Review Assessment and Recommendations Regarding Terrestrial Riparian Vegetation Monitoring in the Colorado River Corridor of Grand Canyon

Final Report

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Part 1:
Review and Synthesis of
Colorado River Riparian Vegetation Monitoring

ABSTRACT

We reviewed literature on riparian vegetation studies in the Colorado River corridor of Grand Canyon National Park and surrounding areas, including the vegetation-related elements of other studies, such as bird- and herpetofaunal surveys, and summarize it in a number of ways. First, a narrative account of the development of riparian vegetation is presented and related to five more or less distinct periods in the hydrograph of the Colorado River and emphases of projects: pre-dam, colonization era I (exotics), colonization era II (natives), high flows era, recovery era, and an intensive study era. An attempt is made to relate the status and trends in riparian vegetation in each era to peculiarities of the hydrograph at that time.

Second, retrospective power analyses are performed on vegetation monitoring methods used most recently by Kearsley and Ayers. Using data from 1998 and 1999, we determined the minimum detectable effect size and the number of years of trend required to detect change in total vegetation volume, species richness, total foliar cover, and Shannon diversity for four broad vegetation types in the sites. In general, the tests had moderately low power. Most comparisons would have required from three to seven times the number of sites currently used to determine that the 1998 / 1999 changes were significant, although many would have required more than 100 sites. Also, between two and seven years of consistent changes like those seen between 1998 and 1999 would have been required to find a significant trend in the data, and two (TVV change in dense tamarisk stands and species richness in sandy bar-top areas) would have required more than 10 years.

Third, to determine the representativeness of the 10 monitoring sites above Diamond Creek, the proportions of vegetation types in them was compared to random samples along the river. First, monitoring site vegetation classifications were cross-walked to a previously used bird habitat classification. This was compared to proportions of these same vegetation types encountered in transects from the water's edge to the bottom of the old high water zone in more than 320 random points from Lees Ferry to Diamond Creek. Paired t-tests showed no difference in proportions, and a regression of sampling site proportions on random point proportions had a slope not different from one and an intercept not different from zero. However, an examination of the studentized residuals from the regression analysis indicated that some vegetation types (most notably dense tamarisk stands) were oversampled while others (containing significant amounts of Baccharis) were undersampled.

Fourth, we used published habitat descriptions for juvenile native fish to determine the suitability of current vegetation sampling for aquatic resources. Due to limited overlap with the reaches considered critical to fish survival, only one reach was studied in detail. Tests for change in the length and area of vegetated shoreline were relatively weak, with a power of 20 - 30 percent. A consistent change of the size seen between 1998 and 1999 for four to seven years would be required before a determination could be made of a significant change in these measures.

Fifty, data from a several sources was used in producing a vegetation / plants database for the Colorado River corridor of Grand Canyon National Park. Published floras and checklists were used to determine which species would be included. Information on synonymy and the current status of nomenclature for all species was taken from the USDA Plants database. Information on the distribution of the plant both locally (upper and lower elevation limits, seasonality) and globally (source of the taxa) was included. Finally, the wetland indicator status of each species was taken from the U.S. Fish and Wildlife Service's database and the growth form and other life history data were included as well.

We are awaiting information on bird community / habitat relationships from the Colorado Plateau Research Station for testing the utility of vegetation data currently collected for bird monitoring purposes. In their description of habitats, Sogge et al. (1998) published only the statistical significance of predictive equations of bird community attributes based on habitat variables, and not the equations themselves. A request for the equations was made in early November, but the author with the files (M.W.) has not yet been able to locate them.

INTRODUCTION

The Grand Canyon Monitoring and Research Center's (GCMRC) objectives for vegetation monitoring are to measure change in riparian vegetation as they relate to several important biotic and social elements. These include the nesting and other habitat requirements for endangered terrestrial and aquatic fauna, the recreation needs of the boating public such as open beaches, aesthetics, and access to camping areas, and the introduction and spread of invasive exotic plant species. These objectives define a very specific set of information needs.

In contrast, in the century since the first ecological survey of the region (Merriam 1890), the information generated by riparian studies in Grand Canyon have been as diverse as the goals and methods identified by the individual investigators. There have been taxonomic surveys (e.g., Clover and Jotter 1944, Ayers et al. 1995), plant databases (e.g., Theroux 1976), ecological studies (Brian 1982, Stevens et al. 1995), wildlife habitat assessments (Sogge et al. 1993, 1998), and monitoring studies (Phillips, B.G. et al. 1986, Kearsley and Ayers 1998). Investigators have used various sampling methods (points, transects, quadrats, plotless rélévés, entire river trips) and made decisions about both sample locations (fixed, random, or regular) and the scale to be employed (individuals, patches, and landscapes). Furthermore, individual investigators have chosen methods for reasons which are based primarily on their experiences and the history of the study in question (see Kearsley and Ayers 1996a, p. 2).

Data from all these studies have both spatial and temporal components. For example, collection records, habitat descriptions, plot records, and transect data all have dates and localities associated with them. In addition, many of these have had overlapping study sites (e.g., Phillips, B.G. et al. 1986, Waring and Stevens 1986). Thus, since the objectives of the GCMRC vegetation monitoring program concerns measuring changes over time, at least some of this information should be applicable. However, to date there has been no systematic attempt to organize all this information.

Here we summarize the results of habitat, ecological, and monitoring studies which have been conducted in the riparian corridor of Grand Canyon and adjacent areas and, to the extent possible, integrate them and relate them to the GCMRC monitoring goals for terrestrial vegetation. By doing so, we provide other investigators, managers and stakeholders with a better picture of what vegetation data has become available so that a) unnecessary repetition of previous work can be avoided and b) gaps in the understanding of the dynamics of the riparian system can be identified. We also assess the relevance of existing data to habitat requirements for endangered terrestrial and aquatic vertebrate species, primarily the southwest willow flycatcher and humpback chub. Finally, we make recommendations regarding the testing and use of methods and technologies to be applied to future vegetation monitoring based on their ability to provide information needed by GCMRC managers.

Objectives

The objectives of the work reported on here are:

1. To review previous riparian vegetation related monitoring and research literature from the Grand Canyon ecosystem This includes the Colorado River corridor between Glen Canyon

Dam and the Grand Wash Cliffs and adjacent similar areas (e.g., Havasu Canyon, pre-dam Glen Canyon). This review will include a summarization of the results of taxonomic, ecological, wildlife habitat, and monitoring studies, both published and unpublished.
2. Determination of how previously collected data relate to habitat requirements for endangered terrestrial species (southwestern willow flycatcher) and aquatic species (native fish populations), with special reference to both floristic and structural requirements for successful nesting, reproduction, and growth.
3. Assessment of the sampling used in previous monitoring studies with regard to their ability capture and adequately represent the whole riparian vegetation of the river corridor and to measure change in vegetation units over time.
4. Review of the methods which have been used to study vegetation to assess their relative merits for monitoring at the scales imposed by a riparian system constrained by a deeply incised canyon. From this review, we make recommendations regarding protocols which should be tested or applied to future long-term monitoring in this system.

Narrative Account

One of the elements required for the synthesis of historical and more recent information on vegetation dynamics was the construction of a narrative account based on many sources. Literature databases, including the GCMRC library, the Grand Canyon Association's Bibliography of the Grand Canyon and Lower Colorado River (Spamer 2000), Northern Arizona University's Special Collections Archive, the library at Grand Canyon National Park, and the library of the Museum of Northern Arizona were searched for relevant published and unpublished material. When available, staff associated with these resources and researchers who had conducted studies in the canyon were consulted for information about other potential sources of data.

The studies of vegetation dynamics in the post-European settlement era of Grand Canyon could be divided into five periods which were separated based on differences in patterns hydrology of the river and the emphasis of those conducting the studies (Table Z). These are 1) pre-dam surveys (before 1963), 2) initial colonization descriptions (1963 - 1973), 3)second wave colonization descriptions (1973 - 1983), 4) high flows impacts and recovery studies (1983 - 1990), and 5) science and monitoring studies (1990 - 1999). Some studies overlap into more than one category, but motivations for the studies seemed to follow a predictable, time-related sequence. Accordingly, the narrative follows this general framework.

Table 1. Literature sources for river corridor vegetation.

Year	Citation	Papref	Time	Type	Relevant Information	Ancillary Data
1936	Dodge 1936	2814	1936	L	A list of the tree species found in Grand Canyon National Park, mostly rim species, but some mention of river corridor presence.	None

Table 1. Literature sources for river corridor vegetation.						
1936	Patraw 1936	2760	1936	L	Early plant species list for the entire park	None.
1941	Clover and Jotter 1941	2821	1941	L	A list of the cactus species found in the Grand Canyon of the Colorado, plus other reaches from Green River to the Davis Dam	Location map
1941	Haring 1941	2878	1940	S	List of mosses found in Grand Canyon, including the river corridor and spring areas.	Some plant associations described.
1946	Clover and Jotter 1946	2699	1938 - 1939	L	A floristic inventory of the Canyon of the Colorado and its tributaries, with a description of physical factors leading to unique habitats	Some site photos
1946	Haring 1946	2879	1940, 1944	S	Enlarged list of mosses found in Grand Canyon, including the river corridor and spring areas.	Some plant associations described.
1968	Nichols 1968	2828	1968	I	Botanical description of soils-defined regions of Unkar Delta, along with observations on cover and productivity of major plant species.	None
1971	Martin 1971	2811	1971	S	List of tree and shrub species found in the river corridor	None

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1975	Phillips 1975	2728	1970 - 1974	S	Vegetation description of the lower Grand Canyon in Pleistocene and present.	Species lists from middens and surveys.
1975	Werthiemer and Overturf 1975	2810	1975	O	Review of biological research done in Grand Canyon since the earliest work in 1850s (Ives)	Unusual References
1976	Aitchison et al. 1976	2813	1973 - 1976	M	Repeat photography of campsites for monitoring human impacts and impacts of dam operations.	Original photographs at MNA
1976	Bain 1976	????	????		List of grand canyon plants at MNA Cited by Ohmart 1982.	
1976	Carothers and Aitchison 1976	1730	1972 - 1976	S	Ecological survey of the river corridor including major terrestrial and aquatic components.	Appendix II-2 by Theroux has species by 10-mile increments
1976	Karpiscak 1976	2716	1889 - 1974	I	Analysis of change in vegetation after completion of Glen Canyon Dam.	Repeat photography examples.
1976	Theroux 1976	2717	1976	S	Description of the vascular plants of the Colorado River corridor of Grand Canyon National Park.	Appendices have occurrences sorted by 10-mile increments.
1977	Phillips et al. 1977	2837	1973	O	Riparian vegetation map of Grand Canyon National Park from Lees Ferry to Diamond Creek.	Some species lists

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1978	Green 1978	2809	1978	I	Line intercept and belt transects for cover, density and frequency at 5 camping beaches to detect human and other impacts.	None
1980	Green et al. 1980	2700	1978 - 1979	M	Line intercept and belt transects through 5 beaches to detect impacts of humans on density and cover of major species	Aerial maps of beaches,
1982	Brian 1982	1727	1981	S	Line intercept data from 34 stands of willow above Phantom Ranch for both canopy and understory species	Summary info available, map and data sheets missing.
1982	Ohmart 1982	2844	1977	O	Map of new high water zone vegetation in Grand Canyon and lists of plant species and listed and candidate species in the Park.	Maps, species lists, candidate species lists.
1982	Warren et al. 1982	2804	1982	S	Vegetation map derived for all of Grand Canyon National Park from aerial photos and ground work. No river corridor work.	Map itself was not found
1983	Ruffner, 1983	2533	1979 - 1983	I	Study of effects of beaver on willow includes estimates of numbers of willow patches in geomorphic settings.	List of number of willow patches by short “reach” in 1979 and 1983.

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1985	Anderson and Ohmart 1985	1779	1972 - 1979	O	Predictive models of bird community variables (density, S, H’, guild structure) based on veg data over a 5 year monthly sampling routine.	None
1985	Stevens and Waring 1985	2406	1980 - 1984	I	Large quadrat data comparing species numbers, plant height and condition in zones, reach types, substrates and river miles (‘83 flood effects)	Data files appended to report
1986	Anderson et al. 1986	2831	1985	S	Remote sensing of shoreline habitats from Lees Ferry to Diamond creek, and checking for impacts of high vs. low flows	Video footage of entire river corridor shoreline.
1986	Phillips et al 1986	1733	1981 - 1982	I	Aesthetic indices in quadrats and transect data on vegetation used to assess human impacts on camping beaches	Clear methods section, some aerial photographs and maps
1986	Pucherelli 1986	1734	1965 - 1984	M	Mapping of old- and new-high water zone vegetation from aerial photos of 7 sites in 1965, 1973, 1980, 1985.	Data files on vegetated area by zone and river mile
1986	Stevens and Waring 1986	1737	1983 - 1985	I	Analysis of data from plant transects and quadrats and soil samples aimed at determining the effects of high flows from 1983 - 1985.	None

Table 1. Literature sources for river corridor vegetation.						
1986	Waring and Stevens 1986	2696	1984, 1986	I	Quadrat data on seedling establishment, flood-induced mortality, and flowering phenology of major woody species	Data analysis files appended to report.
1986	Warren and Schwalbe 1986	2806	1983 - 1986	S	Description of vegetation zonation is incidental to description of herp. habitats, but describes effects of fluctuations. Very high densities.	None
1987	Anderson and Ruffner 1987	2835	1983 - 1986	I	Transect and tree data on age class distribution, seedling survival, reproductive effort and growth of woody old high water zone species.	None
1987	Brian 1987	2803	1982 - 1984	M	Aerial photo interpretation with ground truthing at 7 sites to measure impact of 1983 and 1984 high flows.	Site maps and species lists.
1987	Phillips et al 1987	2701	1987	O	Annotated checklist of plants based on herbarium records and a computerized list.	None
1989	Stevens 1989	1735	1989	I	Pot studies and censuses of quadrats aimed at understanding the interactions among willow and tamarisk in Grand Canyon.	None

Table 1. Literature sources for river corridor vegetation.						
1992	SWCA 1993	2834	1992	S	Initial plot survey data for transects, quadrats, and marsh belt transects measuring basal area and transect width.	None
1993	Stevens and Ayers 1993	1738	1991 - 1993	M	Quadrats, belt transects, plus soil samples, plant moisture stress, and root densities in the river corridor	Plot photos and descriptions, Data at GCMRC / CPRS (?)
1993	Weiss 1993	2808	1991, 1995	I	Assessment of impacts of low and high steady flows on return current channel marsh habitat sizes and numbers.	Video used in MIPS analysis at GCMRC.
1993	Werth et al. 1993	1741	1991	I	The origninal GIS database for Grand Canyon resources, including aquatic, terrestrial, cultural, historical, etc.	Three large maps: GIS sites, Site 5 Resource sites, Site 5 GIS topography.
1994	Carpenter et al. 1994	2786	1995	I	Groundwater fluctuations as a function of river fluctuations.	Soil profiles in beach cross-sections
1994	SWCA 1994	2832	1992, 1993	M	Repeat sampling of a reduced number of transects and quadrats in lower Grand Canyon	None
1995	Ayers et al. 1995	1748	1988 - 1994	L	New species records from Grand Canyon, including those incidental to research of Stevens, Ayers, Phillips, and Brian.	None.

Table 1. Literature sources for river corridor vegetation.						
1995	Bowers et al. 1995	2333	1889 - 1995	M	Repeat photography of river corridor, especially old high water and upland species, was used to estimate longevity / turnover.	Some photographs available.
1995	Stevens et al. 1995	1739	1995	O	Soil data and plant transects describing natural history of riverine wetland areas in Grand Canyon: development, colonization, and succession.	None
1995	Stoffle et al. 1995	2830	1995	M	Setup for monitoring of culturally significant plants in the river corridor. Site descriptions and sampling descriptions.	None
1996	Austin et al. 1996	2829	1996	M	Repeat sampling of belt transects and line-intercept transects to test for changes in abundance of culturally significant species	None
1996	Austin and Osife 1996	2824	1995-1996	I	Repeat sampling of belt transects, line transects, and plots to detect changes in abundance and / or condition of vegetation	None

Table 1. Literature sources for river corridor vegetation.						
1996	Kearsley and Ayers 1996	2353	1995 - 1996	I	Assessment of impacts of 1996 controlled flood via survey of vegetation floristics and structure, and soil seed banks using air photo based maps.	Data files and digital maps at GCMRC / appended to report.
1996	Kearsley and Ayers 1996	1725	1993 - 1994	M	Data on change in quadrats and belt transects in marshes extending Stevens and Ayers (1993) data sets by 10months.	Data files at GCMRC, Species list from river corridor appended.
1996	Kearsley et al. 1996	2412	1994 - 1995	M	Monitoring of 11 sites using aerial photo-based maps and transects / quadrat data from Stevens and Ayers (1993) and earlier.	Data files and digital maps at GCMRC / appended to report.
1996	Waring 1996	2394	1965 - 1992	M	Aerial photo interpretation with ground truthing to examine changes in area covered by New- and Old- high water zone vegetation	GIS maps in GCMRC database, data files appended in hard copy
1996	Webb 1996	2359	1889 - 1995	M	Repeat photography of river corridor to document changes in geomorphology and biology since the Brown / Stanton expedition	Some photographs included.

Table 1. Literature sources for river corridor vegetation.						
1997	Bowers et al 1997	2549	1995 - 1996	O	Repeat photography and carbon dating for description of succession processes on debris flow terraces.	Species lists, areal extent estimates, and lifespan estimates for many upland species encountered in the study.
1998	Converse et al. 1998	2687	1990 - 1993	S	Description of vegetated shoreline incidental to fish habitat descriptions.	None.
1998	Sogge et al. 1998	2705	1994 - 1995	S	Vegetation maps and description of vegetation types incidental to description of bird habitats	Predictive models for bird community attributes.
1999	Kearsley and Ayers 1999	2768	1997 - 1999	M	Interpretation of vegetation change using survey of vegetation floristics and structure based on vegetation maps from 11 sites.	Data files appended to report, digital maps and data at GCMRC
1999	Kearsley and Ayers 1999	2838	1996 - 1997	M	Interpretation of aerial photographs of 11 monitoring sites based on extensive ground work.	Data tables in hard copy and GIS maps in the GCMRC database
1999	Kearsley and Ayers 1999	2667	1995 - 1996	I	Assessment of impacts of 1996 controlled flood via survey of vegetation floristics and structure, and soil seed banks using air photo based maps.	None

Table 1. Literature sources for river corridor vegetation.						
1999	Parnell et al. 1999	2763	1996	I	Analysis of soil and soil water data on the fate of vegetation buried in the 1996 controlled flood.	None
2000	Brian 2000	2779	2000	L	Special status plants in Grand Canyon National Park	Plant and habitat photographs of several river corridor species.

