to maintaining instream energy flow and fish production (pages 70 - 71). Some of the woody riparian communities that will be described below occupy stream margin environments that cannot be considered floodplains.

Influence of Catastrophic Forces

Without doubt, the species composition of riparian ecosystems is a response to multiple factors that are in some way related to hydroperiod and the energy of flowing water. However, in many cases more catastrophic forces create abrupt episodes of severe and destructive stress that dominate community development. Major floods may eliminate large stands of forest by erosion and bank undercutting, creation of new channels, and burial under deep deposits of sediment. Wolman and Leopold (1957) report that the Kosi River in India migrates across its valley at the rate of 750 m/yr. The disordering effects of these events serve to maintain an array of community types in floodplains that would otherwise mature into more homogeneous, even aged stands. Vogl (1980) cites numerous examples of perturbation-dependent ecosystems where the maintenance of certain species is assured by catastrophic events such as floods, wind, storms, fire, volcanism, and glaciation.

The abrasive force of ice can be particularly destructive to vegetation when ice floes occur in combination with flooding. Damaged and partially buried trees in floodplains can be used to reconstruct past flood events (Sigafoos 1964). In Alaska and other areas of high latitude, the paucity of large woody vegetation is possibly due to ice stress and partly due to massive outburst floods from glacier dammed lakes (Post and Mayo 1971). The spectacular annual floods from 1918-1963 from Lake George into the Knik River, Alaska were so regular that the area was designated as a Natural Landmark by the National Park Service (Post and Mayo 1971). It is doubtful if many vascular plants are able to survive this stress in the riparian zone. However, Brice (1971) cites an example where balsam poplar trees survived burial to 2.4 m depth and later scour that exhumed the trees. The nearly ubiquitous occurrence of young, even aged stands of willow and cottonwood on point bars and river islands are evidence of new or renewed environments created by sediment redistribution (Lindsey et al. 1961). Thus, the diversity of vegetation both within and among floodplains is dependent, in part, on episodes of destructive hydrologic forces.

Ecological Succession

Some reports of ecological succession in riparian ecosystems have suggested that open water features will progressively fill in with sediments and eventually develop into a mixed hardwood forest community. This is often interpreted to mean that the floodplain ecosystem is always approaching some static and idealized climax condition. This perception is often in error given the dynamically changing nature of most riparian ecosystems. Point bars of migrating meanders of streams continually create new conditions for pioneer communities to become established. If the stream is in a mode of downcutting through floodplain alluvium, terraces will form and become isolated from the effects of hydroperiod. In the absence of more frequent flooding, species composition will gravitate toward less flood tolerant species. If the stream channel is undergoing aggradation, backwater areas will become less well drained and be replaced gradually by a community of species more tolerant to flooding. On the other hand, increases in flow and sediment deposition, such as that experienced by the Atchafalaya River in the past two decades, may result in massive amounts of siltation, a process that leads to better drained and more elevated conditions (O'Neil et al. 1975). Catastrophic floods and ice floes uproot and prune vegetation providing "open" conditions for species of plants and animals adapted to rapid population growth and resource exploitation (Lindsey et al. 1961, Sigafoos 1964).

Increasing beaver activity in the last two decades, particularly in the bottomland hardwood forests of the Southeast, have demonstrated the impact that small changes in hydroperiod can have on forest communities. It is probable that elimination of original beaver populations reduced the heterogeneity of floodplain forests and created the more uniform forests that are generally perceived to be the natural condition.

In the absence of hydrologic and geomorphic changes in a floodplain, there is some evidence that secondary succession will occur more rapidly in floodplains than it does in upland areas. For example, Frye and Quinn (1979) found that the rate of forest development on high floodplain areas of the Raritan River, New Jersey, occurred more rapidly than in nearby upland sites. The floodplain showed greater species diversity, equitability, basal area, mean stem diameter, and tree height.

Thus, changes in riparian ecosystems can be subtle and slow or catastrophic and abrupt, but seldom are they as directional as the classical aquatic-to-terrestrial models of ecological succession would imply. Since a multiplicity of factors are involved in community development, the probability is low that these will remain static in a natural floodplain. Some manipulations by humans tend to accelerate changes while others mute the forces that are responsible for the maintenance of cyclic phenomena. Since riparian ecosystems are subjected at different times to a variety of hydrologic regimes, geomorphic processes, and catastrophic forces, generalizations to broad geographic areas are sometimes difficult to apply to site specific situations. Climate and biogeography ultimately play a critical role in species composition of floodplain communities.

Description of Plant Communities

The species composition of some common riparian plant communities in the United States will be described by geographical regions (Figure 16). This is not a classification system for riparian vegetation, but merely an overview of the dominant species that are most likely to be encountered in each of the ecoregions (Teskey and Hinckley 1977a, 1977b, 1978a, 1978b, 1978c; Walters et al. 1980a, 1980b). That similar species and genera recur for many regions is not surprising; it merely confirms that the environmental conditions shared by these ecosystems may be more important than climatic differences.

Southern Forest Region. Bottomland hardwood forests are located in the floodplains along major and minor streams of the Southeast. Vegetation varies from communities adapted to extremely long hydroperiods, such as the water tupelo-baldcypress association, to oak-hickory communities of "second bottom" forests, some of which may not flood annually (Figure 17). If the stream channel has undergone recent reorientation, newly formed point bars and levee deposits may support monospecific stands of willow (*Salix* spp.) and mixtures of this and cottonwood (*Populus heterophylla*), river birch (*Betula nigra*) and silver maple (*Acer saccharinum*). If the river channel remains stable, species composition may change to that normally found at higher elevations because the coarsely textured sediments drain rapidly after saturation.

Areas in deeper depressions that have long hydroperiods, such as sloughs and oxbows, will develop water tupelo (*Nyssa aquatica*), baldcypress (*Taxodium distichum*), and frequently water elm (*Planera aquatica*). Communities where overcup oak (*Quercus lyrata*) and water hickory (*Carya aquatica*) occur are usually among the next most poorly drained sites. With even shorter hydroperiods, laurel oak (*Q. laurifolia*), hackberry (*Celtis laevigata* and *C. occidentalis*), red maple (*A. rubrum*), American elm (*Ulmus americana*) and green ash (*Fraxinus pennsylvanica*) may be common. Low ridges in the first bottom may be dominated by sweetgum (*Liquidambar styraciflua*) while higher ridges that have quite short hydroperiods may be occupied by blackgum (*N. sylvatica*), hickories (*Carya* spp.), and white oak (*Q. alba*).

The flats of the second bottom are likely to have poorer internal drainage than the high ridges of the first bottom. As a result the species composition may appear similar to that of the low ridges of the first bottom. Where cherrybark oak (*Q. falcata* var. *pagodaefolia*), swamp chestnut oak (*Q. michauxii*), and water oak (*Q. nigra*) occur, hydroperiods are among the shortest or drainage the best among all bottomland sites. Live oak (*Q. virginiana*) and loblolly pine (*Pinus taeda*) are usually confined to the highest "islands" in floodplain topography.

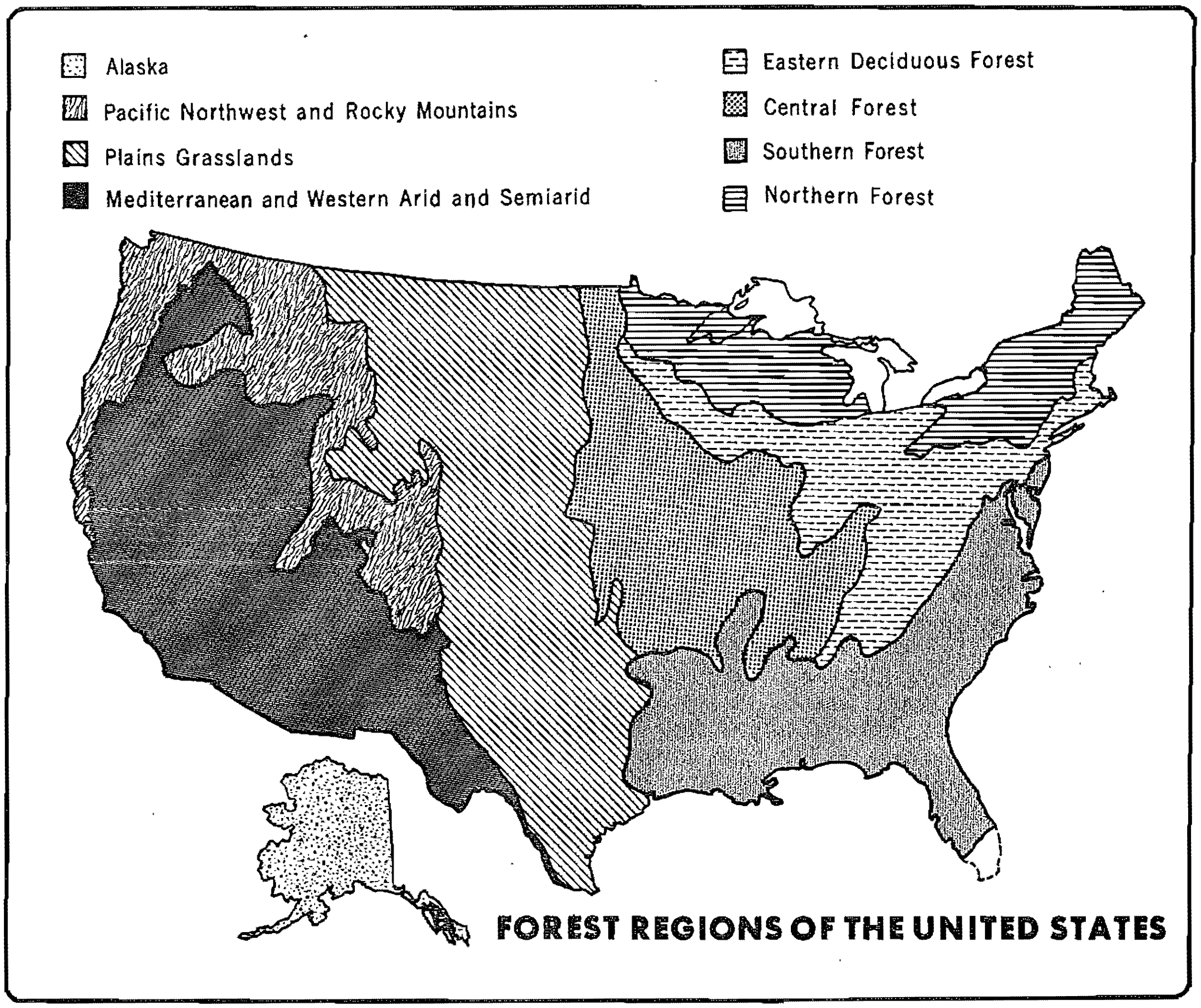


Figure 16. Forest regions of the United States for which riparian plant communities are described. Terminology after Bailey (1976).

So few virgin bottomland hardwood stands now exist that cyclic changes in ancient stands are difficult to reconstruct. In the Congaree Swamp of South Carolina, where 11 distinct communities can be delineated, Gaddy et al. (1975) suggest that shade tolerant hardwoods such as laurel oak eventually overtop sweetgum and other hardwoods for protracted periods of time. Tree fall is offered as a mechanism to create canopy openings so that a mosaic pattern of communities on the floodplain is maintained.

Point bar deposition and other new land forms are initially stocked with cottonwood and willow. These are succeeded by silver maple, ash, elm, and boxelder (A. negundo), a community which may persist indefinitely in southern Illinois (Hosner and Minckler 1965). For more poorly drained sites of the same region, secondary succession has been observed to be initiated by buttonbush (Cephalanthus occidentalis), cottonwood, swamp privet (Forestiera acuminata), cypress, water tupelo, willow, green ash, and pumpkin ash (Fraxinus caroliniana). According to Hosner and Minckler (1965), further fluvial deposition or other events that lead to improved drainage will result in replacement of this community by species found

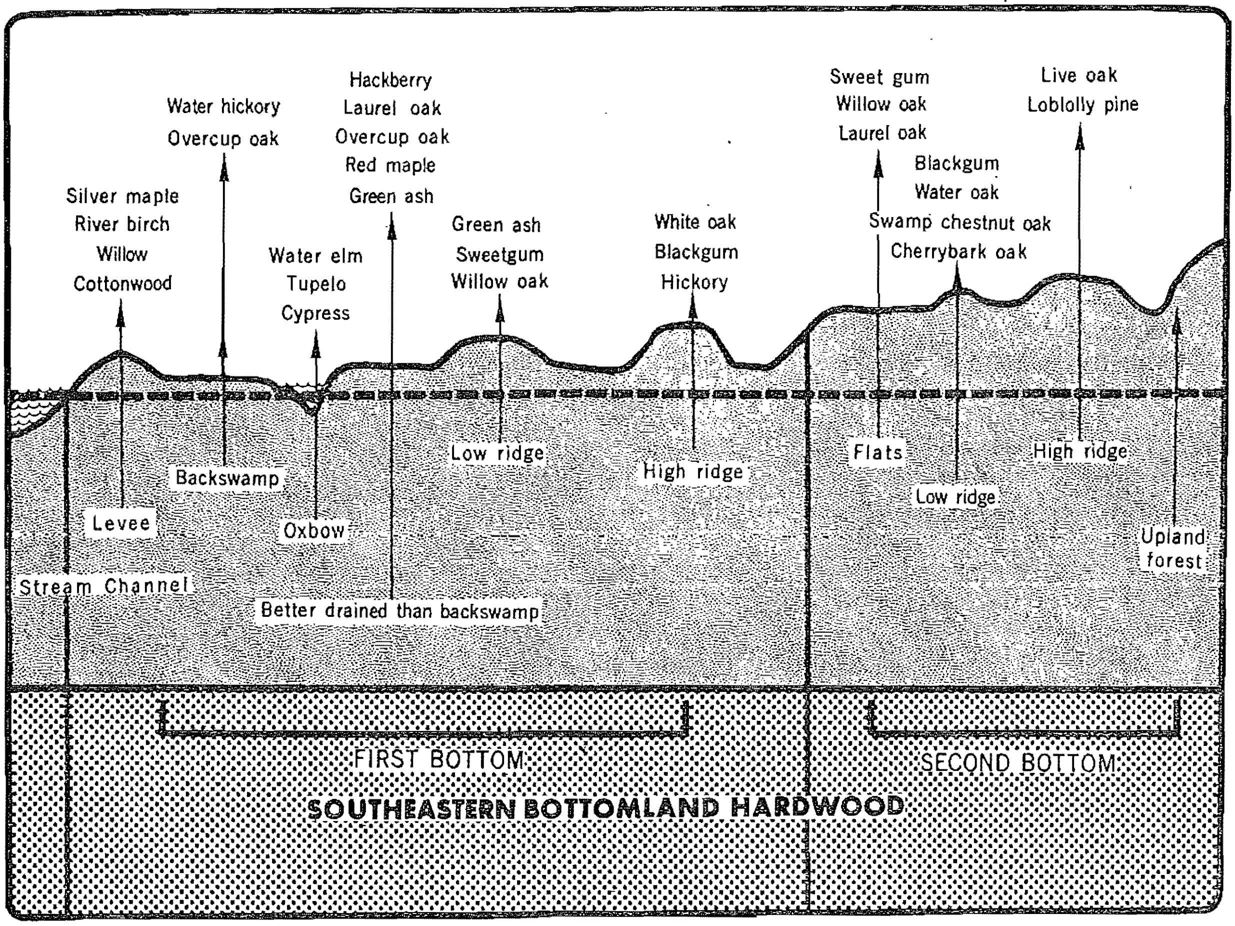


Figure 17. Idealized profile of species associations in southeastern bottomland hardwood forests. After Wharton (1978).

on successively better drained sites (Figure 17).

In narrow bottoms of small streams where the alluvial soils may be moderately well drained, cypress and tupelo generally are absent. The mixture of tree species includes those from the large bottomlands discussed above, from moist coves, and from mesic uplands (Golden 1979). After agricultural abandonment there is a distinct trend toward dominance by light seeded hardwoods [sweetgum, red maple, tulip poplar (Liriodendron tulipifera)] that is provided by mature individuals in uncut strips left over from incomplete clearing for agriculture.

The geographic distribution of baldcypress corresponds approximately with the distribution of southern floodplain forests. However, baldcypress is not an important component of many of the major floodplains since it tends to be restricted to the wettest and most deeply flooded conditions. Some of the most extensive floodplain areas are along the lower Mississippi River as well as large tributaries such as the Arkansas, Red, Ouachita, Yazoo, and St. Francis Rivers. Some of the larger rivers draining in a southerly direction into the Gulf of Mexico are the Pearl, Tombigbee, Alabama, Pascagoula, Chattahoochee, Apalachicola, and the Suwanee Rivers. Those draining from the south Atlantic coast in a southeasterly direction include the Altamaha, Ogeechee, Santee-Cooper, Pee Dee, Cape Fear, Neuse, and Roanoke Rivers.

Central Forest Region. Bottomland forests in this region have strong affinities with those described for adjacent regions (Figure 16). For example, the studies by Hosner and Minckler (1965) in southern Illinois have already been used to characterize the floodplain vegetation of the Southern Forest Region. Robertson et al. (1978) show that the southern floodplain forest type extends up the Mississippi valley to southern Illinois and further northward up the Ohio and Wabash Rivers. To the east, studies by Lindsey et al. (1961) conducted on the Wabash River are equally applicable to the Eastern Deciduous Forest Region and are discussed below. The western part of the Central Forest Region approaches areas where floodplain forests in the Plains Grassland Region have been studied intensively (see below). The admixture of floral components from the south, east and west in the Central Forest Region makes generalizations about riparian vegetation and community succession difficult.

In central Illinois, the vegetation along the Sangamon River illustrates the rapid transition from floodplains to upland forests in species composition, biomass, and annual biomass accumulation (Table 18). Silver maple is clearly dominant in the floodplain, shingle oak (*Quercus imbricaria*) and hackberry (*Celtis occidentalis*) are codominants in the transition zone, and white oak dominates the upland community. Total tree biomass and estimated net biomass accumulation were greatest in the floodplain followed by the upland and transition stands. Dutch elm disease and phloem

Table 18. Tree biomass, net annual accumulation, and distribution among species (%) for a floodplain, transition site and upland along a stream in Illinois. Biomass percentages less than 2% are omitted. From Johnson and Bell (1976).

Species	Percent of total biomass		
	Floodplain	Transition	Upland
Acer saccharinum	73.6	15.7	--
Gleditsia triacanthos	10.9	--	--
Fraxinus pennsylvanica	9.4	--	--
Platanus occidentalis	3.6	--	--
Euonymus atropurpureus	--	9.3	--
Quercus imbricaria	--	22.3	--
Carya cordiformis	--	2.2	--
Celtis occidentalis	--	27.5	--
Prunus serotina	--	4.7	--
Ulmus rubra	--	6.2	--
Ulmus americana	--	5.8	3.3
Quercus velutina	--	--	6.2
Quercus alba	--	--	84.9
Total tree biomass (t/ha)	289	135	227
Estimated net biomass accumulation of trees (t/ha·yr)	11.5	7.0	10.0
Frequency of flooding	3-25%	0.5-3%	0.5%

necrosis have contributed to low biomass of the transition zone by eliminating all large elm, which probably dominated the zone prior to 1950 (Johnson and Bell 1976).

Eastern Deciduous Forest Region. Floodplain forests in this region range from those located along small to moderate sized streams draining the Appalachians to rivers that are relatively large by the time they pass through the region. Some of these larger rivers include the upper Mississippi, Ohio, Susquehanna, Potomac, and Delaware. Because of this diversity, generalizations on riparian vegetation are difficult to make.

The most intensively studied floodplain forests are those on the Wabash and Tippecanoe Rivers in Indiana (Lindsey et al. 1961, Schmelz and Lindsey 1965) which could be included in the Central Forest Region just discussed since a few of the study sites are located there. First bottoms of the floodplains tend to be dominated by black willow (Salix nigra), American elm, and cottonwood. Second bottoms that are infrequently flooded are heavily represented by sugar maple (Acer saccharum), beech (Fagus grandifolia), American elm, redbud (Cercis canadensis), buckeye (Aesculus glabra) as well as 16 other species exceeding 10 cm dbh.

In stands on the floodplain of the Raritan River, New Jersey, Buell and Wistendahl (1955) mention 14 woody species. On the inner floodplain where erosion has produced a series of ridges and poorly drained sloughs, silver maple was the dominant tree, followed by American elm, red maple, and white ash (Fraxinus americana). In less frequently inundated and less severely scoured portions of the floodplain, beech and tulip poplar were abundant along with silver maple.

By comparison, the narrow floodplains of the Little Tennessee River system in the Appalachians of western North Carolina are dominated by river birch (Wolfe and Pittillo 1977). Other common species are wild cherry (Prunus serotina), red maple, black locust (Robinia pseudo-acacia), and tulip poplar.

Successional development on new sites created by stream migration, as described for Wissahickon Creek in southeastern Pennsylvania, may be initiated by silver maple and sycamore (Platanus occidentalis) following the herbaceous ragweed cover (Sollers 1973). This is replaced by a community dominated by white ash, American elm, red maple, black walnut (Juglans nigra), and spicebush (Lindera benzoin). With improved drainage, oak-hickory stands eventually develop. Highest bottoms, or areas which are inundated only by the most severe floods, are dominated by typical upland species (Lindsey et al. 1961). The species composition of stands at this stage will depend heavily on the composition of upland forests. Because of the great diversity in flora throughout the Eastern Deciduous Forest Region, there will be a great deal of geographic variation in the species composition of well drained riparian forests.

Northern Forest Region. Riparian forests in this region have received little study, possibly because attention has been diverted to extensive peat bogs located in the western portion. In comparison with the other regions, rivers tend to be small because many represent either headwater drainages of the Mississippi River or terminate in the Great Lakes after flowing a short distance. The Hudson and Connecticut Rivers in New England are exceptions.

In blanket peat areas where mineral rich soil and distinctive water flow occur, riparian communities develop that differ from the surrounding low-lying shrub and sphagnum bog areas. Heinselman (1970) describes these areas with water flow as rich swamp forest. They have high densities of northern white cedar (Thuja occidentalis) which may be overtopped by black ash (Fraxinus nigra), larch (Larix laricina) or black spruce (Picea mariana). Except where white cedar is dense, speckled alder (Alnus rugosa) forms a shrub layer. Speckled alder and black ash usually disappear in transition from marginal fen to poorer swamp where water flow is more sluggish, water is less mineral rich, and peat is deeper and contains less inorganic matter. Where more apparent floodplain features exist and there is little peat accumulation, American elm may play a larger role, although black ash is still important (Janssen 1967).

In the riparian forests along the Susquehanna, Chemung, and Delaware Rivers in Applachian Uplands of New York, there are five characteristic floodplain features that influence the species composition of plant communities (Morris 1977, Morris et al. 1978). They include:

1. Floodbasins with poorly drained silts and high organic matter content that are dominated by willows, silver maple, cottonwood and wild cherry.

2. Point bars and stream confluence areas with well drained silts that lack willow but have, in addition to the species listed above, sycamore and ash.

3. Frequently and destructively flooded point bars and confluence areas of sand and silt mixtures that support black locust, silver maple, sugar maple, and American elm.

4. Less frequently flooded stable point bars of coarsely textured sands that support hickories, in addition to the two maple species.

5. Seldom flooded Pleistocene terraces where pines, oaks, red maple, and wild cherry dominate.

Plains Grassland Region. As precipitation decreases from the eastern to western U.S.A., the isopleth of runoff reaches 2.5 cm per year in this region, a value arbitrarily chosen to distinguish between humid and arid riparian ecosystems. Since the natural upland vegetation is usually savanna, riparian zones become conspicuous features of the landscape. Some of the major rivers that cross this area are the Missouri, Platte, upper Arkansas, and Canadian Rivers.

One of the greatest floristic differences between arid floodplains and those of the more eastern regions is the general absence of oak species, a group that is particularly abundant in the bottomland hardwood forests of the Southern Forest Region and further north in the Mississippi valley area of the Central Forest Region.

Transitions due to moisture are particularly well illustrated in Oklahoma where Bruner (1931) distinguished between the riparian vegetation of the eastern, central, and western parts of the state. Species that occur in more than one of the parts show decreasing height in the east to west gradient (Figure 18). In the east, continuous flow of even smaller streams supports forests rich in species of trees, shrubs, vines and herbs. Baldcypress, sweetgum, sycamore, river birch, and black gum are common. Dominants of the central Oklahoma floodplains, such as elms, hackberry, walnut, black locust, and honey locust (Gleditsia triacan-

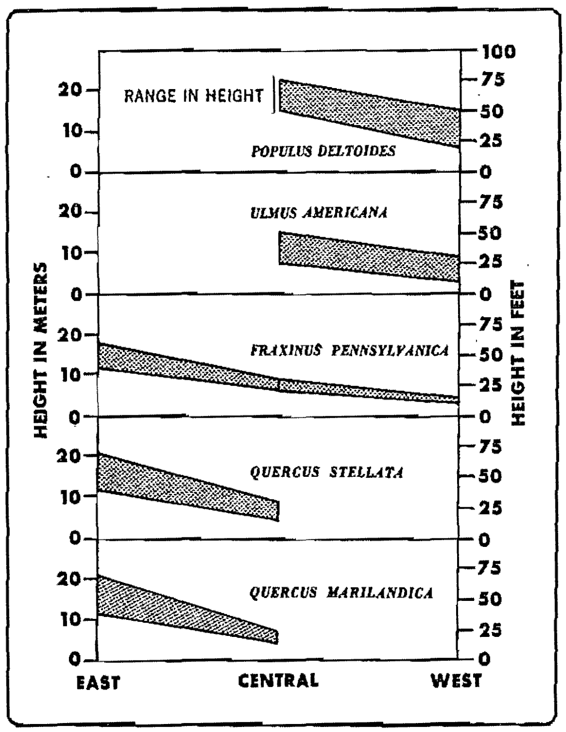


Figure 18. Changes in height and species composition of floodplain forest stands in a west to east gradient in Oklahoma. Adapted from Bruner (1931).

thos), occur also in the east and augment the species diversity there. In the arid west, trees are usually rather widely spaced and neither willows nor cottonwoods reach the stature that they attain eastward. Elm and boxelder are usually found only in valleys or near streams where the water supply is constant. With only a 2 degree change in longitude but a 24 cm change in precipitation in central Oklahoma, floodplain tree species increase from 11 in the west to 23 in the east (Rice 1965).

In the Missouri River floodplain of North Dakota where floodplain width varies from about 1 to 11 km, three forest types can be distinguished (Figure 19) (Keammerer et al. 1975, Johnson et al. 1976). On the lowest and most frequently flooded area, young cottonwood-willow forests have many small trees 6-12 m tall but have few other woody species. At higher elevations, forests consist of older cottonwood whose tall open canopies overtop bur oak (*Q. macrocarpa*) and boxelder. At the

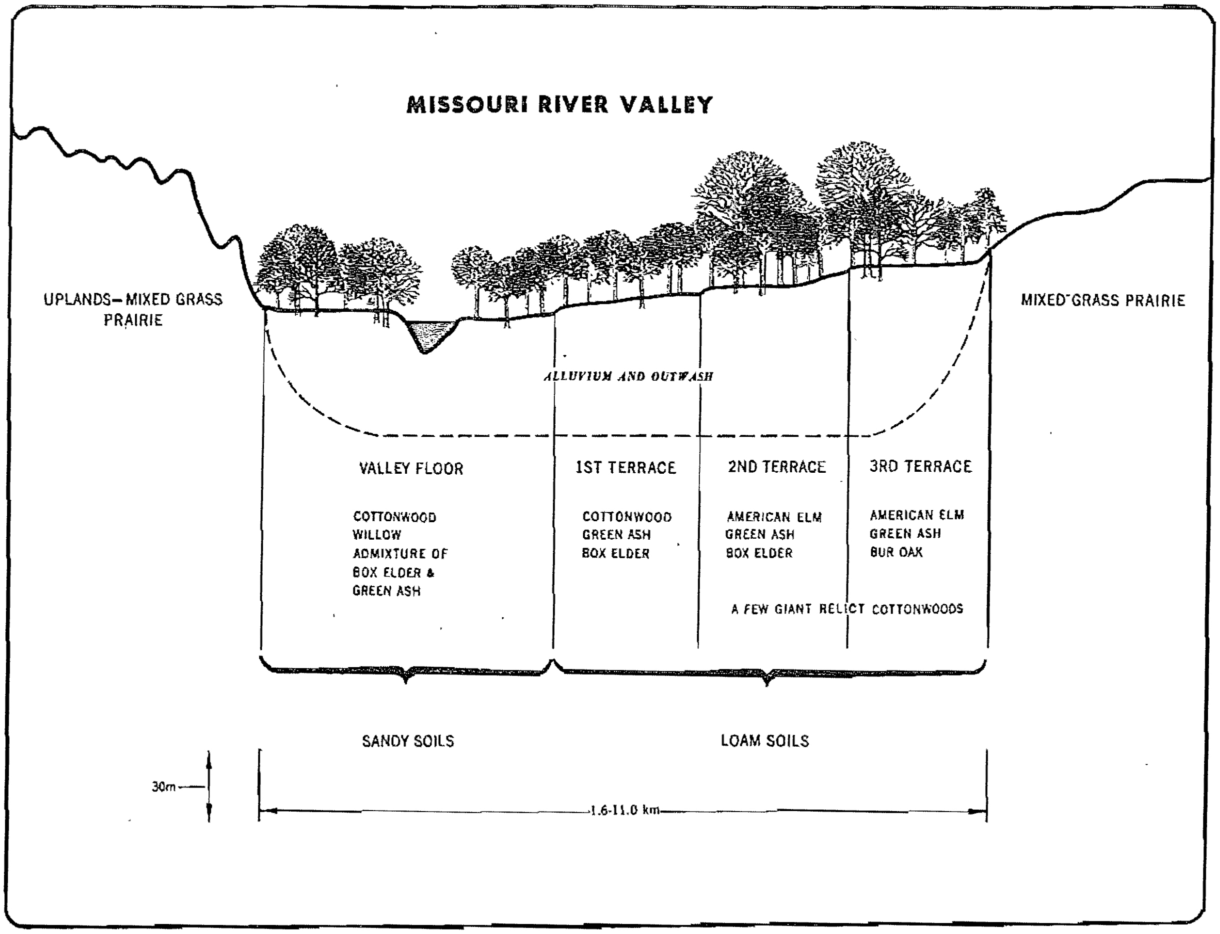


Figure 19. Cross section of the Missouri River in North Dakota showing the distribution of important tree species. From Keammerer et al. (1975).

highest elevations, floodplain forests are dominated by green ash, boxelder, American elm, and bur oak. Canopies are relatively closed and lack the tall shrub and sapling layer characteristic of cottonwood forests.

In the absence of rejuvenation by flooding due to upstream impoundment in May 1954, Johnson et al. (1976) state that cottonwood forests will eventually disappear since seedbed requirements for regeneration are lacking. The change from cottonwood-willow dominance in the lower floodplain with regulation of flooding will lead to higher species diversity but lower landscape diversity.

<u>Mediterranean and Western Arid Forest Regions.</u> Some of the major drainages of the arid West are the San Joaquin, Sacramento, Salt-Gila, and Rio Grande-Pecos Rivers. Along these rivers and their tributary streams, riparian vegetation provides a striking contrast to the drought-stressed semidesert and chaparral of uplands. Species composition of floodplains includes those that are confined to more moist areas as well as those that can survive under drier upland conditions (Campbell and Green 1968). Differentiation between valley floor and upland vegetation increases with increasing drainage area (Zimmermann 1969). Headwaters of intermittent streams have available little more water than well drained upland slopes. There are also dramatic changes in riparian vegetation with increasing elevation.

Along the Rio Grande between El Paso and Albuquerque, a distance of 480 km, five vegetation classes can be described (Campbell and Dick-Peddie 1964). These form a continuum from south to north with gradual and almost imperceptible changes between dominant and subdominant species (Figure 20).

Class 1. In the most xeric class of riparian vegetation, screwbean (<u>Prosopis pubescens</u>) dominates and the cover or density is determined by age of the stand and moisture availability.

Class 2. Where moisture is greater and flooding during the growing season may occur, tamarisk (also called saltcedar - (<u>Tamarix pentandra</u> or <u>T. chinensis</u>) becomes a competitor with screwbean. In areas with a high water table and occasional flooding during the growing season, tamarisk thrives at the exclusion of screwbean.

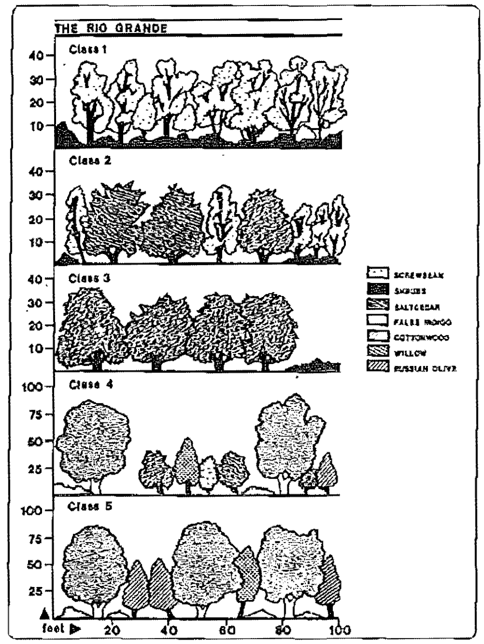


Figure 20. Profiles of five vegetation types along the Rio Grande from El Paso to Albuquerque. The transition from class 1 to class 5 is from xeric to more mesic conditions. Diagrams represent strips about 25 ft wide and 100 ft long. From Campbell and Dick-Peddie (1964).

Class 3. In these dense covers of tamarisk, few shrubs and grasses occur as they do in classes 1 and 2. Class 3 predominates in the southern sector of the river and in disturbed areas to the north.

Class 4. Cottonwood (<u>Populus fremontii</u>) stands attain great height relative to other floodplain species. Russian olive (<u>Elaeagnus angustifolia</u>), tamarisk, and Goodding willow (<u>Salix gooddingii</u>) may become codominants. Mesquite (<u>Prosopis juliflora</u>) occurs occasionally in the northern localities.

Class 5. These are stands with a dense overstory of cottonwood and a separate understory of Russian olive and Goodding willow. Tamarisk is found only in disturbed areas.

The introduction of tamarisk and Russian-olive in the last 50 years has changed succession and ultimate dominants in some communities. Tamarisk is in more than 50% of the floodplain plant communities of the lower Gila River (Haase 1972).

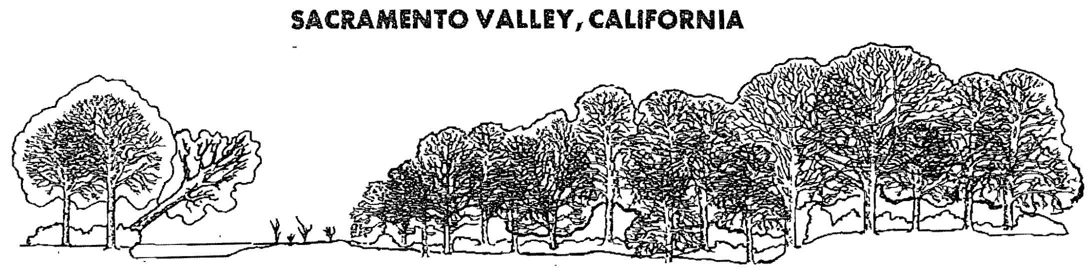
Elsewhere, Freeman and Dick-Peddie (1970) noted a trend toward shrub dominance at lower and upper elevations in southern New Mexico, while trees dominate intermediate elevations. This supports Zimmermann's (1969) observations of increasing upland-riparian differentiation with larger drainage area, though not indefinitely. At the highest elevations studied (1400 m), species such as douglas fir (Pseudotsuga menziesii) and ponderosa pine (Pinus ponderosa) occur, but are restricted from distribution at lower elevations because of high temperatures (Cambell and Green 1968). The transition of vegetation across a floodplain in the Mediterranean Region (Figure 21) illustrates the instability of streamside communities. Forest vegetation develops only in areas that have not been frequently flooded or that have not undergone recent lateral erosion. However, future generations of cottonwoods are dependent on the open, moist sand bars that have resulted from stream instability.

Pacific Northwest and Rocky Mountain Regions. Because of the rugged local relief of much of these regions, stream gradients are frequently steep and channel degradation often predominates. Riparian zones may consist of narrow interrupted bands along small streams or as uninterrupted zones in broad river valleys (Walters et al. 1980b). In mesic sites along streams, gradients of riparian vegetation are probably more a result of stand age, as dictated by time since the last disturbance, than the limiting effects of flooding. Distinct streamside communities are either a result of new land being exposed by destructive floods or the higher local groundwater source along streams (Fonda 1974). A typical gradient beginning at streamside for the western hemlock zone of the Olympic Mountains is: (1) gravel bars dominated by Scouler willow (Salix scouleriana); (2) elevated flats dominated by red alder (Alnus rubra); with time pioneer alder gives way to Sitka spruce (Picea sitchensis), bigleaf maple (Acer macrocarpum) and black cottonwood (Populus trichocarpa); and (3) second terraces occupied typically by Sitka spruce and western hemlock (Tsuga heterophylla). This trend is similar to that of the riparian zone along the McKenzie River in Oregon (Figure 22). Flooding may occur annually on the lowest floodplain.

Some species occur only as riparian species at higher elevations. For example, western hemlock and western redcedar (Thuja plicata) are restricted generally to under 550 m but will reach altitudes of 600 m only along waterways. In the coastal region of northern California, redwood (Sequoia sempervirens) replaces the position of western hemlock, Sitka spruce, and Pacific silver fir (Abies amabilis) found in Oregon and Washington riparian forests. Not only is redwood adapted to survive rapid sedimentation by producing additional roots, but it is also fire tolerant.

In the Rocky Mountains, species on wet sites include cottonwood (P. angustifolia), balsam poplar (P. balsamifera), aspen (P. tremuloides), willows, thinleaf alder (Alnus tenuifolia), and berry bushes (Rubus spp.). At lower elevations Colorado blue spruce (Picea pungens) may replace the wet site species with improved drainage and lack of disturbance. At even lower elevation there is a transition to the drier western arid regions for which the riparian vegetation has already been discussed.

Alaska. At least two climatic zones in Alaska relate to the development of riparian vegetation. On the Arctic slope north of the Brooks Range where permafrost prevails, willow-alder communities along streams are in striking contrast to the shorter tussock-heath tundra and sedge-grass marsh that surrounds them (Bliss and Cantlon 1957). In contrast, riparian vegetation



River Undercutting	Open Plain	Gravel Bar	Riparian Forest	Valley Oak Forest
	low percent cover	*low thicket*	*overstory 25M*	*overstory 15-35M*
	grasses	sandbar willow	cottonwood	valley oak
	herbaceous annuals and perennials	sierra alder	box elder	sycamore
		box elder	sierra alder	ash
		ash	willow	walnut
		cottonwood		poisonoak
				pipe vine
				greenbriar
				blackberry
				wild rose

Figure 21. Profile of vegetation along major rivers in the Sacramento Valley, California. From Conard et al. (1977).

in the maritime climate of southeastern Alaska has similarities in physiognomy to that of the Pacific Northwest Forest Region. Changes in floodplain vegetation from streamside to upland communities in Alaska depend largely on whether the uplands are forested or non-forested. On the Arctic slope, Sage (1974) describes three riparian plant communities. On alluvial deposits that form gravel and silt bars and islands in braided streams, usually no vegetation develops, but in areas not regularly submerged, Equisetum spp. will develop as will occasional dwarf willows. Along small drainage streams, shrub communities of up to 100 cm in height are composed of dwarf birch (Betula nana), stunted Sitka alder (Alnus crispa), and willows (Salix pulchra and S. lanata). A less common community is restricted to streams and drainage canals in the foothills region which is described as tall shrub (90-100 cm), dominated by felt-leaved willow (Salix alaxensis).

In regions where black spruce forests replace the tussock-heath tundra, more elevated portions of the floodplain support stands of balsam poplar which

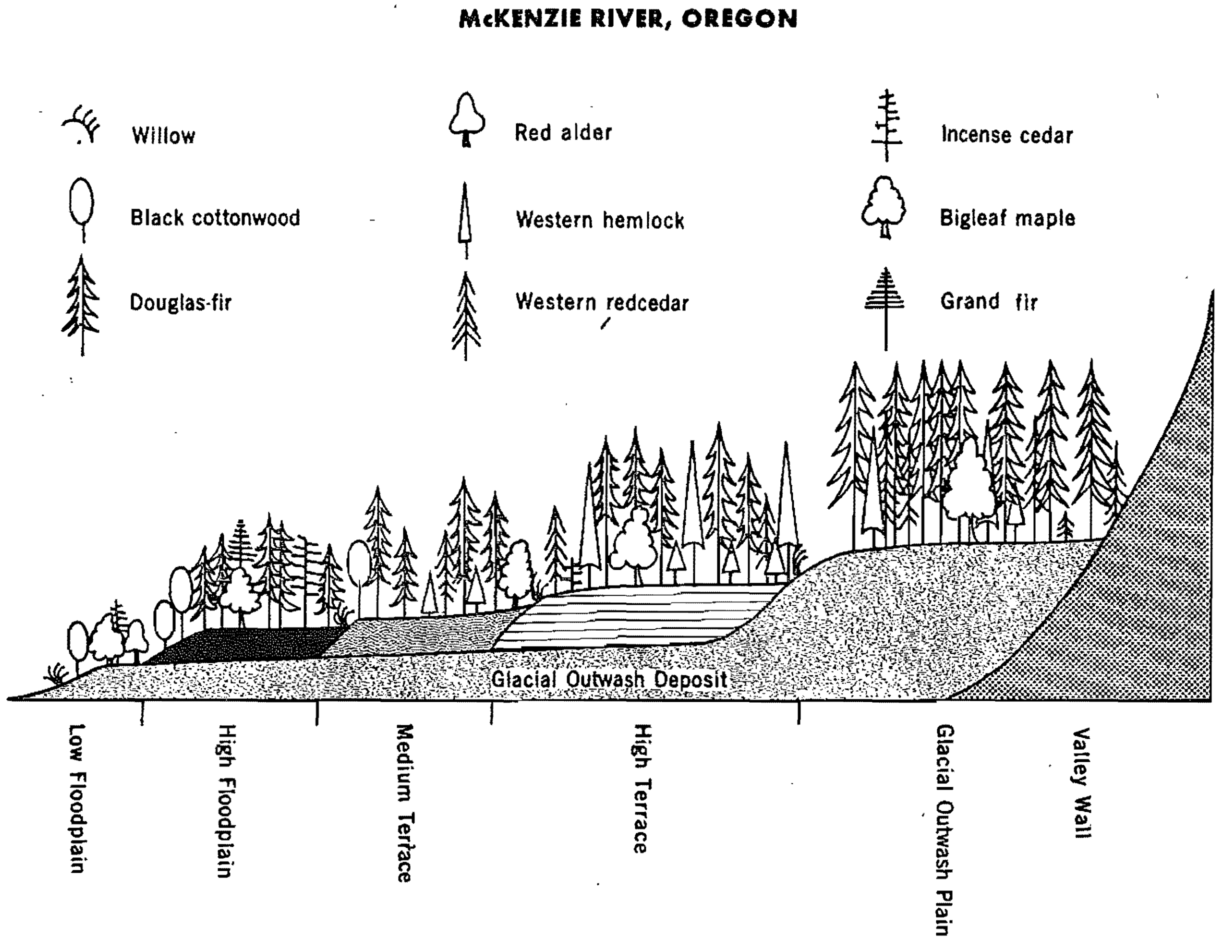


Figure 22. Cross section of floodplain and terrace communities of the McKenzie River, Oregon. From Hawk and Zobel (1974).

are eventually replaced by white spruce (Picea glauca). Figure 23 illustrates an idealized profile for riparian vegetation of the Mackenzie River, N.W.T. (Gill 1972a). In the felt-leaved willow zone, other species of willow (e.g., Salix glauca, S. pulchra, S. arbusculoides) and Sitka alder may assume importance with increasing stand age. White spruce appears to assume dominance only after longer periods without disturbance from flooding. Black spruce will occur at only the uppermost floodplain elevations as described by Drury (1956) for the upper Kuskokwim River region just northwest of the Alaska Range.

