



The Structure and Functioning of Riparian and Aquatic Ecosystems of the Colorado Plateau– Conceptual Models to Inform Monitoring

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LIST OF FIGURES.....	iii
LIST OF TABLES	vii
I. INTRODUCTION AND BACKGROUND	1
A. Purpose and Content of This Report	1
B. Vital Signs Definition	1
C. Purposes of Conceptual Models	3
D. Physiographic, Hydrologic and Ecological Scope	3
II. STRUCTURAL AND FUNCTIONAL RELATIONSHIPS.....	10
A. Background: The Jenny-Chapin Model of Ecosystem Sustainability.....	10
B. Conceptual Model.....	11
1. Hierarchy of System Drivers.....	14
<i>a. State Factors</i>	14
i. Time, Initial Topographic Relief, Geology, and Global Climate	14
<i>b. Interactive Controls-Physical</i>	15
i. Regional Climatic and Atmospheric Conditions.....	15
ii. Upland Watershed Characteristics	17
iii. Stream-flow Regime	19
iv. Fluvial Geomorphic Processes.....	25
v. Natural Disturbance Regime	28
vi. Flood plain Soils	31
vii. Instream Physical and Chemical Conditions	32
<i>c. Interactive Controls-Biotic Functional Groups</i>	35
i. Background	35
ii. Flood plain Soil Biota	36
iii. Riparian Vegetation	37
<i>d. Other Biotic Components</i>	39
2. Ecosystem Dynamics	43
<i>a. Riparian Ecosystem Dynamics</i>	44
i. Longitudinal-Scale Gradients.....	44
ii. Transverse-Scale Gradients.....	45
<i>b. Aquatic Ecosystem Dynamics</i>	45
i. River Continuum Concept.....	47
ii. Network Dynamics Hypothesis	50
iii. Shifting Habitat Mosaic	50
III. ENVIRONMENTAL MONITORING OF RIPARIAN AND AQUATIC ECOSYSTEMS.....	51
A. Characteristics of Healthy Riparian and Aquatic Ecosystems	51
B. Monitoring to Evaluate the Impact of Stressors	52
1. Key components for monitoring to assess impacts of stressors on riparian systems.....	52
2. Key components for monitoring to assess impacts of stressors on aquatic systems.....	52
IV. PRIMARY STRESSORS AND DEGRADATIONAL PROCESSES	54
A. Overview	54
B. Stressors.....	54
1. Climate Change.....	54
2. Stream-flow Alteration.....	56
<i>a. Stream-flow Depletion</i>	56
<i>b. Reduced Streamflow Variability</i>	58
3. Invasion of Non-native Plant Species	59
4. Livestock Grazing	60

<i>a. On-site Grazing</i>	60
<i>b. Trampling</i>	61
<i>c. Nutrient Enrichment</i>	61
5. Visitor Activities	62
<i>a. Trails and Roads</i>	62
<i>b. Walking in streams (slot canyons)</i>	62
<i>c. Driving up streams or frequent stream crossings</i>	63
6. Fire	63
7. Stream Channel Alteration	64
8. Alteration of Upland Watershed	65
<i>a. Off-site Grazing</i>	65
<i>b. Organic contaminants</i>	65
<i>c. Mining</i>	65
C. Degradational Pathways and Processes	66
V. IMPLICATIONS FOR ECOSYSTEM CLASSIFICATION TO SUPPORT MONITORING	77
A. Overview	77
B. Proposed Stream Classification Framework	78
1. Channel Processes	78
2. Hierarchical Channel Classification	78
3. Differences in Reach-level Responses	80
4. Channel Networks	80
VI. LITERATURE CITED	82

LIST OF FIGURES

- Figure 1. Map showing location of NCPN and SCPN parks in relation to ecoregions. Ecoregion designations follow The Nature Conservancy and NatureServe (Groves et al. 2002), as modified from Bailey (1995). See Table 1 for key to four-letter park codes. 4
- Figure 2. Map showing mean annual precipitation (symbol diameter) and mean seasonal precipitation (symbol segments) for National Weather Service Cooperative Network stations located at or near NPS units of the Southern and Northern Colorado Plateau Networks.. Shaded zone approximates the mean northwestern extent of summer monsoon moisture (from Mitchell 1976). Precipitation data were acquired from the Western Regional Climate Center (<http://www.wrcc.dri.edu/climsum.html>). See Table 1 for key to four-letter park codes. 6
- Figure 3. Distribution of the 67 Geological Survey Cataloging Hydrologic Units (HUCs) contained within the Colorado Plateau physiographic province. 7
- Figure 4. Modified version (a) of the Jenny-Chapin model that serves as the general ecosystem model for the SCPN and NCPN, and (b) the array of stressors affecting SCPN/NCPN ecosystems arranged in relation to their first-order effects. Complex, higher order effects occur as the four major controls interact via ecosystem processes. The circle represents the boundary of the ecosystem. 11
- Figure 5. General conceptual model depicting the structure and functioning of riparian ecosystems. Symbols represent the following: solid, rounded rectangles = state factors or major drivers of ecosystem change and variability; ellipses = interactive controls; and rectangles = biotic components. Arrows indicate functional relationships among components and line weights indicate relative importance of relationships. The model is constrained by global climatic and atmospheric conditions, topography, parent material and potential biota. 12
- Figure 6. General conceptual model describing the structural components and functional relationships that characterize aquatic ecosystems. Symbols represent the following: ellipses = interactive controls; and rectangles = biotic components. Arrows indicate functional relationships among components and line weights indicate relative importance of relationships. The model is constrained by global climatic and atmospheric conditions, topography, parent material and potential biota. 13
- Figure 7. Map depicting regional variations in the estimated maximum intensity of precipitation (mm/hr) that can be expected to occur over a 5-min period with 50 percent probability during any given year at NWS stations located at or near NCPN and SCPN units. Shaded zone approximates the mean northwestern extent of summer monsoon moisture (from Mitchell 1976). Data were acquired from NOAA's Precipitation Frequency Data Server (<http://hdsc.nws.noaa.gov/hdsc/pfds/>). See Table 1 for key to four-letter park codes. 16
- Figure 8. An idealized diagram of a fluvial system featuring: (1) a zone of sediment production (watershed); (2) a zone of transport (rivers and streams); and (3) a zone of deposition (alluvial fans, deltas) From: Schumm (1981). 18

Figure 9. Idealized flow paths of water moving from a watershed to a stream. Adapted from: Dunne (1978) and Ziemer and Lisle (1998). 20

Figure 10. Stream hydrographs. (a) Idealized relationship between stormflow and baseflow components of a stream hydrograph, relative to a discrete rainfall event (Modified from Dunne 1978). (b) Idealized hydrograph of an ephemeral stream of the Colorado Plateau, featuring highly variable and temporally unpredictable peak flows with no baseflow. (c) Idealized hydrograph of an unregulated, large perennial stream of the Colorado Plateau, featuring a variable but temporally predictable snowmelt peak flow..... 20

Figure 11. Mean monthly flows for the Yampa river (from 1922), the Green River (from 1929-1963), and the Green River (after 1963). Flaming Gorge Dam began regulating flows on the Green River after 1963. From: Adair et al. (2002). 21

Figure 12. Representation of how a natural flow regime creates and maintains the ecological integrity of riverine ecosystems. From: Poff et al. (1997). 22

Figure 13. A qualitative classification of stream channels based on pattern (straight, meandering, or braided) and type of sediment load, along with flow and sediment variables and relative stability with regard to average erosional activity. From Schumm (1981). 26

Figure 14. Conceptual model of riparian succession relative to the creation of freshly deposited alluvial surfaces. Solid arrows represent possible successional pathways and dashed arrows represent erosional conversion to alluvium. After: Johnson (1992). 29

Figure 15. According to the intermediate disturbance hypothesis, intermediate levels of disturbance result in the highest levels of diversity. Adapted from Hutchinson (1961). 31

Figure 16. Planview maps of the superficial geology of the Green River in Dinosaur National Park, reflecting differences in surfaces supporting riparian vegetation (post-dam flood plain, Intermediate bench, and Cottonwood-boxelder terrace) between a) fan-eddy dominated, and b) restricted meander, channel types. From: Grams and Schmidt (2002). 44

Figure 17. Schematic of the ecological structure of riparian vegetation in relation to the stream channel, surface water, the alluvial water table, and upland vegetation, for a) an alluvial valley, and b) a confined bedrock canyon. Modified from: Goodwin et al. (1997). 46

Figure 18. Model of stream ecosystem identifying major physical and biological components (Miller et al. 2003, adapted from Davis et al. 2001). Aquatic communities are structured by both biotic and abiotic factors, the relative influence depending upon the environmental setting and disturbance events..... 48

Figure 19. The river continuum concept (adapted from Vannote et al. 1980). CPOM is coarse particulate organic matter, FPOM is fine particulate organic matter, DOM is dissolved organic matter..... 49

Figure 20. Conceptual ecological model depicting the processes by which primary stressors to the Salt Creek system of CANY affect riparian and communities. From Schelz (2001). 55

Figure 21. (a) Annual maximum discharge of the Green River near Greendale, Utah and (b) the Green River near Jensen, Utah. The horizontal lines represent the 2-, 5-, 10-, and 25-year recurrence flows. At the Greendale gage, changes in peak flows between 1895-1929 and 1930-1963 reflect the influence of climate (Allred and Schmidt 1999), whereas flows after 1963 reflect the effect of flow regulation from Flaming Gorge Dam. The influence of the Yampa in restoring peak flows to the Green River, can be see by comparing post-1963 flows at the Greendale gage with those at the Jensen gage, located downstream of the Yampa River confluence. From: Grams and Schmidt (2002). 57

Figure 22. Ecosystem dynamics model describing degraded conditions (B-E) characteristic of riparian and aquatic ecosystems of the Colorado Plateau. Ecological factors and processes responsible for various degradational pathways (numbered arrows) are described in subsequent conceptual process models. 69

Figure 23. Conceptual model illustrating the processes by which moderate streamflow depletion, resulting from water management and drought can interactively lead to changes in the structure and functioning of riparian and aquatic ecosystems. Symbols are as follows: hexagon = degraded condition; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components. 70

Figure 24. Conceptual model illustrating the processes by which reduced stream-flow variability, resulting from water management and drought, interactively lead to the dominance of non-native tamarisk and related changes in riparian and aquatic ecosystems. Symbols are as follows: hexagon = degraded condition; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components. 72

Figure 25. Conceptual model illustrating the processes by which on-site grazing, roads and trails, and off-site climate factors, land-clearing, wildfire, grazing, roads and trails, interactively lead to the process of channel incision and severe bank erosion. Symbols are as follows: hexagon = degraded condition; ellipses = interactive controls; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components. 73

Figure 26. Conceptual model illustrating the processes by which channel instability resulting from riparian and upland wildfires interactively lead to degradative changes in riparian and aquatic ecosystems. Symbols are as follows: hexagon = degraded condition; ellipses = interactive controls; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components. 74

Figure 27. Conceptual model illustrating the processes by which stream-flow depletion resulting from water management activities and drought interactively lead to terrestrialization of the riparian zone and associated changes in aquatic and riparian ecosystems. Symbols are as follows: hexagon = degraded condition; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components. 75

Figure 28. Conceptual model illustrating the processes by which heavy off- and on-site grazing and trampling by large herbivores, roads and trails and heavy rain events interactively lead to dewatering of the channel and flood plain. Symbols are as follows: hexagon = degraded condition; ellipses = interactive controls; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components..... 76

Figure 29. The influence sediment supply, stream transport capacity, and the direct and indirect effects of vegetation on stream channel morphology. From: Montgomery and Buffington (1998).. 79

Figure 30. Hierarchical levels of channel classification, reflecting progressively finer spatial scales from the Geomorphic Province to Channel Reach and Channel Units. Adapted from: Montgomery and Buffington (1998). 79

Figure 31. Idealized valley segment channel morphologies, including (a) colluvial, (b) confined alluvial, (c) unconfined alluvial, and (d) bedrock channels. Adapted from: Montgomery and Buffington (1993). 81

LIST OF TABLES

Table 1. Distribution of the 67 Geological Survey Cataloging Hydrologic Units (HUCs) contained within the Colorado Plateau physiographic province. 2

Table 2. Fluvial system variables over geologic time scales (millions of years). Modified from: Schumm (1981). 14

I. INTRODUCTION AND BACKGROUND

A. Purpose and Content of This Report

This report presents conceptual ecological models describing the structure and functioning of aquatic and riparian ecosystems¹ of the Colorado Plateau and adjacent physiographic provinces. These models have been developed to support the Northern and Southern Colorado Plateau Networks (NCPN and SCPN) of the National Park Service's Inventory and Monitoring Program (I&M Program). As part of the I&M Program, the NCPN and SCPN are tasked with identifying a suite of "vital signs" for use in the long-term monitoring of NPS resources in 35 units located in Utah, Arizona, Colorado, New Mexico, and Wyoming (Table 1). The SCPN and NCPN are working closely together to coordinate planning and implementation of their respective monitoring programs (Miller et al. 2003).

This report starts with background information concerning vital signs, the intended purposes of conceptual models, and the geographical and ecological scope of the report. Following this background section, a general conceptual model and literature review are presented in section two, which characterizes important functional relationships among biotic and abiotic components of riparian and aquatic ecosystems. In the third section, physical and biological processes, typical of healthy riparian and aquatic ecosystems, are described and used to identify key elements of a monitoring program designed to detect the effects of anthropogenic stressors on these systems. Regionally important anthropogenic stressors are briefly described in section four, followed by an ecosystem dynamics model depicting degraded conditions, commonly observed in riparian and aquatic ecosystems throughout the region, relative to unimpaired conditions. Conceptual models are then used to illustrate how key ecological processes are linked to various, and sometimes interrelated, degradational pathways. The fifth and final section presents a discussion of stream geomorphic classification systems, including how a hierarchical stream classification system could be used to increase the efficiency of a long-term monitoring program by stratifying the large diversity of Colorado Plateau streams into a more manageable set of characteristic stream types. Stream types would be based, in part, on fluvial geomorphic settings, which influence the relative susceptibility of different reach types to various degradational processes.

B. Vital Signs Definition

As defined by the NPS, vital signs are a subset of physical, chemical, and biological elements, and park ecosystem processes that are selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values. The elements and processes that are monitored are a subset of the total suite of natural resources that park managers are directed to preserve "unimpaired for future generations," including water, air, geological resources, native plants and animals, and the various ecological, biological, and physical processes that act on those resources. Vital signs may occur at any level of organization including landscape, community, population, or genetic level, and may be compositional (referring to the variety of elements in the system), structural (referring to the

¹An *ecosystem* is a spatially explicit unit of the Earth that includes all of the organisms, along with all components of the abiotic environment within its boundaries (Likens 1992, cited by Christensen et al. 1996:670). *Ecosystem structure* refers to the types, amounts, and spatial arrangement of biotic and abiotic components of an ecosystem. *Ecosystem functioning* refers to the flow of energy and materials through the arrangement of biotic and abiotic components of an ecosystem (includes processes such as primary production, trophic transfer from plants to animals, nutrient cycling, water dynamics and heat transfer). In a broad sense, ecosystem functioning includes two components: ecosystem resource dynamics and ecosystem stability (Díaz and Cabido 2001).

Table 1. Distribution of the 67 Geological Survey Cataloging Hydrologic Units (HUCs) contained within the Colorado Plateau physiographic province.

Network / Park	Code	State	Size (ha)	Elevation (m)	Ecoregion
Southern Colorado Plateau Network					
Aztec Ruins National Monument	AZRU	NM	130	1705 - 1764	Colorado Plateau
Bandelier National Monument	BAND	NM	13,254	1626 - 3081	Southern Rocky Mts.
Canyon de Chelly National Monument	CACH	AZ	37,448	1687 - 2336	Colorado Plateau / Arizona-New Mexico Mts.
Chaco Culture National Historic Park	CHCU	NM	14,090	1832 - 2096	Colorado Plateau
El Malpais National Monument	ELMA	NM	46,559	1950 - 2554	Arizona-New Mexico Mts.
El Morro National Monument	ELMO	NM	518	2183 - 2304	Arizona-New Mexico Mts.
Glen Canyon National Recreation Area	GLCA	AZ/UT	505,868	930 - 2319	Colorado Plateau
Grand Canyon National Park	GRCA	AZ	493,050	348 - 2798	Colorado Plateau
Hubbell Trading Post National Historic Site	HUTR	AZ	65	1920 - 1946	Colorado Plateau
Mesa Verde National Park	MEVE	CO	21,093	1833 - 2613	Colorado Plateau
Navajo National Monument	NAVA	AZ	146	1658 - 2294	Colorado Plateau
Petrified Forest National Park	PEFO	AZ	38,024	1618 - 1891	Colorado Plateau
Petroglyph National Monument	PETR	NM	2,915	1519 - 1838	Arizona-New Mexico Mts.
Rainbow Bridge National Monument	RABR	UT	65	1129 - 1492	Colorado Plateau
Salinas Pueblo Missions National Monument	SAPU	NM	433	1815 - 2058	Arizona-New Mexico Mts.
Sunset Crater Volcano National Monument	SUCR	AZ	1,227	2076 - 2441	Arizona-New Mexico Mts.
Walnut Canyon National Monument	WACA	AZ	1,456	1896 - 2106	Arizona-New Mexico Mts.
Wupatki National Monument	WUPA	AZ	14,350	1304 - 1744	Colorado Plateau
Yucca House National Monument	YUHO	CO	14	1767 - 1805	Colorado Plateau
		TOTAL	1,189,205		
Northern Colorado Plateau Network					
Arches National Park	ARCH	UT	30,966	1206 - 1725	Colorado Plateau
Black Canyon of the Gunnison National Park	BLCA	CO	12,159	1636 - 2752	Southern Rocky Mts.
Bryce Canyon National Park	BRCA	UT	14,502	2000 - 2777	Utah High Plateaus
Canyonlands National Park	CANY	UT	136,610	1140 - 2189	Colorado Plateau
Capitol Reef National Park	CARE	UT	97,895	1182 - 2730	Colorado Plateau
Cedar Breaks National Monument	CEBR	UT	2,491	2461 - 3247	Utah High Plateaus
Colorado National Monument	COLM	CO	8,310	1411 - 2160	Colorado Plateau
Curecanti National Recreation Area	CURE	CO	17,433	1982 - 2898	Southern Rocky Mts.
Dinosaur National Monument	DINO	CO/UT	85,097	1442 - 2747	Utah-Wyoming Rocky Mts.
Fossil Butte National Monument	FOBU	WY	3,318	2012 - 2466	Wyoming Basins
Golden Spike National Historic Site	GOSP	UT	1,107	1317 - 1613	Great Basin
Hovenweep National Monument	HOVE	CO/UT	318	1548 - 2056	Colorado Plateau
Natural Bridges National Monument	NABR	UT	3,009	1702 - 2019	Colorado Plateau
Pipe Spring National Monument	PISP	AZ	16	1495 - 1559	Colorado Plateau
Timpanogos Cave National Monument	TICA	UT	101	1669 - 2452	Utah-Wyoming Rocky Mts.
Zion National Park	ZION	UT	59,900	1112 - 2661	Colorado Plateau
		TOTAL	474,709		

organization or pattern of the system), or functional (referring to ecological processes) (from <http://science.nature.nps.gov/im/monitor/vsm.htm#Definitions>).

C. Purposes of Conceptual Models

In the vital-sign identification and selection process, conceptual models are used to summarize existing knowledge and hypotheses concerning the structure and functioning of park ecosystems. A conceptual model is a diagram of a set of associations among certain factors that are believed to impact or lead to a target condition (Margoluis and Salafsky 1998). An important goal of the model is to depict how natural drivers (e.g., climate) and anthropogenic stressors affect ecosystem structure and functioning. The ability of the monitoring program to detect the ecological effects of anthropogenic stressors is dependent upon interpreting trends in resource condition against the backdrop of intrinsic variation. Hypotheses concerning the effects of anthropogenic stressors on ecosystem structure and functioning must be grounded in an understanding of the relationship between natural drivers and the structure, functioning and dynamics of ecosystems. Ecosystems and their components can be characterized on the basis of far more structural and functional attributes than can be monitored affordably. Thus another important goal of the models is to guide the identification of a parsimonious set of “information-rich” attributes that provides information concerning multiple aspects of ecosystem condition (Noon 2003).

No single conceptual model can satisfy all needs. Spatially explicit applications, such as ecological resource assessments, monitoring design, and landscape-level ecological modeling will ultimately require site-specific models, but the monitoring program also requires generalized ecological models to facilitate communication among scientists, managers, and the public regarding ecosystems and how they are affected by human activities and natural processes. Together, the NPS and USGS have adopted an iterative approach of first developing general conceptual models for broadly defined ecosystem types, and then adapting and refining those models with site-specific data concerning abiotic constraints, land- and water-use history, current condition, and specific patterns of ecosystem dynamics. Models presented in this report are necessarily generalized to circumscribe the diversity of riparian and aquatic ecosystems found in SCPN and NCPN units.

Previous NPS reports established a framework for the conceptual modeling effort associated with the NCPN and SCPN monitoring programs (see Evenden et al. 2002, Thomas et al. 2003, Miller et al. 2003). The overall conceptual framework presented in these earlier documents provides a basis for the material included in this report, but it is not repeated here in detail.

D. Physiographic, Hydrologic and Ecological Scope

Given the distribution of parks included in the NCPN and SCPN, the geographic extent of this report ranges from northern Utah and southwestern Wyoming, southward through Utah and western Colorado, to north-central Arizona and New Mexico (Figure 1). This region is centered on the Colorado Plateau physiographic province, but also includes portions of the Wyoming Basin, Southern and Central Rocky Mountains, the Utah High Plateaus, and the Basin and Range physiographic provinces (Hunt 1974). Generally, the region is arid to semi-arid, but steep topographic gradients and complex terrain contribute to considerable spatial variability in temperature and the distribution of precipitation. Park elevations range from 348 m at the lower end

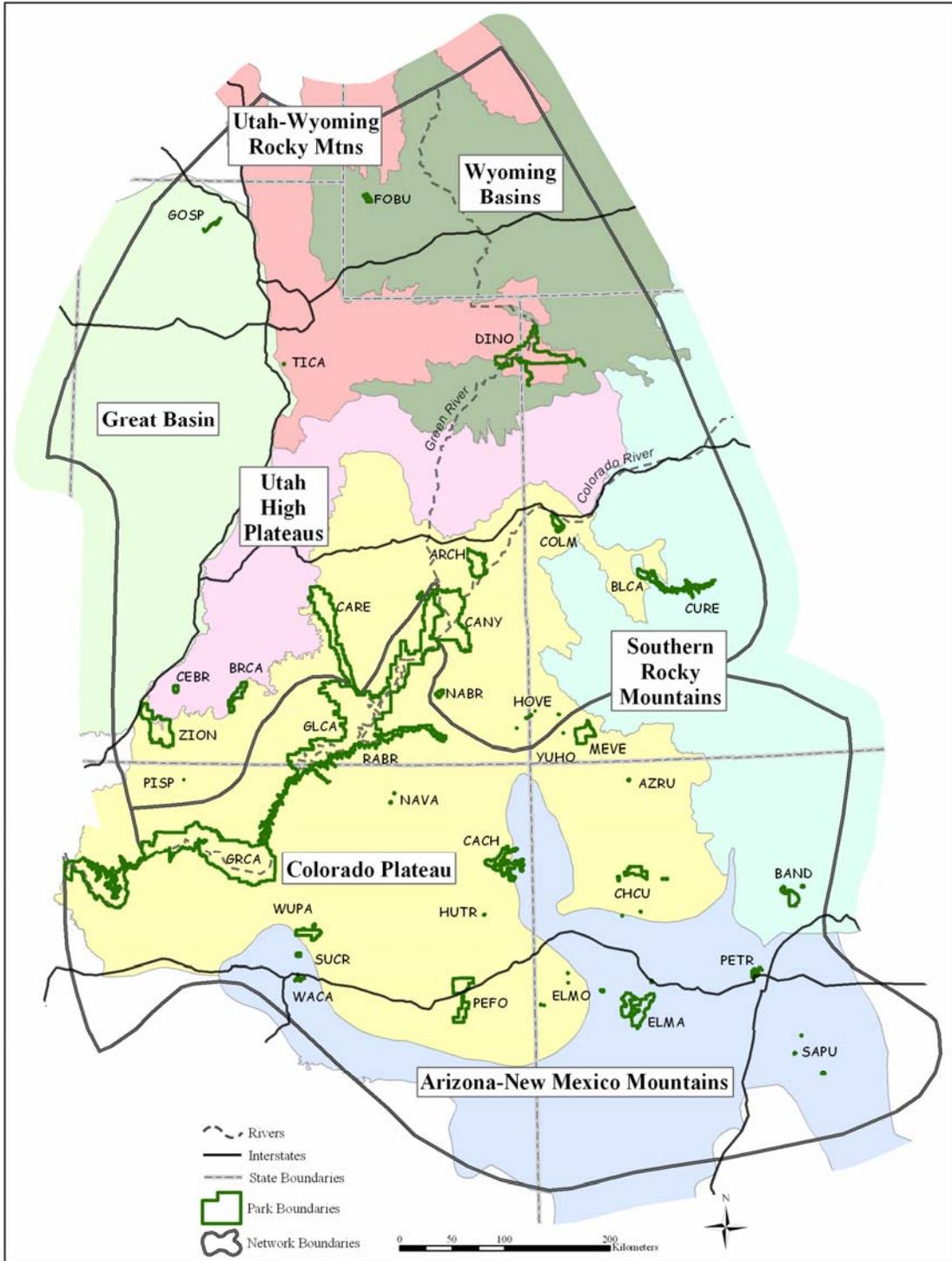


Figure 1. Map showing location of NCPN and SCPN parks in relation to ecoregions. Ecoregion designations follow The Nature Conservancy and NatureServe (Groves et al. 2002), as modified from Bailey (1995). See Table 1 for key for four-letter park codes.

of the Colorado River corridor in Grand Canyon (GRCA) to 3247 m at Cedar Breaks (CEBR) in southwestern Utah (Table 1). [Four-letter codes for park names will be used throughout this report. Table 1 provides a key.] Mean annual precipitation (MAP) at National Weather Service (NWS) Cooperative Network stations located in or near NCPN and SCPN parks ranges from 162 mm at Page, Arizona (GLCA), to 752 mm at CEBR (Figure 2).

The Colorado Plateau physiographic province is a thick, relatively horizontal package of sedimentary rocks that has been uplifted more than 2 km during the last 66 million years (Beghoul and Barazangi 1989). The plateau dips to the north and is broadly deformed into large anticlines, synclines, and monoclines. The central Rocky Mountains occur to the northeast and the southern Rocky Mountains occur to the east and southeast. The Basin and Range occur to the south and west. Large streams of the Colorado Plateau arise in the Rocky Mountains, in the Utah High Plateaus that occur in the transition region between the Colorado Plateau, and the Basin and Range province.

Other streams arise from the volcanic island mountains that dot parts of the Plateau or arise from highlands at the Plateau's southern and eastern rim. The Colorado Plateau covers approximately 20 million hectares, across portions of 30 counties in 4 states. It includes all or part of 67 U.S. Geological Survey Cataloging Hydrologic Units (HUCs; Figure 3). A Cataloging HUC is the smallest element in a hierarchy of hydrologic units and may include part or all of a surface drainage basin, a group of basins, or a distinct hydrologic feature (For more information on HUCs see Seaber et al. 1987, or <http://water.usgs.gov/GIS/huc.html>). Based on information from the U.S. Census Bureau, Geography Division 2001 (TIGER/Line Files, Redistricting Census 2000; <http://www.census.gov/geo/www/tiger/>), there are 124,095 stream kilometers within the province, 99.07% of which drain to the Colorado River. In addition to natural perennial and intermittent streams, the region also contains 757 and 1,804 kilometers of perennial and intermittent human-constructed canals, respectively. The aridity of the region is emphasized by the fact that intermittent or ephemeral streams² represent 94% of the total stream length for the region, and the larger in-flowing or through-flowing perennial streams, including the Colorado, Green, Gunnison, San Juan, Dirty Devil, Escalante, and Virgin Rivers derive much of their stream flow from snowmelt in the Rocky Mountains and Utah High Plateaus.

The following is a brief summary of water sources in both the NCPN and SCPN. Based on a detailed review by Lyn Cudlip and Paul von Guerard (Miller et al. 2003), NCPN parks have perennial, intermittent, and ephemeral (water pockets/tinajas) water sources and support groundwater discharges, such as seeps, hanging gardens, and springs. In contrast to many parks in the SCPN, several NCPN parks (BLCA, CURE, CARE, CANY, DINO, and ZION) have large river systems flowing through them. These rivers are major drivers affecting both the physical and biological components of the parks' riparian and aquatic ecosystems (Miller et al. 2003).

² The terms *perennial*, *intermittent* and *ephemeral* refer to the flow characteristics of a stream under average conditions. Although the boundary between these stream types may be fuzzy, distinctions are based on the concepts of influent and effluent flow. Influent streams "lose" surface flow to the groundwater whereas effluent streams "gain" water from groundwater sources. The term *perennial* here applies to those streams that have flow year-round, and retain a base flow even during dry periods. Such streams are predominantly effluent. In contrast, *intermittent* streams have surface flow for only certain times of the year when they receive water from precipitation and/or groundwater. Thus, they may be effluent at certain times of the year and influent at other times. *Ephemeral* streams have surface flow for relatively short periods of time in response to precipitation. Their channels are influent in that the groundwater surface remains below the channel at all times (Gordon et al. 1992).

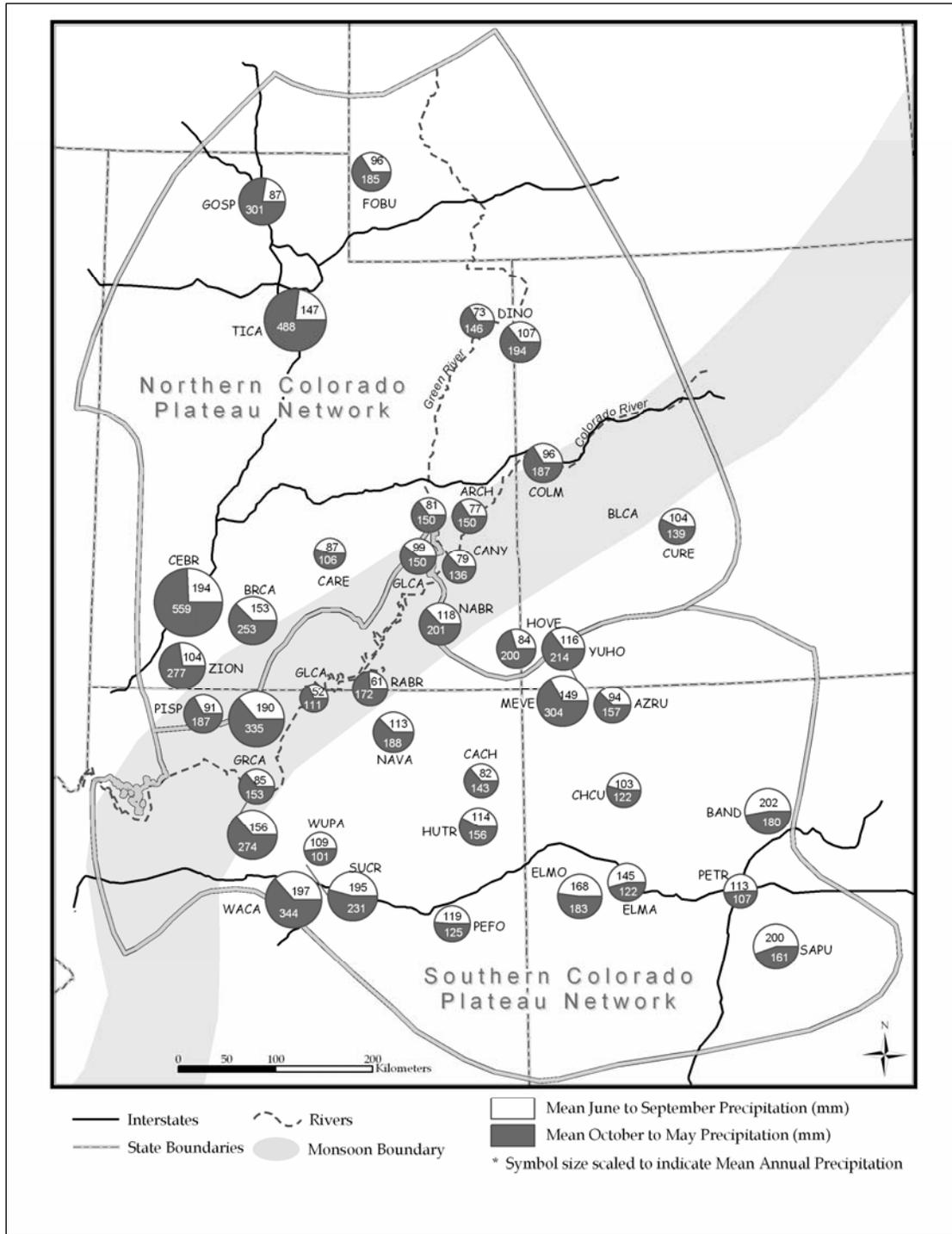


Figure 2. Map showing mean annual precipitation (symbol diameter) and mean seasonal precipitation (symbol segments) for National Weather Service Cooperative Network stations located at or near NPS units of the Southern and Northern Colorado Plateau Networks.. Shaded zone approximates the mean northwestern extent of summer monsoon moisture (from Mitchell 1976). Precipitation data were acquired from the Western Regional Climate Center (<http://www.wrcc.dri.edu/climsum.html>). See Table 1 for key to four-letter park codes.