Because the hydrograph is the most important driving variable in terrestrial riparian systems (Malanson 1993), the discussion of each era in the following section begins with a description of the hydrograph. In addition to unusual flow events, both low and high, the description includes the general pattern of dam releases. Vegetation changes over the entire period are described as well, and the attempt is made to tie these to flows.

Table Z. Hydrograph characteristics of five eras of vegetation research and monitoring in the Colorado River corridor of Grand Canyon.

Period	Hydrograph Characteristics
Pre-dam (before 1963)	Essentially unregulated. Peak flows (3500 cms) in spring and late summer, little intra-day variability
Colonization I (1963 - 1973)	Load-following discharges. Reduced peak flows (1250 cms) with high intra-day range (1050 cms)
Colonization II (1973 - 1983)	Load-following discharges. Reduced peak flows (1250 cms) with high intra-day range (1050 cms)

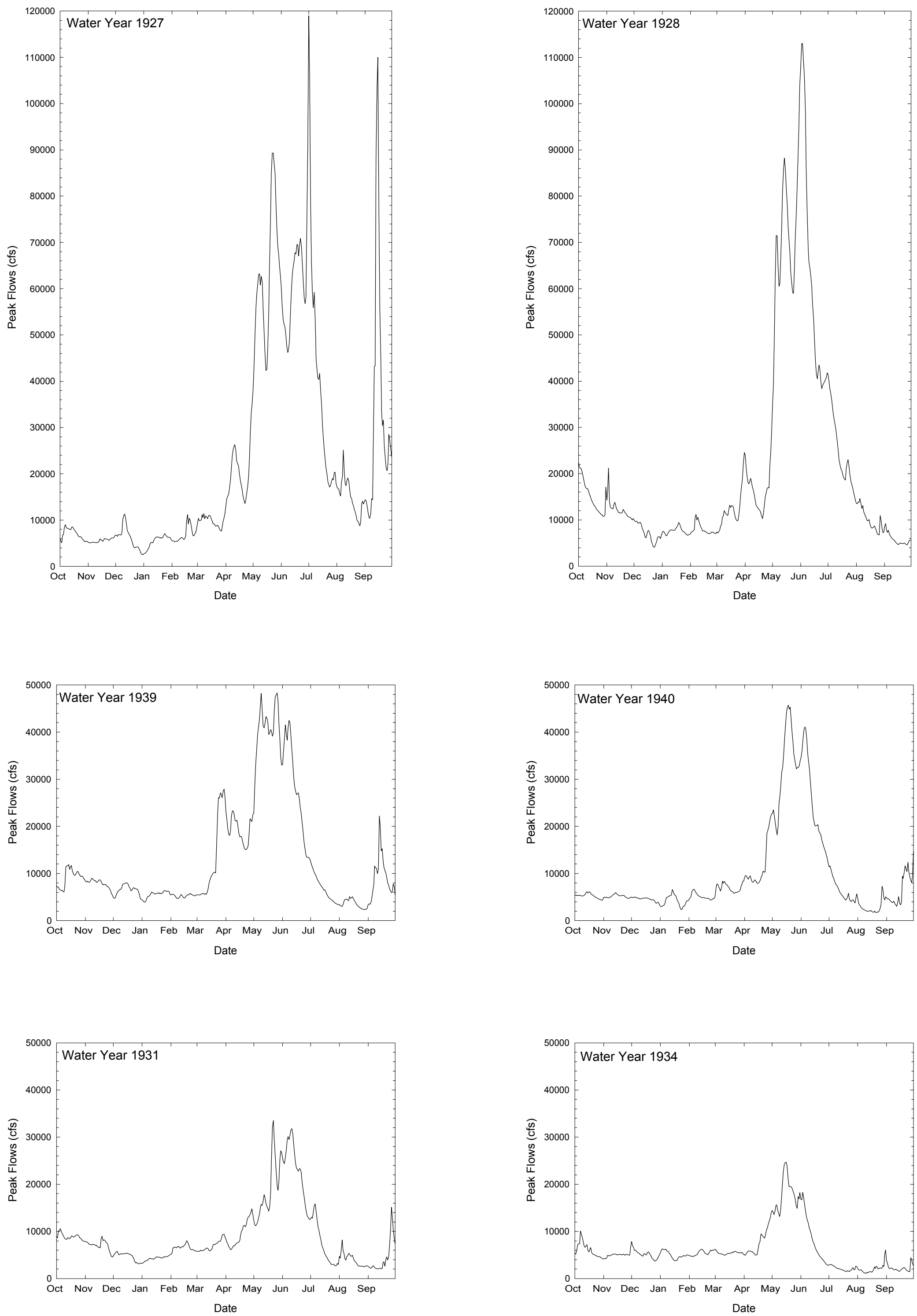
High Flow Impacts (1983 - 1990)	Sustained high flows. High peak flow in 1983 (3500 cms) and 1984 - 1987 (1300 - 1700 cms) then load following.
Interim Flows (1990 - 1995)	Reduced peak flows (1150 cms) and intraday ranges (200 cms). Test flows in 1990 and 1991
Adaptive Management (1996 - 1999)	Similar to interim flows, but with controlled flood (1996) and high steady flows (1997)

Narrative Account

Pre-dam survey era (pre-1963). Before the completion of Glen Canyon Dam, the hydrograph of the Colorado River in Grand Canyon was driven by spring snowmelt floods and occasional large tributary inflows produced by monsoonal late-summer rains (Dolan et al. 1974, Carothers and Aitchison 1976, Webb et al. 1999). Annual return floods during this time were on the order of 2500 cubic meters per second (cms). The 10-year flood was on the order of 3500 cms and the estimate of the historical high flows was roughly 8500 cms (Turner and Karpiscak 1980). The median flow during this same period was less than 350 cms and flows less than 50 cms were not uncommon during the winter (Webb et al. 1999).

Although most discussions of the pre-dam riparian zone written during the last 30 years or so have presumed that these floods removed all vegetation below the annual return flood elevation (Carothers and Aitchison 1976, Stevens et al. 1995, Bureau of Reclamation 1996), an examination of botanical reports from riparian areas showed that most of the elements of present day riparian vegetation were present before dam construction. Several early publications describe stands of willow, baccharis and tamarisk at the river's edge in Grand Canyon (Dodge 1936, Clover and Jotter 1944). Similarly, written accounts and photographs of riparian areas in

Glen Canyon during the same period show the same species at and below the high water mark (Flowers 1959, Lindsay 1959, McDougall 1959a, b, Gaines 1960). The Glen Canyon accounts (Flowers 1959, Lindsay 1959) also describe river's edge herbaceous elements, including species of Equisetum, *Juncus*, *Panicum* and *Carex*. Both Patraw (1936) in Grand Canyon and Gaines (1960) in Glen Canyon collected other wetland species, including *Typha*, *Phragmites*, *Plantago*, *Eleocharis*, *Epipactis*, and *Allium*, although locations for the former in the river corridor are not



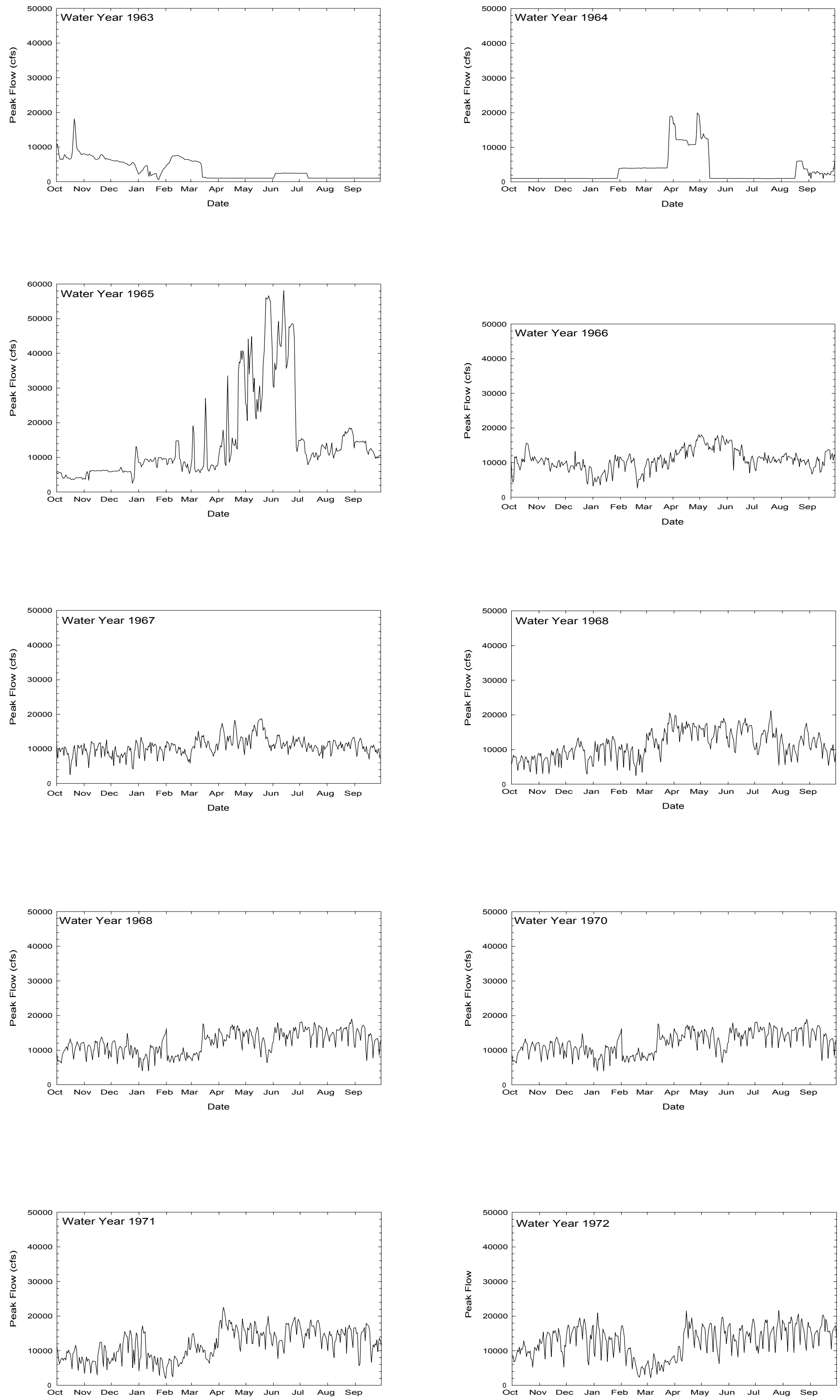
1. Hydrographs from six water years (Oct. 1 - Sept. 30) during the pre-dam era representing high, intermediate, and low flow years

certain. Stands of riparian vegetation generally were between 3 and 20 meters wide, but often expanded to dense stand more than 50 meters wide at tributary mouths (Flowers 1959). Mosses occurred in damp sand and clay throughout all near-shore vegetation types, and created "dense coatings" on intermittently submerged sandstone surfaces, especially where cliffs created shady conditions (Flowers 1959).

It is not the contention here that pre-dam vegetation was a continuous verdant belt extending down to the water's edge at 300 cms. It is clear from the original Brown / Stanton photographs used in Grand Canyon rephotography studies (Turner and Karpiscak 1980, Webb 1996) and the pre-dam Glen Canyon photographs in Woodbury (1959) that vegetation in the zone between 500 and 2500 cms was usually sparse and that the "moist sand" vegetation described by Clover and Jotter (1944) was spotty and discontinuous. However, the species which comprise the bulk of the low riparian zone vegetation were present before the filling of Lake Powell began, and it is likely that during periods of several consecutive years where flows did not exceed 1500 cms (1938 - 1940, 1958 - 1961) that vegetation density increased to moderate levels.

<u>Colonization era I: (1963 - 1973)</u>. With the completion of Glen Canyon Dam in 1963, the hydrograph of the Colorado River in Grand Canyon changed dramatically (Dolan et al. 1974, Carothers et al. 1979, Carothers and Dolan 1982). In addition to limitation of the annual peak flow to roughly 900 cms, daily fluctuations in discharge related to peaking electrical power demands created "tides" of up to several meters in vertical variation each day (Dolan et al. 1974). These "load following" flows would often result in fluctuations between 140 cms and 900 cms. High and low flow levels varied seasonally, with the highest levels in the summer and progressively lower flows in the winter, spring, and fall. Exceptions to this general rule came in 1963 and 1965, when political and administrative incidents led to an unusual flows; 1993 had extremely low peak releases (ca. 200 cms) because the Upper Basin States were attempting to fill Lake Powell and 1965 had unusually high flows in the fall (ca. 1850 cms) related to "equalization" deliveries of water to the Lower Basin States (M. Yard, GCMRC personal communication.

The initial effects of stabilizing the flows in Grand Canyon on the riparian zone were well documented by a series of studies and descriptions of the river corridor vegetation (Carothers and Aitchison 1976, Theroux 1976, Johnson and Carothers 1982a). Reduction of the peak annual flows from 3500 cms to 1250 cms exposed habitats which had previously been regularly flooded by high spring flows. This allowed the expansion of perennial species from above the 3500 cms elevation and protected areas below it into open habitats above the 1250 cms elevation. These new riparian habitats were referred to as the "new high water zone," to distinguish them from the high, pre-dam riparian habitats which were termed the "old high water zone." Evidence from repeat photographic studies (Karpiscak 1976, Turner and Karpiscak 1980), multitemporal remote sensing studies (Pucherelli 1986, Waring 1996) and general surveys (Carothers and Aitchison 1976) clearly showed vegetation spreading into the new high water zone. A vegetation polygon map (Phillips, B.G. et al. 1977), generated from 1973 aerial photographs, shows well-developed shoreline vegetation with abundant woody perennial species. From this data, Ohmart (Ohmart 1982) estimated a total riparian vegetation coverage of approximately 1700 ha of which 1100 ha was new high water zone vegetation. Much of the colonization was by exotic species, especially tamarisk, Bermuda grass and camelthorn (Martin 1971). Ohmart's (1982) estimates put approximately 85% (910 ha) of the new high water zone vegetation in the categories of tamarisk and sparse tamarisk. The rapidity with which colonization and succession occurred caused some



2. Hydrographs of water years in the initial colonization era.

to wonder whether this new riparian habitat would collapse on itself due to a lack of nutrients because the sediment-laden spring flows no longer replenished the riparian soils (Carothers and Dolan 1982).

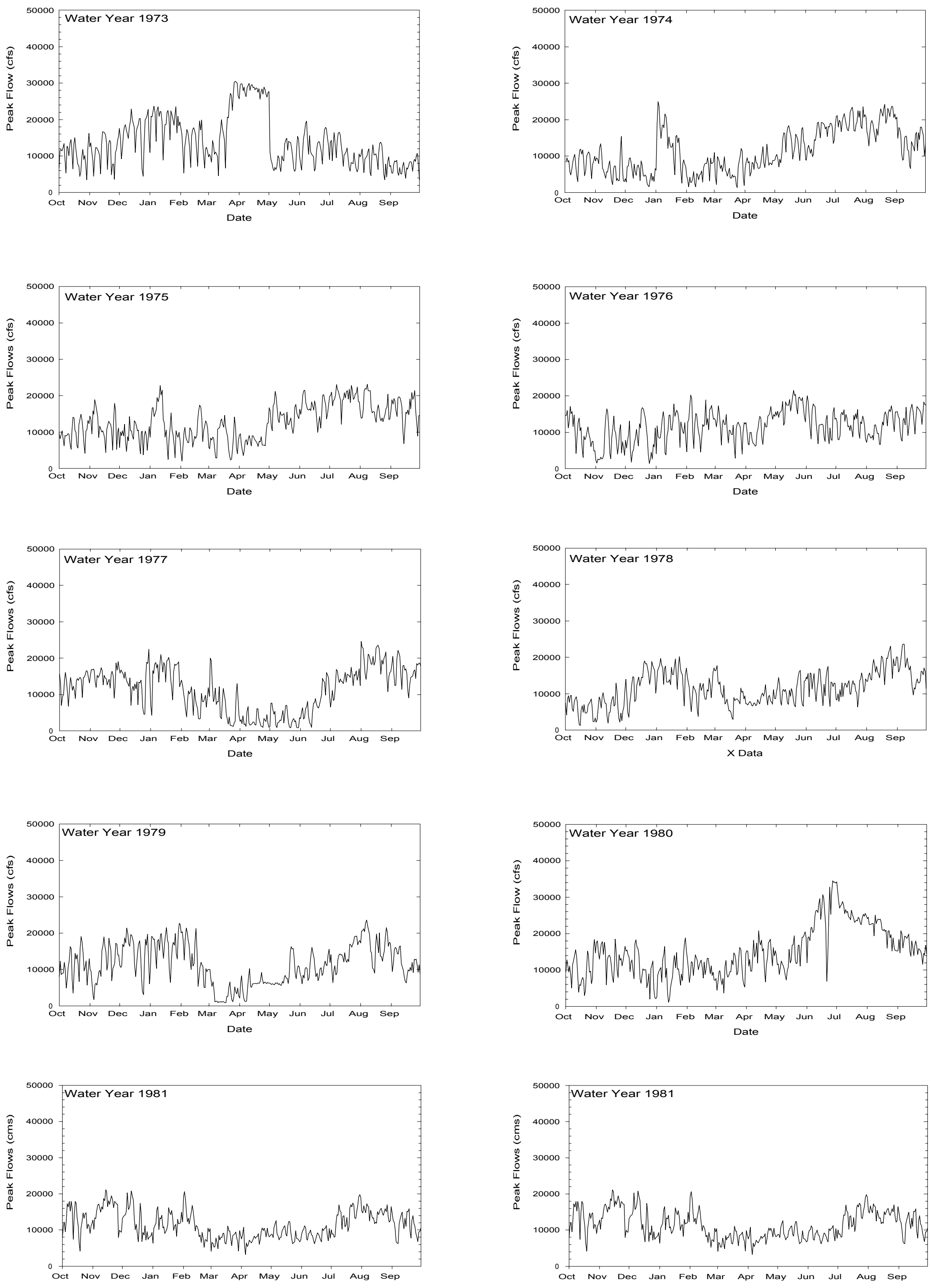
The new load-following flow regime, in which discharge was driven by daily peaks in regional electrical power demand, was producing relatively stable eddy return-current channel habitats (Schmidt and Graf 1988). These low-elevation areas, scoured by the upstream movement of water at eddy margins during periods of higher flows, became still backwaters at lower flows. Nutrient-rich silts and clays from tributary flash-floods were deposited in these slack-water areas (Rubin et al. 1990). In these wet areas with rich soils, wetland patches developed (Stevens et al. 1995). A close examination of this map produced by Phillips et al (1977) shows 52 wetland patches of various sizes and compositions in the river corridor by 1973 (Stevens et al. 1995). Although there were no scales indicated on the maps, and the original photographs are no longer available, Stevens et al. (1995; Table 3) somehow estimated the total are of these wetland patches to be 5 ha in total.