In southeastern Alaska where a comparatively mild marine climate prevails, Hurd (1971) described the successional forest stands that followed the recession of Mendenhall Glacier. Species composition of the youngest to oldest communities were quite similar to what might be expected in a floodplain gradient from streamside to upland. The youngest stand was dominated by Sitka alder, with lesser amounts of willow (Salix sitchensis and S. alaxensis). Balsam poplar occasionally contributed to the composition. Later, the poplar and Sitka spruce dominated. The oldest stand was a western hemlock--Sitka spruce mixture which is common throughout the coastal uplands of southeast Alaska. It appears that successional stages after glacial retreat result in similar gradients in species composition

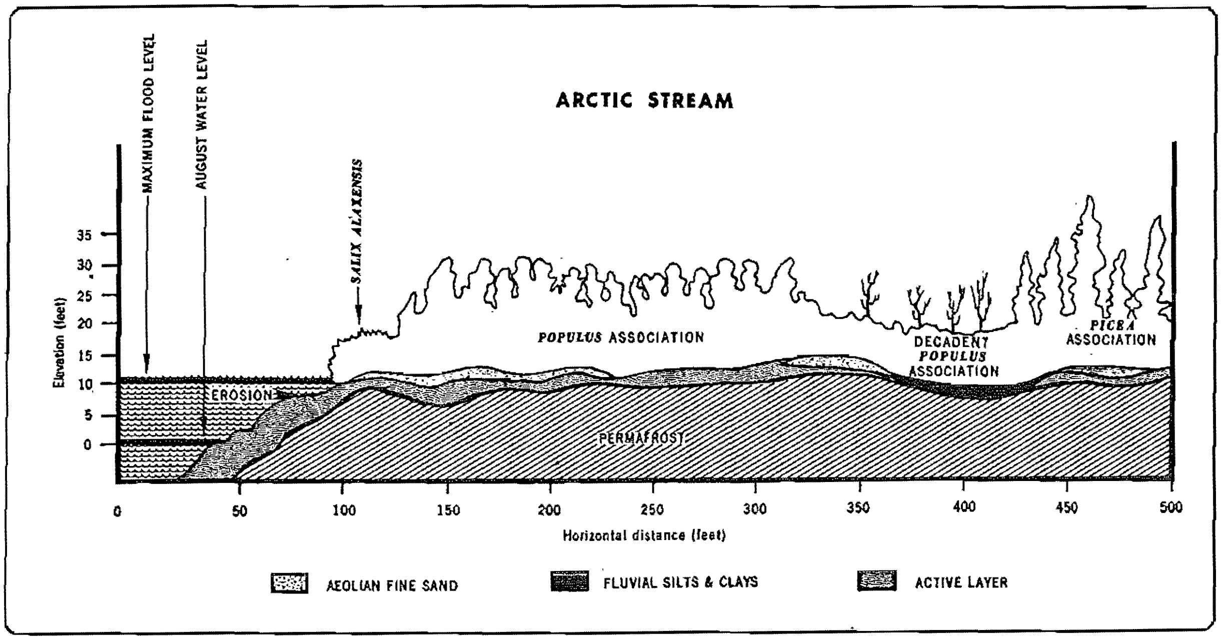


Figure 23. Zonation of plant communities along an arctic stream. From Gill (1972a).

as the time since disturbance along streams.

EFFECTS OF ALTERATION ON THE PROPERTIES OF RIPARIAN ECOSYSTEMS

Although all ecosystems produce and respire organic matter, cycle nutrients, and carry on other processes just described, floodplain ecosystems are unique because these processes are superimposed on the historical and contemporary work performed by flowing water. Few other land forms change as rapidly as floodplains where the channel adjusts its capacity to the natural episodes of large, infrequent floods and variations in sediment load. Diverse topographic features such as oxbow lakes, meander scrolls, and abandoned channels are relicts of this work. Although topographic relief is muted in comparison to many upland landforms, the presence of surface water and natural flood events impose strong control over the microenvironments to which plant and animal communities adapt.

There is sufficient information on these unique floodplain features and their related ecological properties to predict changes that will occur when riparian ecosystems are altered by management of water delivery patterns and by other human intrusions. These alterations can be perceived as stresses which change the pattern of energy flow and the movement of materials to and from riparian ecosystems.

To better understand the way in which these alterations interact with natural ecosystem components, a simplified model of energy flow is used to identify major ecosystem processes of riparian ecosystems (Figure 24). Major sources of energy and materials are shown in the circles on the upper left hand side of the figure -- water, sediments, nutrients, wind, and sun. Other symbols represent storages of material and energy within the ecosystem that are supplied by the outside sources. Exchanges among these storages and interactions with outside sources are indicated by connecting lines of flow. Where two flows interact, whereby one flow augments another in a multiplicative fashion, a large arrow is used to indicate an acceleration of flow. For example the kinetic energy source of water flow interacts with sediments and nutrients to deliver them to riparian

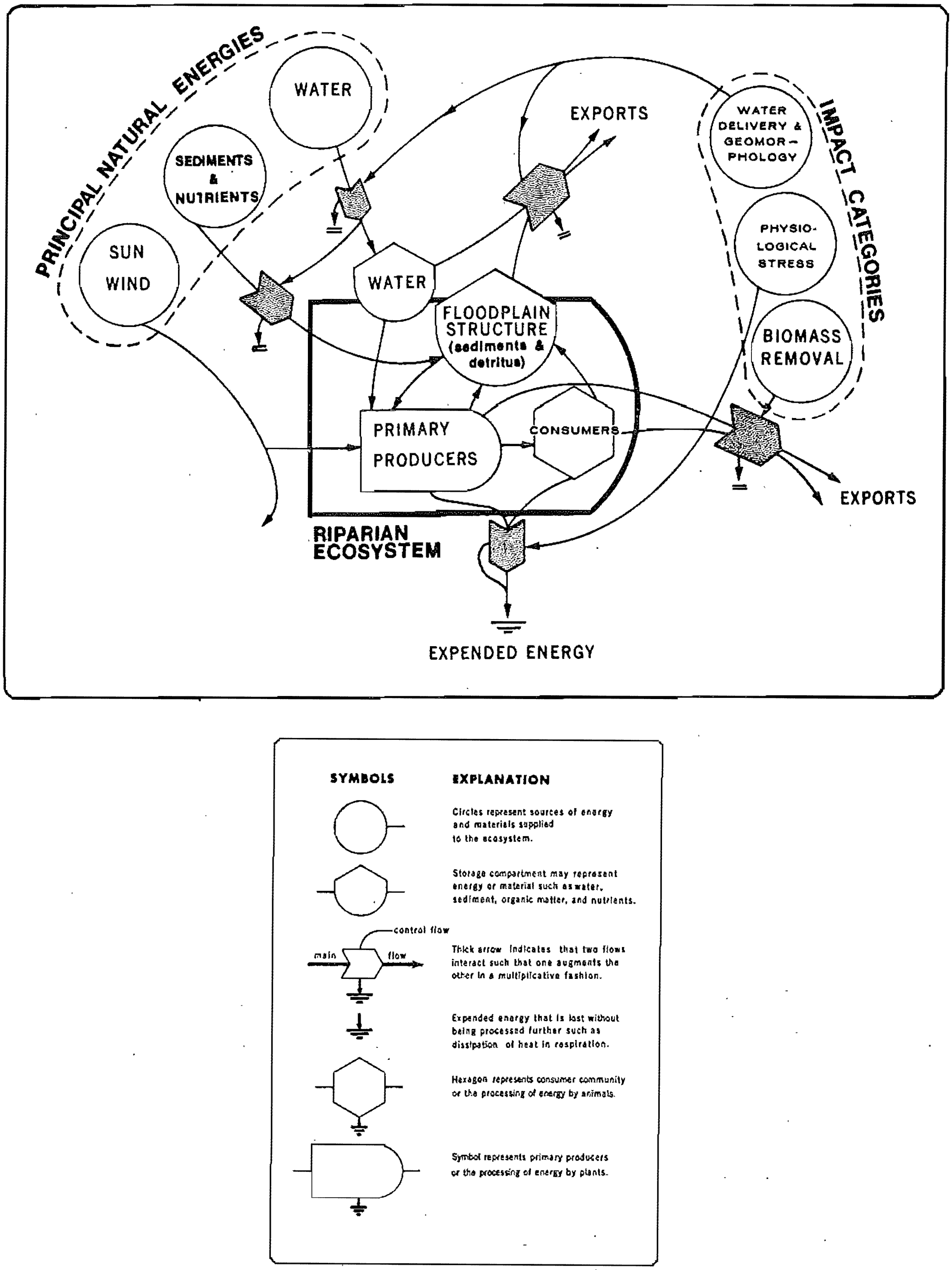


Figure 24. Major flows of energy in a floodplain ecosystem. Adapted from Brown et al. (1978). Refer to text for explanation.

ecosystems. (Many subtle, yet important interactions and feedbacks in this model have been omitted for simplicity.) Smaller, downward pointing arrows are energy sinks that represent necessary losses of thermal energy, such as through respiration, for useful work to occur. When disorder occurs in the flows of energy among ecosystem components, or these components undergo stresses that prevent proper functioning, excessive and wasteful losses of expended energy to these sinks may occur.

To the right side of the figure disruptive energy sources are indicated, again as circles. These represent categories of alteration or impact that drain energy away from the stabilizing flows that maintain ecosystem integrity. The three groups of alterations--water delivery and geomorphology, physiological stress, and biomass removal--all interact at different places in the left to right hand flow of energy.

The closer the alterations interact with the sources of energy, the greater the impact on subsequent flows to the right. Thus, water delivery and geomorphic changes will be expressed at all levels of ecosystem organization. In contrast, biomass removal will have far less effect. If the alterations result in changes of flows close to the primary energy sources, recovery to the original unaltered condition will be slow if recovery is even possible. Energy drains more distant from primary energy sources are less disruptive, and the ecosystem has a high probability of recovering to its original condition.

Most real world alterations of riparian ecosystems and associated stream channels correspond to one or more of the three energy drains in Figure 24. If the alteration can be interpreted as changes in water delivery or geomorphology, severe and long lasting changes can be expected from which there is only a low probability of recovery to the original ecosystem. Physiological stress and biomass removal, depending on the magnitude and frequency at which they are imposed, are more likely to be repaired through natural ecosystem processes (succession) or through mitigation techniques.

Mitigation of damage caused by water delivery and geomorphic changes is extremely costly and time consuming. The costs to restore ecosystems to their original condition after damage may be one indicator of the original value of the work that the ecosystem supplies at no cost to society if it is allowed to function naturally.

Examples of riparian ecosystem alteration and their relationship to the model are outlined in Table 19 and will be explained in the following discussion. Specific effects may differ depending on individual peculiarities of the ecosystem undergoing alteration as well as the nature and severity of the alteration.

Stream Channelization

One of the purposes of stream channelization is to improve the downstream conveyance of water. This is usually achieved by deepening, widening, and straightening the channel. It represents initially a disruptive geomorphic change that would never occur under natural conditions regardless of the time frame. In combination with the effect on water delivery, all essential sources of energy, with the exception of sunlight, are either completely eliminated or greatly diverted. Delivery of water, nutrients, and sediments to the floodplain ecosystem no longer occurs through stream channel-floodplain exchanges. Absence of the natural hydroperiod and water availability imposes severe physiological stress on plant and animal communities.

Increases in channel gradient by reducing sinuosity will result in sharper pulses in flow and concentrate the kinetic energy of flowing water in time and space. This may initiate erosion and cause gullying, depending on soil structure and stream gradient, and result in downstream transport of soil and nutrients. In small, lower order stream channels, removal of streamside vegetation precludes influxes of leaf litter, the principal energy source for instream animal communities. Transformation to a more autotrophically based food web that might be expected upon removal of shade will be of little consequence if benthic structure of the

Table 19. Examples of riparian ecosystem alteration and their relationship to categories of alteration shown in Figure 24. Alterations are listed in approximate direct order to the severity of their impacts on riparian ecosystems, and in inverse order to the time required for recovery following cessation of perturbations.

Intrusions and alterations	Riparian ecosystem component affected: Structure	Riparian ecosystem component affected: Function	Category of alteration
Stream channelization	Channel depth increased	Decreases in floodplain-channel exchanges of water, nutrients and organisms	Water delivery and geomorphology
	Channel gradient increased and sinuosity decreased	Sharper pulses in flow, increased effectiveness of material transport, loss of sinuosity	Water delivery and geomorphology
Containment of streamflow and channel constriction	Restricted floodplain storage	Increased channel scour and greater deposition in narrowed floodplain	Water delivery and geomorphology
Impoundments and diversions:			
Upstream in flooded area	Biomass and water depth	Primary productivity, nutrient cycling, upstream-downstream exchange of organisms	Water delivery and geomorphology
Downstream	Channel depth increased	Sediment supply decreased, scour continues	Water delivery
Introduction of toxic compounds:			
Herbicides	Plant biomass	Primary productivity, trophic structure, & nutrient cycling	Physiological stress
Insecticides	Animal biomass	Trophic structure	Physiological stress
Heavy metals	Plant and animal biomass	Primary productivity, trophic structure, and nutrient cycling	Physiological stress
Timber harvest followed by agriculture	Standing stocks of plant biomass, nutrients, and streambank deterioration	Decreased primary productivity, increased nutrient export, and increased sediment supply and transport	Biomass removal and geomorphology
Grazing by livestock	Plant age structure	Primary productivity and biomass accumulation	Biomass removal
	Streambank deterioration	Increased sediment supply and transport	Geomorphology
Timber harvest followed by silviculture	Standing stocks of plant biomass and nutrients	Temporarily decreased trans- and primary productivity	Biomass removal
Hunting and fishing	Standing stocks of animal biomass	Grazing and predation	Biomass removal

stream channel deteriorates and if greater pulses of water flow and turbidity prevent establishment of primary producers. Mitigation of these damages is clearly not possible because the floodplain has been deprived of the sources of energy that make it unique from upland ecosystems.

Snagging, or the removal of woody obstructions to improve water conveyance, has been suggested in the SCS/FWS Channel Modification Guidelines (44 FR 76299, December 1979) as a preferred alternative to more severe forms of channel modification. However, removal of woody substrates likely causes significant declines in overall animal productivity, animal diversity, and capacity of the stream to assimilate particulate organic matter (Benke et al. 1979). In a southeastern blackwater stream, snags were the most productive habitat available for invertebrates and many fish species are highly dependent on this food source.

Containment of Streamflow and Channel Constriction

Again, geomorphic and water delivery are the principal changes in the natural functioning of the riparian ecosystem when streamflow is contained. Restricted floodplain storage by levee containment increases water velocity in the stream channel and may result in scour and downcutting. However, the deposition of sediments, which originally occurred in the floodplain, will be concentrated between levees and more rapidly obliterate remaining topographic features of the floodplain. Large scale examples of this are occurring along the Mississippi River (Belt 1975) and its distributary, the Atchafalaya River (van Beek 1979). The tendency for these large rivers to build elevated channels and levees accelerates when floodplains are no longer available as areas of sedimentation. Floodplains outside the levees will be deprived of materials in the same way channelization alters exchanges between the stream channel and the floodplain.

Even in the absence of levees, dikes and jetties contribute to the containment of streamflow and channel constriction. Other, more subtle, human activities have resulted in a general tendency toward stabilization of channel meandering, narrowing of channel width, and swifter currents. Not only are fundamental geomorphic and water delivery processes affected, but shifts in food chains can be deduced from activities that convert broad, sometimes braided and often intermittent streams into relatively narrow and swift channels. If no further alteration occurred, there would be an increase in riparian vegetation at the expense of open water; however other human uses such as agriculture may supplant natural floodplain vegetation.

For example, the Platte Rivers in Nebraska and Colorado have undergone a reduction in width by 80-95% during the past 100 years (Williams 1978). The amount of floodplain vegetation has increased considerably at the expense of aquatic surface area and vegetated islands. Nadler (1978) attributed this trend to irrigation practices that produce more stable flow regimes. Irrigation water, which is withdrawn from the river and reduces its sediment load, raises water tables and produces more uniform streamflow. As a result, riparian vegetation becomes more dense and may invade channels during drought years. The result has been the transition from relatively straight, wide, and intermittent streams to narrow and swift channels with more sinuous configuration.

Likewise, in a 830 km reach of the Missouri River, surface area of the river was reduced to half (24,618 ha) of the original area between 1879 and 1972 (Funk and Robinson 1974). Islands, sandbars, snags, and marshes have been virtually eliminated (Figure 25a). Construction of dikes and revetments have been responsible for the surface area lost, but levees, mainstem dams, and tributary reservoirs also contributed to change in channel configuration. Much of the recently accreted floodplain has been put into cultivation of crops. The overall result has been a narrower, swifter and deeper channel accompanied by a reduction in habitat diversity, elimination of some species of fish, and precipitous declines in commerical catches of fish.

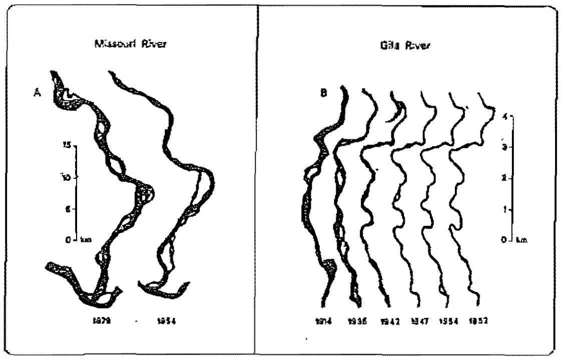


Figure 25. Changes in channel morphology of (A) Missouri River between 1879 and 1954, and (B) Gila River from 1914 to 1962. After Funk and Robinson (1974) and Turner (1974).

In a similar manner, the Gila River, Arizona has undergone a gradual narrowing since 1914 (Fig. 25b). Some of the obvious reasons for changes in configuration include changes in stream discharge and periodicity due to water impoundment. Other less understood changes involve modification of the riparian plant community by increased fire frequency and introduction of exotic species like saltcedar (Turner 1974). Instream primary and secondary productivity is probably reduced in greater proportion than surface area because of swifter abrasive currents and reduced penetration of light under the more turbulent and turbid conditions. Instead of the energy of flowing water being dissipated over broader areas by shifting sand bars and eroding banks, the energy is concentrated in the channel resulting in scour that disorders stream communities.

Impoundments and Diversions

In the inundated reaches upstream from impoundments, changes from lotic to lentic conditions are so extreme that they are too obvious to describe in detail. However, the transformation can be perceived as a change from a structurally complex riparian ecosystem to a relatively simple aquatic system. Although the ecological attributes of the two systems are quite different and difficult to compare objectively, reservoirs usually have construction and maintenance costs (water weed control, dam maintenance, etc.) that must be offset by benefits if society is to gain from the transformation. In comparison, floodplain ecosystems require only protection for them to yield consumables such as flood water storage, water quality maintenance, and products from fish, wildlife, and timber.

Water delivery patterns are altered downstream from impoundments and the sediment supply is held mostly in the reservoir. Other well documented effects in reservoir regulated streams are changes in water chemistry (Hannan 1979, Krenkel et al. 1979,), effects on channel morphology (Simons et al. 1975, Simons 1979), and temperature effects (Fraley 1979). Although direct effects on riparian ecosystems may not be as acute as with other alterations, secondary impacts such as changes in land use to agricultural crop production are freqently the result. Even if the floodplain is not subjected to land use change, the decrease in sediment supply below the impoundment will result in channel scouring and greatly reduce or eliminate sediment delivery to the floodplain. For example, the Shasta Dam on the Sacramento River, California, has reduced the sediment supply below the dam and initiated a phase of degradation (California Department of Water Resources 1979). Erosional-depositional processes currently in effect have lowered the channel by 0.3 m at a distance of 250 km below the dam and are reducing many high terrace riparian lands to lower terrace gravel bars.

Changes in the hydrologic regime also have been dramatic for the Colorado River in the Grand Canyon (Turner and Karpiscak 1980). Before Glen Canyon Dam was built, seasonal variations in discharge were large and daily variations were low (Figure 26). The variations were reversed after the dam began operating in 1963. The result has been establishment of riparian vegetation along the river, especially exotic species such as saltcedar and Russian

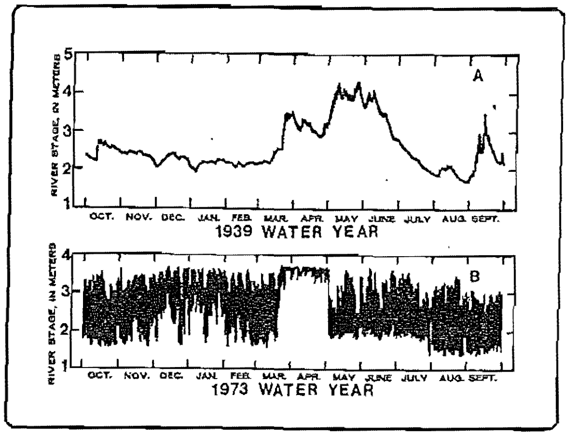


Figure 26. Daily variation in river stage for the Colorado River at Lees Ferry during water year 1939 (A) and water year 1973 (B). From Turner and Karpiscak (1980).

olive. Stream regulation has had an enormous detrimental impact on specialized fishes that have a narrow temperature tolerance (Holden 1979).

For decades water diversion and withdrawal for irrigation in the arid West has resulted in problems with salt balance in the Rio Grande (Wilcox 1955), the lower Colorado, and other major streams (Skogerboe 1973). Under natural conditions, floods occasionally rejuvenate floodplain soils (Babcock and Cushing 1942) by leaching salts and reducing salinity levels. With flood control and the increased evapotranspiration that results from irrigation, there is an increase in soil salinity, particularly during periods of low precipitation. Choices of agricultural crops must necessarily narrow to those tolerant of higher salinity until the problem becomes so acute that agriculture must be abandoned.

Introduction of Toxic Compounds

Herbicides, insecticides, and toxic metals, introduced directly to riparian ecosystems or from the stream by overbank flooding can be regarded as a source of physiological stress to organisms. If accompanying water delivery and geomorphic changes are not imposed, the primary energy sources are maintained and recovery is possible if disruptions are not chronic. In fact, the capacity of water saturated floodplains to immobilize heavy metals in organic rich sediments and to retain pesticides until they are detoxified (Pionke and Chesters 1973) can be a useful and important service for maintaining water quality (Schlesinger 1979). With uplands being managed and utilized at greater intensity, spills, leaks, and appearance of man-made products in runoff are occurring more frequently. The capacity of floodplains for processing these residues and the extent to which they are effective seasonally are not known. However, alterations that accelerate water conveyance will reduce the capacity of floodplains to perform this function.

Grazing by Livestock

Grazing effectively removes plant biomass, alters plant population age structure, and may change the species composition of plant communities. These effects are not restricted to riparian ecosystems; where rangeland has deteriorated under heavy grazing, riparian vegetation also will be under greater grazing pressure. Cattle spend more time in riparian ecosystems than they do in adjacent uplands in the arid west (Martin 1979). Reproduction of tree populations are affected most by heavy browsing on young plants (Dahlem 1979). Without population recruitment of young trees, riparian forests develop unstable age structure and are biased toward large, older trees. Along many streams of arid regions, small stands of relict cottonwood and sycamore are the only forest vegetation remaining. Primary productivity and biomass accumulation of forests necessarily decline under these conditions. Owing to the importance of structural complexity of riparian forests in arid regions (Figure 8), regionwide abundances of vertebrates and invertebrates are dependent on the maintenance of these ecosystems.

Recovery of arid riparian forests from plant biomass removal in many areas is prevented by livestock grazing. Moreover, cottonwood, a major component of these forests, requires special conditions for regeneration. Barren and moist sandbars, which are abundant in shifting, unstable floodplain streams, provide an ideal seedbed for regeneration of cottonwood. Stream channel constriction and flood control considerably reduce conditions for germination. Cottonwood is particularly well adapted to colonization following large floods that may obliterate streamside forests.

Secondary effects of overgrazing may result in increased runoff from uplands and reduction in the stability of stream channels. Restoration of riparian vegetation may require not only reducing or eliminating grazing, but structural measures to control erosion as well. Reduction of livestock grazing, construction of check dams, and other rehabilitation procedures can be successful in retarding soil erosion and rapid channel downcutting. A rangeland restoration study in Colorado demonstrated that streams were transformed from intermittent to perennial flow regimes when restoration procedures resulted in retention of alluvial fill and re-establishment of riparian vegetation (Heede 1977). In this situation an increase in water storage capacity of the newly acquired alluvial fill outweighed water losses that may have resulted from evapotranspiration by riparian vegetation. Other management options are available for improving riparian vegetation and instream conditions (Martin 1979, Platts 1979). Furthermore, fish populations improve rapidly when cattle are excluded (Keller et al. 1979, Van Velson 1979).

Timber Harvest

Forest management practices can range from the selective removal of mature trees to the replacement of natural forest stands by intensive silviculture. Transformation to intensive agriculture may follow timber harvest. The capacity of riparian ecosystems to recover from plant biomass removal will depend partly on the extent to which propagules of native species are available for succession, provided that drainage patterns and hydroperiod are not seriously altered. Clearcutting will cause temporary decreases in evapotranspiration, primary productivity, and probably the capacity to recycle nutrients, whereas selective cutting will have negligible effects on these processes. However, in bottomland hardwood forests of the Southeast, selective cutting has deteriorated the quality of wood products ("highgrading") (Maki et al. 1980) and clearcutting is a preferred practice by foresters (Putnam et al. 1960).

In the wettest portions of southeastern river swamps, regeneration of water tupelo by stump sprouting may result in rapid growth and recovery of plant biomass. This is possible because the root stock is maintained alive and there is less need for the vegetation to initially divert large amounts of photosynthate to belowground parts for growth. In mixed hardwood floodplain forests where regeneration may occur by seeding, the species composition of the forest will depend on a number of factors including available seed source, conditions for germination, competition among young plants, and light availability. Ecological succession in bottomland hardwood forests is poorly understood.

Conversion of forested floodplain ecosystems to agriculture results in a severe and more or less permanent reduction in plant biomass as long as the affected area is farmed. For example, aboveground biomass of a cypress-tupelo stand in Louisiana is 38 kg/m (Conner and Day 1976) whereas a corn crop ranges from near zero in the winter to only 0.4 kg/m at peak biomass (Odum 1971). Secondary practices of flood control and drainage are more seriously damaging to ecosystem function than that of biomass removal. Water delivery changes are involved (Figure 24); consequently there is little opportunity for ecosystem recovery.

Pure stands of saltcedar have replaced many native cottonwood-willow communities in arid regions. Saltcedar is an aggressive competitor and extremely well adapted to floodplain conditions. It has been successful in dominating large sectors of rivers where cottonwood-willow communities existed. Harvest of the original timber, increased frequency of fire, stream channel constriction, and flood control are all alterations induced by humans that have accelerated the dispersal of saltcedar in arid riparian ecosystems (Turner 1974, Everitt 1980).

In efforts to divert water from maintenance of riparian ecosystems to use in agriculture, phreatophyte eradication projects have intentionally removed biomass. There is a great deal of literature that unequivocally advocates the benefits of water yield from streams by means of removing riparian vegetation (Gatewood et al. 1950, Turner and Skibitzke 1952, Bowie et al. 1968, Culler et al. 1970) and most focuses on an intensively studied reach of the Gila River in Arizona. Even if the values of riparian vegetation for organic matter production, shading and temperature amelioration of surface water, and habitat structure (Campbell 1970) are completely disregarded, extrapolating the results to unstudied ecosystems is not warranted because findings vary greatly under the same climatic circumstances (Horton 1972). As early as 1963, it was pointed out that streamflow augmentation could only be expected through manipulation of riparian vegetation under very specific conditions. These are areas in which (1) the water supply is adequate to exceed evapotranspiration losses after treatment, (2) the water table or zone of saturation is within reach of woodland-riparian vegetation, and (3) canyon bottom soils overlaying the water table are of sufficient extent and depth to permit reduction in evapotranspiration if deep rooted vegetation is eliminated (Rowe 1963).

Even if vegetation is removed, it can be considered only a temporary condition (Culler 1970) because revegetation is a predictable consequence of ecological succession. However, continual removal of vegetation does not ensure water salvage. When windspeeds and temperatures are extremely high, evapotranspiration from saltcedar diminishes due to stomatal closure, even though water is freely available (van Hylckama 1980). Estimates of salvageable water based on the assumption that riparian vegetation always uses water at a potential rate may at times be far too large. The long-term effect of these disruptive intrusions may be more severe than just affecting animal biomass and primary and secondary productivity. Riparian ecosystems of the arid West, partly because of widespread deterioration of upland ecosystems, may be extremely important to the survival of many species throughout the region.

Hunting and Fishing

Removal of animal biomass is an alteration that has an excellent opportunity for recovery as long as the habitat structure and life support system of the animals are maintained by the principal flows of energy. Special considerations must be given to providing sufficient contiguous ecosystem area if viable populations of predators are to be maintained. Peculiarities of endangered and threatened species must be given special attention in addition to the maintenance of riparian ecosystem structure and function.

Riparian ecosystems are frequently managed for game species so that additional reproductive success of selected animal populations will support higher rates of harvesting. When management techniques cause water delivery or geomorphic changes, the primary energy sources of the ecosystem are being diverted. Both short- and long-term changes of ecosystem function and structure are predictable under these conditions and they may result in suboptimal levels of natural function and work. Since some wildlife management practices are oriented toward a few game species, little consideration is given to values and functions of the ecosystem that support a high diversity of wildlife species.

CHAPTER FOUR

FISH AND WILDLIFE RESOURCES IN RIPARIAN ECOSYSTEMS

Biologists, naturalists, and other outdoor enthusiasts have long recognized the high value of streams and riparian ecosystems to fish and wildlife. However, quantitative information in support of these observations has surfaced only recently. Research conducted in various areas of the country has confirmed that riparian ecosystems are consistently very important to fish and wildlife on local, regional, and national scales.

Riparian ecosystems differ from upland ecosystems in terms of plant community type, hydrologic features, soil type, and topography. These attributes, along with more subtle environmental parameters, largely determine the potential abundance of animal populations at any particular site. This chapter: (1) discusses the ecological attributes of riparian ecosystems that are most important to fish and wildlife; (2) presents a general characterization of riparian wildlife communities; and (3) examines the overall significance of riparian ecosystems to fish and wildlife.

HABITAT VALUES OF RIPARIAN ECOSYSTEMS FOR FISH AND WILDLIFE

Undisturbed riparian ecosystems normally provide abundant food, cover, and water, and often contain some special ecological features or combination of features that are not found in upland areas (see Chapter 3). Consequently, riparian ecosystems are extremely productive, and have diverse habitat values for fish and wildlife.

The importance of riparian ecosystems can be attributed to specific biological and physical features, including:

1. Predominance of woody plant communities;

2. Presence of surface water and abundant soil moisture;

3. Close proximity of diverse structural features (live and dead vegetation, water bodies, nonvegetated substrates), resulting in extensive edge and structurally heterogeneous wildlife habitats; and

4. Distribution in long corridors that provide protective pathways for migrations and movements between habitats.

Most floodplain ecosystems have some or all of these common attributes that distinguish them from other ecosystems. The relationships of these basic features to fish and wildlife are described below.

Predominance of Woody Plant Communities

Riparian areas often support a variety of plant communities, ranging from mature hardwood forests to alder swamps and cattail marshes. However, woody vegetation predominates in most riparian environments, while herbaceous riparian communities are more limited in extent. Woody riparian communities offer a variety of wildlife habitat values, and are very critical to animal populations where extensive forests are lacking. In grasslands, rangelands, and intensively farmed regions of the U.S.A., woody vegetation along waterways is essential for the survival of many fish and wildlife populations, especially forest-dwelling species (Michny et al. 1975, Boerr and Schmidly 1977, Korte and Fredrickson 1977, Best et al. 1978, Heller 1978, Thomas et al. 1979b). In areas where shrub communities and forests have been cleared for agriculture, woody riparian vegetation may be the only available cover for farmland edge species such as pheasant, dove, and cottontail (Leite 1972).

Woody vegetation is a primary structural feature of riparian wildlife communities. Trees and shrubs are required for roosting or foraging by most riparian bird species, ranging from bald eagle to great blue heron to a variety of small songbirds (Heller 1978, Swift 1980). Mammals such as white-tailed deer, beaver, squirrels, and cottontail are dependent on woody plant materials for shelter and as part of their diet. Woody vegetation on the floodplain increases humidity and provides shade that is attractive to some wildlife species. The attraction of deer, elk, and other wild and domestic ungulates to riparian areas is a result of the thermal cover and microclimate produced by that vegetation (Thomas et al. 1979b).

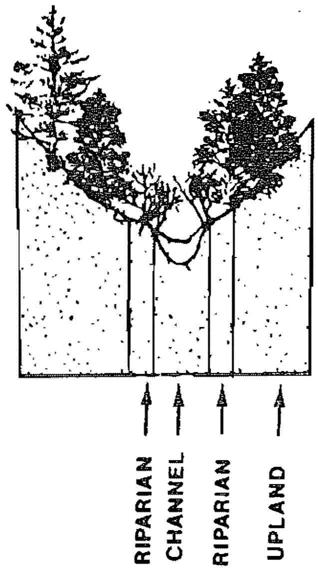
Dead woody vegetation is an important component of wildlife habitat in most forest ecosystems, including riparian woodlands (Noble and Hamilton 1975, Conner 1978, Thomas et al. 1979a, Maser et al. 1980). Standing dead trees or "snags", which are used extensively by wildlife, are especially abundant in beaver ponds (Hair et al. 1978) and where elms occur (Blem and Blem 1975). Snags provide nest sites for cavity-dwelling birds, den trees for small and medium sized mammals, and feeding or perching sites for many species. Fallen logs function as cover for wildlife and as feeding and reproduction sites, but may hinder movement of larger mammals if there is too much downed timber. Dead woody material that is partially submerged in water provides excellent habitat for aquatic, amphibious and certain terrestrial species, although too many logs in a stream channel can act as a barrier to fish passage (Marzolf 1978, Maser et al. 1980).

To varying degrees, aquatic invertebrate and fish communities are influenced by streamside vegetation (Figure 27). Roots of woody vegetation along streams are especially important in bank stabilization and may provide cover for fish and other aquatic animals. Leaf litter from riparian vegetation provides a substantial proportion of food for aquatic invertebrates, particularly in small streams, which in turn constitute a significant proportion of many fish species' diets (Table 20). Terrestrial invertebrates of the riparian zone are often found in streams and become important in the diet of fishes there. The shading of streams by woody riparian vegetation has a dramatic effect on water temperature and the productivity of the aquatic invertebrate community. In all but the coldest regions of the U.S.A., riparian vegetation has a positive influence on salmonid fishes (Meehan 1970, Hunt 1979, Chapman and Knudsen 1980).

<u>Presence of Surface Water and Abundant Soil Moisture</u>

The mere presence of surface water is a requirement of many wildlife species, as an environment for feeding (e.g., waterfowl, fish-eating birds), reproduction (e.g., amphibians), travel (e.g., beaver, muskrats), and escape (e.g., amphibians, muskrat, and beaver). Consequently, many species are rarely found far from water (Figure 28). Water bodies add a dimension of habitat to riparian ecosystems (MacArthur 1964, Hair et al. 1978); increasing the abundance and variety of water bodies contributes to wildlife productivity and diversity (Beidleman 1954, Hardin 1975, Fredrickson 1978).

Seasonal inundation of floodplains increases potential availability of food and breeding habitat for some stream fishes. During annual high water, some species migrate laterally into floodplains to feed among tree roots (e.g., catfish, centrarchids), or to spawn on the inundated forest floor (e.g., blueback herring), returning to the channel when flows slacken and water levels drop (Figure 29) (Wharton and Brinson 1978, Welcomme 1979). At the same time,



RIPARIAN VEGETATION

SITE	COMPONENT	FUNCTION
above ground above channel	canopy & stems	1. Shade-controls temperature & in stream primary production 2. Source of large & fine plant detritus 3. Source of terrestrial insects
in channel	large debris derived from riparian veg	1. Control routing of water and sediment 2. Shape habitat-pools, riffles, cover 3. Substrate for biological activity
streambanks	roots	1. Increase bank stability 2. Create overhanging banks-cover
floodplain	stems & low lying canopy	1. Retard movement of sediment, water and floated organic debris in flood flows

Figure 27. Functions of riparian vegetation as they relate to aquatic ecosystems. From Meehan et al. (1977).

flooding facilitates transport of organic detritus to the channel and downstream (Welcomme 1979).

Even in the absence of surface water, soil moisture (during the growing season at least) may be ultimately responsible for major differences in species composition and productivity between riparian and upland ecosystems. Abundance and diversity of various songbird and small mammal species are related to soil moisture of plant communities (Johnston and Odum 1956, Armstrong 1977, Miller and Getz 1977, Smith 1977, Swift 1980). Several small mammal species are physiologically restricted in distribution to areas with high soil moisture, while others that use underground runways cannot inhabit wet sites (Miller and Getz 1977). Moist soils are required by some bird species for feeding (e.g., woodcock) and for

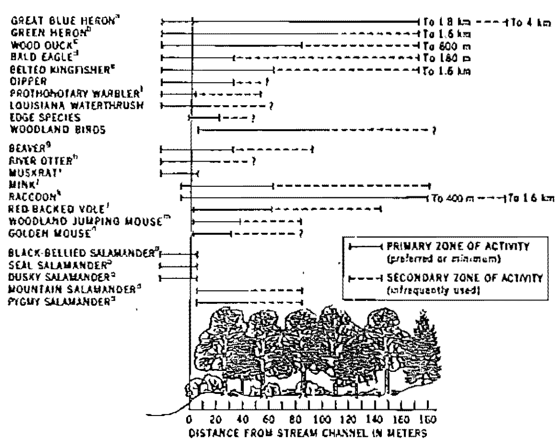


Figure 28. Distribution of riparian wildlife species in relation to streams. Sources: (a) Mathisen and Richards (1978); (b) Mengel (1965); (c) McGilvrey (1968); (d) Steenhof (1978); (e) White (1953), Cornwell (1963); (f) Simpson (1969); (g) Bradt (1947), Hall (1960); (h) Liers (1951); (i) Errington (1937); (j) Schladweiler and Storm (1969); (k) Schwartz and Schwartz (1959); (l) Miller and Getz (1977); (m) Iverson and Turner (1973); (n) Handley (1948); (o) Organ (1961).

Table 20. Importance of aquatic and terrestrial invertebrates in diets of North American stream fishes. Insect orders represent aquatic life stages unless indicated otherwise.

Species	Stream location and size	Stomach contents
Mountain whitefish (*Prosopium williamsoni*)[a]	Sheep R., Alberta; 16.5 m width	Ephemeroptera, Trichoptera, Plecoptera and Diptera made up 89% of the items overall. For larger size classes (300 mm), contents were up to 40% of total.
Coho salmon (*Oncorhynchus kisutch*)[b]	Whitefish R. estuary; L. Michigan tributary	Ephemeroptera most important by weight for yearling fish.
Cutthroat trout (*Salmo clarki*)[c]	Logan R., Utah	Ephemeroptera, Trichoptera, and Diptera were major volume of food items.
Brook trout (*Salvelinus fontinalis*) and cutthroat trout (*Salmo clarki*)[d]	Four streams in northern Idaho; 5-8 m max. width	Ephemeroptera, Coleoptera, Trichoptera, Diptera and Plecoptera comprised 92% of items for 2 species. Terrestrial insects insignificant.
Brook trout (*Salvelinus fontinalis*)[e]	Unnamed stream, Vermont; 5 m wide	Diptera, Trichoptera, Ephemeroptea, and Plecoptera major items except during June and Aug.-Nov. when terrestrial beetles, grasshoppers, and ants dominated.
Black sculpin (*Cottus baileyi*)[f]	Upper S. Fork of Holston R., Virginia; 9.4 m wide at low discharge.	Ephemeroptera, Diptera, Trichoptera, Coleoptera, and Plecoptera comprised 99% of total food items.
Northern mottled sculpin (*Cottus b. bairdi*) and barred darter (*Etheostoma f. flabellare*)[g]	Rock Cr., Oregon; 6 m wide	Ephemeroptera and Diptera were major food items of both species.

[a]Thompson & Davies (1976); [b]Peck (1974); [c]Fleener (1951); [d]Griffith (1974); [e]Lord (1933); [f]Novak & Estes (1974); [g]Pasch & Lyford (1972).

preferred nesting habitats of others (e.g., prothonotary warbler). Generally, moister sites are more productive of wildlife, because foods (vegetation, seeds, insects) are presumably more abundant there, and vegetation structure is more favorable to a greater number of species (Odum 1950, Gaines 1974, Curtis and Ripley 1975, Hardin 1975, Dickson 1978, Swift 1980).