Riparian/Aquatic Ecosystems

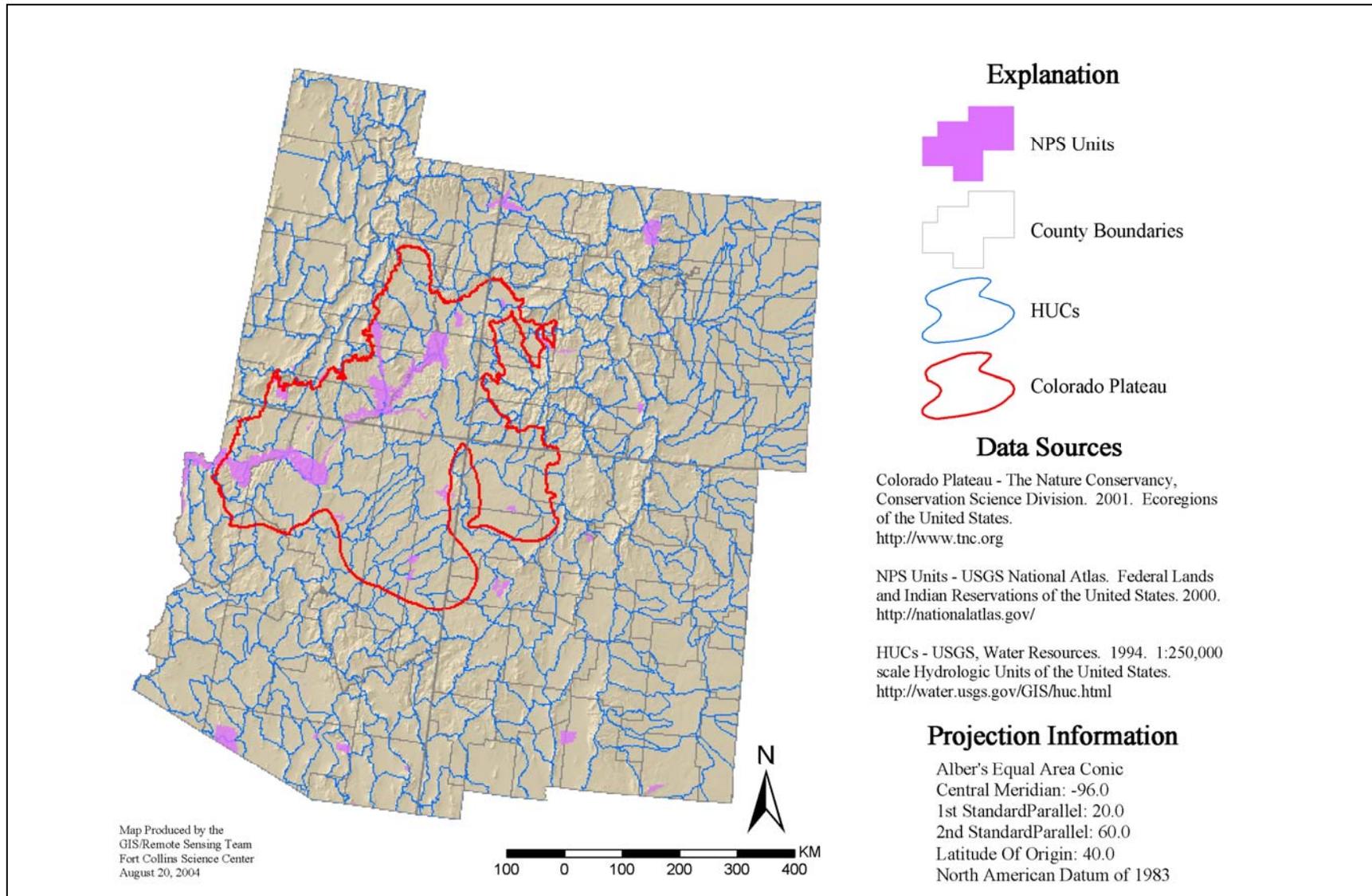


Figure 3. Distribution of the 67 Geological Survey Cataloging Hydrologic Units (HUCs) contained within the Colorado Plateau physiographic province.

Some perennial rivers or streams that flow through or adjacent to NCPN parks include the Colorado River (CANY), the Green River (CANY and DINO), the Yampa River (DINO), the Fremont River and Pleasant, Sulfur, Halls and Oak Creeks (CARE), the American Fork River (TICA), Blue Creek (GOSP) and the Gunnison River (CURE, BLCA), which is dammed by three units of the Colorado River Storage Project and is a gold-medal trout fishery. Several perennial rivers occur in ZION, including the East and North Forks of the Virgin River, the Left and Right Forks of North Creek, Kolob Creek, Orderville Canyon, and La Verkin Creek, Deep Creek, Pine Creek, and Shunes Creek. Intermittent creeks occur at numerous NCPN parks (ARCH, CANY, HOVE, NABR, BRCA, COLO, and FOBU). Springs, seeps, hanging gardens, and tinajas occur in almost all of the NCPN parks.

None of the rivers within the NCPN have Wild and Scenic designation (Miller et al. 2003). Sections of the lower Gunnison River, below Black Canyon, and the Colorado and Green rivers are suitable for recovery of four endangered fish species. CURE is seeking Outstanding National Resource Water (ONRW) designation for several tributaries flowing into reservoirs at the Park. Both Forks of North Creek in ZION are currently listed as impaired waters; however, this may be due to an erroneous classification and they may soon be removed from the list (Miller et al. 2003). The Fremont River is also impaired and 303(d) listed. Red Rock Canyon within BLCA has high ammonia levels, but is not currently on the 303(d) list.

In SCPN parks, most primary water sources are intermittent or ephemeral, flowing during spring runoff or following monsoon rainfall events. Nine SCPN parks (CHCU, ELMA, ELMO, HUTR, PEFO, PETR, SAPU, WACA and WUPA) have no perennial streams and rely entirely on intermittent streams and washes. The Colorado River, which is a large, flow-managed river ecosystem and an essential water source for much of the Southwest, is already the subject of long-term monitoring and research efforts in GRCA and consequently will not be included in the SCPN I & M program (Thomas et al. 2004). Perennial rivers or streams that flow through or adjacent to SCPN parks include the Animas River (AZRU), Rio Grande, Rito de los Frijoles, and Capulin Creek (BAND), Mancos River (MEVE), the Dirty Devil, Escalante, Paria and San Juan Rivers (GLCA) and numerous smaller tributaries including Harris and Twentyfive Mile Washes and Coyote Gulch in GLCA and Bright Angel, Clear, Crystal, Garden, Havasu, and Hermit Creeks in GRCA. Small perennial stream also flow through CACH, NAVA, and RABR.

Springs occur on 14 of the 19 SCPN parks and are viewed as a significant aquatic resource by park managers. Across the Colorado Plateau, springs are ecologically important as critical water and food resources for wildlife, and as important point sources of biodiversity and productivity in otherwise low productivity desert landscapes (Stevens and Nabhan 2002a, b). Arid land springs often function as keystone ecosystems, exerting a disproportionate impact on adjacent ecosystems and regional ecology compared to non-springs habitats (Perla and Stevens 2002). Additional discussion and conceptual models for spring ecosystems on the Colorado Plateau are provided by Stevens and Springer (2004).

Four SCPN parks (AZRU, BAND, GLCA, GRCA) include waters that are 303(d) listed as water quality impaired (Thomas et al. 2004). There are no currently Outstanding Natural Resource Waters (ONRW) designations among SCPN parks. It is anticipated that Level 1 inventories and

initial water quality monitoring activities will support the identification and nomination of potential ONRW designations of NPS waters (Thomas et al. 2004).

In this report we include a continuum of aquatic and riparian environments from small headwater streams to large perennial rivers. Because of their dominance in the regional landscape and importance to ecosystem dynamics of perennial, master streams, we also consider intermittent and ephemeral drainages.

The region under consideration encompasses several distinct floristic areas with different biogeographic histories (McLaughlin 1986, 1989). As a consequence, SCPN and NCPN parks are found in seven broadly-defined ecoregions (Table 1, Figure 1). Numerous regional classification schemes for upland and riparian plant communities have been developed, including those for individual park units (e.g., Romme et al. 1993, Spence et al. 1995). These schemes typically are based on various combinations of floristic, physiognomic, topographic and climatic parameters (Spence et al. 1995). Because a focus area of this report is ecosystem structure and function, this report takes a functional approach that generally corresponds with physiognomy. However, as a tool for linking general concepts with specific ecosystems on the ground, this report also recognizes the classification of *terrestrial ecological systems* (TES) currently being developed by NatureServe (Comer et al. 2003, NatureServe 2003). Comer and colleagues (2003:10) define a TES "...as a group of plant community types that tend to co-occur within landscapes with similar ecological processes, substrates, and/or environmental gradients." Plant community types nested within TES types are association- and/or alliance-level vegetation classification units included in the U.S. National Vegetation Classification (USNVC). USNVC units are used in the USGS-NPS vegetation mapping efforts that are on-going in several NCPN and SCPN parks. The TES classification currently is used by the Southwest USGS Gap Analysis Program and in ecoregional conservation assessments developed by The Nature Conservancy (Groves et al. 2002, Tuhy et al. 2002). Additional information concerning the characteristics of these TES can be found in the accompanying NatureServe report (NatureServe 2003) and in NatureServe's Ecological Systems Database.

In our treatment of riparian ecosystems in this report, we emphasize flood plain and streamside plant communities that are directly influenced by a river or stream through enhanced water supply, flooding, and erosional and depositional processes. Using a hydrogeomorphic classification (HGM) of wetland ecosystems, riparian ecosystems discussed in this report would be considered Riverine Wetlands (Brinson et al. 1995). Such a functional classification emphasizes the primary importance of hydrologic and geomorphic factors responsible for many of the unique structural and functional characteristics of riparian and wetland ecosystems and has been used as a basis for organizing regional classifications of wetland and riparian plant associations (Carsey et al. 2003). Rivers of the southwestern U.S. are inherently dynamic systems that typically experience dramatic variation in stream flow within and between years. Because flow variability tends to increase with increasing aridity and decreasing watershed size, floods tend to have relatively long-lasting influence on channel form and bottomland vegetation in dry regions (Schumm and Lichty 1963, Burkham 1972, Hereford 1984, Friedman et al. 1996, Friedman and Lee 2002). Stream courses of mid- to lower-elevations on the Colorado Plateau are typically dominated by early successional members of the Salicaceae including Willow (*Salix* spp.) and Cottonwood (*Populus* spp.). The composition and structure of regional riparian and

aquatic ecosystems addressed in the report are generally more uniform than corresponding upland ecosystems. This is chiefly attributable to the frequency and intensity of fluvial disturbances, which tend to sustain dominance of early successional, disturbance-dependent species. Of 44 riparian species sampled at 475 randomly selected stream gauging stations across the western U.S., *Salix exigua* and *Populus deltoides* were the first and second most frequently occurring riparian species (Friedman et al. 2005). In fact, riparian corridors, dominated by species of cottonwood and willow (*Populus* and *Salix* spp.), extend from west-central Canada, through the U.S. to northwestern Mexico and represent a striking uniformity of habitat on a continental scale. This is especially important to transcontinental migratory species.

Excluded from the riparian component of this report, are non-riverine wetlands such as springs, seeps, lake margins, and other surface expressions of water that are typically referred to as riparian zones in arid and semi-arid areas of the western U.S. Although these systems have important natural resource values, and may be structurally and compositionally similar to riparian ecosystems, from a functional perspective, these systems have more in common with non-flowing water systems of more humid regions (Brinson et al. 1981). Thus, any management or monitoring efforts should explicitly recognize these fundamental hydrological and geologic differences.

II. STRUCTURAL AND FUNCTIONAL RELATIONSHIPS

This section begins with a brief overview of the ecosystem framework adopted by the SCPN and NCPN. Following this overview, general conceptual models for riparian and aquatic ecosystems and a literature review are presented to characterize important structural functional relationships among biotic and abiotic components of these systems. The section concludes with a review of key theoretical concepts which are commonly used to characterize and interpret riparian and aquatic ecosystem dynamics.

A. Background: The Jenny-Chapin Model of Ecosystem Sustainability

Jenny (1941, 1980) proposed that soil and ecosystem processes are determined by five *state factors* – climate, organisms, relief (topography), parent material, and time since disturbance. Jenny’s state-factor approach has been widely applied as a framework for examining temporal and spatial variations in ecosystem structure and functioning (e.g., Walker and Chapin 1987, Vitousek 1994, Seastedt 2001). Chapin and colleagues (1996) recently extended this framework to develop a set of ecological principles concerning ecosystem sustainability. They defined “...a sustainable ecosystem as one that, over the normal cycle of disturbance events, maintains its characteristic diversity of major functional groups, productivity, and rates of biogeochemical cycling” (Chapin et al. 1996:1016). These ecosystem characteristics are determined by a set of four “interactive controls” – climate, soil-resource supply, major functional groups³ of organisms, and disturbance regime – and these interactive controls both govern and respond to ecosystem attributes. Interactive controls are constrained by the five state factors, which determine the “constraints of place” (Dale et al. 2000). The SCPN and NCPN have adopted a modified version of the Jenny-Chapin model as a general ecosystem framework for informing the development of additional conceptual models and the consideration of vital signs (Figure 4). For vital-signs monitoring, a key aspect of the interactive-control model is the associated

³ *Functional groups* are groups of species that have similar effects on ecosystem processes (Chapin et al. 1996). This concept is generally synonymous with *functional types*.

hypothesis that interactive controls must be conserved for an ecosystem to be sustained. Large changes in any of the four interactive controls are predicted to result in a new ecosystem with different characteristics than the original system (Chapin et al. 1996). For example, major changes in flow regime (e.g., through impoundment or diversion) can greatly affect vegetation establishment and survival patterns, productivity, and competitive interactions among species, and thus can cause significant changes to the structure and functioning of riparian plant communities and higher trophic levels. Changes in riparian vegetation composition and structure can affect the ecosystem’s disturbance regime (e.g., through altered fire frequencies and intensities). These factors and processes in combination can result in an altered system which is fundamentally different from the original system in terms of composition, structure, functioning, and dynamics.

B. Conceptual Model

Expanding on the framework of the Jenny-Chapin model, Figure 5 serves as a general conceptual model describing structural components and functional relationships that characterize riparian ecosystems. Structural components and functional relationships that characterize aquatic ecosystems are depicted in Figure 6. Climatic and atmospheric conditions, soil resources, major functional groups, and disturbance regimes characteristic of riparian and aquatic ecosystems are reviewed in this section. Because natural disturbance regimes in riparian and aquatic ecosystems are inextricably linked to stream flow and fluvial geomorphic processes, the disturbance regime, as an interactive control, is considered here in the context of these two model components.

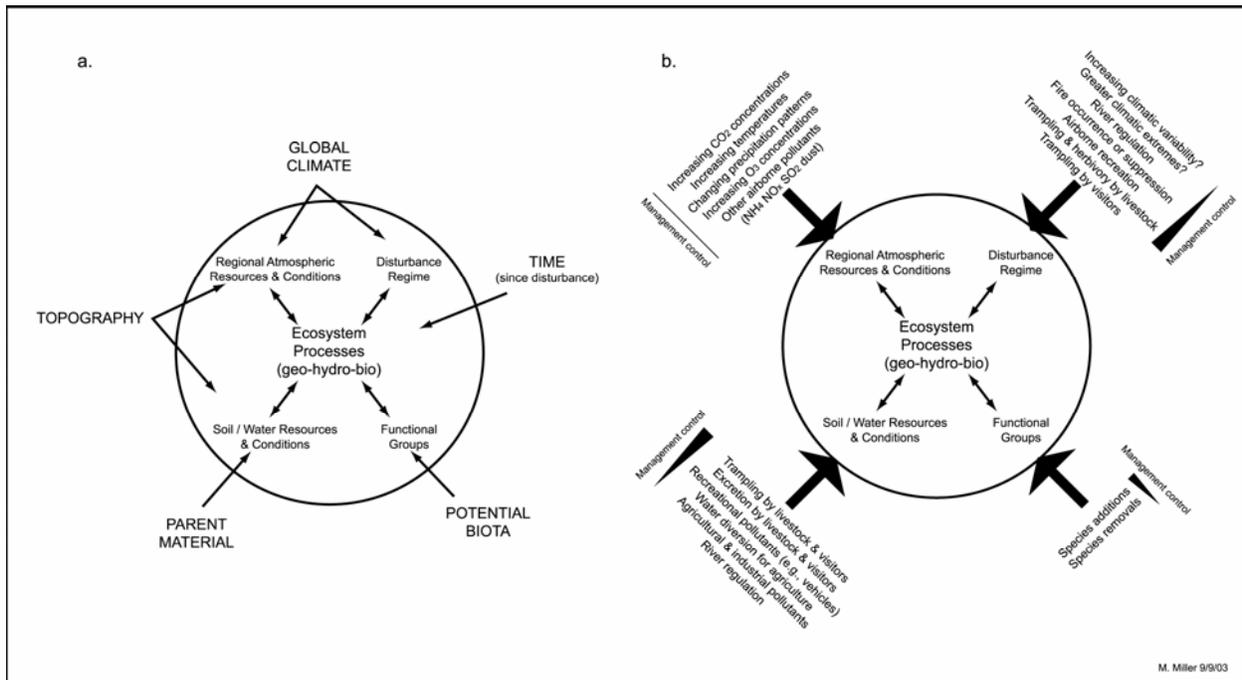


Figure 4. Modified version (a) of the Jenny-Chapin model that serves as the general ecosystem model for the SCPN and NCPN, and (b) the array of stressors affecting SCPN/NCPN ecosystems arranged in relation to their first-order effects. Complex, higher order effects occur as the four major controls interact via ecosystem processes. The circle represents the boundary of the ecosystem.

Riparian/Aquatic Ecosystems

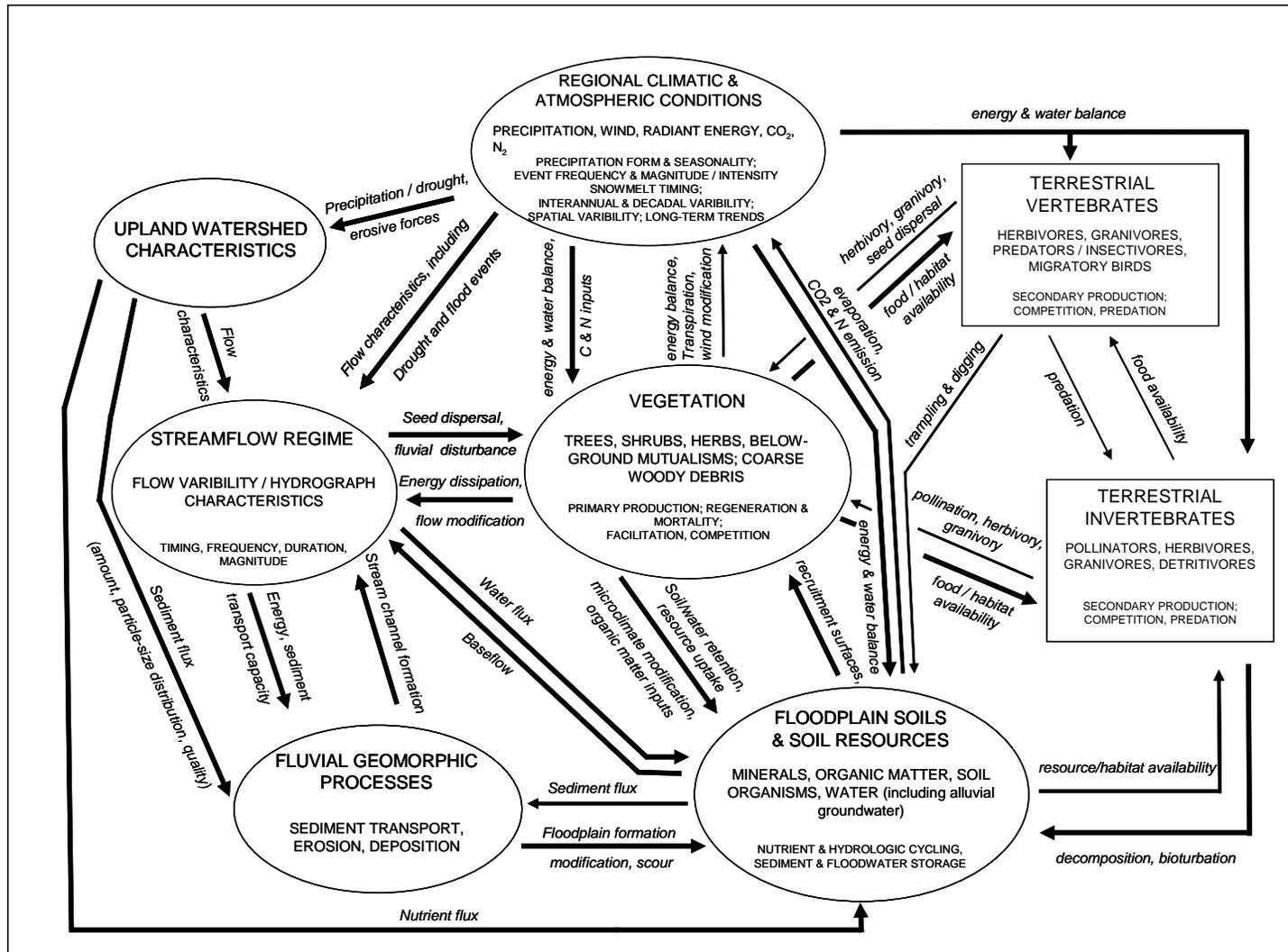


Figure 5. General conceptual model depicting the structure and functioning of riparian ecosystems. Symbols represent the following: solid, rounded rectangles = state factors or major drivers of ecosystem change and variability; ellipses = interactive controls; and rectangles = biotic components. Arrows indicate functional relationships among components and line weights indicate relative importance of relationships. The model is constrained by global climatic and atmospheric conditions, topography, parent material and potential biota.

Riparian/Aquatic Ecosystems

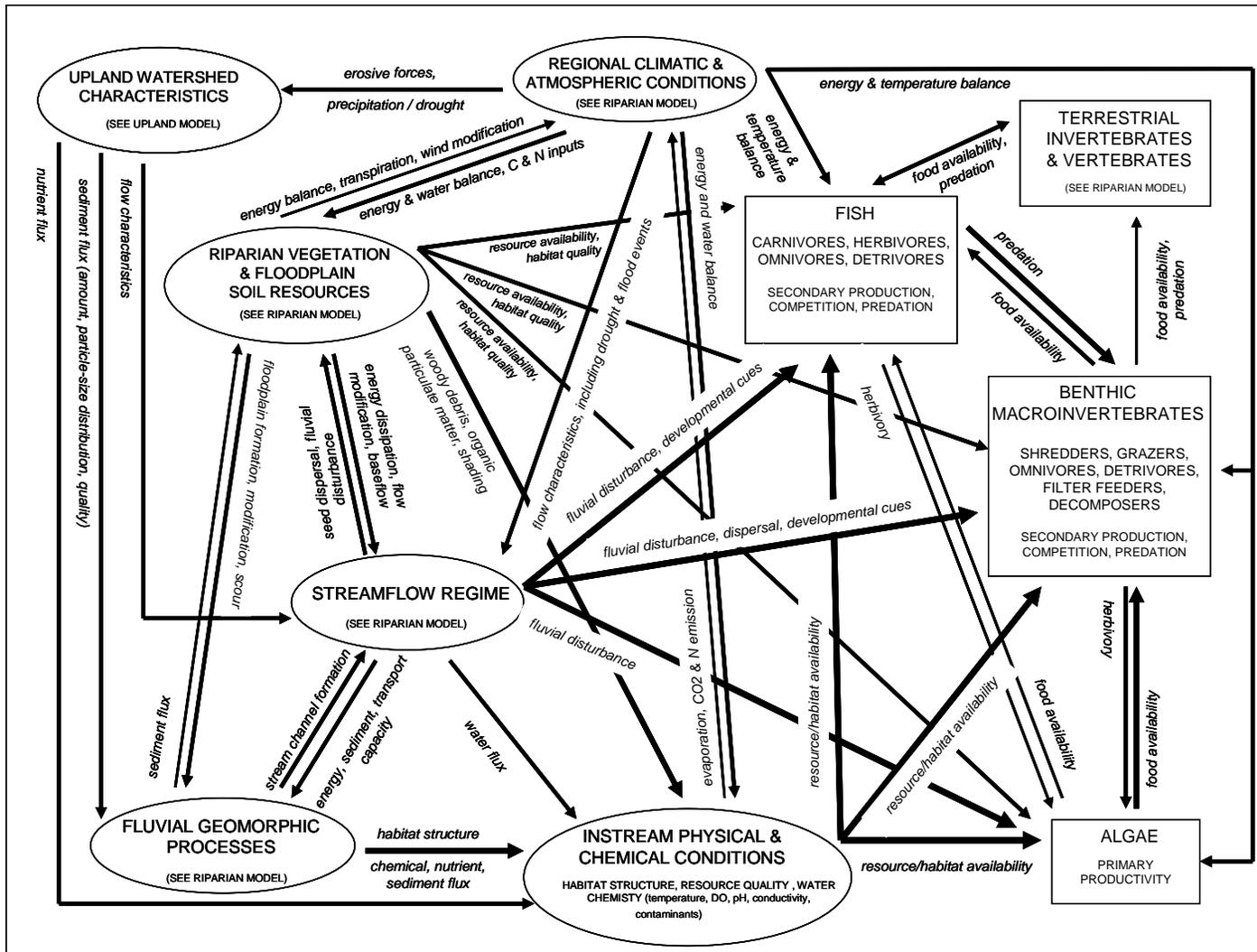


Figure 6. General conceptual model describing the structural components and functional relationships that characterize aquatic ecosystems. Symbols represent the following: ellipses = interactive controls; and rectangles = biotic components. Arrows indicate functional relationships among components and line weights indicate relative importance of relationships. The model is constrained by global climatic and atmospheric conditions, topography, parent material and potential biota.

1. Hierarchy of System Drivers

a. State Factors

i. Time, Initial Topographic Relief, Geology, and Global Climate

Over geologic time scales, Schumm and Lichty (1965) describe four independent variables that influence the erosional evolution of a landscape and its hydrology; (1) time, (2) initial topographic relief, (3) geology, and (4) global climate (Table 2) (Figures 5 and 6). Except for potential biota, these correspond with the state factors in the modified Jenny-Chapin model (Figure 4). The initial relief of a landscape represents potential energy. Over time, this energy is transformed to kinetic energy as climate, acting upon the underlying geological materials (e.g., erodable shales or resistant sandstones), progressively modifies landscape morphology through the process of erosion. Eight additional dependent variables, viewed as elements of a fluvial system, influence the nature of riparian and aquatic ecosystems through their effects upon flows of water and sediments. These variables, discussed in more detail in Section b. *ii*, below, include: (5) vegetation, (6) watershed relief, (7) watershed hydrology, (8) drainage network morphology, (9) hillslope morphology, (10) runoff and sediment flux, (11) valley morphology and channel/flood plain form, and (12) depositional processes and patterns (Schumm 1981) (Table 2).

Table 2. Fluvial system variables over geologic time scales (millions of years). Modified from: Schumm (1981).

<i>Fluvial System Variables</i>	Dependence of Variables
1. Time	Independent
2. Initial Relief	Independent
3. Geology (rock type and geologic structure)	Independent
4. Climate (global)	Independent
5. Vegetation (type and cover)	Dependent on climate and geology (soils)
6. Relief (percentage of watershed remaining above baselevel)	Dependent on preceding variables
7. Runoff and sediment yield (from upland watershed)	Dependent on preceding variables
8. Drainage network morphology (stream density, channel shape, gradient and slope)	Dependent on preceding variables
9. Hillslope morphology (hillslope angle and length)	Dependent on preceding variables
10. Discharge of water and sediment (from the watershed to the valleys)	Dependent on preceding variables
11. Valley and Channel morphology (pattern and extent of alluvial deposits, channel width/depth ratio, planform)	Dependent on preceding variables
12. Depositional system (alluvial fan, delta)	Dependent on preceding variables

b. Interactive Controls-Physical

i. Regional Climatic and Atmospheric Conditions

The precipitation regime is the most important climatic factor shaping the characteristics of riparian and aquatic ecosystems of the Colorado Plateau, and is the focus of our discussion here. Precipitation inputs are the key drivers of fluvial geomorphic processes and support water-limited ecological processes such as primary production, nutrient cycling, and plant reproduction in riparian and upland systems (Noy-Meir 1973, Comstock and Ehleringer 1992, Whitford 2002). These factors also determine how energy moves through the aquatic system and directly influences processes regulating macroinvertebrates. Precipitation seasonality (i.e., timing in relation to the annual cycle of potential evapotranspiration) is particularly important because it strongly controls the partitioning of precipitation into various compartments of the hydrologic budget – evaporation, transpiration, runoff, soil-water storage, and streamflow. Because of its effects on moisture availability, precipitation seasonality is a major determinant of dominant plant life forms and functional groups found within riparian ecosystems (Bagstad et al. 2005).

The precipitation regime strongly influences the seasonality of macroinvertebrate composition and abundance, as well as constraining life history characteristics of the macroinvertebrates. As an example, the North Atlantic Oscillation (NAO) has significant effects on growth and phenology of aquatic macroinvertebrates (Briers et al. 2004). Variation in growth and phenology, associated with the NAO, may influence temporal fluctuations in the composition and dynamics of stream communities. Such variations could result in mismatches between the timing of life history stages and changes in the biotic or physical environment could have important long-term consequences for stream ecosystem function (Briers et al. 2004).

Across the region encompassed by this report, precipitation seasonality varies due to the influence of the Arizona monsoon (Mitchell 1976, Peterson 1994). The Arizona monsoon (also referred to as the “southwest monsoon” or the “Mexican monsoon”) is recognized by climatologists as the northernmost portion of an extensive summer monsoon region that extends to central Mexico and the western slopes of the Sierra Madre Occidental (Higgins et al. 1998). Areas affected by monsoon circulation receive greater amounts of summer precipitation from moist air masses derived from the gulfs of Mexico and California. Notably, the mean northwestern extent of summer monsoon moisture is approximated by a band which cuts across the Colorado Plateau (Figure 7). Areas situated well northwest of this band are predictably dominated by cool-season precipitation (e.g., the parks GOSP, TICA, FOBU), whereas areas southeast of this band (e.g., BAND, ELMA, SAPU) receive higher amounts of summer monsoon precipitation from convective thunderstorms. Areas close to the band are generally characterized by a bimodal precipitation regime, with summer monsoon precipitation that is highly variable from year to year. The runoff regimes of large rivers that flow in the southern portions of the Colorado Plateau have a hydrologic signature reflecting the monsoon season. For example, summer and early fall floods are typical of some years on the Paria and Little Colorado Rivers in GRCA whereas summer floods are not significant on the Green or Yampa Rivers in DINO.

Ehleringer et al. (2000) hypothesized that effects of global change on atmospheric circulation patterns and precipitation may be seen relatively early in the Colorado Plateau region because of the presence of this significant climatic boundary. In an analysis of regional precipitation trends

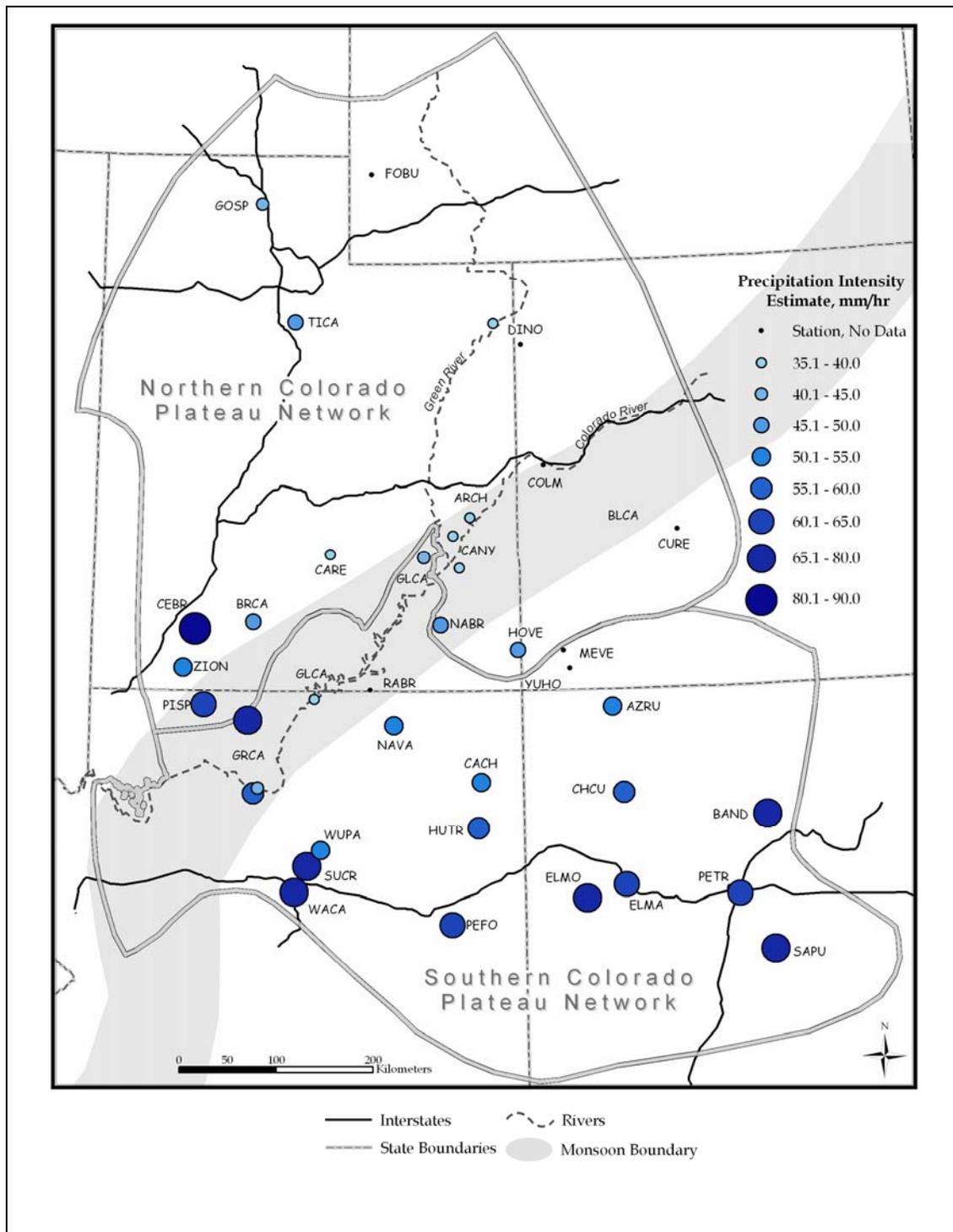


Figure 7. Map depicting regional variations in the estimated maximum intensity of precipitation (mm/hr) that can be expected to occur over a 5-min period with 50 percent probability during any given year at NWS stations located at or near NCPN and SCPN units. Shaded zone approximates the mean northwestern extent of summer monsoon moisture (from Mitchell 1976). Data were acquired from NOAA’s Precipitation Frequency Data Server (<http://hdsc.nws.noaa.gov/hdsc/pfds/>). See Table 1 for key to four-letter park codes.

for the central Colorado Plateau, Spence (2001) found a weak trend towards increasing winter precipitation, but no evidence for significant changes in monsoon precipitation patterns since the 1960's. Annual minimum temperatures, however, were found to have increased significantly across the region during the same time period (Spence 2001).

Regional precipitation patterns are affected by global-scale fluctuations in sea-surface temperatures, atmospheric pressure, and atmospheric circulation patterns that vary at two different time scales (Hereford et al. 2002). Short-term, inter-annual variations in precipitation are related in part to the occurrence of El Niño and La Niña conditions – the two contrasting phases of the El Niño – Southern Oscillation (ENSO) phenomenon that is driven by variations in sea-surface temperatures in the eastern tropical Pacific Ocean (Hereford and Webb 1992, Cayan et al. 1999, Hereford et al. 2002). Hereford and colleagues (2002) found that the detailed relationships were complex, but that strong El Niño episodes generally increased the variability of warm-season precipitation or the frequency of above-normal cool-season precipitation. In contrast, strong La Niña episodes tended to cause normal, low-variability warm-season precipitation and below-normal cool-season precipitation.