In the habitats between the new high water line and the old high water line (ca. 1500 - 3000 cms) little colonization took place in the first ten years following the completion of Glen Canyon Dam. This area had few plants in it before flow stabilization due to scouring, and was too high above the new water level to support germination and establishment of new individuals. Photographs taken during the late 1960s and early 1970s show expanses of bare sand in this zone (Aitchison et al. 1976, Karpiscak 1976, Turner and Karpiscak 1980). Diagrams depicting zonation of vegetation in that era show little other than scattered annuals in this zone (Carothers et al. 1979, Johnson and Carothers 1982b).

<u>Colonization era II: (1973 - 1983)</u>. With the exception of a single 10-day high flow (1275 cms) in 1980, the hydrograph in the second decade of dam operations was very similar to that in the first. Load-following discharges from Glen Canyon Dam varied widely within a given day, and flow ranges varied seasonally depending on power demands.

Vegetation which had established during the first decade continued to thrive. New high water line plants, especially tamarisk, camelthorn, and Bermuda grass appear in photographs taken during monitoring of vegetation and campsites (Aitchison et al. 1976, Carothers and Aitchison 1976). Patches of marsh vegetation in eddy return-current channels and in protected channel margin habitats also continued to develop. Inventories of wetland patches in 1980 and 1982 showed roughly equivalent numbers and total areas (Stevens et al. 1995) when compared to the 1973 maps of Phillips et al (1977).

The major change which took place during this period was the establishment and spread of native clonal woody species in the new high water zone, especially coyote willow *(Salix exiguua)* and arrow weed (*Pluchea sericia*). Both Brian (Brian 1982) and A.M. Phillips (personal communications cited in Brian 1982) described the infiltration of new high water zone habitats by coyote willow long after the exotic species had become well established. Two campsite dynamics studies (Green et al. 1980, Phillips, B.G. et al. 1986) encountered both species in sandy areas commonly used for camping. The major change between 1980 and 1982 photographs of campsites was the increase in stem densities and canopy coverage of these species (Phillips, B.G. et al. 1986). Brian (1982) attributed the shift to a coarsening of the new high water zone substrates which resulted from the loss of silts and clays which were being trapped behind Glen Canyon Dam. The fine lateral roots of coyote willow seedlings acted as anchors by wrapping around the coarse sand grains and preventing seedlings from being dislodged by waves and



3. Hydrographs of water years during the second part of the colonization era.

fluctuations (Brian 1982). Without lateral roots, tamarisk seedlings, though far outnumbering willows, could not become established in the coarser sediments. Germination trials in which the soil texture and inundation frequency were varied confirmed this general pattern (Stevens 1989).

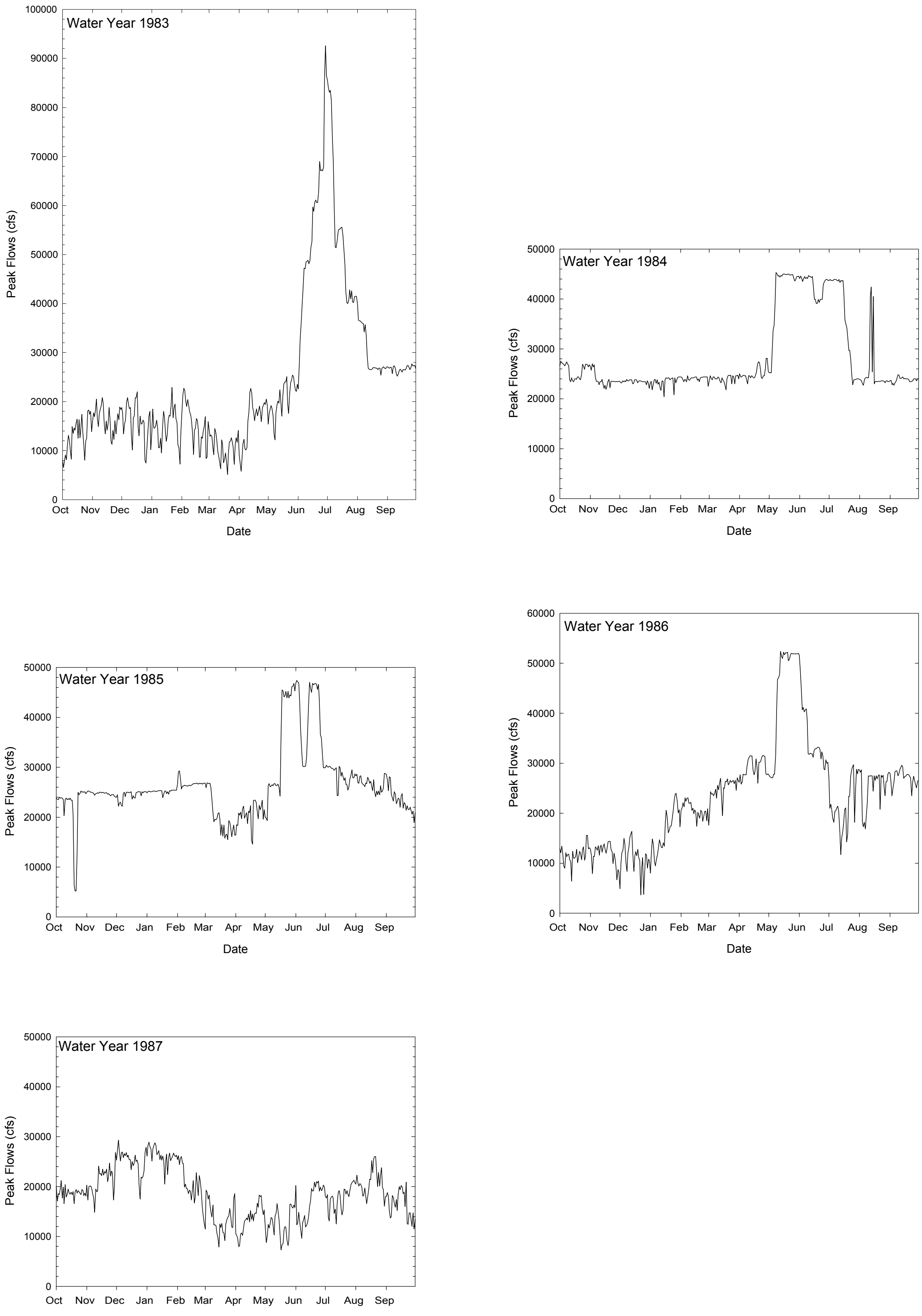
High flows era: (1983 - 1987). Because the filling of Lake Powell occurred in the first of four consecutive extremely wet years, the period 1983 - 1987 was characterized by sustained high flows with occasional extremely high flows from Glen Canyon Dam (Bureau of Reclamation 1996 p. 91). In 1983, a unexpectedly rapid melt and runoff from a series of large late-winter storms caused a rapid rise in Lake Powell and forced dam operators to release water at more than 1270 cms for extended periods and at more than 2650 cms for several days. High inflows to a nearly full Lake Powell in the three following years led to releases above 1250 cms through much of those summers. Flows in 1987 were also relatively high for the post-dam era, but never exceeded powerplant capacity (892 cms).

The effects on new high water zone vegetation growing in areas above the 700 cms stage elevation which had been relatively undisturbed for the previous 18 years were striking. A comparison of aerial photographs of seven sites in 1980 and 1985 showed a loss of 35% of the woody riparian vegetation and nearly 90% of the beach zone vegetation (Brian 1987). More than 55% of all vegetation below the 1200 cms stage elevation were removed and almost half of all vegetation below the 2600 cms stage elevation present in 1980 was gone in 1985 (Brian 1987). A similar study with a slightly broader scope (Pucherelli 1986) which examined vegetation losses over the same period across eight 3 - 10 km reaches found that approximately 40% of new high water zone vegetation acreage, measured as the product of patch area and patch density, was lost during the first three years of these high flows (Figure X). By the end of 1985, only 20% of shoreline habitats at 800 cms supported woody plants (Anderson et al. 1986). Of that, most was concentrated between the dam and Lees Ferry and between National Canyon and Diamond Creek (Anderson et al. 1986) where 35% of the shoreline at 800 cms was vegetated.

In areas where some plants survived, species responses to the high flows differed . For example, willows and arrowweed were completely scoured from beaches in eddy complexes, but because they were not removed by drowning or scouring in other settings, they did not change significantly in repeated transects in a series of geomorphic settings (Stevens and Waring 1985). However, tamarisk and baccharis were both scoured and drowned and so densities of both were significantly reduced (Stevens and Waring 1985).

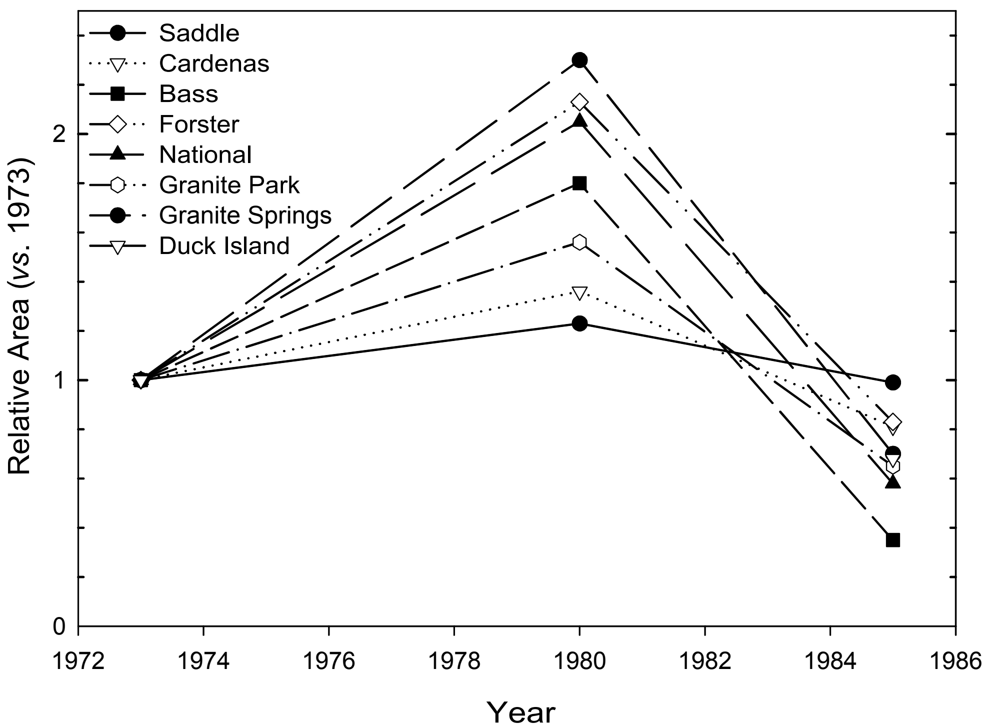
The germination and early growth of most species was stimulated by the high flows. With the exception of the clonal wetland species (*Typha*, *Phragmites*, and *Scirpus*), new high water zone species, especially tamarisk, increased after the flood (Stevens and Waring 1985). With the removal of tamarisk from silt and sand terraces and the increase in germination and establishment in cobble bars, their distribution seemed to be shifting as a result of the flows. Old high water line plants, especially mesquite, germinated and grew vigorously during the high flow years, offsetting some of the losses of saplings and seedlings in hard freezes in 1978 and 1984 (Anderson and Ruffner 1987).

Herbaceous communities in marshes lost most of their vegetation too. Essentially all vegetation in the lowest elevation habitats was lost in a comparison of 1980 and 1985 aerial photographs of 7 sites (Brian 1987). A whole-corridor inventory of marshes from 1980 and 1984 aerial photographs showed a loss of 65% of marsh patches and a reduction of 86% in total marsh area (Stevens et al. 1995). Only 2% of the shoreline at 140 cms was vegetated by the end of 1985, the third consecutive year of sustained high flows (Anderson et al. 1986). These areas



4. Hydrographs of water years during the high flows era.

were not recolonized until 1987 and afterwards due to the near complete removal of important herbaceous clonal wetland species and sustained high flows in 1996 (Waring and Stevens 1986, Stevens et al. 1995).

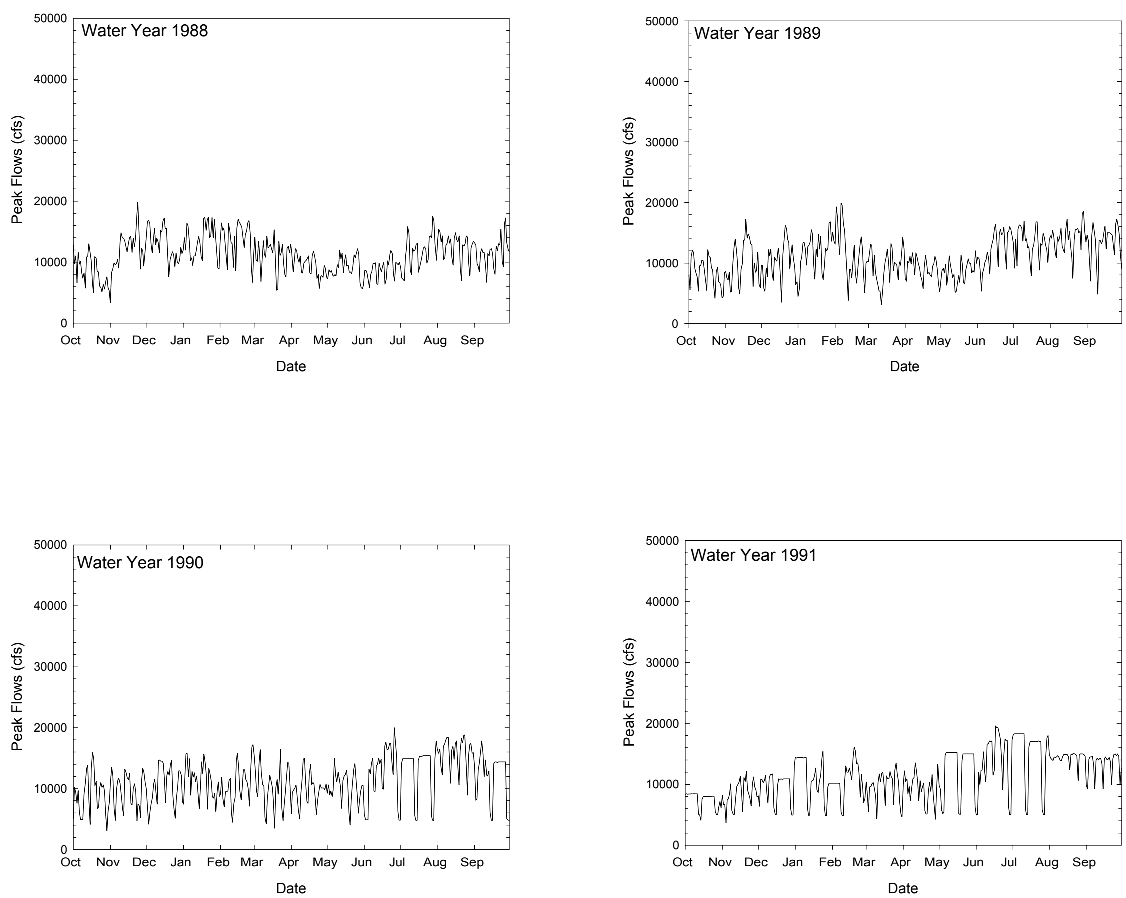


5. Growth of new high water zone vegetation at eight sites before the high flows of the early 1980s, and vegetation losses resulting from scour and inundation.

Recovery era: (1987 - 1991). Flows during the next five years were similar to those during the initial colonization periods. Load-following flows with no discharges above powerplant capacity were the norm. Almost no releases were above 700 cms, although power generation-related fluctuations in releases often ranged from 50 - 700 cms in a single day. In water year 1990 and early water year 1991, a series of test flows were conducted to quantify the effects of several different types of dam operations on sediments in the river corridor. These discharges included different flow extremes and ramping rates, including steady low and high flows, moderate fluctuations, and low fluctuations.

During this period, vegetation returned to habitats from which it had been removed or buried during the high flow period. By 1990, there were more than 15 times as many wetland patches in the river corridor as there had been in 1984 after the highest and longest sustained high flow periods in 1983 and 1984. Likewise, the total area of wetland patches in 1990 was more than 16 times that in 1984. By the end of 1991, the number and area of wetland patches had increased even farther. The number of patches had increased by more than 25% in that year, and the total area had more than doubled in that same year (Stevens et al. 1995).