Diversity and Interspersion of Habitat Features

Within riparian ecosystems, there are a great variety of habitat features that are used by a relatively large number of fish and wildlife species. Riparian areas are able to support dense growths of herbaceous, shrub and forest vegetation, the arrangement of which determines suitability of a site for many species. In addition, riparian environments often provide various aquatic habitats and nonvegetated substrates that are important to fish and wildlife.

Riparian ecosystems tend to be very complex wildlife habitats, due to the interspersion of the many physical and biological features present. With

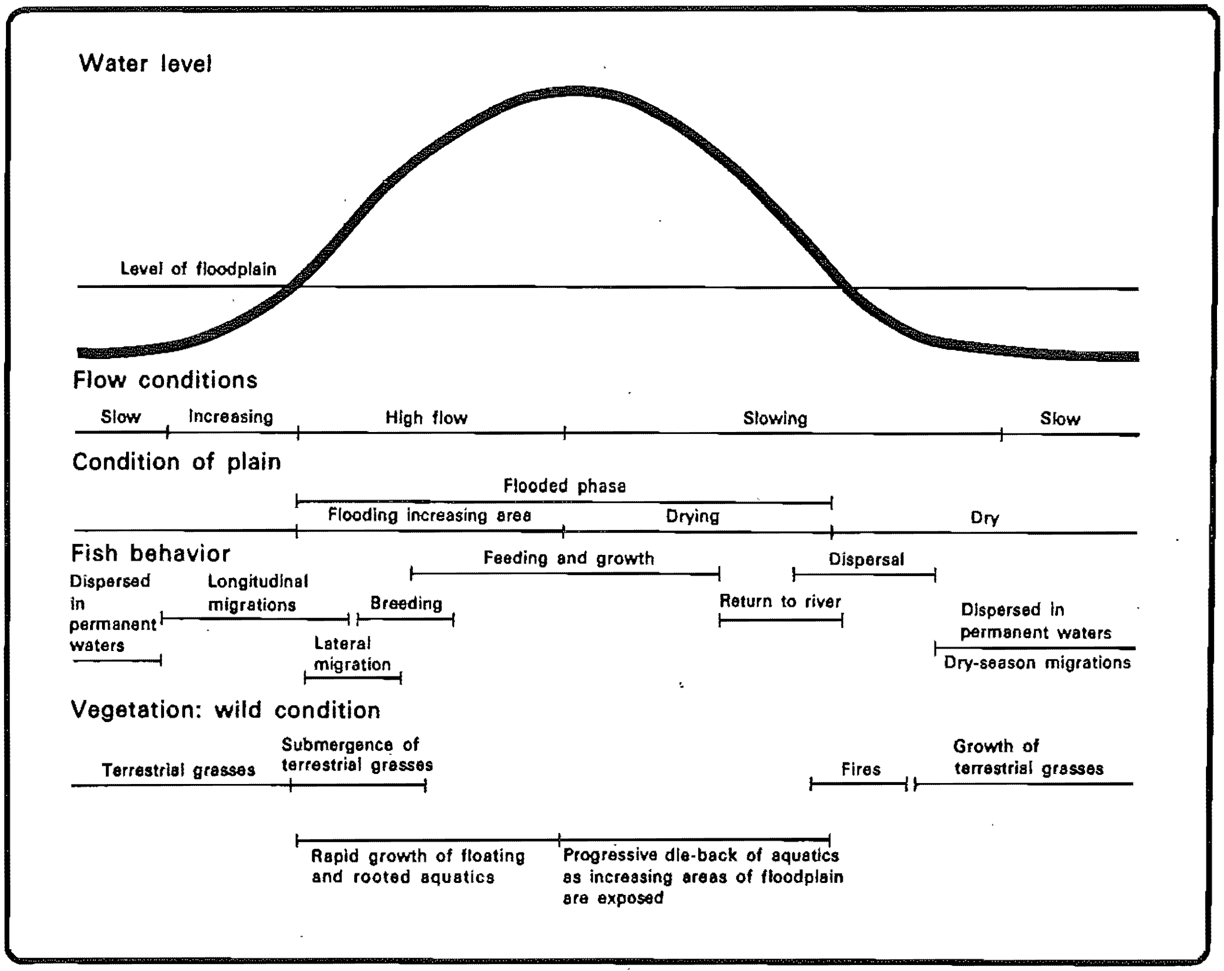


Figure 29. Synchrony of events related to flooding in a floodplain-river system in the tropics. From Welcomme (1979).

maturity, and as a result of natural flooding, riparian woodlands often become interspersed with natural drainages, marshes, ponds, and brushland. This is especially evident at beaver ponds which are used by a great diversity of wildlife (Kirby 1975, Hair et al. 1978). Inevitably, wildlife species that require a combination of riparian habitat features are more sensitive to alterations than those requiring only one component.

Associated with most riparian ecosystems is substantial development of edge at the interface between stream channel and riparian vegetation, and in the transition from floodplain to upland plant communities (Figure 30). The interface between stream and woody plant communities may be one of the greatest values to wildlife of riparian ecosystems; many species occur almost entirely in this zone (Figure 28). Riparian-upland edges are very important for many upland and edge species of wildlife, at least where woody riparian communities adjoin relatively open rangeland, grassland, or farmland (Thomas et al. 1979b).

Because edges and their ecotones are usually richer in wildlife than adjoining areas (Figure 31), they are an important component of riparian wildlife habitats (Hardin 1975, Thomas et al. 1979c). However, excessive manipulation of floodplain forests to maximize edge

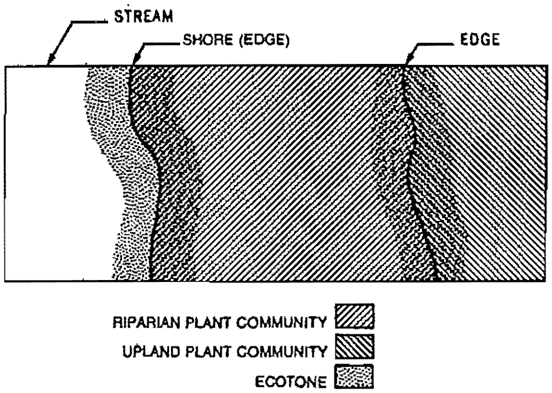


Figure 30. Edges and ecotones in riparian ecosystems. Adapted from Thomas et al. (1979c).

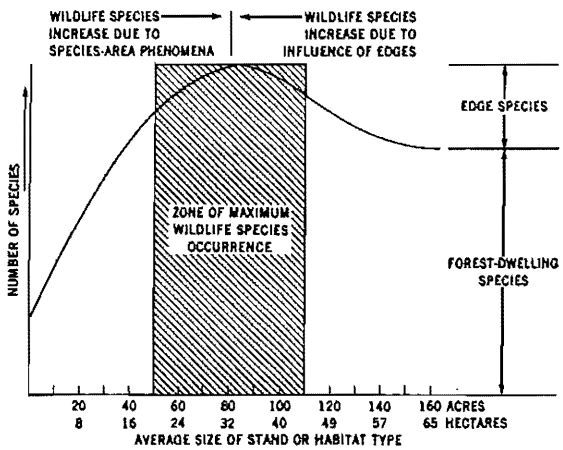


Figure 31. Relationship of wildlife diversity to size of a plant community type. From Thomas et al. (1979c).

development would adversely affect the more uncommon species that require continuous riparian forest cover.

Corridors for Dispersal and Migration

The linear nature of riparian ecosystems provides distinct corridors that are important as migration and dispersal routes and as forested connectors between habitats for wildlife such as birds, bats, deer, elk, and small mammals (Figure 32) (Blair 1939, Rappole and Warner 1976, Stevens et al. 1977, Wauer 1977, Willson and Carothers 1979). Woody vegetation must be present for terrestrial species to find needed cover while travelling across otherwise open areas. Animals involved in population dispersal may utilize food and water from riparian areas during their movements. The value of waterway corridors for migratory movements may be more accentuated in arid regions than in humid, more heavily vegetated areas (Wauer 1977).

Maintenance of fish populations often depends on localized dispersal movements over short distances and spawning migrations covering hundreds of kilometers. Fish migrate to satisfy nutritional and reproductive requirements that may not be met in a particular

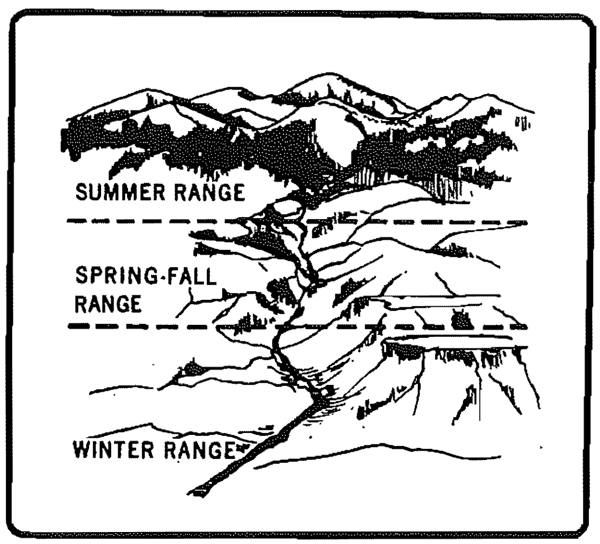


Figure 32. Riparian zones are frequently used as migration routes by wildlife, such as mule deer (Odocoileus hemionus) which travel along streams between high elevation summer range and low elevation winter range. From Thomas et al. (1979b).

stream segment, and to maintain populations throughout a stream (Hall 1972, Durbin et al. 1979). Reproductive success of many species requires unobstructed access to migration (Davis and Cheek 1966), which depends on structural integrity of the stream and its associated riparian communities.

RESPONSES OF FISH AND WILDLIFE TO HABITAT VARIABLES

Despite various environmental attributes common to riparian ecosystems, there are many ecological variables that further determine their relative values as fish and wildlife habitats (Short and Shamberger 1979). Those variables often reflect suitability of a site for wildlife species, and can be used to evaluate and compare riparian habitats with one another or with nonriparian ecosystems. Most important among riparian fish and wildlife habitat variables are vegetation type (composition and structure), size and shape, hydrologic patterns, adjacent land use, and elevation.

Vegetation Type

Generally, riparian wildlife communities are influenced more by structural form of vegetation than by species composition of the plant community. The type, size, and arrangement of canopy, shrub, and herbaceous vegetation largely determine the suitability of a site for wildlife. Most songbird species have specific requirements of vegetation (e.g., dense understory, closed canopy), as do deer (e.g., twigs within browsing height), black bear (Landers et al. 1979), bald eagle (Steenhof 1978), a few small mammals (Miller and Getz 1977), and many other species. Other species are able to inhabit several community types or successional stages. The variety of wildlife habitats, especially for birds, is greatest in structurally diverse woodlands where all three vegetation layers are present and where those layers are distributed in patches throughout an area (Beidleman 1954, MacArthur and MacArthur 1961, Austin 1970, Glasgow and Noble 1971, Carothers et al. 1974, Carothers and Johnson 1975, Whitmore 1975, Anderson and Ohmart 1977, Gaines 1977, Stevens et al. 1977, Dickson 1978). However, homogeneous riparian woodlands, such as even-aged plantations, may support a few species not commonly found in heterogeneous stands (Dickson 1978).

Riparian wildlife communities are influenced to some degree by plant species composition of an area, especially where there are clear differences in the food values of the various vegetation types. There is probably much less variation in the riparian community types of a region than there is in the structural forms that each type may take. However, presence of mast (fruits and nuts) producing trees in a bottomland community is especially favorable to use by wood duck, wild turkey, squirrels, and other wildlife. Furthermore, various plant species may host very different invertebrate populations among the foliage and branches; this directly affects their value to many songbird species.

Preferences for certain riparian vegetation types is most prevalent among passerine (perching) birds. In Louisiana and eastern Texas, oak-gum swamps had many yellow-billed cuckoos, tufted titmice, Carolina wrens, and cardinals, while none of these were among the most numerous birds in a tupelo swamp (Dickson 1978). Cottonwood and willow communities are the most favorable riparian bird habitats in the West (B. W. Anderson et al. 1977). Saltcedar, an exotic plant species, has a low value to most riparian bird species (Beidleman 1978, Cohan et al. 1978, Conine et al. 1978), but it is valuable as nesting habitat for white-winged dove (Shaw and Jett 1959), and a few of the more rare species, such as Bell's vireo, blue grosbeak, black-tailed gnatcatcher, and Gila woodpecker (B. W. Anderson et al. 1977, Cohan et al. 1978). Addition of native trees to saltcedar stands would greatly enhance the value of those sites, as would maintenance of mature communities rather than early seral stages (B. W. Anderson et al. 1977). It is generally believed that hardwoods support greater breeding bird densities and number of bird species than softwoods (Thomas et al. 1975).

Although little information is available on herbaceous and non-vegetated areas of riparian ecosystems it seems reasonable that their values to fish and wildlife differ little from structurally similar areas in non-riparian zones. Wildlife communities in riparian marshes are likely dominated by waterfowl (especially dabbling ducks and geese), shorebirds (e.g., avocet, rails), a few songbirds (e.g., blackbirds, wrens, and sparrows), furbearing mammals, and various amphibians (Hardin 1975, Flake and Vohs 1979). Value of marshes to wildlife is largely influenced by water regimes, interspersion of cover and open water, and the composition and structure of the emergent marsh plants (Weller 1978).

As a result of continual erosion and deposition, streams commonly produce at least two kinds of nonvegetated substrates: barren streambanks; and stream channel alluvial areas (e.g., outwashes and sandbars). Prior to invasion by herbaceous or woody plants, steeply sloped streambanks provide required nesting sites for the bird species such as belted kingfisher, bank swallow, and rough-winged swallow (Cornwell 1963, Gaines 1974). Mid-channel sandbars along the Missouri River provide resting grounds for migrating waterfowl, basking areas for softshell turtle, and nesting sites for the least tern (U.S. Fish and Wildlife Service 1980). Sandy shoals are important to turtles for nesting (Dodd 1978), and for killdeer, spotted sandpiper and upland sandpiper which feed near the sand-water interface. The sandbar-channel combination serves as a feeding ground and nursery area for many species of fish. Bald eagle and osprey feed on fish concentrated in those shallow water areas (U.S. Fish and Wildlife Service 1980a).

Size and Shape of Riparian Area

The size (width and/or area) of a plant community has a direct relation to its ecological values. There is no clear concensus on the minimum size of a riparian stand that is needed to accomodate wildlife populations, protect water quality, or provide recreation. Various minimum dimensions have been recommended for these purposes (Table 21), but additional research is needed to provide a more comprehensive data base.

Even very narrow strips of riparian vegetation are important to instream aquatic communities and for certain kinds of wildlife. Species commonly occurring along streams or shorelines, such as mink, belted kingfisher, and riparian edge species, are often able to establish territories in narrow riparian woodlands (Curtis and Ripley 1975). However, narrow riparian woodlands are unsuitable for species requiring large areas of forest or considerable isolation from man, such as black bear (Landers et al. 1979), osprey (Swenson 1979), great blue heron (Scott 1980), the presumed extinct ivory-billed woodpecker (Korte and Fredrickson 1977), and many forest dwelling songbirds. Reduction in size of southwestern riparian woodlands is at least partly responsible for the regional decline of several species; Cooper's hawk, red-shouldered hawk, and yellow-billed cuckoo were found only where patches were more than 100 m wide (Gaines 1974).

The area of riparian vegetation most heavily used by terrestrial wildlife is that within 200 m of a stream (or open water), although some species travel as much as 4 km from nesting to feeding area (Figure 28). A 200 m wide vegetative strip is apparently able to accomodate breeding territories of most songbirds (Stauffer and Best 1980). Many vertebrates, especially riparian mammals, reptiles, and amphibians, concentrate their activities well within 60 m of water (Hairston 1949, Organ 1961, Tilley 1973, Krzysik 1979).

Along with the lateral dimension of riparian wildlife habitats, the overall size is also important to many species. Size of animal territories varies widely among species, ranging from less than a hectare for small terrestrial animals to several square kilometers for birds of prey and large mammals. Reducing the size of a community type progressively eliminates species requiring large areas of the particular type and favors expansion of species associated with the new land use and the edges created. For example, prothonotary warblers are generally absent from waterways where the border of deciduous trees is less than 30 m (100') deep (Simpson 1969). In

Table 21. Width of riparian buffer strips recommended[a] to protect water quality and aquatic life in streams.

Function of buffer strip	Recommended width	Recommended by
Protect water quality from logging	8 m (25') plus .6 m (2') per 1% of slope	Trimble 1959
Protect water quality from logging in municipal watersheds	16 m (50') plus 1.2 m (4') per 1% of slope	Trimble and Sartz 1957
Protect aquatic life from logging	at least 30 m	Erman et al. 1977
Protect water quality and fish	25 m (75') plus any additional width that supports riparian vegetation.	USDI Bureau of Land Management 1979
Protect streams from adverse land management practices	30 m (100')	U.S. Dept. Agriculture 1980
Maintain wild or scenic values of river corridors	400 m (.25 mile)	Wild and Scenic Rivers Act (P.L. 90-542)
Protect aquatic environment	at least 15 m	Canada Fisheries and Marine Service 1978

[a]These recommendations do not represent conclusions or recommendations of the FWS or the authors of this report.

contrast, red-shouldered hawks are found primarily in forested stream valleys with adjacent clearings (Stewart 1949, Craighead and Craighead 1956), and are absent from the center of extensive forest stands (Brown and Amadon 1968). While edge species tend to be very ubiquitous, species that require large riparian stands are generally less common, and face declining population levels as riparian alterations continue. Where riparian "islands" are created, the size needed to support potential songbird diversity near maximum values is at least 5-6 ha, but is probably as large as 10 ha for maintaining a diversity of all wildlife forms (Gaines 1974, Galli et al. 1976, Emmerich 1978, McElveen 1978, Willson and Carothers 1979). Larger areas will support additional species because interspecific competition and territoriality in a small stand limit the number of large species that can coexist.

Width of a riparian woodland also determines the degree to which impacts of adjacent land use on water quality are buffered before reaching the stream. Optimum width for a riparian buffer zone varies with stream width, topography, soil type, type of impact, sensitivity of the resource, and water quality standards. Buffer strips reduce erosion (and pollution), preserve the stream channel's stability, retard runoff, trap sediments and nutrients, maintain suitable water temperatures for aquatic life, and provide vegetation and invertebrates as food for birds, and other wildlife (Curtis and Ripley 1975).

Stream Type and Hydrologic Pattern

Riparian communities are found along many kinds of streams, varying in size, shape, velocity, flow patterns, and water quality. The importance of stream type to fish and wildlife is largely a function of the relation between these variables and habitat components already discussed.

As one moves downstream from tributary to river, flow volume increases, overbank flooding is more widespread, and riparian communities are broader and more distinct than in headwater areas (especially in mountainous regions). At the same time, the influence of riparian vegetation on the adjacent stream decreases downstream. Middle-order perennial streams and their riparian communities may be the most heavily used wildlife areas in a watershed because they provide very sizable and diverse habitats (both instream and riparian).

Riparian wildlife are also sensitive to differences in stream type that are not always reflected by vegetation. Ephemeral streams often support valuable woody riparian growth, but lack fish, the aquatic food base upon which certain riparian species depend. Similarly, clear slow-moving water is important to beaver and muskrat (Flood et al. 1977), belted kingfisher (Cornwell 1963), and water snakes (Lagler and Salyer 1947) because it enhances the production of aquatic food organisms and the ability of these species to find food.

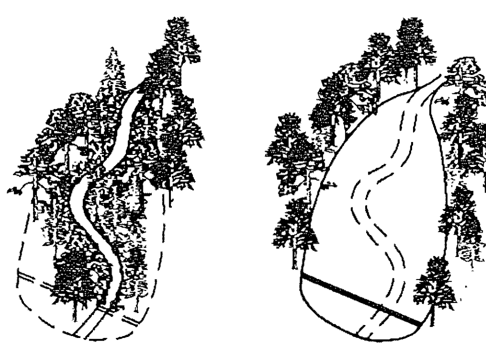
Periodic flooding is one of the most significant phenomena affecting the use of riparian ecosystems by fish and wildlife. Although floodplains are very unpredictable environments, annual flooding has a generally favorable effect on productivity of fish (Wharton and Brinson 1978, Welcomme 1979) and wildlife (Wharton 1970, Batzli 1977, Gaines 1977, Fredrickson 1979). The overflow of streams onto floodplains directly influences both animal populations and their habitats. Overbank flooding is critical for the exchange of energy, nutrients, and animal populations between aquatic and terrestrial portions of riparian ecosystems.

The composition and structure of riparian plant communities is dependent upon the prevailing hydrologic regime. Many bottomland tree species must be flooded periodically to produce seeds, and for subsequent development into seedlings and mature trees (Teskey and Hinckley 1977a, 1977b, 1978a, 1978b, 1978c; Walters et al. 1980a, 1980b). However, no woody plants are able to reproduce on sites that are flooded throughout the year. Development of understory vegetation in wetland forests is reduced by widely fluctuating water levels during the growing season (Flinchum 1977, Brown et al. 1978, Swift 1980). Clearly, the long-term maintenance of existing riparian wildlife habitats depends on the continuation of natural flooding patterns.

Seasonal and short-term overbank flooding has profound effects on terrestrial wildlife. Distributions of ground dwelling vertebrates are often more closely related to hydrologic patterns than to vegetation features. Riparian mammal populations may be generally impoverished (Barclay 1980) or relatively dense (Arnold 1940), depending in part on recent hydrologic events (Blair 1939, Armstrong 1977, Batzli 1977, Miller and Getz 1977). Short-term floods (several days) often have little detrimental effect on wildlife; deer mice, tree squirrels, and box turtles apparently take refuge in unflooded sites or trees (Stickel 1948, Hoslett 1961, Ruffer 1961). In contrast, severe flooding (several weeks) temporarily eliminates and may limit resident small mammal populations in a floodplain. Recolonization by individuals from nearby unflooded areas occurs slowly (Blair 1939, Wetzel 1958, McCarley 1959, Turner 1966, Iverson et al. 1967).

Depth and duration of flooding in a riparian ecosystem also determines the availability of foods for waterfowl and wading birds. In the southeastern U.S.A., inundation of bottomland hardwoods during winter creates excellent feeding areas for hundreds of thousands of ducks, especially wood ducks and mallards, which feed on the fallen mast crop (acorns). Wood duck, great blue heron, and green heron feed primarily in water less than 0.5 m deep (Martin et al. 1951, Palmer 1962, Webster and McGilvrey 1966); a gradual rise or fall of water levels in the riparian zone allows maximum use of the area by these and other species.

Permanent impoundment of streams has very dramatic consequences on fish and wildlife habitats in the inundated floodplain (Figure 33). A rapid increase in fish populations commonly follows reservoir construction as food resources on the freshly inundated floodplain are exploited. A subsequent decline results as those resources are depleted, without rejuvenation by alternating wet and dry phases, as occurred previously. Long-term impoundment of streams by man or beaver eliminates habitat of ground nesting, canopy feeding, and ground foraging birds, including prothonotary warbler (Simpson 1969), Kentucky warbler, and white-throated sparrow (Dickson 1978), and many riparian reptiles and amphibians (Dodd 1978). However, partially impounded riparian communities can enhance areas for bald eagle, waterfowl, cavity-nesters, and flycatching birds (Hair et al. 1978), and provide protection from predators for herons, egrets, and red-winged blackbirds (Dickson 1978).



BEFORE IMPOUNDMENT	AFTER IMPOUNDMENT
Habitat for stream-dwelling fish	Habitat for lake-dwelling fish
Predominantly floodplain/terrestrial wildlife habitat	Predominantly aquatic fish habitat
Streambank habitat for many specialized wildlife species	Streambank habitat replaced by extensive, often unstable shoreline; altered species assemblage
Natural hydrologic regime provides exchange pathways for nutrients, detritus and organisms between channel and floodplain	Permanent inundation eliminates floodplain vegetation and vital pathways of exchange
Downstream transport of detritus and sediments	Retention of detritus and sediments behind dam
Corridor for fish and wildlife movements	Corridor altered and interrupted

Figure 33. Fish and wildlife values at small stream impoundments.

Adjacent Land Use

Wildlife use of riparian ecosystems can be influenced by adjacent land use. Riparian ecosystems surrounded by low quality wildlife habitats often support higher density and diversity of birds during migration than would otherwise be expected, because populations do not spread out over the entire area to feed (Stevens et al. 1977). Nesting birds can inhabit riparian communities in higher densities where adjacent agricultural lands produce an abundant food supply but lack nesting sites (Carothers et al. 1974). Similarly, carrying capacity of deer in bottomland hardwood forests of the lower Mississippi Valley may double where agricultural crops are readily available (Glasgow and Noble 1971). Riparian ecosystems surrounded by forest land do not usually exhibit such obvious influences of adjacent wildlife habitat, because resources are more similar and competing species are normally present there.

Many bird species find shelter in riparian vegetation, but feed extensively in surrounding agricultural lands (Glasgow and Noble 1971, Carothers et al. 1974, Whitmore 1975, Conine et al. 1978). Of 63 riparian species along the lower Colorado River, 41 travelled varying distances into adjacent agricultural lands (Figure 34). Within those farmed lands, bird densities increased towards the floodplain, and were positively correlated with presence of canals, weedy margins, and alfalfa. However, total encroachment of agriculture into the riparian zone would completely eliminate many species (Conine et al. 1978).

Effects of adjacent land uses are limited primarily to the vegetative edges of riparian ecosystems, and are

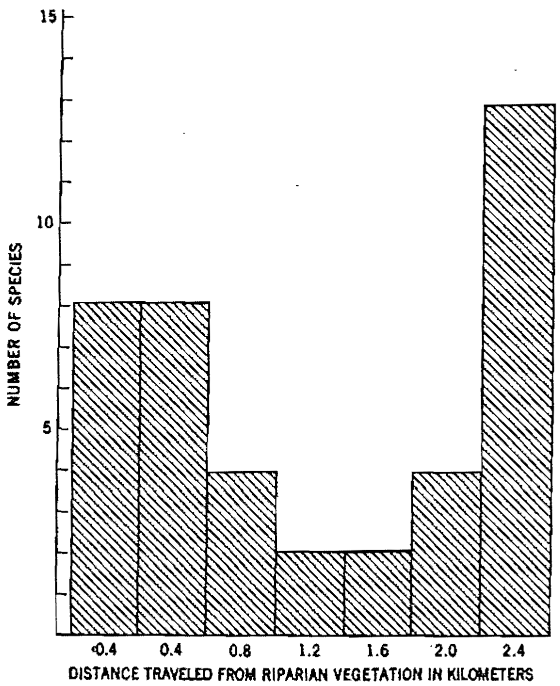


Figure 34. Distance travelled by riparian bird species into agricultural areas. From Conine et al. (1978).

most important to wildlife in narrow or small patches of riparian vegetation. In narrow corridors of streambank vegetation, most wildlife species must extend their territories into adjacent lands, and are directly affected by the food resources and wildlife populations that occur there.

Elevation

The composition of riparian wildlife communities is affected by elevation, especially in the West, where dramatic changes in climate, topography, and vegetation are associated with altitude (Noon and Able 1978). In addition, riparian dependence of many species is reduced at higher elevations, because moisture is readily available in nonriparian communities as well (Hairston 1949, Johnson et al. 1977).

Abundance and diversity of birds in lowland riparian ecosystems is significantly greater than in high elevation riparian areas (Finzel 1964, Wooding 1973, Stevens et al. 1977, Burkhard 1978). A similar phenomena may exist among other vertebrates (Burkhard 1978), but this has not been confirmed.

Elevation effects on riparian wildlife communities are often associated with riparian habitat variables that have already been discussed (e.g., size, productivity, and diversity of vegetation, hydrologic patterns, value as travel corridors). For example, perennial streams in relatively flat areas usually support large, distinct riparian corridors, while riparian vegetation along mountain streams may be lacking or barely noticeable. Riparian woodlands that extend between high mountain and lowland areas may be important for seasonal movements by elk and deer (Thomas et al. 1979b), but may not be used by migrating birds simply because birds fly between mountain ranges rather than over them (Stevens et al. 1977).

CHARACTERISTIC RIPARIAN WILDLIFE COMMUNITIES

Surveys of animal communities in riparian ecosystems reveal that these areas are inhabited by a great variety of birds, mammals, amphibians, and reptiles. Certain groups of wildlife tend to predominate in undisturbed riparian ecosystems across the U.S.A. However, the presence or absence of particular species is often determined by specific habitat variables, geographic location, and site specific alterations from human disturbance.

Partial descriptions of riparian wildlife communities have been reported for many areas of the country, but thorough characterizations are not readily available for most (Table 22). The value of riparian ecosystems to wildlife has been most intensively studied in western arid regions, the Midwest, and the lower Mississippi Valley where threats to riparian ecosystems tend to be greatest.

Table 22. References for information on riparian wildlife communities in the U.S.A.

Region	State	References[a]
California	California	AB(12), Gaines (1974, 1977), Goldwasser (1978), Hehnke and Stone (1978), Ingles (1950), Michny et al. (1975), Roberts et al. (1977), Sands (1977, 1978).
Pacific Northwest	Oregon	Hinschberger (1978), Thomas (1979)
	Washington	Lewke (1975), McKern (1976)
Rocky Mountain	Colorado	Armstrong (1977), Beidleman (1948, 1954), Fitzgerald (1978), Wooding (1973)
	Montana	AB(2)
	Utah	AB(2), Whitmore (1975)
	Wyoming	AB(1), Brown (1967)
Arid Southwest	Arizona	Anderson and Ohmart (1977), Arnold (1940), Carothers et al. (1974), Johnson et al. (1977), Johnson (1978), Johnson and Simpson (1971), Stevens et al. (1977), Szaro (1980)
	Nevada	Austin (1970)
	New Mexico	Hubbard (1971), Schmidt (1976)
	Texas	AB(2), Boerr and Schmidly (1977), Engel-Wilson and Ohmart (1978), Wauer (1977)
Plains-Grasslands	Colorado	AB(7), Beidleman (1948, 1954), Crouch (1961)
	Kansas	AB(2), Tubbs (1980), Zimmerman and Tatschl (1975)
	North Dakota	AB(4)
	Oklahoma	AB(7), Barclay (1978, 1980), Blair (1939), Heller (1978)
	South Dakota	Emmerich (1978)
Corn Belt	Illinois	AB(1), Blem and Blem (1975a,b), Yeager (1949), Yeager and Anderson (1944), Wetzel (1958)
	Iowa	AB(1), Best et al. (1980), Geier (1978), Geier and Best (1980), Hoslett (1961), Stauffer and Best (1980)
	Indiana	AB(6), New (1972)
	Ohio	AB(6), Leite (1972)
Lake States	Minnesota	Dawson (1979), Iverson et al. (1967), Kirby (1975)
	Wisconsin	Dawson (1979), Faanes (1979), Prellwitz (1976)
Mississippi Delta	Arkansas	AB(5)
	Louisiana	AB(3), Glasgow and Noble (1971), Kennedy (1977), Ortego et al. (1976)
	Missouri	Fredrickson (1979)
Northeast-Appalachian	Connecticut	AB(1), Golet (1976), Miller and Getz (1977)
	Delaware	AB(2)
	Maine	AB(2)
	Maryland	AB(7)
	Massachusetts	AB(3), Golet (1976), Swift (1980)
	New Jersey	AB(2)
	New Hampshire	AB(1)
	New York	AB(3), Hardin (1975), Malecki and Eckler (1980), Webb et al. (1972)
	Pennsylvania	AB(2)
	Tennessee	Hooper (1967)
	Vermont	AB(1), Dodge et al. (1976), Miller and Getz (1977), Possardt and Dodge (1978)
	Virginia	AB(2), Ellis (1976), Gill et al. (1975), Hooper (1967)
	West Virginia	AB(5)
Southeast	Alabama	AB(1)
	Georgia	AB(1), Wharton (1970, 1978)
	Louisiana/Texas	Dickson (1978)
	North Carolina	AB(2)
	South Carolina	AB(1), Hair et al. (1978)
Alaska	Alaska	Kessel and Cade (1958), Maher (1959), Sage (1974)

[a]"AB" indicates that breeding bird census data have been published from one or more sites (number of sites in parenthesis) in American Birds or Audubon Field Notes.

Three groups of wildlife are described here: birds, mammals, and herps (reptiles and amphibians). The purpose of this section is to identify wildlife species or groups that are commonly found in riparian ecosystems. Where possible, the relative abundance and diversity of animal communities are described. Although some fish communities are dependent on riparian vegetation, they are not characterized in this report.

Birds

Birds are probably the most common, conspicuous, and easily studied form of wildlife in riparian ecosystems. As a result, and because of their general aesthetic popularity, there has been much research that describes riparian bird communities.

Community Characteristics. Birds using riparian ecosystems can be categorized into at least four groups based on their seasonal occurrence: (1) summer (breeding) residents; (2) winter residents; (3) transients (passing through during fall and/or spring migrations; and (4) permanent residents (nonmigratory species). As a result of many factors (migratory and local movements, reproduction, mortality, and seasonally changing habitat requirements), bird populations are distinctly different from season to season.

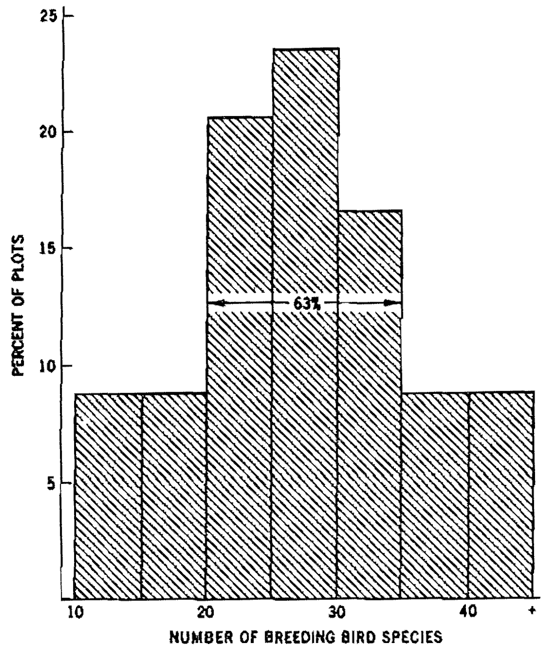


Figure 35. Number of breeding bird species on 98 riparian census plots (from Breeding Bird Census data published in Audubon Field Notes and American Birds).

Riparian ecosystems are valuable as breeding habitats for birds everywhere in the U.S.A. Individual stands of riparian woodland usually have 10 to 50 breeding bird species, with most having between 20 and 34 (Figure 35, Table 23). Population densities of birds breeding in riparian areas generally fall between 40 and 900 pairs per 40 ha (Table 24), but most often are between 150 and 550 pairs per 40 ha (Figure 36). Presumably, bird density reflects productivity and is a good measure of the availability of birds for observation by birdwatchers, photographers, hikers, etc.

The value of riparian ecosystems to winter bird populations has received increased attention from biologists recently (Dickson 1978, Szaro 1980). The species richness of bird communities in riparian vegetation during winter is generally comparable to that in summer, except in the most interior areas of the U.S.A. (Table 25). The abundance of winter residents is commonly equal to or greater than that of summer birds (Table 26), especially where there is a major influx from northern and inland breeding grounds (Lewke 1975, Kennedy 1977, Barclay 1980, Szaro 1980).

Riparian ecosystems are also important to birds during migration (Rappole and Warner 1976, Stevens et al. 1977, Fitzgerald 1978). Many riparian birds use the same habitats, when available, during migration as they do on their nesting grounds (Parnell 1969). Consequently, the number of species found in a riparian ecosystem during spring and fall is increased, because it includes departing and incoming seasonal resi-

Table 23. Number of breeding bird species on riparian study areas.

Community type and location	No. of species	Source
Riparian vegetation, Texas	38	Wauer 1977
Cottonwood-willow, Texas	27	Engel-Wilson and Ohmart 1978
Saltcedar, Texas	28	Engel-Wilson and Ohmart 1978
Desert riparian, California	13	Berry 1977
Willow-cottonwood, California	20	Ingles 1950
Cottonwood-willow, California	27	Gaines 1977
Various types, Arizona	18-35	B. Anderson et al. 1977
Bottomland forest islands, Okl.	11-15	Barclay 1978
Mature floodplain forest, Mo.	31	Zimmerman and Tatschl 1975
Young floodplain forest, Mo.	19	Zimmerman and Tatschl 1975
Bottomland hardwoods, Louisiana	16-23	Dickson 1978
Beaver ponds, South Carolina	15	Hair et al. 1978
Riparian forests, New York	33	Malecki and Eckler 1980
Riparian corridor, New York	24	Malecki and Eckler 1980
Alder, New York	26	Hardin 1975
Shrub, Alaska	8	Sage 1974

Table 24. Breeding bird densities in riparian ecosystems.

Plant community type and location	Density (pairs per 40 ha)	Source
Cottonwood-willow forest, Ca.	840	Gaines 1977
Willow-cottonwood streambottom, Ca.	197	Ingles 1950
Sacramento Valley riparian, Ca.	240-450	Gaines 1977
Desert riparian, California	863	Berry 1977
Desert bosques, Nevada	44-49	Austin 1970
Floodplain vegetation, Arizona	200-325	Cohan et al. 1978
Cottonwood, Arizona	425-847	Carothers et al. 1974
Mixed riparian vegetation, Arizona	193-322	Carothers et al. 1974
Willow, Colorado	100	Fitzgerald 1978
Cottonwood-willow, Colorado	525-589	Fitzgerald 1978
Cottonwood-willow, Colorado	225-900	Beidleman 1954
Cottonwood, Colorado	319	Bottorff 1974
Saltcedar, Colorado	131-503	B. Anderson et al. 1977
Saltcedar, Texas	486	Engel-Wilson and Ohmart 1978
Cottonwood-willow, Texas	708	Engel-Wilson and Ohmart 1978
Bottomland forests, Oklahoma	400	Barclay 1978
Bottomland hardwoods, Louisiana, Tx.	300-590	Dickson 1978
Riparian vegetation, New York	59-167	Malecki, and Eckler 1980
Riparian communities, Great Plains	137-748	Szaro (1980)

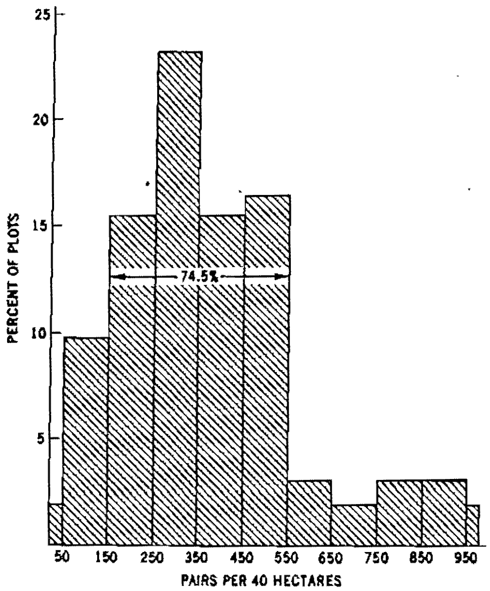


Figure 36. Breeding bird densities on 98 riparian census plots. From Breeding Bird Census reports published in Audubon Field Notes and American Birds.

dents in addition to any totally transient species. During migration periods, density of birds in riparian ecosystems depends heavily on availability of avian foods; riparian areas often provide optimal feeding areas needed for successful migrations by waterfowl, shorebirds, and other assemblages.

Characteristic Species. Riparian bird communities are generally comprised of numerous passerine species, several birds of prey, several upland game birds, and a variety of birds associated with aquatic feeding areas. These species can be grouped into one of various "guilds" according to their feeding habits (Table 27). During the breeding season, over half of the birds are species that forage for insects on foliage (vireos, warblers) or species that forage for seeds on the ground (doves, orioles, grosbeaks, sparrows) (Anderson and Ohmart 1977, Gaines 1977, Heller 1978, Swift 1980). Next in abundance are the ground feeding and bark feeding insectivorous species, such as the wood

Table 25. Number of winter bird species on riparian study areas.

Plant community type and location	No. of species	Source
Cottonwood-willow, Texas	23	Engel-Wilson and Ohmart 1978
Saltcedar, Texas	13	Engel-Wilson and Ohmart 1978
Desert riparian, California	26	Berry 1977
Mature floodplain forest, Missouri	16	Zimmerman and Tatschl 1975
Young floodplain forest, Missouri	9	Zimmerman and Tatschl 1975
Semi-disturbed woodland, California	62	Ryder and Ryder 1979
Willow, California	12,20	Ryder and Ryder 1979
Desert riparian, California	13,26	Ryder and Ryder 1979
Paloverde-ironwood-smoketree, Ca.	36,40	Ryder and Ryder 1978
Coast live oak, California	17-34	Ryder and Ryder 1978
Cottonwood, Colorado	19	Ryder and Ryder 1978
Cottonwood-willow, Colorado	27	Ryder and Ryder 1979
Oak-juniper canyon, Arizona	64	Ryder and Ryder 1978
Mesquite-juniper canyon, Texas	40	Ryder and Ryder 1978
Mixed habitat-disturbed, Oklahoma	40	Ryder and Ryder 1979
Floodplain forest, Illinois	25,28	Ryder and Ryder 1978
Oak-gum-cypress, Mississippi	39	Ryder and Ryder 1978
Hickory-oak ash, Maryland	38	Ryder and Ryder 1979
Mature foodplain forest, Maryland	31	Ryder and Ryder 1978
Disturbed coastal floodplain, Va.	40	Ryder and Ryder 1978
Woodland floodplain, New York	26	Ryder and Ryder 1978
Riparian woodlands, South Dakota	14	Emmerich 1978
Riparian vegetation, Washington	46	Lewke 1975
Bottomland forests, Illinois	37,47	Graber and Graber 1978

Table 26. Densities of riparian bird populations in winter. From Ryder and Ryder 1978, 1979.

Plant community type and location	No. per 40 ha
Bottomland oak-gum-cypress, Mississippi	475
Floodplain forest, Illinois (two locations)	148, 226
Riparian woodland, New York	99
Hickory-oak-ash forest, Maryland	274
Mature floodplain forest, Maryland	240
Coastal disturbed floodplain, Virginia	272
Mixed habitat-disturbed bottomland, Oklahoma	183
Floodplain cottonwood, Colorado	186
Cottonwood-willow riverbottom, Colorado	311
Oak-juniper canyon, Arizona	1016
Semi-disturbed riparian, California	728
Willow riparian, California (two locations)	209, 1606
Desert riparian willows, California (two locations)	345, 609
Coast live oak riparian, California (various locations)	366-659
Blue paloverde-ironwood-smoketree, California (two locations)	219, 405
Mesquite-juniper canyon, Texas	503

Table 27. Foraging guilds of riparian birds.