Decadal-scale variations in precipitation patterns are related to a recently recognized phenomenon known as the Pacific Decadal Oscillation, or PDO (Mantua and Hare 2002, Hereford et al. 2002). Precipitation variability associated with the PDO is partly related to cyclical variations in sea-surface temperatures in the northern Pacific Ocean, although mechanisms driving PDO variability remain poorly understood (Mantua and Hare 2002). In their analyses of 20th-century precipitation patterns on the Colorado Plateau, Hereford and colleagues (2002) found evidence for three relatively distinct precipitation regimes that appeared to be in phase with the PDO. The first of these was a period of relatively wet conditions from 1905 to 1941. Following this wet period, two other distinctly recognizable precipitation regimes occurred from 1942 to 1977 (dry) and from 1978 to 1998 (wet). The marked shift to dry conditions that began in 1999 and continues through the present suggests a transition to the dry PDO phase that could continue for the next 2-3 decades (Hereford et al. 2002). This has important implications for ecosystem management⁴ and monitoring in the region due to the effects of precipitation patterns on disturbance regimes (Swetnam and Betancourt 1998) and on the capacity of ecosystems to resist or recover from natural disturbances and human land- and water-use activities (Ehleringer et al. 2000, Whitford 2002).

ii. Upland Watershed Characteristics

Schumm (1981) described an idealized fluvial system consisting of three zones: (1) watersheds or zones of net sediment production, (2) streams and rivers representing zones of transport of water and sediment from the watershed, through valleys, to, (3) zones of net deposition, such as deltas and alluvial fans (Figure 8). These zones are not as spatially segregated as represented by Figure 8 because in reality there is a rather complex interpenetration of zones. For example, alluvial sediments may be temporarily stored as channel or flood-plain deposits within the channel network of a watershed or in the valley of a large river (zones 1 and 2) (Schumm 1981,

⁴ *Ecosystem management* is the process of land- and water-use decision making and land- and water-management practice that takes into account the full suite of organisms and processes that characterize and comprise the ecosystem and is based on the best understanding currently available as to how the ecosystem works. Ecosystem management includes a primary goal of sustainability of ecosystem structure and function, recognition that ecosystems are spatially and temporally dynamic, and acceptance of the dictum that ecosystem functioning depends on ecosystem structure and diversity (Dale et al. 2000:642).

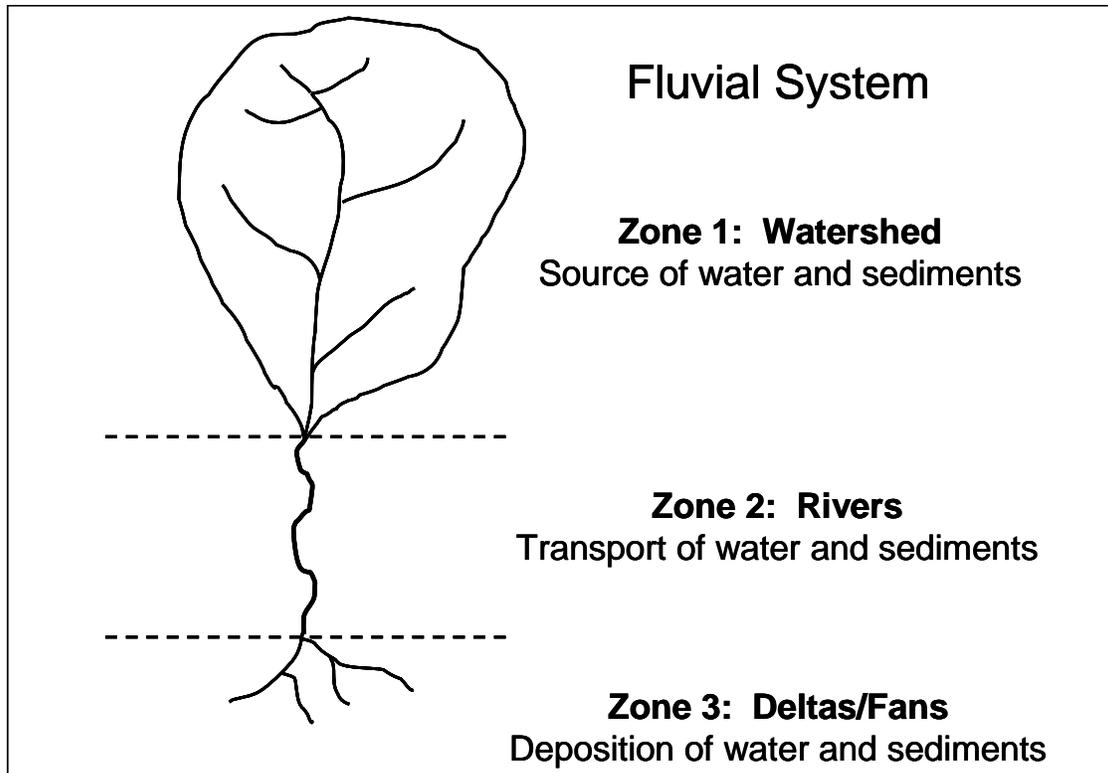


Figure 8. An idealized diagram of a fluvial system featuring: (1) a zone of sediment production (watershed); (2) a zone of transport (rivers and streams); and (3) a zone of deposition (alluvial fans, deltas) From: Schumm (1981).

Benda et al. 2004). Likewise, reservoir deltas (zone 3) may be actively eroded as declining reservoir pools lower local baselevels (J. Schmidt, personal communication).

The upland watershed contains a diversity of landform features including drainage divides, hillslopes, stream channels and flood plains. Water and sediment are ultimately derived from the upland watershed (zone 1) through the interaction of the eight dependent watershed variables listed in Table 2. The four independent or state variables of time, initial relief of the watershed, geology, and global climate influence the type and cover of vegetation, and watershed topography. These in turn influence the runoff and sediment flux from the watershed, the development of stream network and hillslope morphologies, and thus the discharge of sediment and water to receiving streams and rivers (zone 2). The amount and timing of flow and the amount and size of sediment, delivered from the watershed to the valleys, establishes channel and flood-plain form and processes, which provides the physical template for riparian and aquatic ecosystems (Figures 5 and 6) (Frisell et al. 1986).

Given the number of interactive controlling variables, watershed characteristics can be endlessly diverse; however, regional characteristics of the Colorado Plateau allow for some generalized inferences about the influence of watershed characteristics on stream-flow patterns and sediment flux. For example, the Colorado Plateau represents a geologically young landscape, of high-relief, thin, patchy soils, extensive exposures of bedrock, and sparse vegetation. In such a landscape, drainage densities within a watershed will be relatively high and stream gradients

steep, resulting in high unit runoff for a given precipitation event, with high, flashy peak discharges, relatively low baseflows, and high sediment yield. On the Colorado Plateau, thunderstorm events deliver high precipitation rates that cannot infiltrate the soils of typical watersheds, and short-duration overland flow events are characteristic of the monsoon season. Land use activities like livestock grazing that increase the area of exposed bedrock, or which decrease soil stability and infiltration rates, result in increased delivery rates of water to stream channels, which in turn lead to more rapid runoff and larger flood events. High surface runoff rates tend to increase soil erosion, and the removal of vegetation also leads to soil erosion by raindrop impact. Delivery of larger amounts of water and sediment from the watershed (zone 1) to stream channels (zone 2) has the potential to alter channel form and process and thus alter the structure and functioning of riparian and aquatic ecosystems.

iii. Stream-flow Regime

Stream flow originates from precipitation falling within a watershed. However, resulting stream flow patterns, or the stream hydrograph, can be highly variable across streams, because of underlying differences in climate, geology, topography, soils and vegetation cover within a watershed. Precipitation reaches a stream through various pathways, including direct precipitation, unsaturated or Horton overland flow, ground-water flow, shallow sub-surface flow, and saturated overland flow (Figure 9) (Dunne 1978). Each of these flow paths respond differently to precipitation events (rain or snow) and thus contribute differentially to two important components of stream flow; *baseflow* and *stormflow*. Because rates of groundwater flow are slow and flowpaths are relatively long, water moving to streams along these paths contribute to the baseflow of streams between precipitation events. Surface runoff from precipitation reaches streams much more quickly, contributing to stormflow during and shortly after precipitation events (Figure 10a). Because of the potential for high intensity (monsoon)

rainfall events, steep terrain, thin, patchy soils, exposures of relatively impermeable bedrock, and sparse vegetation, the hydrographs of streams originating within the Colorado Plateau are dominated by relatively high-magnitude, short-duration, temporally unpredictable stormflow hydrographs with little or no baseflow (Figure 10b). In contrast, the large extraregional rivers that traverse the Plateau, feature snowmelt hydrographs with temporally predictable, long-duration snowmelt peaks and baseflow (Figure 10c). Ultimately, the stream-flow regime determines the mechanical forces available in the valley that erode, transport and deposit sediment and which maintain channel form and process.

Flow Variability. Although stream flows of virtually all perennial rivers, originating within or flowing through the Colorado Plateau, have been modified by humans, the Yampa River remains the only relatively un-regulated, extraregional stream in the Colorado River Basin. Flows on the Yampa and pre-dam Green Rivers provide an example of a relatively natural snowmelt hydrograph (Figure 11), featuring a temporally predictable snowmelt runoff peak in late May to early June, followed by flow recession to relatively low base flows throughout the remainder of the year. In contrast, flow regulation from large, in-channel dams, dramatically reduces these peaks while typically increasing baseflows, as illustrated by the post-dam Green River (Figure 11). Annual variations in the magnitudes of high and low flows characterize the natural flow variability that is important to maintaining the ecological integrity of riparian and aquatic ecosystems (Figure 10c).

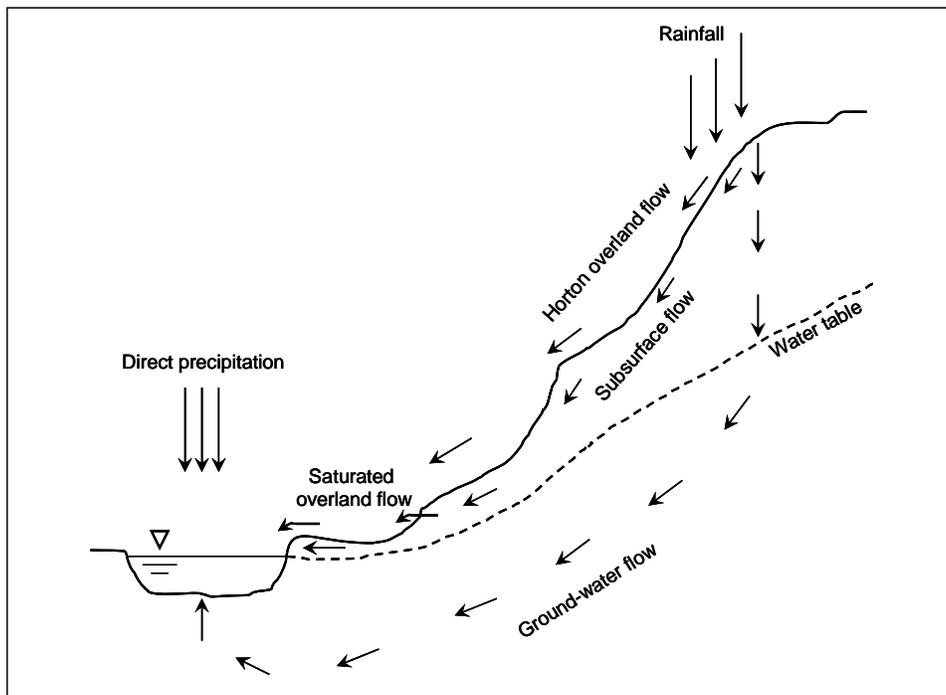


Figure 9. Idealized flow paths of water moving from a watershed to a stream. Adapted from: Dunne (1978) and Ziemer and Lisle (1998).

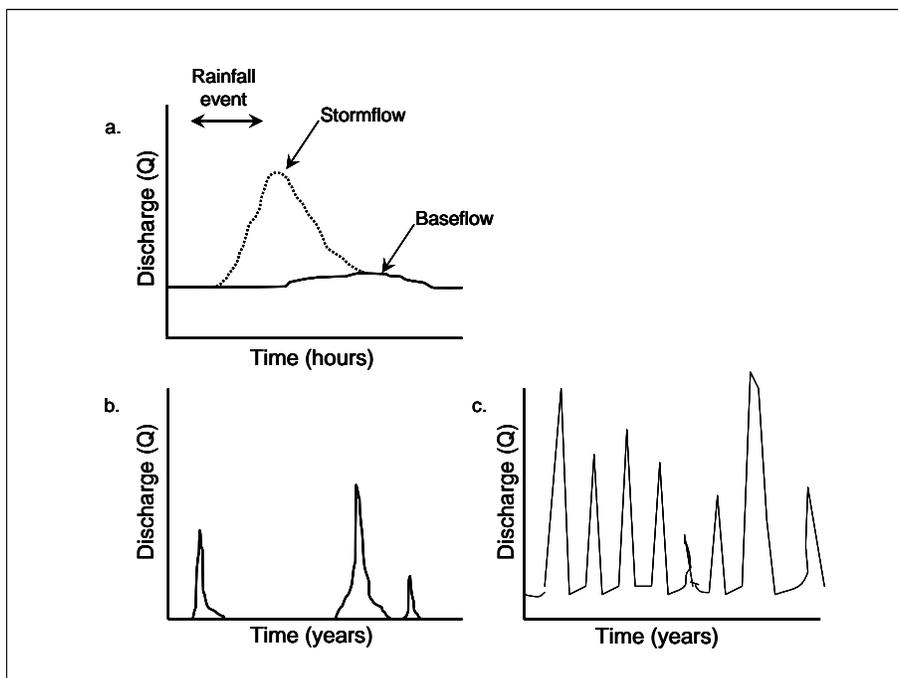


Figure 10. Stream hydrographs. (a) Idealized relationship between stormflow and baseflow components of a stream hydrograph, relative to a discrete rainfall event (Modified from Dunne 1978). (b) Idealized hydrograph of an ephemeral stream of the Colorado Plateau, featuring highly variable and temporally unpredictable peak flows with no baseflow. (c) Idealized hydrograph of an unregulated, large perennial stream of the Colorado Plateau, featuring a variable but temporally predictable snowmelt peak flow.

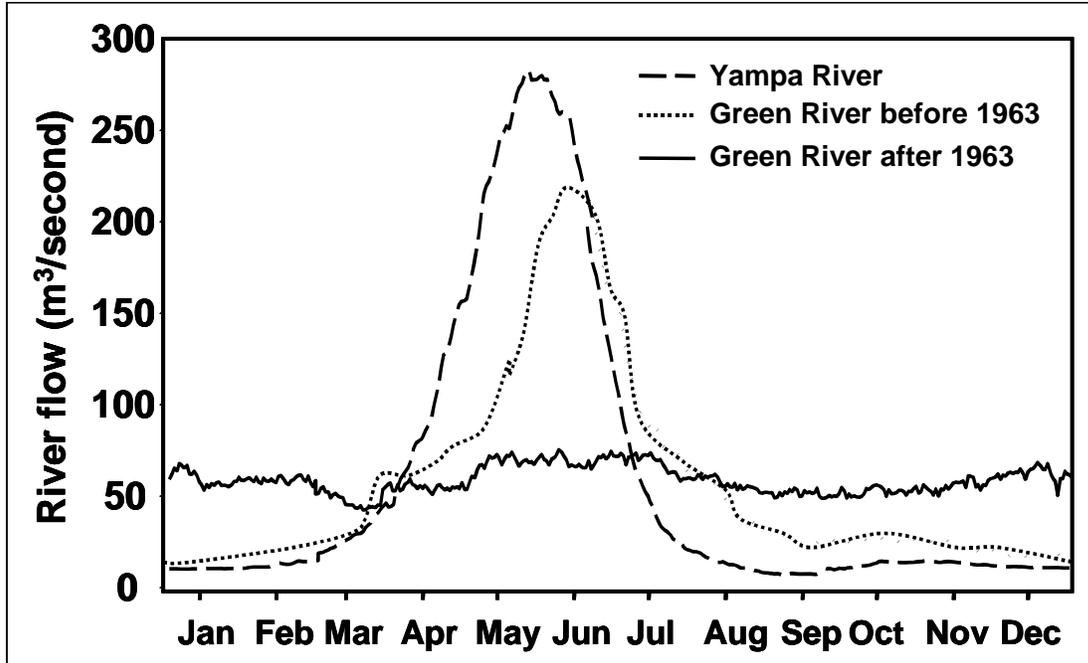


Figure 11. Mean monthly flows for the Yampa river (from 1922), the Green River (from 1929-1963), and the Green River (after 1963). Flaming Gorge Dam began regulating flows on the Green River after 1963. From: Adair et al. (2002).

The natural flow regime paradigm holds that natural flow variability is primarily responsible for structuring and maintaining the physical and biotic integrity of aquatic and riparian ecosystems (Figure 12) (Richter et al. 1996, Stanford et al. 1996, Poff et al. 1997). Ecologically relevant elements of stream flow include the magnitude, frequency, duration, timing, and change rate of flow. These elements have been used to describe regional stream-flow patterns, which vary as a function of climate and watershed characteristics (Poff and Ward 1989). They may also be used to characterize specific hydrologic events, such as extreme high or low flows, or human-modified flow patterns, both of which can exert lasting influence on the ecological integrity of these systems (Richter et al. 1996).

Although extreme flow variation can eliminate species (Zimmerman 1969, Bain et al. 1988), episodic floods and droughts are necessary for persistence of some species of fish (Meffe 1984) and plants (Nilsson et al. 1991, Friedman, et al. 1996). In fact, the high biological diversity characteristic of riparian and aquatic ecosystems may be maintained by relatively frequent hydrologic disturbance events, which would act to limit the process of competitive exclusion of species in these environments (Huston 1979). In aquatic ecosystems, for example, tinaja fauna has been studied at CARE where existing macroinvertebrate communities seem to be resistant to natural disturbance, such as flooding and drying. Baron et al. (1998) hypothesized that hydrologic variability (which limits the success of many otherwise dominant species) is necessary for the maintenance of these unique systems. In a study of riparian systems, empirically derived distributions of plants along cross-valley gradients of inundation frequency and duration were used in conjunction with hydraulic models to simulate the response of wetland, riparian and upland vegetation to changes in flow along the Gunnison River in BLCA and Fremont River in CARE. Model results predicted reductions in the area of wetland and

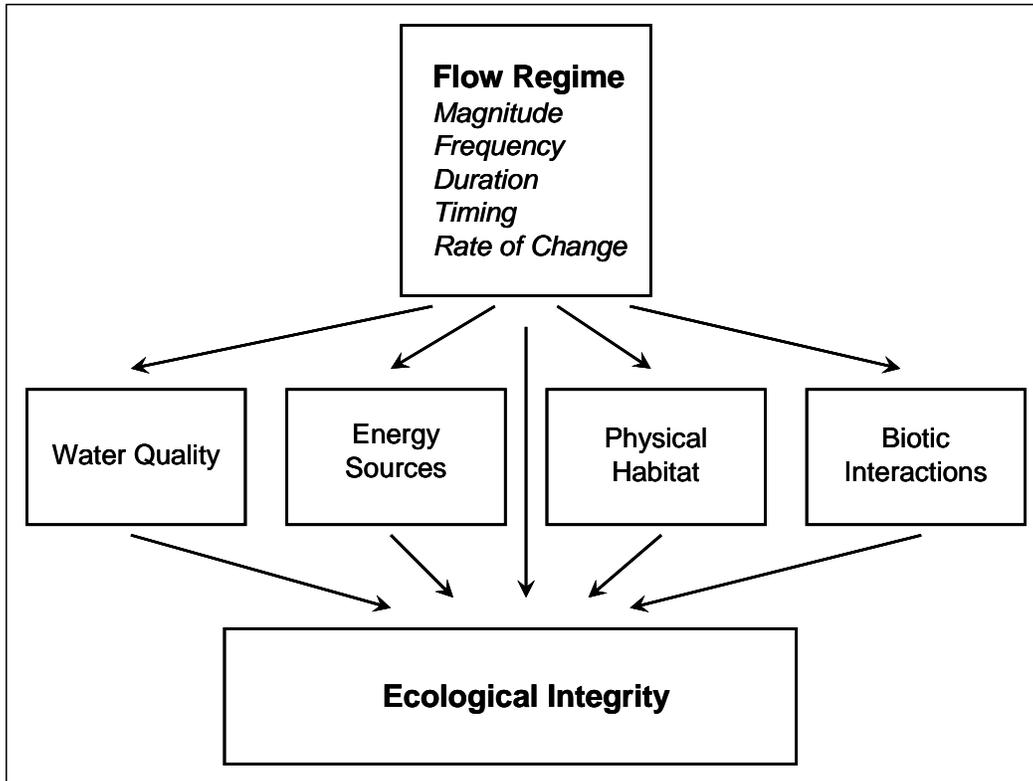


Figure 12. Representation of how a natural flow regime creates and maintains the ecological integrity of riverine ecosystems. From: Poff et al. (1997).

riparian vegetation zones in response to diminished flow variability, as frequency and duration of inundation decreased away from the stream (Auble et al. 1994, 2005). These predictions match observed changes in vegetation along the Green River below Flaming Gorge Dam and Reservoir (Merritt and Cooper 2000).

Given the importance of flow variability in structuring and maintaining riparian and aquatic ecosystems, identification of a parsimonious set of hydrologic indicators that are sensitive to anthropogenic disturbances, would be an important element of any efforts to monitor, manage, and restore riparian and aquatic ecosystems (Olden and Poff 2003).

Floods. The reproductive traits of early successional riparian trees are tightly linked with fluvial disturbances. Seeds of *Populus* spp. and *Salix* spp. germinate and grow on moist, freshly deposited alluvial sediments following floods of appropriate timing, magnitude, and rate of flow recession (Mahoney and Rood 1998, Stromberg et al. 1991, Scott et al. 1997, Auble and Scott 1998, Cooper et al. 2003). The physical disturbance and increased moisture availability provided by floods is also positively associated with species richness and cover of herbaceous species in arid and semi-arid riparian zones. Whereas some studies have reported reduced diversity of riparian herbs following flooding (Smith et al. 1998), flood-related increases in the cover and diversity of annual and some perennial riparian herbs along the San Pedro River, Arizona, was attributed to creation of safe sites for germination, increased water availability, and the possible transport of seeds and vegetative propagules by flood waters (Bagstad et al. 2005). Flood transport of seeds, or hydrochory, may play an important role in maintaining high species

diversity in riparian landscapes by preferentially delivering the seeds of species, or groups of species, to specific riparian landscape positions at times suitable for establishment and growth. Fluvial surface features and simple hydrologic variables explained significant proportions of the observed variability in seed deposition patterns along stream channel margins in flume experiments (Merritt and Wohl 2002).

High magnitude floods, in particular, can produce dramatic, long-term transformations in riparian ecosystem structure and functioning by inducing widespread geomorphic change and plant mortality that may in turn initiate extended episodes of establishment of relatively long-lived riparian species (Schumm and Lichty 1963). Flood-induced widening of ephemeral streams by infrequent, large-magnitude floods was followed by decades of post-flood channel narrowing, and establishment of riparian cottonwoods on portions of the former channel during intervening low flow periods (Friedman and Lee 2002). Thus, individual floods may influence the reproductive patterns of riparian species for decades following the event. Along many southwestern U.S. streams, channel narrowing and flood-plain formation since at least the 1940's, has been accompanied by the establishment of extensive stands of tamarisk (primarily *Tamarix ramosissima*; Burkham 1972, Hereford 1984). The degree to which tamarisk has facilitated such narrowing is the nexus of a long-standing debate (Graf 1978, Everitt 1980). However, the regional nature of channel narrowing and flood-plain construction in several southern Colorado Plateau streams led Hereford (1987) to conclude that this channel-change process is primarily under the control of larger-scale factors such as climate.

Alluvial Groundwater. Water from surface flow and associated shallow alluvial aquifers is essential to the persistence of most low-elevation woody riparian species in the southwestern U.S. Thus, an integrated understanding of surface and alluvial groundwater flows, and their interactions, is fundamental to understanding establishment and survival processes of existing riparian and wetland ecosystems (Winter 1999, Woessner 2000). On coarse substrates in dry regions, early establishment and growth of *Populus* spp. seedlings, and other woody riparian pioneer species, may require groundwater within 1-2 m of the establishment surface (McBride and Strahan 1984, Mahoney and Rood 1992, Segelquist et al. 1993, Stromberg et al. 1996), but lenses of finer alluvial material may allow seedlings to survive the first few growing seasons without making contact with the groundwater (Cooper et al. 1999). Following initial establishment, root growth allows young trees to survive gradual groundwater declines. Depth to the groundwater may increase as a result of subsequent flood-plain accretion or channel incision (Everitt 1968, Hereford 1986), and *Populus* species have been observed at sites where depth to groundwater is 7 - 9 m (Robinson 1958). However, mature native riparian species such as *Populus*, *Salix* and *Tamarix* are typically found in riparian settings where depth to water is < 4 m (Meinzer 1927, Busch et al. 1992, Scott et al. 1997, Stromberg et al. 1997, Horton et al. 2001a). Close proximity to groundwater was important in the establishment and persistence of some wetland and riparian herbs along the San Pedro River, Arizona (Bagstad et al. 2005).

Alluvial groundwater is the principle source of water for riparian trees (Busch et al. 1992, Snyder and Williams 2000) and even relatively modest fluctuations or declines (1.5-3 meters) can induce lethal moisture stress (Scott et al. 1999, Shafroth et al. 2000). Seasonal groundwater declines of 2.5-3 meters, in a dry year, along the free-flowing Hasayampa River, Arizona, produced moisture stress in the native riparian cottonwood (*Populus fremontii*), and willow (*Salix*

gooddingii) and non-native tamarisk (*Tamarix ramosissima*). All species responded to this stress with lowered shoot water potentials, decreased leaf gas exchange rates, increased canopy die-back, and some tree mortality. Compared to native riparian trees, however, tamarisk had much higher rates of leaf gas exchange and stem growth under shallow groundwater conditions, and exhibited less crown die-back and mortality when groundwater declined. The combination of high leaf gas exchange rates and stem growth when water is available, and greater moisture stress tolerance under dry conditions, help explain the competitive success of tamarisk in southwestern riparian ecosystems, particularly those subject to large within and across-year fluctuations in water availability (Horton et al. 2001b).

The rate, depth, and duration of alluvial ground-water decline and the water holding characteristics of the soil interact with atmospheric water demand (i.e., temperature, humidity, wind speed) to influence the intensity and duration of water stress in groundwater-dependent plants. The few studies that quantitatively link alluvial groundwater dynamics to riparian vegetation response suggest that along rivers in arid and semi-arid regions: (1) riparian trees are sensitive to seasonal or longer-term alluvial groundwater declines (Groeneveld and Griepentrog 1985, Stromberg et al. 1996), (2) they exhibit moisture stress responses ranging from short-term physiological adjustments to stand-wide mortality (Busch et al. 1995, Scott et al. 1999, Shafroth et al. 2000, Horton et al. 2001a,b), (3) stress responses can be deferred by short-term increases in streamflow and corresponding rises in the groundwater (Cooper et al. 2003), (4) tree physiological condition deteriorates rapidly when groundwater declines cross a threshold depth ranging from 1.5-3 meters (Scott et al. 1999, Shafroth et al. 2000, Horton et al. 2001a), (5) the non-native tamarisk is more tolerant of groundwater-induced moisture stress than native cottonwoods and willows (Busch and Smith 1995, Cleverly et al. 1997, Shafroth et al. 2000, Horton et al. 2001b), and (6) that the intensity of the physiological response appears to be conditioned by the influence of the historical, site-specific groundwater regime on root architecture (Shafroth et al. 2000, Scott et al. 2000).

Drought. The effects of regional climatic drought on riparian ecosystems are expressed most directly through reduced surface flows and depletion of alluvial groundwater aquifers. Thus, the stress effects of naturally occurring drought mimic those produced by anthropogenic stressors such as damming and diversion of streamflow, groundwater pumping, and channel incision resulting from altered flows of water and sediments, bank stabilization, or in-stream gravel mining (Bravard et al. 1997, Kondolf 1994 & 1997, Rood et al. 1995, Stromberg et al. 1996 & 1997, Scott et al. 2000).

The response of any plant to gradually increasing water stress involves progressive and integrated physiological and morphological responses, beginning with stomatal closure, reduced leaf and canopy development, and ending with death (Bradford and Hsiao 1982, Braatne et al. 1992). Mild water stress can reduce plant productivity by limiting CO₂ assimilation through stomatal closure, lowering net photosynthesis, and through the death of leaves and fine roots. Under more severe drought conditions, trees exhibit reduced radial stem increments, wilting and abscission of leaves, and branch death. Tree mortality may follow directly or secondarily as the result of insects or other pathogens (Albertson and Weaver 1945). Because these changes occur at different levels of water stress and on different time scales, accurate quantification of longer-term water stress is problematic (Pallardy et al. 1991).

Despite widespread occurrence in arid and semi-arid landscapes, riparian cottonwood species are susceptible to drought-induced cavitation of xylem vessels (Tyree et al. 1994), and suffer higher mortality during drought than several eastern deciduous forest species (Kaylor et al. 1935, Albertson and Weaver 1945) or non-native tamarisk (Busch and Smith 1995, Cleverly et al. 1997, Horton 2001a, b). In water stressed cottonwood species, Smith et al. (1991) found significantly reduced stomatal conductance and reduced midday leaf water potential (Ψ_1) for *Populus trichocarpa* compared with non-stressed trees. These trends were particularly pronounced for juvenile trees. Busch and Smith (1995) found moderately higher rates of stomatal conductance and transpiration and slightly higher predawn and midday Ψ_1 in comparing *Populus fremontii* and *Salix gooddingii* from a gaining reach with those from a losing reach of the Bill Williams River, Arizona. Riparian *Populus* can exhibit morphological and growth responses to chronic water stress, including reduced leaf size, increased leaf thickness, reduced leaf area, reduced annual stem elongation, and reduced radial stem increments (Smith et al. 1991, Stromberg and Patten 1991, Busch and Smith 1995). Under conditions of acute water stress associated with severe climatic drought or ground-water declines, *Populus* display more extreme morphological responses such as crown die-back (branch sacrifice), and ultimately stand mortality (Ellison and Woolfolk 1937, Albertson and Weaver 1945, Stromberg 1993, Rood et al. 1995, Rood et al. 2000).

iv. Fluvial Geomorphic Processes

Stream Channel and Flood plain Form. Stream channels adjust to variations in the discharge of water and the size and amount of sediment supplied to the stream from the watershed (Figure 5). Flow governs channel dimensions like width, depth and meander patterns. Channel form is mostly determined by the amount and size of bedload⁵, even though bedload may be a small proportion of the total sediment flux. In the case of Colorado Plateau streams, gravel is typically a small proportion of the total sediment flux which is primarily composed of sand, silt, and clay. Five general channel types have been identified, based on plan-view pattern and sediment load, and are presented in Figure 13, in terms of the relative stability of their erosional patterns, as well as how channel shape and gradient relate qualitatively to the variables of sediment size, sediment load, flow velocity and stream power. These channel types include: (1) straight channels with migrating sand waves; (2) straight channels with alternate bars; two meandering channel types (3a) highly sinuous channels of approximately equal width and (3b) channels that are wider at the bends than between bends; (4) channels transitional between meandering and braided form; and (5) braided channels (Schumm 1981). Because of regional watershed characteristics contributing to high, flashy peak flows and high sediment loads, channels of Colorado Plateau streams are typically composed of low bars and the active channels are often braided.

Abrupt changes in channel patterns, from straight through braided forms, can occur in response to a range of factors, as critical geomorphic thresholds are exceeded by changes in external variables such as stream power, channel gradient, and sediment (Schumm and Kahn 1972). Such channel pattern-shifts can be triggered by episodic events, which may have long-lasting effects on stream and valley morphology, erosional and depositional processes, and riparian and aquatic ecosystems. Rare, large floods have eroded flood plains and terraces and transformed

⁵ *Bedload* refers here to sediment moving on or near the bed of a channel.

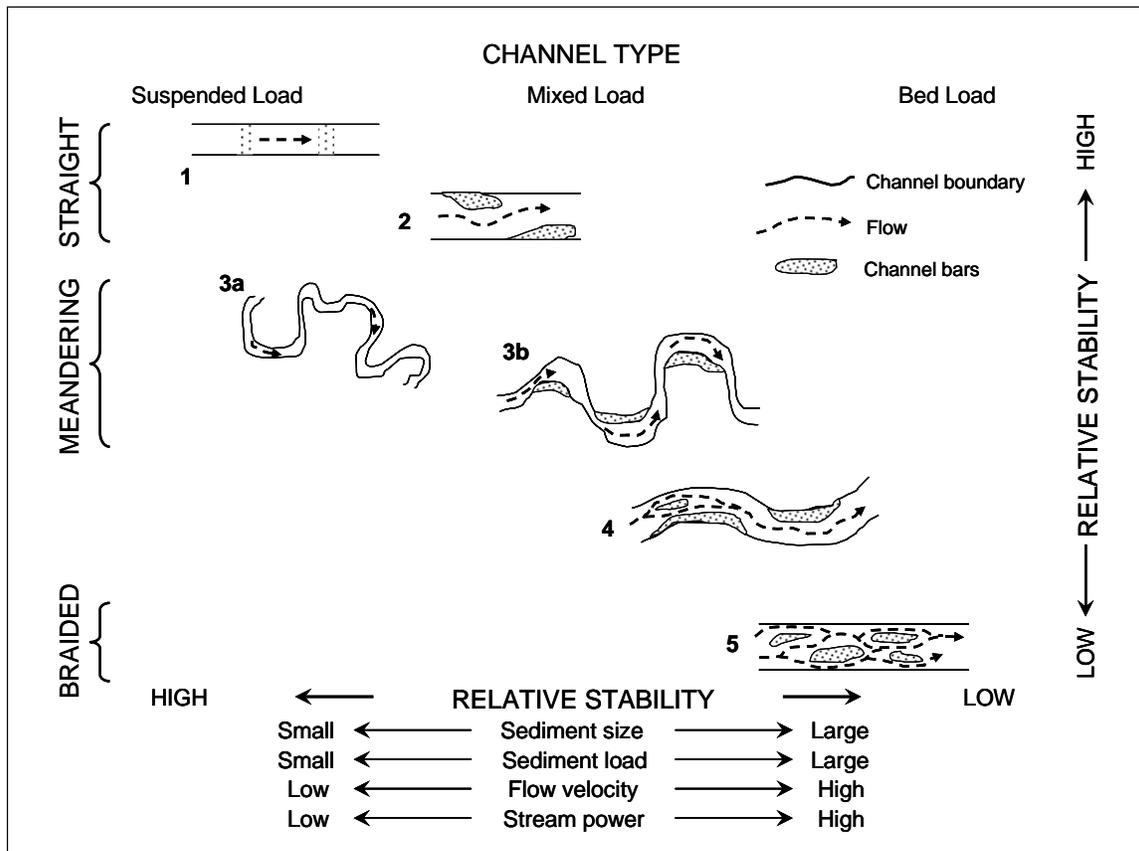


Figure 13. A qualitative classification of stream channels based on pattern (straight, meandering, or braided) and type of sediment load, along with flow and sediment variables and relative stability with regard to average erosional activity. From Schumm (1981).

meandering channels, near the threshold of a pattern-change, to a braided pattern (Schumm and Licity 1963). Following such floods, subsequent channel narrowing and re-establishment of a meandering channel form, have been shown to occur through the process of flood plain construction and the establishment of riparian vegetation on portions of the former channel bed (Schumm and Licity 1963, Friedman et al. 1996). Similarly, channel narrowing, by flood-plain construction and the widespread establishment of tamarisk, has been noted across the Colorado Plateau. These changes have been attributed to natural and human-induced shifts in stream flow and sediment delivery caused by climate change, land use changes, and completion of the Colorado River Storage Project (CRSP). These changes also resulted from the establishment and spread of the non-native shrub tamarisk (*Tamarix ramosissima*) (Allred and Schmidt 1999, Graf 1978, 1980, Grams and Schmidt 2002, Hereford et al. 2002).