At the same time, non-wetland vegetation increased as well (Waring 1996). In three sections of Glen and Marble Canyons, the total area of vegetation below the old high water line was 7 -

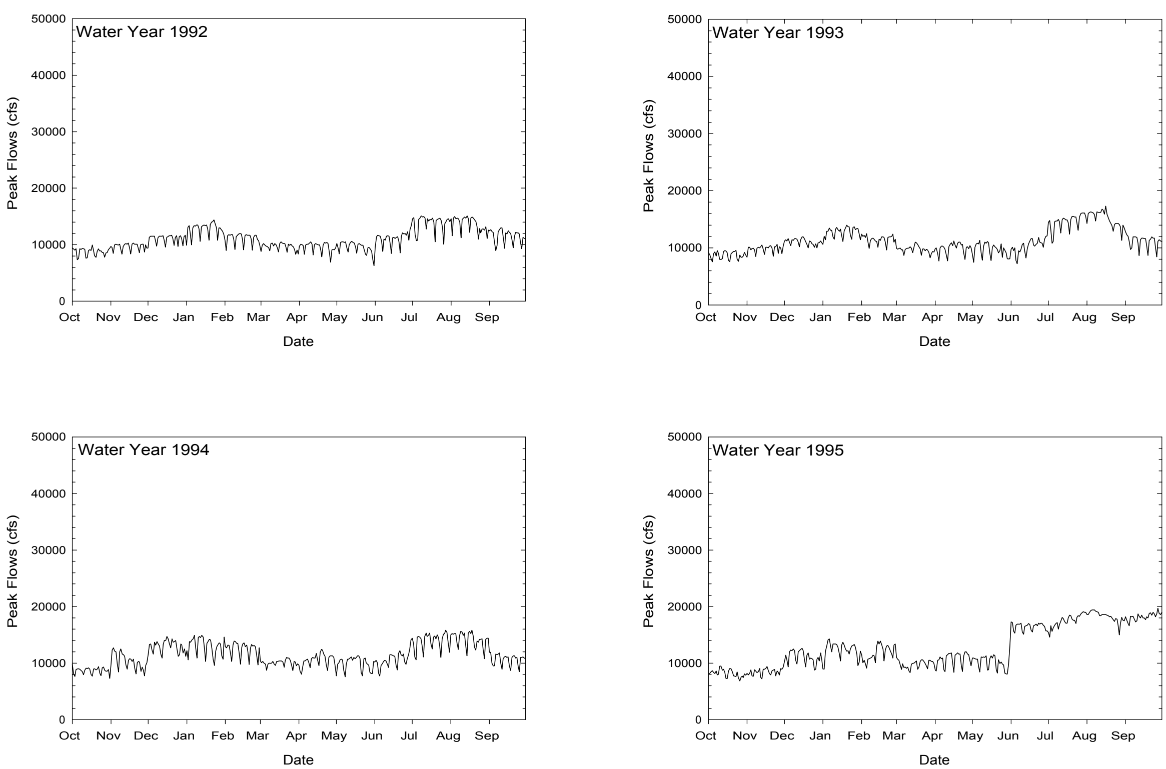


6. Hydrographs from water years during the post-high flows recovery era.

10% higher than it had been in 1984. However, it was still between 5 and 10% below its levels in 1973, indicating that the overall effects of the high flows on new high water zone vegetation had not been negated by this relatively benign periods of flows.

Interim flows era (1992 - 1995). As a result of the Grand Canyon Protection Act, a new pattern of discharges from Glen Canyon Dam were begun on August 1, 1991. Termed the interim flows operating criteria, these flows were designed to conserve sediment resources by reducing the mobilization of sand in sandbar and channel deposits. These flows included moderated non-emergency bounds on upper and lower flows and reduced the possible up- and down ramping rates (Bureau of Reclamation 1995). The specific operating criteria are outlined in Table Z.

The reduced range of fluctuations between 1992 and 1994 had an impact of the distribution of new high water zone vegetation in two ways. First, riparian vegetation moved into habitats exposed by the reduction of maximum discharges of roughly 100 cms (450 versus 540 cms). Examinations of aerial photographs showed an increase of roughly 30% in the total area of new high water zone vegetation in the 37 km of river corridor studied by Waring (1996).



7. Hydrographs from water years during the interim flows operating criteria era.

Table 2. Comparison of interim flows operating criteria to post-dam flows which preceded them. Figures are in cubic meters per second.

	Post-dam norm		Interim flows criteria	
Minimum flow	28	(Sept. to Apr.)	142	(night)
	85	(Apr. to Sept.)	227	(7 a.m. - 8 p.m.)
Maximum flow	892		566	
Change per day	863	(Sept. to Apr.)	142	(low vol. months)
	807	(Apr. to Sept.)	170	(med vol. months)
			227	(high vol. months)
Ramping rate (cms / hr)	No restrictions		71	

Second, the distribution of some species changed as well. The drawdown of groundwater associated with reductions in river stage (Carpenter et al. 1994) caused a shift towards more upland species in nearly all habitats. Vegetation in debris fan quadrats did not change significantly, but that in channel margin, bar-top, and water's edge patches all showed an increase in upland and facultative upland species and a loss of moisture-loving species (Kearsley and Ayers 1996a). In belt transects across return-current channel marshes, higher elevation transects showed much more of this change than the low-elevation transects in the same sites (Kearsley and Ayers 1996a). That these changes were a function of reduced flows was reinforced by the fact that high elevation transects in fall marsh surveys, which followed summer flows of roughly 450 cms were less affected than they were in spring surveys, which followed flows of roughly 340 cms (Kearsley and Ayers 1996a). In addition, plant water potential measurements of willow and tamarisk in two sites showed that river stage accounted for a small but significant portion of the total variability in moisture stress (Stevens and Ayers 1993). Willows growing at higher elevation in these sites had reduced annual stem growth as well (Stevens and Ayers 1993).

Vegetation studies conducted in the lower Grand Canyon (Rkm 268 - Grand Wash Cliffs) showed slightly different patterns (SWCA Inc. 1993, SWCA 1994, 1995). In general there were no overall patterns of change detected in the upland / wetland plant ratios in quadrats and transects, perhaps the result of half of the sites sampled being on Lake Powell. Most of the effects of these reduced flows on the lake plots (below Rkm 378) were the result of erosion removing large sections of shoreline rather than the gradual drying of soil and drops in the water table (SWCA 1995). A drop in the level of the river caused the sandbars to become unstable and erode through either seepage processes or by calving off catastrophically (Budhu and Gobin 1994).

During this period, Stevens and Ayers (1993) described several important relationships between new high water zone vegetation and environmental variation. First, inundation frequency (or its proxy, elevation) and geomorphic setting were the primary determinants of vegetation composition and productivity in the river corridor. Low elevation areas in return-current channels trapped fine sediments during periods of reduced flows and created habitats which fostered the development of wetland vegetation. Channel margin habitats, where sediments were slightly coarser but access to the water table was equivalent to marsh settings, supported fairly dense stands of phreatophytes. In the higher elevation habitats on bar tops, sediments were similar to channel margin areas, but the water table was further below the surface and sparser, less diverse communities developed. Debris fan habitats, which had coarser substrates and less access to ground water, were particularly sparsely vegetated, but supported a very species-rich flora.

Soil characteristics differed in these areas as well (Stevens and Ayers 1993). Marsh areas had the highest soil nitrogen and phosphorus and the finest particles sizes, reflecting the periodic inputs from sediment-laden waters during tributary flash floods. Channel margin habitats had the next ighest levels, and bar top and debris fan settings had the lowest. These numbers coincided with the relative frequency of inundation of these plots, a 15 year echo from earlier researchers who had expressed concern that a lack of flood inundation and consequent fertilization of the soils would lead to a decline in riparian productivity (Carothers and Dolan 1982).

Although habitat variables were shown to affect plant growth and species composition in these sites, no reciprocal effects were found. For example, the density of neither above- nor below ground plant tissues had an effect on sandbar stability (Stevens and Ayers 1993). Two factors could explain this unexpected result. First, the unconsolidated riparian soils in Grand Canyon consist mostly of sand and coarse sand which are very difficult to stabilize. These studies were conducted on transects through sites chosen by surveyors as representative of normal, hydraulically created sandbars (Beus et al. 1992). Second, the range of above-ground stem densities (0.5 - 5.0 / m^2) and root densities (1.1 - 2.0 / m^2) may have been lower than the threshold at which stabilization will occur.

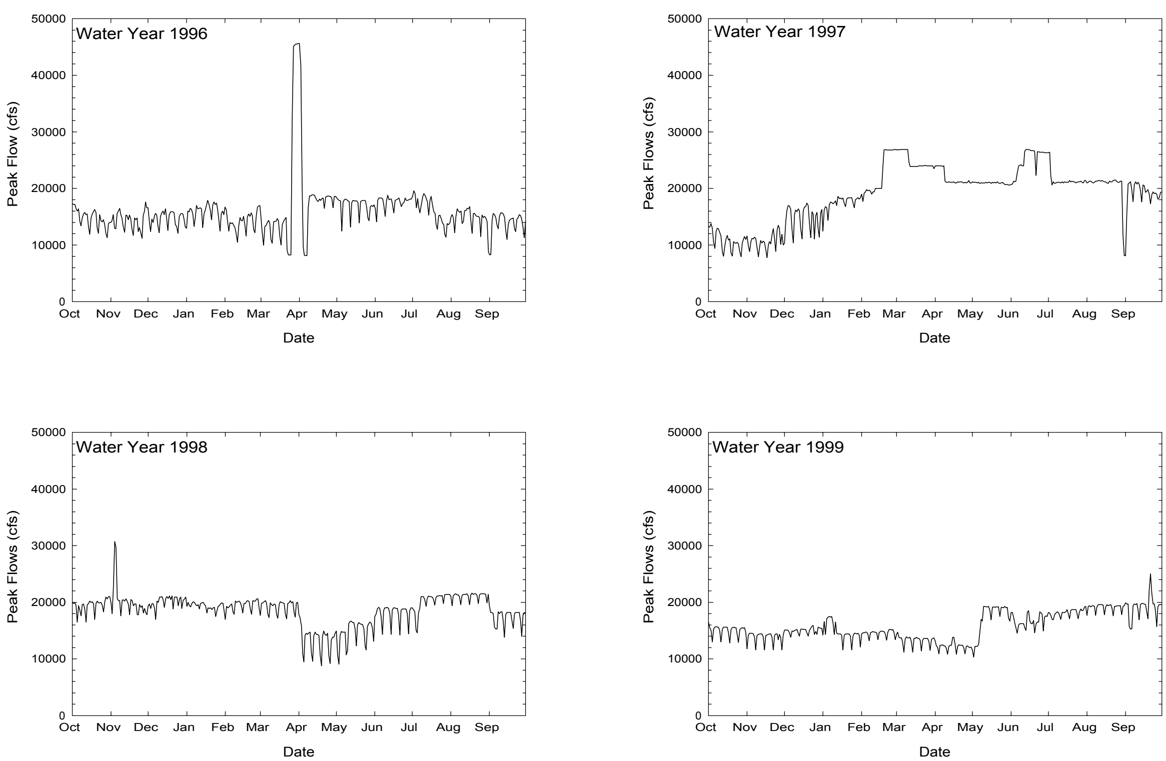
It was also during this period that the first detailed consideration of the impacts of exotic species in the river corridor was considered. Although numerous authors had previously described the presence and impacts of tamarisk and camelthorn, it was not until detailed studies of vegetation began that other species, notably ravenna grass (*Saccharum ravennae* (L.) L.) and broadleaved pepperweed (*Lepidium latifolium* L.) drew attention (Stevens and Ayers 1993, Kearsley and Ayers 1996a). Tamarisk and camelthorn were described as “naturalized” and “manageable”, respectively (Stevens and Ayers 1993). However, ravenna grass and pepperweed were both branded potential problems deserving immediate attention before they achieved their potential as noxious weeds in the river corridor.

After their identification as problem plants, neither spread very widely, but each for a different reason. By 1994, ravenna grass populations in the river corridor were brought under control by crews who dug up and disposed of adult plants wherever they were found (pers. obs.). New sightings were reported to NPS trail- and campsite maintenance crews who removed them as part of annual river trip work. Pepperweed, on the other hand, did not spread in a detectable way either, but it was not removed from the system either. Whether for reasons related to small population sizes, patchy distribution, or limited sampling programs, no change in the distribution or abundance of this species was detected (Kearsley and Ayers 1996a).

<u>Adaptive management era (1996 - 1999)</u>. The hydrograph in this period was driven by several active management decisions. The non-emergency limits on the hydrograph were similar to those during the interim flows period, although maximum flows were increased to 680 cms and upramping rates were increased to 110 cms/hr. Three significant events occurred in 1996

and 1997: the controlled flood of 1996, the steady high flows of 1997, and the 1997 habitat maintenance flows. In the first, a 10-day steady release of 1275 cms was bracketed by three day periods of 225 cms steady flows. The purpose of the high flows was to test whether sediments stored in the channel could be transferred and stored in high-elevation deposits (i.e., sandbars or beaches) by high releases. The low bracketing flows were for the purpose of obtaining aerial photographs at a consistent discharge. The high flows of 1997 resulted from high runoff into a nearly full Lake Powell. These flows included steady 765 cms releases in February and March and again in June. During the remainder of the growing season (March - June, July - September), releases were nearly constant at around 565 cms. A November, 1997 three-day flow of 880 cms came during an attempt to move sediment which had been deposited in the main channel during four periods of unusually high inflows from the Paria River in August and September. Three-day low flows of 227 cms in September of 1996 and 1997 were for the purpose of taking annual aerial photographs of the river corridor. In 1997 and 1998, photographs were taken during three-day steady flows of 425 cms which did not have a noticeable effect on the hydrograph.

During this period of variable flows, there was little opportunity for vegetation trends to develop because there were no consistencies in the hydrograph of consecutive years for vegetation to adjust to. The impacts of individual events on vegetation were idiosyncratic rather



8. Hydrographs of water years during the adaptive management era.

than summing to a overall trend.

The controlled flood of 1996 buried most sites under 0.5 - 2.0 m (avg. = 0.64m) of sand scoured from the channel bottom (Kearsley and Ayers 1998, Hazel et al. 1999). The immediate impacts on riparian plant communities were the burial of the herbaceous layer (grasses and forbs) and the loss of significant portions of the soil seed bank (Kearsley and Ayers 1996b, 1998). Transects in lower canyon sites all lost herbaceous vegetative cover (Phillips, A.M.,III and Jackson 1996). Backwater areas in return current channels, which had supported wetland vegetation were buried under up to 2 m of soil rather than being scoured clear of vegetation and sediment (Kearsley and Ayers 1996b, Parnell et al. 1999).

Longer-term effects of the flood were far less dramatic. Most of the woody perennial species were unaffected by the flood (Kearsley and Ayers 1996b). Six months after the flood, there was no detectable change in the areal extent of any of the major vegetation patch types (Kearsley and Ayers 1996b). Regrowth of willow and horsetail during the high flows in the summer of 1996 resulted in an increase in cover and density of both species in line-intercept transects (Austin et al. 1996), and a culturally significant Goddings willow showed good growth as well (Phillips, A.M.,III and Jackson 1997). In the two downstream-most mapping sites, there was a flush of Bermuda grass, camelthorn, and willow in response to the elevated summer flows which resulted in a finding of an increase in vegetation density in the lowest two meters of the canopy (Kearsley and Ayers 1996b). Six months after the high flow, the overall reduction in cover was on the scale of 20%, and surviving vegetation grew in enough so that wetland patches in 1995 were classified again as wetland based solely on the surface vegetation in them (Kearsley and Ayers 1996b). New sand in the upper elevation areas of the new high water zone was repeatedly reworked by wind, preventing most plant establishment (Phillips, A.M.,III and Jackson 1996).

The high flows in 1997 had mixed effects on riparian vegetation. The total areal extent of wetland and riparian scrubland patches did not change significantly during the year, even though higher water levels had inundated parts of these patches (Kearsley and Ayers 1999a). Within these patches, structural density increased, especially with increases in the shrub layer (both willow and tamarisk), although no detectable changes were seen in the total foliar cover of these species (Kearsley and Ayers 1999a). There were increases in foliar cover in lower canyon line-intercept transects, dur mostly to flushes of Bermuda grass and camelthorn (Austin et al. 1997, Phillips, A.M.,III and Jackson 1997). Native species, notably horsetail and willow, increased on these same transects as well, but only in areas above the highest water mark (Austin et al. 1997, Phillips, A.M.,III and Jackson 1997). A culturally significant Goodding's willow did not show any negative effects of the high flows in either 1996 or 1997, although there was some indication of roots being undermined (Phillips, A.M.,III and Jackson 1997). Composition of patches and transects, measured by species richness and diversity, did not change appreciably (Austin et al. 1997, Phillips, A.M.,III and Jackson 1997, Kearsley and Ayers 1999a), although there was a great deal of species turnover among rare species in patches (Kearsley and Ayers 1999a).

Although Bermuda grass and camelthorn grew well in place where they had occurred prior to the high flows (Austin et al. 1997, Phillips, A.M.,III and Jackson 1997), these flows did not spread these exotics. There was no increase in the between-site distributions in either vegetation or soil seed banks resulting from the high flows for either of these species (Kearsley and Ayers 1998, 1999a). Nor was there a change in the distribution or abundance of three other potential problem exotics: ravenna grass, pepperweed, and weeping lovegrass (*Eragrostis curvula* (Schad.) Nees) (Kearsley and Ayers 1998, 1999a).

The three-day spike flow in 1997 and subsequent moderately high flows in water years 1998 and 1999 (450 - 570 cms) did not have a dramatic effect on sediments (Kaplinski et al. 1997). Nor did it have much of an effect on vegetation in vegetation monitoring sites. Plants at the water's edge continued to grow well, but the reduction in maximum flows in 1999 caused some mortality of horsetail populations (Austin et al. 1998). Methodological problems prevented the finding of any significant changes in foliar cover of vegetation mapping sites, but measures of mapping density showed increases in the high water year of 1998 and no significant change after water levels dropped in 1999 (Kearsley and Ayers 1999b). There were no significant changes in the areal extent of the major patch types (wetland, scrubland, bar top), nor of their diversity or species richness.

Because there have been no reports produced on vegetation monitoring during water year 2000, this concludes the narrative account of vegetation changes. Some vegetation work was conducted during this year, but it will be described elsewhere in this document.

METHODS

Power Analysis

In order to determine the ability of monitoring methods to detect year-to-year change in measurement levels within polygons, a series of power analyses were conducted. For vegetation attributes, including foliar cover, species richness, Shannon diversity, and structure (TVV), we first generated within-site means from 1998 and 1999 data for each of four broad vegetation types: wetland, mixed riparian scrub, tamarisk scrub, and bar-top. Site means were calculated as the average difference between 1998 and 1999 values for each polygon of a type in the site. To avoid problems arising from differences among within-site variance, sites which had only one polygon of a given type were excluded from the analysis of that type, as were sites which had none in one year or both. The mean and standard deviation of differences between 1998 and 1999 site means were calculated and, together with the number of sites, were used to estimate the power of a one-sample t-test with a null hypothesis mean of zero. The probabilities of type I and II errors were set at 0.05 and 0.20, respectively, and a standard power equation of (equation 7.8; Zar 1999 p. 107), was used to estimates minimum number of sites required for concluding that the 1998 - 1999 changes were significant. The minimum sample size required for the following levels of change were determined as well: loss of gain of three TVV hits, loss or gain of 10% cover, loss or gain of one and two species, and a change (+ or -) of 0.20 in Shannon H' diversity. The minimum detectable percentage changes for each measure from 1998 levels were also determined using another standard equation (equation 7.9; Zar 1999).