Foraging substrate	Major component of diet	Example
Generalist	Omnivore	Starling, jays
Ground	Seed	Cardinal
Ground	Insect	Wood thrush
Ground	Mammal	Red-shouldered hawk
Foliage	Seed	Tufted titmouse
Foliage	Insect	Red-eyed vireo
Foliage	Bird	Screech owl
Foliage	Nectar	Hummingbirds
Bark	Insect	Woodpeckers
Air	Insect	Wood pewee
Water	Fish	Belted kingfisher
Water	Omnivore	Wood duck

thrush and Gila woodpecker, respectively. During winter when most insect populations are low, bark foraging birds and ground-foraging seedeaters are most abundant.

Riparian ecosystems also support a variety of birds with specialized foraging techniques such as herons, wood duck, kingfishers, hummingbirds, and raptors. Although most of these species are only locally abundant, they are important members of the ecosystem because each is uniquely adapted to inhabit the riparian environment. Aquatic areas are the primary habitat component for many of these riparian birds, especially wintering waterfowl (Fitzgerald 1978, Hair et al. 1978) and migrating marsh and shorebirds in inland areas of the continent.

Riparian ecosystems are inhabited by a fairly predictable set of feeding guilds. However, due to the uncertainty of the presence of habitat features, the exact species composition of a given area cannot be accurately predicted. Nationwide, over 250 species of birds have been observed using riparian vegetation for cover or feeding during some part of the year. However, in any given region, vegetation type, or season, the number of species is considerably less than the nationwide total.

In each region of the U.S.A., there are certain species that are commonly abundant or frequently seen in riparian ecosystems (Tables 28 and 29). Included among these are common forest and edge species, and others that are clearly dependent on the aquatic-woodland interface. Because these latter species require aquatic habitat and have a more restricted distribution, they are most seriously affected by hydrologic alterations of streams. In contrast, forest-dwelling birds are adversely affected by activities that reduce the size of riparian woodlands, a situation that could create additional habitat for the already common edge species (Table 30).

Mammals

Mammals are important in most riparian ecosytems, as part of various food chains, in their ability to modify riparian communities (e.g., beaver), and because they provide much opportunity for observation or harvesting by man. Although mammals are seen less often than birds, indirect evidence of their presence may be easily found.

The number of mammal species in a riparian woodland generally ranges from 5 to 30 (Table 31) with population densities varying greatly. A typical riparian mammal community may include several furbearers, a few small and medium sized mammals, and one or more large mammals (Table 32). While some of these are abundant in nonriparian areas, many depend on or prefer riparian ecosystems. Water-oriented mammals, especially the furbearers and certain small mammals are almost entirely restricted to riparian zones of streams, rivers, and lakes. Without healthy riparian ecosystems, the survival of many mammal species would be threatened.

Amphibians and Reptiles

Researchers have generally neglected studying amphibians and reptiles in favor of more economically important animals. However, these groups, collectively referred to as "herps," are important in riparian food chains, and are now being recognized as valuable indicators of environmental quality (Orser and Shure 1972, Dodd 1978). Much additional information is needed to better understand the role of reptiles and amphibians in riparian ecosystems (Patton 1977).

Nearly all amphibians (salamanders, toads, frogs) depend on aquatic habitats for reproduction and overwintering, and many species are specifically adapted and restricted to riparian environments (Hairston 1949, Organ 1961, Tilley 1973, Fredrickson 1979, Wharton 1978, Krzysik 1979). Although reptiles are generally less restricted in relation to water, a clear preference for riparian ecosystems is displayed by various turtles, snakes, alligator, and many others.

The diversity of amphibians and reptiles in riparian ecosystems is probably comparable to that of mammals, except in the Southeast, where a tremendous variety of herps occur in riverbottom riparian areas. Reptiles and amphibians that are commonly or typically

Table 28. Most abundant breeding birds on 98 census plots in riparian vegetation. From breeding bird census reports published in Audubon Field Notes and American Birds.

Species	No. of plots observed on	Habitat preference Edge	Forest
Red-eyed vireo	40		x
Northern cardinal	31	x	
Common yellowthroat	28	x	x
Song Sparrow	26	x	
Wood Thrush	23		x
American redstart	22	x	x
Acadian flycatcher	21		x
Red-winged blackbird	20	x	
European starling	20	x	x
American robin	19	x	x
Gray catbird	19	x	
Tufted titmouse	17		x
House wren	16	x	
Mourning dove	16	x	
Eastern wood pewee	15		x
Yellow warbler	15	x	
Rufous-sided towhee	12	x	x
Northern oriole	12	x	
Indigo bunting	11	x	
Ovenbird	11		x
Parula warbler	11		x
Common flicker	11	x	x
Blue jay	11		x

found in riparian ecosystems have been identified by Beidleman (1954), Conant (1958), Stebbins (1966), Hardin (1975), McKern (1976), Roberts et al. (1977), Wharton (1978), and Barclay (1980) (Tables 33 and 34).

SIGNIFICANCE OF RIPARIAN ECOSYSTEMS TO FISH AND WILDLIFE

The importance of riparian ecosystems to fish and wildlife has been evaluated by two basic approaches: (1) by comparing the productivity (abundance) and diversity of wildlife in riparian versus other ecosystems, and (2) by establishing the dependence of species on riparian habitats. Based on available information, riparian ecosystems can be regarded as extremely valuable to fish and wildlife.

Comparison of Riparian and Nonriparian Wildlife Communities

Riparian areas are fairly consistent in having relatively high productivity and diversity of animal species. This results from the abundance of wildlife foods, and the presence of very diverse wildlife habitats within the riparian zone.

Riparian ecosystems are among the most productive areas for wildlife in the U.S.A., with few exceptions. For example, the density of birds observed in riparian forests exceeds that in upland vegetation by as much as two-fold in many states (Table 35). This is not totally attributable to the abundance of riparian-dependent species; riparian communities are regarded as the most productive or preferred habitat type for many "upland" species, including white-winged dove (Shaw and Jett 1959), screech owl (VanCamp and Henny 1975), red-shouldered hawk (Stewart 1949), woodcock (Horton and Causey 1979), white-tailed deer (Glasgow and Noble 1971, Fitzgerald 1978, Short and Shamberger 1979, Zwank et al. 1979), elk (Oregon Department of Fish and Wildlife 1980), squirrels (Gill et al. 1975), wild turkey (Glasgow and Noble 1971, Gill et al. 1975), and a variety of songbirds.

Becuase riparian ecosystems are suitable for many upland as well as riparian species, a majority of the species in any given region may be found there (Table 36). Riparian ecosystems support a greater diversity of wildlife than nearly all non-water related habitats.

Table 29. Distribution of common bird species in riparian ecosystems[a].

Species	California	Pacific Northwest	Rocky Mountain	Arid Southwest	Plains-Grasslands	Lake States	Corn Belt	Mississippi Delta	Northeast-Appalachian	Southeast	Alaska
	Region										
Double-crested cormorant	-	-	-	-	d	-	-	-	-	-	-
Great blue heron	d	d	-	d	p	d	d	p	d	p	-
Green heron	-	d	-	d	d	d	-	p	d	d	-
Black-crowned night heron	-	d	-	d	d	-	-	-	-	-	-
Yellow-crowned night heron	-	-	-	-	d	-	-	d	-	p	-
Mallard	-	-	-	-	d	d	-	p	d	p	-
Black duck	-	-	-	-	-	-	-	-	d	-	-
Blue-winged teal	-	-	-	-	d	-	-	-	-	-	-
Shoveler	-	-	-	-	-	-	-	d	-	-	-
Wood duck	-	d	-	-	d	d	d	p	d	p	-
Mergansers (Hooded&Common)	-	d	-	d	d	d	-	-	-	-	-
Cooper's Hawk	d	-	d	d	d	d	-	-	-	-	-
Red-tailed hawk	d	-	-	-	d	d	-	-	-	-	-
Red-shouldered hawk	d	-	-	-	-	d	-	p	d	p	-
Broad-winged hawk	-	-	-	-	-	d	-	-	d	-	-
Bald eagle	-	d	-	-	d	d	-	-	d	d	d
Osprey	-	d	-	-	-	d	-	-	d	-	d
Ptarmigans (Willow&Rock)	-	-	-	-	-	-	-	-	-	-	p
Ruffed grouse	-	-	-	-	-	d	-	-	-	-	-
Bobwhite quail	-	-	-	-	d	-	d	-	-	-	-
California quail	p	-	-	-	-	-	-	-	-	-	-
Gambel's Quail	-	-	-	p	-	-	-	-	-	-	-
Ring-necked pheasant	d	-	-	-	p	-	p	-	-	-	-
Turkey	-	-	-	-	d	-	-	p	d	p	-
Gallinules (Purple&Common)	-	-	-	-	d	-	-	d	-	-	-
Killdeer	d	p	d	-	p	-	-	-	d	-	-
Solitary sandpiper	-	d	-	-	-	-	-	-	-	-	-
Spotted sandpiper	d	d	d	d	d	d	d	-	d	-	d
Woodcock	-	-	-	-	d	d	-	d	d	d	-
Common snipe	-	d	-	-	d	-	-	-	-	d	p
White-winged dove	-	-	-	p	-	-	-	-	-	-	-
Mourning dove	d	d	d	p	p	-	d	-	-	-	-
Ground dove	-	-	-	d	-	-	-	-	-	-	-
Yellow-billed cuckoo	-	d	-	d	d	d	d	p	d	p	-
Black-billed cuckoo	-	-	-	-	p	d	-	-	-	-	-
Roadrunner	-	-	-	d	-	-	-	-	-	-	-
Screech owl	p	d	-	d	d	d	d	-	d	-	-
Barred owl	-	-	-	-	-	-	-	d	-	-	-
Nighthawks (Common&Lesser)	-	d	-	d	-	-	-	-	-	-	-
Chimney swift	-	-	-	-	-	-	-	d	-	d	-

Ruby-throated hummingbird	-	-	-	d	d	-	d	-	-	-	-
Black-chinned hummingbird	d	d	p	-	-	-	-	-	-	-	-
Anna's hummingbird	d	-	-	d	-	-	-	-	-	-	-
Broad-tailed hummingbird	d	d	-	-	-	-	-	-	-	-	-
Rufous hummingbird	d	-	-	-	-	-	-	-	-	-	-
Belted kingfisher	d	d	d	-	d	d	-	-	d	d	d
Common flicker	d	d	d	p	p	d	d	d	d	d	-
Pileated woodpecker	-	-	-	-	d	d	-	d	d	d	-
Red-bellied woodpecker	-	-	-	-	d	d	d	p	d	p	-
Gila woodpecker	-	-	d	-	-	-	-	-	-	-	-
Red-headed woodpecker	-	-	-	p	-	d	-	-	-	-	-
Acorn woodpecker	d	-	-	-	-	-	-	-	-	-	-
Lewis' woodpecker	d	-	-	d	-	-	-	-	-	-	-
Yellow-bellied sapsucker	d	d	-	d	d	-	d	-	d	-	-
Hairy woodpecker	d	d	-	p	d	d	p	d	d	-	-
Downy woodpecker	d	d	d	-	p	d	p	p	d	d	-
Ladder-backed woodpecker	-	-	d	d	-	-	-	-	-	-	-
Nuttall's woodpecker	d	-	-	-	-	-	-	-	-	-	-
Eastern kingbird	-	d	-	-	p	d	d	-	d	-	-
Western kingbird	p	d	d	d	d	-	-	-	-	-	-
Great-crested flycatcher	-	-	-	p	d	d	p	d	d	-	-
Wied's-crested flycatcher	-	-	d	-	-	-	-	-	-	-	-
Ash-throated flycatcher	p	d	-	p	d	-	-	-	-	-	-
Eastern phoebe	-	-	-	d	-	d	d	d	-	-	-
Black phoebe	d	-	-	d	-	-	-	-	-	-	-
Say's phoebe	d	-	d	-	-	-	-	-	-	-	-
Yellow-bellied flycatcher	-	-	-	-	d	d	-	-	-	-	-
Dusky flycatcher	d	-	-	-	-	-	-	-	-	-	-
Acadian flycatcher	-	-	-	-	d	d	-	p	d	-	-
Willow or Alder flycatcher (Traill's)	d	p	d	d	-	-	-	-	-	-	-
Least flycatcher	-	-	-	d	-	-	-	d	-	-	-
Hammond's flycatcher	-	d	-	-	-	-	-	-	-	-	-
Vermillion flycatcher	-	-	d	-	-	-	-	-	-	-	-
Gray flycatcher	d	-	-	-	-	-	-	-	-	-	-
Olive-sided flycatcher	-	-	-	-	-	d	-	-	-	-	-

Table 29. Continued.

Species	California	Pacific Northwest	Rocky Mountain	Arid Southwest	Plains-Grasslands	Lake States	Corn Belt	Mississippi Delta	Northeast-Appalachian	Southeast	Alaska
	Region										
Eastern pewee	-	-	-	-	p	d	d	-	d	-	-
Western pewee	d	d	d	d	d	-	-	-	-	-	-
Tree swallow	d	p	d	-	d	d	-	-	d	-	-
Bank swallow	d	d	-	d	-	d	-	-	d	-	d
Rough-winged swallow	d	d	-	d	d	d	-	-	-	-	-
Cliff swallow	d	-	d	d	-	d	-	-	-	-	-
Blue jay	-	-	-	-	p	d	d	d	d	p	-
Stellar's jay	d	d	-	-	-	-	-	-	-	-	-
Scrub jay	d	-	-	-	-	-	-	-	-	-	-
Yellow-billed magpie	d	-	-	-	-	-	-	-	-	-	-
Blue-billed magpie	d	d	-	d	-	-	-	-	-	-	-
American crow	d	d	-	-	d	d	-	-	-	-	-
Black-capped chickadee	d	p	-	p	-	d	-	d	-	-	-
Carolina chickadee	-	-	-	d	-	-	p	d	p	-	-
Mountain chickadee	-	d	-	-	-	-	-	-	-	-	-
Tufted titmouse	-	-	-	-	d	d	d	p	d	p	-
Plain titmouse	d	-	-	-	-	-	-	-	-	-	-
Verdin	-	-	p	-	-	-	-	-	-	-	-
Bushtit	d	d	-	-	-	-	-	-	-	-	-
White-breasted nuthatch	d	-	d	d	p	d	d	-	d	d	-
Brown creeper	d	-	-	-	d	d	-	d	d	d	-
Wrentit	d	-	-	-	-	-	-	-	-	-	-
Dipper	-	d	d	-	-	-	-	-	-	-	d
House wren	d	d	d	d	p	d	p	-	d	-	-
Winter wren	d	-	-	d	d	d	d	d	-	-	-
Bewick's wren	p	d	-	d	d	-	-	-	-	-	-
Carolina wren	-	-	-	d	-	-	p	d	p	-	-
Marsh wrens (Long&Short-billed)	d	-	d	-	-	-	d	-	-	-	-
Canyon wren	d	-	-	-	-	-	-	-	-	-	-
Northern mockingbird	-	-	d	d	-	-	-	-	-	-	-
Gray catbird	-	d	d	-	p	d	d	d	d	d	-
Brown thrasher	-	-	-	p	-	d	d	-	-	-	-
Crissal thrasher	-	-	d	-	-	-	-	-	-	-	-
Sage thrasher	d	-	-	-	-	-	-	-	-	-	-
American robin	d	d	p	d	p	d	d	p	d	d	p
Wood thrush	-	-	-	-	d	d	d	d	p	d	-
Hermit thrush	d	-	-	-	-	d	-	d	-	d	-
Swainson's thrush	d	d	-	-	d	-	-	-	-	-	-
Gray-cheeked thrush	-	-	-	-	-	-	-	-	-	p	-

Veery	-	d	-	d	d	d	-	p	d	-	-
Eastern bluebird	-	-	-	d	-	-	-	-	-	-	-
Western bluebird	d	-	-	-	-	-	-	-	-	-	-
Bluethroat	-	-	-	-	-	-	-	-	-	p	-
Townsend's solitaire	d	-	-	-	-	-	-	-	-	-	-
Arctic warbler	-	-	-	-	-	-	-	-	-	p	-
Blue-gray gnatcatcher	-	-	d	d	d	d	p	d	d	-	-
Black-tailed gnatcatcher	-	-	p	-	-	-	-	-	-	-	-
Golden-crowned kinglet	d	p	-	-	d	d	-	d	d	d	-
Ruby-crowned kinglet	p	d	-	p	d	d	-	d	d	d	-
Yellow wagtail	-	-	-	-	-	-	-	-	-	p	-
Cedar waxwing	d	d	d	-	d	d	-	d	-	-	-
Phainopepla	-	-	p	-	-	-	-	-	-	-	-
Northern shrike	d	-	-	-	-	-	-	-	-	-	d
Loggerhead shrike	d	-	d	-	-	-	-	-	-	-	-
Starling	p	p	d	d	p	-	d	d	d	d	-
Black-capped vireo	-	-	-	d	-	-	-	-	-	-	-
White-eyed vireo	-	-	-	-	d	-	-	p	d	p	-
Bell's vireo	-	-	p	-	-	-	-	-	-	-	-
Yellow-throated vireo	-	-	-	d	-	d	d	-	d	-	-
Solitary vireo	d	d	-	-	-	d	d	p	d	-	-
Red-eyed vireo	-	d	p	-	d	-	d	p	p	p	-
Warbling vireo	-	d	p	p	d	d	d	-	d	-	-
Black and white warbler	-	-	-	d	d	d	-	d	-	-	-
Pronthonotary warbler	-	-	-	-	d	d	-	p	d	p	-
Swainson's warbler	-	-	-	-	-	-	d	d	d	-	-
Tennessee warbler	-	-	-	d	d	-	-	-	-	-	-
Orange-crowned warbler	d	d	d	p	-	-	-	d	-	-	-
Nashville warbler	d	p	d	-	-	d	-	-	-	-	-
Lucy's warbler	-	-	p	-	-	-	-	-	-	-	-
Parula warbler	-	-	-	-	-	d	-	p	d	p	-
Yellow warbler	d	p	p	d	d	d	d	-	p	-	-
Magnolia warbler	-	-	-	-	d	d	-	-	-	-	-
Black-throated blue warbler	-	-	-	-	-	-	-	-	-	-	-
Yellow-rumped warbler	p	-	d	p	-	d	-	d	d	p	-

Table 29. Concluded.

Species	Region										
	California	Pacific Northwest	Rocky Mountain	Arid Southwest	Plains-Grasslands	Lake States	Corn Belt	Mississippi Delta	Northeast-Appalachian	Southeast	Alaska
Black-throated green warbler	-	-	-	-	-	d	-	-	p	-	-
Cerulean warbler	-	-	-	-	d	d	-	d	-	-	-
Blackburnian warbler	-	-	-	-	d	-	-	d	-	-	-
Chestnut-sided warbler	-	-	-	-	d	-	-	-	-	-	-
Yellow-throated warbler	-	-	-	-	-	-	d	-	-	-	-
Bay-breasted warbler	-	-	-	-	d	-	-	-	-	-	-
Pine warbler	-	-	-	-	-	-	d	-	d	-	-
Ovenbird	-	-	-	-	d	d	d	-	d	-	-
Northern waterthrush	d	-	-	d	d	d	-	p	-	-	-
Louisiana waterthrush	-	-	-	d	d	d	d	d	p	-	-
Kentucky warbler	-	-	-	d	d	-	p	d	-	-	-
MacGillivray's warbler	p	p	-	-	-	-	-	-	-	-	-
Common yellowthroat	-	p	-	p	p	d	d	-	p	-	-
Yellow-breasted chat	d	-	p	d	-	-	d	-	d	-	-
Hooded warbler	-	-	-	d	-	-	p	d	p	-	-
Wilson's warbler	d	d	d	-	-	-	-	-	-	-	p
Canada warbler	-	-	-	-	p	-	-	p	-	-	-
American redstart	-	d	p	-	d	d	d	d	d	d	-
House sparrow	d	-	d	p	-	d	-	-	-	-	-
Western meadowlark	-	p	d	-	p	-	-	-	-	-	-
Red-winged blackbird	-	p	d	d	d	d	d	d	d	d	-
Orchard oriole	-	-	p	p	-	-	d	-	d	-	-
Northern oriole	d	p	d	p	p	d	d	d	d	-	-
Rusty blackbird	-	-	-	d	-	-	d	d	p	-	-
Great-tailed grackle	-	-	d	-	-	-	-	-	-	-	-
Common grackle	-	-	-	p	-	-	p	d	d	-	-
Brown-headed cowbird	d	d	p	p	p	d	d	d	d	-	-
Scarlet tanager	-	-	-	d	-	d	-	d	-	-	-
Summer tanager	-	-	p	-	-	-	d	-	-	-	-
Northern cardinal	-	-	-	d	d	-	d	p	d	p	-
Rose-breasted grosbeak	-	-	-	p	d	d	-	d	-	-	-
Black-headed grosbeak	p	d	d	p	d	-	-	-	-	-	-
Blue grosbeak	-	-	p	d	-	-	-	-	-	-	-
Indigo bunting	-	-	-	d	d	d	d	-	-	-	-
Lazuli bunting	d	d	-	d	d	-	-	-	-	-	-
Painted bunting	-	-	p	d	-	-	-	-	-	-	-
Dickcissel	-	-	-	-	-	-	-	-	-	-	-
Evening grosbeak	d	-	-	-	-	-	-	d	-	-	-
Purple finch	d	d	-	-	d	d	-	d	d	d	-
Cassin's finch	d	d	p	-	-	-	-	-	-	-	-

House finch	p	d	-	-	d	-	-	-	-	-	-
Hoary redpoll	-	-	-	-	-	-	-	-	-	p	-
Pine siskin	d	p	-	d	-	d	-	-	-	-	-
American goldfinch	d	p	d	-	p	-	d	d	d	-	-
Lesser goldfinch	d	-	-	-	-	-	-	-	-	-	-
Green-tailed towhee	-	d	d	-	-	-	-	-	-	-	-
Rufous-sided towhee	p	d	-	d	d	d	d	d	d	p	-
Brown towhee	d	-	-	p	-	-	-	-	-	-	-
Abert's towhee	-	-	p	-	-	-	-	-	-	-	-
Savannah sparrow	d	-	-	-	-	-	-	-	-	p	-
Sage sparrow	-	-	d	-	-	-	-	-	-	-	-
Northern junco	p	p	d	p	d	d	-	-	p	-	-
Tree sparrow	-	-	-	d	-	-	-	d	-	p	-
Chipping sparrow	d	d	d	-	d	d	-	-	-	-	-
White-crowned sparrow	p	p	d	p	d	-	-	-	-	-	p
White-throated sparrow	-	-	-	d	d	d	-	d	p	p	-
Fox sparrow	d	d	-	d	-	-	-	-	-	p	-
Lincoln's sparrow	d	d	d	d	d	-	-	-	-	-	-
Swamp sparrow	-	-	d	d	-	d	d	d	d	-	-
Song sparrow	d	p	p	p	p	d	d	d	p	d	-
Lapland longspur	-	-	-	-	-	-	-	-	-	p	-

[a]Distribution indicated as follows:
"d" - documented as abundant or characteristic of riparian ecosystems.
"p" - probably found in riparian ecosystems to a lesser extent.
"-" - no documented evidence of regular occurrence in riparian ecosystems.

Table 30. Bird species that are locally abundant along streams and rivers.

Species	Primary distribution (FWS region)						
	1	2	3	4	5	6	Alaska
Great blue heron	x	x	x	x	x	x	-
Green heron		x	x	x		-	-
Black-crowned night heron		x	x	x		x	-
Wood duck	x	-	x	x	x	-	-
Red-shouldered hawk	x[a]	-	x	x	x	-	-
Gray hawk	-	x	-	-	-	-	-
Bald eagle	x	-	x	x	x	x	x
Osprey	x	-	x		x		x
American woodcock	-	x	x	x	x	-	-
Spotted sandpiper	x	x	x	x	x	x	x
White-winged dove		x	-	-	-	-	-
Common screech owl		x	x	x	x		-
Belted kingfisher	x	-	x	x	x	x	x
Bank swallow	-	-	x	-	x	x	x
North American dipper	x	x	-	-	-	x	x
Phainopepla	-	x	-	-	-	-	-
Prothonotary warbler	-	-		x	x	-	-
Louisiana waterthrush	-	-	x	x	x	-	-

[a]California.

Table 31. Number of mammal species in riparian ecosystems.

Community type and location	Total no. of species	Source
Cottonwood-willow stands, Colorado	43	Beidleman 1954
Various riparian types, Colorado	23	Fitzgerald 1978
Streamside areas, Colorado	33	Burkhard 1978
Mesquite stands, Arizona	35	Arnold 1940
Bottomland forests, Oklahoma	29	Barclay 1980
Riparian forest, Iowa	6[a]	Best et al. 1978
Streamside vegetation, Vermont	8[a]	Possardt & Dodge 1978
Bottomland hardwoods, Georgia	7[a]	Boyd 1976 (in Wharton 1978)

[a]Small mammals only.

Table 32. Distribution of common riparian mammals.

		Region[b]									
Species[a]	Riparian dependence	California	Pacific Northwest	Rocky Mountain	Arid Southwest	Plains-Grasslands	Lake States	Corn Belt	Mississippi Valley	Northeast-Appalachian	Southeast
Eastern mole		-	-	-	-	d	-	D	-	-	-
Common shrew		-	d	-	-	d	-	d	-	d	-
Water shrew	x	-	-	d	-	-	D	-	-	-	-
Short-tailed shrew	x	-	-	-	-	p	D	d	-	p	p
Little brown bat		D	D	-	D	-	D	-	-	D	D
Big brown bat		D	D	-	D	-	D	-	-	D	-
Pallid bat		D	D	-	D	-	-	-	-	-	-
Raccoon	x	d	p	p	p	p	D	D	D	p	p
Mink	x	d	p	p	-	d	D	d	D	D	p
River otter	x	d	p	p	-	-	D	-	D	d	d
Striped skunk		d	p	-	d	d	D	d	D	D	d
Coyote		D	p	-	p	p	D	d	D	-	d
Gray squirrels	x	d	-	-	-	-	-	d	p	d	p
Fox squirrels		-	-	d	-	p	-	D	p	-	d
Pocket gophers		D	d	d	p	d	D	d	-	-	-
Pocket mice		-	d	-	p	-	-	-	-	-	-
Kangaroo rats		-	d	-	p	-	-	-	-	-	-
Beaver	x	p	p	d	d	p	D	d	D	d	p
Harvest mouse (western)		D	p	-	d	d	-	d	-	-	-
Deer mice		-	p	d	d	p	D	d	D	p	p
White-footed mouse	x	-	-	-	p	p	D	p	-	p	-
Hispid cotton rat		-	-	-	p	d	-	-	-	-	-
Woodrats (Neotoma)		-	d	-	p	-	-	-	-	-	d
Red-backed vole	x	-	-	d	-	d	D	-	-	p	-
Voles		D	p	p	-	p	D	d	-	d	-
Muskrat	x	p	p	d	d	d	D	d	D	d	d
Meadow jumping mice		-	-	p	-	d	D	-	-	d	-
Woodland jumping mice		-	-	-	-	-	D	-	-	p	-
Cottontails		D	p	d	p	p	-	d	p	D	-
Swamp rabbit	x	-	-	-	-	d	-	-	p	-	p
Jackrabbits and hares		p	d	-	d	d	D	-	-	d	-
White-tailed deer		-	d	-	-	p	D	d	p	p	p
Mule deer		D	p	-	d	d	-	-	-	-	-

[a] Species noted with an "x" depend on or prefer riparian sites nationwide because of habitat requirements. Other species may be restricted to riparian ecosystems where other suitable habitats are unavailable locally.
[b] Symbols indicate the following: "D" - documented abundance in riparian areas within parts of region; "d" - documented presence, not necessarily abundant in region; "p" - probably abundant in riparian areas in the region; "-" - no evidence of presence in region's riparian communities.

Table 33. Common reptiles in riparian ecosystems.

Species[a]	Region[b]									
	California	Pacific Northwest	Arid Southwest	Rocky Mountain	Plains-Grasslands	Lake States	Corn Belt	Mississippi Valley	Northeast-Appalachian	Southeast
Alligator	-	-	-	-	-	-	-	d	-	D
Snapping turtle	-	-	-	-	D	D	d	D	D	D
Musk turtles	-	-	-	-	-	-	d	D	d	D
Mud turtles	-	-	D	-	d	-	d	D	-	D
Sliders, cooters, water and box turtles	p	D	d	d	D	D	d	D	d	p
Softshell turtles	-	-	d	-	D	-	D	D	-	D
Earless lizards	-	-	-	-	d	-	-	-	-	-
Spiny lizards	p	p	D	-	d	-	-	-	-	-
Tree lizard	-	-	d	d	-	-	-	-	-	-
Skinks	p	D	D	D	D	d	D	D	D	p
Whiptails and racerunners	p	d	D	-	D	-	d	d	-	d
Alligator lizards	p	d	d	-	-	-	-	-	-	-
Gila monster	-	-	d	-	-	-	-	-	-	-
Boas	d	D	d	D	-	-	-	-	-	-
Water snakes	-	-	-	-	d	D	D	D	D	D
Black swamp snake	-	-	-	-	-	-	-	-	-	p
Red-bellied and brown snakes	-	-	-	-	d	D	d	D	D	D
Garter snakes	p	p	D	d	p	D	D	D	p	D
Striped swamp snake	-	-	-	-	-	-	-	-	-	D
Rainbow and mud snakes	-	-	-	-	-	-	-	D	-	p
Racers	p	p	D	d	D	-	-	-	-	-
Whipsnakes	-	d	p	-	-	-	-	d	-	d
Green snakes	-	-	-	-	D	D	D	D	D	D
Rat snakes	-	-	D	-	D	-	d	D	-	D
Bullsnake (gopher snake)	p	p	d	d	D	-	-	-	-	-
King snakes	D	-	D	-	D	-	d	D	d	D
Cottonmouths	-	-	-	-	-	-	-	p	-	p
Massassauga & pygmy rattlesnakes	-	-	-	-	d	D	D	p	-	-
Rattlesnakes	p	p	D	-	d	-	d	D	d	D

[a] Common names for species or groups from Conant (1958) and Stebbins (1966).
[b] Symbols indicate the following:
"D" - documented abundance in riparian areas within parts of region.
"d" - documented presence, not necessarily abundant in region.
"p" - probably abundant in riparian areas in the region.
"-" - no evidence of presence in region's riparian communities.

Table 34. Common amphibians in riparian ecosystems.

	Region[b]									
Species[a]	California	Pacific Northwest	Arid Southwest	Rocky Mountain	Plains-Grasslands	Lake States	Corn Belt	Mississippi Valley	Northeast-Appalachian	Southeast
Mole salamanders & relatives	d	D	d	d	d	d	d	d	d	d
Newts	-	-	-	-	-	-	-	D	d	D
Pacific newts	d	d	-	-	-	-	-	-	-	-
Dusky salamanders	-	-	-	-	-	-	-	d	p	D
Woodland & slimy salamanders	-	d	-	-	-	-	-	-	d	d
Spring salamanders	-	-	-	-	-	-	-	-	D	-
Red & mud salamanders	-	-	-	-	-	-	-	-	D	D
Brook salamanders	-	-	-	-	-	-	-	-	D	D
Tailed frog	-	d	-	d	-	-	-	-	-	-
Spadefoot toads	d	d	D	-	d	-	-	-	-	-
True toads	p	D	D	d	p	D	D	d	d	D
Cricket frogs	-	-	-	-	D	-	D	d	-	D
Tree frogs	p	D	D	d	d	D	D	D	D	D
Chorus frogs	-	-	-	-	p	D	D	D	D	D
Narrow-mouthed toad	-	-	-	-	-	-	-	-	-	D
True frogs	p	D	d	D	D	D	D	D	p	D

[a] Common names for species or groups from Conant (1958) and Stebbins (1966).
[b] Symbols indicate the following:
"D" - documented abundance in riparian areas within parts of region.
"d" - documented presence, not necessarily abundant in region.
"p" - probably abundant in riparian areas in the region.
"-" - no evidence of presence in region's riparian communities.

Table 35. Comparisons of bird densities between riparian and upland ecosystems.

Location	Synopsis	Source
California	Breeding bird densities in cottonwood-willow equal or exceed those in any California vegetation type	Gaines 1974
Colorado	Breeding and winter bird densities are well in excess of all other terrestrial habitat types.	Bottorff 1974
Arizona	Density of passerines (migrant and breeding) during spring were 1.3 to 21 times higher in riparian woodlands than in adjacent nonriparian habitats.	Stevens et al. 1977
Louisiana and east Texas	Breeding bird densities in bottomland hardwoods were 2 to 4 times higher than in the best pine and pine-hardwood stands.	Dickson 1978
Illinois	Average total biomass of birds in floodplains was nearly twice that found in uplands.	Blem and Blem 1975
Virginia	Breeding birds were 44% more numerous on bottom slope transects as compared to midslope transects.	Hooper 1967
Southwest U.S.	Desert riparian vegetation supported an average of 3.8 times more birds per 40/ha than desert scrub vegetation.	Austin 1970
South Carolina	Beaver ponds had 1.5 to 2 times the number of birds found in upland sites during all seasons of the year.	Hair et al. 1978
Washington	Number of birds in riparian communities was 2 to 4 times that in an equal area of nonriparian communities.	Lewke 1975

Table 36. Proportion of wildlife species using riparian ecosystems.

Location	Species in area (number)	Species using riparian (number)	(percent)	Source
Sacramento Valley, Calif.	277 nesting birds in State	67	24	Hehnke and Stone 1978
Gila Valley, N.M.	112 breeding birds	80	71	Hubbard 1971
San Juan Valley, N.M.	105 breeding birds	79	75	Schmidt 1976
Colorado	600+ birds in State	245	40	Beidleman 1978
South Platte Valley, Colo.	151 vertebrates	147	97	Fitzgerald 1978
Roaring Fork, Colo.	124 breeding birds	42	34	Wooding 1973
South-central Okla.	22 large mammals	18	82	Barclay 1980
South-central Okla.	60 reptiles & amphibians	34	49	Barclay 1980
Louisiana	383 bird species	225	59	Glasgow and Noble 1971
Mississippi	54 mammal species	42	82	Glasgow and Noble 1971
Kaolak River Valley, Alaska	9 breeding passerine birds	5	56	Maher 1959
2 river valleys, Alaska	17+ breeding passerine birds	8	47	Sage 1974
Missour River Basin, Neb.	300+ species occur in basin	115	33	USDI Heritage Conserv. and Recr. Serv. 1979
Great Plains	325 breeding birds	136	42	Tubbs 1980
Southwestern	58 species in study area	40	69	Ports 1979

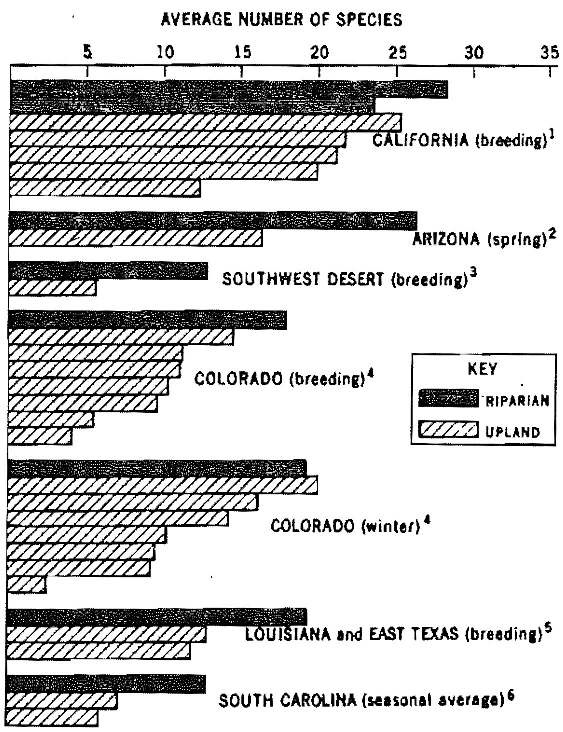


Figure 37. Number of bird species in riparian and upland vegetation types. Sources: (1) Gaines 1974; (2) Stevens et al. 1977; (3) Austin 1970; (4) Bottorff 1974; (5) Dickson 1978; (6) Hair et al. 1978.

This is especially true for birds (Figure 37), and amphibians.

In arid regions, riparian communities are more productive of wildlife than non-forested upland ecosystems, but the former are probably comparable to typical upland eastern forests. Wildlife productivity and diversity of riparian zones in humid climatic regions are also greater than on adjacent forested uplands, but probably to a lesser degree than in the arid West (Johnson 1978).

Dependence of Fish and Wildlife on Riparian Ecosystems

Without riparian ecosystems, many fish and wildlife species would be unable to survive, or would do so at lower densities. Of all the terrestrial wildlife species that occur in a locale or region, 10-80% depend on or prefer riparian ecosystems (Table 37). As many as 50% of bird species in some western states are found primarily in riparian vegetation; and may be dependent on those habitats. Dependence on riparian ecosystems is based on requirements for

Table 37. Number of terrestrial wildlife species dependent on or preferring riparian ecosystems.

Location	Synopsis	Source
Gila Valley, New Mexico	49% of breeding bird species are restricted to or prefer riparian.	Hubbard 1971
San Juan Valley, New Mexico	46% of breeding bird species are restricted to or prefer riparian.	Schmidt 1976
Upper Williams Fork Basin, Colorado	44% of the small mammal species have a primary affinity to riparian ecosystems.	Armstrong 1977
Great Basin, southeastern Oregon	79% of the vertebrate species are dependent on or prefer riparian zones.	Thomas et al. 1979b
Roaring Fork Watershed, Colorado	18% of the breeding bird species were entirely restricted to riparian vegetation.	Wooding 1973
Snake River Valley, Washington	50% of the bird species are dependent on riparian vegetation.	Lewke 1975
California	10% of the nesting bird species have a primary affinity to riparian forests.	Gaines 1974, 1977
California	39 mammal species, 19 herp species and 17 species of butterflies depend on riparian forests.	Sands 1978
Southwest lowlands	52% of nesting bird species are "obligate" or "preferential" riparian, and an additional 19% use wetlands and riparian areas extensively	Johnson et al. 1977

open water and/or riparian vegetation, as shown by herons, belted kingfisher, prothonotary warbler, several furbearing mammals, and most amphibians.

Extensive alteration of riparian ecosystems has occurred throughout the U.S.A., as described at the beginning of this report, accompanied by declining populations of many dependent species of fish and wildlife. At present, riparian ecosystems are important to about 80 (29%) of the 276 species or subspecies listed as threatened or endangered by the U.S. Fish and Wildlife Service (1980b) (Table 38). Included are 18 terrestrial species, such as Yuma clapper rail, bald eagle, gray bat, alligator, whooping crane, and Bachman's warbler, and a variety of aquatic species that are directly influenced by the amount and condition of riparian ecosystems. Many riparian species are candidates for future federal listing as threatened or endangered, including Bell's vireo, western populations of yellow-billed cuckoo, and an undetermined number of plants and invertebrates.

Table 38. Threatened and endangered animal species in riparian ecosystems (from U. S. Fish and Wildlife Service 1980b).

Mammals	Fishes (continued)
Bat, Indiana	Gambusia, San Marcos
Bat, gray	Killifish, Pahrump
Deer, Columbian white-tailed	Madtom, Scioto
Manatee, West Indian (Florida)	Madtom, yellowfin (T)
Pronghorn, Sonoran	Pupfish, Owens River
	Pupfish, Tecopa
Birds	Pupfish, Warm Springs
	Squawfish, Colorado River
Crane, whooping	Stickleback, unarmored threespine
Duck, Hawaiian	Sturgeon, shortnose
Eagle, bald	Topminnow, Gila
Falcon, American peregrine	Trout, Arizona (T)
Falcon, Arctic peregrine	Trout, Gila
Goose, Aleutian Canada	Trout, greenback cutthroat (T)
Kite, Everglades	Trout, Lahontan cutthroat (T)
Rail, Yuma clapper	Trout, Little Kern golden (T)
Warbler, Bachman's	Trout, Paiute cutthroat (T)
Woodpecker, ivory-billed	Woundfin
Reptiles/Amphibians	Snails
Alligator, American (T)[a]	Snail, Chittenango ovate amber (T)
Crocodile, American	
Salamander, San Marcos	Clams
	Pearly mussel, Alabama lamp
Fishes	Pearly mussel, Appalachian monkeyface
	Pearly mussel, birdwing
Bonytail, Pahranagat	Pearly mussel, Cumberland bean
Chub, bonytail	Pearly mussel, Cumberland monkeyface
Chub, humpback	Pearly mussel, Curtis
Chub, Mojave	Pearly mussel, dromedary
Chub, slender (T)	Pearly mussel, green-blossom
Chub, spotfin (T)	Pearly mussel, Higgin's eye
Cui-ui	Pearly mussel, orange-footed
Dace, Moapa	Pearly mussel, pale lilliput
Darter, bayou (T)	Pearly mussel, pink mucket
Darter, fountain	Pearly mussel, Sampson's
Darter, leopard (T)	Pearly mussel, tubercled-blossom
Darter, Maryland	Pearly mussel, turgid-blossom
Darter, Okaloosa	Pearly mussel, white cat's eye
Darter, slackwater (T)	Pearly mussel, white wartyback
Darter, snail	Pearly mussel, yellow-blossom
Darter, watercress	Pigtoe, rough
Gambusia, Big Bend	Pigtoe, shiny
Gambusia, Clear Creek	Pocketbook, fat
Gambusia, Goodenough	Riffle shell clam, tan
Gambusia, Pecos	

[a]Threatened species.