Flood plains represent one of a number of river-deposited features and are typically composed of vertically stacked fine-grained layers of sediment left by discrete floods. By definition, flood plains are level surfaces constructed by a river under prevailing climatic conditions, and are relatively frequently inundated by high flows (Leopold 1994); however, there is no regionally consistent recurrence of inundation of these features, as is found in laterally accreting flood plains along meandering rivers in other regions. Riparian vegetation establishment and

succession is intimately linked to the lateral and vertical accretion of sediments that lead to flood plain formation across a range of channel forms (Schumm and Lichty 1963, Hereford 1984, Bradley and Smith 1986, Boggs and Weaver 1994). This linkage between fluvial geomorphic processes and riparian vegetation dynamics creates the topographic diversity, soil moisture gradients, fluvial disturbance patches, and distinctive microclimates that characterize riparian ecosystems (Figure 5). The spatial extent of flood plains along rivers and streams of the Colorado Plateau is highly variable and dependent on geomorphic setting. Along channels confined by colluvium⁶ or bed rock, flood plain deposits may be narrow and discontinuous, or even non-existent. In contrast, channels in large alluvial basins may have large, spatially extensive flood plains.

Vertically aggraded flood plains progressively become disconnected from surface flows in adjacent channels, and may be abandoned if the regional climate becomes drier. Abandoned flood plains are referred to as terraces. Remnant terrace sequences from across the arid and semi-arid western U.S., including the Colorado Plateau, record several climatically driven valley cut-and-fill cycles during the Holocene period (within the last 10,000 years). These changes have dramatic effects on rivers and their flood plains. Geologic evidence indicates that during relatively cool, wet periods, valleys fill by deposition of alluvial (river-derived) sediments. When a period of deposition is followed by a comparatively dry period, the channel incises into the alluvium, abandoning the previously constructed flood plain as a terrace. Whereas valley deposition or aggradation is a slow process (thousands of years), corresponding valley erosion is rapid (tens to hundreds of years) (Leopold 1994). A more detailed description of valley cut-fill cycles is provided in the following section.

Arroyo Cutting and Filling. Arroyos are steep-walled gullies with inset stream channels. They are typically incised into cohesive, fine-textured valley fill materials and occur throughout the western U.S.; however, they are especially common in arid and semi-arid regions of the southwest. Arroyos can form rapidly in response to floods, cutting to depths of up to 20 meters and widths of greater than 50 meters. Because of the often dramatic physical changes and related economic impacts resulting from arroyo formation, this process has received considerable scientific attention. The most recent episode of active arroyo formation in the southwestern U.S. occurred between about 1880 and 1930, although stratigraphic evidence points to earlier cycles of arroyo cutting and subsequent filling that occurred approximately 2000 and 700 years before the present (YBP) (Emmett 1974). A number of hypotheses have been proposed to explain arroyo formation, of which climate change, anthropogenic impacts, and intrinsic adjustments of the channel system have emerged as leading factors (Cottam and Stewart 1940, Schumm and Hadley 1957, Cook and Reeves 1976, Graf 1988). In this interpretation, some streams systems in arid and semi-arid regions are inherently unstable as a function of intrinsic variables such as climate, geology, vegetation cover, valley slope and other factors that influence the fluvial system (see Table 2). As these systems approach critical geomorphic thresholds, fluctuations in climate, such as seasonal precipitation patterns, or land use, such as grazing intensity, may act as triggers, initiating a cycle of arroyo cutting as systems cross erosional thresholds (Bull 1997, Elliott et al. 1999, Miller et al. 2004)

⁶ *Colluvium* is material typically found at the foot of a slope and deposited there as a result of gravitational action.

Arroyo cutting is used here to describe the process by which a stream channel is significantly lowered and widened by erosion. Once initiated, arroyo formation tends to be a self-sustaining, long-term process that can propagate through a drainage network. A description of arroyo formation along the Fremont River, Utah, and major tributaries, illustrates the long-term, complex response of a stream network to arroyo cutting. According to Graf (1980), arroyo development in the Fremont River basin involved five key processes: (1) development of discontinuous arroyos in the early to mid-1800s, as stream gradient adjusts to a) the crossing of intrinsic stability thresholds, b) shifts in climate, or c) land-use changes; (2) erosion of a deep arroyo system along the channel of the Fremont River following a major flood in 1897; which (3) triggered the headward erosion of arroyos up the major tributaries; (4) the formation of soil pipes (Fletcher et al. 1954); together with (5) continued failure of the arroyo walls following flow events, which adds new sources of sediment to the system. The process of arroyo cutting within the Fremont River basin remains active more than 100 years following the initiating event.

The formation of arroyos along stream networks can have important indirect effects on riparian vegetation through their influence on alluvial ground water (Bravard et al. 1997, Scott et al. 2000). Throughout the southwestern U.S., many river valley bottoms that once supported riverine marshes (Cienegas) and riparian forests have been converted, by channel incision and consequent alluvial ground water declines, to dry terraces dominated by drought-tolerant vegetation (Cook & Reeves 1976, Hendrickson & Minckley 1985). Channel change processes like arroyo cutting are often discontinuous in time and space, involving lags in geomorphic and biological responses to changes in physical conditions. Thus, without accurate historical information, the initiation of arroyo cutting events and their ecological consequences remain largely unaccounted for (Graf 1980).

v. Natural Disturbance Regime

We consider stream-flow variability and fluvial geomorphic processes to be key elements of the natural disturbance regime for riparian and aquatic ecosystems (Figures 5, 6). Within riparian corridors, the availability of water and nutrient rich soils, along with relatively frequent fluvial disturbance, contribute to high rates of productivity and confer both resistance and resilience to natural disturbance processes (Stromberg 1993). In addition, uniquely high levels of biological diversity associated with riparian ecosystems are attributed to several factors, including 1) variation in the frequency and intensity of flooding, 2) large-scale variation in climate as streams traverse elevational gradients, 3) small-scale topographic diversity and related soil and moisture gradients, created by channel change processes, and 4) upland disturbance processes, which together, produce a diverse array of habitat patch types (Naiman et al. 1992).

Early successional woody riparian species like cottonwood and willow, as well as a host of herbaceous species, are disturbance-dependent, requiring bare, moist stream deposits for seed germination and establishment. These are restrictive conditions in arid and semi-arid environments and such conditions are produced most frequently and extensively on a landscape scale by fluvial geomorphic processes. Thus, models of riparian ecosystem dynamics typically begin with un-vegetated alluvial landforms, which in regions like the Colorado Plateau, are typically colonized by cottonwood and willow species. These early successional vegetation patches are either replaced by later successional riparian or upland species, or returned to bare

alluvium by intense fluvial disturbance (Johnson 1994, Friedman et al. 1997, Richter and Richter 2000) (Figure 14).

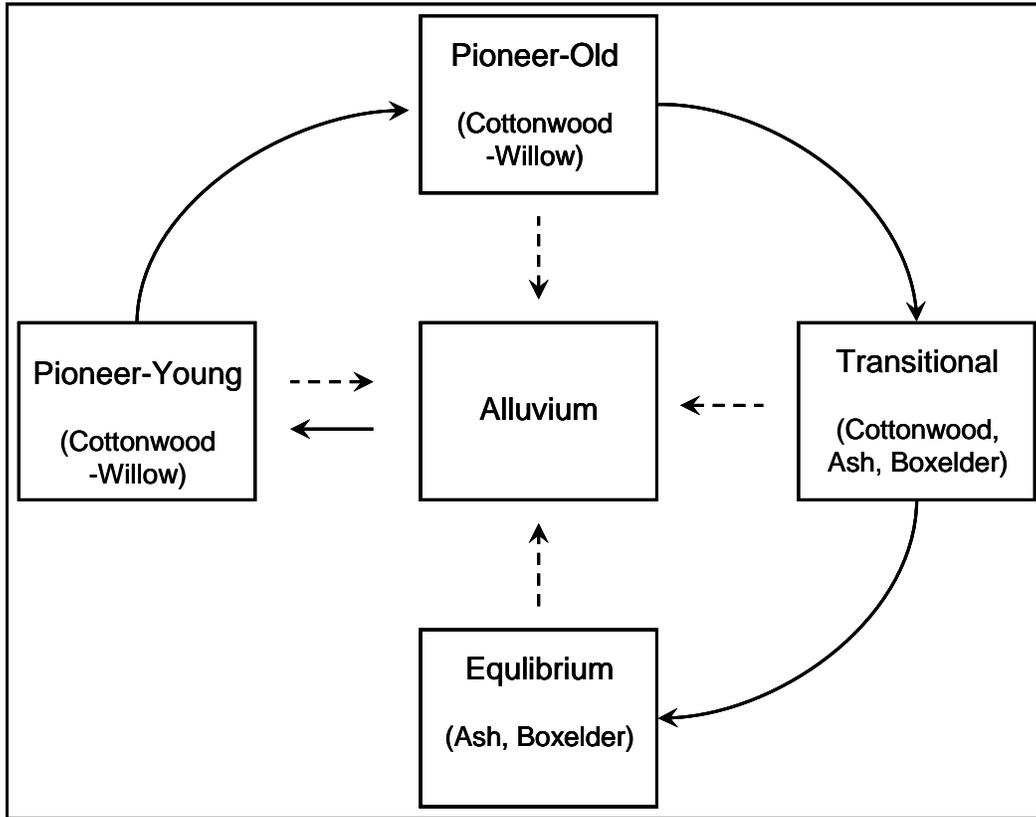


Figure 14. Conceptual model of riparian succession relative to the creation of freshly deposited alluvial surfaces. Solid arrows represent possible successional pathways and dashed arrows represent erosional conversion to alluvium. After: Johnson (1992).

Upland disturbances primarily influence riparian and aquatic ecosystems indirectly, through hydrological and geomorphological processes that control the timing and amount of water and sediments delivered to streams from the watershed. Upland wildfire is an important disturbance factor, which reduces or eliminates vegetation cover and alters soil properties, contributing to accelerated hill-slope runoff, soil erosion and debris flows. This in turn results in altered stream flow and geomorphic processes in receiving streams, including higher stream peak discharges and increases in erosional and depositional processes (Christensen et al. 1989). Climate can have important influences on fire regimes, and reconstruction of Holocene fire history suggests that periodic climatic fluctuations, with the most recent being a rapid rise in temperatures during the twentieth-century, in conjunction with decreased precipitation, has contributed to increased occurrence of severe fires in the southwestern U.S. Thus, under warmer, drought-prone climatic conditions, severe stand-replacing upland fires and related erosional events (Pierce et al. 2004) would be expected to increasingly influence natural disturbance processes in riparian and aquatic ecosystems.

The historical importance of wildfire in riparian forests of the southwestern U.S. is not well established. However, it has been suggested that riparian fires have increased in both frequency

and intensity, due in part to reduced stream flows, lowered groundwater levels, and accumulations of dead and senescent plant materials produced by drought stress (Busch 1995, Ellis 2000). The efficient post-fire recovery of Tamarisk, relative to native riparian trees (Busch and Smith 1993), emphasizes the potential importance of fire in the structure and functioning of riparian ecosystems on the Colorado Plateau (Busch 1995).

Disturbance in aquatic ecosystems can be described in terms of frequency, intensity, predictability, time since disturbance, predation intensity, resource variability, and environmental heterogeneity. Responses to both natural and anthropogenic disturbances vary regionally, due to constraints imposed by geomorphic and hydrologic regimes. The role of disturbance in structuring aquatic communities has been described by a number of hypotheses including: the equilibrium hypothesis, the intermediate disturbance hypothesis, and the dynamic equilibrium hypothesis (Resh et al. 1988).

Historically, the equilibrium hypothesis, which assumes a constant environment, was viewed as the appropriate model for describing aquatic community structure. This model assumes that community structure is controlled by biotic processes. Therefore, in the absence of disturbance, community structure is the direct result of competitive, mutualistic, and trophic interactions among species (Resh et al. 1988). The equilibrium hypothesis is most suitable for relatively stable environments. In the arid southwest, the highly variable physical environment plays an important role in structuring aquatic communities, and consequently disturbance models are more appropriate for describing the processes regulating biological diversity. The intermediate disturbance hypothesis (Hutchinson 1961, Connell 1978, Ward and Stanford 1983) suggests that intermediate levels of biotic or abiotic disturbances (e.g., frequency of substrate shifting or periodic flooding events) can promote maximum species diversity under certain circumstances (Figure 15). The intermediate disturbance hypothesis assumes a competitive hierarchy of species; thus, in the absence of disturbance, superior competitors will eliminate inferior ones and reduce species richness. In contrast, if disturbances are too frequent or too intense, the resident competitors will be eliminated and colonizing species will dominate the system (Resh et al. 1988). Maximum biotic diversity is maintained in aquatic systems by a level of disturbance that maintains environmental heterogeneity, but also allows biotic communities to become established (Ward and Stanford 1983).

In the dynamic equilibrium model, Huston (1979) suggested that if the recurrence interval of disturbance was shorter than the time necessary for competitive exclusion, then species that were poorer competitors would persist and maintain high species richness. In some cases, however, disturbance could be severe or frequent enough to eliminate species with long life cycles and species richness would decline. This model allows for differentiation between rarely disturbed systems, and those, including many stream systems of the Colorado Plateau, with “opportunistic” community types associated with frequent and/or intense disturbance (Resh et al. 1988). In an example of one such system, Reice (1985) found that frequent floods or spates kept the macroinvertebrate community in a state of perpetual disequilibrium, which limited competitive exclusion and thus maintained high species richness.

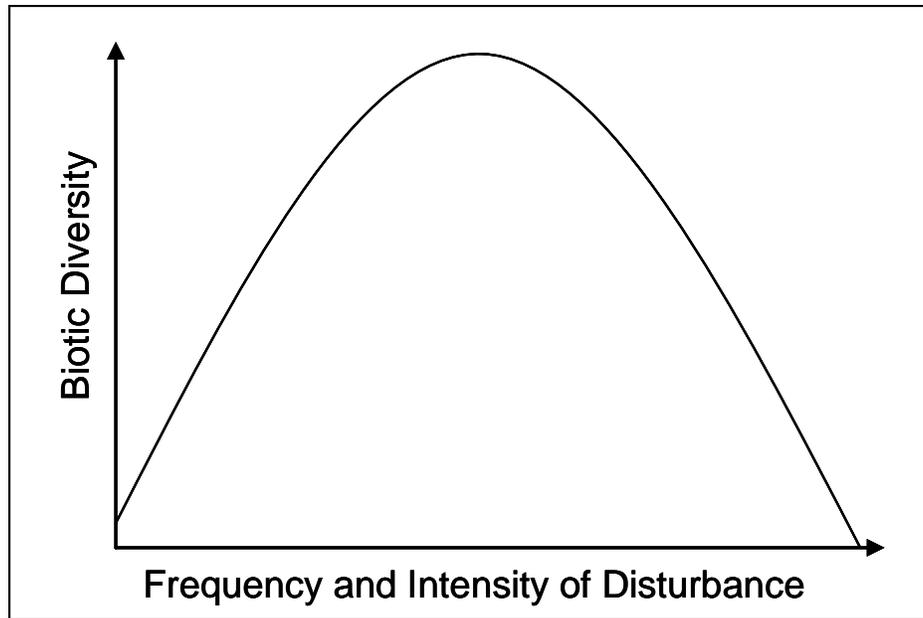


Figure 15. According to the intermediate disturbance hypothesis, intermediate levels of disturbance result in the highest levels of diversity. Adapted from Hutchinson (1961).

vi. Flood plain Soils

The soils of a riparian ecosystem differ from those of both upland systems and permanently flooded bottomlands. Shallow alluvial groundwater is a unique and important functional feature of riparian flood plain soils, and is tightly linked to surface water dynamics (see *Alluvial Groundwater*, under Section II. B. 1. b. iii). Native and non-native woody phreatophytes, like cottonwood, willow and tamarisk are dependent, to varying degrees, upon shallow alluvial groundwater sources. Spatially complex moisture gradients resulting from flood-plain topographic diversity and surface and ground-water dynamics, influence the diversity of herbaceous riparian plants and soil organisms (Meinzer 1927, Scott et al. 1997, Stromberg et al. 1997, Pollock et al. 1998, Horton et al. 2001a, Bagstad et al. 2005, Beauchamp 2004). Because of their dynamic nature, flood plain soils of riparian ecosystems in dry regions of the U.S. are typically young and poorly developed, often lacking the distinct horizons of soils formed by the interaction of weathering processes and living organisms over time. Many of these soils, particularly at lower elevations of the Plateau, may lack an aquic moisture regime, which requires that soils be saturated long enough to become anoxic and to develop distinctive redoximorphic features such as gleying (Brady 1974). The combination of fine-textured soils, high organic matter and high nutrient content, alternating periods of wetting and drying, and anaerobic versus aerobic conditions, which make flood plain soils in more humid regions so biogeochemically dynamic, are generally lacking in southwestern flood plain soils (Mitsch and Gosselink 1993). In dry-region riparian ecosystems, nutrient availability is likely more related to nutrient flux in stream flow than soil stores, although these fluxes are poorly understood (Schade et al. 2002). Freshly deposited alluvial sands are typically low in nitrogen and riparian plants colonizing these surfaces are nitrogen-limited (Adair and Binkley 2002). In general, the periodic wetting and drying of riparian soils is considered important in the release of nutrients from leaf litter in riparian environments (Mitsch and Gosselink 1993). On the flood plain of the Yampa

River, Colorado, however, no pulse of N was detected from leaf breakdown and instead, leaf litter appeared to remove N from floodwaters (Anderson et al. 2003). Retention of floodwater-N in litter, subsequent N mineralization, and uptake by riparian plants or soil organisms, was hypothesized as a mechanism for the net accumulation of N in nitrogen-poor, dryland flood plain soils (Anderson et al. 2003).

vii. Instream Physical and Chemical Conditions

Associations among biological stream communities and habitat characteristics, specifically physical and chemical conditions, have been well described in recent studies outside of the Colorado Plateau (e.g. Lyons 1996, Lohr and Fausch 1997, Maret et al. 1997, Brown 2000, Waite and Carpenter 2000). These studies provide general information on how physical and chemical parameters influence macroinvertebrate community structure. For example, the combination of stream flow variability and channel features such as pools, riffles, meander loops, and bars, create a diversity of instream microhabitats that vary spatially in terms of water depth and velocity. This microhabitat diversity plays an important role in structuring macroinvertebrate and fish communities (Baltz et al. 1991, Gordon et al. 1992, Munn et al. 2002, Brown 2002). Habitat homogenization reduces microhabitat diversity, resulting in detrimental effects on macroinvertebrates, as shown by a decrease in species richness and diversity. The key characteristics of water resources within the Colorado Plateau are that they are relatively limited and isolated, and that many of the streams and pools in the area are characterized by drought and flooding regimes, and consequently support an aquatic biota adapted to such flashy systems (Stanley and Fisher 1997).

Chemical Conditions. Due to its influence on habitat quality, water chemistry significantly affects composition, abundance, and diversity of macroinvertebrate species (Johnson et al. 1993). Of these chemical features, dissolved oxygen and conductivity (salinity or hardness) are the most influential (Thorp and Covich 1991). Increases in salinity and alkalinity and decreases in dissolved oxygen are correlated with decreases in macroinvertebrate density and diversity (Earl and Blinn 2003). In unaltered streams, dissolved oxygen is rarely limiting, but can become a critical environmental variable in polluted or diverted streams, where oxygen concentrations can decrease precipitously (Allan 1995). Following forest wildfires in southwestern New Mexico, ash input from the fires resulted in increased alkalinity, potassium, nutrients, pH, conductivity and turbidity, and decreases in dissolved oxygen (Earl and Blinn 2003). Associated with these changes in water chemistry, macroinvertebrate density was reduced in the ashed reach for nearly a year. In addition, macroinvertebrate drift was substantially higher when compared with a reference reach, and there were significant differences in macroinvertebrate community structure between ashed and non-ashed reaches (Earl and Blinn 2003).

Anthropogenic pollution can degrade the integrity of aquatic ecosystems by either altering natural chemical parameters, or by introducing organic and inorganic toxicants (Thorp and Covich 1991). Because macroinvertebrates are affected by water chemistry and show greater sensitivity to toxicity than other aquatic organisms, they have been used as bio-indicators of water quality in place of direct water chemistry analysis (Rosenberg and Resh 1993). Certain macroinvertebrate taxa are more sensitive than others to specific chemicals. For example, plecopterans and baetids (Ephemeroptera) are very sensitive to insecticides, whereas other taxa are more sensitive to chemicals such as herbicides, fungicides, and industrial chemicals

(Rosenberg and Resh 1993). Macroinvertebrates along Salt Creek in CANY have been inventoried (Banta 2002) and continue to be monitored in relation to water quality (Charlie Schelz, personal communication). Water chemistry is covered extensively in associated Colorado Plateau Inventory and Monitoring water quality protocols and therefore is not discussed further here.

Physical Conditions - Flow Regime. Flow regime is an important determinant of aquatic community structure. Streamflow can strongly affect habitat characteristics, dispersal, resource acquisition, competition and predation of macroinvertebrates. The mechanisms by which flow affects benthic organisms include both direct and indirect paths. One direct effect of flow on macroinvertebrates occurs when the hydrodynamic forces physically displace the organism (drift). Many species have adapted morphological and behavioral responses to certain flow characteristics, and invertebrates in the Virgin River and associated tributaries seem to be adapted to the frequent water level fluctuations and flooding within the Zion Narrows (Workman 1980). Indirect effects of flow occur via an intermediate abiotic or biotic variable, which in turn affects the local macroinvertebrate community. For example, flow can determine the distribution of sediment particle sizes available in a stream reach, which in turn may affect macroinvertebrates adapted to specific substrate sizes (Hart and Finelli 1999).

Many human activities (dams, diversions, channelization, ground-water withdrawal) modify the natural flow regime in streams and rivers. Dams can disrupt the longitudinal linkages and decrease the availability and complexity of available habitats (Grubbs and Taylor 2004). Dams and diversions often reduce flow variability which can negatively affect fauna adapted to highly variable flow conditions (McIntosh et al. 2002). This can result in decreased macroinvertebrate abundance and diversity, and cause a shift from lotic species to lentic species (McIntosh et al. 2002, Grubbs and Taylor 2004). Interestingly, Pippin and Pippin (1980, 1981) found that there were slight increases in macroinvertebrate diversity and abundance in Frijoles and Capulin Creeks associated with decreased flood events. Regulated streams may have slower current velocity and increased sediment loads, resulting in a decrease in filtering-collector macroinvertebrates and an increase in sediment-tolerant species (Grubbs and Taylor 2004). Increasing the duration of dewatering may increase the dessication of potential colonization sources (in wetted upstream areas and the hyporheic zone) and thus restrict community recovery, with longer dewatering periods favoring colonizing macroinvertebrates (Fowler 2004).

Indirect effects of flow alteration may occur as associated changes in fish and algal communities result in changes to macroinvertebrate community structure (Meffe 1984, McDowell 2003). Altered flow regimes often favor introduced fish species that are generalists and can tolerate a wide array of environmental conditions (Meador et al. 2003). Studies have shown that when flows are reduced, fish species richness is also reduced (Cuffney et al. 1997). These changes in predator community (fish) can result in changes to the prey community (macroinvertebrates). Because of different feeding habits, introduced fish can alter macroinvertebrate community structure (McDowall 2003). Algal community structure will also change in response to flow regime (Munn et al. 2002), which can in turn alter macroinvertebrate communities. Algae may take considerable time to recover following drying of a stream, so macroinvertebrates that rely on algae as a food source (grazers) may be less abundant initially after rewatering (Fowler 2004). In one study, changes in the algal community resulting from low flow in a Colorado stream

resulted in a shift from a collector-gatherer macroinvertebrate community to a shredder community (Canton et al. 1984).

Physical Conditions - Substrate. Watershed factors, including flow regime and geology, determine channel substrate composition (Allan 1995), an important habitat feature for macroinvertebrates (Thorp and Covich 1991). Substrate provides sites for resting, food acquisition, reproduction, and development, as well as refuge from predators and physical disturbance. Different groups of macroinvertebrates require different substrate types and microhabitats (Gordon et al. 1992). These groups also play different functional roles in their environment. For example, detritivores such as oligochaetes and crustaceans, live in fine sediments, mix fine particulate organic matter, and stabilize soil structure, whereas shredders such as stoneflies, shred coarse particulate organic matter and prepare it for decomposers (Freckman et al. 1997).

In general, diverse substrate characteristics promote diverse taxonomic assemblages, and both diversity and abundance of aquatic macroinvertebrates have been shown to increase with substrate stability and the presence of organic detritus (Allan 1995). Flow diversion, erosion, or trampling can reduce substrate diversity and can thus reduce macroinvertebrate diversity. Substrate embeddedness (increased siltation) can result in reduced fish and macroinvertebrate species diversity, and alter algal assemblages (Cuffney et al. 1997). A study of macroinvertebrate communities in the Gore Creek Watershed, Colorado, documented low species abundance at sites with high sediment loads (Wynn et al. 2001). However, many aquatic plants may prefer finer substrates and, once established, may act as substrate for other organisms (Gordon et al. 1992). High sediment loads can minimize light availability and create an allochthonous riparian-based macroinvertebrate community (Haden et al. 2003). With small amounts of sediment, density and abundance of macroinvertebrates may be decreased due to reduction of interstitial habitat, although structure and species richness may not change. Greater sediment amounts that drastically change substrate type (i.e., from cobble-gravel to sand-silt) will change the number and type of taxa to more sediment-tolerant species, thus altering community structure and species diversity but often with increasing densities of macroinvertebrates (Lenat et al. 1979).

Physical Conditions - Temperature. Numerous factors including climate, elevation, extent of riparian vegetation, and relative importance of groundwater inputs have an effect on water temperature regimes (Allan 1995). Aquatic insects respond to the entire thermal regime, which is a composite of patterns of absolute temperatures, diel and seasonal amplitudes, and rates of change (Ward and Stanford 1982). Water temperature plays a major role in the ecology and evolution of aquatic macroinvertebrates because it directly influences the metabolic rates, physiology, and life-history traits and helps to determine rates of important processes such as nutrient cycling and productivity (Ward and Stanford 1982, Poole and Berman 2001). Because temperature influences fecundity, dormancy, growth and maturation, time of emergence, and survival, it will ultimately alter macroinvertebrate community structure (Ward and Stanford 1982, Vinson 2001).

The temperature of large rivers is unlikely to be affected by riparian shading, as their size conveys considerable thermal inertia and results in reaches that are largely exposed to the sun

(Allan 1995). In small streams, such as many of those along the Colorado Plateau, shading can play an important role in regulating water temperatures. Many anthropogenic stressors, such as grazing, roads, and stream channelization, reduce or eliminate riparian vegetation cover along the banks and thus reduce shading, leading to increased water temperature.

Fluctuations in water temperature induce behavioral and physiological responses in macroinvertebrates, and permanent shifts in stream temperature regimes can create habitat unsuitable for temperature-sensitive species (Poole and Berman 2001). For example, downstream from a cold water-release dam, macroinvertebrates tolerant of lower temperatures, such as chironomids and amphipods, were dominant, and less tolerant species of the orders Plecoptera, Ephemeroptera, and Trichoptera were uncommon (Stevens et al. 1997). Similarly, if water temperature is increased, native cold-water taxa can be replaced by non-native warm water taxa (Maret 1995). Changes in water temperature also alter fish and algal assemblages, which will result in changes to macroinvertebrate communities as they respond to changes in algal and fish communities (Baltz et al. 1987, Stevens et al. 1997).

c. Interactive Controls-Biotic Functional Groups

i. Background

Chapin and colleagues (1996) identified biotic functional groups (hereafter described as *functional types*) as one of the four interactive controls of ecosystem sustainability because of the capacity of dominant functional types to shape the structure and functioning of whole ecosystems. Associated with efforts to model ecological consequences of global climate change, a vast literature has developed concerning different approaches to deriving or classifying functional types – particularly with respect to vegetation (e.g., Smith et al. 1997). Identification and use of a particular functional-type scheme depends on the ecosystem function(s) of interest. It has been proposed that the most important functions in dryland terrestrial ecosystems are those that control the retention of water and nutrient resources because productivity and diversity cannot be sustained in systems that fail to retain resources (Ludwig and Tongway 1997, Whisenant 1999, Whitford 2002). Because of their landscape position and highly connected linear forms, riparian and aquatic ecosystems receive large fluxes of water and sediment from upland and upstream sources. Similarly, their potential to store flood water and nutrient-rich sediments, are considered key functional attributes (Mitsch and Gosselink 1993). Functions affecting the cycling and retention of water and nutrient resources will be emphasized here, but other functions will not be excluded. For purposes of this report, it is less important to adopt a specific functional-type classification scheme than it is to include a broad functional perspective when considering the biotic components of riparian and aquatic ecosystems.

Without adopting a particular classification scheme, it remains useful to identify two general categories of functional types that are equally important for ecosystem dynamics. These are (1) *functional effect types* –organisms with similar effects on ecosystem functions such as primary production, nutrient cycling, and sediment trapping, and (2) *functional response types* – organisms with similar responses to environmental factors such as climate, resource availability, natural disturbances, and water management activities (Walker 1997, Walker et al. 1999, Díaz and Cabido 2001). The distinction between these two types is important for considering how biotic composition affects the resistance and resilience of ecosystems to climatic fluctuations and changes, natural disturbances, and anthropogenic stressors (Walker et al. 1999). Although some

workers have emphasized the importance of overall functional diversity for sustaining ecosystem processes (Tilman et al. 1997), the effect-response distinction suggests that long-term ecosystem functioning may be favored when different functional response types are nested within the same functional effect type (Walker et al. 1999, Díaz and Cabido 2001). Thus, functional redundancy and functional diversity may both be important for long-term persistence of ecosystem structure and functioning.

ii. Flood plain Soil Biota

Flood plain soil biota represents a broadly defined group of organisms that is an important contributor to the structure and functioning of riparian ecosystems. Most of the ecosystem processes associated with soil resources (i.e., nutrient cycling, water infiltration and storage, soil aggregate stability, water and nutrient uptake by plants) are mediated by soil organisms (Skujins 1984; Whitford 1996, 2002; Lavelle 1997; Wardle 2002). Although the general significance of soil biota for ecosystem processes (particularly nutrient cycling) has long been acknowledged, there is increasing recognition that this diverse group of organisms must be considered much more explicitly in order to develop a better understanding of the structure and functioning of terrestrial (Wardle 2002, Reynolds et al. 2003) and likewise riparian ecosystems. Because of their intimate association with other components of riparian ecosystems, soil biota in Figure 5 are included in components identified as flood plain soils and soil resources, vegetation, and invertebrates.

Soil biota include microfloral components (bacteria, algae, and fungi), microfaunal components (nematodes, microarthropods, and protozoans), and macrofaunal components (earthworms, ants, termites, and larval stages of several insect families) that are involved in a variety of processes essential for litter decomposition and nutrient cycling. Functioning of these belowground processes is dependent on the amounts and types of organic-matter inputs from vegetation and on soil conditions such as moisture availability (which is strongly influenced by surface and ground-water dynamics), soil structure, soil aeration, and soil temperature (Whitford 1996, 2002; Mitsch and Gosselink 1993).

Mycorrhizal fungi, which form symbiotic associations with roots of many plant species, are another important element of the soil biota. The mycorrhizal symbiosis is one in which the fungal partner provides nutritional benefits to the host plant, and the plant provides carbohydrates to the fungi (Smith and Read 1997). Roots colonized by mycorrhizal fungi acquire phosphorus, zinc, and possibly copper and N more efficiently than un-colonized roots. There is also evidence that mycorrhizae can increase water uptake in plants due to the greater soil volume accessed by colonized roots (Smith and Read 1997). Arbuscular mycorrhizal fungal communities have been described for a number of ecosystems, however comparatively little is known about the structure and composition of these communities in riparian ecosystems. Recent research in cottonwood/willow forests along regulated and unregulated reaches of the Verde River, Arizona, indicates that fungal colonization rates and diversity increased with increases in the diversity of perennial plant species and decreased with increases in stand age, as well as distance from and elevation above the channel. Stand age, soil moisture and soil texture appeared to be important environmental determinants of fungal community structure, and whereas most species found in these riparian settings are also found in adjacent desert uplands,

diversity was higher in the riparian zone and two species were restricted to these sites (Beauchamp 2004).

Some species common to riparian ecosystems have been identified as mycorrhizal when inspected by botanists (Trappe 1981). Families with a high frequency of mycorrhizal colonization among inspected species included the Asteraceae, Fabaceae, Rosaceae, Poaceae, and Solanaceae. The Brassicaceae stands out as a relatively common riparian plant family in which most inspected species were non-mycorrhizal (Trappe 1981).

Another important symbiotic relationship involving soil biota is that between plants and N-fixing bacteria. Several native and non-native shrubs that are locally common or abundant in riparian ecosystems throughout the Colorado Plateau region are capable of forming a symbiotic association with N-fixing actinomycetes. Actinorhizal shrub genera found in riparian zones throughout the region include *Alnus* (Betulaceae), *Purshia* (Rosaceae), *Shepherdia* and *Elaeagnus* (Elaeagnaceae) (Schwencke and Carú 2001). The frequency of actual actinorhizal colonization in these genera and the overall contributions of this relationship to riparian N cycling are poorly understood.

iii. Riparian Vegetation

At a broad level, vegetation is generally recognized as the dominant functional type in riparian ecosystems. In addition to conducting photosynthesis, the aboveground structure of vascular plants increases roughness and thus protects flood plain soils from erosion and enhances the deposition and retention of nutrient-rich sediments during floods. Litter from plants reduces the erosive impacts of rainfall on soil surfaces and provides inputs to soil organic matter for nutrient cycling. Aboveground structures of riparian plants modify the physical environment by shading and litter deposition, strongly affecting spatial and temporal patterns of soil-resource availability to other organisms. Vegetation structure helps create gradients of moisture and temperature that are important to maintaining biotic diversity. Roots stabilize soils and stream-banks, are conduits for resource acquisition and redistribution, and provide organic-matter inputs to soil food webs. Vegetation also provides fuel for fire, as well as resources and habitat structure for belowground and aboveground consumers and decomposers ranging from fungi and bacteria to birds and mammals (Brinson et al. 1981, Whitford 2002, Wardle 2002). Finally, carbon storage and the mediation of earth-atmosphere energy / water balances are additional ecosystem functions performed by vegetation that are increasingly important with respect to global-change processes (Breshears and Allen 2002, Asner et al. 2003).

A large number of vegetation attributes affects the manner and extent to which these functions are performed. Size, biomass, photosynthetic rate, relative and absolute growth rates, tissue chemistry, stem basal area, canopy cover, vertical canopy structure, spatial arrangement and contiguity, leaf area, leaf longevity, and plant life-span are some of the more important vegetation attributes for ecosystem functioning (Chapin 1993). Root distribution, reproductive traits, moisture requirements, and phenology are additional functional attributes of vegetation that are particularly important in riparian ecosystems. With respect to disturbance interactions, important functional attributes include palatability, flammability, and mode of post-disturbance regeneration.