A second set of power analyses were conducted to determine the ability of our methods to detect changes in the areal extent of the four major vegetation types from one year to the next. The total area of each type in each of the 10 upstream sites (no site or patch area data were available for the site at 249L) was calculated for 1998 and 1999. Sites which lacked a type of vegetation were excluded from the analysis of that type. The mean and standard deviation of changes and the number of sites used for each type were then calculated. A standard change of 10% from 1998 levels, a 0.05 probability of a type I error, and the calculated standard deviation were used to generate power curves for wetland, mixed riparian scrub, tamarisk scrub, new high water zone scrub (combined mixed riparian and tamarisk scrub polygons), and bar-top change detection. The point on the each curve at which the calculated power was 0.80 (beta = 0.20) was used to determine the minimum number of sites required.

The final power analyses were done for the vegetated shoreline data. Because the number of geomorphic reaches is fixed and because sampling at the rate of 20 samples per reach provided a good estimate of the variance (Kearsley and Ayers 1999b), a power curve based on variable sample sizes or standard deviations did not make sense. For each geomorphic reach, means for total length, total area, and number of vegetated patches per 50 meter sample were calculated from 1998 and 1999 data. Means and standard deviations of changes in reach means were then calculated, and an estimate was made of the magnitude of change from 1998 to 1999 for each measure. A power calculation package (Bface; Lenth 1996) was used to determine the power of a one-sample t-test (a paired t-test) to detect a range of magnitudes of change which seemed reasonable, basedon the 1998 / 1999 change. For example, between 1998 and 1999, the length of vegetated shoreline per 50 meters increased, on average, 33%. We therefore calculated power for changes of 25%, 35%, 45% and 55%. Although the average change of vegetated area and number of patches of vegetation per 50 meters was much greater over the same period (87% and 85%, respectively) we calculated power over the same interval as for the length data (25% - 55%) to keep things conservative.

Sampling site representation

In order to determine whether the vegetation in sampling sites visited during the past several years is representative of vegetation in the river corridor generally, the relative abundances in the sites were compared to random samples of vegetation from Lees Ferry to Diamond Creek. A classification employed by Sogge and colleagues (Table 3; Sogge et al. 1998) for bird habitat data was used as the basis for the comparison. First, the *ad-hoc* monitoring site polygon classifications were cross-walked to the Sogge classification. Because the original list of patch types was intended to cover only the sampling sites from the 1998 paper (M. Sogge, pers. comm.), several new classes had to be created. The final list of types and their descriptions are listed in Table 3. Data on polygon area from digitized and rectified 1999 vegetation maps of the sampling sites from Kearsley and Ayers (1999b) was combined with the classification data and pooled across all sites to produce a total area per vegetation class data set.

To determine the relative abundance of these same types of vegetation in the river corridor, we sampled 30 points randomly located in each of the 11 geomorphic reaches of Schmidt and Graf (1988). For each reach we generated 30 three-digit random numbers between zero and one and 30 random integers between 1 and 10 using a random number generator from a spreadsheet software package (Lotus 1-2-3 release 9 for Windows). The three digit numbers were multiplied by the number of river miles in a reach and the products were added to the beginning mileage of the reach to determine the location (to the nearest 0.1 mile) at which the samples were to be taken. The integers were used to determine the side on which the samples were to be taken (odd = left, even = right, looking downstream). All points were then located on a 1990 set of color aerial photographs on which the U.S. Fish and Wildlife Service had marked river mileages, in 0.1 mile increments, from Lees Ferry to Diamond Creek. Using landmarks from the color aerials, the locations were transferred to a set of 400% xerox enlargements of the black and white 1:4800 aerial photographs taken in September 1998 which were to be taken into the field. The approximate locations were also marked in a river guide (Belknap and Evans 1989) to facilitate locating the points in the field.

Table 3. New high water zone vegetation types modified from Sogge et al. (1998) used in the comparisons of vegetation sampling sites and random locations.	
Symbol	Characteristics
a	Monotypic ARROWWEED on beach / sandy sites
ab	Mixture of ARROWWEED / BACCHARIS with no obvious dominant. Linear patches along river
abt	ARROWWEED, BACCHARIS, and TAMARISK with no obvious dominant. Linear patches.
ag	ARROWWEED - GRASS patches; Linear patches where % cover is dominated by grass
at	ARROWWEED - TAMARISK mixture in linear patches along river with no consistent dominant.
aw	ARROWWEED - WILLOW mixture in moist sites, linear near river.
ax	ARROWWEED - DESERT BROOM strip at top of new high water zone, dominated by arrowweed.
b	BACCHARIS, either *B. emoryi* or *B. salicifolia* in monotypic stands in deep, moist soil.
bm	BACCHARIS - MESQUITE mixture in large patches, Baccharis usually dominant.
bmt	BACCHARIS - MESQUITE - TAMARISK mixtures with a few large mesquite in large patches.
bt	BACCHARIS - TAMARISK mixture with no consistent dominant along river in moist soils.
btw	BACCHARIS - TAMARISK - WILLOW in even mixtures in linear patches along river.
d	SAND / ROCK / CLIFF / TALUS with no vegetation (not included in analyses)
e	EQUISETUM / SEDGE wetland patches with no woody vegetation.
f	DEBRIS FAN, BOULDERS, ROCKY AREAS with little or no vegetation (not included in analyses)
g	GRASS patches, usually monotypic *Bromus* or *Cynodon* stands.
m	MESQUITE as monotypic or dominant with grass associations in drier sites.
mt	MESQUITE - TAMARISK mixtures in large patches, usually with tamarisk dominant.
p	REED / CATTAIL marsh patches dominated by *Phragmites*, *Typha*, or other emergents.
t	TAMARISK in large, monotypic stands
tw	TAMARISK - WILLOW mixtures in moist soils along river, with no consistent or obvious dominant.
tx	TAMARISK - DESERT BROOM mixtures at top of new high water zone, no consistent dominant.
w	WILLOW stands in moister sites, not usually linear patches.

In the field, the amounts of all types of new high water zone vegetation were recorded. Cues from the river guide and landmarks on the 400% enlargements were used to determine the location of the sample points. At each sample point, a line was drawn perpendicular to the river from the water's edge through the new high water zone vegetation up to the 40 kcfs stage elevation. Along this line, the extent and identity of each vegetation type (from the augmented

Sogge classification) was marked on the enlargement.

To determine the amounts of all types of vegetation encountered in this sampling, the sample lines were recreated with ARC View on 1998 digital color infra-red orthophotos in the GCMRC GIS department. Using cues from the river guide and the xerox enlargements, points were located on the orthophotos which had been enlarged to a scale between 1:750 and 1:900. The ARC View measuring tool was used to determine the length of the line as it passed through each vegetation type. Bare rock, talus or sand (category "D") and debris fan / rocky / boulder areas (category "F") were not included in the analysis. Lengths of each vegetation type were summed across all sample lines in all reaches to produce an overall total length for each type.

To compare the relative amounts of each vegetation type in the sampling sites with relative amounts in the river corridor, the totals in both were first converted to proportions of the total vegetated length or total vegetated area and arcsine-square root transformed. Amounts from the two areas were then compared in two ways. First a paired t-test was used to check for systematic differences between the two groups. Because we had excluded the "D" and "F" cagetories from the analyses, the independence problems associated with summing to 1.00 did not have as much of an effect. Second, regression analysis with a test of the null hypothesis of a slope of 1 and an intercept of zero using standard methods (Zar 1999) was applied to the data. If monitoring sites were indeed a reflection of the composition of the river corridor vegetation, we predicted no significant difference from these parameter estimates.

To further describe vegetation patterns in the transects and monitoring sites, k-dominance plots (Platt et al. 1984) were generated from the two data sets. By comparing plots of cumulative proportions *versus* the natural log of rank order for both data sets, differences in dominance relationships would become apparent. If the two sets of data were drawn from communities with abundances distributed among vegetation types in a similar way, then plots of the two lines would appear similar with one or more points of intersection (Platt et al. 1984). On the other hand, if one or a few vegetation types dominated data from one of the sets far more than in the other, the curves may have had a similar shape, but would not overlap.

Fish Habitat Data

In order to assess the suitability of vegetated shoreline data for use in monitoring habitat availability for juvenile native fish, habitat descriptions from the most relevant publication were consulted (Converse et al. 1998). Information on shoreline conditions in the three habitat "reaches" described in the published material were pulled from the 1998 and 1999 vegetation monitoring data. No substrate data (e.g. amount of talus, cliff, debris fan, etc. shoreline) were collected which would apply to the descriptions of juvenile fish habitat, so only the vegetation data was used. One of these reaches (Reach 3) was represented in only one sample in the two years, and so was excluded from the rest of the calculations. Descriptive statistics, including means and standard deviations of estimates of the length of vegetated shoreline and total area of vegetated per 50 meter sample in the near-shore area were generated as well as the difference between 1998 and 1999 levels in each reach. After confirming that variances did not differ between years, pooled standard deviations were calculated for each reach. The pooled standard deviations and the 1998 / 1999 difference were used in a power analysis for a random effects one-way ANOVA (Bface; Lenth 1996) to determine the power of the comparison with and five and ten observations per reach per year, and the number of years required for a finding of significant differences, based on a constant rate of change. Five and ten observations per reach was selected because one sample per 0.5 - 2 km seemed an appropriate level of effort.

Bird Habitat Data

To check the suitability of vegetation monitoring data for bird habitat quality monitoring we will use the areal extent data from several of the monitoring sites as inputs for predictive equations from earlier work. Mathematical models for predicting abundance and richness of bird communities have been developed by Sogge et al. (1998). Some vegetation data collected during the previous several years is directly applicable to these models, and could be used to predict bird community measures. The equations for predicting abundance and species richness from "NHWZ" (new high water zone patch areas) will be used for three reasons: a) it minimizes problems arising from different definitions of such distinctions as a "tamarisk" vs. a "tamarisk / grass" patch, b) it includes as much information as possible from the entire vegetation site, and c) it has the best explanatory power of any of the patch area measures used by Sogge et al (1998 , chapter 7, p. 33).

Polygon area data from Forster beach (122.8 L), the only vegetation monitoring site which overlapped with the bird habitat data, will be used. The equations for predicting bird abundance, species richness, and Shannon diversity (H') based on habitat are (**These equations have not yet been provided to us**):

Abundance = *****(NHWZ) + ****(SEG) + ****(NHWZ x SEG)
Richness = ****(NHWZ) + ****(SEG)
Diversity = ****(NHWZ) + ****(SEG)

where SEG is a fixed factor accounting for differences based on location in the river corridor, and "NHWZ x SEG" was the a significant interaction term between river segment and areal extent of new high water zone vegetation determined in the model for predicting bird abundance. In this case, both SEG and NHWZ x SEG are essentially constants because we were considering only one site in one segment.

Once these numbers were calculated, two comparisons will be made. First, to assess the utility of the measures, abundance, diversity, and richness estimates will be compared to numbers from Sogge et al. (1998) to determine if numbers were close to actual bird counts. Second, the 1998 and 1999 estimates will be compared to assess the dynamics in bird community measures predicted by habitat variability.

RESULTS

Power Analysis

Polygon vegetation attribute data showed that the power of year-to-year comparisons varied with both measures and type of vegetation (Table 4). The power of tests to detect changes of the

Table 4. Power analysis of TVV, cover, richness, and diversity estimates for vegetation sampling sites indicating minimum numbers of sites required and minimum detectable change based on data structure in 1998 and 1999.

Measure	Type[1]	1998	Change	N (min) [2]	N (hyp) [3]	Min (yr.) [4]
TVV	M Scrub	29.99	-1.42	77	21	4.30 (4)
	T Scrub	15.73	+0.08	**	20	3.31 (42)
	Wetland	16.60	-2.70	54	31	4.48 (2)
	Bar Top	4.00	-1.13	50	9	3.12 (3)
Cover	M Scrub	59.54	-32.06	4	17	13.01 (1)
	T Scrub	61.06	-22.81	9	34	19.50 (1)
	Wetland	54.68	-13.5	12	18	14.15 (2)
	Bar Top	11.56	-4.52	49	12	11.47 (3)
Richness	M Scrub	13.91	+0.348	**	17 / 6	1.30 (4)
	T Scrub	10.47	-0.796	**	72 / 20	3.13 (4)
	Wetland	14.74	-0.408	**	43 / 13	2.57 (7)
	Bar Top	8.4	+0.484	**	** / 48	5.01 (11)
Diversity (H')	M Scrub	1.277	-0.009	**	14	0.23 (26)
	T Scrub	1.020	-0.207	12	13	0.23 (2)
	Wetland	1.211	0.073	**	17	0.30 (5)
	Bar Top	0.899	0.200	30	30	0.39 (2)

[1] Vegetation type: M Scrub = mixed riparian scrub (*Salix*, *Tamarix*, *Baccharis*), T Scrub = dense *Tamarix* patches

[2] Minimum number of sites to conclude 1998/1999 change significant (alpha = 0.05, power = 80%); ** = greater than 100

[3] Minimum number of sites to conclude significant (alpha = 0.05, power = 80%) hypothetical TVV change of 3, cover change of 10%, richness change of 1 and 2 species, H' change of 0.20; ** = greater than 100

[4] Minimum detectable change for alpha = 0.05 and beta = 0.20, given the variability in the 1998 and 1999 data and the number of years required for this change given measured rates of change.

magnitude observed between 1998 and 1999 was relatively low; a minimum of 50 sites would have been required to find the 28% change in Bar Top TVV significant. Greater numbers of samples would have been required to detect significant changes in the TVV for mixed scrub, tamarisk scrub, and wetland patches. To detect a change of 3 TVV "hits" per sample point would have required between 9 and 21 sites.

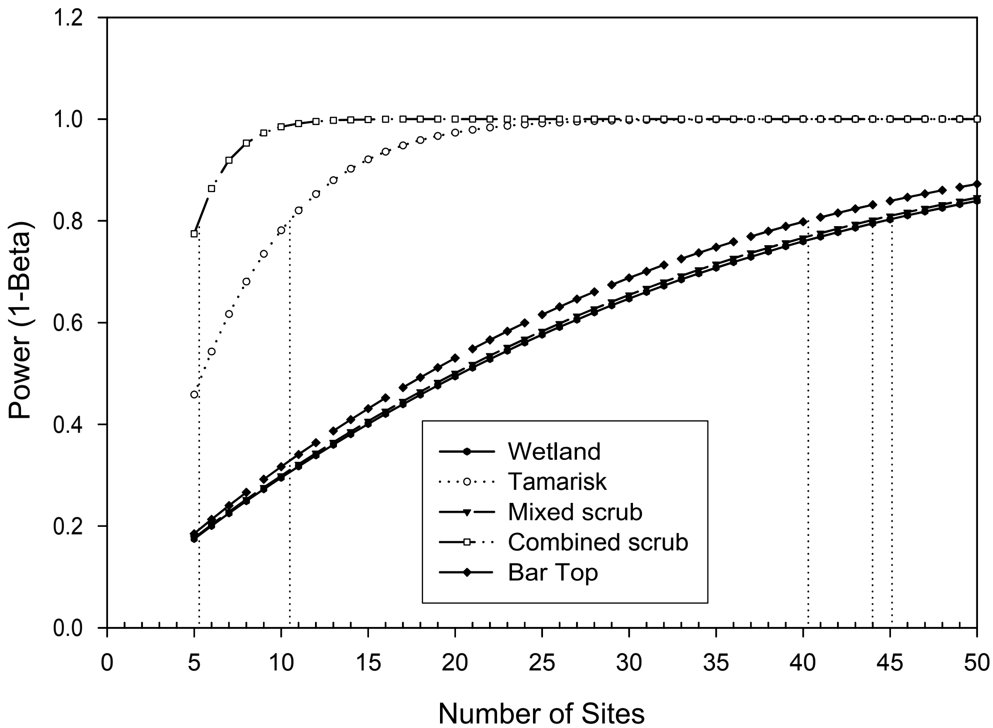
Except for the case of sparse, bar-top patches, tests of total cover change had reasonable power. The large cover changes reported in 1999 could be detected with as few as four sites (Table 4). Changes of a more reasonable magnitude would have required12 to 34 sites depending on the patch type. Current sampling practices would have captured changes of between 19.5% cover in tamarisk scrub patches (29% of 1998 values) and 11.47% cover in bar top patches (99.2% of 1998 values).

Power was much lower in comparisons of species richness. There would have been no finding of significant differenceswith fewer than 100 sites given the small amount of change and the high variability in the 1998 and 1999 data (Table 4). More moderate levels of change (two species) would have been detected with fewer than 50 sites. The minimum detectable change in richness varied form 1.3 species (in mixed scrub patches; 8.5% of 1998 values) to 5.01 species (bar top patches; 59.7% of 1998).

Differences in variability among patch types led to great differences in the power of comparisons of H' (Shannon diversity) between 1998 and 1999. Relatively large changes in tamarisk scrub and bar top patches H' measures resulted in tests powerful enough to detect 1998 / 1999 H' changes with 12 and 30 sites, respectively. However, more than 100 sites would have been required to find statistical significance in the small changes in H' seen in mixed scrub and wetland patches. When the power of tests were determined for comparable amounts of change (0.20 units of H'), the minimum number of sites for all four types were comparable as well (13 to 30).

Surprisingly, there was very little variation across all measures in the number of years required to find statistical significance. With the exception of cases where there was very little change in 1999 (TVV in tamarisk scrub and H' in mixed scrub) and a case of high among-site variability (richness in bar top patches), most measures would have shown significant differences after five or fewer years of monitoring (Table 4). The large 1999 changes in cover would have been picked up in three or fewer years, TVV changes would have been uncovered in fewer than four years, and H' changes in less than 5 years. Other than the bar top patches, changes in richness would have been detectable in four to seven years, depending on patch type.

Power curves generated from the total areal extent by vegetation type data showed that while tests for changes in tamarisk patches and a combined scrub patches type were adequately powerful, tests for wetland, mixed scrub, and bar top patches were much less so (Figure 9). Only 11 and six sites, respectively, were required to achieve a power of 80% in comparisons of the area tamarisk and new high water zone scrub patches. Wetland patches would have required 46 sites, mixed riparian scrub patches 45 sites, and bar top patches 41 sites to achieve the same level of power.