CHAPTER FIVE

THE VALUE OF RIPARIAN ECOSYSTEMS
INSTITUTIONAL AND METHODOLOGICAL CONSIDERATIONS

Numerous land and water uses affect the character and vitality of riparian ecosystems throughout the United States. The examples are familiar ones. To accomplish a variety of public and private purposes rivers are dammed and large amounts of land are inundated; streams are channelized and otherwise altered; the vegetation of riparian lands are cleared; and large quantities of water are withdrawn from rivers and streams. In many cases, the result of such practices is the physical alteration of riparian systems and the elimination of the natural functions performed by them. However, another look at the same examples tells us that such alterations are "improvements" which benefit people through increased water supply and hydroelectric power, new flood protection, enhanced navigation, increased agricultural production, and more sites for homes and commercial activities. Yet, both views of the problem beg more basic questions. In the case of alteration, do the social benefits of these developments exceed the social costs? For that matter, are all of the benefits and costs even considered? In the case of preservation, is not society denying itself significant benefits by failing to exploit rivers, streams, and related land resources? Therefore, a central issue for decisionmakers - public and private - concerns the value of riparian resources in different and often competing uses.

In recent years there has been a growing awareness that riparian ecosystems provide many useful goods and services to humans. Many of these are summarized in Table 39 and include natural flood storage capacity, water quality maintenance, recreational opportunities, habitat for fish and wildlife, and various aesthetic and scientific values. Riparian ecosystems also have the potential of providing goods and services that are available through their alteration. Consumer demand exists for the goods and services of riparian ecosystems both in their natural and their altered state. However, identifying goods and services that can be derived from natural or altered riparian areas is relatively simple when compared with the task of specifying the relative value of these goods and services.

The significant extent of riparian ecosystem alteration, as discussed in Chapter 2, might lead one to automatically conclude that the goods and services which result from such alteration have a greater value to society. Yet, there is reason to suggest that biases inherent in our institutional mechanisms for allocating society's resources favor alteration of natural ecosystems. Ideally, institutional systems of resource allocation should provide for the most valued uses of natural resources whether the goods and services derive from the natural sector, the result of alteration, or a mixture of the two.

The purpose of this chapter is to provide an overview of the key issues concerning the valuation of nature's goods and services, particularly those associated with riparian areas. In addressing this topic, it is necessary to come to grips with two aspects of the valuation problem - one institutional and the other methodological. The institutional aspect concerns the manner in which different institutions approach

Table 39. Qualitative list of values of riparian ecosystems. Adapted from Lugo and Brinson (1978).

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 and/or clays
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 they become 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

valuation. Resources which are allocated through the marketplace, regulated markets (i.e., those controlled by significant regulatory constraints), and by agencies with public resource management authority may approach the task of resource valuation somewhat differently. This chapter will review features of these institutions which cause such differences.

The second aspect actually includes a bundle of issues all related to the selection of valuation methodologies. To avoid needless confusion with basic terms, a few definitions will be useful. In the riparian wetland and floodplain literature, the term "value(s)" has been used to refer to many different things. Frequently, values are attributes such as flood storage capacity, groundwater recharge, water quality maintenance, habitat for fish and wildlife and others associated with natural or unaltered ecosystems (Jahn 1978, Greeson et al. 1978). Having identified these "values," one can proceed in assessing the extent to which an area supports or exhibits these values. This is typically accomplished through "valuation" procedures, some of which will be discussed later in the chapter.

Value and valuation, however, take on a more comprehensive meaning when viewed from a social perspective. The approach, most frequently associated with the discipline of economics, not only asks what values are supported within an area, but also what uses of that area, new or existing, are <u>most</u> valuable to society. This approach requires an abundance of information including ecological. It also requires that methods be designed to make ecological information meaningful to people - especially the decisionmakers who make actual choices about resource utilization (Comptroller General Feb. 8, 1979). Therefore, in the socio-economic and socio-political spheres, one cannot end with a recitation and ranking of ecological values. An attempt should be made to ascertain the importance of those values to society whether through enlightened benefit-cost analysis or some other form of analysis.

At the outset, it is important to inform the reader that much of this chapter addresses riparian values from an economic perspective.[1] Attacks on the use of economics to analyze environmental problems are abundant. While the economic approach is no panacea, the discipline can contribute much to the meaningful evaluation of goods and services of natural environments. There is nothing in economic theory that prevents consideration of natural values, intangible benefits, and aesthetic contributions to human welfare. Rather, the obstacles to appropriate consideration of these values are institutional and methodological. This chapter will attempt, among other things, to focus on the important issues which give rise to these obstacles.

This leads to one other point about economic analysis - its philosophical underpinnings. The basis of economic theory is profoundly utilitarian in that it concerns itself with the fulfillment of human needs and wants in a world of resource scarcity. Whether this is a good or bad thing is largely a question of philosophy. Certainly, other bases of value exist. For example, living or non-living resources may have value for their own sake and not just because they provide utility for people. This debate has gained some prominence in the environmental arena and is aptly discussed in a series of articles by Krieger (1973), Tribe (1974), and Sagoff (1974).

VALUATION: AN INSTITUTIONAL PERSPECTIVE

<u>Overview</u>

It is likely that resource valuation would not be a problem if resources

[1]There has been little economic analysis of riparian ecosystems as a distinct problem within the areas of natural resource and environmental economics. However, this chapter attempts to make general principles and concepts from these fields of study relevant to the analysis of riparian ecosystems. Wetland resource allocation has been examined by economists. Since issues concerning wetlands and riparian areas are similar, relevant information from those studies will be applied to issues raised in this chapter.

were available in unlimited quantities. In that ideal situation, resources could be consumed for some purposes without making them less available for others. However, it is generally accepted that society possesses scarce resources which must be used to satisfy the unlimited wants and needs of people (Barnett and Morse 1963). This is true of environmental resources as well. As Freeman et al. (1973) note, "managing the environment can be viewed as a problem of allocating the services of scarce environmental resources among competing ends or uses." For example, a secluded forested area might be used to provide people with "wilderness" experiences (and other compatible uses such as maintenance of wildlife habitat). Or, the area might be logged for timber or mined for its subsurface resources. However, the area cannot be used for all purposes. Some choices are mutually exclusive. As another example, a wetland might be used to provide a quality nesting area for migratory waterfowl or it might be drained, filled, and cultivated for production of a cash crop. Again, a single resource can be allocated for some but not all purposes.

While all uses of resources are not necessarily "mutually exclusive," tradeoffs do result when resources are used for one or a combination of purposes. The choice of using a resource for one purpose has a cost - the foregone opportunity for the other use. Activities in riparian areas provide a telling illustration of this situation. Yet, for reasons to be explained in this chapter, the characteristics of resource values emanating from natural environments make accurate assessment of these tradeoffs difficult in many circumstances.

Tradeoffs that are particularly difficult to evaluate are those involving irreversible consequences (Krutilla and Fisher 1975). In these circumstances, use of an area or a resource for one purpose makes irreversible the lost opportunity to use a resource for some other uniquely valuable purpose. For example, mining the geothermal energy of Yellowstone National Park would irreversibly eliminate the use of this resource for other purposes (e.g., watching "Old Faithful"). A project that results in the destruction of the last remaining habitat of an endangered species has irreversible consequences. Species extinction withdraws fauna and/or flora from the earth's reservoir of potential resources...including the gene pool. The long-term costs of such incidents are extremely difficult to evaluate (Bishop 1978).

It might be argued that no technical means are available to reproduce a naturally functioning riparian ecosystem that provides some combination of benefits listed in Table 39. Therefore, decisions which lead to the destruction of these ecosystems pose "irreversible" consequences.[2]

<u>Environmental Problems as Economic Problems.</u> Economics provides one useful approach to analyzing land and water use practices which affect the extent and vitality of riparian areas. Basically, economics is a study of choice in a world of resource scarcity. Of economics' many branches, one is devoted to the study of resource allocation among competing uses to achieve maximum social welfare.[3] Central to this field of study is the criterion of economic performance called "efficiency." Reduced to its simplest form, economic efficiency is achieved when resources are gravitating to their most valuable uses at the least possible cost to society (Freeman et al. 1973). All institutions which perform the function of allocating scarce resources can be evaluated against this criterion.

One institutional approach to resource allocation is the private market where individuals own and exchange goods and services. Here, private resource owners are guided by incentives to use resources in particular ways including the production of goods and services. Consumers, on the other hand, make individual choices regarding what combination of goods, services, and other amenities will satisfy their needs and

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[2]Although it may be possible to replace several functions of a riparian ecosystem through artificial means and on an individual basis.

[3]The use of the term "welfare" should not be confused with welfare as income redistribution.

wants. The price system, in theory, reflects the relative values of that which is being produced, consumed, or devoted to a specific purpose. In other words, prices result from the interaction of individuals making choices about what is or is not valuable.

Another method for allocating resources is a variation of the market, except that market activities are regulated by public bodies. These bodies perform any number of functions from determining prices, issuing permits and licenses, specifying land uses appropriate for certain areas, and setting standards of quality for land, air, and water. Finally, another approach is public ownership of resources where public bodies invest, manage, and dispose of resources. We find all of these institutional approaches at work in the economy, interacting to perform the allocative function. The manner in which these institutions "value" resources - either explicitly or implicitly - will affect the way resources are used.

The Performance of Institutions. Until recently, there has been a lack of good empirical work assessing: 1) the costs and benefits of particular resource uses; and 2) the performance of institutions in allocating scarce resources.[4] This is particularly true in the environmental arena where conceptual and methodological advances have been relatively recent phenomena (Fisher and Peterson 1976). Nevertheless, assessments of the costs of environmental degradation and the benefits of environmental improvement are being made with respect to topics as diverse as health, aesthetics, recreation, property value, fish and wildlife, and others.

Tracing real world accounts of environmental degradation to flaws in the institutions which allocate resources is frequently a rather subjective exercise. Usually, the analysis begins and ends by pointing an accusing finger at the industrialist who discharges waste into open water, the farmer who clears and drains wet areas for agricultural production, or the developer who mars a scenic vista with rows of condominiums.

In many cases, the high value which consumers place on specific goods and services may well account for these alterations of natural systems. For example, it is believed that demand for soybean production causes conversion of bottomland hardwood areas to agricultural production (MacDonald et al. 1979a). However, it is not unreasonable to suspect that in other cases, failures within both private and public institutions create significant incentives to ignore environmental values, and therefore cause a misallocation of resources. In other words, the polluter, land clearer, and developer may be merely reacting to the incentives with which they are faced. The end result of these failures is the over- or under- production of specific goods and services (e.g., the over production of soybeans or the under production of natural flood storage capacity). Stated another way, existing scarce resources are not being used efficiently in that they are not being put to more valued uses. In these situations, producers and consumers are not faced with or do not realize the actual costs or benefits of their activities. In addition, society is not realizing an optimal use of its resources.

Valuation Problems in the Private Sector: Market Failure

In market economies resources are allocated to a variety of uses including the production of goods and services demanded by the consuming public. The value of a resource, or a good resulting from some combination of resources and productive factors (capital, labor, etc.), is reflected in the price it can command in the market place. Ideally, a producer will be faced with all of the actual costs of production or development (so that price reflects the level of cost) and all the actual benefits (so that he has proper incentives to produce that which society demands). On the other side, consumers faced with accurate prices will react accordingly

[4] Several reports prepared by the President's Council on Environmental Quality provide readable accounts of methods and actual studies on economic assessment of environmental quality (Council on Environmental Quality 1971, 1975, 1978, 1979).

and consume that level of goods and services which satisfies their self-interest.

Since Adam Smith wrote The Wealth of Nations in 1776, it has been apparent to many that markets allocate many goods and services with admirable efficiency. If the market were to allocate environmental and/or natural services efficiently, there would be no need for concern. However, a growing body of conceptual and empirical work indicates that the market will misallocate environmental resources.

Causes of Resource Misallocation. For markets to allocate resources efficiently, economists generally agree that a few basic requirements must be satisfied. These include: (1) markets must be competitive (i.e., no monopolies); (2) there must be information about present and future prices, and about alternatives available to producers and consumers; (3) there must be no externalities or, in other words, the costs and benefits of an activity must be realized only by those participating in a market exchange or transaction; and (4) there must be mobility (transferability) of resources so that they may be moved from less valuable to more valuable activities. Of these four requirements, the third - "externalities" - presents the most persistent obstacle to the proper valuation of environmental resources (Freeman et al. 1973).

Externalities, quite simply, are the costs and/or benefits of an activity that are not or cannot be restricted to the individuals making the resource use decisions. In other words, the costs or benefits of an activity become "external" to those making resource use decisions. In these situations, the price system is not allowed to perform its critical function of placing accurate values on resources put to various uses.

> "When the meat-packing firm dumps its unused animal parts into a river, downstream swimmers and fishermen are the objects of spillover costs; when your next-door neighbor plays his phonograph loudly and it annoys you, you are the object of a spillover cost; when the coal-burning industry in a community fills the sky with coal dust smog, residents of the community are the objects of a spillover cost; when the next semitruck pulls onto the freeway with the effect of delaying your arrival and that of all other freeway motorists, you and your fellow drivers are the objects of a spillover cost. It is characteristic that in each of these cases, the person harmed bears identifiable 'costs' for which he is not compensated. Moreover, in each case, this person would be willing to pay something to avoid bearing the spillover cost" (Haveman 1970).

The literature on environmental issues is rife with examples of this problem. The use of air and water as a place for waste disposal was seldom figured into the production costs of industries and, therefore, into the price of the product produced. While competing uses of air and water may well have been more valuable, there was no way to determine that because other competing uses could not be valued through the price system.

In the riparian area, potential examples of externalities can be identified:

(1) A farmer clears, ditches, drains, and dikes his land located adjacent to a river in order to capitalize on a lucrative soybean market. Such actions by individual farmers along a watercourse frequently involve stream channelization as well.[5] These land use practices tend to direct flood waters downstream subjecting individuals located there to greater flood risk. The negative impact of these practices on fish and wildlife species has been well documented. Those who derive pleasure

[5]Such activities are often performed with the assistance of public subsidies. In these situations, it might be argued that a double subsidy is involved - the assistance from a government agency and the uncompensated use of or damage to other resources.

from these species will realize the costs of such activities (Brown 1975).

(2) Appropriators along a western stream place increasing demands on the supply of water available in the stream. Gradually, the demands become so severe that river flows are too low to support a fishery during some periods of the year. In addition, the low flows result in damage to streamside vegetation critical to wildlife species.

(3) Landowners fill in wetlands along a river to build attractive homesites. Replacement of the wetland area along a significant portion of the river results in several unintended effects. A sudden decline in the river's fishery is detected. Monitors of water quality notice an increase in sediment and pollutants in the municipal water supply requiring increased treatment costs.

These are but a few examples of what might happen when riparian areas are altered. The point is not that the uses resulting from alteration are not valuable, but that the other existing benefits derived from the natural environment <u>are not valued</u>. A look at the causes of externalities reveals the institutional basis of the problem.

<u>Property Rights, Public Goods and Transaction Costs.</u> Externalities are symptoms of more fundamental institutional failures. For markets to allocate resources efficiently, property rights must be defined, assigned, and enforced (Posner 1977). With full ownership, the owner can prevent others from using, benefiting from, or damaging the resource without making compensation (Freeman et al. 1973). However, some environmental resources are not easily appropriated as private property.

> "...many environmental resources are still unpriced and remain outside the market. Because ownership rights have not been assigned to them, or because they are not easily broken up into units that can be bought and sold, such valuable environmental assets as watercourses, the air mantle, landscape features, and even silence are 'used up' but their use is not accurately reflected in the price system" (F. Anderson et al. 1977).

If exclusion cannot be implemented through the assignment and enforcement of legal rights, no market will form to provide or maintain such services (Krutilla, 1979).

Garret Hardin's <u>The Tragedy of the Commons</u> (1968)[6] is a classic statement on the effects of open and unlimited access to a resource base. Where no rights of exclusion are enforced, the resource base is labeled "common property." In this situation, "everybody's property is nobody's property." Rational individuals exploit this common resource to benefit themselves to the collective detriment of other resource users. Common grazing lands (Hardin 1968) and open access to fisheries and clam beds (North and Miller 1978) are familiar examples. However, there has been a tendency to lump many environmental problems under the label of the "commons" dilemma. Godwin and Shepard (1979) use water pollution and timber harvesting on public lands to caution against this liberal use of "commons" analysis. Careful attention must be paid to the facts of a resource problem before putting it into the commons category.

Some environmental resources are held in the public domain. But where the public has not defined and exercised its right to exclude certain uses and users of these resources, the effect is the same as if no right of exclusion existed at all. For many years, watercourses, the atmosphere, and some public land resources were considered standard examples of this (Dales 1970). Recent attempts to define and enforce the rights of the public to specific resources through legislation represent attempts to correct this institutional void. Whether these are effective responses is a separate issue for analysis.

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[6]This article and several others which build on its central theme are contained in Hardin and Baden (1977).

Finally, some environmental resources possess characteristics which prevent the assignment of private ownership rights and efficient allocation in the marketplace. Economists refer to these as "public goods." As Bish (1971) states:

> "Public goods are goods that can be consumed by one person without diminishing the consumption of the same good by another and where exclusion of potential consumers is not feasible. For example, national defense is a service that is available to every citizen and an increase in population does not cause a decrease in services for original citizens. The qualifying clause differentiates this case from the situation where exclusion is feasible because the good can be packaged and sold on the private market...Examples of public goods include national defense services, flood control, and the legal structure...This use of the term "public" relates only to the nature of the good and has nothing to do with the nature of the producer, whether it is a public government or a private firm. The public aspect relates only to the form of consumption of the goods." (emphasis added)

Some individuals cannot be economically excluded from the benefits of a public good once it is produced. Therefore, private entities have little or no incentive to produce and market these resources, goods, or services.

In some cases, riparian ecosystems display convincing examples of public goods. A landowner who maintains his riparian lands for natural flood storage, water quality maintenance, and fish and wildlife habitat cannot sell the service to one buyer without making it available[7] to others. Potential buyers of the service cannot exclude others from benefiting as well. In essence, nonbuyers can take a "free ride" on the other buyer's investment. Given this dilemma, maintenance of these services from the natural sector is extremely difficult (Krutilla and Fisher 1975). There is little incentive for landowners to maintain their riparian lands for these purposes because they receive no return on this type of use. On the consumer side, there is little incentive for one person to invest in or buy these services from landowners because he cannot exclude others from taking a "free ride" on his purchase. It might be argued that the public should try to organize and negotiate with riparian landowners to maintain these services where it seems appropriate. However, the costs of organizing people, devising a legal agreement, and enforcing the contract (i.e., transaction costs) are sometimes so high that such activity is prohibited. Coercive arrangements created through public laws have generally been used as a substitute for this approach.

Valuation in the Public Sector: Opportunities and Problems

During this century, government has been given a substantial role in allocating society's scarce resources among competing uses. Not only does government intervene to regulate markets, but it participates in the direct provision of goods and services to society. Typically, the presence of market failure such as that discussed earlier in the chapter has been used as a justification for government intervention in market allocation. However, two points must be clarified here. First, market failure is only a necessary and not a sufficient justification for public intervention. Since government or public allocation also fails, what is needed for objective analysis is a careful comparison between market and nonmarket solutions to resource allocation problems (Wolf 1979). Second, indicating a justification for

[7]One possible exception here is the property owner who maintains his land as a game or fish preserve and then sells rights to hunt and fish. Some measure of exclusion is possible to make such a use of land profitable in a financial sense. However, once a species strays from the confines of the preserve, there is nothing except State hunting regulations to prevent the capture of the animal by an outsider.

market intervention and specifying the appropriate form of intervention (i.e., regulation, taxation, or public ownership) are two very different tasks. The latter is probably a much more difficult job. An evaluation of alternatives is reviewed by Stewart and Krier (1978).

Government is usually called to justify its resource allocation decisions in terms of benefit and costs (broadly speaking). All public agencies must perform their duties in a manner consistent with the statutory authority granted to them by legislative bodies. However, in many cases, these statutes contain broad substantive goals which have the effect of conferring considerable discretion on the agencies. Therefore, many agencies that perform regulatory and/or management functions are put in a position of deciding how to allocate resources in the "public interest." This is true of the State water board that permits new uses and transfers of water; the environmental regulatory agency that sets standards for air and water quality; or the agency that must determine the best uses of publicly owned and managed resources.

Government also affects resource allocation in ways very different from direct regulation or public resource management. Through taxes, subsidies, and other policies, government encourages or provides incentives for certain types of land and water use practices. In these situations, government implicitly makes a judgement that land and water are more valuable in some uses rather than others....if, in fact, the effect on other resource uses is evaluated at all. This is reflected in the kinds of land and water use activities that result from such policies.

This section looks briefly at these two general aspects of public resource allocation. Examples of opportunities for resource valuation by regulatory, development, and management agencies are identified. Also, examples of implicit statements by government about the value of certain resource uses through tax, subsidy, and other policies will be mentioned.

Regulations, Resource Investments, and Public Management. Opportunities for valuing resource uses arise in every situation where government makes decisions about resource allocation. Whether resource valuation is compelled by legislative mandate or whether it results from an internal agency decision to follow such a procedure is a separate issue. Also the specific approach to be used in evaluating decisions may vary among and even within public agencies.

Examples of agencies at the State level that conduct some form of resource assessment before granting permits for development of wetlands are reviewed by Kusler (1978). Similar reviews of State water allocation procedures for granting permits for new water uses and transfers can be found in Clark (1972).[8] At the Federal level, the U.S. Army Corps of Engineers conducts a "public interest review" (33 C.F.R. 320) when deciding whether to permit activities that affect waters of the United States (pursuant to Sections 9 and 10 of the Rivers and Harbors Act of 1899 and Section 404 of the Clean Water Act). This is perhaps one of the more celebrated attempts to consider all possible variables in determining the highest and best use of water and related land resources. Economic values of resource uses are not the only factors considered in the public interest review. Arguably, however, the items for consideration under the review procedures are broad enough so as not to be inconsistent with the goal of economic efficiency. As another example, the Federal Energy Regulatory Commission (FERC) is required to consider alternative uses of a waterway before deciding to grant or deny licenses for hydroelectric power projects. FERC must consider the effects of the project on commerce, water power development, recreation, and

[8] Since states in the eastern and western United States follow essentially different principles in allocating water resources, it is useful to consider these separately. A review of eastern riparian water right jurisdictions is contained in Ausness (1977). The U.S. Fish and Wildlife Service has conducted a survey of state laws, procedures, and strategies as they relate to instream flow problems in 13 western appropriation doctrine states (Enviro Control, Inc. 1978).

other beneficial uses of the waterway [16 U.S.C. 803(a)].

The U. S. Army Corps of Engineers, the USDI Bureau of Reclamation, and the USDA Soil Conservation Service invest in, and provide technical assistance for, projects which result in navigation improvements, flood control, hydropower development, irrigation, watershed development, and recreation. For many of the projects, Congress requires an analysis of project costs and benefits before considering the project for authorization. For example, the Watershed Protection and Flood Prevention Act (P.L. 83-566) requires SCS to perform a cost-benefit analysis for its small watershed programs. Similarly, the Flood Control Act of 1936 requires the Corps of Engineers to conduct an analysis for many of its public works projects.[9] In addition, the National Environmental Policy Act (42 U.S.C. 4321-4361), the Water Resources Council's Principles and Standards for Water and Related Land Resource Planning (developed in accordance with the Water Resources Planning Act, 42 U.S.C. 1962-1962d-s), and the Fish and Wildlife Coordination Act (16 U.S.C. 661-667e) have established broad requirements for the evaluation of the pros and cons of project development including impacts on the environment.

For any major Federal action significantly affecting the quality of the human environment, section 102(2)(c) of the National Environmental Policy Act (NEPA) requires an environmental impact statement (EIS). NEPA does not require that a cost-benefit analysis be contained in every EIS. However, where cost-benefit analysis is being conducted for project justification, NEPA regulations require that the analysis be incorporated into the EIS. In addition, the statement shall discuss the relationship between the cost-benefit analysis and any unquantified environmental impacts, values, and amenities (43 C.F.R. 1502.23).

Amendments to the Fish and Wildlife Coordination Act (FWCA) in 1958 brought about substantive and procedural changes to Federal and Federally-permitted or licensed water resource project planning. First, the development of fish and wildlife benefits are now to be considered a co-equal purpose of water projects along with the more traditional purposes such as flood control, hydropower generation, and water supply. To achieve this goal, several procedural steps are now built into the planning process including: (1) mandatory consultation by development agencies with State and Federal wildlife agencies; (2) full consideration by development agencies of the wildlife agencies' project recommendations stemming from consultation; and (3) authority for development agencies to implement the recommendations of wildlife agencies concerning fish and wildlife protection, enhancement, and mitigation as they find acceptable (Stutzman 1980).

A persistent problem under the FWCA planning process concerns the basis for justifying fish and wildlife enhancement and mitigation measures of water projects. Wildlife agencies assert that valuation procedures utilized by development agencies consistently result in low and inappropriate enhancement and mitigation measures (Comptroller General 1974). The debate has focused on the use of traditional valuation methods such as "recreation-use days" to evaluate wildlife losses and benefits associated with water projects. The development of Habitat Evaluation Procedures (HEP) by the U.S. Fish and Wildlife Service is the first major attempt to address this methodological problem (U.S. Fish and Wildlife Service 1980c). However, instead of confronting the problem primarily from an economic perspective, HEP attempts to employ a biological basis for habitat evaluation.

Perhaps the most comprehensive analysis for water development projects is that conducted pursuant to the Principles and Standards (P & S) developed by the U.S. Water Resources Council. The P & S establish the basic process to be followed by the Federal agencies when planning activities and projects affecting water and related land resources (45 Fed. Reg. 64366: Sept. 29, 1980). Two

[9]A straightforward discussion tracing the development of economic analysis in public project evaluation is provided by Krutilla (1975).

primary objectives are targeted by P & S: National Economic Development (NED) outputs and Environmental Quality (EQ) impacts of water projects. Separate manuals have been developed for the NED and EQ analysis of water projects. The NED manual follows traditional cost-benefit analysis procedures for measuring the contribution of water projects to social welfare. These methods have been criticized by economists on selected grounds...some of which concern the analysis of costs which result from project development (Duffield et al. 1979).

The separation of NED and EQ considerations emphasizes the methodological problems of incorporating environmental considerations into traditional cost-benefit analysis. Shabman (1979) analyzes this issue as it relates to the general problem of mitigation. Methodological problems which bias cost-benefit analysis against environmental considerations are discussed later in this chapter.

Public land managers are faced with similar opportunities for resource valuation in their land and water management decisions. Legislation concerning wilderness preservation (Bigelow 1979), national forest land management (Krutilla and Haigh 1978) and management of other public lands (Hagenstein 1979) all encourage, if not require, resource valuation as a prerequisite for land and water management. In addition to the requirements of the public land management statutes such as the Forest and Rangeland Renewable Resources Planning Act of 1974 as amended by the National Forest Management Act of 1976 (16 U.S.C. 1600-1676), the Multiple-Use Sustained-Yield Act of 1960 (16 U.S.C. 528-531), the Federal Land Policy and Management Act of 1976 (43 U.S.C. 1701-1782, as amended) and the Wilderness Act of 1964 (16 U.S.C. 1131-1136), NEPA and P & S may require resource valuation as a prerequisite to specific land management decisions.

Problems with "Implicit" Valuation. Several studies have suggested that government programs involving taxes and subsidies encourage some uses of land and water over others. There is nothing unique about tax and subsidy programs, per se, as a form of government activity. They have been used to achieve environmental and developmental goals (F. Anderson et al. 1977). However, careful evaluation of the side-effects on land and water use is important before any program involving taxes and subsidies is adopted.

Analysis of wetland drainage programs in the upper midwest by Leitch and Danielson (1979) and Goldstein (1971) have identified a relationship between these land use practices and government subsidies to agriculture. In a similar vein, Shabman (1980) suggests a correlation between subsidies to agriculture in the form of price supports and insurance and the clearing of bottomland hardwood areas in the southeastern U.S.

The National Flood Insurance Program (NFIP), has been criticized as a major contributor to floodplain development resulting in the alteration of riparian ecosystems and increased flood hazard (Plater 1974). Through attractive insurance premiums, the NFIP subsidizes the cost of risk associated with locating in a floodplain. Although the goal of flood insurance is a noble one - to spare flood victims any economic disaster - it may tend to encourage floodplain development. The Federal Emergency Management Agency now requires communities to adopt floodplain management programs as a prerequisite for membership in the NFIP. The success of this program in encouraging wise use of floodplains is yet to be evaluated.

Tripp (1977) asserts that Corps of Engineers flood control programs and SCS drainage programs amount to a subsidy that encourages increased agricultural activities near rivers and streams. The result is the alteration of the riparian zone and, in some cases, increased water pollution and fewer benefits from fish and wildlife resources. Brown (1975) conducted an economic analysis of government subsidies for stream channelization projects and concluded that federal subsidies for such projects should be terminated. Because the benefits of channelization projects are generally very localized, such projects should be financed at that level. Only where the effects of these projects extend beyond local jurisdictions should federal intervention be considered. An analysis of cost-sharing provisions for Soil Conservation Service projects has been performed by the Comptroller General (Nov. 13, 1980).

These are but a few examples of subsidy programs which affect riparian areas. Needless to say, subsidies exist through other government programs which encourage preservation of natural areas. However, the total dollar amount expended for such purposes appears to be relatively small. Nevertheless, this situation exemplifies the tensions which exist between government programs which appear to promote opposing uses of land and water (Comptroller General Feb. 8, 1979).

Other Variables Affecting Public Evaluation. There is no question that resource valuation, even when conducted with skill and objectivity, provides only one set of information for agencies and legislative bodies to consider before making decisions about resource allocation. Certainly, water resource projects with unattractive cost-benefit ratios have been approved and implemented. This should not be surprising. Once the function of resource allocation shifts from the private to the public sector, a whole new set of variables may affect the outcome. An influential politician, vocal constituents, effective lobby groups may all pave the way for approval of questionable projects. Indeed, it may be very rational conduct for a local concern to demand a project providing very limited and localized benefits be subsidized by Federal funds. (Comptroller General Nov. 13, 1980).

Even where cost-benefit analysis plays a significant role in decisionmaking, methodological flaws could reduce its overall value as a decisionmaking tool. Problems associated with the use of cost-benefit analysis in Federal decisionmaking have been reviewed by the Comptroller General (June 2, 1978 and August 7, 1978). As Haveman (1972) pointed out in his study of navigation improvement, hydroelectric and flood control projects, preproject estimation of benefits by development agencies at times bare little resemblance to the actual accounting of benefits once the projects were in place. The same could be said for cost estimates. Whether discrepancies were the result of faulty methodology, uncertain information or inappropriate application, significant questions about the careful use of such analyses must be addressed.

A METHODOLOGICAL PERSPECTIVE ON VALUATION

The Basis for Resource Valuation

Resource valuation becomes an important exercise at both a theoretical and practical level. In theory, where markets fail to produce efficient results, resources are not being valued at appropriate levels. The implication of this is that costs and benefits of resource use are not properly reflected and, therefore, are not providing the appropriate incentives to resource users. The theoretical purpose of non-market allocation (or government intervention) is to allocate the resource in question as if an efficient market were allocating it.

The practical implications of the theoretical ambitions of non-market allocation are numerous. One central task of public resource allocators (regulators, permit grantors, lease grantors, public land managers) is to assess the costs and benefits of various competing resource uses. To perform this analysis, the value of resources in different uses must be identified. Where market prices do not exist for reasons discussed earlier, the necessary dollar values for such analysis are not available. There are several methods for obtaining surrogate values for resources so that an attempt at some form of cost-benefit analysis can be made. However, the methodological problems are significant and may, in many circumstances, impede evaluation of all those potential costs and benefits of resource use. In addition, such analyses may be very expensive to perform. Still other methods do not rely on economic analysis at all. The development of an Environmental Quality Account for the Principles and Standards recognizes implicitly that economic methods of valuation do not exist for some values. This necessitates the use of other methodologies.

Ecological Values and Their Assessment

Because water and other materials from the landscape converge in riparian zones, a given area of riverine ecosystem tends to support a greater production of natural goods and services than an equivalent area of upland in the same geographic region. Many ecological functions in riparian ecosystems, such as primary productivity and nutrient cycling, are accelerated because of greater fertility and the higher availability of water than in adjacent upland areas. In addition, riparian ecosystems have a profound influence on the condition of aquatic ecosystems to which they export material and energy. These characteristics have been covered in detail in Chapter 3. However, there is no reason why the assessment of ecological values should differ between upland or riparian ecosystems. Indeed, approaches should be consistent and applicable to all natural resources. Once an acceptable approach to valuation is established, its application to specific sites should be responsive to differing values that exist among natural systems.

Rather than attempting to assign monetary values to specific riparian ecosystems, this section will review approaches to valuation of goods, services, and amenities which result from ecological processes. Ideally, valuation of natural and altered ecosystems should be as comprehensive as possible. One of the problems is our ability to measure and quantify natural functions and to assess the extent to which society values the life support services and other benefits that ecosystems provide. The existence of these functions and the benefits which society derives from them may not be perceived until the goods and services are no longer being supplied and the functions must be replaced by technological substitutes. Even if current human use of the resource is not being realized, the value of preserving options for future generations could be taken into consideration (Krutilla 1967, Page 1977).

It is generally accepted that a number of valuable goods, services, and amenities are attributable to riparian areas. Not all riparian ecosystems generate all of these outputs nor do all provide the same quantity of value for each category. For that matter, it is difficult to say, given our present state of knowledge on the subject, whether all natural functions of riparian areas and their resulting benefits are recognized. For example, in considering ecosystems from a global perspective, there is no question that they function to control levels of atmospheric gases, are essential to the circulation of water, and regulate the movement of nutrients. Similar functions in life support have been demonstrated on smaller scales. For example, Turner (1977) has shown that yields of offshore shrimp fisheries correlate with the amount of intertidal coastal marsh area. While the benefit of some of these functions appear immediate and direct, others may seem remote or marginally applicable to human well being and survival. They are, nevertheless, present and part of the life support system.

Identifying and Organizing Information for the Valuation Process

As Krutilla and Fisher (1975) state:

> "In the extractive industries - forestry, agriculture, minerals - there are specialized branches of economics that can provide professionally competent estimates of the economic value of services provided by that extractive output. The present value of service flows will give the resource value of a tract of land when it is used for commercial extractive activities. The costs of the extractive activities today, of course, include the opportunity returns lost in transforming the tract of wildland (and/or reach of stream) into the developmental alternative. And what is the opportunity cost of this land transformed from its natural state? It is the value of the service flows that

the public would derive from the land in its natural state."

Therefore, we begin with the premise that natural environments may function in ways that bring benefits to humans. A necessary precedent to the analysis Krutilla and Fisher beg is information explaining the manner in which land and water use practices can affect the flow of goods and services to humans; and the effect of such changes on social welfare. This will help determine what uses of an area are optimal from society's point of view.

Drawing from Freeman (1979), three sets of information are required to make these evaluations: (1) information about ecological processes and functions which ultimately result in benefits to humans; (2) specific information about the nature of these goods and services and the manner in which changes in land and water use practices will lead to changes in the flow of environmental goods and services; and (3) how changes in levels of environmental services lead to changes in economic welfare. The first set is almost entirely derived from the biological and physical sciences. The third set is largely within the realm of economics. The second set represents the interface between social and natural sciences.

Table 40 attempts to summarize information about riparian areas that would be needed to begin the analysis mentioned above. Category I refers to the natural processes at work in riparian areas. Category II identifies specific goods, services, or amenities that would be provided for human consumption. The final category (III) indicates the type of data that would be necessary to assess changes in the level of human welfare (value) caused by changes in levels of the consumable. As will be discussed later in the section, two general types of information can be used to perform economic valuation assessments of natural goods and services. These include market data (prices of related and/or substitute goods and services) and nonmarket data (information from surveys, questionnaires, interviews, voting, etc.). Category III identifies which type of information has been most commonly used in valuing natural goods and services.[10]

Many of the functions listed in Table 40 can occur simultaneously in natural and partially altered riparian ecosystems. However, some may conflict with each other. For example normal agricultural practices require forest removal and flood protection activities which preclude perpetuation of many of the values listed in other categories. As illustrated in Figure 24, activities that alter the geomorphic and hydrologic characteristics of riparian ecosystems are most likely to have a lasting and irreversible effect on the natural services provided by riparian ecosystems. The ultimate purpose of developing studies based on the information in Table 40 is to trace the relationship between individual land/water use activities and the flow of goods and services from the natural environment that contribute to human welfare. An example might ask: (1) how do riparian functions result in natural flood storage; (2) how do alterations in the riparian zone affect the provision of this natural service; and (3) how can we measure changes in economic welfare that result from such changes? It seems as though most of our efforts in recent years have been focused on the first question. Approaches to answering the remaining two questions, including valuation techniques (question 3), are in early stages of development.

Economists and ecologists have been largely responsible for developing approaches to resource valuation. In this review of methods, three categories are considered: (1) qualitative and other statements of value; (2) economic approaches; and (3) life support or energetic approaches. In reviewing these methods, several factors will be briefly considered including: the conceptual

[10] Thomas et al. (1979d) have developed a model for evaluating the interactions of these categories of information. For a more detailed discussion of information that is needed to perform an economic evaluation of natural areas, the reader should consult this study.

TABLE 40. Information requirements for the economic assessment of riparian ecosystem values.

	Natural element or energy source	I. ECOLOGICAL Natural function	II. ECOLOGICAL/ECONOMIC INTERFACE Goods, services, or amenities produced	III. ECONOMIC[a] Type of data utilized to measure value Market	 Non-market
1.	Primary productivity of plants	Accumulation of biomass	Timber products: Source of high quality lumber and veneer.	+[b]	-
		Contribution to atmospheric CO_2 balance	Air quality	+	+
		Structure for wildlife habitat and basis for food webs	Wildlife (Important for food, recreation, and existence values - see #2 below)	+	+
2.	Secondary productivity of animals	Fish production	Commercial fisheries (food) Sport fisheries (food and recreation)	+[c] +[d]	- +[f]
		Wildlife production	Furbearers, waterfowl, other game (food, pelts, and recreation)	+[e]	+[f]
		Basis for other food webs	Support for fish and wildlife diversity and other food (See above)	+	+
3.	Hydrologic cycle	Area of convergence for upland runoff	Groundwater supply for municipal, agricultural and industrial consumption (particularly important in arid riparian zones)	+[g]	+
		Groundwater supply supports forests in desert and prairie regions	All forest uses in arid region (timber, fish, wildlife, recreation, existence values)	+[h]	+[h]
		Presence of flowing surface water	Disposal and dispersion of sewage effluent and other waste	+	-
		Surface water (flooding)	Most goods, services, and amenities of #1 and #2	+	+
4.	Geomorphic work of streamflow	Maintains landscape and floodplain topographic diversity	Recreation and existence values as a result of natural features	+[i]	+[j]
		Maintains channel structure	Navigation and drainage	+[k]	-
		Maintenance of ecotones and transition zones	Animal and plant diversity (particularly significant for endangered species)	-	+
5.	Floodwater storage	Ameliorates downstream flood peaks during excessive runoff events	Protection from flood damage	+[l]	+
6.	Nutrient cycling	Accumulation of nutrients derived from upland runoff and erosion	Contributor to primary productivity (See #1)	See #1	
			Water quality (protected in ecosystems downstream from riparian ecosystems through nutrient accumulation, denitrification, and other transformations)	+[m]	-
7.	Soil/sediment processes	Sediment deposition in floodplain maintains rich soils	High primary productivity; rich agricultural soils for food and fiber (see #1)	+	-
		Sediment deposition in floodplain maintains low suspended solids in streamwater	Water quality (reduces treatment requirements)	+[m]	-
		Metabolic transformation of organic compounds	Water quality (transforms organic toxins such as pesticides to nontoxic products)	+	-
		Accumulation of heavy metals in anaerobic soils	Water quality	+	-

[a]Several bibliographies and other summaries of research on this subject are available. The reader should consult the following for citations and discussions of specific attempts to value natural goods, services, and amenities: Dwyer et al. 1977, Leitch and Scott 1977, Clawson 1977, and Thomas et al. 1979. Citations are provided for a few examples of studies relevant to this category. For additional citations, consult the research summaries and bibliographies referenced in this chapter. [b]Clawson 1977a, Clawson 1977b; [c]Batie and Wilson 1978, Gillick and Scott 1975; [d]Brown et al. 1964; [e]Goldstein 1971, Raphael and Jaworski 1979; [f]Hammack and Brown 1974, Knetsch and Davis 1977; [g]Gupta and Foster 1975; [h]Clawson 1977a, Clawson 1977b; [i]Krutilla and Fisher 1975; [j]Krutilla and Fisher 1975, Brookshire et al. 1976; [k]Martin and Casavant 1980; [l]U.S. Army Corps of Engineers 1971, 1976; [m]Delorme and Wood 1974.

basis of the analysis, attempts to apply the analysis, and special problems that seem to limit the utility or practical application of the methods.

It must be mentioned that all three types of approaches to valuation are not necessarily relevant to the analytical process represented in Table 40. Economic methods, of course, would be used to perform the analysis in Category III of Table 40. Where economic data is unavailable, costly to acquire, or if a suitable and reliable methodology has not yet been developed, then another type of valuation approach may be required (such as a qualitative approach). Life support or energetic approaches generally rest on the assertion that qualitative and economic approaches do not accurately capture the true significance of ecological systems.

Approaches to Valuation (I): Qualitative and Other Statements of Value

Table 39 contains qualitative statements of a variety of values attributed to riparian ecosystems. Not all riparian ecosystems have all these values nor do all of them provide the same "quantity of value" for each category. The relative importance of each entry cannot be determined from qualitative statements such as these. In addition, values are not necessarily additive. The "total value" of a riparian ecosystem is not always equal to the total number of entries from the table. Furthermore, many of the values listed in Table 39 conflict with each other and thus illustrate some of the shortcomings of general qualitative statements of value. Impetus to include these types of values for consideration in water development projects has been provided by the Environmental Quality procedures of Principles and Standards for Planning Water and Related Land Resources developed by the U.S. Water Resources Council. The procedures encourage the assessment of values that are not readily amenable to economic assessment, especially resources that contribute to overall social well-being and quality of human life. While a trained ecologist can evaluate the merits of a qualitative statement, most people are used to a scale of value based on dollar values and, thus, cannot appreciate qualitative ecological statements. This system of evaluation, although useful, lacks universality and is susceptible to biased judgment and incomplete analysis.

Methods have been developed which seem to make effective use of this approach in limited situations, particularly field studies and on-site investigations. For the most part, these approaches rely on ranking systems based on qualitative valuations of resource values within a particular area.

Wetland Ranking Methods. Approaches developed by Reppert et al. (1979) and Larson (1976) for evaluating wetland resources begin with a comprehensive survey of functions, characteristics, and values associated with wetland areas. These values are generally the same as have been mentioned throughout this document (see Table 39). The approaches then detail suggested procedures for documenting necessary information, compare the resource values with other wetlands in a given area, and rank the overall value of the wetland (i.e., usually by employing a simple numerical ranking or a descriptive "high", "moderate", or "low" value approach).