In dry regions like the Colorado Plateau, riparian ecosystems often appear as visually distinctive landscape elements, with the structure, composition, and density of riparian vegetation standing in sharp contrast to plant communities on adjacent uplands. Whereas the vertical and horizontal structure provided by woody riparian trees and shrubs functions as important habitat for an array of animal species, much of the plant diversity in riparian systems is found in the herbaceous community. The composition of the herbaceous community varies temporally within and across seasons, as well as spatially along moisture gradients created by fluctuations in stream flow. A diversity of obligate wetland herbs and grasses occupy channel bars, channel margins and backchannels, while a host of upland species occur on flood plains and alluvial terraces (Auble et al. 1994, 2005; Stromberg and Chew 2002). Textural differences within alluvial deposits, across a range of surface elevations, also influences plant species composition and diversity (Friedman et al. 1996, Jansson et al. 2000), emphasizing the importance of fluvial geomorphic processes in maintaining overall plant species richness in riparian ecosystems.

The two most frequently- occurring native tree genera in riparian ecosystems of the western U.S. are *Populus* and *Salix*. The non-native trees, tamarisk (*Tamarix*) and Russian-olive (*Elaeagnus*), represent the third and fourth most frequently occurring riparian genera (Friedman et al. 2005). Other important native tree genera in the region include *Acer*, *Fraxinus*, *Celtis*, *Alnus*, and *Betula*. Important native shrub genera include *Prunus*, *Cornus*, *Shepherdia*, *Chrysothamnus*, *Sarcobatus*, *Symphoricarpos*, *Rosa*, *Purshia*, and *Fallugia*. Some of the more important native herbaceous genera include *Bidens*, *Cardamine*, *Carex*, *Eleocharis*, *Equisetum*, *Glycyrrhiza*, *Juncus*, *Schoenoplectus*, *Scirpus*, *Solidago*, *Typha*, *Veronica*, and *Xanthium*. Genera of annual and perennial grasses include *Agrostis*, *Alopecurus*, *Calamagrostis*, *Distichlis*, *Echinochloa*, *Glyceria*, *Hordeum*, *Leersia*, *Phragmites*, *Spartina*, and *Sporobolus*.

Provision of habitat for a diverse array of secondary consumer and decomposer communities is another important functional attribute of riparian vegetation. Undisturbed riparian ecosystems are recognized as being especially diverse biologically. The importance of riparian ecosystems in this regard is attributed to a unique combination of physical and biological characteristics, including: (1) a predominance of woody plants; (2) at least a seasonal presence of surface water and high soil moisture; (3) an interspersed of diverse structural elements that create high habitat patch diversity; and (4) a linear form with high upstream-downstream connectivity, that provides for uniform, protected pathways for migration and movements between different habitat types (Brinson et al. 1981).

Many of the functional attributes described above differ greatly among vegetative life forms. For example, there are relatively large differences among riparian trees, shrubs and herbs in terms of canopy height, architecture and spatial arrangement, as well as in their responses to climate, fire and herbivory. As a consequence, ecosystems characterized by different proportions of trees, shrubs, herbs, and grasses can be expected to differ greatly in terms of associated ecosystem processes including nutrient cycling, hydrologic regimes, disturbance regimes, and wildlife-habitat relationships. Likewise, temporal shifts in the relative abundance and spatial configuration of vegetative life forms can significantly affect the functioning of an array of ecosystem processes.

d. Other Biotic Components

Terrestrial Invertebrate and Vertebrate Communities. The presence of water, nutrient-rich soils, and the interspersed nature of a variety of successional aquatic and terrestrial biotic communities make riparian ecosystems, particularly in arid regions, more productive and biologically diverse than surrounding uplands (Lugo et al. 1990; Knutson et al. 1996). The physical and biotic components of riparian ecosystems have important influence on the biota of stream ecosystems, but in this section we focus on non-aquatic invertebrate and vertebrate communities. Vertebrate and invertebrate communities are significant contributors to the biological diversity of riparian ecosystems in arid regions (e.g., Stevens et al. 1977, Brode and Bury 1984, Falck et al. 2003, Fleishman et al. 1999). There are numerous ways in which above-ground consumers can directly or indirectly affect the structure and functioning of riparian ecosystems. Activities associated with herbivory, trampling, and ponding are among those that have the greatest ecosystem-level consequences for riparian and aquatic ecosystems due to their many effects on vegetation structure and flood plain soil processes. Processes of competition and predation can likewise have important ecosystem-level consequences by altering the structure of consumer food webs, but these processes are not reviewed here.

Herbivory can have numerous direct and indirect effects on ecosystem properties. Native herbivores in riparian ecosystems of the region include insects (grasshoppers, chrysomelid beetles, and others), and mammals such as beaver (*Castor canadensis*), deer mouse (*Peromyscus maniculatus*), montane vole (*Microtus montanus*), and mule deer (*Odocoileus hemionus*). In some locations in the region, use of riparian systems by elk (*Cervus elaphus*) and moose (*Alces alces*) also can be significant, particularly in winter (Allen 1989, Hobbs 1996). Herbivorous insects and small to medium-sized mammals can have significant effects on riparian and wetland vegetation structure, reproductive patterns, and ecosystem processes such as decomposition and nutrient cycling (Wallace and O'Hop 1985, Scott and Haskins 1987, Anderson and Cooper 2000). Perhaps the greatest ecosystem-level consequences for riparian ecosystems are those activities associated with biophysical alterations, such as dam building by beaver and structural habitat modifications resulting from herbivory and trampling, caused by large-bodied browsers and grazers, including mule deer, elk and domestic livestock. At certain levels, these activities contribute to the overall biodiversity of riparian ecosystems by creating a dynamic mosaic of different habitat patch types (Naiman and Rogers 1997). However, chronic, high densities of large-bodied browsers and grazers may ultimately lead to habitat simplification and loss of biodiversity (Kauffman and Krueger 1984, Taylor 1986, Scott et al. 2003).

Large herbivores can affect individual plants both directly and indirectly through a variety of mechanisms. Direct impacts include altered physiological function and morphology attributable to defoliation and trampling (Briske 1991, Briske and Richards 1995). Defoliation and trampling by large herbivores may indirectly influence plant performance as a consequence of altered micro-environmental conditions, soil properties (Thurow 1991), mycorrhizal relations (Bethlenfalvay and Dakessian 1984), competitive relations, and through effects on ecosystem processes such as nutrient cycling and channel and flood plain formation. Seed dispersal is yet another indirect mechanism by which large herbivores and other animals may affect vegetation structure. Through time, combined direct and indirect impacts can result in altered plant population dynamics (e.g., altered rates of reproduction, recruitment, and mortality) and

consequent changes in plant community composition, structure, and distribution (Brinson et al. 1981, Naiman and Rogers 1997). Due to strong interactions of vegetation with nutrient cycling, hydrologic processes, disturbance regimes, and geomorphic processes, herbivore-driven changes in vegetation structure can have cascading effects on multiple ecosystem processes and properties.

Large herbivores also can affect the productivity and composition of plant communities through numerous indirect and direct effects on nutrient cycling in upland (Archer and Smeins 1991) and riparian systems. Herbivore-driven shifts in plant community structure can affect nutrient cycles by altering the capacity of vegetation to capture and retain soil and water resources (Whitford 2002) and by altering the quantity and quality of organic-matter inputs (Bardgett and Wardle 2003). Herbivory removes foliage and directly diverts nutrients from litter and physiological processes of intra-plant cycling. Nutrients acquired from foliage may be incorporated in animal biomass or spatially redistributed across the landscape in urine and dung. Where excreta are deposited, productivity may be enhanced if nutrients contained in the excreta are accessible to nearby plants. In other portions of the landscape, productivity may be reduced due to the removal of nutrients in foliage.

Aquatic Invertebrate and Vertebrate Communities. Aquatic biota includes four components: algae, benthic macroinvertebrates, fish and amphibians. These groups interact directly with each other as well as with terrestrial vertebrates and invertebrates and riparian vegetation. Monitoring of aquatic biota at parks within the Colorado Plateau is based upon macroinvertebrates, and consequently, the focus for this discussion is on benthic macroinvertebrates.

Macroinvertebrates play a key role in stream ecosystems due to their intermediate position in the food chain, linking allochthonous/autochthonous production with higher trophic levels, such as fish (Munn and Brusven 1991). Trophic dynamics regulate the movement of carbon, nutrients, and energy among organisms in an ecosystem (Chapin et al. 2002). In complex food webs, nutrients and energy of one trophic level are utilized by organisms from several different trophic levels (Wetzel 1983). The transfer of energy and nutrients from their original sources to successive trophic levels occurs through photosynthesis, bacterial decomposition, or the feeding of herbivorous and carnivorous animals (Goldman and Horne 1983).

In a simplified aquatic food web, energy inputs might include coarse particulate organic matter (CPOM), fine particulate organic matter (FPOM), dissolved organic matter (DOM), and light. Microbes and shredders break down CPOM, such as fallen leaves, and create more FPOM, which serves as an energy source for collectors. Light is an energy source for algae that then become energy for grazers. Grazers, collectors, and shredders are energy sources for predators such as fish and carnivorous macroinvertebrates (Alan 1995). In order to trace the energy through a food web, such as the one described above, the contribution of each trophic level to the diet of each animal in the ecosystem must be known (Chapin et al. 2002). This has become easier to test in recent years through stable isotope analysis. Typically, organic sources of an aquatic food web are determined by the ratio between ^{12}C and ^{13}C ($\delta^{13}\text{C}$), while trophic position is assigned with a ratio between ^{14}N and ^{15}N ($\delta^{15}\text{N}$) (Shannon et al. 2001). Trophic linkages have been detailed in the lower Colorado River in Glen and Grand Canyons using $\delta^{13}\text{C}$, $\delta^{15}\text{N}$, and $\delta^{34}\text{S}$ (Angradi 1994, Shannon et al. 2001).

The relative abundance of different types of primary producers (algae) depends on many factors including nutrient availability, water depth and velocity, the stability of the substrate, and disturbance regime. Unshaded streams can support dense algal growth (autochthonous productivity), but shaded streams rely more on riparian (allochthonous) inputs where algal growth is minimal (Covich et al. 1999). Although the function of algal assemblages is similar in both shaded and non-shaded systems, the magnitude of algal contribution to aquatic primary productivity is much higher in systems with minimal riparian input. Primary producers can act as an interface between the physical environment and macroinvertebrate communities. Macroinvertebrate grazers consume algae and therefore the type and abundance of algae can strongly influence macroinvertebrate community structure. Conversely, macroinvertebrate grazing will determine the type of algal communities present (Steinman 1996). An increase in algal abundance is often associated with an increase in macroinvertebrate density and growth, while decreases in algal abundance are associated with reduced macroinvertebrate densities (Feminella and Hawkins 1995).

Algal communities can be substantially altered by disturbances such as dam construction, grazing, and agriculture. (Shannon et al. 1994, Haefner and Lindahl 1991). These changes in algal community structure can result in changes in macroinvertebrate species composition and abundance. A number of studies on algal communities (Smith and Piccin 2004) and algal response to disturbance regime (e.g. Shannon et al. 1994, Haefner and Lindahl 1991, Angrandi 1994, Stevens et al. 1997, Benenati et al. 2000) have taken place within the Colorado Plateau.

Fish communities play an essential role in aquatic systems throughout the Colorado Plateau. Macroinvertebrates are an important food source for fish, thus influencing fish community structure while predation by fish, in turn, influences macroinvertebrate community structure. Isotopic analysis has confirmed three main trophic levels in aquatic systems on the Colorado River: algae, macroinvertebrates, and fish (Angradi 1994). Because these trophic levels are mutually dependent, disturbance to one affects the other, as demonstrated by experiments on the Green River that demonstrated that fish exert a large influence on macroinvertebrate community structure (Collins and Shiozawa 2001).

Over the last century, there has been a rapid decline in populations of native fishes in the Southwest, largely due to habitat changes associated with human modifications (eg. dams and irrigation) (Minckley and Deacon 1968) and the introduction of exotic fish species. Abiotic factors (flash floods and water temperatures) play an important role in mediating the outcome of biotic interactions between native and introduced fish in fluctuating streams throughout the southwest (Castleberry and Cech 1986). In this arid region, native fish are better adapted to floods than introduced fish, and if flooding is frequent, populations of introduced fishes are reduced, allowing coexistence of native and introduced species (Meffe 1984). When abiotic disturbances occur less frequently, native populations decline as introduced fish populations increase (Meffe 1984). This cycle is intensified because many introduced fish prey on native fry. Macroinvertebrate communities are affected by the change in fish communities due to the different feeding habits of native and introduced species (McDowell 2003). Most of the aquatic ecological studies conducted on the Colorado Plateau have focused on large rivers such as the Colorado and Green Rivers, and on fish communities (Haden et al. 2003) and the status of native

fish communities are currently a high priority vital sign for a number of parks throughout the plateau.

Amphibians provide an obvious link between aquatic and terrestrial systems in a riparian setting because their life history includes an aquatic larval stage (that feeds on algae and macroinvertebrates) and an adult stage that feeds on terrestrial invertebrates. A variety of amphibian species have declined in number throughout the Southwest in recent years, and some species are federally listed or candidate species under the Federal Endangered Species Act (Thomas et al. 2004). In addition, several different amphibian species have been studied in National Parks and Monuments in the region (Haefner and Lindahl 1988, Woodbury 1933, Berghoff 1995, Lafrancois 1996, Fridell et al. 2000, Graham 2002), including endangered leopard frogs (Thomas et al. 2004). Amphibians are further discussed in the Inventory and Monitoring protocol (Graham, in preparation).

Benthic macroinvertebrates play a crucial role in both aquatic and riparian systems. They are a food source for fish, amphibians and birds, and they also act as a consumer, as grazers on algae, shredders of plants and leaves (riparian input), and collector-gatherers (consuming detrital material). Macroinvertebrates respond to physical parameters (temperature, substrate, and flow velocity) (Covich et al. 1999) and chemical conditions (pH, conductivity, contaminants, and dissolved oxygen), both of which influence resource availability and habitat quality. Biotic factors (predation, parasitism, competition) and food availability (the relative contribution of autochthonous versus allochthonous inputs) in the system can influence species composition and abundance (McCafferty 1998, Power 1990). Macroinvertebrates are also directly influenced by riparian vegetation which provides nutrients and physical habitat.

Macroinvertebrates are frequently used to assess stream quality as: (1) they are ubiquitous and consequently can be affected by environmental perturbation in a variety of aquatic systems and habitats, (2) the large number of macroinvertebrate species offers a wide spectrum of responses to environmental stressors, because different species require different habitat and water conditions, (3) their basic sedentary nature allows effective spatial analysis of disturbance effects, and (4) they have relatively long life cycles, which allows elucidation of temporal changes caused by perturbations (adapted from Rosenberg and Resh 1993).

Few ecosystems possess either the frequency or intensity of environmental change that are observed in stream systems (Power et al. 1988), particularly in the arid southwest with extreme environmental conditions including drought and floods. The native macroinvertebrates of the desert southwest are well adapted to this disturbance regime. Since macroinvertebrate fauna in the streams of the Colorado Plateau are frequently exposed to unpredictable floods and dry periods, these streams are dominated by mayflies, small diptera, and other taxa with shortened aquatic developmental stages or the ability to rapidly recolonize disturbed habitats. Although such disturbance events can alter the structure of aquatic communities, they are critical to the life histories of many native macroinvertebrates in the Colorado Plateau. It has been suggested that streams with flashy hydrology should have less abundant and less varied fauna than non-flashy systems (Hynes 1970, Baron et al. 1998). This has been supported by aquatic surveys conducted in the Colorado Plateau (Workman 1980, Baron et al. 1998, Benenati 1998).

There have been a number of surveys that have examined the general taxonomy of macroinvertebrate species across the Colorado Plateau (e.g. Wiersema et al. 2004, Moulton and Stewart 1997, Moulton et al. 1994). Other studies have examined macroinvertebrate response to a single variable in areas with known disturbance. For example, macroinvertebrate response to mining (Peterson et al. 2002), dams (Shannon et al. 2001, Benenati et al. 2000), drought (Dahm et al. 2003, Canton et al. 1984), and fire (Viera et al. 2004, MacRury and Clements 2002, MacRury 2002) have been examined. Studies on algal response to damming also provide limited macroinvertebrate inventories in the area around Glen Canyon Dam (Shannon et al. 1996, Benenati et al. 2000, Shannon et al. 2001). Macroinvertebrate community structure has been described for a number of streams in National Parks throughout the Colorado Plateau, including the Fremont River, Pleasant Creek, and Sulphur Creek at CARE (Kirby and McAllister 2000, Brammer and MacDonald 2003), the Virgin River and tributaries at ZION (Workman 1980, Shakarjian and Stanford 1998), Salt Creek at CANY (Banta 2002, Charlie Schelz, personal communication), and Capulin and Frijoles Creeks at BAND (Pippin and Pippin 1980, 1981).

In addition to studies on stream systems within the parks, baseline macroinvertebrate data is available for ephemeral systems in several of the National Parks across the Colorado Plateau (Berghoff 1995, Baron et al. 1998, Haefner and Lindahl 1988, 1991). Macroinvertebrate communities have been documented in seeps, springs, hanging gardens, ponds, pools and emergent wetlands in a number of National Parks including ZION (Woodbury 1933), GRCA (Sorensen and Kubly 1997), CHCU (Freehling and Johnson 2002), and in tinajas at CARE (Haefner and Lindahl 1988).

2. Ecosystem Dynamics

We conclude this section with a review of some of the leading concepts used to define and quantify riparian and aquatic ecosystem dynamics. Identifying and characterizing sources of variability in pattern and process within these systems is a critical first step in recognizing degradational changes resulting from anthropogenic stressors versus changes that result from natural variability.

Whereas riparian and aquatic systems of the Colorado Plateau can be relatively simple in structure and composition, they may be highly variable across temporal and spatial scales. The magnitude, frequency and duration of fluvial disturbance events, for example, can vary significantly within and across years. Additionally, other physical gradients, such as elevation, lithology, water temperature and depth to ground water are highly variable across spatial scales. Thus, heterogeneity in these systems is typically scale dependent.

The relationship between scale and heterogeneity in the riparian and aquatic ecosystems of the Colorado Plateau is well recognized. As a result, a number of concepts, hypotheses and / or theories have been proposed to help organize how we perceive variability in these systems across spatial or temporal scales. The following discussion of ecosystem dynamics is divided into two parts: (1) riparian ecosystem dynamics, conceptualized by longitudinal- and transverse-scale gradients (Bendix 1994) and (2) aquatic ecosystem dynamics, as described by the river

continuum concept (Vannote et al. 1980), the network dynamics hypothesis (Benda et al. 2004), and the habitat mosaic hypothesis (Stanford et al. 2005).

a. Riparian Ecosystem Dynamics

Two physical environmental gradients have been used to describe riparian ecosystem dynamics and diversity at different scales; longitudinal, or up-down valley gradients and transverse or cross-valley gradients. Longitudinal-scale variables including elevation, valley slope, valley width and lithology, influence riparian ecosystem dynamics at larger spatial scales, whereas smaller, transverse-scale variables include depth to the water table, flood frequency, flood intensity, and substrate texture (Bendix 1994). We briefly illustrate the influence of these factors on riparian ecosystem dynamics and diversity.

i. Longitudinal-Scale Gradients

In addition to longitudinal variation in elevation, the flow paths for most of the large extra-regional streams of the Colorado Plateau cross rock types or lithologies that vary considerably in their resistance to erosion, and as a consequence, these rivers occupy a range of valley types from deep, narrow bedrock canyons to broad alluvial valleys. These lithologic discontinuities exert strong influence on the longitudinal pattern and extent of riparian ecosystems. For example, planview maps of fluvial landforms supporting riparian vegetation, illustrate the influence of channel type and valley setting on the spatial pattern and extent of riparian vegetation along the Green River in DINO (Figure 16). Figure 16a illustrates the influence of tributary debris fans on the geomorphic organization of the channel and associated fluvial landforms in a narrow bedrock canyon. In these fan-eddy dominated canyons, fluvial landforms supporting riparian vegetation are restricted to small, spatially limited surfaces formed in association with pools and re-circulating eddies, that develop immediately up and downstream of

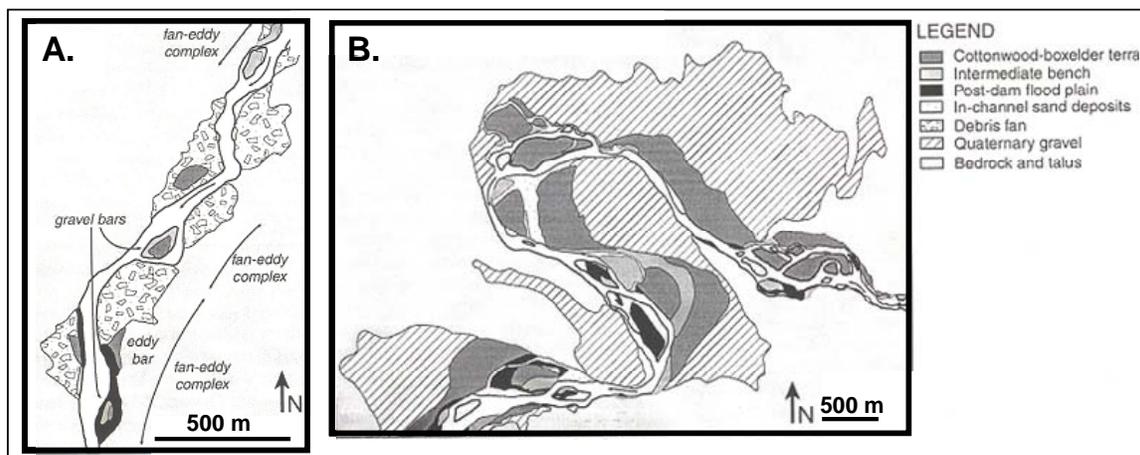


Figure 16. Planview maps of the superficial geology of the Green River in Dinosaur National Park, reflecting differences in surfaces supporting riparian vegetation (post-dam flood plain, Intermediate bench, and Cottonwood-boxelder terrace) between a) fan-eddy dominated, and b) restricted meander, channel types. From: Grams and Schmidt (2002).

debris fans (Schmidt 1990). In an alluvial reach consisting of restricted meanders (Figure 16b), landforms with riparian vegetation are larger and more spatially complex, reflecting active channel meandering and narrowing processes (Grams and Schmidt 2002).

ii. Transverse-Scale Gradients

The distinctive species composition and spatial patterning of riparian vegetation is in part related to transverse gradients of disturbance intensity and moisture availability created primarily by variations in surface flow, topographic diversity of fluvial landforms, and valley setting (Hupp and Osterkamp 1985, Malanson 1993, Auble et al. 1994, Bendix 1994). Figure 17, illustrates the ecological structure of riparian ecosystems, in two different hydrogeomorphic settings, typical of the Colorado Plateau. The riparian ecosystem depicted in Figure 17a, represents an alluvial valley in which longer-term alluvial sediment supply has exceeded transport. Here, riparian vegetation occupies two distinct fluvial landforms; a currently active flood plain with a shallow water table adjacent to the stream. Under the current climate regime, unconfined stream erosional and depositional processes create disturbance patches across the flood plain, maintaining a spatial mosaic of fluvial surfaces and vegetation patches of different sizes and ages. Older, deeply rooted cottonwoods, occupy an alluvial terrace which represents a higher, formerly active flood plain under a different climate regime. The riparian zone in alluvial valleys can be very wide, dependent in part upon the depth and stability of the water table. There is typically a tight connection between surface flows and the alluvial water table, and in alluvial valleys, the water table typically slopes away from the stream channel as water is “lost” to the alluvium. In such settings, riparian vegetation, particularly on higher surfaces, are especially vulnerable (less resistant) to natural or human-induced declines in the alluvial water table (Scott et al. 1999, Shafroth et al. 2000).

In narrow, bedrock valleys, sediment transport tends to exceed supply and thus, fluvial landforms and associated riparian vegetation occur as narrow, often discontinuous patches on or adjacent to colluvial or side-valley materials (Figure 17b). In these hydrogeomorphic settings, the water table typically slopes to the stream and flows “gain” water from groundwater. Thus, riparian vegetation in bedrock canyons is more resistant to flow depletions than in alluvial valleys.

b. Aquatic Ecosystem Dynamics

Few ecosystems possess either the frequency or intensity of environmental change that are observed in stream systems (Power et al. 1988). The seasonal fluctuation in discharge represents a key abiotic control in the structuring of stream communities (Welcomme 1988). As water levels rise, the availability of food increases for grazers, insectivores, and detritivores that forage over inundated flood plains. Inundated flood plains also provide temporary nurseries and refugia for many aquatic species. The pattern and extent of these habitats depend on the stream hydrograph, channel morphology, and on the ability of various species and size classes to cross barriers under certain hydrologic conditions (Power et al. 1988). Extreme natural events (such as scouring floods or episodes of low flow) can eliminate much of the biota, and set the stage for periods of biotic recovery or succession between these disturbances (Fisher 1983, Power et al. 1988).

The actual effects of extreme flow events on benthic communities depend upon both precipitation and hydrogeologic characteristics of a given watershed (see section II, B, 2).

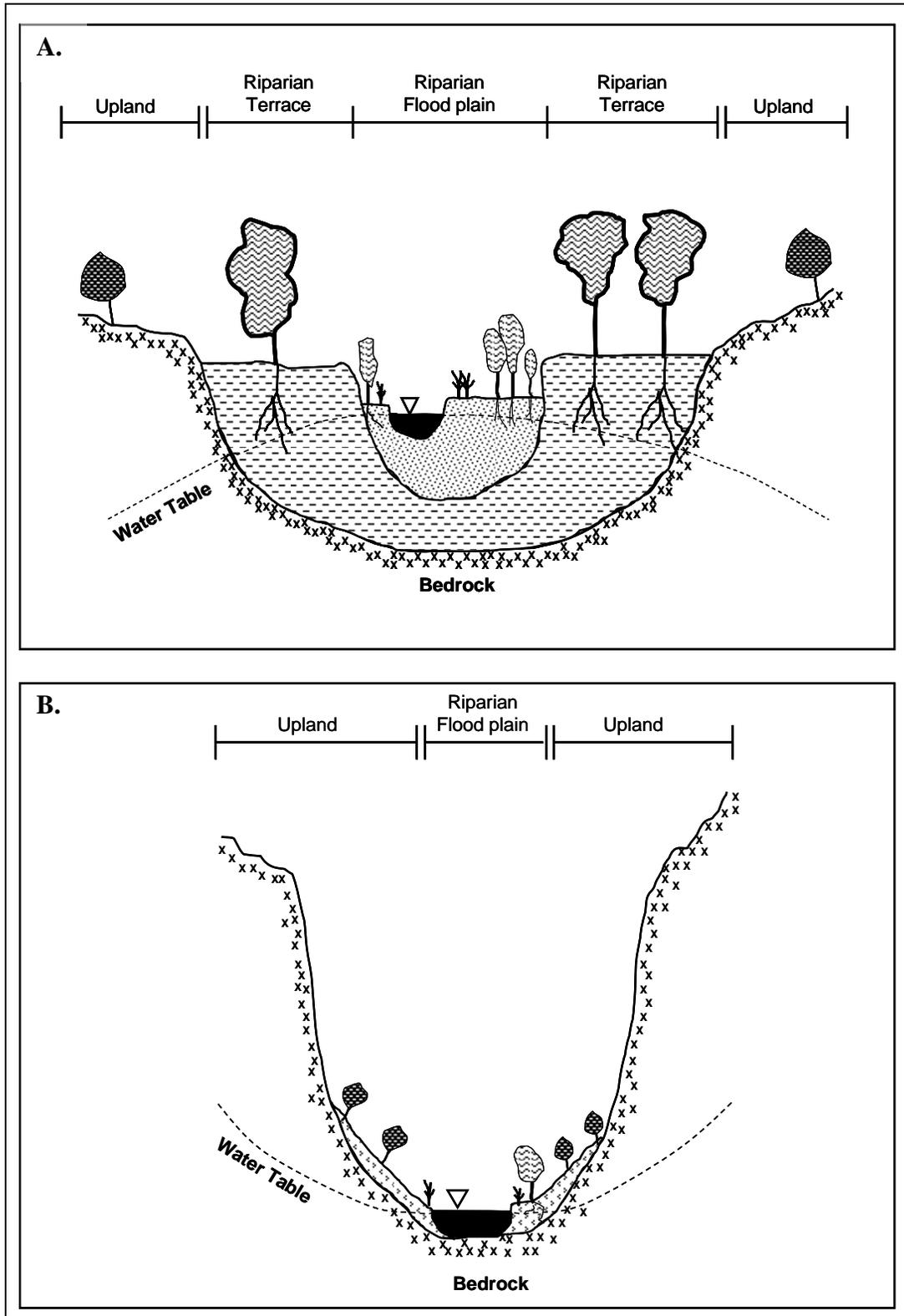


Figure 17. Schematic of the ecological structure of riparian vegetation in relation to the stream channel, surface water, the alluvial water table, and upland vegetation, for a) an alluvial valley, and b) a confined bedrock canyon. Modified from: Goodwin et al. (1997).

Scouring floods may enhance co-existence of species by maintaining an intermediate level of disturbance. In watersheds with steep topography and little soil development, stream discharge can exhibit extreme variability, including scouring floods or extended low flows that can eliminate or greatly reduce instream biota (Power et al. 1988), although many native aquatic species have physiological and behavioral adaptations to resist the effects of flooding or drought. Such disturbance events can determine the structure of aquatic communities, and they are critical to the life histories of many stream organisms. For example, because macroinvertebrate faunas in the streams of the Colorado Plateau are frequently exposed to unpredictable floods and dry periods, these streams are dominated by mayflies, small diptera, and other taxa with shortened aquatic developmental stages or the ability to rapidly recolonize disturbed habitats. Thus, functional relationships among aquatic species may change with both density and ontogeny (developmental stage) (Power et al. 1988).

Aquatic ecosystems are typically abiotically controlled because physical disturbance maintains populations at such low densities that biotic interactions are not as important. Biotic interactions, however, may be important in allowing populations to endure abiotic disturbances (Power et al. 1988). In fact, the relative importance of abiotic and biotic factors controlling aquatic community structure and function may shift with dynamic changes in density of organisms and environmental conditions (Figure 18). Power et al. (1988) suggest that many of these processes are not well understood for aquatic ecosystems and will require much additional research in order to develop a full understanding of the dynamics of these systems.

i. River Continuum Concept

The river continuum concept (Figure 19) describes a transition in ecosystem structure and functioning from narrow headwater streams to broad rivers, with a continuum of physical gradients and biotic responses linked longitudinally through waterflow and ecosystem processes (Vannote et al. 1980). Based on the principles of fluvial geomorphology, the river continuum concept emphasizes gradual adjustments of biota and ecosystem processes in rivers in accordance with gradual downstream changes in hydrologic and geomorphic properties (Benda et al. 2004). This links a gradient of physical factors, mostly generated by stream morphology and hydrology, with life-history strategies of benthic invertebrates and the dynamics of nutrient inputs and their utilization by functional feeding groups of invertebrates.

Headwater streams are often shaded by riparian vegetation. These plants reduce light availability to aquatic primary producers (algae) and provide most of the organic input to the stream (coarse particulate organic matter, CPOM). Leaves and wood (allochthonous input) that fall into the stream are colonized by aquatic fungi and to a lesser extent by bacteria. The resulting leaf packs and woody debris that accumulate are consumed by invertebrate shredders that break leaves and other detritus into pieces (fine particulate organic matter, FPOM) and digest the microbial particles. As material is carried downstream, the fine particles are consumed in suspension by filter feeders or from benthic sediments by collectors, and eventually excreted as dissolved organic matter (DOM). As headwater streams merge to form broader streams, the greater light availability supports more instream production (algal growth/autochthonous productivity), and the input of terrestrial detritus contributes proportionately less to stream energetics. This coincides with a change in the invertebrate community from one dominated by shredders to one

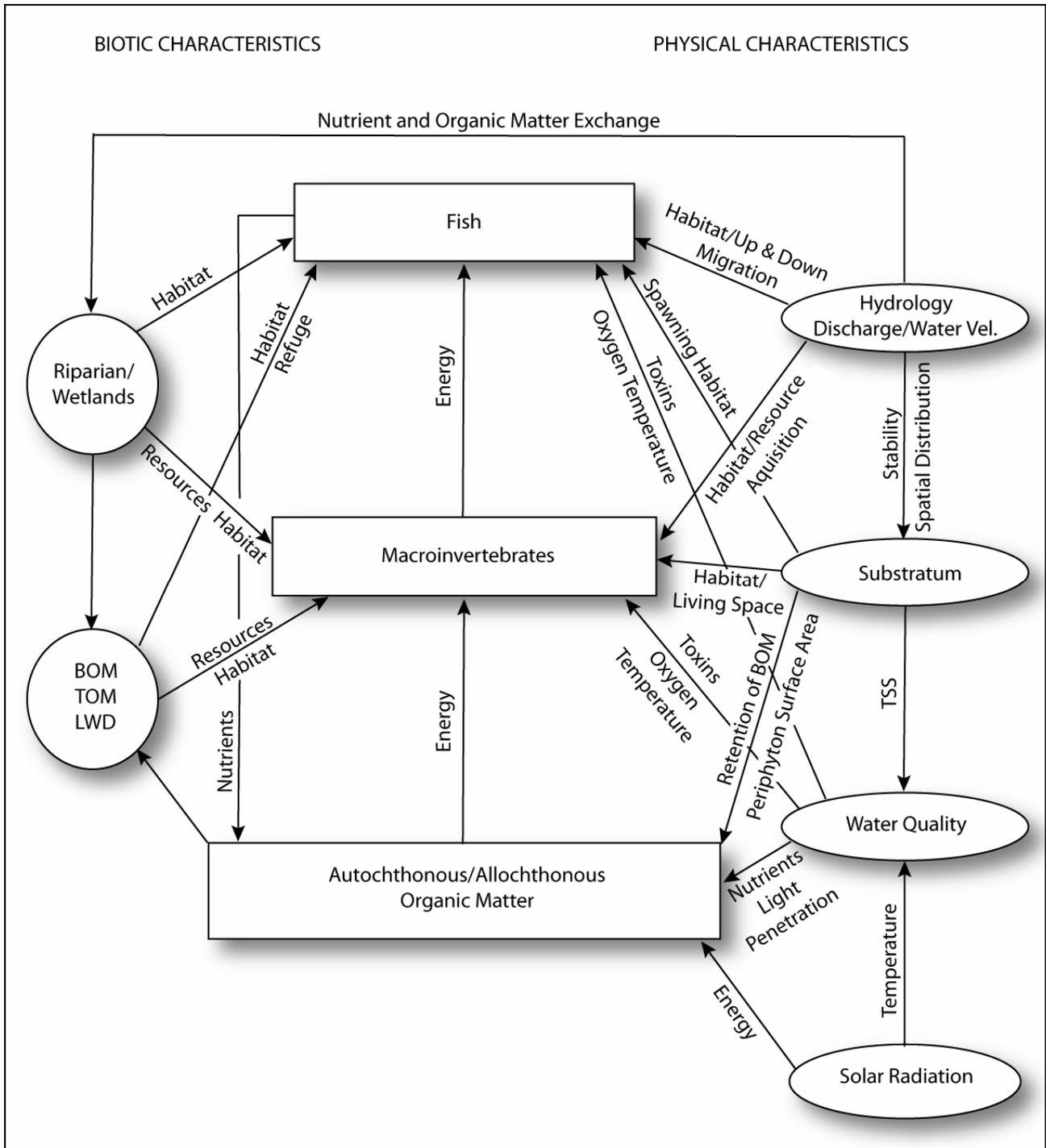


Figure 18. Model of stream ecosystem identifying major physical and biological components (Miller et al. 2003, adapted from Davis et al. 2001). Aquatic communities are structured by both biotic and abiotic factors, the relative influence depending upon the environmental setting and disturbance events.