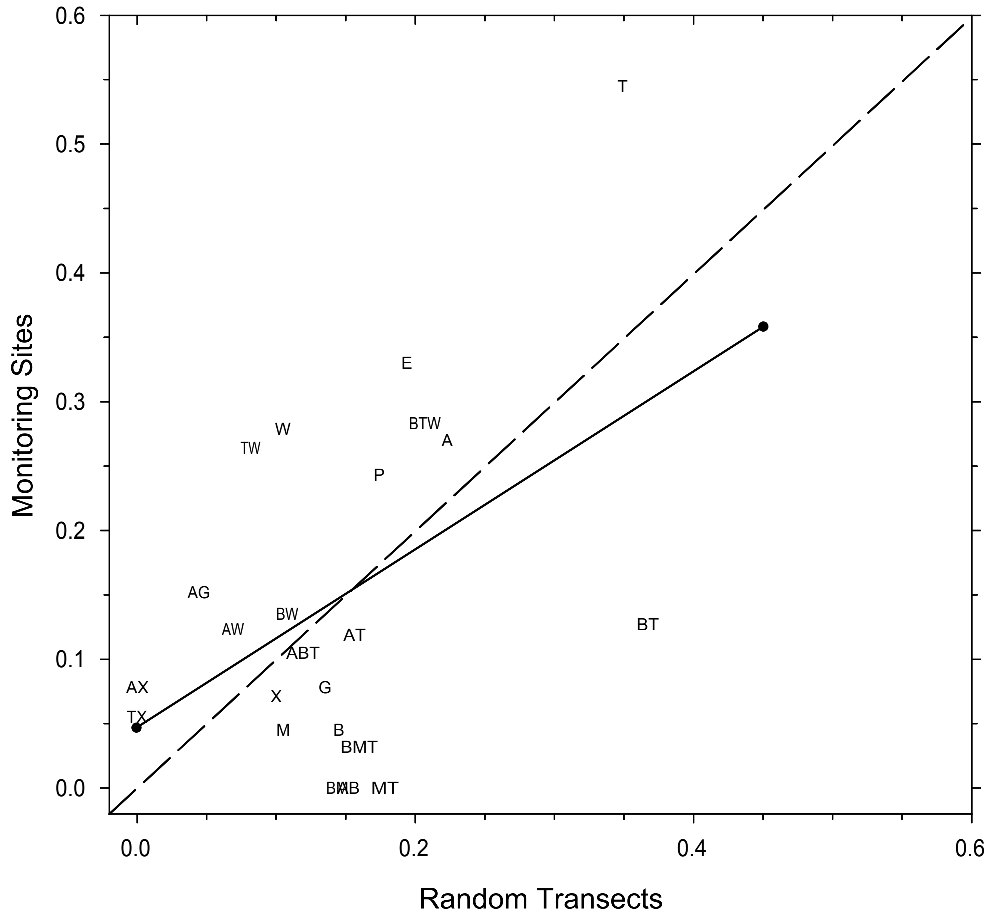


9. Power curves for detecting change in the total areal extent of major vegetation types as a function of the number of sites sampled.

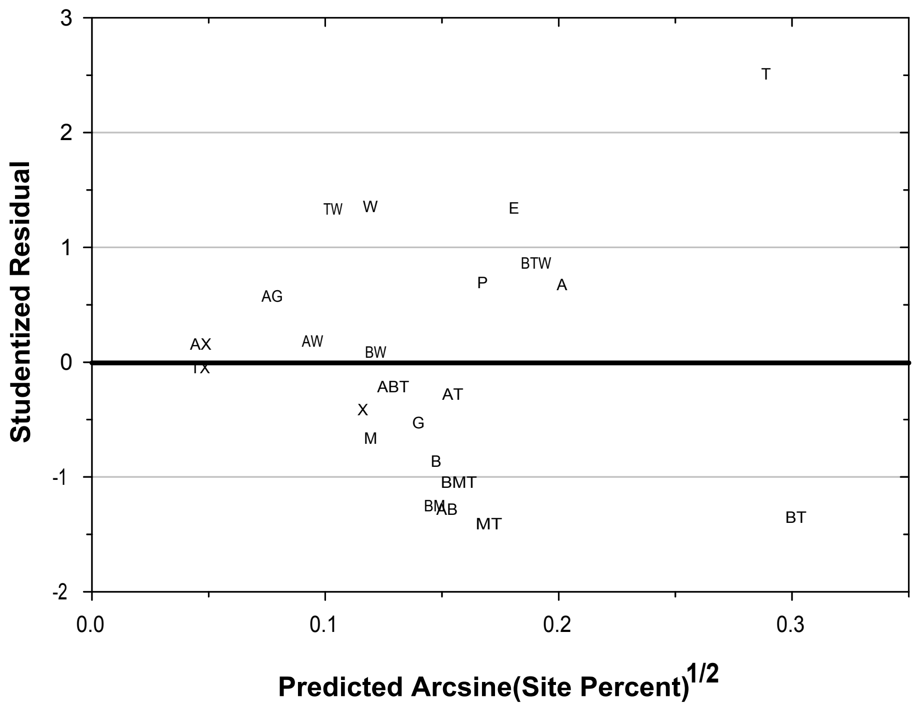
Sampling site representation

Data from the sampling sites and random transects show similar, but not identical, vegetation types. The paired t-test showed no significant difference between the two groups ($t_{(22)} = 0.091$, n.s.), indicating that there were no overall differences in percentages of each vegetation type between the two groups. The regression analysis showed no significant difference from a unit slope (Figure 10; $t_{(25)} = -1.055$, n.s.) and an intercept of zero ($t_{(25)} = 0.950$, n.s.). However, an examination of the studentized residuals (residual / std. error; Figure 11) shows that there are some individual departures from complete agreement. Tamarisk patches ("T") are over represented (studentized residual = 2.51), and many of the baccharis vegetation types are under represented in the monitoring sites (ab, bmt, bm, and bt all < -1.0). Surprisingly, *Equisetum* (E) and *Phragmites* (P) dominated marsh vegetation are only marginally over represented in monitoring sites (studentized residuals = 1.34, 0.69, respectively).

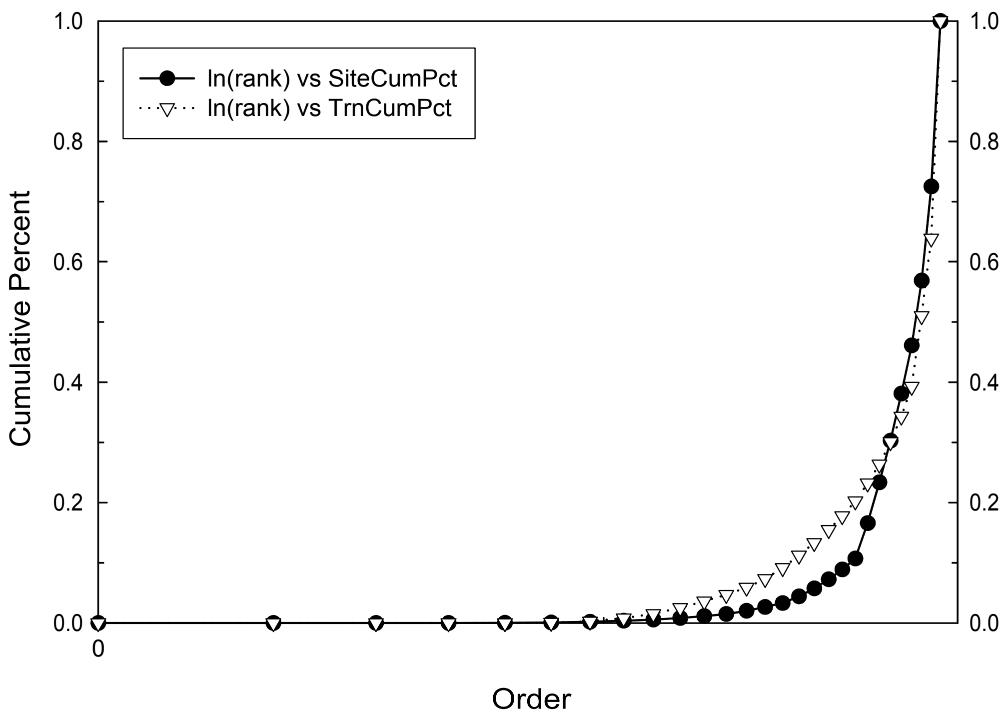
The k-dominance plot indicates that there is no simple difference between monitoring sites and random sites in the dominance structure of vegetation types (Figure 12). The dominance plot lines in the two samples cross at one point. While this indicates that the more common vegetation types account for slightly more of monitoring site vegetation than the common types encountered in transects, it also indicates that dominance relationships do not differ significantly in the two groups (Platt et al. 1984)



10. Comparison of proportion of 10 monitoring sites and 330 random transects under different vegetation types. Dashed line represents the line of complete agreement with unit slope and zero intercept, solid line represents the regression equation.



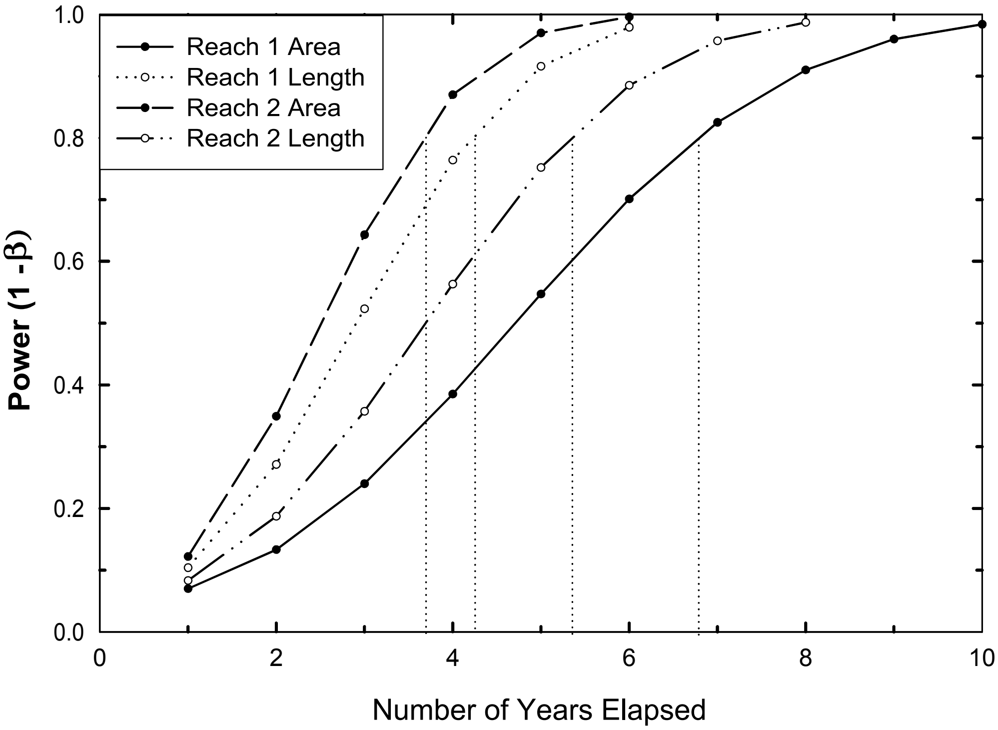
11. Studentized residuals (residual / standard error of prediction) for vegetation types from Figure 10 showing over representation of tamarisk patches (T) and under representation of baccharis patches.



12. K-dominance plots for data from monitoring sites (filled circles) and random transects. Cumulative percent is plotted against the natural log of rank order of abundance (lowest to highest).

Fish habitat data

The power analysis of the fish habitat data showed that neither of the tests in either reach was very powerful (Figure 13, Table 5). Comparisons of the length of vegetated shoreline between years had a power of only 17 to 23 percent, and would require five or six years of consistent change before producing an 80% chance of finding a significant difference at " = 0.05. Likewise, the power of tests for changes in the area of vegetated shoreline ranged from 14 to 28%, and would have required four to seven years of change before it was likely that a significant change would be found. Increasing the sampling intensity to 10 sites per reach per year had only a marginal effect; the power of the tests increased approximately 10% and the number of years required to detect change decreased only 1 or 2 years in each case.



13. Power curves for total length and total area of vegetated shoreline in the first two reaches of Converse et al (1988) as a function of number of years elapsed under constant trends calculated from 1998 and 1999 data. Drop lines indicate points at which power = 0.80.

Table 5. Power (1 - $) of tests comparing length and area of vegetated shoreline in the first two reaches of Converse et al. 1998.

		Length		Area	
Reach	Miles	Power	Years[1]	Power	Years[1]
1	61.4 - 65.5	.230	5	.142	7
2	65.5 - 73.4	.176	6	.274	4

[1] Number of years at the observed (1998 / 1999) rate of change required to demonstrate a significant difference where " = 0.05 and power = (1 - $) = 0.80.

Nor was the system-wide monitoring for change in shoreline a powerful design. None of the three measures, length, total area, or number of patches of vegetated shoreline would have detected a 10% change, based on the data from 1998 and 1999 (Table 6). Tests of all of the three aspects of vegetated shoreline had power less than 80% for changes of less than 45% of the 1998 values (Table 7, Figure 14).

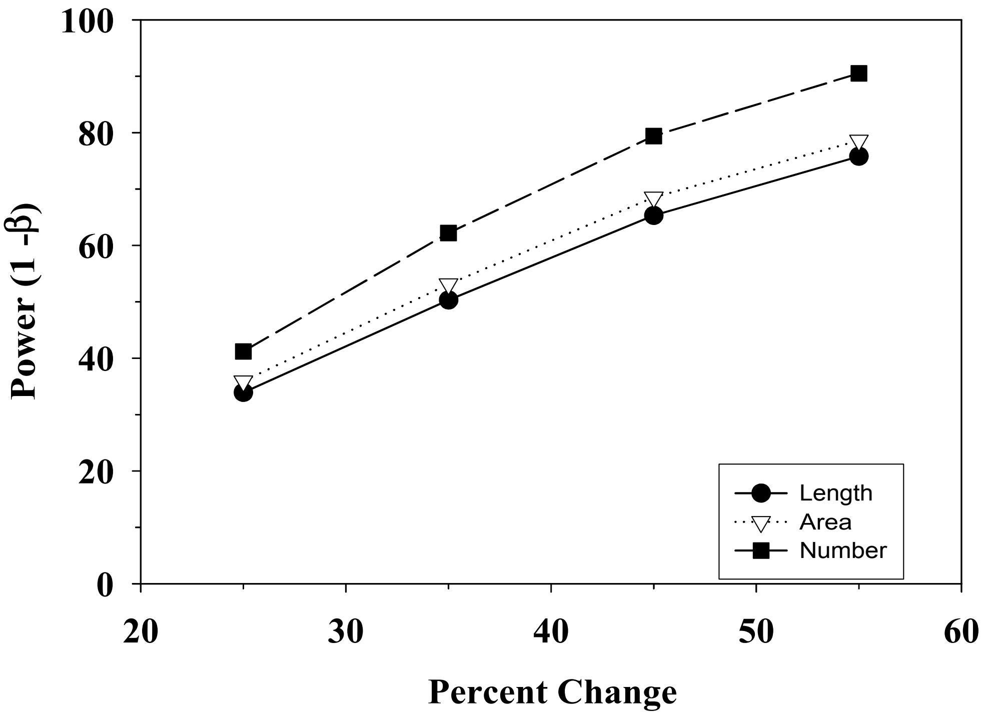
Table 6. Power analysis of vegetated shoreline data . Columns include 1998 levels, probability of detecting a 10% change and the range of no detectable change based on data in 1998 and 1999.

Measure	1998	Detect 10% [1]	Lower – Upper, (%) [2]
Length per 50 meter sample	13.79	No	11.0 – 18.5 (25.7)
Area / 50 meter sample	15.50	No	9.5 – 25.3 (50.9)
Number / 50 meter sample	1.51	No	0.88 – 2.30 (47.3)

[1] Indicates whether a change of 10% in the 1998 levels would have been significant at alpha = 0.05 and beta = 0.80 (one tailed)
[2] Detransformed lower and upper limits of the range of no detectable change based on alpha = 0.05 and beta = 0.20 (one tailed)

Table 7. Size of change from 1998 to 1999 and power ($1 - \beta$) of paired sample test to detect changes in aspects of vegetated shoreline across a range of magnitudes.

	1998 to 1999	25%	35%	45%	55%
Length	33.1%	33.9	50.3	65.3	75.8
Area	86.7	35.9	56.1	68.5	78.6
Number	85.1	41.2	62.2	79.4	90.5



14. Power to detect change in length, area, and number of vegetated shoreline patches in canyon-wide surveys of all 11 reaches of Schmidt and Graf (1990) as a function of the size of between-year changes expressed as a percentage of 1998 values.

Bird Habitat Data

Because the authors of the original study have not yet been able to provide us with the equations generated from their data, we cannot evaluate the question of the suitability of the vegetation monitoring data on patches to inter-annual fluctuations in bird populations. It does not appear likely that the equations will be forthcoming, because the author with that information (M.W.) is currently committed to other projects to the point that he cannot attend to our requests.

Section 2:
VEGETATION SAMPLING

Many methods have been applied to the measurement of vegetation or plant species' abundance in the river corridor of Grand Canyon. Most have estimated some aspect of plant cover using one of three methods. Pre-dam descriptions of vegetation in Glen Canyon were done using line-intercept methods to determine relative abundances of the common dominant plant species (Flowers 1959, Woodbury et al. 1959). Many workers since the early Grand Canyon ecological work in the 1970s have used line-intercept methods to describe vegetation. L.T. Green, A.M. Phillips and their colleagues used line-intercept data to establish monitoring sites to study impacts of recreational use (Green, L. T.,III 1978, Green, L. T. et al. 1980, Phillips, B. G. et al. 1986). Brian (1987)used line intercept transects to sample willow stands as part of a description of willow communities. Hualapai tribal resources personnel have been using repeat line-intercept transects to monitor vegetation in culturally important areas (Phillips, A. M.,III and Jackson 1996, 1997, 1999, Jackson et al. 1997). The Southern Paiute Consortium has also been monitoring vegetation with line-intercept methods (Stoffle et al. 1995, Austin and Osife 1996, Austin et al. 1996, 1997, 1998, 1999).