These approaches are attractive for several reasons. They are fairly straightforward and can be implemented by individuals who are not necessarily "experts" in the field. The method can be relatively cheap to apply. Finally, it does provide a method of comparing wetland areas within a region.

There are several disadvantages, however. The technical sophistication is at such a level that truly "difficult" scientific questions regarding resource values or ecosystem functions cannot be addressed. In perhaps a more significant light, it does not address the question of what is the value of the resource to society. Having identified a wetland with levels "x, y, and z" flood storage, wildlife habitat, and water purification characteristics will not help decisionmakers compare the use of wetlands for these purposes with developmental or other uses. This does not mean that the approaches are not useful for what they were designed. It simply means that they can provide answers for only a limited number of decisionmaking problems.[11]

Approaches to Valuation (II):
Economic Methods

Frequently, public officials have the responsibility of making decisions that will affect the allocation of resources. This can come about when a decision must be made to grant a permit or a license for private development. In the case of public land management, the responsibility may be even greater since many land and water decisions can be made only by those responsible for management. Finally, agencies like the Corps of Engineers or the Bureau of Reclamation participate in and implement decisions which result in the investment of considerable funds and resources for flood control, navigation, hydropower, and irrigation.

As discussed earlier, few major actions affecting the environment, water and related land resources, and fish and wildlife resources escape the planning and analysis requirements of NEPA, Principles and Standards, and the Fish and Wildlife Coordination Act. In many if not most cases, some kind of cost-benefit (c-b) analysis must be made to justify the efficacy of the proposed project. Generally speaking, c-b analysis might refer to virtually any analytical method that organizes information on alternative causes of action and displays the trade-offs associated with those actions (Conservation Foundation 1980). Rosen (1977) makes the following distinctions among these analyses:

1. Cost-benefit analysis - any quantitative or qualitative analysis which seeks to compare costs and benefits of various alternative projects;

2. Quantitative cost-benefit analysis - cost-benefit analysis in which all costs and benefits are measured quantitatively, although not by a single parameter such as dollars. For example, costs can be measured in dollars or acres of farmland while benefits can be measured in lives saved or property damage avoided.

3. Monetized cost-benefit - quantitative cost-benefit analysis in which all costs are ultimately measured in a single unit, such as dollars. Monetized cost-benefit analysis, then, is analysis in which all criteria have been quantified and converted to a common unit of measure by some scheme of valuation.

While few people have problems with using a level of analysis represented in category 1, opinion diverges sharply when categories 2 and 3 are proposed for application. At these levels, the practical methodological complexities of c-b analysis become apparent.

Economic methods must be used to provide the resource values necessary for performing c-b analysis. The goal or purpose of this analysis is defined by Stroup et al. (1976):

> "In order to assure that society is receiving the greatest attainable value from the increasingly limited resources, it has become common practice to compare the net benefits which society will receive from opposing uses, where net benefits are defined as total benefits minus total costs. The use which results in the greatest net benefit to society is the use to which that resource should ideally be put. If this were done with all resources, people's total benefits would be maximized."

The goal of resource allocation guided by c-b analysis is a familiar one: economic efficiency. By determining the

[11]The Habitat Evaluation Procedures (HEP), have been designed by the U.S. Fish and Wildlife Service to assign "habitat values" for specific geographical areas and wildlife species. This approach is somewhat, although not completely, similar to these methods in terms of the questions the method is designed to address. The approach allows one to assign a habitat value for a specific area. It does not compare the value of the area as habitat with the values of other uses. HEP procedures do call for economic values for relevant fish or wildlife species in the evaluation study.

relative values of resources in different uses, we can, in theory, determine the highest and best use of those resources. While the marketplace values many resources in terms of prices, this same measure is quite often not available for goods not allocated by the market. These are the "non-market goods" discussed earlier. Goods which the market cannot allocate efficiently because of problems with property right assignment or public good characteristics, do not carry prices reflecting their true value to society. Therefore, to perform c-b analysis, methods must be used to find substitute or surrogate values for these resources. Some assert that, conceptually, there is nothing in the economic basis for c-b analysis that prevents consideration of any resource, good, service, or amenity that society might value. The real problem is finding practical methods to meaningfully measure and incorporate these values into the analysis (Freeman 1979).

Assumptions, Methods, and Limitations. This section will not provide a detailed review of c-b methods and procedures. For that purpose, the reader should consult any number of useful references including Mishan (1976) and Peskin and Seskin (1975). Traditional approaches for evaluating river basin development projects are contained in Krutilla and Eckstein (1958). Cogent accounts of the application of c-b analysis to environmental situations are provided by Ackerman et al. (1974) and Krutilla and Fisher (1975).

Rodgers (1980) identifies some major assumptions under which c-b analysis proceeds. First, c-b analysis assumes that all interests can be adequately expressed in dollars. Regardless of the difficulty of this task, it is important to use a common denominator to avoid the problem of comparing "apples and oranges." Second, the value of any commodity to an individual is accurately reflected in his willingness to pay for it. The central importance of "willingness to pay" is that it is an actual expression of an individual's weighted preference among given alternatives. Finally, the aggregate willingness to pay of many individuals can be measured or inferred from market prices. To arrive at a "social" determination of value, one must aggregate the preferences or dollar votes of its citizens. The "price," in theory, reflects the final outcome of this vote.

Baram (1980), Peskin and Seskin (1975), Jaffe (1980) and Rodgers (1980) have aptly described some of the alleged theoretical and practical limitations of c-b analysis. This is not to say, however, that these authors concur on all the criticisms. These include:

1. C-B analysis assumes efficiency is the only social goal worthy of analysis. There are other goals with which society might be concerned such as a more equal distribution of wealth.

2. An inordinate amount of information is necessary to assess all the impacts and results of resource utilization. Reducing the analysis to a single value obscures the complexity and uncertainty of the decision being analyzed particularly with respect to costs that might accumulate with those of future projects or those that become apparent only in the distant future.

3. Some costs and benefits cannot be measured. Intangibles, human life,[12] and non-market goods do not carry market prices and are inherently difficult to value. In these areas moral and ethical considerations may be far more important.

4. C-B analysis cannot account for the future effects of decisions even with the use of discount rates for cost-benefit measurement. This fact biases decisions against future generations (Page 1977).

5. Because the c-b analysis is filled with uncertainties, imprecision, and opportunities for misleading conclusions as well as manipulation, those

[12]Although it must be noted that courts of law have been attempting to deal with this problem on a practical level for many years. The law allows and juries frequently award monetary damages for loss of life and limb (Dobbs 1973).

conducting the analysis can tailor it to their own self-interested purposes. In short, the analysis can be abused.

After citing its numerous shortcomings, Rodgers (1980) attempts to put the use of c-b analysis in perspective.

> "Despite its conceptual and practical frailties, cost-benefit analysis begins to look better when compared to the obvious alternatives. Uninformed intuition undoubtedly plays a major role in administrative decisionmaking today. In particular, legislative-type judgements by the agencies are classic intuitive balancing acts."

While advances in the tools and techniques of formal agency decisionmaking have been significant over the last several decades, commentators continue to use increased consistency, refinement and philosophical awareness in methods such as c-b analysis.

<u>Valuation Techniques</u>. To evaluate land and water use practices in the riparian zone through c-b analysis, methods are required to value the services of these areas. If the analysis is to be complete, all aspects must be evaluated especially those which normally command no price in the market place.

There are essentially two general approaches to obtaining estimates of values for natural goods and services: the use of "market techniques" which analyze the relationship between marketed goods and services and those that are not marketed, and the use of non-market techniques. The first approach attempts to draw inferences about the value of natural environments from their relationship to marketed (priced) goods and services. The second approach is called a "non-market" mechanism because actual prices and buying behavior are not used as such. Rather, individuals are asked to reveal their preferences through questionnaires, voting, interviews, and other means. These general approaches are surveyed and critiqued by Freeman (1979).

In addition to Dwyer et al. (1977), Thomas et al. (1979) provides a thorough review of existing quantitative economic techniques for valuing goods and services of natural environments including wetland areas and other valuable habitats. A few of these techniques which fall under the general categories mentioned above are reviewed here. This discussion relies heavily on their review.

1. Market techniques. This involves observation of the market prices of relevant goods and services. Since natural areas are generally not valued on the basis of all the goods and services they provide (i.e., due to market failure), this method has little direct relevance. However, there are variations which have proven useful in certain circumstances.

One such method is the <u>cost of least-cost alternative</u>. In the event a land/water use decision would result in the loss of some good and service providing real benefits to people, this approach uses market data to estimate the cost of substituting the same goods and services by the least-cost alternative (Thomas et al. 1979). Ecological control of water quality is an example of services that economists have traditionally considered to be "free" or a service for which the market fails to command a price. Yet, when the wetland disappears, the "free" service vanishes, and people may choose to use energy and technology to perform the same work.

The value of the lost service may be evaluated by calculating the cost to society to do the same work using technology. The value of the service rises or falls in proportion to the intensity of human activity and the cost of the technology. The ecosystem is normally evaluated on the basis of only one service which is considered important to society.

Examples of these calculations are given by Gosselink et al. (1974) for waste treatment and by the U.S. Army Corps of Engineers (1976) for flood control. The calculations have major shortcomings, while they quite possibly underestimate the total value of the ecosystem. However, they do point out the high cost of substituting human technologies for ecosystem services (Comptroller General Feb. 8, 1979). Odum (1978), for example, compared the energy costs for waste treatment by tertiary sewage treatment plants and by certain wetlands in Florida. He found that treatment using wetlands used 25 times less fossil fuel energy than treatment using the tertiary plant. Natural ecosystems are obviously much more efficient than human technologies in accomplishing tasks for which they are adapted. The real economic value of all vital services (Table 39) performed by riparian ecosystems for which human technologies cannot feasibly compete may be astronomical.

Other market techniques include those which calculate dollar values of habitat services and yields. Examples of yields that have a dollar value are fisheries, aqua-cultural production, timber, organic fuels such as peats, yields from hunting or trapping activities, and so on. Value, according to this method, is equal to the price that such products command in the market place. There are several variations on this theme including: net market value (direct), total value of output, total expenditure, unit cost or average cost, net economic rent. Most of these have been soundly criticized on a conceptual basis (Thomas et al. 1979).

Statistical inferential methods attempt to use market data about the demand for goods and services related to the non-market good or service being studied. Once the data are obtained, inferences about the value of the nonmarket good can be made. The travel cost method is a basic form of this approach. In order to acquire the value of a nonpriced good or service, the cost of travel to obtain the good or service is first calculated. Non-market methods such as contingent valuation (discussed later) and the travel cost method have been combined to value recreation (Stroup et al. 1976) and waterfowl habitat (Krutilla and Fisher 1975).

Finally, expensive but potentially effective tools for representing the economic value of goods and services of natural areas are hedonic methods. Where individuals have opportunities to choose a level of natural services or environmental quality through their consumption of a private good, valuation of the natural service may be possible. As Freeman (1979) explains:

> "...if air quality varies across space in an urban area, individuals may choose their exposure to air pollution through their residential location decisions. Residential housing prices may include premiums and discounts for locations in clean or dirty areas. It may be possible to estimate the demand for public goods such as clean air from the price differentials revealed in private markets."

Models are established to analyze individual decisions pertaining to the allocation of time and money to the relevant private good and service. From here the researcher attempts to deduce the value of nonmarketable goods or services from the available market data.

2. Nonmarket Data Techniques. Another way to obtain surrogate or substitute values for nonmarket goods and services is through means such as surveys, interviews, questionnaires, and voting. These approaches must be used when market prices do not exist for the relevant good or service, or when values cannot be deduced from related market goods and services. The central problem with this general category of techniques is to induce people to reveal clearly and directly the value they place on the good or service in question.

One method, referred to as contingent valuation, attempts to collect individual responses to a hypothetical situation. Personal interviews, mail surveys, and experimental testing procedures are employed to record the preferences of individuals with respect to the hypothetical allocation of goods, services, and/or amenities. A general level consensus with respect to the overall willingness to pay for the service is reached by taking an average value. This serves then as an aggregate social value for the good, service, or amenity in question. There are numerous methodological obstacles to obtaining sound information from such approaches. The absence of a "real" situation to which to react may not provide a clear incentive for a serious and well reasoned response. Hypothetical questions regarding a person's willingness to pay for a service may result in little more than the respondents "pulling numbers off the tops of their heads."

Other techniques using this general concept are employed. In some cases, individuals are asked to reveal prefered quantities of a given good, service, or amenity rather than their willingness to pay. Voting or referenda are also used to reveal individual preferences. The considerable methodological problems with converting this information to dollar values is discussed by Freeman (1979) and Thomas et al. (1979).

Selected Studies of Natural Values. A number of studies have attempted to implement the several methods discussed to obtain values of nonmarket goods, services, and amenities. The studies selected for review here are particularly relevant to riparian and wetland areas.[13]

Gosselink et al. (1974) compared two approaches to calculating the value of tidal marshes along the southeastern coastline of the United States. In one instance, they identified and separated individual products, uses and functions that have value to man (fishery production, agriculture, waste treatment). Dollar values were placed on these components to arrive at a value for the marsh area. The second approach called the "life support value approach," attempted to place a dollar value on marsh areas by multiplying the calories of energy resulting from primary production of an acre of marsh by a dollar value per calorie. The authors assert that this represents the true value of the work performed by the ecosystem.

The study has been criticized by ecologists and economists. Fundamental shortcomings of the approach from an ecological perspective have been pointed out by Odum (1978). He asserts that the value of the yield is based upon an artificial, marketplace evaluation which does not represent the true value in terms of the ecosystem which produces the yield. Depending on the conditions of the market, the values of fish and recreation may fluctuate widely even though the natural cost to the ecosystem to produce that yield does not change. In addition, market value fluctuations can lead to ecosystem damage if the actual net yield is increased above the sustainable yield because of increased market demand. Another shortcoming of the approach is that the natural ecosystem is often evaluated solely on the basis of one or two functions that are readily identifiable as useful to human beings rather than on the multiplicity of functions which are performed.

Shabman and Batie (1978) attack the economic value estimates as neither conceptually nor empirically correct. Their fundamental criticism focuses on the attempted use of energy as a determinant of economic value. Prices are not solely and ultimately determined by energy but rather by the normal market forces of supply and demand. Papers by Odum (1979) and Shabman and Batie (1979) contribute to the extensive debate on these issues.

In their study of Michigan coastal wetlands, Raphael and Jaworski (1979) made estimates of the economic value of associated fish, wildlife, and recreation. Having identified the relationship between coastal wetlands and dependent goods/services, they extrapolated the gross annual economic return per acre directly attributed to wetland uses. On the basis of selected commercial products and recreational use (sport fishing, nonconsumptive recreation, waterfowl hunting, trapping of furbearers, and commercial fishing), the authors concluded that wetlands produced a gross annual return of $198.25 per hectare.

Delorme and Wood (1974) considered the benefits of navigation improvement

[13]As general references, the reader should consult the extensive bibliographies developed by Leitch and Scott (1977) and Thomas et al. (1979).

(dredging, channelization, locks and dams) on the Savannah River and the opportunities foregone for use of the adjacent river swamp areas for natural waste treatment. Based on preliminary estimates of the waste treatment effectiveness of one acre of river swamp, the authors concluded:

> "...the [4050 hectares] of river swamp...could effectively treat a BOD loading of about 1.96 kg/day/ha while maintaining river quality at a level necessary for recreational use. Based upon the cost of artificial tertiary treatment, this natural system's alternative worth is approximately $12.5 million per year."

No formal comparison with navigation benefits were made. Values of the waste treatment service of the estuarine wetland areas were also calculated.

Gupta and Foster (1975) developed and applied economic criteria for freshwater wetland policy in Massachusetts. The criteria were developed for use by wetland boards to aid decisions regarding the permitting of development activities in wetlands. The technique involves a comparison between the social opportunity cost of a permit denial, as revealed by market prices, and the social value of four groups of wetland benefits including wildlife production, visual-cultural benefits (i.e., recreational, education, and aesthetic benefits), water supply, and flood control potential. Wildlife values were calculated on the basis of the costs of wetland acquisition by the Massachusetts Division of Fisheries and Wildlife. These data were juxtaposed with a ranking system for determining the wildlife productivity of various wetland areas. Visual cultural values were also calculated on the basis of the acquisition costs for wetland purchases by town conservation commissions (wetlands were purchased for open space). Municipal water supply benefits from preserved wetlands compared the cost of wetland water with that of an alternative water source. Finally, flood control benefits were derived from an Army Corps of Engineers study of the Charles River Basin in Boston. The study recommended preservation of 3410 hectares of natural storage areas (wetlands) and estimated that the flood control benefits (avoided losses) would be $647,000 per year (U.S. Army Corps of Engineers 1971). After calculation of the trade-off prices for various uses of wetlands, the authors concluded that "90% of the wetlands in the State should be preserved."

A somewhat similar study was conducted by Shabman et al. (1979) of wetland preservation in the State of Virginia. Not being able to find certain evidence of the relationship between wetland areas and the benefits typically associated with them, the authors argue that wetland "boards may now be making preservation oriented decisions without an understanding of the opportunity costs of foregone development." The conclusions were based on studies of residential home developments in Virginia Beach wetland areas and recreational home subdivision development in Accomack County.

A number of studies have assessed the value of prairie wetlands (potholes) as nesting areas for migratory waterfowl.[14] The standard resource allocation issue addressed in these studies concerned the value of prairie wetlands in their natural state versus their value as drained lands put into agricultural production (Leitch and Danielson 1979). Hammack and Brown (1974) employed an interview method while Goldstein (1971) employed the travel cost technique to derive values for waterfowl. These studies are important for their approach to comparing the values of natural and agricultural uses of wetlands.

A rather unique study by Brown (1976) evaluated the impact of the wetland easement program on agricultural land values in North and South Dakota. The purpose of the easement program is to prevent farmers from draining wetlands for agricultural production. Brown concluded that market prices fully reflected the effect of the easement program. Differences in agricultural

[14]Other wetland values such as water quality maintenance, groundwater recharge, and flood control were recognized but not evaluated.

land values were based on the net income foregone by the land owner as a result of the easement. However, Brown also concluded (based on waterfowl hunting benefits of prairie wetlands provided by Hammack and Brown 1974) that the easement program resulted in an efficient allocation of wetland resources even where the value of the studied agricultural lands was relatively high. Brown cautioned that his findings could not be used to generalize for all wetlands in the region.

Gillick and Scott (1975) performed an economic analysis of the relationship between buffer strips and fishery resources along Miller Creek in the Olympic Peninsula. The study attempted to estimate the size of an optimal buffer strip given the value of timber resources, fishery resources, and the impact of timber harvest practices on the aquatic environment. The only aquatic-related values calculated were those for sport and commercial fisheries. The value of harvested timber was the only riparian land value calculated.

Questionnaires were sent to sport fishermen requesting information about their willingness-to-pay for the fishery resource. The economic value of the commercial fishery was estimated by deducting costs of production from catch value to the fishermen in Washington State. The study concluded that no buffer strip or a selectively cut buffer strip resulted in greater joint value than did a 15 or 30 m natural buffer.

The studies reviewed here suggest a number of things for similar research in riverine riparian areas. Since many values of riparian ecosystems are not allocated by markets because of the inherent institutional failures discussed at the beginning of this chapter, careful study of the typical values associated with these natural areas must be performed. Once the goods, services, and other functions of an area have been established, natural resource and environmental economics specialists should be consulted. Since the economic methods discussed are complex and the pitfalls on the road to sound analysis are many, the need for a trained specialist is apparent. Trained economists should be consulted to insure that the methods employed reflect the best state-of-the-art.[15]

The need for site specific studies is especially important. Ambitious studies attempting to make gross generalizations about the value of natural riparian areas are probably doomed to failure. Studies of this sort must recognize and utilize information from the natural sciences to affirm relationships between the natural environment and the resulting benefits realized by humans (Freeman 1979, Shabman et al. 1979).

Approaches to Valuation (III): Life Support or Energy Analysis

Values Based on Life Support. All yields and services provided by riparian ecosystems, as well as the ability of these ecosystems to recover from stress, are a function of the energy flow patterns through these ecosystems. Primary productivity is an important but not the exclusive component of the energy flow pattern. All organisms of an ecosystem depend upon the energy rich productions of photosynthesis. These organisms, in turn, perform many of the services that we value. Using this reasoning, Odum (1971) proposed that value should be calculated on the basis of the magnitude of energy flow associated with the primary productivity of the ecosystem. Using primary productivity data, Odum found that the work required to develop the recreational value of the bays near Corpus Christi in Texas was equal to 32.63 Kcal/m • day. Primary productivity contributed 22.5 times more work to the recreational value than people did through management and purchase of recreation-related goods and services. In these calculations, the calorie is used as a common unit of measure since calories of potential energy are required to perform all kinds of work in the real world. In addition, dollar values for ecosystems are derived from the relationship between energy flow in the eco-

[15]The most complete study of the economics of natural environments including discussion of concepts and several case studies are found in Krutilla and Fisher (1975).

system and the total energy flow of the nation. It is assumed that the dollar value of the ecosystem bears the same relationship to the gross national product, i.e.,

$$\frac{\text{dollar value of ecosystem}}{\text{gross national product}} = \frac{\text{energy flow of ecosystem}}{\text{energy flow of nation}}$$

Using this factor, Odum calculated the value of both human and natural ecosystem work with the same units.

Conceptually, the approach differs from those discussed so far because it takes into consideration the total amount of work that ecosystems perform. It also attempts to standardize the measurement of value with an energy unit (the calorie) that is governed by absolute physical laws rather than being vulnerable to the problems of inflation or deflation associated with the dollar.

The life support approach, however, has at least three difficulties. First, using primary productivity as a measure of value implies that ecosystems with higher primary productivity are more valuable than those with lower rates of primary productivity. The ranges of primary productivity values for riparian ecosystems are probably proportional to litterfall shown in Table 11. Is a streamside forest in Minnesota less valuable than a cypress swamp in Florida? It may be incorrect to consider systems with low primary productivity as less valuable than systems with high primary productivity, because the slower system may be compensating for its lower productivity with high quality products. Second, this approach accounts only for flows of energy associated with primary productivity; however, in many riparian ecosystems the work of the river in transporting and depositing sediments may be more important (in terms of quality) than solar energy capture by photosynthesis. While other energy flows are indirectly included in the primary productivity response of the system, their value should not be ignored in a calculation of ecosystem value. Third, the approach also fails to account for the variations in the quality of the energies that converge on an ecosystem. Variations in energy quality reflect variations in the capability to do work. Thus, energy flows with different energy qualities cannot be added without a correction for the quality difference.

Values Based on Energy Analysis, Corrected for Quality. Calculations of value based on life support discriminate against systems with low productivity. By correcting for energy quality the problem is partly resolved. For example, in arid riparian ecosystems where goods and services are limited by water supply, the energetic value of water in driving other ecosytem processes may be much higher than in the southeastern floodplain forests, where water is normally in abundance. Corrections for energy quality account for differences in energy value of the same substance under different conditions, the energy cost of concentrating energy by means of energy transformations, and the regional role or value of ecosystems (Odum 1970, 1973, 1978; Odum and Odum 1976). For example, the energy value of sun and wind, nutrients and sediments, and water shown as forcing functions in Figure 24 must all be converted to equivalent energy quality units. By expressing all energy flows in the same units of quality, the method allows summation of flows. Other examples of aspects of this approach are reviewed in Lugo and Brinson (1978).

The contribution of natural ecosystems to a regional economy can also be measured by the ratio of fossil-fuel energy use to the sum of all natural energies dissipated in the region. The ratio for the United States is 2.5, but ratios range from values as high as 10 for urban areas, such as Miami, to as low as 0.3 for the world as a whole. The competitiveness of an economy may ultimately depend on the free energy contribution from the natural sector. A high ratio means a small energy contribution from natural ecosystems, a high dependency on purchased energy resources from outside suppliers, and a poor competitive position if the economy is dependent on unreliable resources. A low ratio may not be competitive, because the region may be limited technologically. As fossil fuel sources of energy become more expensive, society will become more dependent on ecosystems that are driven by solar energy and other energy sources such as water flow.

Odum (1977) suggests that energy could be used in c-b analysis in much the same way that money is used. Decisions using least cost alternatives would be made on the basis of the useful work performed by the whole system, which is assumed to contribute to the economic vitality of a region. For example, Kemp et al. (1977) discuss the application of c-b analysis to the options available for cooling water from a three-unit nuclear power plant at Crystal River, Florida. The alternative of using the estuary as a recipient of heated water resulted in an estimated loss of about 0.002% of the total regional flow of natural energies (decreases in primary productivity, death of organisms by entrainment, etc.). However, by using mechanical draft cooling towers to cycle the water, a diversion of fossil fuel energy of about 160 times that lost by the estuary was predicted. Interestingly, 30% of the energy cost of cooling tower construction and maintenance was from environmental impacts outside the affected region, a value about two times greater than the projected impact on the estuary if it was used to receive discharges of heated water.

One of the shortcomings of energy analysis is that it has not been tested in enough situations to gain widespread acceptance. Using energy as the basis for c-b analysis does not seem to be generally accepted among economists as an alternative for traditional monetary c-b analyses, although, some writers have suggested common bases for economic and energy analysis.

LITERATURE CITED

Ackerman, B., S. Rose-Ackerman, J. Sawyer, Jr., and D. Henderson. 1974. The Uncertain Search for Environmental Quality. The Free Press, New York.

Allen, J. R. L. 1965. A review of the origin and characteristics of recent alluvial sediments. Sedimentology 5:89-191.

Anderson, B. W., A. Higgins, and R. D. Ohmart. 1977. Avian use of saltcedar communities in the Lower Colorado River valley, pp. 128-136. In R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-43, Ft. Collins, Colo.

Anderson, B. W. and R. D. Ohmart. 1977. Vegetation structure and bird use in the lower Colorado River Valley. pp. 23-34. In R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-43, Ft. Collins, Colo.

Anderson, B. W., R. D. Ohmart, and J. Disano. 1978. Revegetating the riparian floodplain for wildlife, pp. 318-331. In R. R. Johnson and J. F. McCormick (tech. coord.). Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Anderson, F., A. Kneese, P. Reed, S. Taylor, and R. Stevenson. 1977. Environmental Improvement Through Economic Incentives. Johns Hopkins University Press, Baltimore.

Anderson, R. C. and J. White. 1970. A cypress swamp outlier in southern Illinois. Trans. Ill. State Acad. Sci. 63:6-13.

Armstrong, D. M. 1977. Ecological distribution of small mammals in the upper Williams Fork Basin, Grand County, Colorado. Southwest Naturalist 22:289-304.

Arnold, L. W. 1940. An ecological study of the vertebrate animals of the mesquite forest. M.S. Thesis, Univ. of Arizona, Tucson. 79 pp.

Ausness, R. 1977. Water use permits in a riparian state: problems and proposals. Kentucky Law Journal 6:191-265.

Austin, G. T. 1970. Breeding birds of desert riparian habitat in southern Nevada. Condor 72:431-436.

Babcock, H. M. 1968. The phreatophyte problem in Arizona. 12th Annual Ariz. Watershed Symposium: Proceedings 12:34-36.

Babcock, H. M. and E. M. Cushing. 1942. Recharge to ground-water from floods in a typical desert wash, Pinal County, Arizona. Trans. Amer. Geophys. Union 1942:49-56.

Bailey, R. G. 1976. Ecoregions of the United States. USDA Forest Service, Intermountain Region, Ogden, Utah. U.S. GPO 777-152.

Baram, M. 1980. Cost-benefit analysis: an inadequate basis for health, safety, and environmental regulatory decisionmaking. Ecology Law Quarterly 8:473-532.

Barclay, J. S. 1978. The effects of channelization on riparian vegetation and wildlife in south central Oklahoma; pp. 129-138. In R. R. Johnson and J. F. McCormick (tech. coord.). Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Barclay, J. S. 1980. Impact of Stream Alterations on Riparian Communities in Southcentral Oklahoma. U.S. Fish and Wildlife Service, FWS/OBS-80/17, Washington, D.C. 91 pp.

Barnett, H. and C. Morse. 1963. Scarcity and Growth: The Economics of Natural Resource Availability. Johns Hopkins Press, Baltimore.

Batie, S. and J. Wilson. 1978. Economic values attributable to Virginia's coastal wetlands as inputs in oyster production. Southern J. of Agric. Econ. 10:111-118.

Batzli, G. O. 1977. Population dynamics of the white-footed mouse in floodplain and upland forests. Am. Midl. Nat. 97:18-32.

Beck, K. C., J. H. Reuter and E. M. Perdue. 1974. Organic and inorganic geochemistry of some coastal plain rivers of the southeastern United States Geochim. Cosmochim. Acta 38: 341-364.

Bedinger, M. S. 1979. Forests and Flooding with Special References to the White River and Ouachita River Basins, Arkansas. U.S. Geol. Survey Water Resources Investigations, Open-file Report 79-68, Washington, D.C.

Beidleman, R. G. 1948. The vertebrate ecology of a Colorado plains cottonwood river bottom. M.S. thesis, Univ. of Colorado, Boulder. 351 pp.

Beidleman, R. G. 1954. The cottonwood river-bottom community as a vertebrate habitat. Ph.D. Thesis. Univ. of Colorado, Boulder. 358 pp.

Beidleman, R. G. 1978. The cottonwood-willow riparian ecosystem as a vertebrate habitat, with particular reference to birds, pp. 192-195. In W. D. Graul and S. J. Bissell (tech. coord.), Lowland River and Stream Habitat in Colorado: A Symposium. Colo. Chap. Wildl. Soc. and Colo. Audubon Counc., Denver, Colo.

Bell, D. T. and F. L. Johnson. 1974. Ground-water level in the floodplain and adjacent uplands of the Sangamon River. Trans. Ill. State Acad. Sci. 67:376-383.

Bell, D. T. and S. K. Sipp. 1975. The litter stratum in the streamside forest ecosystem. Oikos 26:391-397.

Belt, C. B. Jr. 1975. The 1973 flood and man's constriction of the Mississippi River. Science 189:681-684.

Benke, A. C., D. M. Gillespie, and F. K. Parrish. 1979. Biological Basis for Assessing Impacts of Channel Modification: Invertebrate Production, Drift, and Fish Feeding in a Southeastern Blackwater River. Environmental Resources Center, Rep. 06-79, Georgia Inst. Techn., Atlanta. 187 pp.

Berry, K. H. 1977. The east Mojave Desert: an example of inventory design and habitat analysis, pp. 495-519. In A. Marmelstein (ed.), Classification, Inventory, and Analysis of Fish and Wildlife Habitat. U.S Fish & Wildlife Service FWS/OBS-78/76.

Best, L. B., D. F. Stauffer and A. R. Geier. 1978. Evaluating the effects of habitat alteration on birds and small mammals occupying riparian communities, pp. 117-124. In R. R. Johnson and J. F. McCormick (tech. coords.). Proceedings of the Symposium on Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Bigelow, R. 1979. Selection of wilderness areas: an economic framework for decision-making. Utah Law Review 1979:269-294.

Bilby, R. E. and G. E. Likens. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. Ecology 61:1107-1113.

Bish, R. 1971. The Public Economy of Metropolitan Areas. Markham Publishing Co. Chicago.

Bishop, B. 1978. Endangered species and uncertainty: the economics of a safe minimum standard. American J. of Agric. Econ. 60:10-18.

Blair, W. F. 1939. Some observed effects of stream-valley flooding on mammalian populations in eastern Oklahoma. J. Mammal. 20:304-306.

Blem, C. R. and L. B. Blem. 1975. Density, biomass, and energetics of the bird and mammal populations of an Illinois deciduous forest. Trans. Ill. Acad. Sci. 68:156-164.

Bliss, L. C. and J. E. Cantlon. 1957. Succession on river alluvium in northern Alaska. Am. Midl. Nat. 58: 452-469.

Boeer, W. J. and D. J. Schmidly. 1977. Terrestrial mammals of the riparian corridor in Big Bend National Park. p. 212-217. In R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-43, Ft. Collins, Colo.

Bormann, F. H. and G. E. Likens. 1979. Pattern and Process in a Forested Ecosystem. Springer-Verlag, New York. 253 pp.

Bormann, F. H., G. E. Likens, T. G. Siccama, R. S. Pierce, and J. S. Eaton. 1974. The export of nutrients and recovery of stable conditions following deforestation at Hubbard Brook. Ecol. Monogr. 44:255-277.

Bottorff, R. L. 1974. Cottonwood habitat for birds in Colorado. Am. Birds 28:975-979.

Bowie, J. E., W. Kam, E. A. Branson, and R. S. Aro. 1968. Use of Water by Riparian Vegetation, Cottonwood Wash, Arizona. U.S. Geol. Surv. Water-Supply Pap. 1858. Washington, D.C. 62 pp.

Bradt, G. W. 1947. Michigan Beaver Management. Mich. Dept. Cons. Game Div. 56 pp.

Bray, J. R. and E. Gorham. 1964. Litter production in forests of the world. Adv. Ecol. Res. 2:101-157.

Brice, J. 1971. Measurement of lateral erosion at proposed river crossing sites of the Alaska pipeline. U.S. Geol. Surv., Water Resource Div., Alaska District. 39 pp.

Brinson, M. M. 1973. The Organic Matter Budget and Energy Flow of a Tropical Lowland Aquatic Ecosystem. Ph.D. Thesis, University of Florida, Gainesville, FL. 251 pp.

Brinson, M. M. 1976. Organic matter losses from four watersheds in the humid tropics. Limnol. Oceanogr. 21: 572-582.

Brinson, M. M. 1977. Decomposition and nutrient exchange of litter in an alluvial swamp forest. Ecology 58: 601-609.

Brinson, M. M., H. D. Bradshaw, and R. N. Holmes. 1981a. Significance of floodplain sediments in nutrient exchange between a stream and its floodplain. In T. O. Fontaine III, S. M. Bartell (eds). Dynamics of Lotic Ecosystems. Symposium of the Savannah River Ecology Lab, Augusta, Ga. Oct. 19-22, 1980. (In press).

Brinson, M. M., H. D. Bradshaw, R. N. Holmes, and J. B. Ekins, Jr. 1980. Litterfall, stemflow, and throughfall nutrient fluxes in an alluvial swamp forest. Ecology 60:827-835.

Brinson, M. M., H. D. Bradshaw, and E. S. Kane. 1981b. Nitrogen Cycling and Assimilated Capacity of Nitrogen and Phosphorus by Riverine Wetland Forests. Water Resour. Res. Inst. - Univ. North Carolina, Rep. No. 167. Raleigh. 90 pp.

Broadfoot, W. M. 1967. Shallow-water impoundment increases soil moisture and growth of hardwoods. Soil Sci. Soc. Amer. Proc. 31:562-564.

Brookshire, D., B. Ives, and W. Schultze. 1976. The valuatión of aesthetic preferences. Journ. of Environmental Economics Management 3: 325-346.

Brown, J. 1975. Stream channelization: The economics of a controversy. Nat. Resources Jour. 14:557-576.

Brown, L. N. 1967. Ecological distribution of mice in the Medicine Bow Mountains of Wyoming. Ecology 48: 677-680.

Brown, L. and D. Amadon. 1968. Eagles, Hawks and Falcons of the World. McGraw Hill Book Co., N.Y. Vol. 2. 945 pp.

Brown, R. 1976. A study of the impact of the wetlands easement program on agricultural land values. Land Economics 52:509-517.

Brown, S. 1978. A comparison of cypress ecosystems and their role in the Florida landscape. Ph.D. Diss., Univ. of Florida, Gainesville. 569 pp.

Brown, S., M. M. Brinson, and A. E. Lugo. 1978. Structure and function of riparian wetlands, pp. 17-31. In R. R. Johnson and J. F. McCormick (tech. coord.). Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA Forest Service Gen. Tech. Rep. WO-12. Washington, D.C.

Brown, W., A. Singh, and E. Castle. 1964. An Economic Evaluation of the Oregon Salmon and Steelhead Sport Fishery. Tech. Bull. No. 78. Oregon State Univ. Agricultural Experiment Station, Corvallis.

Bruner, W. E. 1931. The vegetation of Oklahoma. Ecol. Monogr. 1:99-188.

Buell, M. F. and W. A. Wistendahl. 1955. Floodplain forests of the Raritan River. Bull. Torrey Bot. Club. 82:463-472.

Bulkley, R. V., R. W. Bachmann, K. D. Carlander, H. L. Fierstine, L. R. King, B. W. Menzel, A. L. Witten, and P. W. Zimmer. 1976. Warmwater Stream Alteration in Iowa: Extent, Effects on Habitat, Fish, and Fish Food, and Evaluation of Stream Improvement Structures (Summary Report). U.S. Fish and Wildlife Service FWS/OBS-76/16. Washington, D.C. 39 pp.

Burkhard, W. T. 1978. Vertebrate associations in lowland versus high elevation river and stream habitat in Colorado, pp. 52-55. In W. D. Graul and S. J. Bissell (tech. coord.), Lowland River and Stream Habitat in Colorado: A Symposium. Colo. Chap. Wildl. Soc. and Colo. Audubon Counc., Denver, Colo.

Burns, L. A. 1978. Productivity, Biomass, and Water Relations in a Florida Cypress Forest. Ph.D. dissertation, Univ. of North Carolina, Chapel Hill. 170 pp.

California Dept. of Fish and Game. 1966. California Fish and Wildlife Plan, Vol. III. Calif. Dept. Fish Game, Sacramento.

California Dept. of Water Resources. 1979. Land Use Change in the Sacramento River Riparian Zone, Redding to Colusa: An update-1972 to 1977. The Resources Agency, State of California, Sacramento. 38 pp.

Campbell, C. J. 1970. Ecological implications of riparian vegetation management. J. Soil Water Conser. 25: 49-52.

Campbell, C. J. and W. A. Dick-Peddie. 1964. Comparison of phreatophyte communities on the Rio Grande in New Mexico. Ecology 45:492-502.

Campbell, C. J. and W. Green. 1968. Perpetual succession of stream-channel vegetation in a semiarid region. J. Ariz. Acad. Sci. 5:86-98.

Canada Fisheries and Marine Service. 1978. Guidelines for Land Development and Protection of the Aquatic Environment. Fisheries and Marine Service Tech. Rep. 807, Fisheries and Oceans Canada, Vancouver, B.C. 55 pp.

Carlson, W. A. and R. T. Runnels. 1952. A study of silt deposited by the July 1951 flood, central Kansas River valley. Kans. Acad. Sci. Trans. 55: 209-213.

Carothers, S. W. and R. R. Johnson. 1975. Water management practices and their effects on nongame birds in range habitats, pp. 210-222. In D. Smith (tech. coord.), Proc. of Symposium on Management of Forest and Range Habitats for Nongame Birds. USDA For. Serv. Gen. Tech. Rep. WO-1, Washington, D.C.

Carothers, S. W., R. R. Johnson and S. W. Aitchison. 1974. Population structure and social organization of Southwestern riparian birds. Amer. Zool. 14:97-108.

Carter, M. R., L. A. Burns, T. R. Cavinder, K. R. Dugger, P. L. Fore, D. B. Hicks, H. L. Revells, T. W. Schmidt, and R. Farley. 1973. Ecosystem analysis of the Big Cypress Swamp and estuaries. EPA 904/ 9-74-002, Atlanta. 478 pp.

Chalk, D. E. 1979. Predicting impacts of a proposed irrigation water conservation project on wildlife habitat, pp. 305-309. In G. A. Swanson (tech. coord.). The Mitigation Symposium: A National Workshop on Mitigating Losses of Fish and Wildlife Habitats. USDA For. Serv. Gen. Tech. Rep. RM-65. Ft. Collins, Colo.

Chapman, D. W. and E. Knudsen. 1980. Channelization and livestock impacts on salmonid habitat and biomass in western Washington. Trans. Am. Fish. Soc. 109:375-363.

Childers, J. M. 1970. Flood Frequency in Alaska. U.S. Geol. Surv. Open-File Rep. Alaska District. 30 pp.

Clark, R. E. 1972. Waters and Water Rights, Vols. 5, 6, 7. The Allen Smith Company, Indianapolis.

Clawson, M. (ed.). 1977a. Research in Forest Economics and Forest Policy. Resources for the Future, Inc., Washington, D.C.

Clawson, M. 1977b. Decision Making in Timber Production, Harvest, and Marketing. Resources for the Future, Inc., Washington, D.C.

Coble, R. W. 1979. A Technique for Estimating Heights Reached by the 100-year Flood in Unregulated, Nontidal Streams in North Carolina. U.S. Geol. Surv. Water-Resources Investigations, 79-69, Raleigh.

Cohan, D. R., B. W. Anderson, and R. D. Ohmart. 1978. Avian population responses to saltcedar along the lower Colorado River, pp. 371-381. In R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain, Wetlands, and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Comptroller General. (March 8, 1974). Improved Federal Efforts Needed to Equally Consider Wildlife Conservation with other Features of Water Resource Development. U.S. General Accounting Office, Washington, D.C.

Comptroller General. (June 2, 1978). Better Analysis of Uncertainty Needed for Water Resource Projects. U.S. General Accounting Office, Washington, D.C.

Comptroller General. (August 7, 1978). An Overview of Benefit-Cost Analysis for Water Resources Projects -- Improvements Still Needed. U.S. General Accounting Office, Washington, D.C.

Comptroller General. (Feb. 8, 1979). Better Understanding of Wetland Benefits Will Help Water Bank and Other Federal Programs Achieve Wetland Preservation Objectives. U.S. General Accounting Office, Washington, D.C.

Comptroller General. (Nov. 13, 1980). Congressional Guidance Needed on Federal Cost Share of Water Resource Projects Where Project Benefits Are Not Widespread. U.S. General Accounting Office, Washington, D.C.

Conant, R. 1958. A Field Guide to Reptiles and Amphibians of the United

States and Canada East of the 100th Meridian. Houghton Mifflin Co., Boston, Mass. 366 pp.

Conard, S. G., R. L. MacDonald, and R. F. Holland. 1977. Riparian vegetation and flora of the Sacramento Valley, pp. 47-55. In A. Sands (ed.). Riparian Forests in California: Their Ecology and Conservation. Publ. No. 15, Institute of Ecology, Univ. of Calif., Davis.

Conine, K. H., B. W. Anderson, R. D. Ohmart, and J. F. Drake. 1978. Responses of riparian species to agricultural habitat conversions, pp. 248-263. In R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain, Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Conner, W. H., and J. W. Day, Jr. 1976. Productivity and composition of a baldcypress-water tupelo site and bottomland hardwood site in a Louisiana swamp. Am. J. Bot. 63:1354-1364.