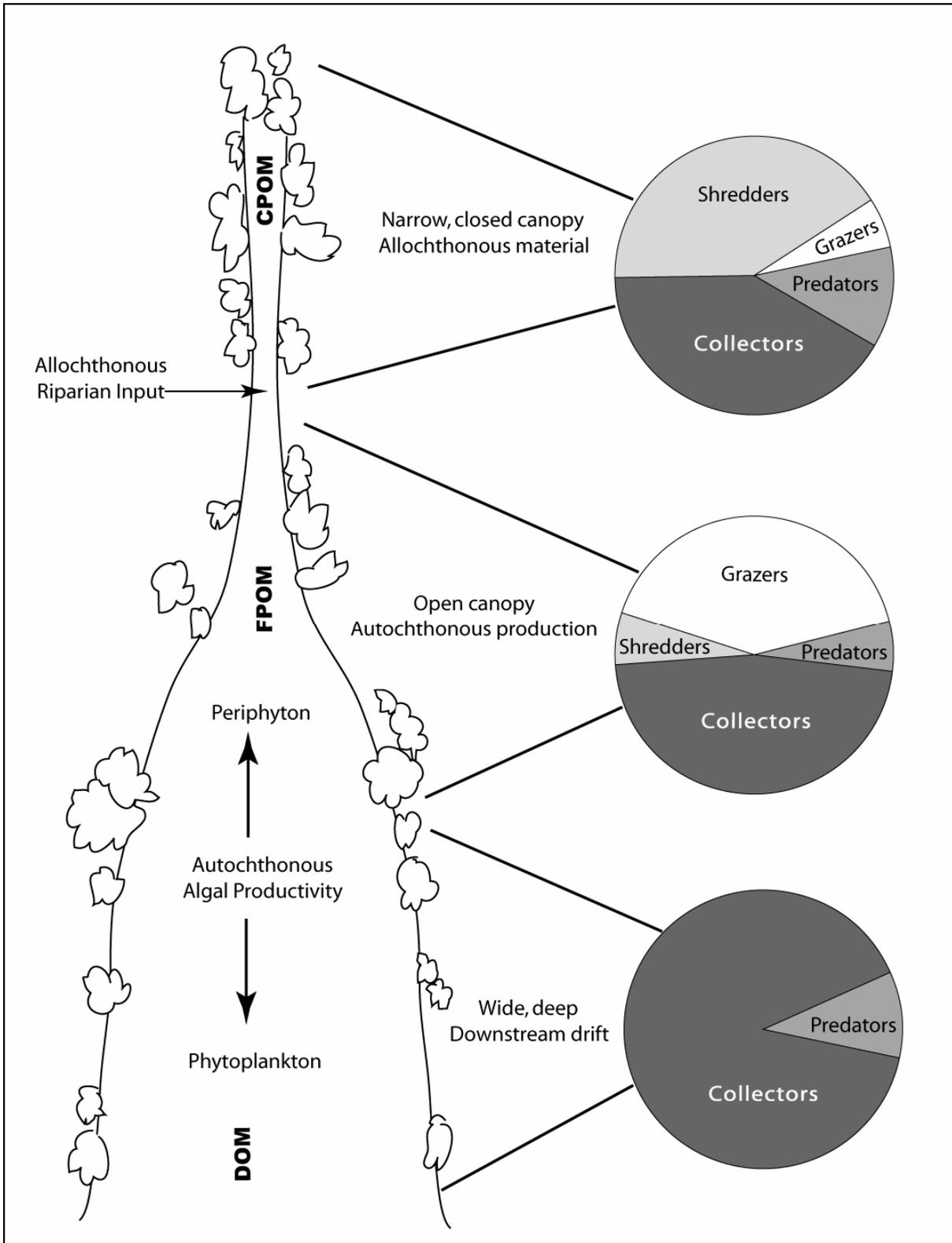


Figure 19. The river continuum concept (adapted from Vannote et al. 1980). CPOM is coarse particulate organic matter, FPOM is fine particulate organic matter, DOM is dissolved organic matter.

dominated by collectors and grazers. The middle reaches of rivers are typically less steep than headwaters and begin to store sediments from upstream erosion. These sediments support rooted vascular plants and a benthic detrital community of collectors. The largest downstream reaches are typically deep and slow moving, dominated by collectors and detritivores that live in the sediment.

ii. Network Dynamics Hypothesis

According to the network dynamics hypothesis, riparian and aquatic habitats are structured by branching river networks, channel intersecting confluences that alter channel and flood plain morphology, and the frequency and intensity of episodic disturbances (Benda et al. 2004). A river network can be thought of as the landscape template within which climatically induced events effect the supply and transport of water, sediment and organic material. Based on the concept of a river network of channels and their confluences, Benda et al. (2004) have developed a model describing how geomorphic characteristics (basin-size, basin-shape, drainage-density, and network geometry) interact to regulate the spatial distribution and characteristics of tributaries throughout a watershed. The attributes of this basin-wide network determine local habitat characteristics and ultimately determine species richness and diversity in riparian and aquatic ecosystems. Confluences within a river network create morphological conditions at these junctions that interact with the underlying spatial network, to create dynamic habitats. Abrupt changes in water and sediment flux that occur at channel confluences trigger changes in channel and flood plain morphology. An increase in the number of confluences adds sediment, water, nutrients, and organic debris further downstream, increasing heterogeneity of habitats and resulting in higher productivity in aquatic and riparian systems.

River networks also interact with episodic watershed disturbances, (such as fires, storms, and floods), to temporally alter riparian and aquatic habitats, typically resulting in increases in biological diversity and productivity. For example, the episodic nature of sediment-related disturbance creates erosional and depositional landforms that contribute to physical heterogeneity leading to biological diversity and increased productivity in riparian habitats. Increasing the heterogeneity of habitat conditions, including channel width and depth, bed substrate, wood storage, and water velocity, will result in an increase in species richness of fish and macroinvertebrates. Likewise, for riparian communities, greater topographic variation in flood plains and terraces creates local variation in inundation and soil moisture regimes, thereby increasing plant diversity.

iii. Shifting Habitat Mosaic

The shifting habitat mosaic model describes riparian and aquatic ecosystems as dynamic, non-linear interactions among physical and biological processes. Changing successional states (or gradients) across a landscape are mediated by interactive physical and biological drivers resulting in a shifting habitat mosaic that can be seen throughout riparian zones (Stanford et al. 2005). The key processes driving biotic processes and related geochemical cycles include flood-caused sediment erosion and deposition (cut and fill alluviation), routing of river water and nutrients above and below ground, channel movement (e.g., avulsion), and production and entrainment of large wood. Fluvial geomorphic processes, are constantly modifying the riparian zone, creating a mosaic of different habitat conditions across the flood plain. Strong interactions between short-duration high-stream-power floods, channel and sediment movement, increased

roughness due to the presence of vegetation and dead wood, and the upwelling of ground water, in concert with riparian plant successional processes creates a complex, dynamic distribution of resource patches and associated biota. Frequent floods maintain high habitat patch heterogeneity, resulting in higher species richness in both riparian and aquatic ecosystems. In contrast, reduced flooding contributes to more stable habitats, which can in turn lead to a decline in species richness.

III. ENVIRONMENTAL MONITORING OF RIPARIAN AND AQUATIC ECOSYSTEMS

In this section, we use conceptual models of ecosystem structure, functioning and dynamics described in Section II, to identify a key set of vital signs to be used in developing a monitoring strategy for riparian and aquatic ecosystems of the Colorado Plateau. One important goal of resource monitoring is to discriminate changes in a system resulting from normal ecosystem dynamics from degradational change associated with specific anthropogenic stressors. To that end, we use structural and functional characteristics typical of healthy, naturally functioning riparian and aquatic ecosystems of the Colorado Plateau, to identify a specific set of vital signs that would serve as sensitive indicators of important degradative changes. Regionally important stressors and degradative pathways leading to several degraded conditions typical of these systems are detailed in Section IV.

A. Characteristics of Healthy Riparian and Aquatic Ecosystems

Because of their unique landscape position, and tight linkages between fluvial and upland disturbance processes, riparian and aquatic ecosystems are potentially sensitive indicators of landscape-level environmental change (Naiman et al. 1988). A hallmark of these ecosystems is their resilience to frequent and sometimes intense physical disturbances. Rapid recovery of the structural and functional elements of riparian and aquatic ecosystems following disturbance is distinct from the observed recovery of upland ecosystems in arid and semi-arid regions, where productivity is typically lower, plant recruitment slower, more episodic, and where vegetation dynamics may include multiple stable states (Scheffer and Carpenter 2003, Miller *in press*). High rates of recovery in riparian and aquatic ecosystems result primarily from *adequate moisture*, which supports relatively high rates of productivity, *high linear connectivity*, which allows for rapid re-colonization of disturbed sites from upstream and downstream refugia, and *disturbance-adapted species*, whose life-history characteristics allow them to reestablish and re-colonize quickly following disturbance. Thus, relatively healthy, naturally functioning riparian and aquatic ecosystems of the Colorado Plateau would be generally characterized by: 1) surface flow variability; 2) active erosional and depositional processes; and 3) relatively shallow alluvial groundwater; which together would support; 4) riparian vegetation patch dynamics; and 5) aquatic habitat diversity; which would in turn support 6) riparian and aquatic communities dominated by native species, representing a diversity of structural and functional groups. Based on the conceptual models and this general characterization of healthy riparian / aquatic ecosystems, a summary of the vital signs that will be monitored in order to evaluate the impact of the major anthropogenic stressors is described in Section IV, below.

It should be noted here, that because many human-related disturbances mimic large-scale, natural disturbance processes, it may be difficult to clearly separate human-related changes in riparian and aquatic ecosystems from natural variability in some cases. Indeed, studies on the Colorado Plateau record change in geomorphic process and riparian vegetation structure and composition, coincident with climate-related shifts in regional precipitation and stream-flow patterns, as well as human-related changes in stream flow and the establishment and spread of the non-native tamarisk (Hereford 1984, Allred and Schmidt 1999, Grams and Schmidt 2002). Similarly, climate change and land-use practices such as grazing and land-clearing, can act together, over broad spatial scales, to reduce vegetation cover and thus alter the delivery of water and sediment to receiving streams (Trimble and Mendel 1995). This, in turn, alters the rate, magnitude, and style of channel processes, which ultimately structure and maintain riparian and aquatic ecosystems (Brinson et al. 1981, Frissell et al. 1986). These points argue strongly for a hierarchical approach to monitoring that links reach-specific monitoring of riparian and aquatic vital signs with watershed-scale monitoring of land-use and land-cover.

B. Monitoring to Evaluate the Impact of Stressors

1. Key components for monitoring to assess impacts of stressors on riparian systems

The riparian component of the proposed monitoring protocols focuses on detecting significant ecological changes in the key physical drivers of riparian ecosystems, such as stream flow and associated channel and flood plain forming processes, and the composition and structure of riparian vegetation. Metrics involving both physical drivers and riparian vegetation will be monitored across a range of spatial scales.

Measurements of physical variables at the reach or channel unit scale include repeat channel cross-section topographic surveys, mapping of alluvial deposits, including characterization of surface texture, and development of reach-specific stage-discharge relationships. Valley segment to watershed-scale mapping of river level geology, channel confinement, valley slope, channel planform and alluvial deposits provides a broader, longitudinal-scale understanding of key physical processes and geomorphic organization (Grams and Schmidt 2002, in press; Montgomery and Buffington 1998).

Methods for monitoring transverse-scale patterns in riparian vegetation include woody and herbaceous plant cover (Daubenmire 1959), species richness or composition, including relative importance of non-native and upland species (Innis et al. 2000), size / age structure of dominant riparian trees, and total vegetation volume (Mills et al. 1991). Variables, sampled remotely at larger scales, include width and area of riparian forest (Stromberg and Patten 1990, Lite and Stromberg in press), and woody riparian vegetation composition, including non-native species (Friedman et al. 2005).

2. Key components for monitoring to assess impacts of stressors on aquatic systems

The primary focus of the aquatic component of the protocols is the assessment of the status of benthic macroinvertebrate communities and associated habitat characteristics. Benthic macroinvertebrates will be collected from the stream following procedures described in the protocols, and identified by a contract laboratory to the lowest feasible taxonomic level. Physical habitat characteristics will be measured at each site where benthic macroinvertebrates

are collected. These include substrate, embeddedness, stream width, depth and velocity, water temperature, riparian canopy cover, and geomorphic channel units along the stream reach.

Macroinvertebrates are useful indicators of aquatic ecosystem quality and have been used for bio-monitoring since the early 1900's (Cairns and Pratt 1993). Recent efforts focus on the development of indicator species, diversity indices, and multivariate techniques, which link macroinvertebrate communities with habitat conditions. Because factors such as riparian vegetative structure, geology, and climate determine the state of a stream and therefore the community of organisms that occupy that stream (Townsend et al. 1997), it is important to also understand regional climatic and atmospheric conditions, as well as any drivers or stressors in the system, whether anthropogenic or natural when evaluating the status of an aquatic system. It is also important to assess the integrity of a stream system on a site-specific basis, as macroinvertebrate community structure will naturally vary from site to site and across regions.

Macroinvertebrates are of central importance in streams because of their variable functional roles as detritivores, herbivores, predators, competitors, and prey. In addition to their significance within the biotic environment, macroinvertebrates are sensitive to the physical and chemical environment, and can serve as indicators of water quality. Invertebrate response to stressors and drivers can be rapid and provides an excellent means to examine temporal and spatial variation in aquatic ecosystem quality. Because of their link to other components of the biotic and abiotic environment, macroinvertebrate monitoring should be conducted in conjunction with evaluation of riparian condition, physical habitat, and water quality whenever feasible.

When using macroinvertebrates to assess stream quality, it is important to examine communities and populations in addition to specific indicator species. A metric is an enumeration representing an assemblage (community) characteristic or combination of characteristics that changes in a predictable way with increased human influence. Biological metrics relate specific measures of assemblage structure, composition, and functional attributes to a minimally disturbed system. A multimetric approach has been advocated because several different metrics, each measuring a different component of the assemblage, are believed to provide a more robust assessment of ecological integrity. Some commonly used metrics include species diversity, percent tolerant species, percent sensitive species, percent shredders, percent grazers, and percent introduced or non-native species. The Index of Biological Integrity (IBI) is a collection of metrics that examines entire community and species assemblages. The IBI employs metrics of certain characteristics, such as trophic composition, native and non-native species composition, and species diversity and abundance, to determine "scores" that indicate the biological integrity of a given site compared to the integrity of a comparable "least-disturbed" site (Karr 1991). A diverse environment promotes a diverse macroinvertebrate community and a loss of species diversity or abundance, may indicate environmental degradation (Covich et al. 1999). An advantage of the biological metrics approach is that it is the most amenable to non-experts. A disadvantage is that the appropriate set of metrics and an IBI for use in the Colorado Plateau region have yet to be determined.

IV. PRIMARY STRESSORS AND DEGRADATIONAL PROCESSES

A. Overview

This section includes: (1) brief descriptions of the predominant anthropogenic and natural stressors affecting the structure and functioning of riparian and aquatic ecosystems of the Colorado Plateau in general, and park units in particular; (2) an ecosystem dynamics model depicting degraded conditions and degradational pathways, commonly observed for these systems; and (3) conceptual models summarizing the role stressors play in ecosystem processes that lead to degradation of these systems. Additional information on the significance of stressors in specific parks can be found in associated NPS reports (Evenden et al. 2002, Thomas et al. 2003, Miller et al. 2003).

B. Stressors

Here we present a general discussion of key stressors which are most likely to influence a range of riparian and aquatic ecosystems across the Colorado Plateau and within park units of the SCPN and NCPN. However, it should be recognized that in part because of their landscape position, riparian and aquatic ecosystems are typically influenced by multiple stressors, which interact with other natural physical and biotic processes across a range of scales, and that these interactions are often site specific. For example, Figure 20 is a general conceptual model summarizing the primary stressors affecting Salt Creek in CANY (Schelz 2001).

1. Climate Change

Persistent changes in climate, especially altered precipitation patterns and increases in soil and air temperatures, can have direct effects on the composition, structure and functioning of riparian and aquatic ecosystems by limiting moisture availability and altering fluvial geomorphic processes. Furthermore, climate-related changes in precipitation patterns and temperature typically interact with existing anthropogenic stressors. For example, atmospheric drought can intensify the effects of stream flow depletion on riparian and aquatic communities by reducing stream flows. Also, heavy livestock grazing on the uplands, in combination with intense rainfall events, can trigger channel erosion and incision processes (see discussion of *Arroyo Cutting and Filling*, pg. 28). Shifts in precipitation patterns appear to have important implications for riparian and aquatic ecosystems of the Colorado Plateau. Using discharge records and tree ring chronologies, Hereford (1984) described a period of erosion along the Little Colorado River between 1905 and 1937, corresponding with an early-century wet period. This was followed by channel narrowing, flood-plain aggradation, and riparian vegetation establishment, primarily involving *Tamarix* on portions of the former channel, as precipitation, mean annual discharge and flood frequency decreased in a subsequent dry period. A similar temporal pattern of channel narrowing and riparian vegetation establishment has been described on the upper (Grams and Schmidt 2002) and lower Green River (Allred and Schmidt 1999), suggesting that region-wide geomorphic processes and related changes in riparian vegetation dynamics may be sensitive to relatively small shifts in climate in arid and semi-arid environments. Whether characterized by dry or wet climatic conditions, extreme years featuring floods or droughts can have long-lasting consequences for riparian and aquatic ecosystem structure and functioning by causing episodes of plant mortality or establishment (Burkham 1972, Ehleringer et al. 2000, Friedman and Lee 2000). Such events also can directly affect macroinvertebrate communities through mortality (Wood and Petts 1994) and indirectly through changes in aquatic microhabitat structure. For

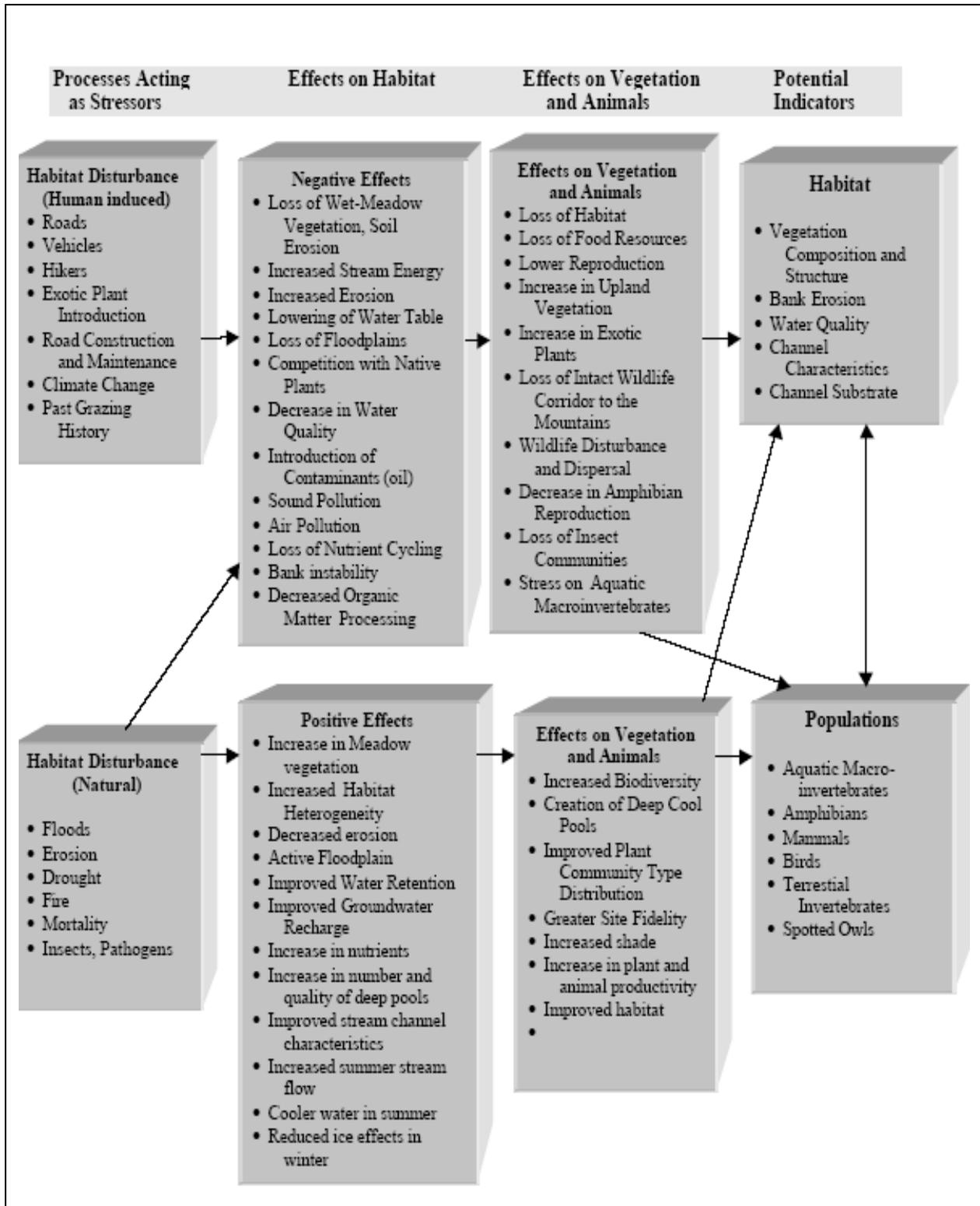


Figure 20. Conceptual ecological model depicting the processes by which primary stressors to the Salt Creek system of CANY affect riparian and communities. From Schelz (2001).

example, coarse particulate organic material (CPOM) is retained during droughts (increasing availability to shredders and detritivores) and more rapidly flushed out during floods.

2. Stream-flow Alteration

Human alterations of natural flow regimes (also see section II. B. 1. b. iii., Stream flow regime) can vary widely, depending on the nature and intensity of water-use activities. However, the effects of large storage dams on stream flow regimes of perennial rivers of the Colorado Plateau have been pervasive and profound, typically involving significant reductions in the magnitude of annual snow-melt peak flows and increases in base flows. A comparison of pre- and post-dam flows on the Green River, Utah, illustrates the extent to which flows on large interregional streams have been altered (Figure 21a). In such cases, flows of sediment and water from unregulated tributaries become especially important in maintaining system integrity. For example, the Yampa River, with a small headwater reservoir (Stagecoach Reservoir), is the only relatively un-regulated interregional stream in the Colorado Plateau region, and stream flow from the Yampa re-establishes a relatively natural hydrograph for the middle and lower Green River through DINO and CANY (Figure 21b).

Large, in-channel dams have significantly altered riparian and aquatic ecosystems throughout the southwestern U.S. by disrupting flows of water and sediment and fragmenting once-continuous riparian corridors. Because water storage behind dams is large relative to runoff, the alteration of riparian and aquatic ecosystems is correspondingly greater in this region (Graf 1999). Most of the large interregional streams that carry stream flow derived from the Rocky Mountains are dammed, many by the facilities of the Colorado River Storage Project. Large trans-basin diversions affect the headwaters of the upper Colorado and Gunnison Rivers and deplete stream flow through CANY. Stream flow on regional streams such as Tsaile Creek and the Escalante, and Dirty Devil Rivers, is affected by dams and diversions upstream of some park units. These diversions deplete base-flows and in some cases channel forming flood-flows. There is also a large dam and reservoir on the Rio Grande, just downstream from BAND. Thus, parks with riparian and aquatic ecosystems on highly flow-regulated rivers include BAND, BLCA, CANY, DINO, GLCA, MEVE, WACA, WUPA and GRCA.

Channel adjustments, involving changes in cross-sectional form, the size and distribution of bed and bank materials, slope and planform, accompany stream flow alteration and reflect complex adjustments to temporal variations in stream flow and the amount and size of sediment particles supplied to the stream from the watershed. Complex interactions among flow, channel response, and plant and animal life history traits contribute to considerable spatial and temporal variability in the response of riparian and aquatic ecosystems to flow alteration. Biotic changes typically show a lagged response to driving physical variables, frustrating efforts to develop simple predictive models of ecosystem response (Petts 1987). This suggests the potential importance of using measures of physical processes or attributes, like flow and channel form, as leading indicators of degradational change in riparian and aquatic ecosystems. Two important aspects of stream-flow alteration are discussed below.

a. Stream-flow Depletion

Reduction or depletion of stream flow may result from diversion or damming of surface water, ground-water abstraction, climatic drought, or a combination of anthropogenic activities and

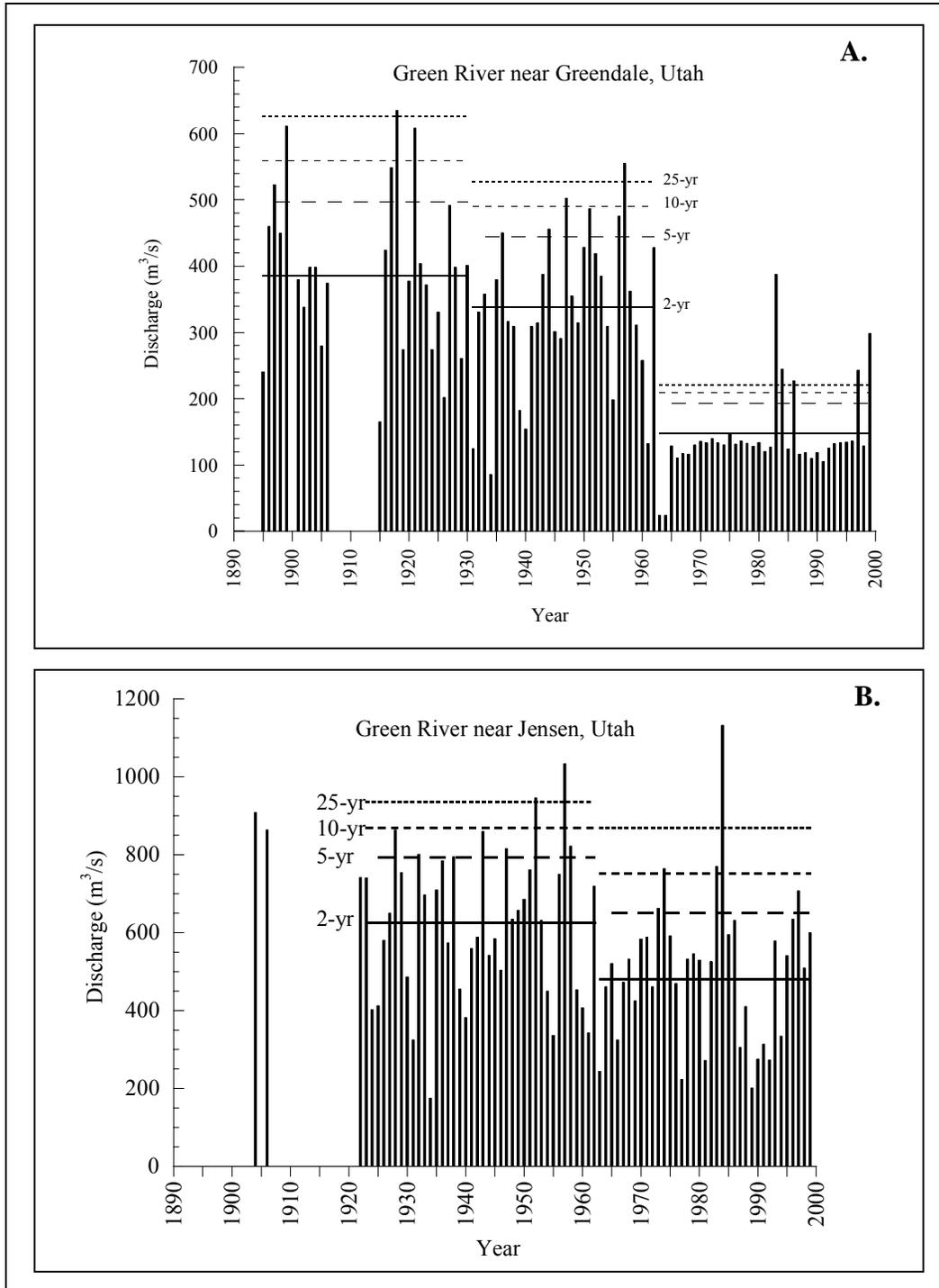


Figure 21. (a) Annual maximum discharge of the Green River near Greendale, Utah and (b) the Green River near Jensen, Utah. The horizontal lines represent the 2-, 5-, 10-, and 25-year recurrence flows. At the Greendale gage, changes in peak flows between 1895-1929 and 1930-1963 reflect the influence of climate (Allred and Schmidt 1999), whereas flows after 1963 reflect the effect of flow regulation from Flaming Gorge Dam. The influence of the Yampa in restoring peak flows to the Green River, can be seen by comparing post-1963 flows at the Greendale gage with those at the Jensen gage, located downstream of the Yampa River confluence. From: Grams and Schmidt (2002).

natural processes. Surface water extraction for agricultural use or upstream water diversion projects contribute to stream flow alteration and depletion at a number of SCPN parks, including MEVE and CACH. Declining water tables related to regional aquifer drawdown by external mineral developments are management concerns for both NAVA and PEFO. For example, water use by Peabody Coal Mine exceeds the aquifer recharge rate, thus raising the concern of lowered ground-water levels at NAVA (Christensen 1979). Stream-flow depletion can have a range of effects on riparian and aquatic ecosystems, depending in part upon the intensity and duration of the depletions. The effects of moderate or episodic flow depletions on riparian and aquatic ecosystems can be subtle, ranging from transient physiological stress to structural and functional changes involving reductions in species richness and primary productivity, and increases in non-native species. Significant depletions of surface and groundwater can lead to dewatering of the channel and flood plain, resulting in a variety of structural and functional changes, including the mortality of riparian vegetation and aquatic biota, destabilization of channel banks, and the encroachment of upland vegetation and non-native weeds into the riparian zone (Auble et al. 1997 and 2005, Innis et al. 2000, Kondolf and Curry 1986, Rood and Mahoney 1990, Scott et al. 1999).

Concerns about the effects of stream-flow depletions on aquatic biota are not limited to surface water. Groundwater abstractions for municipal and agricultural uses also may alter aquatic communities (Erman and Erman 1995, Armitage and Petts 1992). Biotic community alterations have been observed in response to ground-water withdrawal (Wood and Petts 1994, Bickerton et al. 1993), but studies are scarce, especially within the Colorado Plateau region.

b. Reduced Streamflow Variability

Stream flow variability is the principle force that creates and maintains the integrity of riparian and aquatic ecosystems (Brinson et al. 1981, Poff et al. 1997, Bunn and Arthington 2002; Figure 12). Thus, any anthropogenic activity that alters the natural flow regime represents a significant threat to the structural and functional integrity of these ecosystems, both directly and indirectly. Reduced stream-flow variability associated with large in-channel dams (see Figure 21), represents one of the single most important threats to riparian and aquatic resources on the Colorado Plateau. A river channel in quasi-equilibrium with its stream flow and sediment discharge regimes will generally reflect a form adjusted to the combination of flow and sediment discharges that produce the greatest change in channel bed and bank material (Leopold and Maddock 1953). Dams alter two key elements of the fluvial geomorphic system: (1) the transport capacity of the river and (2) the amount of sediment available for transport (Grant et al. 2003). The style, degree and timing of channel change below a dam are dependent on the relative balance among post dam sediment inputs, the altered hydrologic regime, and the geomorphic setting of a particular reach (Williams and Wolman 1984, Church 2002, Grams and Schmidt 2002). Stream reaches in sediment surplus exist where sediment delivery to the stream exceeds the sediment-transport capacity. Sediment deficit occurs in reaches where the sediment transport capacity exceeds sediment delivery (Williams and Wolman 1984). Whereas all streams are in sediment deficit in the reach immediately below a dam, tributary inputs can make significant contributions to the total supply (Topping et al. 2000, Grams and Schmidt, *in press*). On the Colorado Plateau, river reaches in sediment deficit exist on the Colorado River in GLCA and GRCA and on the Gunnison River in BLCA. River reaches in sediment surplus include the Green River in DINO and CANY. Reaches in sediment surplus are characterized by narrowing

channels, vertically-accreting flood plains, fine-grained sediment deposition on gravel bars, and expanded non-native riparian vegetation, particularly tamarisk, on post-dam fluvial landforms. Narrowing and simplification of channel form, in combination with fundamental changes to riparian forest structure, has resulted in the loss of both aquatic and riparian habitats along these river corridors (Hammerson 1999, Schmidt and Brim-Box 2004, Tyus and Haines 1991).

Damming of streams and rivers can also alter macroinvertebrate community structure by altering instream temperatures. Temperatures may either decrease or increase depending on where the water is drawn from in the reservoir for release (Vinson 2001, Benenati et al. 2000), and changes in that location within the reservoir can modify the effect of the dam (Hart et al. 2002). Altered temperature affects macroinvertebrate community structure because life-history characteristics such as fecundity, growth rate, survival, and time of emergence are strongly regulated by water temperature (Vinson 2001). Glen Canyon Dam on the Colorado River has lowered water temperatures below the dam, resulting in altered algal and macroinvertebrate communities, with few temperature sensitive species of orders Ephemeroptera, Plecoptera or Trichoptera. Instead, more temperature tolerant invertebrates such as chironomids and amphipods dominate this cold-water section of the river (Stevens et al. 1997).

Aside from changes in flow regime and temperature, altered sediment fluxes downstream from dams may cause changes in fish and macroinvertebrate community structure (Blinn et al. 1998, Stevens et al. 1997). On the Colorado River, both algae and macroinvertebrate communities are significantly different at sites with low turbidity and suspended sediment immediately below Glen Canyon Dam, compared with sites with higher turbidity and suspended sediment below the confluence with the Paria River (Stevens et al. 1997, Shannon et al. 1994). Communities that existed before river regulation were characterized by those tolerant of high sediment loads, which inhibits autochthonous (algal) productivity and favors allochthonous inputs from terrestrial organic material (Haden et al. 2003). This also can be seen in the least regulated portions of the Green and Colorado Rivers, where macroinvertebrate community structure reflects allochthonous productivity, as compared to regulated portions of those rivers, where autochthonous communities dominate (Haden et al. 2003, Shannon et al. 1996). Similarly, allochthonous productivity is positively correlated with distance downriver from Glen Canyon Dam on the Colorado River (Benenati et al. 2001).

3. Invasion of Non-native Plant Species

Riparian corridors are generally more productive and have higher plant species richness than surrounding upland ecosystems. However, because of naturally high rates of hydrological disturbance and high edge-to-area ratios at both the landscape and localized patch scales, these systems are susceptible to invasion by non-native plants, which may constitute 25-30% of all species (Malanson 1993, Planthy-Tabacchi 1996). Tamarisk (*Tamarix ramosissima*) and Russian-olive (*Elaeagnus angustifolia*) have and are invading riparian areas along most of the perennial waterways in SCPN, including the Escalante, Fremont, Little Colorado, Rio Grande, Animas, Chaco, and Colorado Rivers and now constitute the third and fourth most frequently occurring woody riparian plants in the western U.S. (Friedman et al. 2005). Whereas the spatial and temporal heterogeneity created by natural disturbance may accommodate moderate levels of invasion by non-native species, without displacement of natives, significantly altered disturbance regimes may advantage non-native species. For example, although tamarisk is known to invade

relatively undisturbed settings, the most dense and extensive invasions result primarily from altered stream flow and geomorphic processes, as well as other factors including land clearing, livestock grazing and climate change (Hereford 1984, Everitt 1998, Shafroth et al. 2005). Although factors controlling the invasiveness of Russian-olive are less well known, reduced levels of physical disturbance resulting from stream flow management is considered to be a leading cause (Katz and Shafroth 2003).