Beginning in the early 1990s, several groups used basal, rather than foliar, cover estimates to describe vegetation. Stevens and Ayers measured basal cover of all species in quadrats and segmented belt transects in new high water zone habitats (Stevens and Ayers 1993, 1995). Their methods were adopted by Hualapai Department of Natural Resources personnel working in sites between National Canyon and Pearce Ferry (SWCA 1994, 1995, Christensen 1997). Kearsley and Ayers (1996a) used the same methods to continue Stevens and Ayers for another year and, with other colleagues from N.P.S. and the C.P.R.S. used basal cover methods and ocular estimates during the same year (Kearsley et al. 1996).

Ocular estimates of cover have also been used for assessment and monitoring. In the 1980s and 1990s, several authors used computer assisted assessments of extent and cover of new high water zone vegetation in aerial photographs to determine medium-term effects of high flows and interim flows (Pucherelli 1986, Brian 1987, Waring 1996). Kearsley, Ayers, and their colleagues used ocular estimates of cover in 3 m radius circular plots to monitor new high water zone vegetation in 11 sites between Glen Canyon Dam and Diamond Creek (Kearsley and Ayers 1996a, b, 1999a, b, c, Kearsley et al. 1996).

Methods and Aims of Floristic Vegetation Sampling

Sampling for floristic aspects of vegetation monitoring involves measuring one or more of the basic vegetation attributes: biomass, frequency, density, cover and structure (Shimwell 1971, Causton 1988, Bell et al. 1991). Biomass is measured in some form of plant standing mass or mass production in a year for a given area.. Density is the number of individual plants or, depending on definitions, the number of stems per unit area. A species frequency is the probability of finding it in a quadrat of a given size. A related measure, cover, is the proportion of the ground surface which would be covered by a vertical projection of all foliage on to it, measuring either leaves and stems (foliar cover) or trunks / basal clumps (basal cover). Plant structure is usually a quantitative description of the three-dimensional or at least vertical distribution of plant parts.

Biomass. Biomass is generally measured for range studies or in applied management settings (Kelly et al. 1974). It is probably the best single measure of a species' importance within a community (Bonham 1989), but suffers from several shortcomings for monitoring uses. First, direct- and double sampling involve consistent or periodic destructive sampling of above-ground tissues (Catchpole and Wheeler 1992) and many samples are required to reduce the variance of estimates (Gillen and Smith 1986, Causton 1988, Heidelbaugh and Nelson 1996). And second, between-year variation generally reflects short-term, weather related factors and herbivore population fluctuations more than long-term trends (Mueggler 1992). When these regional, short-term fluctuations are large, they tend to severely limit the power of monitoring studies to detect trends (Urquhart et al. 1993). Biomass is lognormally distributed, since mass is a multiplicative, rather than additive, quantity, so data should be log transformed before analysis (Bonham 1989).

Density. Plant density measures are very useful for comparing trends within individual species or among species with similar growth forms (Daubenmire 1959, Oldemeyer and Regelin 1980), and where composition, rather than the absolute amount of vegetation is important (Friedel and Shaw 1987a). However, comparisons across growth form types (trees vs. annual herbs, shrubs vs. vines) do not allow measurements to directly measure the importance of different types of vegetation (Bonham 1989), and interpretations can become hazy in the cases of clonal and rhizomatous species (Daubenmire 1959, Wein and Rencz 1976). Distance based methods, such as nearest neighbor and angle order, are extremely quick to use in the field because they do not require that plots, transects, or lines be located (Good and Good 1971), but they have very strict assumptions about the distribution of individuals which, if violated, result in extremely high variability and biased estimates (McNeill et al. 1977, Bonham 1989). Density estimates are Poisson distributed, where variance is a function of the mean, so data should be square-root transformed before analysis (Bonham 1989).

Frequency. Plant frequency is a very rapid method for assessing plant distribution, since it involves only a presence / absence measure for each quadrat (Sutherland 1996). However, frequency has a number of shortcomings for a monitoring setting which need to be acknowledged. First, it is sensitive to quadrat size, shape, and number, and results in extremely variable estimates (high between-observer variance) unless very large numbers of samples are taken (Shimwell 1971, Arzani and King 1994). And second, it is biased for plant size and distribution, tending to record disproportionately higher levels for large, regularly distributed species than smaller, more clumped species so that comparisons across taxa and growth forms are unreliable (Shimwell 1971, Sutherland 1996). Since it is a presence / absence phenomenon, frequency is a binomial quantity, and data need to be transformed before standard linear methods can be applied (Bonham 1989).

Cover. Plant cover is the oldest, and probably the most commonly used attribute of plant abundance (Bonham 1989). It is measured using either points, lines, or polygons as sampling units (Floyd and Anderson, J. E. 1987) and can be measured as either foliar / vegetative cover or as basal cover (Shimwell 1971). It is not biased for plant size or distribution (Floyd and Anderson, J. E. 1987) and is therefore appropriate for comparisons involving many taxa and growth forms and for defining dominance relationships (Floyd and Anderson, J. L. 1982, Floyd and Anderson, J. E. 1987, Arzani and King 1994). Although it is related to biomass, cover, especially basal cover, is far less subject to inter-annual environmental fluctuations (Heidelbaugh and Nelson 1996). Cover, like biomass, can be related directly to remote sensing techniques (Arzani and King 1994).

Structure. Measuring habitat structure is a common way to tie information on vegetation to animal components of the community. It is distinct from floristic sampling because it involves only the arrangement of objects (plant parts) in space, without any reference to the taxonomic status of the objects (Bell et al. 1991). Habitat structure serves as an appropriate surrogate for many compositional variables because it integrates information across all species and eventually describes both the quantity and distribution of resources (Noon 1999).

Cover Measurement Methods

Because plant foliar cover is the most commonly used attribute of plant abundance, and because of its minimal bias for plant size, distribution, and growth form, we would recommend that it be used for monitoring riparian vegetation in the Colorado River corridor. Cover is often used as a proxy for biomass so that structure, foodbase, and other wildlife-related measures can be related to monitoring data. Furthermore, because it can be directly related to remote sensing techniques, it will allow process - based monitoring of local changes in plant abundance and composition measured on the ground to be linked to larger-scale, system-wide patterns.

As mentioned in the previous section, cover can be measured in a number of ways which generalize into point-, line-, and quadrat methods. In the sections below, we will describe the mechanics, variants, problems, and typical applications for each cover estimation type . We will also compare their utility based on three important monitoring criteria: sensitivity, precision, and speed..

Point methods assess cover on the basis of the proportion of very small sampling locations which are covered by leaves or stems. First applied to rangeland studies in the 1920s in New Zealand (references in Goodall 1952), the standard apparataus includes some linear or rectangular frame which holds fine pins which are lowered vertically or at a fixed angle into the vegetation (Causton 1988, Bonham 1989, Sutherland 1996). For each pin, contacts with live vegetation counts as a "hit" and either one or all contacts with each species per pin are recorded, depending on whether cover, or relative cover measures are desired (Goodall 1952). Cover is usually expressed as the percentage of all pins which contact at least one species. Cover values range from 0 - 100 percent for any one species, and may exceed 100 percent for total cover due to layering of canopies of multiple species.

The basic method has been modified many times for applications in specific habitats. To survey grassland areas, a wheelpoint apparatus is often used. A wheel with a known circumference and one or several "survey spokes" is rolled along a transect, and point intercept data is recorded whenever a survey spoke strikes the ground and goes vertical (Walker 1970, Friedel and Shaw 1987a, b, Arzani and King 1994). It has been recommended that such a device requires a field crew of three (roller, reader, recorder) to operate a wheelpoint sampler (Friedel and Shaw 1987a). Step-point methods and transect-point methods evaluate cover at the toe of the shoe or along one side of a transect tape using pins, knives, bayonets and other sharp-edged instruments (Walker 1970, Poissonet et al. 1973, Glatzle et al. 1993). For marine data collections in subtidal situations, a rig consisting of knotted string connected at both ends to a heavy bar is dropped to the bottom, and contact between the knots and benthic algae and invertebrates can be recorded (Leonard and Clark 1993). For data collection above the water in the intertidal zone, rectangular plexiglass squares with dot grids have been laid on the substrate, and the proportion of dots overlying algae and sessile invertebrates has been used to measure cover (Meese and Tomich 1992). Finally, because pin and point diameters greater than 1.5 cm have large effects on cover estimates, "sighting frames" with two superimposed grids of fine string or wire have been used to locate points in the vegetation without any real pin or needle points being used (Floyd and Anderson, J. L. 1982, Floyd and Anderson, J. E. 1987). Tree canopy measurements can be made with samplers consisting of mirrors and prisms mounted on blocks or tubes so that vertical sightings determine whether there is foliage or branches directly above sample points (Lemmon 1956, Vales and Bunnell 1988).

Several authors have pointed out problems with the point interception method. Mostly, these involve the impact of pin diameter on cover estimation, because any pin with a measurable diameter samples an area larger than a true point, and so will overestimate true cover in the same way "loop" samples do in range settings (Goodall 1952, Becker and Crockett 1973, Catchpole and Wheeler 1992). This includes the case of marine samples where dots on plexiglass have a measurable width (Meese and Tomich 1992). Methods for the location of sampling points which are neither regular or random, as with step-point locations, results in biased samples (Becker and Crockett 1973, Bonham 1989). Plant morphology and growth form can affect cover estimates made with point intercept methods; vertically oriented grasses and graminoids tend to be undersampled relative to broad-leaved species with horizontal architectures, and fine- or dissected leaved morphology tend to be encountered more than entire-leaves species with the same absolute cover (Goodall 1952, Johnston 1957, Poissonet et al. 1973). In addition, pin-based methods tend to become unwieldy in tall, dense woody vegetation where frame setting becomes awkward, a wheelpoint apparatus is difficult to maneuver, and vegetation must be sampled both above and below the point (Walker 1970, Causton 1988, Catchpole and Wheeler 1992, Arzani and King 1994). Sighting frames are also difficult to use in diverse, multilayered vegetation where plants must be moved to see vegetation below (Sutherland 1996).

Point interception methods with pin frames or wheels are most often used in herbaceous and grassland settings with low (< 1m) canopies (Goodall 1952, Friedel and Shaw 1987a, Sutherland 1996). Similarly, point interception quadrats in marine settings are best suited for low-lying and prostrate taxa, although they are capable of sampling where some vertical layering occurs (Meese and Tomich 1992, Leonard and Clark 1993). The use of vertical sighting tubes and gridded mirrors allows point data to be taken from canopy layers more than 40 m overhead (Lemmon 1956, Vales and Bunnell 1988), although identification of species responsible for the point interceptions becomes more difficult as canopy height increases (Lemmon 1956).

Line interception methods determine cover by measuring the proportion of the length of a line through a habitat which intersects live vegetative cover (Hormay 1949). The model underlying these measurements assumes a vertical plane through the line, but measures only the vertical projection of the vegetation contacts with the plane onto that line. Usually, line intercept data is collected off of a measuring tape stretched between two points. The beginning and end points of where each individual plant, or unbroken group of plants, contacts the line are recorded. Percent cover can be defined in several ways. First, percent cover for the transect is defined as one minus the proportion of the length of the line which intercepts bare ground with no vegetation above it. Percent cover for each species is measured as the proportion of the line which intercepts that species. As with point methods, the total cover can be greater than 100 percent because of canopy layering.

There are many ways to measure along a line where vegetation begins and ends. Typically, readers either use ocular estimates of where a plant first contacts the line, or uses a plumb line to get accurate measures (Hormay 1949, Causton 1988). In cases where field time is expensive and limited, especially in marine settings where data is collected while diving, transects are videotaped in the field, and measured on a screen (Leonard and Clark 1993).

Problems with the use of line-intercept data were pointed out early in the development of the method. Subjective, ocular estimates of contact endpoints can lead to great variability between observers (Hormay 1949, Daubenmire 1959, Floyd and Anderson, J. L. 1982, Causton 1988). This problem is especially acute for trees and shrubs where breaks in the canopy must be interpreted as either open or occupied, depending on whether observers are measuring actual interception of the line, or an idealized canopy dripline in which canopy gaps are filled. As such, they tend to overestimate the canopy of shrubs and underestimate the cover of bare ground and herbaceous species. One proposed solution, using some “normal density” of foliage and stems to determine whether a segment is occupied (Hormay 1949), would vary for each species. It would seem to require subjective assessments as to whether each point met some criteria of “normal density” or not based on past experience and would, therefore, merely substitute one subjective decision for another.

Line intercept methods have been used in many different habitat types. Most often, they are applied in some type of shrub-dominated community including shrub / grasslands and savannahs (Johnston 1957, Heady et al. 1959, Brun and Box 1963, Friedel and Shaw 1987a, Sutherland 1996). Video recordings of benthic algal communities have been used to perform line-interception transects in marine systems (Leonard and Clark 1993). Desert, and sparse vegetation are generally not suited to line-interception methods owing to open canopies of species and the high number of replicates required to account for patchy distributions (Friedel and Shaw 1987b).

Quadrat cover methods use small circular or rectangular frames to delineate samples in which ocular estimates of cover are made. Cover is simply the observers’ estimates of the proportion of the ground surface within the frame which would be covered by a vertical projection of all vegetation (Shimwell 1971). Frames are generally less than one square meter in area, typically 0.1 m^2 (20 x 50 cm; Daubenmire 1959), although larger sizes, up to 25 m^2 have been employed (McNeill et al. 1977). Usually there is some form of visual calibration associated with the estimates involving separate small squares or markings on the quadrat frame which represent fixed proportions of the total framed area (e.g., 1%, 5%, 10%, 25%; Daubenmire 1959, Causton 1988). Often during estimation, subsamples or off-site samples are harvested to calibrate visual estimates as the observation process proceeds (Smith 1944).

Most of the variations of the quadrat method concerns the size and shape of the quadrat, and the way in which cover is recorded. Daubenmire (1959) gave a standard method of using a

Table 1. Daubenmire’s (1959) cover class scale

Cover Range	Cover Scale
0	0
0 % < cover ≤ 5 %	1
5 % < cover ≤ 25 %	2
25 % < cover ≤ 50 %	3

50 % < cover ≤ 75 %	4
75 % < cover ≤ 95 %	5
95 % < cover ≤ 100 %	6

20 x 50 cm rectangular frame and a cover scale for converting estimates (Table 1). Both were designed to limit between-observer error. The frame size was presented as a compromise between wanting a quadrat large enough to contain a meaningful sample while being small enough that observers would not have to move their eyes to take in the whole frame and thus rely on memory to determine cover values. Further, by converting estimates to broad cover classes, inter-observer differences in estimates would be reduced as well (Daubenmire 1959). For example, rather than two observers making estimates of 30% and 45% cover for the same species (a difference of 50%), in the same quadrat, both observers would give cover class scores of 3. Others have suggested the use of regular quadrat shapes to minimize edge / area ratios which determine the number of judgements about whether plants are “in” or “out” of the quadrat (Greig-Smith 1983). Others have argued that irregularly shaped quadrats avoid being trapped by plant traits (shape, dispersion) that occur over regular intervals (Shimwell 1971). Finally, it has been suggested that quadrat size be flexible, and mostly dependent on the size of the plants in the vegetation being surveyed (Gauch 1982).

As one might expect, most of the problems with quadrat data result from inter-observer variability due to the subjectivity in assessment of cover. In an early test for observer bias in visual estimation, Smith (1944) demonstrated observer bias even among trained individuals who were allowed to calibrate their observations by discussing estimates. He also showed that while training could reduce variability among observers, it remained high even after 7 consecutive days of estimating the cover of the same plots. On the final day of the study, estimates ranged from 15 % to 35 % on either side of the mean, and 95 % conficence bands ranged from 20 % to 50 % of mean values (Table 2). Other, less rigorous studies have shown similar and discouraging levels of inter-observer variability (Heady et al. 1959, Wein and Rencz 1976, Block et al. 1987, Floyd and Anderson, J. E. 1987, Friedel and Shaw 1987a), although some authors have argued that the visual estimation benefits from being conceptually straightforward and that observer training can reduce inter-observer error (Kennedy and Addison 1987, Dethier et al. 1993).

Table 2. Measures of variability (and percentage of mean value) among eight observers' ocular cover estimates of cover in 3 habitats. From Smith (1944).

Vegetation	Mean	S.D.	Half Range [1]	95% C.L. [2]
Sagebrush	5.25	1.14 (22 %)	1.95 (37 %)	2.69 (51 %)
Winterfat	7.52	1.00 (13 %)	1.58 (21 %)	2.36 (31 %)
Grassland	8.31	0.80 (9 %)	1.23 (15 %)	1.88 (23 %)

[1] One half of the range from lowest to highest estimates.
[2] One half the width of the 95% confidence band.

Visual estimation is probably the most broadly applied method of determining foliar cover. The earliest monitoring work on rangeland conditions used visual estimation (references in Smith 1944). It has been applied in polar deserts, wet meadows, shrub communities, tropical rainforests and savannahs, and in marine benthic studies (Wein and Rencz 1976, Bowman and Minchin 1987, Friedel and Shaw 1987a, Dethier et al. 1993). In addition to work in the field, visual methods can be applied to stereo photo pairs taken in situations where field time for estimation will be limited (Pierce and Eddleman 1973, Ratliff and Westfall 1973).

Comparisons of Methods

Each of the methods described in the previous section has advantages and drawbacks inherent in the way information on cover is collected. Table 3 is a compilation of the results of all studies listed in Elzinga and Evenden (1997) which compared two or more of the three methods directly. Comparisons here are based on three vegetation elements which would be important in a monitoring context. First, data types are compared on the basis of sensitivity – the ability to detect rare species when the most common species have been adequately sampled according to some pre-established criteria. This is important because for better or worse, maintenance of biodiversity, expressed as species richness, is often a primary objective of land managers. Therefore, monitoring methods should be able to adequately describe the contents of the system being monitored. Second, comparisons are made on the basis of precision – which, as used here, is the opposite of inter-observer variability (i.e., "measurement error" and "observer bias"). Because monitoring, by definition, takes place over long time periods, it is likely that there will be turnover of field personnel measuring vegetation. Therefore, it is important that methods have minimal inter-observer variation brought on by ambiguity, application of subjective judgement, and so on. Third, methods are compared on the basis of the speed with which they can be finished. Although comparisons can be made based on completing the same number of units (point frames, line transects, quadrats), here the question relates to the amount of time required to sample enough that a) estimates were within a fixed percentage of a "true" value, b) confidence limits came down to some fixed percentage of the true value, c) the coefficient of variation had stabilized, or d) the power of the test (1 - $) to detect differences at some level of significance had reached a specified point.