Conner, R. N. 1978. Snag management for cavity nesting birds, pp. 120-128. In R. M. DeGraaf (tech. coord.), Proceedings of the Workshop Management of Southern Forests for Nongame Birds. USDA For. Serv. Gen. Tech. Rep. SE-14, Atlanta.

Conservation Foundation. 1980. Cost-Benefit Analysis: A Tricky Game. Conservation Foundation Letter, December. Washington, D.C.

Cooper, R. H. and A. B. Hollingshead. 1973. River bank erosion in regions of permaforst, pp. 272-283. In Fluvial Process and Sedimentation. Proc. Hydrol. Symp. at Univ. Alberta. Information Canada, Ottawa.

Cornwell, G. W. 1963. Observations on the breeding biology and behavior of a nesting population of belted kingfisher. Condor 65:426-431.

Correll, D. L. 1978. Estuarine productivity. Bioscience 28:646-650.

Council on Environmental Quality. 1971, 1975, 1978, 1979. Environmental Quality (Annual Reports). Washington, D.C.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of Wetlands and Deepwater Habitats of the United States. U.S. Fish and Wildlife Service, FWS/OBS-79/31, Washington, D.C. 103 pp.

Craighead, J. J. and F. C. Craighead, Jr. 1956. Hawks, Owls and Wildlife. Stackpole Co., Harrisburg. 443 p.

Crites, R. W. and J. E. Ebinger. 1969. Vegetation survey of floodplain forests in east-central Illinois. Trans. Ill. Acad. Sci. 62:316-330.

Crouch, G. L. 1961. Wildlife populations and habitat conditions on grazed and ungrazed bottomlands in Logan County, Colorado. M.S. thesis, Colorado State Univ., Ft. Collins, Colo. 144 pp.

Crouch, G. L. 1979. Changes in the vegetation complex of a cottonwood ecosystem on the South Platte River, pp. 19-22. In Great Plains Agricultural Council, Forestry Committee (Program Committee), Riparian and Wetland Habitats of the Great Plains. Great Plains Agricultural Council Publication No. 91. USDA For. Serv. Rocky Mt. For. and Range Exp. Stn., Fort Collins, Colo. 88 pp.

Culler, R. C. 1970. Water conservation by removal of phreatophytes. Trans. Amer. Geophys. Union 51:684-689.

Culler, R. C. and Others. 1970. Objectives, Methods, and Environment--Gila River Phreatophyte Project, Graham County, Arizona. U.S. Geol. Surv. Prof. Pap. 655-A. 25 pp.

Cummins, K. N. 1974. Structure and function of stream ecosystems. Bioscience 24:631-641.

Curtis, R. L. and T. H. Ripley. 1975. Water management practices and their effect on nongame bird habitat values in a deciduous forest community, pp. 128-141. In D. R. Smith (Tech. Coord.). Proceedings of the Symposium on Management of Forest and Range Habitats for Nongame Birds. USDA For.

Serv. Gen. Tech. Rep. WO-1, Washington, D.C.

Dahlem, E. A. 1979. The Mahogany Creek watershed--with and without grazing, pp. 31-34. In O. B. Cope (ed.), Grazing and Riparian/Stream Ecosystems. Trout Unlimited, Denver, Colo.

Dales, J. H. 1970. Pollution, Property and Prices. Univ. of Toronto Press, Toronto.

Davis, J. R. and R. P. Cheek. 1966. Distribution, food habits, and growth of young clupeids, Cape Fear River system, North Carolina. Proc. Annual Conf., SE Assoc. Game Fish Comm. 20: 250-259.

Dawson, D. K. 1979. Bird communities associated with succession and management of lowland conifer forests, pp. 120-131. In R. M. DeGraaf (tech. Coord.), Workshop Proceedings: Management of North Central and Northeastern Forests for Nongame Birds, USDA For. Serv. Gen. Tech. Rep. NC-51, St. Paul, Minn.

Day, J. W., Jr., T. J. Butler, and W. H. Conner. 1977. Productivity and nutrient export studies in a cypress swamp and lake system in Louisiana, pp. 255-269. In M. Wiley (ed.). Estuarine Processes, Vol. 2. Circulation, Sediments, and Transfer of Material in the Estuary. Academic Press, New York.

Delorme, Jr., C., and N. J. Wood. 1974. Savannah river improvement and environmental preservation. Land Economics 50:284-288.

deMarch, L. 1975. Nutrient budgets for a high arctic lake (Char Lake, N. W. T.). Verh. Internat. Verein. Limnol. 19:496-503.

Dickson, J. G. 1978. Forest bird communities of the bottomland hardwoods, pp. 66-73. In R. M. DeGraaf (tech. coord.), Proceedings of the Workshop Management of Southern Forests for Nongame Birds. USDA For. Serv. Gen. Tech. Rep. SE-14, Atlanta.

Dobbs, D. 1973. Remedies. West Publishing Co, St. Paul.

Dodd, C. K., Jr. 1978. Amphibians and reptiles: the declining species. Water Spectrum 10:24-32.

Dodge, W. E., E. E. Possardt, R. J. Reed and W. P. MacConnell. 1976. Channelization Assessment. White River, Vermont: Remote Sensing, Benthos, and Wildlife. USDI FWS/OBS-76/07. Washington, D.C. 73 pp.

Drury, W. H., Jr. 1956. Bog Flats and Physiographic Process in the Upper Kuskokwim River Region, Alaska. Contrib. Gray Herbarium, Harvard Univ. 1128. 130 pp.

Duever, M. J., J. E. Carlson and L. A. Riopelle. 1975. Ecosystem analysis at Corkscrew Swamp, pp. 627-725. In H. T. Odum, K. C. Ewel, (eds.). Cypress Wetlands for Water Management, Recycling and Conservation, Univ. of Florida, Gainesville.

Duffield, J., J. Carroll, L. Fischer, S. Hanke, R. Haveman, C. Howe, W. O'Neil, T. Power, and V. K. Smith. 1979. Benefits or Costs II: An Analysis of the Water Resources Council's Manual of Procedures for Evaluation of Benefits and Costs. Natl. Wildl. Fed., Washington, D.C.

Durbin, A. G., S. W. Nixon, and C. A. Oviatt. 1979. Effects of the spawning migration of the alewife, *Alosa pseudoharengus*, on freshwater ecosystems. Ecology 60:8-17.

Dury, G. H. 1964a. Principles of Underfit Streams. U.S. Geol. Surv. Prof. Pap. 452A. 67 pp.

Dury, G. H. 1964b. Subsurface Exploration and Chronology of Underfit Streams. U.S. Geol. Surv. Prof. Pap. 452B. 56 pp.

Dury, G. H. 1965. Theoretical Implications of Underfit Streams. U.S. Geol. Surv. Prof. Pap. 452C. 43 pp.

Dwyer, J.F., J. R. Kelley, and M. O. Bowes. 1977. Improved Procedures for Valuation of the Contribution of Recreation to National Economic Development. Research Report No. 128. Water Resources Center, University of Illinois, Urbana.

Eckblad, J. W., N. L. Peterson, and K. Ostlie. 1977. The morphometry, benthos and sedimentation rates of a floodplain lake in Pool 9 of the upper Mississippi River. Amer. Midl. Nat. 97:433-443.

Efford, I. E. 1972. An interim review of the Lake Marion Project, pp. 89-109. In Z. Kajak and A. Hillbricht-Ilkowska (eds.). Productivity Problems of Freshwaters. PWN-Pol. Sci., Warsaw.

Eggler, W. A. 1938. The maple-basswood forest type in Washburn County, Wisconsin. Ecology 19:243-263.

Ellis, R. W. 1976. The Impact of Stream Alteration on Wildlife Along Tributaries of the Roanoke River, Charlotte County, Virginia. M. S. Thesis, Virginia Polytechnic Institute and State Univ., Blacksburg. 105 pp.

Emmerich, J. M. 1978. Bird Utilization of Woodland Habitat in the Eastern Quarter of South Dakota. M.S. Thesis. South Dakota State Univ., Brookings. 130 pp.

Engel-Wilson, R. W. and R. D. Ohmart. 1978. Floral and attendant faunal changes in the Lower Rio Grande between Fort Quitman and Presidio, Texas, pp. 139-147. In R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain, Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Enviro Control, Inc. 1978. Instream Flow Strategies for Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, North Dakota, Oregon, South Dakota, Utah, Washington, and Wyoming. U.S. Fish and Wildlife Service FWS/OBS-78/35-47, Ft. Collins, Colo.

Erman, D. C., J. D Newbold and K. B. Roby. 1977. Evaluation of Streamside Buffer strips for Protecting Aquatic Organisms. California Water Resources Center Technical Completion Report Contribution No. 165, Univ. of Calif., Davis. 48 pp.

Errington, P. L. 1937. Habitat requirements of stream-dwelling muskrats. Trans. N. Am. Wildl. Conf. 2: 411-416.

Everitt, B. L. 1968. Use of the cottonwood in an investigation of the recent history of a flood plain. Am. J. Sci. 266:417-439.

Everitt, B. L. 1980. Ecology of saltcedar--a plea for research. Environ. Geol. 3:77-84.

Faanes, C. 1979. New bird haunts 91, Lake Pepin region. In D. D. Tessen (eds.), Wisconsin's Favorite Bird Haunts Supplement. Wis. Society Ornith., Madison. 424 pp.

Faanes, C. A. 1981. Birds of the St. Croix River Valley: Minnesota and Wisconsin. U.S. Fish and Wildlife Service, North Am. Fauna Series Number 73, Washington, D.C. 196 pp.

Finzel, J. E. 1964. Avian populations of four herbaceous communities in southeastern Wyoming. Condor 66: 496-510.

Fisher, A., and F. Peterson. 1976. The environment in economics: A survey. The Journ. of Econ. Lit. 14:1-33.

Fisher, S. G. 1977. Organic matter processing by a stream-segment ecosystem: Fort River, Massachusetts, U.S.A. Int. Revue ges. Hydrobiol. 62:701-727.

Fisher, S. G. and S. R. Carpenter. 1976. Ecosystem and macrophyte primary production of the Fort River, Massachusetts. Hydrobiologia 47: 175-187.

Fisher, S. G., L. J. Gray, N. B. Grimm, and D. E. Busch. 1980. Post-flood recovery of a desert stream ecosystem. Abstracts of 43rd Annual Meeting of Amer. Soc. Limnol. Oceanogr., June 16-19, 1980, Knoxville, Tennessee.

Fisher, S. G. and G. E. Likens. 1973. Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. Ecol. Monogr. 43:421-439.

Fisk, H. N. 1944. Geological Investigation of the Alluvial Valley of the Lower Mississippi River. Miss. River Comm., Vicksburg, Miss. 78 pp.

Fisk, H. N. 1952. Geological Investigation of the Atchafalaya Basin and the Problem of the Mississippi River Diversion. Volume I. Miss. River Comm., Vicksburg, Miss. 145 pp.

Fisk, H. N. and E. McFarland, Jr. 1955. Late Quaternary deltaic deposits of the Mississippi River. Geol. Soc. Amer. Spec. Pap. 62:279-302.

Fitzgerald, J. P. 1978. Vertebrate associations in plant communities along the South Platte River in northeastern Colorado, pp. 73-88. In W. D. Graul and S. J. Bissell (tech. coord.), Lowland River and Stream Habitat in Colorado: A Symposium. Colo. Chap. Wildl. Soc. and Colo. Audubon Counc., Denver, Colo.

Flake, L. D. and P. A. Vohs Jr. 1979. Importance of Wetland Types to Duck Production and to Non-game Bird Populations. Completion Report, Project No. B-045-SDAK, Wildlife and Fisheries Dept., South Dakota State Univ., Brookings. 50 pp.

Fleener, G. G. 1951. Life history of the cutthroat trout, Salmo clarki Richardson, in Logan River, Utah. Trans. Am. Fish. Soc. 81:235-248.

Flinchum, D. M. 1977. Lesser Vegetation as Indicators of Varying Moisture Regimes in Bottomland and Swamp Forests of Northeastern North Carolina. Ph.D. Diss., North Carolina State Univ., Raleigh. 49 pp.

Flood, B. S., M. E. Sangster, R. D. Sparrowe, and T. S. Baskett. 1977. A Handbook for Habitat Evaluation Procedures. U.S. Fish and Wildlife Service, Resource Publication 132, Washington, D.C. 77 pp.

Fonda, R. W. 1974. Forest succession in relation to river terrace development in Olympic National Park, Washington. Ecology 55:927-942.

Fraley, J. J. 1979. Effects of elevated stream temperatures below a shallow reservoir on a cold water macroinvertebrate fauna, pp. 257-272. In J. V. Ward and J. A. Stanford (eds.). The Ecology of Regulated Streams. Plenum Press, New York.

Fredrickson, L. H. 1978. Lowland hardwood wetlands: Current status and habitat values for wildlife, pp. 296-306. In P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland Functions and Values: The State of Our Understanding - Proceedings of the National Symposium on Wetlands, American Water Resources Association, Minneapolis, Minn.

Fredrickson, L. H. 1979. Floral and Faunal Changes in Lowland Hardwood Forests in Missouri Resulting from Channelization, Drainage, and Impoundment. U.S. Fish and Wildlife Service, FWS/OBS-78/91, Washington, D.C.

Freeman, A. M. 1979. The Benefits of Environmental Improvement: Theory and Practice. Johns Hopkins University Press, Baltimore.

Freeman, A., R. Haveman, and A. Kneese. 1973. The Economics of Environmental Policy. John Wiley and Sons, Inc., New York.

Freeman, C. E. and W. A. Dick-Peddie. 1970. Woody riparian vegetation in the Black and Sacramento mountain ranges, southern New Mexico. Southwestern Naturalist 15:145-164.

Frey, H. T. 1979. Major Uses of Land in the United States: 1974. USDA Economics, Statistics and Cooperatives Service. Agricultural Economic Report No. 440. 33 pp.

Frye, R. J. II and J. A. Quinn. 1979. Forest development in relation to topography and soils on a floodplain of the Raritan River, New Jersey. Bull. Torrey Bot. Club 106:334-345.

Funk, J. L. 1970. Warm-water streams, pp. 141-152. In A Century of Fisheries in North America. Am. Fish. Soc. Spec. Publ. No. 7.

Funk, J. L. and J. W. Robinson. 1974. Changes in the Channel of the Lower Missouri River and Effects on Fish and

Wildlife. Aquatic Series No. 11, Missouri Dept. of Conservation, Jefferson City, Mo. 52 pp.

Gaddy, L. L., T. S. Kohlsaat, E. A. Laurent, and K. B. Stansell. 1975. A Vegetation Analysis of Preserve Alternatives Involving the Beidler Tract of the Congaree Swamp. South Carolina Wildl. & Mar. Resour. Dept., Columbia, S.C.

Gagliano, S. M. and J. L. van Beek. 1975. Environmental Base and Management Study, Atchafalaya Basin, Louisiana. U.S. Environmental Protection Agency, EPA-600/5-75-006, Washington, D.C.

Gaines, D. A. 1974. A new look at the nesting riparian avifauna of the Sacramento Valley, California. West. Birds 5:61-80.

Gaines, D. A. 1977. The Valley riparian forests of California: their importance to bird populations, pp. 57-85. In A. Sands (ed.), Riparian Forests in California: Their Ecology and Conservation. Publ. No. 15, Institute of Ecology, Univ. of Calif., Davis.

Galli, A. E., C. F. Leck, and R. T. Forman. 1976. Avian distribution patterns in forest islands of different sizes in central New Jersey. Auk 93:356-364.

Gambrell, R. P. and W. H. Patrick, Jr. 1978. Chemical and microbiological properties of anaerobic soils and sediments, pp. 375-423. In D. D. Hook and R. M. M. Crawford (eds.). Plant Life in Anaerobic Environments. Ann Arbor Science, Michigan. 564 pp.

Gatewood, J. S., J. W. Robinson, B. R. Colby, J. D. Hem, and L. C. Halpenny. 1950. Use of Water by Bottomland Vegetation in Lower Safford Valley, Arizona. U.S. Geol. Surv. Water-Supply Paper 1103.

Geier, A. R. 1978. Habitat Selection by Small Mammals of Riparian Communities: Evaluating the Effects of Habitat Alteration. M.S. thesis, Iowa State Univ., Ames. 46 pp.

Geier, A. R., and L. B. Best. 1980. Habitat selection by small mammals of riparian communities: evaluating effects of habitat alterations. J. Wildl. Manage. 44:16-24.

Gill, D. 1972a. The point bar environment in the Mackenzie River Delta. Can. J. Earth Sci. 9:1382-1393.

Gill, D. 1972b. Modification of levee morphology by erosion in the Mackenzie River Delta, Northwest Territories, pp. 123-138. In R. J. Price and D. E. Sugden (eds.), Polar Geomorphology. Institute of British Geographers Spec. Publ. No. 4.

Gill, J. D., J. W. Thomas, W. M. Healy, J. C. Pack, and H. R. Sanderson. 1975. Comparison of seven forest types for game in West Virginia. J. Wildl. Manage. 39:762-768.

Gillick, T. and B. Scott. 1975. Buffer Strips and the Protection of Fishery Resources: An Economic Analysis. DNR Report No. 32. State of Washington Department of Natural Resources, Olympia.

Gilman, B. A. 1976. Wetland Plant Communities Along the Eastern Shoreline of Lake Ontario. M.S. Thesis, SUNY Coll. Environ. Sci. For., Syracuse, N.Y.

Glasgow, L. L. and R. E. Noble. 1971. The importance of bottomland hardwoods to wildlife, pp. 30-43. In Proceedings of the Symposium on Southeastern Hardwoods (Sept. 15-16). Louisiana State Univ., Baton Rouge.

Godwin, R. and W. Shepard. 1979. The use of the commons dilemma in examing the allocation of common resources. The Western Political Quarterly 22: 265-277.

Golden, M. S. 1979. Forest vegetation of the lower Alabama Piedmont. Ecology 60:770-782.

Goldstein, J. 1971. Competition for Wetlands in the Midwest: An Economic Analysis. Johns Hopkins Press, Baltimore.

Goldwasser, S. 1978. Distribution, Abundance and Reproductive Success of the Least Bell's Vireo in California. Calif. Dept. of Fish and Game, Nongame Wildlife Investigations, Project W-54-R-10, Job IV-1.5.1. 18 pp.

Golet, F. C. 1976. Freshwater wetlands as wildlife habitats, pp. 84-115. In M. W. Lefor, W. C. Kernard, and T. B. Helfgott (eds.). Proceedings: The Wetland Conference. Univ. of Conn. Inst. of Water Resources. Rep. No. 26, Storrs.

Golley, F. B., J. T. McGinnis, R. G. Clements, G. I. Child, and M. J. Duever. 1975. Mineral Cycling in a Tropical Moist Forest Ecosystem. Univ. Georgia Press, Athens. 248 pp.

Gosselink, J., E. Odum, and R. Pope. 1974. The value of the Tidal Marsh. LSU-SG-74-03. Center for Wetland Resources, Louisiana State Univ., Baton Rouge.

Graber, R. and J. Graber. 1978. Letter report to G. A. Paulson, Illinois Nature Preserves Commission. Illinois Natural History Survey, Urbana. 5 pp.

Grannemann, N. G. and J. M. Sharp, Jr. 1979. Alluvial hydrogeology of the lower Missouri River valley. J. Hydrol. 40:85-99.

Greeson, P. E., J. R. Clark, and J. E. Clark. 1978. Wetland Functions and Values: The State of Our Understanding. Amer. Water Resour. Assoc., Minneapolis, Minn. 674 pp.

Griffith, J. S., Jr. 1974. Utilization of invertebrate drift by brook trout (Salvelinus fontinalis) and cutthroat trout (Salmo clarki) in small streams in Idaho. Trans. Am. Fish. Soc. 103:440-447.

Gupta, T. and J. Foster. 1975. Economic criteria for freshwater wetland policy in Massachusetts. Am. J. Agric. Econ. 57:40-45.

Haase, E. F. 1972. Survey of floodplain vegetation along the lower Gila River in southwestern Arizona. J. Ariz. Acad. Sci. 7:66-81.

Hadley, R. F. 1960. Recent Sedimentation and Erosional History of Fivemile Creek, Fremont County, Wyoming. U.S. Geol. Surv. Prof. Pap. 352-A. GPO, Washington, D.C.

Hagenstein, P. 1979. Public lands and environmental concerns. Arizona Law Review 21:449-460.

Hair, J. D., G. T. Hepp, L. M. Luckett, K. P. Reese, and D. K. Woodward. 1978. Beaver pond ecosystems and their relationships to multi-use natural resource management, pp. 80-92. In R. R. Johnson and J. F. McCormick (tech. coord.). Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Hairston, N. G. 1949. The local distribution and ecology of the plethodontid salamanders of the southern Appalachians. Ecol. Monogr. 19:47-73.

Hall, C. A. S. 1972. Migration and metabolism in a temperate stream ecosystem. Ecology 53:585-604.

Hall, J. G. 1960. Willow and aspen in the ecology of beaver on Sagehen Creek, California. Ecology 41: 484-494.

Hall, T. F. and W. T. Penfound. 1939a. A phytosociological study of a cypress-gum swamp in southeastern Louisiana. Am. Midl. Nat. 21:378-395.

Hall, T. F. and W. T. Penfound. 1939b. A phytosociological study of a Nyssa biflora consocies in southeastern Louisiana. Am. Midl. Nat. 22:369-375.

Hall, T. F., and W. T. Penfound. 1943. Cypress-gum communities in the Blue Girth Swamp near Selma, Alabama. Ecology 24:208-217.

Hammack, J. and G. Brown. 1974. Waterfowl and Wetlands: Toward a Bioeconomic Analysis. Johns Hopkins University Press, Baltimore.

Handley, C. O., Jr. 1948. Habitat of the golden mouse in Virginia. J. Mammal. 29:298-299.

Hannan, H. H. 1979. Chemical modifications in reservoir-regulated streams, pp. 75-94. In J. V. Ward and J. A. Stanford (eds.), The Ecology of Regulated Streams. Plenum Press, New York.

Hardin, D. L. 1975. Animal Community Structure of Adirondack Wetlands. MS Thesis, S.U.N.Y. College of Env. Sci. and For., Syracuse. 192 pp.

Hardin, G. 1968. The tragedy of the commons. Science 162:1243-1248.

Hardin, G. and J. Baden (eds.). 1977. Managing the Commons. Freeman, San Francisco.

Harris, W. F., P. Sollins, H. T. Edwards, B. E. Dinger, and H. H. Shugart. 1975. Analysis of carbon flow and productivity in a temperate deciduous forest ecosystem, pp. 116-122. In D. E. Reichle, J. F. Franklin, D. W. Goodall (eds.). Productivity of World Ecosystems Symposium. National Academy of Sciences, Washington, D.C.

Haveman, R. 1972. The Economic Performance of Public Investments. Johns Hopkins University Press, Baltimore

Haveman, R. 1970. The Economics of the Public Sector. John Wiley and Sons, Inc., New York.

Hawk, G. M. and D. B. Zobel. 1974. Forest succession on alluvial landforms of the McKenzie River valley, Oregon. Northwest Sci. 43:245-265.

Heede, B. H. 1977. Case Study of a Watershed Rehabilitation Project: Alkali Creek, Colorado. USDA Forest Serv. Res. Pap. RM-189.

Hehnke, M. and C. P. Stone. 1978. Value of riparian vegetation to avian populations along the Sacramento River system, pp. 228-235. In R. R. Johnson and J. F. McCormick (tech. coord.). Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Heinselman, M. L. 1970. Landscape evolution, peatland types, and the environment in the Lake Agassiz Peatlands Natural Area, Minnesota. Ecol. Monogr. 40:235-261.

Heller, V. J. 1978. The Effects of Stream Alteration and Associated Land Use Changes on Riparian Avifauna in Southcentral Oklahoma. M.S. Thesis, Okla. State Univ., Stillwater. 180 pp.

Hinschberger, M. S. 1978. Occurrence and Relative Abundance of Small Mammals Associated with Riparian and Upland Habitats Along the Columbia River. M. S. thesis, Oregon State Univ., Corvallis. 87 p.

Hirsch, A. and C. A. Segelquist. 1978. Position paper: protection and management of riparian ecosystems: activities and views of the U.S. Fish and Wildlife Service, pp. 344-352. In R. R. Johnson and J. F. McCormick, (tech. coord.). Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12.

Hobbie, J. E. and G. E. Likens. 1973. Output of phosphorus, dissolved organic carbon, and fine particulate carbon from Hubbard Brook watersheds. Limnol. Oceanogr. 18:734-742.

Hodkinson, I. D. 1975. Dry weight loss and chemical changes in vascular plant litter of terrestrial origin, occurring in a beaver pond ecosystem. J. Ecol. 63:131-142.

Holden, P. B. 1979. Ecology of riverine fishes in regulated stream systems with emphasis on the Colorado River, pp. 57-74. In J. V. Ward and J. A. Stanford (eds.), The Ecology of Regulated Streams. Plenum Press, New York.

Hooper, R. G. 1967. The Influence of Habitat Disturbances on Bird Populations. M.S. Thesis, Virginia Polytechnic Inst. and State Univ., Blacksburg. 134 pp.

Horton, G. I. and M. K. Causey. 1979. Woodcock movements and habitat utilization in central Alabama. J. Wildl. Manage. 43:414-420.

Horton, J. S. 1972. Management problems in phreatophyte and riparian zones. J. Soil Water Conserv. 27: 57-61.

Hoslett, S. A. 1961. Effects of floods on mammal distribution. Proc. Iowa Acad. Sci. 68:260-263.

Hosner, J. F. and L. S. Minckler. 1963. Bottomland hardwood forests of southern Illinois--regeneration and succession. Ecology 44:29-41.

Hough, A. F. 1936. A climax forest community on East Tionesta Creek in northwestern Pennsylvania. Ecology 17:9-28.

Hubbard, J. P. 1971. The Summer Birds of the Gila Valley, New Mexico. Nemouria No. 2.

Hunt, R. L. 1979. Removal of Woody Streambank Vegetation to Improve Trout Habitat. Tech. Bull. No. 115, Wisconsin Dept. of Natural Resources, Madison. 36 pp.

Hurd, R. M. 1971. Annual tree-litter production by successional forest stands, Juneau, Alaska. Ecology 52: 881-884.

Ingles, L. G. 1950. Nesting birds of the willow-cottonwood community in California. Auk 67:325-332.

Irizarry, R. A. 1969. The Effects of Stream Alteration in Idaho. Idaho Fish and Game Dept., Federal Aid in Fish and Wildlife Restoration Job Completion Report, Project F55-4-2, Boise. 26 pp.

Iseri, K. T. and W. B. Langbein. 1974. Large Rivers of the United States. USDI Geological Survey Circular 686. 10 pp.

Iverson, S. L. and B. N. Turner. 1973. Ecological notes on Manitoba _Napaeozapus insignis_. Can. Field-Nat. 87: 145-149.

Iverson, S. L., R. W. Seabloom, and J. M. Hnatiuk. 1967. Small mammal distributions across the prairie-forest transition of Minnesota and North Dakota. Am. Midl. Nat. 78:188-197.

Jaffe, A. 1980. Benefit-cost analysis and multi-objective evaluation of federal water projects. Harvard Environmental Law Review 4:58-85.

Jahn, L. 1978. Values of riparian habitats to natural ecosystems, pp. 161-165. _In_ R. Johnson and J. McCormick, (tech. coord.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12. Washington, D.C.

Jahns, R. H. 1947. Geologic Features of the Connecticut Valley, Mass., as Related to Recent Floods. U.S. Geol. Surv. Water-Supply Pap. 996. 158 p.

Janssen, C. R. 1967. A floristic study of forests and bog vegetation, northwestern Minnesota. Ecology 48: 751-765.

Johnson, F. L. and D. T. Bell. 1976. Plant biomass and net primary production along a flood-frequency gradient in the streamside forest. Castanea 41:156-165.

Johnson, R. R. 1978. The lower Colorado River: a western system, pp. 41-55. _In_ R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain Wetlands and Other Ecosystems, Proceedings of the Symposium. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Johnson, R. R., L. T. Haight, J. M. Simpson. 1977. Endangered species vs. endangered habitats: a concept, pp. 68-79. _In_ R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-43. Ft. Collins, Colo.

Johnson, R. R. and J. M. Simpson. 1971. Important birds from Blue Point cottonwoods, Maricopa County, Arizona. Condor 73:379-380.

Johnson, W. C., R. L. Burgess, and W. R. Keammerer. 1976. Forest overstory vegetation on the Missouri River floodplain in North Dakota. Ecol. Monogr. 46:59-84.

Johnston, D. W. and E. P. Odum. 1956. Breeding bird populations in relation to plant succession on the Piedmont of Georgia. Ecology 37:50-61.

Jordan, M. and G. E. Likens. 1975. An organic carbon budget for an oligotrophic lake in New Hampshire, U.S.A. Int. Ver. Theor. Angew. Limnol. Verh. 19:994-1003.

Keammerer, W. R., W. C. Johnson, and R. L. Burgess. 1975. Floristic analysis of the Missouri River bottomland forests in North Dakota. Can. Field Nat. 89:5-19.

Keller, C., L. Anderson, and P. Tappel. 1979. Fish habitat changes in Summit Creek, Idaho, after fencing the riparian area, pp. 46-52. In O. B. Cope (ed.), Grazing and Riparian/Stream Ecosystems. Trout Unlimited, Denver, Colo.

Kemp, W. M., W. H. B. Smith, H. N. McKellar, M. E. Lehman, M. Homer, D. L. Young, and H. T. Odum. 1977. Energy cost-benefit analysis applied to power plants near Crystal River Florida, pp. 508-543. In C. A. S. Hall and J. W. Day, Jr. Ecosystem Analysis in Theory and Practice: An Introduction with Case Histories. John Wiley and Sons, Inc., New York. 684 pp.

Kennedy, R. S. 1977. Ecological Analysis and Population Estimates of the Birds of the Atchafalaya River Basin in Louisiana. Ph.D. Diss. La. St. Univ., Baton Rouge. 201 pp.

Kessel, B. and T. J. Cade. 1958. Birds of the Colville River, Northern Alaska. Univ. Alaska Biol. Pap. No. 2. 83 pp.

Kirby, R. E. 1975. Wildlife utilization of beaver flowages on the Chippewa National Forest, north-central Minnesota. Loon 47:180-185.

Kitchens, W. M., Jr., J. M. Dean, L. H. Stevenson, and J. H. Cooper. 1975. The Santee Swamp as a nutrient sink, pp. 349-366. In F. G. Howell, J. B. Gentry, M. H. Smith (eds.). Mineral Cycling in Southeastern Ecosystems. ERDA Conf-740513, NTIS, Washington, D.C.

Klopatek, J. M., R. J. Olson, C. J. Emerson and J. L. Jones. 1979. Land use conflicts with natural vegetation in the United States. Environmental Conservation 6: 191-200.

Knetsch, J. and R. Davis. 1977. Comparisons of methods for recreation evaluation, pp. 450-468. In Dorfman and Dorfman (eds.), Economics of the Environment (2nd ed.). W. W. Norton and Co., New York.

Korte, P. A. and L. H. Fredrickson. 1977. Loss of Missouri's lowland hardwood ecosystem. Trans. 42nd N. Am. Wildl. Nat. Resour. Conf. 42: 31-41.

Krenkel, P. A., G. F. Lee, and R. A. Jones. 1979. Effects of TVA impoundments on downstream water quality and biota, pp. 289-306. In J. V. Ward and J. A. Stanford (eds.), The Ecology of Regulated Streams. Plenum Press, New York.

Krieger, L. 1973. What's wrong with plastic trees? Science 179:446.

Krutilla, J. 1967. Conservation reconsidered. American Economic Review 57:777-786.

Krutilla, J. 1975. The use of economics in project evaluation. Trans. 40th N. Am. Wildl. and Nat. Resour. Conf. 40:374-381.

Krutilla, J. 1979. Economics and the environment: a time for taking stock, pp. 5-12. In G. Gorrie, (ed.), Environmental Economics: Papers Presented at the National Conference, Canberra, 29-30 May, 1978. Department of Science and the Environment, Canberra, Australia.

Krutilla, J. and O. Eckstein. 1958. Multiple Purpose River Development: Studies In Applied Economic Analysis. Johns Hopkins University Press, Baltimore.

Krutilla, J. and A. Fisher. 1975. The Economics of Natural Environments: Studies in the Valuation of Commodity and Amenity Resources. Johns Hopkins University Press, Baltimore. 292 pp.

Krutilla, J. and J. Haigh. 1978. An integrated approach to national forest management. Environmental Law 8: 373-416.

Krzysik, A. J. 1979. Resource allocation, coexistence, and the niche structure of a streambank salamander community. Ecol. Monogr. 49:173-194.

Kuchler, A. W. 1964. Manual to Accompany the Map - Potential Natural Vegetation of the Conterminous United States. Amer. Geogr. Soc. Spec. Publ. No. 36. 39 pp.

Kuenzler, E. J., P. J. Mulholland, L. A. Ruley, and R. P. Sniffen. 1977. Water Quality in North Carolina Coastal Plain Streams and Effects of Channelization. Water Resour. Res. Inst., Univ. North Carolina, Rep. No. 127. Raleigh.

Kusler, J. 1978. Strengthening State Wetland Regulations. U.S. Fish and Wildlife Service, FWS/OBS-78/98, Washington, D.C. 147 pp.

Lacey, J. R., P. R. Ogden and K. E. Foster. 1975. Southern Arizona Riparian Habitat: Spatial Distribution and Analysis. Office of Arid Land Studies Bull. 8, Univ. of Arizona, Tucson. 85 pp.

Lagler, K. F. and J. C. Salyer. 1947. Food and habits of the common watersnake, Natrix s. sipedon, in Michigan. Papers of the Michigan Academy of Science Arts and Letters. 31: 169-180.

Landers, J. L., R. J. Hamilton, A. S. Johnson and R. L. Marchinton. 1979. Foods and habitat of black bears in southeastern North Carolina. J. Wildl. Manage. 43:143-153.

Lane, E. W. 1955. The importance of fluvial morphology in hydraulic engineering. Proc. Am. Soc. Civil Engrs. 81:1-17.

Langbein, W. B. and Others. 1949. Annual Runoff in the United States. U.S. Geol. Surv. Circular 52.

Langdon, O. G., J. P. McClure, D. D. Hook, J. M. Crockett, and R. Hunt. 1980. Extent, condition, management, and research needs for bottomland hardwood-cypress forests in the Southeast. Unpublished paper presented at the Bottomland Hardwood Wetland Workshop, Lake Lanier, Georgia. June 1-5, 1980.

Larson, J. 1976. Models for Assessment of Freshwater Wetlands. Publ. No. 32, Water Resources Research Center, Univ. of Mass., Amherst. 91 pp.

Lattman, L. H. 1960. Cross section of a floodplain in a moist region of moderate relief. J. Sedim. Petr. 30: 275-282.

Leitch, J., and L. Danielson. 1979. Social, Economic and Institutional Incentives to Drain or Preserve Prairie Wetlands. Economic Report ER79-6. Institute of Agriculture, Forestry and Home Economics, University of Minnesota. St. Paul, Minnesota.

Leitch, J., and D. Scott. 1977. A Selected Annotated Bibliography of Economic Values of Fish and Wildife and Their Habitats. Agricultural Economics Miscellaneous Report No. 27. North Dakota University, Fargo. 132 pp.

Leite, E. A. 1972. A Review and Testing of Techniques for Measuring Farm Game Population Parameters in Riparian and Adjacent Upland Habitats. Ohio Upland Wildlife Ecology Research and Management Project W-103-R-15, Job IV-b, Final Job Report. 57 pp.

Leopold, L. B. and J. P. Miller. 1956. Ephemeral Streams - Hydraulic Factors and Their Relation to the Drainage Net. U.S. Geol. Surv. Prof. Pap. 282A.

Leopold, L. B., M. G. Wolman, and J. P. Miller. 1964. Fluvial Processes in Geomorphology. W. H. Freeman and Co., San Francisco, CA.

Lewke, R. E. 1975. Pre-Impoundment Study of Vertebrate Populations and Riparian Habitat Behind Lower Granite Dam on the Snake River in Southeastern Washington. Ph.D. Diss. Washington State Univ., Pullman. 242 pp.

Liers, E. E. 1951. Notes on the river otter (Lutra canadensis). J. Mammal. 32:1-9.

Likens, G. E., F. H. Bormann, R. S. Pierce, J. S. Eaton, and N. M. Johnson. 1977. Biogeochemistry of a Forested Ecosystem. Springer-Verlag, New York. 146 pp.

Lindauer, I. E. 1978. A comparison of the vegetative communities of the South Platte and Arkansas River drainages in eastern Colorado, pp. 56-72. In W. D. Graul and S. J. Bissell (tech. coord.). Lowland River and Stream Habitat in Colorado: A Symposium, Denver, Colo.

Lindsey, A. A., R. O. Petty, D. K. Sterlin, and W. Van Asdall. 1961. Vegetation and environment along the Wabash and Tippecanoe Rivers. Ecol. Monogr. 31:105-156.

Little, A. D. Inc. 1973. Report on Channel Modifications, Vol. I. Prepared for Council on Environmental Quality, Wash., D.C. 394 pp.

Livingston, R. J. and J. L. Duncan. 1979. Climatological control of a north Florida coastal system and impact due to upland forestry management, pp. 339-381. In R. J. Livingston (ed.), Ecological Processes in Coastal and Marine Systems. Plenum Press, New York.

Livingston, R. J., R. L. Iverson, R. H. Estabrook, V. E. Keys, and J. Taylor, Jr. 1974. Major features of the Apalachicola Bay system: physiography, biota, and resource management. Fla. Sci. 27:245-271.

Lord, R. F. 1933. Type of food taken throughout the year by brook trout in a single Vermont stream with special reference to winter feeding. Trans. Am. Fish. Soc. 63:182-197.

Lugo, A. E., and M. M. Brinson. 1978. Calculations of the value of salt water wetlands, pp. 120-130. In P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland Functions and Values: The State of our Understanding. Am. Water Resour. Assoc., Minneapolis, Minn.

MacArthur, R. H. 1964. Environmental factors affecting bird species diversity. Am. Natur. 98:387-397.

MacArthur, R. H. and J. W. MacArthur. 1961. On bird species diversity. Ecology 42:594-598.

MacDonald, B. C. and C. P. Lewis. 1973. Geomorphic and Sedimentologic Processes of Rivers and Coast, Yukon Coastal Plain. Geol. Surv. of Canada. Cat. No. R72-12173. Information Canada, Ottawa.

MacDonald, P. O., W. E. Frayer and J. K. Clauser. 1979a. Documentation, Chronology and Future Projections of Bottomland Hardwood Loss in the Lower Mississippi Alluvial Plain, Vol. I: basic report. U.S. Fish and Wildlife Service, Division of Ecological Services, Jackson, Miss. 133 pp.

MacDonald, P. O., W. E. Frayer and J. K. Clauser. 1979b. Documentation, Chronology and Future Projections of Bottomland Hardwood Loss in the Lower Mississippi Alluvial Plain, Vol. II: appendices. U.S. Fish and Wildife Service, Division of Ecological Services, Jackson, Miss. 295 pp.

Maddock, T., Jr. 1975. Background summary of basic information, pp. 1-18. In K. J. Sabol (ed.), National Conference on Flood Plain Management (24-25 July 1974, Wash., D.C.). National Association of Conservation

Districts, League City, Texas. 261 pp.

Maher, W. J. 1959. Habitat distribution of birds breeding along the upper Kaolak River, northern Alaska. Condor 61:351-368.

Maki, T. E., A. J. Weber, D. W. Hazel, S. C. Hunter, B. T. Hyberg, D. M. Flinchum, J. P. Lollis, J. B. Kognstad, and J. D. Gregory. 1980. Effects of Stream Channelization on Bottomland and Swamp Forest Ecosystems. Water Resour. Res. Inst. Univ. of North Carolina, Rep. No. 147, Raleigh.

Malcolm, R. L. and W. H. Durum. 1976. Organic Carbon and Nitrogen Concentrations and Annual Organic Carbon Load of Six Selected Rivers of the United States. U.S. Geol. Surv. Water Sup. Pap. 1817-F, GPO, Washington, D.C.

Malecki, R. A. and J. T. Eckler. 1980. An Environmental Assessment of the Wayne County Water Management Program in the Melvin Brook Watershed. Agric. Water Manage. Prog. No. 80-1. NYS College of Agric. and Life Sciences, Cornell Univ., Ithaca. 30 pp.

Mann, K. H., R. H. Britton, A. Kowalczewski, T. J. Lack, C. P. Mathews, and I. McDonald. 1972. Productivity and energy flow at all trophic levels in the River Thames, England, pp. 579-596. In Z. Kajak and A. Hillbricht-Ilkowska (eds.), Productivity Problems of Freshwater. IBP/UNESCO Symp. Kazimierz Dolny, Poland.

Mansfield, G. R. 1939. Flood Deposits of the Ohio River, January-February 1937--A Study of Sedimentation. U.S. Geol. Surv. Water-Supply Pap. 838. pp. 693-733.

Martin, A. C., H. S. Zim, and A. L. Nelson. 1951. American Wildlife and Plants: A Guide to Wildlife Food Habits. McGraw-Hill Book Co., Inc., N.Y. 500 pp.

Martin, M. and K. Casavant. 1980. An initial evaluation of the benefits and costs of navigation as an alternative use of Columbia River basin water. Environmental Law 10:331-348.

Martin, R. O. R. and R. L. Hanson. 1966. Reservoirs in the United States. U.S. Geological Survey Water-Supply Paper 1838. 115 pp.

Martin, S. C. 1979. Evaluating the impacts of cattle grazing on riparian habitats in the national forests of Arizona and New Mexico, pp. 35-38. In O. B. Cope (ed.), Grazing and Riparian/Stream Ecosystems. Trout Unlimited, Denver.

Marzolf, G. R. 1978. The Potential Effects of Clearing and Snagging on Stream Ecosystems. U.S. Fish and Wildlife Service FWS/OBS-78/14, Washington, D.C. 31 pp.

Maser, C., R. G. Anderson, K. Cromack, Jr., J. T. Williams, and R. E. Martin. 1980. Dead and down woody material, pp. 78-95. In J. W. Thomas (tech. ed.), Wildlife Habitats in Managed Forests-the Blue Mountains of Oregon and Washington. USDA For. Serv. Agric. Handbk. No. 533.