The ecological effects of exotic species' invasions vary by species, but typically include major changes in community composition, competitive displacement of native species, and alterations of ecosystem-level properties such as disturbance regimes (D'Antonio and Vitousek 1992, Mack and D'Antonio 1998). Tamarisk has been implicated in ecosystem-level changes including altered erosional and depositional processes (Graf 1978, Allred and Schmidt 1999) and increases in the frequency and intensity of riparian fires (Busch and Smith 1995). The ecological effects of Russian-olive invasion is little studied, and there is no published information on competitive or facilitative interactions with co-occurring riparian species or the potential role that this nitrogen-fixing species may play in ecosystem nutrient dynamics (Katz and Shafroth 2003).

4. Livestock Grazing

a. On-site Grazing

Livestock grazing is one of the most pervasive human stressors of natural ecological systems on the Colorado Plateau. Livestock use is permitted in portions of one SCPN park (GLCA) and four NCPN parks (DINO, CARE, CURE, and BLCA). Seasonal livestock trailing is permitted in FOBU and BRCA, and several other parks repeatedly experience trespass livestock. Most (if not all) other parks were grazed by domestic livestock at one time, and many parks have on-going issues associated with persistent legacies of past livestock grazing and livestock-management practices. Herbivory and trampling by elk (BAND), bison (CARE and GRCA), and feral burros (GRCA) also occur. A study on riparian resources at NAVA links declining recruitment of native tree species to livestock grazing in the area (Brotherson et al. 1983). Trespass livestock are a significant management concern for MEVE and PEFO.

Because of the presence of water and shade, riparian areas are often subject to more intense grazing pressure than adjacent uplands (Platts 1991). Long-term grazing by livestock and other large herbivores can have profound on-site impacts on riparian ecosystems including the removal of plant biomass, alteration of plant population age structures, and simplification of plant compositional and structural diversity (Szaro and Pace 1983, Kauffman and Kruger 1984, Schultz and Leininger 1990). These changes have in turn been related to reduced abundance and diversity of riparian-dependent species, including birds (Taylor 1986, Dobkin et al. 1998, Scott et al. 2003). Because grazing acts as a stressor on an array of intercorrelated variables, altering physical habitat, riparian vegetation, and water chemistry, potential impacts to macroinvertebrate communities are variable (Griffith et al. 2001). Reduced flow, increased sedimentation, and a shift from allochthonous (riparian) to autochthonous (algal) productivity, can cause shifts in macroinvertebrate community structure and decrease macroinvertebrate diversity (Oberlin et al. 1999).

b. Trampling

Trampling of stream banks by cattle and pack animals causes a loss of bank stability and changes in channel morphology as streams generally become wider and shallower (Scrimgeour and Kendall 2003). Increases in turbidity and suspended solids are also associated with livestock trampling (Davies-Colley et al. 2004). Trampled riparian areas are characterized by soil compaction, vegetation removal, and decreased water infiltration rates, which results in increased runoff rates (Trimble and Mendel 1995). A combination of vegetation loss and wider, shallower channels can increase light availability and water temperature, resulting in increased algal growth, and a subsequent increase in macroinvertebrate grazer communities. Trampling has been shown to change the relative abundance of macroinvertebrate species and functional groups, such as from Plecoptera, Ephemeroptera, and Orthocladiinae to other Diptera and non-insects including amphipods, either directly through changes in physical habitat; decreases in water (stream size) and increased sedimentation, or indirectly through changes in algal communities (Griffith et al 2001).

Increases in turbidity, erosion, and suspended solids, conversely decrease light penetration and thus reduce algal growth causing a shift from autochthonous to allochthonous productivity. These contrasting effects on algal communities make it difficult to predict changes in macroinvertebrate community structure without determining which stressors are dominant at a given site. In general, macroinvertebrate diversity typically decreases in response to increasing sedimentation, (Kaller and Hartman 2004) followed by an increase in generalist species and a loss of specialist species. For example, Weigel et al. (2000) found that stream reaches with minimal trampling contained more specialist macroinvertebrate species than did stream reaches with greater trampling. Also, species that prefer fine-grained sediments (e.g. oligochaetes and chironomids) tend to be found in greater abundance in trampled areas (Meadows 2001). Effects of trampling by livestock in streams and pools across the Colorado Plateau should be similar to those in other regions, although few studies have specifically examined these effects on the Plateau.

c. Nutrient Enrichment

Increased nutrients inputs, due to livestock grazing (accompanied by a decrease in riparian vegetation), alter the composition of the macroinvertebrate community. Large inputs of nutrients will eventually result in eutrophication of the system, causing a predominance of nuisance algae, an increase in tolerant invertebrate species, typically grazers, and a decrease in invertebrate richness and diversity. Scrimgeour and Kendall (2003) found a greater total invertebrate biomass at grazed sites vs. non-grazed sites. Non-grazed sites, however, had a greater biomass of shredders (indicative of an allochthonous community) as compared to grazed sites, which had a greater biomass of collectors and grazers (indicative of an autochthonous community). This is consistent with expected changes in macroinvertebrate community structure resulting from decreased bankside vegetation and increased nutrient loading (Scrimgeour and Kendall 2003). Haefner and Lindahl 1991 studied the effects of grazing at CARE and found algal growth increased in response to nutrient inputs, followed by increases in certain macroinvertebrate species. Effects of nutrient inputs from livestock urine and feces also can be particularly detrimental to isolated pools, which can become anoxic (Haefner and Lindahl 1991).

5. Visitor Activities

Visitor use in and around riparian and aquatic resources in the parks tend to be spatially concentrated, magnifying the potential impacts to these systems. Riparian resources at GLCA and CACH are both heavily impacted by visitor activities. Documented impacts from recreation at GRCA and GLCA include bank erosion, contamination from human waste, water pollution, trash, and trampling of plants (Carothers and Aitchinson 1976). Heavily vehicular use degrades riparian areas in CACH.

a. Trails and Roads

Two important on-site recreational activities include hiking through canyons and driving off-road vehicles through canyons, both of which typically involve frequent stream crossings and / or walking and driving up the stream channel. As with cattle trails, hiking trails and roads breach stream banks and levees, increasing hydraulic roughness and removing vegetation. At high flows, turbulence created by these features accelerates erosion, creating more turbulence in a positive feedback loop. Trails and road crossings also serve as preferred flow paths for water onto, and off of the flood plain during rising and falling stream flows, causing further erosion (Trimble and Mendel 1995). Finally, because of reduced resistance to flow, un-vegetated trails or roads crossing flood plain surfaces parallel to the stream, would be expected to erode during high flows and could trigger channel incision processes (Cook and Reeves 1976).

Off-site roads alter abiotic components of aquatic ecosystems by changing soil density and composition, runoff and sedimentation patterns, light and temperature regimes, and water chemistry (Trombulak and Frissell 2000). Biotic alterations in response to these changes can be seen in riparian vegetation structure, and aquatic community structure (Backer et al. 2004). Few studies have examined direct effects of roads on macroinvertebrate communities, especially in the Colorado Plateau region. However, macroinvertebrates are sensitive to the effects mentioned above. Kaller and Hartman (2004) found a threshold level of sediment accumulation, above which macroinvertebrate abundance and diversity were reduced significantly. Increased sedimentation would also tend to favor macroinvertebrates that prefer habitats characterized by fine substrata such as oligochaetes and chironomids (Meadows 2001). Instream salinity levels are greatly increased by roads and certain macroinvertebrate species are more sensitive than others to high levels of road salt (Benbow and Merritt 2004). This sensitivity to road effects makes macroinvertebrate inventories useful for monitoring the status of aquatic systems, in national parks across the Colorado Plateau where roads have been constructed for visitor access.

b. Walking in streams (slot canyons)

Hiking through slot canyons is a popular activity in a number of parks including CARE (Halls Narrows) and ZION (Zion Narrows). Hikers walk through water within the confining walls during most of their time in the narrows, stirring up streambed sediment. No restrooms or water supplies are maintained in these areas, posing a risk of contamination from human waste. Hiker disturbance in the channel may increase turbidity, bacteria, and change the water chemistry, affecting aquatic habitat (Cudlip et al. 1999). Shakarjian and Stanford (1998) demonstrated that visitors hiking in streams at ZION impact macroinvertebrates in that system, as the number of hikers in the North Fork of the Virgin River was negatively correlated with the biomass and density of benthic macroinvertebrates.

c. Driving up streams or frequent stream crossings

Vehicles crossing or driving up streams causes an increase in stream turbidity, total dissolved solids (TDS), total suspended solids (TSS), salinity, and overall erosion (Lane and Sheridan 2002, Sample et al. 1998). Several studies have shown that macroinvertebrate communities respond to these factors. Increased turbidity and the associated decrease in light penetration, result in decreased diversity and / or a complete community shift in both algae and macroinvertebrates (Stevens et al. 1997, Thiere and Schulz 2004). Similarly, increases in TDS and changes in salinity levels can alter benthic invertebrate community structure (Leland and Fend 1998). Several studies have demonstrated species-specific responses to TSS, with certain species showing more resistance to high TSS than others (Thiere and Schulz 2004). Erosion is a direct cause of increases in turbidity, TDS, and TSS, and has been correlated with a decrease in the biointegrity of macroinvertebrate communities where erosion is prevalent (Rothrock et al. 1998). High levels of turbidity, TDS, and TSS do not allow for the establishment of light-dependent algae and associated invertebrate assemblages. Sensitive orders such as Ephemeroptera, Trichoptera, and Odonta are found to be less abundant under these conditions (Thiere and Schulz 2004). Instead, more tolerant invertebrate taxa such as dipterans become established (Stevens et al. 1997, Thiere and Schulz 2004).

Few studies have examined the effects of visitor activities on macroinvertebrate assemblages in streams of the Colorado Plateau. However, monitoring of macroinvertebrates in Salt Creek (CANY) indicates that macroinvertebrate diversity and richness is higher in relatively undisturbed reaches, upstream from reaches with frequent vehicle used (Schelz 2001, Banta 2002).

6. Fire

Small fires either in riparian areas or nearby uplands have occurred at a number of SCPN and NCPN parks. MEVE and BAND, however, have experienced frequent, severe and extensive fires since the early 1990s. As a result, siltation and ash flow in riparian areas is a major management concern for both parks. Also, many burned riparian areas within the park have experienced significant sediment loss (Bouchier 2000). As global climate change and sustained regional drought interact to shorten fire return intervals, particularly in areas with extensive beetle kill, many more parks in the networks may experience the effects of fire.

Depending on severity and extent, upland fire events can degrade riparian and aquatic ecosystems due to erosion, increases in suspended and bed-load sediment and increased peak flows following floods (Veenhuis 2002, Vieira 2004). Although post-fire impacts may be minimal following low or moderate severity fire, degradation of riparian systems following high-severity events can be significant. Erosion rates, for example, following high-severity fire can increase by one or more orders of magnitude (Benavides-Solorio 2003, Moody and Martin 2001). Because upland areas of the southwest are experiencing an increase in severe fire events (Allen et al. 2002), degradation due to severe-fire events represents a significant threat to riparian and aquatic ecosystems of the Colorado Plateau.

The structure and functioning of riparian areas are adversely effected by the sequence of wildfire, increased runoff, erosion and downstream sedimentation. The removal or reduction of the forest canopy, surface vegetation cover and ground cover (especially forest floor litter) all

contribute to accelerated erosion following severe fire (Cipra et al. 2002). The loss of the forest canopy also reduces shading to riparian areas which can raise water temperatures by 3 to 10 °C (Amaranthus et al. 1989). Additionally, snowmelt may be faster from burned areas, resulting in earlier and higher spring runoff events. A several fold increase in peak flows further amplifies surface and mass erosion (Dennis 1989, Tiedemann et al. 1979). Sediment laden flows often induce sheet wash, rill and gully erosion and cause mass movements such as debris torrents. As mass movements travel through the channel network, they can cause intense bank scour, which increases the volume of sediment delivered to downstream areas (Cipra et al. 2002). Post-fire salvage logging can exacerbate these effects, further stressing riparian and aquatic biota (Karr et al. 2004).

Alterations to water chemistry following fire also degrade riparian systems. The ash from fires can temporarily increase nutrients, ions, turbidity, pH, and alkalinity while decreasing dissolved oxygen levels (Earl and Blinn 2003). Macroinvertebrate densities are reduced immediately after a fire, but can recover within a year, whereas community structure and diversity are affected over a long period (Earl and Blinn 2003, Vieira et al. 2004). Because of intense flooding after burns and because of instream physical and chemical changes, generalist macroinvertebrate species with successful and rapid larval dispersal mechanisms tend to dominate over more specialized macroinvertebrate species that were present in the pre-fire system (Vieira et al. 2004). Many of the studies that examine macroinvertebrate response to fire have been conducted on and around the Colorado Plateau (e.g. Earl and Blinn 2003, Vieira et al. 2004, Veenhuis 2002), so existing data provides a good baseline for future studies in the region.

7. Stream Channel Alteration

The process of channel incision or arroyo cutting has been a concern in some park units such as FOBU, where efforts have been made to halt headward erosion of discontinuous arroyos. In contrast, the filling of Chaco Wash arroyo has raised concerns at CHCU, as the possible reduction in flood conveyance poses potential threats to ruins located immediately adjacent to the Wash. In CARE, active headward erosion is occurring along tributaries to the Fremont River (M. Scott, personal observation). Although a number of anthropogenic factors have contributed to contemporary arroyo formation, including the construction of trails and ditches, clearing of bottomlands, and widespread cattle grazing (Cottam and Stewart 1940, Cook and Reeves 1976), stratigraphic evidence dating back several thousand years confirms that region-wide episodes of arroyo cutting and filling, is a long-term, cyclic process that predates human landuse (Schumm and Hadley 1957).

Stream channelization is typically carried out to improve drainage or flood-carrying capacity, resulting in a smooth uniform channel with enhanced water conveyance and more predictable hydraulic behavior. The straightening of channels and reduction in roughness leads to greater flow velocities and higher erosive forces, resulting in increased turbidity and sedimentation (Gordon et al. 1992). Excessive siltation of gravel and cobble beds can lead to suffocation of fish eggs and aquatic insect larvae, and can affect density and composition of periphyton (algal communities) (Gordon et al. 1992). Suspended sediments reduce light penetration and consequently primary productivity. Stream channel alteration is frequently accompanied by removal of riparian vegetation, changing the relative contribution of allochthonous and autochthonous nutrient sources to the system. A decrease in riparian vegetation canopy cover

can also result in increased water temperatures and daily temperature fluctuations. Large algal blooms and daily temperature fluctuations are accompanied by large daily fluctuation in oxygen concentrations.

8. Alteration of Upland Watershed

a. Off-site Grazing

Heavy grazing on the uplands results in soil compaction, which reduces infiltration of precipitation and increases the delivery of water and sediment to streams. The combination of increased upland runoff and reduced channel stability within riparian zones from grazing contributes to increased stream bank and channel erosion, and has been implicated in the initiation of region-wide channel incision or arroyo cutting (Brinson et al. 1981, Cook and Reeves 1976).

b. Organic contaminants

Organic and metal contaminants degrade riparian areas at both MEVE and PEFO. Organic pollutants from pesticide use in urban and agricultural areas act as stressors on aquatic communities. Macroinvertebrates in stream reaches containing pesticides have shown similar numbers of individuals, but lower overall diversity and richness than communities in pesticide-free reaches (Thiere and Schulz 2004, Lenat 1984). Certain taxa are more sensitive than others to contaminants (Sibley et al. 1991, Thiere and Schulz 2004, Carsten von der Ohe and Liess 2004, Lenat 1984). The effects of different chemicals used for pest control are variable. For example, chemicals which are less water soluble to soil particles may be less toxic to macroinvertebrates than they would be if they were available in the water (Schulz and Liess 2001b). Organic contaminants have been shown to negatively affect macroinvertebrate survival and growth, and increase downstream macroinvertebrate drift (Schulz and Liess 2001a). Information about the effects of pesticides on macroinvertebrates across the Colorado Plateau is sparse and comparisons of growth rates or drift rates of macroinvertebrates in streams of the Plateau could provide useful information for managers in areas where chemicals are used within the watershed.

c. Mining

The Mancos River, which flows through MEVE, and its upstream tributaries are listed as impaired due to copper contamination. In addition, PEFO lists radionuclide contamination as a significant management concern. Just as organic contaminants are toxic to many macroinvertebrate species metal contaminants resulting from historic mining activities are also toxic. Community composition of macroinvertebrates is known to change in response to instream metals, with more resistant species dominating and less resistant species reduced or eliminated completely. Specific species sensitivity to toxic effects of metal compounds has been examined (Carsten von der Ohe and Liess 2004, Hoiland and Rabe 1992). In general, macroinvertebrate species richness, abundance, and diversity are reduced in response to stressors from mining activities (Griffith et al. 2001, Hoiland and Rabe 1992). Instream metal contamination can also be assessed by examining trace element bioaccumulation in macroinvertebrate tissues (Cain et al. 1992, Hare 1992).

Examinations of macroinvertebrate community structure across the Colorado Plateau suggest that species that are particularly sensitive to metal contamination include those species of the orders Ephemeroptera and Plecoptera (Clements 2004, Deacon et al. 2001). Macroinvertebrate metal bioaccumulation data has also been useful in determining the spatial and temporal extent of metal contamination around the Colorado Plateau (Deacon et al. 2001, Peterson et al. 2002). However, there have been few studies in the region and future studies of both macroinvertebrate community structure and metal bioaccumulation in tissues could assist managers in determining the effects and extent of metal contamination in streams.

C. Degradational Pathways and Processes

Figure 22 presents an ecosystem dynamics model that conceptualizes interrelationships among four degraded riparian and aquatic conditions, often resulting from human activities and commonly observed in these systems across the Colorado Plateau. Major degradational pathways, along with conceptual models representing the interactive ecological factors and processes leading to specific degraded conditions, are also illustrated. The dynamics model and the following narratives illustrate how these degraded riparian and aquatic conditions differ structurally and functionally from an idealized, naturally functioning condition (Condition A), where a system's resilience to natural disturbances is retained along with a characteristic diversity of abiotic and biotic components and processes (see Section III. A.). Implicit in the characterization of a system operating within a natural range of variation is recognition that such a system may change as a consequence of changing global climate (Hannah et al. 2001). Given this potential background of change, we describe four degraded conditions for riparian and aquatic ecosystems (Figure 22, Conditions B – E) to aid in developing a monitoring program to identify the ecological effects of anthropogenic stressors, as distinct from intrinsic natural variation. These conditions include:

- *Depleted Streamflow (Condition B)*—where major functional groups, flood plain soil resources, geomorphic processes, in-stream physical processes and chemical conditions remain largely intact, but ecosystem structural diversity (e.g., riparian tree height and density) and species richness is simplified and processes are somewhat altered by the presence of non-native riparian and aquatic species. The systems are somewhat less resistant and resilient to stressors and natural disturbances, compared with the natural, relatively undisturbed condition. For riparian and aquatic ecosystems, the consequences of moderate streamflow depletion would be expected to include:

Riparian consequences:

- Decreased cover and productivity of riparian and wetland plants
- Decreased structural complexity
- Increased cover of non-native weeds

Aquatic consequences:

- Decreased species richness and diversity
- Increased low-flow tolerant species
- Change from allochthonous to autochthonous productivity
- Increase in grazers and decrease in shredders

- *Reduced Streamflow Variability (Condition C)*—where there is often a significant increase in cover and density of the non-native riparian species *Tamarix* and a corresponding decrease in cover and richness of native riparian species, representing a major change in riparian functional group character. System structure and functioning (especially the fire disturbance regime and geomorphic processes) are significantly altered. Resistance and resilience to stressors and disturbance are diminished relative to the natural condition. For riparian and aquatic ecosystems, the consequences of reduced streamflow variability would be expected to include:

Riparian consequences:

- Decreased cover, productivity and diversity of riparian and wetland plants
- Increased cover of Tamarisk

Aquatic consequences:

- Decreased species richness and diversity
- Increased generalist species
- Increase in shredders and decrease in grazers

- *Altered Erosional and Depositional Processes (Condition D)*—where significant erosion of stream channel and banks reduces flood plain soil resources and lowers alluvial ground-water tables. Potentially large reductions in riparian and aquatic functional group diversity, species richness. Site potential is typically altered (possibly severely) as a result of reduced resource availability, site productivity, structural complexity and conditions required to support typical functional groups. Systems resistance and resilience to disturbance are greatly reduced relative to the natural condition. For riparian and aquatic ecosystems, the consequences of altered erosional and depositional processes would be expected to include:

Riparian consequences:

- Decreased cover, productivity and diversity of riparian and wetland plants
- Decreased structural complexity
- Increased cover of non-native weeds
- Increased cover of upland species

Aquatic consequences:

- Decreased species richness and diversity
- Increased sediment-tolerant species
- Increased temperature-tolerant species
- Decrease in grazers and shredders

- *Stream Dewatering (Condition E)*—where increases in cover and density of upland species in the riparian zone represent a fundamental change in site hydrology and functional group structure. Riparian and aquatic ecosystem structure and functioning (especially geomorphic processes and disturbance regimes) are greatly altered, reducing productivity and limiting capacity of the site to support characteristic functional groups. Aquatic species will only be present during episodic events of stream flow, and these species will be those that have the

ability to rapidly colonize a stream. The systems resistance and resilience to disturbance and stressors are profoundly altered relative to the natural condition. For riparian and aquatic ecosystems, the consequences of stream dewatering would be expected to include:

Riparian consequences:

- Mortality of riparian and wetland vegetation
- Conversion of wetland and riparian vegetation to upland species

Aquatic consequences:

- Decreased species richness and diversity
- Increased sediment-tolerant species
- Increased opportunistic, generalist species
- Predominance of colonizers

The preceding describes four common degraded conditions we have observed in riparian and aquatic ecosystems across the Colorado Plateau. Clearly, other definable conditions exist, and those we have described represent points on a continuum of change, and will vary in detail across sites as a function of many factors, including, disturbance history, hydro-geomorphic setting, elevation, and local climate.

The following are conceptual models representing ecosystem factors and processes that characteristically define the pathways of degradation illustrated in Figure 22. As with condition descriptions, the models are generalized in order to provide a framework for identifying the role of key natural and anthropogenic factors in these degradational processes. Recognizing that factors and processes will vary site to site, models representing particular, on-the-ground situations will need to be tailored on a case by case basis.

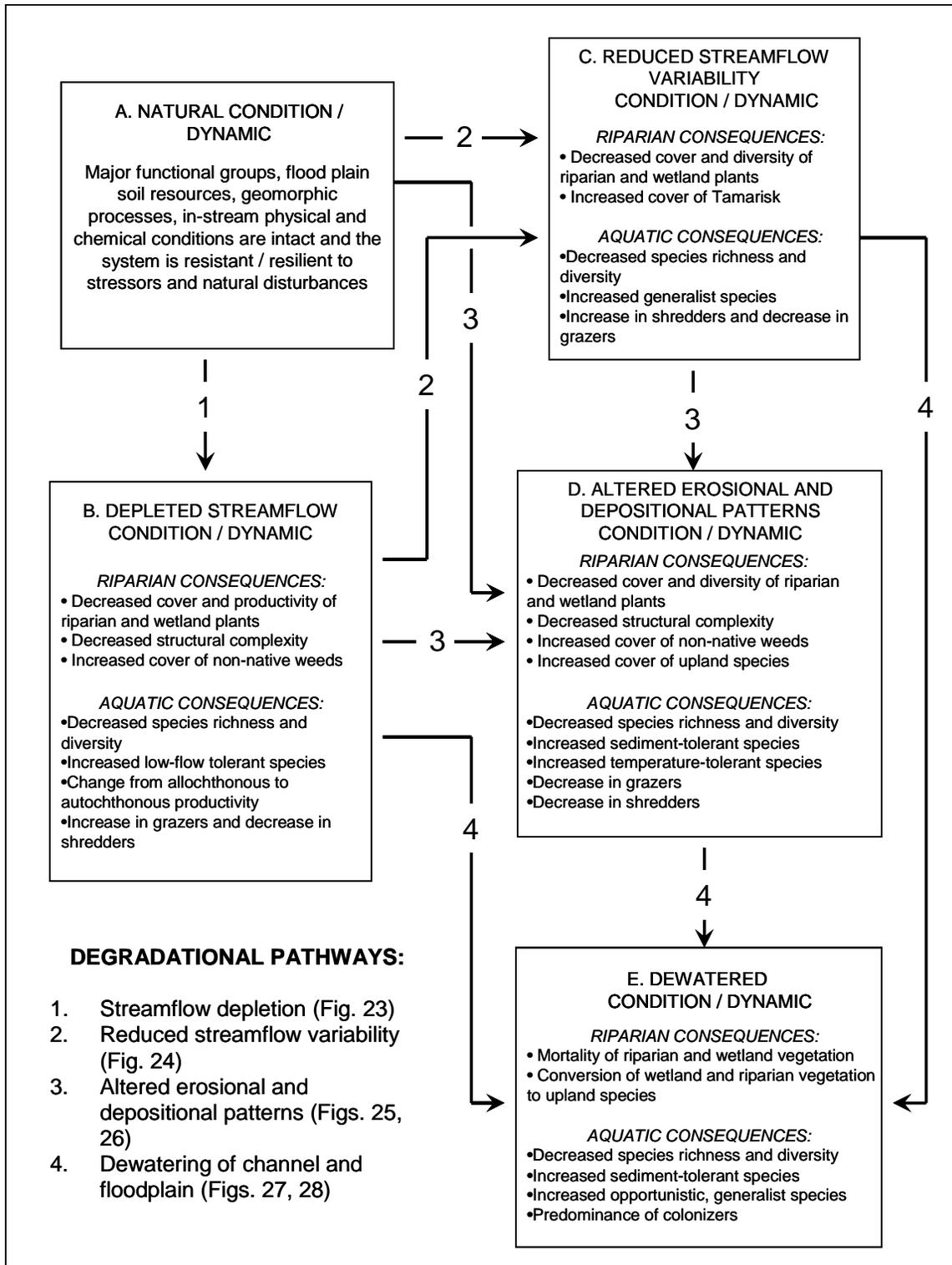


Figure 22. Ecosystem dynamics model describing degraded conditions (B-E) characteristic of riparian and aquatic ecosystems of the Colorado Plateau. Ecological factors and processes responsible for various degradational pathways (numbered arrows) are described in subsequent conceptual process models.

1) **Pathway 1: Streamflow depletion.** The transition from a relatively natural condition to the depleted streamflow condition is common across the Plateau and typically results from three regionally pervasive anthropogenic stressors; damming, streamflow diversion, and groundwater extraction. Climatic drought can influence riparian and aquatic ecosystems directly by depleting streamflow or indirectly, on aquatic systems, through its effects on riparian vegetation cover and productivity (Figure 23). Stream-flow depletions can have a range of effects on riparian and aquatic ecosystems, depending upon the severity of the depletion. Here we describe the effects of small to moderate depletions, which may be subtle, involving reduced over-bank flooding, reduced surface flows, and lowered alluvial groundwater levels. These changes can in turn lead to reduced riparian vegetation cover and site productivity, structural simplification, such as reduced tree height and density (Scott et al. 1999), and reductions in the creation of new riparian vegetation patches, which may favor increases in non-native species, including weeds (Planty-Tabacchi et al. 1995). Direct effects on aquatic macroinvertebrate communities include decreased species richness and diversity and an increase in species adapted to low flow conditions. Indirect effects resulting from a change in riparian community structure include an increase in species that consume algae (grazers) and a decrease in those that consume riparian vegetation (shredders).

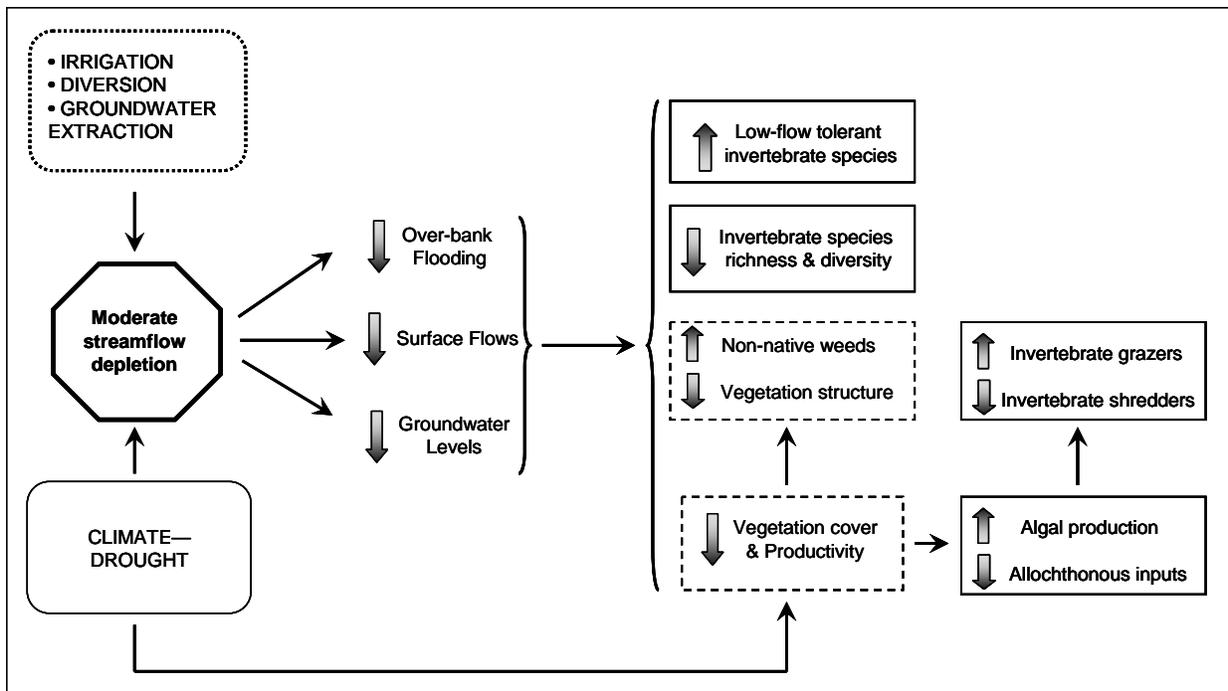


Figure 23. Conceptual model illustrating the processes by which moderate streamflow depletion, resulting from water management and drought can interactively lead to changes in the structure and functioning of riparian and aquatic ecosystems. Symbols are as follows: hexagon = degraded condition; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components.

2) **Pathway 2: Reduced streamflow variability.** A widespread degradational process involving the conversion of riparian cottonwood-willow forest to woodlands dominated by the non-native riparian tree, tamarisk (*Tamarix ramosissima*), has been facilitated by reduced streamflow variability (Figure 24). This represents a common transition in riparian zones throughout the western US (Friedman et al. 2005), particularly along flow-regulated, perennial rivers. The primary mechanism apparently responsible for such a transition involves reduced stream power and sediment transport, resulting in channel narrowing. Establishment of relatively dense stands of tamarisk on un-vegetated portions of the formerly active channel facilitates flood plain formation and channel narrowing through the vertical accretion of sediments. This process may act as a positive feedback mechanism to further reduce sediment transport (Figure 24). Although climate-related fluctuations in precipitation have been implicated as a principle cause of channel narrowing along some un-dammed rivers (Schumm and Lichty 1963, Hereford 1984), damming and diversion of stream flow on most of the larger perennial rivers of the Colorado Plateau have clearly facilitated such a transition (Shafroth et al. 2005). In fact, both climate and flow regulation have likely acted in concert, to varying degrees on different rivers, to produce this transition (Allred and Schmidt 1999, Grams and Schmidt 2002; Figure 24). This conversion appears to be self-promoting to the degree that tamarisk increases the frequency and intensity of fires in the riparian zone, and typically re-sprouts more effectively following fire than native riparian species such as cottonwood (Ohmart and Anderson 1982, Busch and Smith 1995). Thus, the establishment of tamarisk also tends to reduce the cover and diversity of native riparian and wetland plant species (Figure 24). High salinity levels, either natural or human-induced (e.g., by irrigation return flows), also may favor the establishment of tamarisk over native species (Shafroth et al. 1995).

Native fish and macroinvertebrate species have evolved life-history characteristics specifically adapted to natural flow regimes (Bunn and Arthington 2002), and in the arid southwest, flow variability is a critical component of the natural flow regime. Stream-flow alterations that result in an increase or decrease in baseflow, a change in flow patterns (especially peak flows), and the conversion of intermittent to completely dry reaches (Vinson 2001, Weisberg et al. 1990, Blinn et al. 1998) can directly affect native species with specific flow adaptations and requirements, and increase opportunities for the establishment of non-native species that tolerate relatively regulated flows (Blinn et al. 1998, Bunn and Arthington 2002, Haden et al. 2003). This leads to changes in species composition, diversity, abundance, and density of fish, macroinvertebrates, and algae communities (Weisberg et al. 1990, Castella et al. 1995, Benenati et al. 1998, Dahm et al. 2003). Dry stream channels also prevent movement between stream reaches, which can impact species dependent on stream connectivity for population maintenance (Bunn and Arthington 2002). In addition to direct effects of changes in flow regime, changes in macroinvertebrate communities may result from changes in riparian, fish or algal communities. For example, macroinvertebrate functional groups have been shown to shift from grazers to shredders during a period of flow reduction in a Colorado stream as a result of decreased algal production during low flow (Canton et al. 1984).