Mathews, C. P. and A. Kowalczewski. 1969. The decomposition of leaf litter in the River Thames. J. Ecol. 57:543-552.

Mathisen, J. and A. Richards. 1978. Status of great blue herons on the Chippewa National Forest. Loon 59: 104-106.

McCarley, H. 1959. The effect of flooding on a marked population of Peromyscus. J. Mammal. 40:57-63.

McDowell, W. H. and S. G. Fisher. 1976. Autumnal processing of dissolved organic matter in a small woodland stream ecosystem. Ecology 57:561-569.

McElveen, J. D. 1978. The Effect of Different Edge Types and Habitat Sizes on the Distribution of Breeding Birds in Northern Florida. M.S. Thesis, Univ. of Florida, Gainesville. 47 pp.

McEvoy, T. J., T. L. Sharik, and D. W. Smith. 1980. Vegetative structure of an Appalachian oak forest in southwestern Virginia. Am. Midl. Nat. 103:96-105.

McGill, R. R., Jr. 1975. Land Use Changes in the Sacramento River Riparian Zone, Redding to Colusa. Calif. Dept. of Water Resources, Northern District. 23 pp.

McGill, R. R., Jr. 1979. Land use change in the Sacramento River riparian zone, Redding to Colusa, an update - 1972 to 1977. Calif. Dept. of Water Resources, Northern District. 34 pp.

McGilvrey, F. B. 1968. A Guide to Wood Duck Production Habitat Requirements. U.S. Bureau of Sport Fish. Resour. Publ. 60. 32 pp.

McKern, J. L. 1976. Inventory of Riparian Habitats and Associated Wildlife Along the Columbia and Snake Rivers, Vol. I: Summary Report. U.S. Army Corps of Engineers, North Pacific Division, Walla Walla, Wash. 100 pp.

McNatt, R. 1978. Possible strategies for preservation of the San Pedro River riparian community, pp. 201-206. In R. R. Johnson and J. F. McCormick (tech. coord.). Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12. Washington, D.C.

Meehan, W. R. 1970. Some Effects of Shade Cover on Stream Temperature in Southeast Alaska. USDA For. Serv. Res. Note PNW-113. 9 p.

Mengel, R. M. 1965. The Birds of Kentucky. Ornithol. Monogr. 3, A.O.U., The Allen Press, Lawrence, Kan. 581 pp.

Merritt, R. W. and D. L. Lawson. 1978. Leaf litter processing in floodplain and stream communities, pp. 93-105. In R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Michny, F. J., D. Boos, and F. Wernette. 1975. Riparian Habitats and Avian Densities Along the Sacramento River. Calif. Dept. Fish & Game, Admin. Rep. 75-1. 42 p.

Miller, D. H. and L. L. Getz. 1977. Factors influencing local distribution and species diversity of forest small mammals in New England. Can. J. Zool. 55:806-814.

Minshall, G. W. 1978. Autotrophy in stream ecosystems. Bioscience 28: 767-771.

Mishan, E. J. 1976 (2nd ed.). Cost-Benefit Analysis. Praeger Specal Studies, New York.

Mississippi Game and Fish Commission. 1978. Mississippi Statewide Wildlife Habitat and Population Inventory. Federal Aid Project W-56, Mississippi Game and Fish Comm. 172 pp.

Mitsch, W. J. 1978. Interactions between riparian swamp and a river in southern Illinois, pp. 63-72. In R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tec. Rep. WO-12, Washington, D.C.

Mitsch, W. J., C. L. Dorge, and J. R. Wiemhoff. 1977. Forested wetlands for water resource management in southern Illinois. Res. Rep. No. 132. Water Resour. Center, Univ. Illinois, Urbana.

Mitsch, W. J., C. L. Dorge, and J. R. Wiemhoff. 1979a. Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois. Ecology 60:1116-1124.

Mitsch, W. J., W. Rust, A. Behnke, and L. Lai. 1979b. Environmental Observations of a Riparian Ecosystem During Flood Season. Res. Rep. No. 142. Water Resour. Center, Univ. Illinois, Urbana.

Morris, L. A. 1977. Evaluation, Classification, and Management of the Floodplain Forests of South Central New York. M.S. thesis. SUNY Coll. Environ. Sci. For. Syracuse, N.Y. 181 pp.

Morris, L. A., A. V. Mollitor, K. J. Johnson, and A. L. Leaf. 1978. Forest management of floodplain sites in the northeastern United States, pp. 236-242. In R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Mulholland, P. J. 1979. Organic Carbon Cycling in a Swamp-Stream Ecosystem and Export by Streams in Eastern North Carolina. Ph.D. Diss., Univ. of North Carolina, Chapel Hill. 152 pp.

Mulholland, P. J. and E. J. Kuenzler. 1979. Organic carbon export from upland and forested wetland watersheds. Limnol. Oceanogr. 24:960-966.

Nadler, C. T., Jr. 1978. River Metamorphosis of the South Platte and Arkansas Rivers, Colorado. M. S. Thesis, Colorado State Univ., Fort Collins. 151 pp.

Naiman, R. J. and J. R. Sibert. 1978. Transport of nutrients and carbon from the Nanaimo River to its estuary. Limnol. Oceanogr. 23:1183-1193.

Nessel, J. 1978. Distribution and Dynamics of Organic Matter and Phosphorus in a Sewage Enriched Cypress Strand. M.S. Thesis, Univ. of Florida, Gainesville. 159 pp.

New, J. 1972. A nongame bird survey in Indiana on selected portions of natural and channeled streams in the Kankakee and Elkhart River basins. Thirty-fourth Midwest Fish and Wildlife Conf., Des Moines, Iowa. 10 pp.

Noble, R. E. and R. B. Hamilton. 1975. Bird populations in even-aged loblolly pine forests of southeastern Louisiana. Proc. S.E. Assoc. Game Fish Comm. Conf. 29:441-450.

Noon, B. R. and K. P. Able. 1978. A comparison of avian community structure in the northern and southern Appalachian Mountains, pp. 98-117. In R. M. DeGraaf (tech. coord.), Management of Southern Forests for Nongame Birds, Proceedings of the Workshop. USDA For. Serv. Gen. Tech. Rep. SE-14., Asheville, N.C. 176 pp.

North, D. and R. Miller. 1978 (4th ed.). The Economics of Public Issues. Harper and Row, New York.

Novak, J. K. and R. D. Estes. 1974. Summer food habits of the black sculpin, Cottus baileyi, in the Upper South Fork Holston River drainage. Trans. Am. Fish. Soc. 103:270-276.

Odum, E. P. 1950. Bird populations of the Highlands (North Carolina) Plateau in relation to plant succession and avian invasion. Ecology 31:587-605.

Odum, E. P. 1971. Fundamentals of Ecology. W. B. Saunders Co., Philadelphia. 574 pp.

Odum, E. P. 1979. Rebuttal of "Economic Value of Natural Coastal Wetlands: A Critique". Coastal Zone Management Jour. 5:231-244.

Odum, H. T. 1956. Primary production of flowing waters. Limnol. Oceanogr. 1:102-117.

Odum, H. T. 1970. Energy values of water resources, pp. 56-64. Proceedings 19th Southern Water Resource and Pollution Control Conference, Duke Univ.

Odum, H. T. 1971. Environment, Power, and Society. Wiley-Interscience, New York. 331 pp.

Odum, H. T. 1973. Energy, ecology, and economics. Ambio 2:220-227.

Odum, H. T. 1977. Energy, value, and money, pp. 174-196. In C.A.S. Hall and J. W. Day, Jr. (eds.), Ecosystem Modeling in Theory and Practice: An Introduction with Case Histories. John Wiley and Sons, New York. 684 pp.

Odum, H. T. 1978. Energy analysis, energy quality, and environment, pp. 55-87. In M. W. Gilliland. (ed.), Energy Analysis: A New Public Policy Tool. AAAS Selected Symposium, Washington, D.C.

Odum, H. T., and E. C. Odum. 1976. Energy Basis for Man and Nature.

McGraw Hill, New York, New York. 297 pp.

Ohmart, R. D., W. O. Deason and C. Burke. 1977. A riparian case history: the Colorado River, pp. 35-47. In R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-43.

O'Neil, C. P., J. E. deSteiguer, and G. W. North. 1975. Trend Analysis of Vegetation in Louisiana's Atchafalaya River Basin. U.S. Geol. Surv., NSTL, Bay St. Louis, Miss.

Oregon Department of Fish and Wildlife. 1980. Elk Habitat Use in an Unlogged and Logged Forest Environment. Wildlife Research Report No. 9, Oregon Dept. of Fish and Wildlife, Eugene.

Organ, J. A. 1961. Studies of the local distribution, life history, and population dynamics of the salamander Genus Desmognathus in Virginia. Ecol. Monogr. 31:189-220.

Orser, P. N. and D. J. Shure. 1972. Effects of urbanization on the salamander Desmognathus fuscus. Ecology 53:1148-1154.

Ortego, B., R. B. Hamilton, and R. E. Noble. 1976. Bird usage by habitat types in a large freshwater lake. Proc. S. E. Assoc. Game Fish Comm. Conf. 30:627-633.

Outhet, D. N. 1974. Progress report on bank erosion studies in the Mackenzie River Delta, N.W.T., pp. 298-345. In Hydrologic Aspects of Northern Pipeline Development. Environmental-Social Program, Northern Pipeline. Cat. No. R57-1/1974. Information Canada, Ottawa.

Page, T. 1977. Conservation and Economic Efficiency. Johns Hopkins Univ. Press, Baltimore.

Palmer. R. S. 1962. Handbook of North American Birds. Vol. I. Yale Univ. Press, New Haven, Conn. 567 pp.

Parnell, J. F. 1969. Habitat relations of the Parulidae during spring migration. Auk 86:505-521.

Pasch, R. W., and J. H. Lyford, Jr. 1972. The food habits of two species of Cottus occupying the same habitat. Tran. Am. Fish. Soc. 101:377-381.

Pase, C. P. and E. F. Layser. 1977. Classification of riparian habitat in the Southwest, pp. 5-9. In R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-43, Ft. Collins, Colo.

Patrick, W. H., Jr. and M. E. Tusneem. 1972. Nitrogen loss from flooded soil. Ecology 53:735-737.

Patton, D. R. 1977. Riparian research needs. p. 80-83. In R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-43, Ft. Collins, Colo.

Peck, J. W. 1974. Migration, food habits, and predation on yearling coho salmon in a Lake Michigan tributary and bay. Trans. Am. Fish. Soc. 103:10-14.

Penfound, W. T. and T. F. Hall. 1939. A phytosociological analysis of a tupelo gum forest near Huntsville, Alabama. Ecology 20:358-364.

Peskin, H., and E. Seskin, eds. 1975. Cost-Benefit Analysis and Water Pollution Policy. The Urban Institute, Washington, D.C.

Peters, J. C., and W. Alvord. 1964. Man-made channel alterations in thirteen Montana streams and rivers. Trans. 29th N. Amer. Wildl. Nat. Resour. Conf. 29:93-101.

Phipps, R. L. 1979. Simulation of wetlands forest vegetation dynamics. Ecological Modelling 7:257-288.

Pionke, H. B. and G. Chesters. 1973. Pesticide-sediment-water interactions. J. Environ. Qual. 2:29-45.

Plater, Z. 1974. The takings issue in a natural setting: floodlines and the police power. Texas Law Review. 52: 201-256.

Platts, W. S. 1979. Livestock grazing and riparian/stream ecosystems--an overview, pp. 39-45. In O. B. Cope (ed.), Grazing and Riparian/Stream Ecosystems. Trout Unlimited, Denver.

Ploskey, G. R. and R. M. Jenkins. 1980. Inventory of U.S. reservoirs, January 1, 1980. USDI Fish and Wildlife Service, National Reservoir Research Program, Fayetteville, Arkansas. 39 pp.

Ports, M. A. 1979. Occurrence and Density Studies of Nongame Wildlife in Southeastern Kansas. Kansas Fish and Game Commission Report. 55 pp.

Posner, R. 1977 (2nd ed.). Economic Analysis of Law. Little, Brown, and Company. Boston.

Possardt, E. E. and W E. Dodge. 1978. Stream channelization impacts on songbirds and small mammals in Vermont. Wildl. Soc. Bull. 6:18-24.

Post, A. and L. R. Mayo. 1971. Glacier dammed lakes and outburst floods in Alaska. U.S. Geol. Surv. Hydrologic Invest. Atlas HA-455.

Prakesh, A., M. A. Rashid, A. Jensen, and D. V. Subba Rao. 1973. Influence of humic substances on the growth of marine phytoplankton: diatoms. Limnol. Oceanogr. 18:516-524.

Prellwitz, D. M. 1976. Effects of Stream Channelization on Terrestrial Wildlife and Their Habitats in the Buena Vista Marsh, Wisconsin. U.S. Fish and Wildlife Service, FWS/OBS-76/25, Washington, D.C. 64 pp.

Prosser, N. S., R. G. Martin, and R. H. Stroud. 1979. Adequacy and accuracy of fish and wildlife impact assessment at Corps of Engineers projects, pp. 384-390. In G. A. Swanson (tech. coord.), The Mitigation Symposium. USDA For. Serv. Gen. Tech. Rep. RM-65, Ft. Collins, Colo.

Putnam, J. A., G. M. Furnival, and J. S. McKnight. 1960. Management and Inventory of Southern Hardwoods. USDA For. Serv., Agric. Handbk. No. 181, Washington, D.C. 102 pp.

Raphael, C. N. and E. Jaworski. 1979. Economic value of fish, wildlife, and recreation in Michigan's coastal wetlands. Coastal Zone Manage. J. 5: 181-194.

Rappole, J. H. and D. W. Warner. 1976. Relationships between behavior, physiology and weather in avian transients at a migratory stopover site. Oecologia 26(2):193-212.

Rashid, M. A. and G. E. Reinson. 1979. Organic matter in surficial sediments in the Miramichi Estuary, New Brunswick, Canada. Estuarine and Coastal Marine Science 8:23-36.

Reiners, W. A. 1972. Structure and energetics of three Minnesota forests. Ecol. Monogr. 42:71-94.

Reppert, R. T., W. Sigleo, E. Stakhiv, L. Messman, ad C. Meyers. 1979. Wetland Values: Concepts and Methods for Wetlands Evaluation. Research Report 79-R1. Institute for Water Resources U.S. Army Corps of Engineers. Ft. Belvoir, Va.

Rice, E. L. 1965. Bottomland forests of north-central Oklahoma. Ecology 46:708-713.

Roberts, W. G., J. G. Howe, and J. Major. 1977. A survey of riparian forest flora and fauna of California, pp. 3-19. In A. Sands (Ed.), Riparian Forests in California: Their Ecology and Conservation. Institute of Ecology Publ. No. 15, Univ. of Calif., Davis.

Robertson, P. A., G. T. Weaver, and J. A. Cavanaugh. 1978. Vegetation and tree species patterns near the northern terminus of the southern floodplain forest. Ecol. Monogr. 48: 249-267.

Robinson, T. W. 1965. Introduction, Spread and Areal Extent of Saltcedar (Tamarix) in the Western States. U.S. Geol. Surv. Prof. Pap. 491-A. 12 pp.

Rodgers, W., Jr. 1980. Benefits, costs, and risks: oversight of health and environmental decisionmaking. The Harvard Environmental Law Review 4: 191-226.

Rosen, S. 1977. Cost-benefit analysis, judicial review and the national environmental policy act. Environmental Law 7:363-381.

Rowe, P. B. 1963. Streamflow increases after removing woodland-riparian vegetation from a southern California watershed. J. Forestry 61: 365-370.

Ruffer, D. G. 1961. Effect of flooding on a populaton of mice. J. Mammalogy 42(4):494-502.

Russell, D. M. 1967. A Survey of Streambank Wildlife Habitat in Kentucky. Kentucky Department of Fish and Wildlife Resources, Pittman-Robertson Game Management Technical Series No. 16. 51 pp.

Ryder, R. A. and A. Ryder (ed.). 1978. Thirtieth winter bird-population study. Am. Birds 32:22-48.

Ryder, R. A. and A. Ryder (ed.). 1979. Thirty-first winter bird-population study. Am. Birds 33:18-53.

Sage, B. L. 1974. Ecological distribution of birds in the Atigun and Sagavanirktor River valleys, Arctic Alaska. Canadian Field-Naturalist 88:281-291.

Sagoff, M. 1974. On preserving the natural environment. Yale Law Journal 84:125.

Sands, A. (Ed.). 1977. Riparian Forests in California: Their Ecology and Conservation. Institute of Ecology Publ. No. 15, Univ. of Calif., Davis. 122 pp.

Sands, A. 1978. Public involvement in riparian habitat protection - a California case history, pp. 216-227. In R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

Scharpenseel, H. W., C. Ronsani, and F. Pietig. 1968. Comparative age determinations on different humic-matter fractions, pp. 67-73. In Isotopes and Radiation in Soil Organic-Matter Studies. Internat. Atomic Energy Agency, Vienna.

Schladweiler, J. L. and G. L. Storm. 1969. Den use by mink. J. Wildl. Manage. 33:1025-1026.

Schlesinger, R. B. 1979. Natural removal mechanisms for chemical pollutants in the environment. Bioscience 29:95-101.

Schmelz, D. V. and A. A. Lindsey. 1965. Size-class structure of old-growth forests in Indiana. For. Sci. 11: 258-264.

Schmidt, C. G. 1976. Summer Birds of the San Juan Valley, New Mexico. N. Mex. Ornithol. Soc. Publ. No. 4.

Schumm, S. A. and R. W. Lichty. 1963. Channel widening and floodplain construction along Cimarron River in southwestern Kansas. U.S. Geol. Surv. Prof. Pap. 352-D.

Schwartz, C. W. and E. R. Schwartz. 1959. The Wild Mammals of Missouri. Univ. Missouri Press and Missouri Conserv. Comm. 341 pp.

Scott, J. 1980. The beleaguered blues. Colorado Outdoors 29:1-3.

Seki, H., K. V. Stephens, and T. R. Parsons. 1969. The contribution of allochthonous bacterial and organic materials from a small river to a semi-enclosed sea. Arch. Hydrobiol. 66:37-47.

Shabman, L. 1979. Mitigation planning under principles and standards, pp. 196-202. In G. Swanson (tech. coord.), The Mitigation Symposium. USDA For. Serv. Gen. Tech. Rep. RM-65. Ft. Collins, Colo.

Shabman, L. 1980. Economic incentives for bottomland conversion: The role of public policy and programs. Paper Presented to the 45th North American Widlife Conference, Miami Beach, Fla. March 22-26.

Shabman, L. and S. Batie. 1978. Economic value of natural coastal wetlands: a critique. Coastal Zone Manage. J. 4:232-247.

Shabman, L. and S. Batie. 1979. Estimating the Economic Value of Coastal Wetlands. Sea Grant Project Paper VPI-SG-79-08. Dept. of Agricultural Economics, Virginia Polytechnic Institute and State University. Blacksburg, Va.

Shabman, L., S. Batie and C. Mabbs-Zeno. 1979. The Economics of Wetland Preservation in Virginia. Sea Grant Project Paper VPI-SG-79-07. Dept. of Agricultural Economics, Virginia Polytechnic Institute and State University, Blacksburg, Va.

Shaw, H. and J. Jett. 1959. Mourning Dove and White-winged Dove Nesting in the Gila River Bottom Between Gillespie Dam and the Junction of the Salt and Gila Rivers, Maricopa County, Arizona. Arizona Game & Fish Dept. Project W-53-R-10 Special Report, Phoenix. 10 pp.

Shaw, S. P. and C. G. Fredine. 1956. Wetlands of the United States. U.S. Fish and Wildife Service, Circular 39. Washington, D.C. 67 pp.

Short, C. and M. Schamberger. 1979. Habitat analysis methodology as applied to plains riparian habitats, pp. 47-54. In Great Plains Agricultural Council, Forestry Committee (Program Committee), Riparian and Wetland Habitats of the Great Plains: Proceedings of the 31st Annual Meeting. Great Plains Agric. Counc. Publ. No. 91, USDA For. Serv. Fort Collins, Colo. 88 pp.

Sigafoos, R. S. 1964. Botanical Evidence of Floods and Flood-plain Deposition. U.S. Geol. Surv. Prof. Pap. 485-A. 35 pp.

Simons, D. B. 1979. Effects of stream regulation on channel morphology, pp. 95-111. In J. V. Ward and J. A. Stanford (eds.), The Ecology of Regulated Streams. Plenum Press, New York.

Simons, D. B., P. F. Lagasse, Y. H. Chen, and S. A. Shumm. 1975. The River Environment--a Reference Document. CER 75-76 DBS-PFL-YHC-SAS-14. Engineering Research Center, Colorado State Univ., Fort Collins. Prepared for U.S., Fish and Wildlife Service, Twin Cities, Minn.

Simpson, M. B., Jr. 1969. The prothonotary warbler in the Carolina Piedmont. Chat 33:31-37.

Skogerboe, G. V. 1973. Agricultural impact on water quality in western rivers, Chapter 12. In H. W. Shen. Environmental Impact on Rivers. H. W. Shen Publ., Fort Collins, Colorado.

Smith, D. G. 1976. Effect of vegetation on lateral migration of anastomosed channels of a glacier meltwater river. Bull. Geol. Soc. Amer. 87: 857-860.

Smith, K. G. 1977. Distribution of summer birds along a forest moisture gradient in an Ozark watershed. Ecology 58:810-819.

Sollers, S. C. 1973. Substrate conditions, community structure, and succession in a portion of the floodplain of Wissahickon Creek. Bartonia 42: 24-42.

Spencer, J. S., Jr. 1979. Status of timber inventories in the northeastern Great Plains, pp. 37-42. In Great Plains Agricultural Council Forestry Committee (Program Committee), Riparian and Wetland Habitats of the Great Plains: Proceedings of the 31st Annual Meeting. Great Plains Agric. Counc. Publ. No. 91. USDA For. Serv. Ft. Collins, Colo. 88 pp.

Stauffer, D. F. and L. B Best. 1980. Habitat selection by birds of riparian communities: evaluating effects of habitat alterations. J. Wildl. Manage. 44(1):1-15.

Stearns, F. 1951. The composition of the sugar maple-hemlock-yellow birch association in northern Wisconsin. Ecology 32:245-265.

Stebbins, R. C. 1966. A Field Guide to Western Reptiles and Amphibians. Houghton Mifflin Co., Boston, Mass. 279 pp.

Steenhof, K. 1978. Management of Wintering Bald Eagles. U.S. Fish and Wildlife Service FWS/OBS-78/79. 60 pp.

Sternitzke, H. S. 1976. Impact of changing land use on Delta hardwood forests. J. Forestry 74:25-27.

Sternitzke, H. S. and J. F. Christopher. 1970. Land clearing on the lower Mississippi valley. Southern Geographer 10:63-66.

Stevens, L. E., B. T. Brown, J. M. Simpson, and R. R. Johnson. 1977. The importance of riparian habitat to migrating birds, pp. 156-164. In R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-42, Washington, D.C.

Stewart, R. and J. Krier. 1978 (2nd ed.). Environmental Law and Policy. Little, Brown and Co., Boston.

Stewart, R. E. 1949. Ecology of a nesting red-shouldered hawk population. Wilson Bull. 61:26-35.

Stickel, L. F. 1948. Observations on the effect of flood on animals. Ecology 29:505-507.

Stroup, R. L., et al. 1976. Estimation of Amenity Values As Opportunity Costs for Energy Related Water Use in Montana. Report No. 81, Montana Univ. Joint Water Resources Research Center, Bozeman.

Stutzman, K. 1980. Water Resources Planning Under the Fish and Wildlife Coordination Act. U.S. Fish and Wildlife Service, FWS/OBS - 80/44. Washington, D.C.

Swenson, J. E. 1979. Factors affecting status and reproduction of ospreys in Yellowstone National Park. J. Wildl. Manage. 43(3):595-601.

Swift, B. L. 1980. Breeding Bird Habitats in Forested Wetlands of West-Central Massachusetts. M.S. Thesis, Univ. of Mass., Amherst. 90 pp.

Szaro, R. C. 1980. Factors influencing bird populations in southwestern riparian forests, pp. 403-418. In R. M. DeGraaf (tech. coord.), Workshop Proceedings: Management of Western Forests and Grasslands for Nongame Birds. USDA For. Serv. Gen. Tech. Rep. INT-86, Ogden, Utah.

Tan, F. C. and P. M. Strain. 1979. Organic carbon isotope ratios in recent sediments in the St. Lawrence estuary and Gulf of St. Lawrence. Estuarine and Coastal Marine Science 8:213-225.

Tennessee Valley Authority. 1971. Stream Lengths in Tennessee River Basin. TVA Division of Water Quality Planning, Hydraulic Data Branch. 25 pp.

Teskey, R. O. and T. M. Hinckley. 1977a. Impact of Water Level Changes on Woody Riparian and Wetland Communities: Vol. II. The Southern Forest Region. U.S. Fish and Wildlife Service, FWS/OBS-77/59. 46 pp.

Teskey, R. O. and T. M. Hinckley. 1977b. Impact of Water Level Changes on Woody Riparian and Wetland Communities: Vol. III. The central forest region. U.S. Fish and Wildlife Service, FWS/OBS-77/60. 36 pp.

Teskey, R. O. and T. M. Hinckley. 1978a. Impact of Water Level Changes on Woody Riparian and Wetland Communities: Vol. IV. Eastern Deciduous Forest Region. U.S. Fish and Wildlife Service, FWS/OBS-78/87. 54 pp.

Teskey, R. O. and T. M. Hinckley. 1978b. Impact of Water Level Changes on Woody Riparian and Wetland Communities: Vol. V. Northern Forest Region. U.S. Fish and Wildlife Service, FWS/OBS-78/88. 54 pp.

Teskey, R. O. and T. M. Hinckley. 1978c. Impact of Water Level Changes on Woody Riparian and Wetland Communities: Vol. VI. Plains Grassland Region. U.S. Fish and Wildlife Service, FWS/OBS-78/89. 29 pp.

Thomas, J. W. (tech. ed.). 1979. Wildlife Habitats in Managed Forests: the Blue Mountains of Oregon and Washington. USDA For. Serv. Agric. Handbk. No. 533. 512 pp.

Thomas, J. W., G. L. Crouch, R. S. Bumstead, and L. D. Bryant. 1975. Silvicultural options and habitat values in coniferous forests, pp. 272-287. In D. R. Smith (tech. coord.), Proceedings of the Symposium on Management of Forest and Range Habitats for Nongame Birds. USDA For. Serv. Gen. Tech. Rep. WO-1, Washinton, D.C.

Thomas, J. W., R. G. Anderson, C. Maser, and E. L. Bull. 1979a. Snags, pp. 60-77. In J. W. Thomas (tech. ed.), Wildlife Habitats in Managed Forests - the Blue Mountains of Oregon and Washington. USDA For. Serv. Agric. Handbk. No. 533.

Thomas, J. W., C. Maser, and J. E. Rodiek. 1979b. Wildlife Habitats in Managed Rangelands - the Great Basin of Southeastern Oregon: Riparian Zones. USDA For. Serv. Gen. Tech. Rep. PNW-80. 18 pp.

Thomas, J. W., C. Maser and J. E. Rodiek. 1979c. Wildlife Habitats in Managed Rangelands - the Great Basin of Southeastern Oregon: Edges. USDA For. Serv. Gen. Tech. Rep. PNW-85. 17 pp.

Thomas, M., B. Liu and A. Randall. 1979. Economic Aspects of Wildlife Habitat and Wetlands. (Final Report for Water Resources Council). Midwest Research Inst., Kansas City, Mo.

Thompson, G. E. and R. W. Davies. 1976. Observations on the age, growth, reproduction, and feeding of mountain whitefish (Prosopium williamsoni) in the Sheep River, Alberta. Trans. Am. Fish. Soc. 105:208-219.

Thompson, J. C. 1955. Conditions on irrigated sections in the middle Rio Grande in New Mexico, pp. 27-29. In P. C. Duisberg (ed.), Problems of the Upper Rio Grande: An Arid Zone River. Publ. No. 1. U.S. Commission for Arid Resource Improvement and Development.

Thomsen, B. N. and H. H. Schumann. 1968. Water Resources of the Sycamore Creek Watershed, Maricopa County, Arizona. U.S. Geol. Surv. Water-Supply Pap. 1861. 53 pp.

Tilley, S. G. 1973. Life histories and natural selection in populations of the salamander Desmognathus ochrophaeus. Ecology 54:3-17.

Timbol, A. S. and J. A. Maciolek. 1978. Stream Channel Modification in Hawaii, Part A: Statewide Inventory of Streams, Habitat Factors and Associated Biota. U.S. Fish and Wildlife Service, FWS/OBS-78/16. 157 pp.

Tribe, L. 1974. Ways not to think about plastic trees: new foundations for environmental law. Yale Law J. 83:1315.

Trimble, G. R. 1959. Logging roads in northeastern municipal watersheds. J. Am. Water Works Assoc. 51:407-410.

Trimble, G. R., Jr. and R. S. Sartz. 1957. How far from a stream should a logging road be located. J. Forestry 55:339-341.

Tripp, J. 1977. Tensions and conflicts in federal pollution control and water resource policy. Harvard Journal on Legislation 14:225-280.

Tubbs, A. A. 1980. Riparian bird communities of the Great Plains, pp. 419-433. In R. M. DeGraaf (tech. coord.), Workshop Proceedings: Management of Western Forests and Grasslands for Nongame Birds. USDA For. Serv. Gen. Tech. Rep. INT-86, Ogden, Utah.

Turner, R. E. 1977. Intertidal vegetation and commerical yields of penaeid shrimp. Trans. Am. Fish. Soc. 106: 411-416.

Turner, R. E., S. W. Wood, and H. R. Jitts. 1979. Estuarine influences on a continental shelf plankton community. Science 206:218-220.

Turner, R. M. 1974. Quantitative and Historical Evidence of Vegetation Changes Along the Upper Gila River, Arizona. U.S. Geol. Surv. Prof. Pap. 655-H, Washington, D.C.

Turner, R. M. and M. M. Karpiscak. 1980. Recent vegetation changes along the Colorado River between Glen Canyon Dam and Lake Mead, Arizona. U.S. Geol. Surv. Prof. Pap. 1132.

Turner, R. W. 1966. Effects of flooding on the mouse Peromyscus leucopus Trans. Ill. Acad. Sci. 59:390-391.

Turner, S. F. and H. E. Skibitzke. 1952. Use of water by phreatophytes in 2000-foot channel between Granite Reef and Gillespie Dams, Maricopa County, Arizona. Trans. Am. Geophys. Union. 33:66-72.

U.S. Army Corps of Engineers. 1971. Charles River Massachusetts, Main Report and Attachments. U.S. Army Corps of Engineers, New England Division, Waltham, Mass.

U.S. Army Corps of Engineers. 1976. Natural Valley Storage: A Partnership with Nature. New England Division, U.S. Army Corps of Engineers, Waltham, Mass.

U.S. Army Corps of Engineers. 1978. The Streambank Erosion Control Evaluation and Demonstration Act of 1974: Interim Report to Congress. U.S. Army Corps of Engineers, Washington, D.C. 12 pp.

U.S. Department of Agriculture. 1980. Policy on Fish and Wildlife. USDA Secretary's Memorandum No. 2019, Washington, D.C. 6 pp.

USDA Conservation Needs Inventory Committee. 1971. Basic Statistics - National Inventory of Soil and Water Conservation Needs, 1967. USDA Statistical Bulletin No. 461, Washington, D.C. 211 p.

USDA Soil Conservation Service. 1971. Watershed projects involving channelization work-approved and pending, pp. 1621-1749. In House of Representatives, Stream Channelization: Hearings before the Subcommittee of the Committee on Government Operations, 92nd Congress, 1st session, Part 3., Appendix 15.

USDA Soil Conservation Service. 1972. Report on the Inventory of Channel Work - 1972. Unpublished data, USDA Soil Conservation Service, Washington, D.C.

USDA Soil Conservation Service. 1975. Inventory Classification of Channel Work. USDA Soil Conservation Service Watersheds Memorandum 118.

USDA Soil Conservation Service. 1978. National Erosion Inventory Estimates. USDA Soil Conservation Service, Washington, D.C.

USDA Soil Conservation Service. 1980. Report on the Inventory of Channel Work - 1979. Unpublished data, USDA Soil Conservation Service, Washington, D.C.

USDI Bureau of Land Management. 1979. Floodplain management, Section 7221. In USDI Bureau of Land Management Manual. Washington, D.C. 7 pp.

USDI Bureau of Land Management. 1980. Summary of Wildlife Statistical Information, Fiscal Year 1979. USDI Bureau of Land Management, Division of Wildlife and Endangered Species WO-240. 9 pp.

USDI Geological Survey. 1974. Flows of Large Rivers in the United States. U.S. Geol. Surv., Washington, D.C. 9 pp. (mimeo)

USDI Heritage Conservation and Recreation Service (HCRS). 1979. Missouri National Recreational River Management Plan: Gavins Point Dam, South Dakota, to Ponca State Park, Nebraska. USDI Heritage Cons. and Recreation Serv., Denver, Colo.

U.S. Fish and Wildlife Service. 1980a. Missouri River Stabilization and Navigation Project, Sioux City, Iowa to Mouth: Detailed Fish and Wildlife Coordination Act Report. Prepared by FWS Division of Ecological Services Field Office, North Kansas City, Mo. 80 pp.

U.S. Fish and Wildlife Service. 1980b. Republication of lists of endangered and threatened species and correction of technical errors in final rules. Fed. Reg. 45(99):33768-33781.

U.S. Fish and Wildlife Service. 1980c. Habitat Evaluation Procedures. Ecological Services Manual (100 Series). Washington, D.C.

van Beek, J. L. 1979. Hydraulics of the Atchafalaya basin main channel system. U.S. Environmental Protection Agency, EPA-600/4-79-036. Washington, D.C.

VanCamp, L. F. and C. J. Henny. 1975. The Screech Owl: Its Life History and Population Ecology in Northern Ohio. U.S. Fish and Wildlife Service, North American Fauna No. 71. 65 pp.

van Hylckama, T.E.A. 1980. Weather and Evapotranspiration Studies in a Saltcedar Thicket, Arizona. U.S. Geol. Surv. Prof. Pap. 491-F. Washington, D.C. 78 pp.

Van Velson, R. 1979. Effects of livestock grazing upon rainbow trout in Otter Creek, Nebraska, pp. 53-55. In O. B. Cope (ed.), Grazing and Riparian/ Stream Ecosystems. Trout Unlimited, Denver.

Vogl, R. J. 1980. The ecological factors that produce perturbation-dependent ecosystems, pp. 63-94. In J. Cairns, Jr., The Recovery Process in Damaged Ecosystems. Ann Arbor Science, Ann Arbor, Mich.

Walters, M. A., R. O. Teskey, and T. M. Hinckley. 1980a. Impact of Water Level Changes on Woody Riparian and Wetland Communities: Vol. VII. Mediterranean Region, Western Arid, and Semi-Arid Region. U.S. Fish and Wildlife Service FWS/OBS-78/93. 47 pp.

Walters, M. A., R. O. Teskey, and T. M. Hinckley. 1980b. Impact of Water Level Changes on Woody Riparian and Wetland Communities: Vol. VIII. Pacific Northwest and Rocky Mountain Regions. U.S. Fish and Wildlife Service FWS/OBS-78/94.

Wauer, R. H. 1977. Significance of Rio Grande riparian systems upon the avifauna, pp. 165-174. In R. R. Johnson and D. A. Jones (tech. coord.), Importance, Preservation and Management of Riparian Habitat: A Symposium. USDA For. Serv. Gen. Tech. Rep. RM-43, Ft. Collins, Colo.

Webb, W. L., J. R. Bart, and C. A. Komarek. 1972. Technical Report on Wildlife Resources. St. Lawrence-Eastern Ontario Shoreline Study. SUNY College Environ. Sci. Forestry. Syracuse, N.Y. 63 pp.

Webster, C. G. and F. B. McGilvrey. 1966. Providing brood habitat for wood ducks, pp. 70-75. In Wood Duck Management and Research: A Symposium, Wildlife Management Institute, Washington, D.C.

Welcomme, R. L. 1979. Fisheries Ecology of Floodplain Rivers. Longman, N.Y. 317 pp.

Weller, M. W. 1978. Management of freshwater marshes for wildlife, pp. 267-284. In R. E. Good, D. F. Whigham and R. L. Simpson (eds.), Freshwater Wetlands: Ecological Processes and Management Potential. Academic Press, New York.

Wetzel, R. M. 1958. Mammalian succession on midwestern floodplains. Ecology 39:262-271.

Wharton, C. H. 1970. The Southern River Swamp: A Multiple-Use Environment. Georgia State Univ., Atlanta. 48 pp.

Wharton, C. H. 1978. The Natural Environments of Georgia. Georgia Dept. Nat. Resour., Atlanta, Ga. 227 pp.

Wharton, C. H. and M. M. Brinson. 1978. Characteristics of southeastern river systems, pp. 32-40. In R. R. Johnson and J. F. McCormick (tech. coord.), Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA For. Serv. Gen. Tech. Rep. WO-12, Washington, D.C.

White, H. C. 1953. The Eastern Belted Kingfisher in the Maritime Provinces. Fish. Res. Board Canada Bull. 97. 44 pp.

Whitmore, R. C. 1975. Habitat ordination of passerine birds of the Virgin River Valley, southwestern Utah. Wilson Bull. 87:65-74.

Whittaker, R. H., F. H. Bormann, G. E. Likens, and T. G. Siccama. 1974. The Hubbard Brook ecosystem study: forest biomass and production. Ecol. Monogr. 44:233-254.

Wilcox, L. V. 1955. Analysis of salt balance and salt burden data on the Rio Grande, pp. 39-44. In P. C. Duisberg (ed.), Problems of the Upper Rio Grande: An Arid Zone River. Publ. No 1. U.S. Comm. for Arid Resource Improvement and Development.

Williams, G. P. 1978. Historical perspective of the Platte Rivers in Nebraska and Colorado, pp. 11-41. In W. D. Graul and S. J. Bissell (tech. coord.), Lowland River and Stream Habitat in Colorado: A Symposium. Colo. Div. Wildl., Colo. Audubon Counc., Denver, Colo.

Willson, M. F. and S. W. Carothers. 1979. Avifauna habitat islands in the Grand Canyon. Southwestern Nat. 24: 563-576.

Wilson, W. J., E. H. Buck, G. F. Player, and L. D. Dreyer. 1977. Winter Water Availability and Use Conflicts as Related to Fish and Wildlife in Arctic Alaska: A Synthesis of Information. U.S. Fish and Wildlife Service, FWS/OBS-77/06. 243 pp.

Wolf, C., Jr. 1979. A theory of non-market failures. The Public Interest 55:114-133.

Wolfe, C. B., Jr. and J. D. Pittillo. 1977. Some ecological factors influencing the distribution of Betula nigra L. in western North Carolina. Castanea 42:18-30.

Wolman, M. G. and L. B. Leopold. 1957. River Flood Plains: Some Observations on Their Formation. U.S. Geol. Surv. Prof. Pap. 282-C. 107 pp.

Wooding, J. 1973. Census of the breeding birds of the Roaring Creek watershed. Colo. Field Ornithol. 18:36-41.

Yarbro, L. A. 1979. Phosphorus Cycling in the Creeping Swamp Floodplain Ecosystem and Exports from the Creeping Swamp Watershed. Ph.D. Diss., Univ. of North Carolina, Chapel Hill. 231 pp.

Yeager, L. E. 1949. Effect of permanent flooding in a river-bottom timber area. Bull. Ill. Nat. Hist. Surv., 25:33-65.

Yeager, L. E. and H. G. Anderson. 1944. Some effects of flooding and waterfowl concentrations on mammals of a refuge area in central Illinois. Am. Midl. Nat. 31:159-178.

Zimmerman, J. L. and J. L. Tatschl. 1975. Floodplain birds of Weston Bend, Missouri River. Wilson Bull. 87:196-206.

Zimmermann, R. C. 1969. Plant Ecology of an Arid Basin; Tres Alamos-Redington Area, Southeastern Arizona. U.S. Geol. Surv. Prof. Pap. 485-D. 47 pp.

Zwank, P. J., R. D. Sparrowe, W. R. Porath, and O. Torgerson. 1979. Utilization of threatened bottomland habitats by white-tailed deer. Wildlife Soc. Bull. 7:226-232.

BIBLIOGRAPHIC DATA SHEET	1. Report No. FWS/OBS-81/17	2.	3. Recipient's Accession No.
4. Title and Subtitle RIPARIAN ECOSYSTEMS: Their Ecology and Status			5. Report Date 6.
7. Author(s) M. M. Brinson, B. L. Swift, R. C. Plantico and J. S. Barclay			8. Performing Organization Rept. No.
9. Performing Organization Name and Address National Water Resources Analysis Group Eastern Energy and Land Use Team U.S. Department of the Interior Fish and Wildlife Service Route 3, Box 44 Kearneysville, WV			10. Project/Task/Work Unit No. 11. Contract/Grant No.
12. Sponsoring Organization Name and Address Department of the Interior, U.S. Fish and Wildlife Service Office of Biological Services, Eastern Energy and Land Use Team Route 3, Box 44 Kearneysville, WV 25430			13. Type of Report & Period Covered Final report 14.

15. Supplementary Notes

16. Abstracts

This report is a comprehensive review and synthesis of information on the ecological values of riparian ecosystems. Chapters are included on the following topics: status of riparian ecosystems in the U.S.; ecological functions and properties of riparian ecosystems (e.g., geomorphology, primary productivity, nutrient cycling, hydrology, etc.); importance of riparian ecosystems to fish and wildlife; and considerations in valuation (ecologic and economic) of riparian ecosystems. The report is a technical summary of extensive literature reviews and personal communications with Federal and State agencies. It was developed by the U.S. Fish and Wildlife Service to document the natural values and importance of riparian ecosystems for consideration in water resources planning and development.

17. Key Words and Document Analysis. 17a. Descriptors

floodplains, rivers, streams, wetlands, terrestrial habitats, geomorphology, primary productivity, alteration of flow

17b. Identifiers/Open-Ended Terms

ecosystems, wildlife, riparian plants, ecology, evaluation, economics, environmental effects

17c. COSATI Field/Group 48B, G

18. Availability Statement	19. Security Class (This Report)	21. No. of Pages
Unlimited	UNCLASSIFIED	xvi + 154
	20. Security Class (This Page) UNCLASSIFIED	22. Price