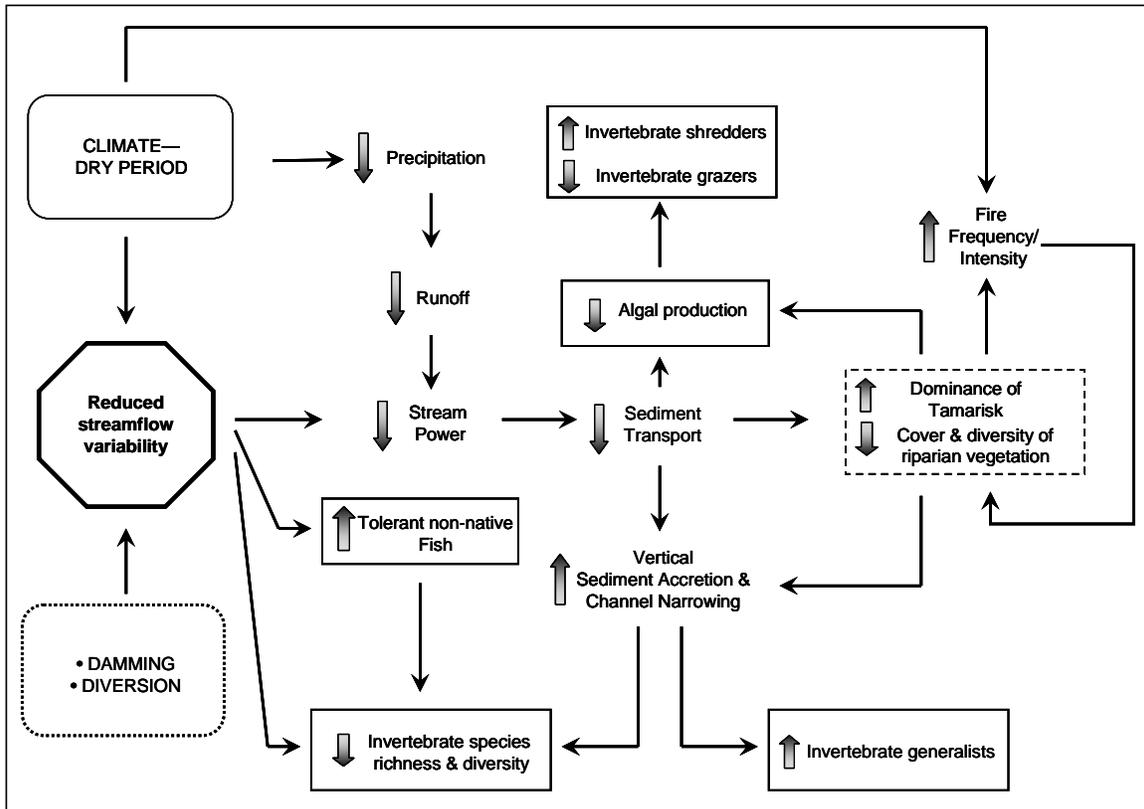


Figure 24. Conceptual model illustrating the processes by which reduced stream-flow variability, resulting from water management and drought, interactively lead to the dominance of non-native tamarisk and related changes in riparian and aquatic ecosystems. Symbols are as follows: hexagon = degraded condition; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components.

3) **Pathway 3: Altered erosional and depositional processes.** This degradational pathway involves significant changes in the balance between sediment delivery to stream channels and the capacity of the channel system to transport the delivered sediments. We present two conditions; one in which there is net erosion or removal of sediment from the system and another in which there is net accumulation or storage of sediment in the system.

Increased erosion of channel, banks and flood plains can have important effects on riparian and aquatic ecosystems and typically result from a number of pervasive on- and off-site stressors as well as natural factors related to climate (Figure 25). Landuse activities including heavy upland grazing, land clearing and the creation of roads and trails are known to decrease upland vegetation cover and increase runoff of water and sediment to receiving streams (Cottam and Stewart 1940, Cook and Reeves 1976). Increased stream power resulting from these factors, can contribute to increased rates of channel, bank and flood plain erosion. These factors typically act in concert with on-site stressors including heaving livestock grazing, trailing, and bottomland clearing, which also promote erosional processes by reducing riparian vegetation cover, increasing hydraulic roughness, and decreasing bank stability (Trimble and Mendel 1995).

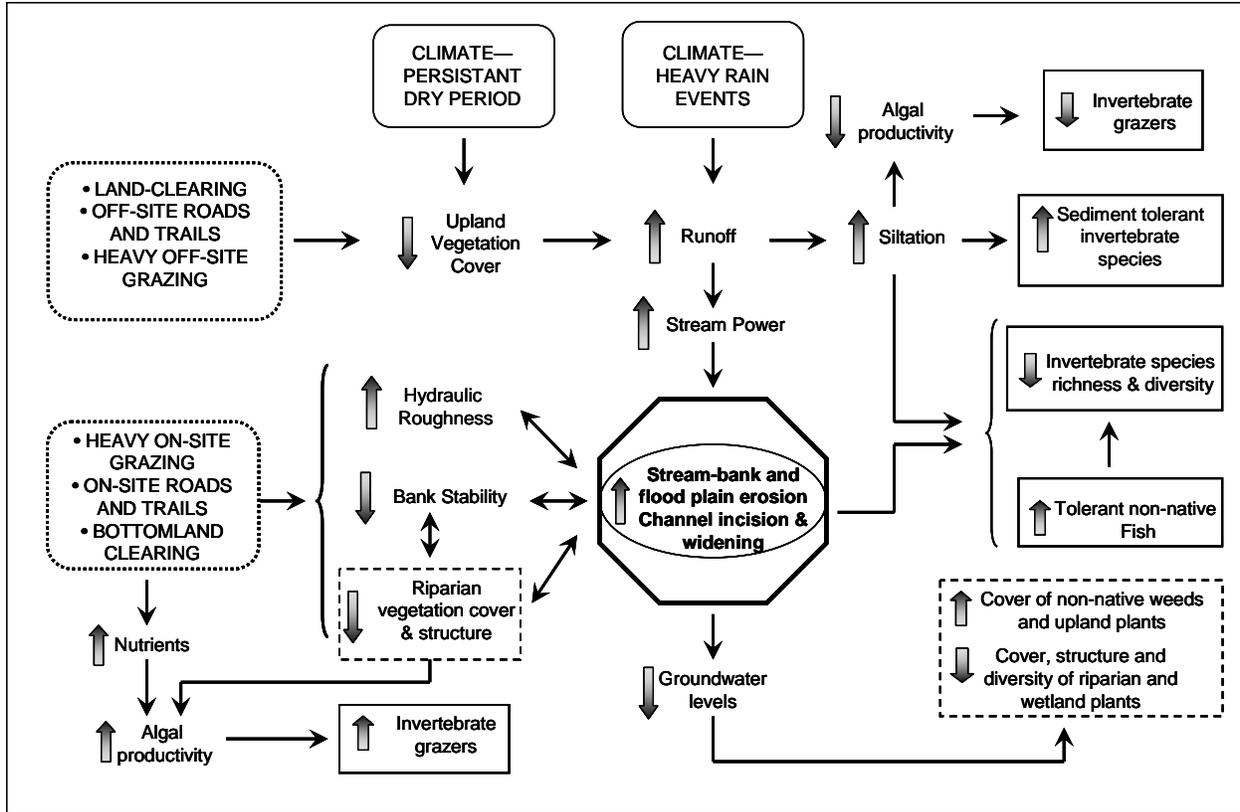


Figure 25. Conceptual model illustrating the processes by which on-site grazing, roads and trails, and off-site climate factors, land-clearing, wildfire, grazing, roads and trails, interactively lead to the process of channel incision and severe bank erosion. Symbols are as follows: hexagon = degraded condition; ellipses = interactive controls; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components.

Increased stream bank and flood plain erosion can create a positive feed-back loop by reducing riparian vegetation cover and structure, resulting in further loss of bank stability, and thus more channel, bank and flood plain erosion (Figure 25). Channel erosion in the form of incision influences riparian ecosystems primarily through its effect on alluvial groundwater levels. Moderate groundwater declines can lead to reductions in the cover, species diversity, and structural complexity of riparian and wetland plants, and increases in the cover of non-native weeds and upland plant species (Figure 25; Scott et al. 1999, Shafroth et al. 2000). Direct effects on aquatic communities include an increase in sediment tolerant macroinvertebrates and fish, and a corresponding decrease in macroinvertebrate species richness and diversity. As in Pathway 2, changes in algal and fish communities result in additional changes to macroinvertebrate community structure. Severe channel erosion associated with arroyo cutting (see Section II. B. 1. b. *iv*) can profoundly alter riparian and aquatic ecosystems by dewatering alluvial aquifers, and is discussed in the context of Pathway 4, below.

A condition of channel instability resulting from increased rates of sediment deposition is often related to severe hillslope erosion and decreased bank stability following wildfire (DellaSala et al. 2004). Under this condition, the loss of upland vegetation cover, in combination with heavy

rains, can increase runoff and upland erosion, delivering more sediment to the channel system than can be transported, leading to channel instability, including aggradation, widening, and incision of accumulated sediments. Loss of riparian vegetation cover to fire can exacerbate this condition by decreasing bank stability, which further contributes to channel widening, bank erosion and channel aggradation (Figure 26). Together, these conditions typically lead to the short-term loss of riparian vegetation on the scale of years. However, on the scale of decades increased sediment storage can lead to increases in riparian vegetation, and enhanced inputs of large woody debris contributes to increased channel stability, complexity and organic matter retention (Robinson et al. 2005). The short term switch from allochthonous riparian input to increased autochthonous algal productivity will result in a change in macroinvertebrate species composition from shredders to grazers. A decrease in riparian cover will also lead to increased water temperatures and temperature-tolerant aquatic species.

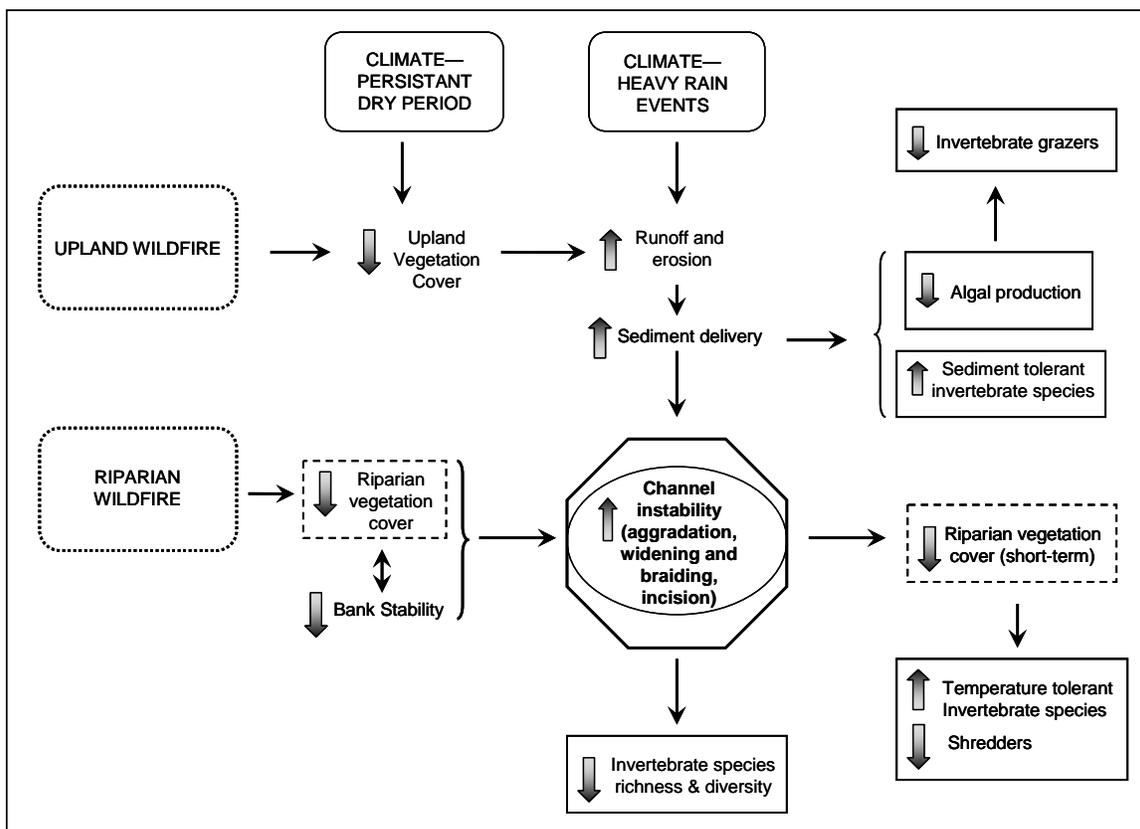


Figure 26. Conceptual model illustrating the processes by which channel instability resulting from riparian and upland wildfires interactively lead to degradative changes in riparian and aquatic ecosystems. Symbols are as follows: hexagon = degraded condition; ellipses = interactive controls; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic riparian components; dashed rectangles = riparian biotic components.

4) **Pathway 4: Terrestrialization of the Riparian Zone.** Conversion of the riparian zone to dominance by upland plants is a common degradational trajectory for riparian and aquatic ecosystems across the region, driven in large part by water management. Terrestrialization is the predicted outcome of reductions in flow variability and / or flow volume (Auble et al. 1997, 2005), and the degree of terrestrialization may serve as an important signal of the extent to which

riparian ecosystems have been altered by water management activities (Innis et al. 2000). Whereas moderate depletions of surface and / or groundwater may alter riparian vegetation composition and structure (Pathway 1; Figure 23), severe surface-water / ground-water depletions can lead to dewatering of the channel and flood plain alluvium, resulting in the mortality of riparian vegetation (Rood and Mahoney 1990, Scott et al. 1999) and ultimately conversion of the site to upland vegetation, often including non-native weeds (Figure 27). Once a system becomes dewatered, the aquatic community will be severely altered, with only generalist, opportunistic, colonizing species present during periodic flow events.

This degradational pathway may also be facilitated by other related processes. Reduced bank stability, resulting from the loss of riparian vegetation, can increase channel and bank erosion (Kondolf and Curry 1986), leading to additional riparian plant mortality. Riparian plant mortality and the terrestrialization process may also be driven by the increased probability of fire, resulting from decreased over-bank flooding, increases in tamarisk and drought, acting alone or in concert (Figure 27).

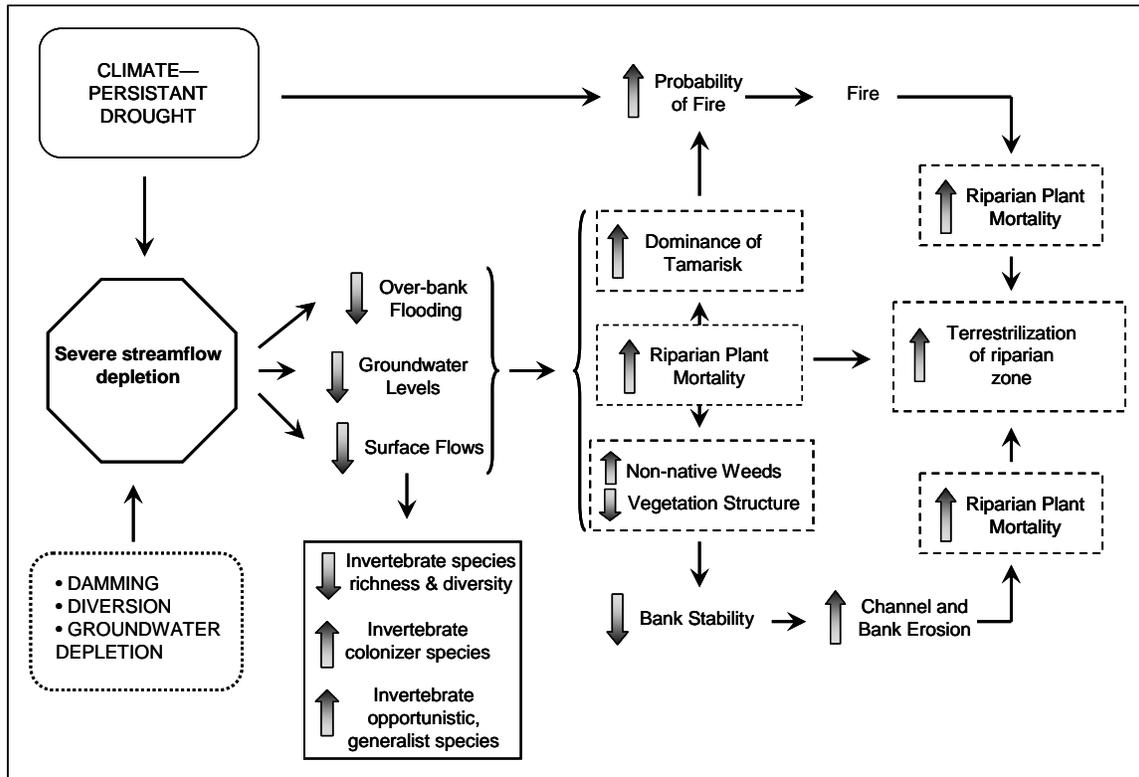


Figure 27. Conceptual model illustrating the processes by which stream-flow depletion resulting from water management activities and drought interactively lead to terrestrialization of the riparian zone and associated changes in aquatic and riparian ecosystems. Symbols are as follows: hexagon = degraded condition; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components.

Another common mechanism driving terrestrialization of riparian ecosystems across the region is channel incision or arroyo-cutting, resulting in part from wide-spread and persistent grazing and

trampling or, somewhat less commonly, roads and trails. Grazing by livestock and other large herbivores are shown to have profound on-site impacts on riparian vegetation including the removal of plant biomass, and simplification of plant compositional and structural diversity (Szaro and Pace 1983, Kauffman and Kruger 1984, Schultz and Leininger 1990). Within riparian zones, grazing reduces the erosional resistance of alluvial surfaces by reducing vegetation cover and trampling directly erodes and destabilizes these surfaces, making them prone to further erosion during high flows (Trimble and Mendel 1995). Ultimately, severe channel incision can dewater channels and flood plains, resulting in riparian plant mortality (Bravard et al. 1997, Scott et al. 2000) and terrestrialization of the riparian zone (Figure 28).

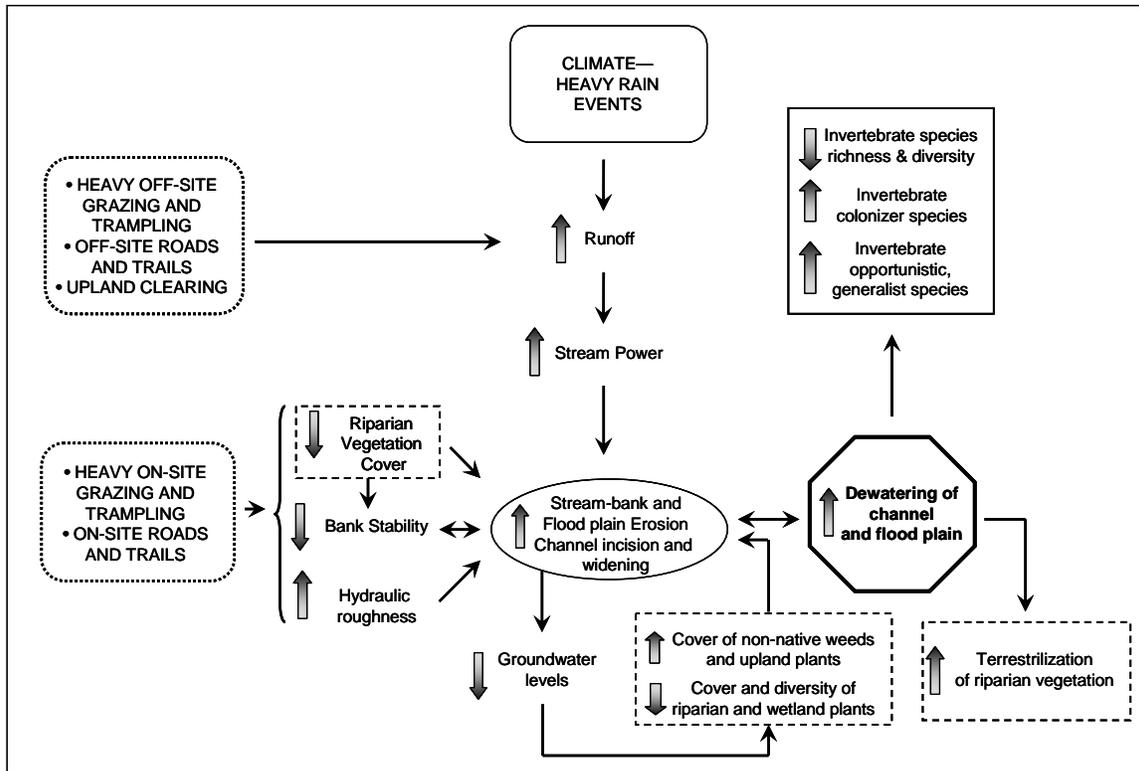


Figure 28. Conceptual model illustrating the processes by which heavy off- and on-site grazing and trampling by large herbivores, roads and trails and heavy rain events interactively lead to dewatering of the channel and flood plain. Symbols are as follows: hexagon = degraded condition; ellipses = interactive controls; dashed rounded rectangles = anthropogenic stressors; solid rounded rectangles = system drivers; solid rectangles = aquatic biotic components; dashed rectangles = riparian biotic components.

Livestock grazing causes increased nutrient loading, reductions in riparian vegetation cover (which changes instream light and temperature regimes), and increased bacterial inputs (Scrimgeour and Kendall 2003, Davies-Colley et al. 2004). These changes will result both directly and indirectly in altered benthic macroinvertebrate communities (Figure 28). The riparian plant community controls the amount of light reaching the stream surface, and strongly influences nutrient cycling and transport, organic matter input, bank stability, stream channel morphology, and subsurface flow into a stream (Gregory et al. 1991). If vegetation is reduced, light and temperature will increase, which may result in greater algal growth. Nutrient loading also contributes to greater algal growth and a potential subsequent change in species

composition. Increased algal growth results in greater invertebrate biomass (Behmer and Hawkins 1986) and a change in community structure (i.e. a change from allochthonous communities to autochthonous communities). For example, certain collector and facultative grazer species utilize areas with high nutrient loads that are associated with changes in algal species communities (Behmer and Hawkins 1986).

As with cattle trails, hiking trails or roads breach stream banks and levees, increasing hydraulic roughness and removing vegetation. At high flows, turbulence created by these features accelerates erosion, creating more turbulence in a positive feedback loop. Trails and road crossings also serve as preferred flow paths for water onto, and off of the flood plain during rising and falling stream flows, causing further erosion (Trimble and Mendel 1995). Finally, because of reduced resistance to flow, un-vegetated trails or roads crossing flood plain surfaces parallel to the stream, would be expected to erode during high flows and could trigger localized channel incision processes (Cook and Reeves 1976), leading to reach-scale terrestrialization of the riparian zone. Off-site roads and trails may contribute to channel incision and terrestrialization by increasing runoff from the uplands, especially during intense rainfall events (Figure 28).

V. IMPLICATIONS FOR ECOSYSTEM CLASSIFICATION TO SUPPORT MONITORING

A. Overview

Stream channel classification systems use similarities of form and / or process to discretely organize complex landscape features that display both relatively continuous longitudinal variation (Vannote et al. 1980) and sharp, local discontinuities (Montgomery 1999, Benda et al. 2004). Although many stream classification schemes have been developed, no single classification system can be expected to adequately serve all purposes. Ultimately, the conceptual basis of any classification system is dependent upon the specific objectives of the particular classification (Mosley 1987, Kondolf et al. 2003). Earlier form-based classifications, like that developed by Rosgen (1994), provide useful physical descriptions of different stream types that facilitate communication about streams among people with different backgrounds and experiences, but is not considered suitable for purposes such as assessing stream stability, inferring geomorphic process, predicting geomorphic response to natural or anthropogenic disturbances, or for guiding stream restoration or monitoring activities (Juracek and Fitzpatrick 2003). Ideally, a geomorphic classification system would be process-based, applicable across a range of spatial and temporal scales, and capable of assessing probable channel responses to a range of natural and anthropogenic disturbances (Naiman et al. 1992). Such a process-based classification could expand the scope of the conceptual models presented here, by identifying the spatial distribution of stream types that differ in their sensitivity, or resistance and resilience, to anthropogenic stressors of concern to Park managers, and thus provide a basis for objectively prioritizing and selecting sites for monitoring that would be more broadly representative of similar stream types across the Colorado Plateau (Frissel et al. 1986).

B. Proposed Stream Classification Framework

Here, we review a hierarchical, process-based approach to channel classification that was initially developed for mountain drainage basins in the Pacific Northwest to assess channel functional condition and potential response to natural or disturbance-related changes in sediment supply, and stream discharge, across a range of spatial scales. The conceptual framework of this classification system is briefly summarized below and presented in detail in Montgomery (1999), Montgomery and Buffington (1993, 1998), Montgomery and McDonald (2002). Such a framework could be adapted to the Colorado Plateau and used to increase the efficiency and effectiveness of riparian and aquatic ecosystems monitoring, as part of the I&M Program for the NCPN and SCPN.

1. Channel Processes

Although stream channel morphologies may vary widely, they respond to the same basic set of factors. On the scale of tens to hundreds of years, channel morphology is shaped by (1) the delivery of sediment from adjacent uplands, (2) the competence of the channel to transport the delivered sediment downstream, and (3) the direct and indirect influence of vegetation on these processes (Figure 29). Channels typically change, in an indeterminate fashion, to variations in sediment inputs and discharge, by adjustments in width, depth, flow velocity, sediment size, bed forms, and channel pattern. However, conceptual models, and a large body of empirical evidence, suggest that differences in channel form, process, and physical setting, influence the probability of a specific response to a particular disturbance. A hierarchical classification of factors influencing channel processes, over a range of spatial and temporal scales, can enhance understanding and assessment of a stream channels potential response to disturbance by grouping functionally similar physical environments and channel types (Frissel et al. 1986, Montgomery and Buffington 1998).

2. Hierarchical Channel Classification

Montgomery and Buffington (1998) present a spatial hierarchy (Figure 30a-d) that provides a basis for comparing channels and channel processes at progressively finer scales of resolution. For example, watersheds within the Canyonlands sub-division of the Colorado Plateau Physiographic Province, tend to be more similar in terms of topographic relief, climate, and geology; thus, channels in this region would generally be more comparable in terms of drainage area, stream-flow patterns, and bed material, than channels elsewhere on the Plateau. Within watersheds, different valley segments can be identified based on sediment production, transport, and the nature of valley fill material. For example, valley types typical of the Colorado Plateau, include bedrock valleys (Figure 30d), which are typically narrow and contain little or no river-deposited sediments, indicating that sediment transport capacity in these valleys exceeds supply. In alluvial valleys, by contrast, supply equals or exceeds transport capacity, and channels may be confined to narrow flood plains (Figure 30b) or unconfined on wide flood plains (Figure 30c). Channel reaches represent similar channel morphologies over distances of tens to thousands of meters within which smaller channel units, classified as distinct channel habitat features such as pools, rapids, or channel bars are found (Figure 30d). Channel reach types would include Bedrock, Colluvial, and a variety of possible Alluvial reach types, which could be part of a classification system developed specifically for the Colorado Plateau.

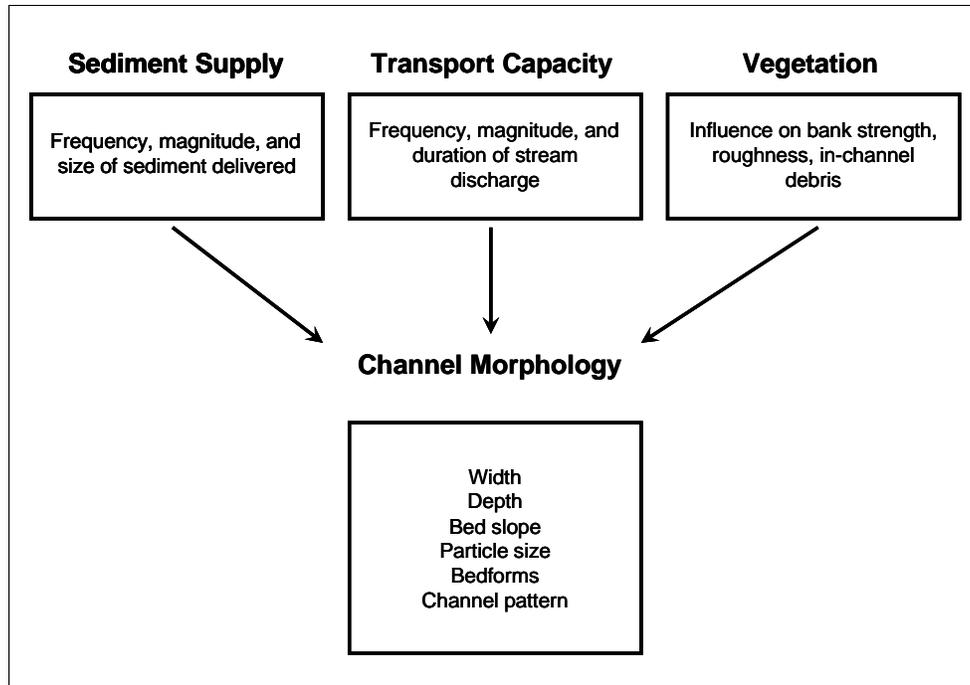


Figure 29. The influence sediment supply, stream transport capacity, and the direct and indirect effects of vegetation on stream channel morphology. From: Montgomery and Buffington (1998).

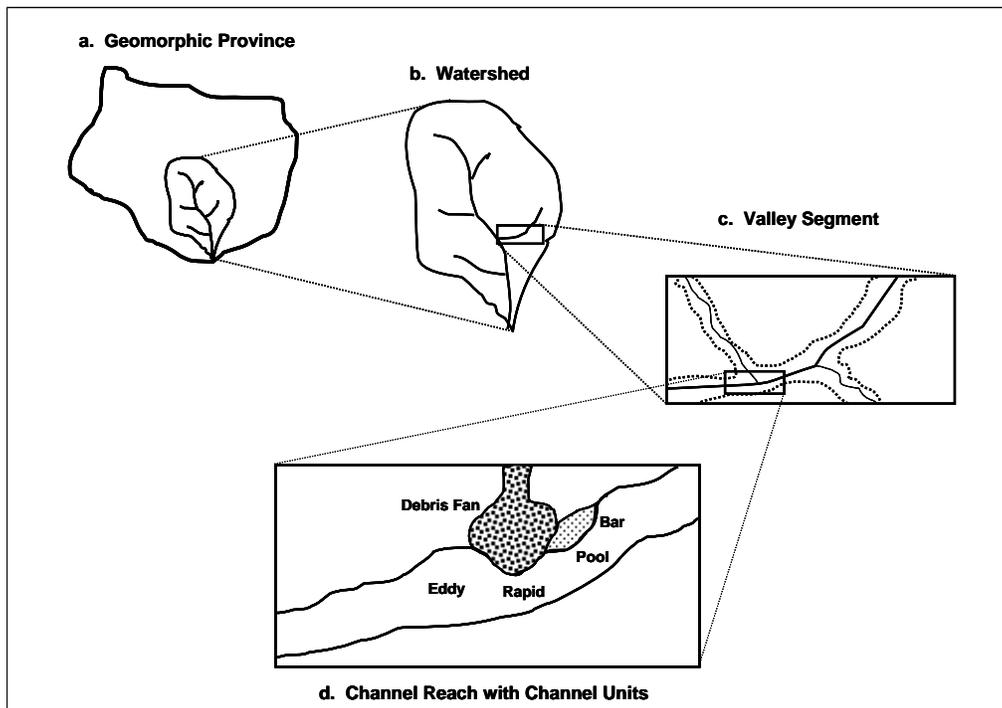


Figure 30. Hierarchical levels of channel classification, reflecting progressively finer spatial scales from the Geomorphic Province to Channel Reach and Channel Units. Adapted from: Montgomery and Buffington (1998).

3. Differences in Reach-level Responses

Differences in reach-level morphology and channel processes result in different responses to similar perturbations in stream discharge or sediment supply. For example, bedrock or steep alluvial reaches with large-sized bed material, are much more resilient to increases in sediment supply or discharge, because of their high sediment transport capacities, than are low-gradient alluvial reaches (Montgomery and Buffington 1998).

4. Channel Networks

Schumm (1977) described three, geomorphically distinct zones within a watershed; headwater zones that are source areas for sediment, zones of sediment transport by rivers, and zones of deposition along lower gradient channel reaches. Thus, the spatial positions of channels within a channel network, condition their potential response to changes in sediment supply or discharges, resulting from natural disturbance events or human land-use activities within a watershed (Benda et al. 2004). For example, a temporary increase in sediment production within a watershed would be expected to move relatively quickly through transport reaches, with transient effects on channel process and form, but would be likely to accumulate as storage in channels and flood plains in depositional reaches, with consequent long-term effects on channel form and process. Thus, channel positions in a channel network, which represent transitions between transport and depositional dominated processes, would be especially sensitive indicators of changes in sediment flux within a watershed and thus serve as important locations for monitoring changes in watershed condition.

Finally, consideration of other factors such as degree of channel confinement (Figure 31a-d), bed slope, and amount of riparian vegetation provide further insight into the relative resistance and resilience of channels to natural or anthropogenic perturbations. Regional development of empirical relations between channel reach types and bed slope, would allow rapid assessment of channel types, at the watershed scale, using topographic maps and digital elevational models (DEMs) (Montgomery and Buffington 1998). Development of a hierarchical classification scheme for the Colorado Plateau, similar to the one described here, would provide a powerful tool for assessing the condition and evaluating the relative resistance and resilience of riparian and aquatic ecosystems to various natural disturbances and anthropogenic stressors, and provide a more rational basis for prioritizing and selecting reaches for vital signs monitoring.

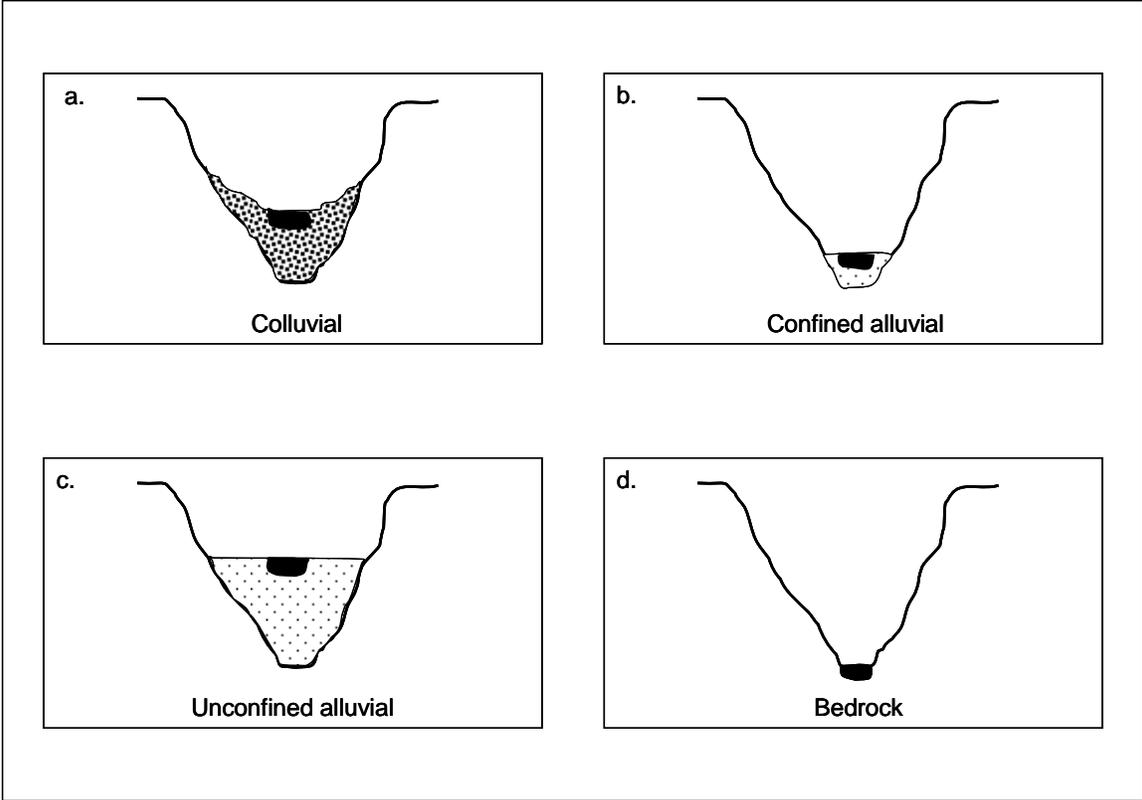


Figure 31. Idealized valley segment channel morphologies, including (a) colluvial, (b) confined alluvial, (c) unconfined alluvial, and (d) bedrock channels. Adapted from: Montgomery and Buffington (1993).

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