For sensitivity to rare species, point and line methods were generally better than quadrat methods. Line intercept transects, wheelpoint transects, point transects and point frames along transects tended to sample many different habitats as they crossed sampling areas containing many different vegetation types. Quadrat methods, which usually randomly disperse samples across habitats, tend to take longer to sample as many different habitats and vegetation types. This is especially true of species which are found in less common habitats.

Observer bias and inter-observer variability tends to be more of a problem with quadrat methods than either line- or point intercept methods. First, transects for line-intercept sampling tend to encounter more different habitats, thereby homogenizing observations among transects (Floyd and Anderson, J. E. 1987, Friedel and Shaw 1987b, Bonham 1989, Glatzle et al. 1993, Arzani and King 1994). A long, narrow sample such as a transect will cross many environmental gradients than a shorter, wider quadrat. Second, judgements on whether a pin contacts a leaf or not or where exactly on a tape a shrub canopy's boundary is will likely vary less than ocular estimates of the proportion of a rectangle occupied by a shrub's canopy projection (Heady et al. 1959, Becker and Crockett 1973). Using broad canopy coverage classes can reduce this source

Table 3. Comparison of sensitivity, precision, and speed of cover estimate methods.	
Line vs. Point Intercept	**The methods give similar richness estimates:** Line methods generally pick up more species, especially rare species (1) although point contact methods sometimes give more (2) and some show no difference (3). **Both methods have low between-observer variance:** Most studies show no difference between the two (4), but some have found differences (5) **Point methods are faster than line methods:** Almost all studies have shown that data collection, reduction, and entry are all two to five times faster with point methods (6), although wheel point fieldwork is slower in heavy brush (7)
Line Intercept vs. Quadrat	**The two methods give similar estimates of richness:** Lines and quadrats yield the same number of species (8), although annual and rare species show up more often in quadrats (9). **Quadrat methods are susceptible to observer bias:** Observer error is much lower in line intercepts (10) although broad cover classes can reduce or eliminate it (11) **Quadrat methods are faster:** In general, visual estimates are faster (12), although experience can affect speed (13).
Point Intercept vs. Quadrat	**Point methods yield far more species:** Richness estimates are higher with point intercept methods (14). **Point methods have far less observer bias:** Point contacts are far less subjective (15), although the simple communities from visual / photo studies can vary less because they have so few elements (16). **Point methods are faster:** The time required to achieve a specified level of precision is shorter for point contacts (17), especially if lab time is considered in photo quadrat studies (18).

1 (Johnston 1957, Heady et al. 1959, Brun and Box 1963, Walker 1970)
2 (Wein and Rencz 1976, Floyd and Anderson, J. E. 1987, Arzani and King 1994)
3 (Brun and Box 1963, Poissonet et al. 1973, Floyd and Anderson, J. L. 1982)
4 (Johnston 1957, Heady et al. 1959, Brun and Box 1963, Becker and Crockett 1973, Floyd and Anderson, J. L. 1982, Floyd and Anderson, J. E. 1987)
5 (Walker 1970, Poissonet et al. 1973, Wein and Rencz 1976)
6 (Johnston 1957, Heady et al. 1959, Brun and Box 1963, Becker and Crockett 1973, Poissonet et al. 1973, Wein and Rencz 1976, Floyd and Anderson, J. L. 1982, Floyd and Anderson, J. E. 1987)
7 (Walker 1970)
8 (Daubenmire 1959, Poissonet et al. 1973)
9 (Daubenmire 1959, Heady et al. 1959)
10 (Heady et al. 1959, Shimwell 1971, Poissonet et al. 1973, Floyd and Anderson, J. L. 1982, Floyd and Anderson, J. E. 1987, Friedel and Shaw 1987a)
11 (Daubenmire 1959)
12 (Daubenmire 1959, Floyd and Anderson, J. L. 1982, Floyd and Anderson, J. E. 1987)
13 (Daubenmire 1959, Heady et al. 1959)
14 (Heady et al. 1959, Friedel and Shaw 1987a, b, Foster et al. 1991, Leonard and Clark 1993)
15 (Heady et al. 1959, Shimwell 1971, Wein and Rencz 1976, Floyd and Anderson, J. E. 1987, Friedel and Shaw 1987a, b, Meese and Tomich 1992)
16 (Foster et al. 1991)
17 (Heady et al. 1959, Shimwell 1971, Wein and Rencz 1976, Friedel and Shaw 1987a, b, Leonard and Clark 1993)
18 (Meese and Tomich 1992, Leonard and Clark 1993)

of inter-observer error (Daubenmire 1959). However, such coding of cover data should be reserved for use in the creation of broader vegetation classifications rather than in monitoring, since it tends to obscure real differences (Floyd and Anderson, J. E. 1987). For example, using the Daubenmire scale in Table 1, a real change of as much as 50% in cover (30% vs. 45%) would not be detected even if all observers were exactly on target because cover would be coded as a “3" in both cases. Finally, the subjectivity of ocular estimates likely results from a great deal of experience which cannot be standardized, regardless of observer training (Smith 1944, Heady et al. 1959).

A further problem with ocular estimates is the accuracy of the estimates, regardless of the precision. A comparison of dry weight of harvested material with ocular estimates of the eight observers in Smith’s (1944) study showed that there was very little correspondence between the two. Individual observers varied in their ability to estimate total and species-specific amounts of vegetation. Furthermore, differences among observers did not vary in a predictable way which would have allowed an observer-specific correction to be applied to the data so that “true” amounts could be calculated (Smith 1944).

Line-intercept and point-intercept methods are roughly equivalent in terms of inter-observer variability (Heady et al. 1959, Floyd and Anderson, J. E. 1987). In shrub- and grassland habitats, no consistent differences found between the two. Some studies found that point-intercept data was more consistent across observers (Poissonet et al. 1973) and others found the opposite (Walker 1970). Most have found no difference between the two (Table 3).

For efficiency, measured as the speed with which a specified level of precision can be achieved, point-intercept methods tend to do a better job. Quadrat methods and charting can be faster than line methods in the field with experienced crews because setup time is minimal (Table 3). Because pin frames require roughly the same amount of time to set up as quadrat frames but reading pins goes faster than estimating species coverages, point-contact methods are usually quicker than quadrat methods and line-intercept methods, in some cases by a factor of two (Floyd and Anderson, J. E. 1987). Point-intercept methods’ time advantage is further increased when office time is considered. It requires only the tabulation of contacts by species, total contacts, and total no-contact pin numbers. With line-interception and quadrat methods, each species total proportion must be calculated from beginning and end points of line segments or sums of individuals’ areas, and bare ground must be calculated from all lengths of the line transect or patch areas with no contacts.

Structural Vegetation Sampling

Measurements of habitat structure fall into one of three general categories: area measurements, synthetic indices, and measures of resource quantity. The utility of habit area data is based on correlations among patch area, resource abundance and species richness. As patch area increases, so does the number of individuals of a species which the patch will support (e.g., Morrison et al. 1992). Similarly, patch area is a good predictor of the number of species to be found in the patch, as the rich literature on species - area relationships shows (MacArthur and Wilson 1967, Diamond 1975).

Indices of habitat structure produce some sort of information-theoretic description of the habitat. Foliage height diversity (MacArthur and MacArthur 1961), based on proportions of the canopy in different vertical strata, predicts bird species diversity in a variety of habitats. Similarly, foliage “profiles” (MacArthur and Horn 1969), provide predictive information about community processes such as succession and productivity (Aber 1979).

Quantity measures of habitat structure describe the abundance of plant resources within patch boundaries. Total vegetation volume (TVV; (Mills et al. 1991) predicts bird species abundance and species richness better than some commonly used indices of structure. Similarly, measures of the amount of decaying logs, vines, density of undergrowth without specifying th species responsible for the structure usually is a better predictor of animal populations than the species themselves and the “fine structure” of their distributions (Wiens 1989, Wagner et al. 2000).

In the Colorado River corridor of Grand Canyon, several methods have been applied to measuring habitat structure. Stevens and Ayers (1993, 1995) measured a multivariate index of structure at a single point in each vegetation patch at nine vegetation monitoring sites. The number of live contacts with a 5 cm diameter survey rod in intervals above the ground (0 - 0.3m, 0.3 - 1.0m, 1.0 - 2.0 m, 2.0 - 4.0 m, and 4.0 m+) were recorded at a point deemed “typical” of the patch. Kearsley and Ayers (1996, 1996a) used the same intervals in a continuation of this method, but expanded the sampling to include 10 - 20 randomly located points in each patch. A multivariate similarity analysis and an examinationof variability as a function of sample size showed that 10 points adequately represented all habitats except sparse bar-top areas where 15 were required (Kearsley and Ayers 1999c). Testing for trends was accomplished using repeated measures ANOVA, with patches as blocks.

Patch area has also been measured to monitor changes in physical habitat characteristics. Stevens and Ayers (1993, 1995) and Kearsley and co-workers (1996, 1996a, b, 1999b) used ArcInfo with the assistance of the GCES / GCMRC GIS department to measure the area of patches which had been classified into a number of ad-hoc vegetation types. Sogge and colleagues (Sogge et al. 1998) and other studying breeding birds in the river corridor (Spence et al. 1998, Spence 2000) have used GIS measurements of patch area to describe habitats.

Several groups have used univariate foliage / canopy quantity to assess habitat quality. Kearsley and colleagues (citations above) changed methods and began using TVV because it was a published method which was being used elsewhere in the southwestern U.S. in general and in Grand Canyon in particular (Spence 2000). In addition, it was shown to have direct management utility (Mills et al. 1991). Sogge (Sogge et al. 1998) used both patch area and vegetation volume (patch area x canopy thickness) to predict breeding bird densities and species richness.

MANAGEMENT RECOMMENDATIONS

From the information in the Request for Proposals to synthesize vegetation research information, the GCMRC has information needs regarding vegetation which fall into four categories:

– effects of dam releases on quantity and identity of riparian plants below the 45,000 cfs stage elevation
– habitat structure for wildlife in the new high water zone, with some information on species composition
– presence and distribution of exotic plant species associated with the river corridor
– presence and density of emergent and near-shore vegetation serving as habitats for juvenile native and non-native fish.

Based on the strengths of methods reviewed above and comparisons of sampling methods, we would recommend that monitoring of vegetation be carried out in several ways, First, due to the success of “quantity” structural data in describing bird communities, we would recommend that

TVV or a closely allied method be used to quantify the structural qualities of habitats. It produces a clear assessment of habitat which integrates across taxonomic boundaries and has direct application to wildlife population issues. The delineation of patches by digitization of hand-drawn boundaries should be review for subjectivity, however. The act of drawing a "bright shiny line" between what is habitat and what is not is fraught with subjectivity (Urquhart et al. 2000). This source of bias will become important as monitoring continues and the likelihood of personnel turnover increases.

Second, when the composition and abundance of riparian vegetation is needed, we recommend that point-interception methods be used. They do not suffer from the observer biases which arise from the ocular estimates of cover (e.g., Kearsley and Ayers 1999c) or where judgements of start and end points in line intercept transects depend on subjective interpretations of the "normal density" of each species (e.g., Phillips, A. M.,III and Jackson 1999). Problems with tall and complex vegetation can make point interception methods difficult, but the information gained will be more repeatable than methods which are easier to apply in the field.

To relate vegetation patterns and changes to the hydrograph, we recommend that point-intercept samples be placed in the context of stage elevation with transects perpendicular to the river (Warner 1983). With known stage-to-discharge relationships in the corridor (Randle and Pemberton 1987), the opportunity should arise to relate changes in vegetation to water management decisions. We do not recommend that individual points be taken along transects, but rather that point-intercept frames be used at intervals along the transects so that discrete information at each sampling point can be taken and related to stage information. The presence and absence of near-shore vegetation can be assessed at each transect as well. Data on the presence of exotics should come from the vegetation data itself. However, because these species have low densities and patchy distributions we would recommend that detailed studies in locations where densities are high be conducted (see Kearsley and Ayers 1999c) in order to capture information on the dynamics of these populations.

Section 3

REMOTE SAMPLING METHODS

Remote sensing methods have a huge potential in the assessment of vegetation resources. Managers can acquire data on vast areas of vegetation at costs as low as $ 0.62 / hectare in relatively short periods of time with very little human impact on the landscape (Doren et al. 1999). The rapid development of airborne and satellite sensor technology has brought many new methods of remote sampling into the realm of potential application, including multispectral and hyperspectral analysis of vegetation composition and lidar assessment of habitat structure (e.g., Lefsky et al. 1999a, Cochrane 2000).

The applicability of these technologies in Grand Canyon is still not completely clear. Good information has been acquired using these techniques in landscapes with moderate levels of topographic relief and varying amounts of vegetation cover. Some of the more experimental methods have shown promise under strictly controlled conditions (Gong et al. 1997) and others have been useful in biologically complex systems (Drake et al. 2000). Questions remain regarding the application of these methods in a constrained canyon system where complex topography and shadowing present problems beyond those of clouds, canopy architecture, microclimate, and surficial soil reflectance encountered in other systems (Lillesand and Kiefer 1987).

Motivated by the review panel on remote sensing technologies (Berlin et al. 1998), GCMRC has requested this review of remote sensing methodologies as part of the FY 2000 review and assessment of vegetation monitoring protocols. For this review, we have selected three sensor types for consideration: multispectral / CIR, hyperspectral, and lidar systems. The first and last were included because they were recommended in the R.F.P. from which this report comes. Hyperspectral systems are being considered because it shows great promise as a method to separate out species-specific signals from reflectance data and because it has been applied in at least part of the canyon in a test situation (Merényi et al. 2000a, b).

For the purpose of this review, our literature search was conducted on two fronts. First, we considered studies in Grand Canyon which had already been conducted and for which reports were available either through the GCMRC library or from the authors of the study themselves. Second, for more rigorous studies, we performed a literature search through two major electronic science databases, Science Direct and TREE-CD. Each contains in excess of 1.2 million published papers and covers a wide range of fields. To avoid characterizing methods on the basis of older papers in a field which is changing rapidly, we limited our searches to primarily studies published in the last 4 years (1997 - 2000). Studies published before this point were primarily based on data collected before 1995 and were generally preliminary in nature, especially those regarding lidar and hyperspectral sensor systems. We have excluded satellite data systems for two basic reasons. First, although the resolution of these systems is improving and in experimental data sets is comparable to that of airborne systems, the operational systems have resolutions on the order of tens of meters which, in Grand Canyon, would often yield riparian vegetation strands of two or fewer pixels in width (Bohnenstiel and Weber 1999). Second, these data will be available through government channels without letting a contract to collect them.

In the next sections we will describe the three remote methods mentioned above. We first describe the theory behind the technology. We then present the case studies of their application to vegetation questions, and include information on the types of vegetation, topography, and other conditions encountered by investigators where they have been applied. We also distill what we can from the studies on the strengths and weaknesses of method, where they would be useful to addressing GCMRC information needs and in what ways they would fall short. In the final section we provide general advice on the use of remote methods in vegetation monitoring based on the literature review and our own experiences. This includes both positive direction on where such methods are likely to enhance the GCMRC mission and caveats.

Multispectral Methods (Color / Color I.R.)

Many studies have been conducted for GCES and GCMRC using information from color and false-color infrared images (Warren et al. 1982, Pucherelli 1986, Waring 1996, Kearsley and Ayers 1999c). A large-format fixed-image camera using panchromatic or CIR film is mounted on an aircraft which collects an image of reflected light in the visible spectrum and portions of the near infra-red spectrum (400 - 1050 nm). Reflectance in these spectra is based on two factors. First, visible spectral reflectance is determined by the abundance of chlorophyll a and b and various carotenoids which are in turn determined by the species involved and the degree of environmental stress. Reflectance in the near-infrared spectrum depends on the size and orientation of intercellular air spaces (Lillesand and Kiefer 1987). Interpretation of the images as to vegetation types is based on field inspection of a limited number of areas. Based on distinct color and / or textural properties, qualitative inferences can be made regarding the presence and abundance of various vegetation types.

Creation of vegetation maps is possible given the presence of adequate numbers of ground georeference points with known coordinates to control for parallax distortion (Lillesand and Kiefer 1987). Scanned images are then rectified using these points with appropriate software (e.g. ARC-INFO, GRASS). From these rectified images, more quantitative assessments about vegetation abundance are possible (Warner and Katibah 1981).

In the Colorado River corridor of Grand Canyon, many studies have used these methods during the past 30 years. The first vegetation map of the river corridor used semi-rectified black and white photographs (Phillips et al. 1977). Early studies of recreational impacts on campsites came from similar methods (Green, L.T.,III 1978, Green, L.T. et al. 1980), as did assessments of the extent of native willow communities (Brian 1982). Pucherelli (1986) and Waring (1996) based assessments of the impacts of dam releases on riparian vegetation abundance on orthorectified black and white photographs. Later researchers began using rectified color aerial images to document changes in the abundance of specific vegetation types in a dozen specific sites (Stevens and Ayers 1995, Stevens et al. 1995, Kearsley and Ayers 1999a, b, c).

A technological step above these methods is the use of digital sensors to collect data on visible and near-infrared reflectivity of vegetation. Fixed image cameras using charge coupling device chips (CCD) or video recorders with sensitivities into the infrared range have been used in vegetation applications. Rangeland condition and the spread of insect outbreaks in native vegetation have been assessed using digital color-IR video (Everitt et al. 1992, 1997a, b). Senescence and the loss of trees in riparian gallery forests in the lower Rio Grande were documented using similar systems (Leonard et al. 2000). The invasion of native scrubland vegetation has been assessed using a digital color-IR camera (Stow et al. 2000).

Digital multispectral sensor systems combine spectral reflectance information with positional information from kinematic global positioning systems (KGPS) which determine the position of the sensor and some variety of an inertial navigation system (INS) which records the orientation of the sensor. Theoretically, no further georeferencing for image rectification is required because pixels are all associated with spatial information.

Current multispectral sensors have high spectral resolution (< 10nm) and are capable of sampling 15 or more distinct wavelengths simultaneously. The theoretical basis for these systems revolves around interspecific differences in the concentration of photosynthetic pigments and leaf ultrastructure. Based on these types of differences, if one looks at reflectivities in enough different wavelengths, one should be able to determine a spectral "signature" for each species. Classification of different species is based on either linear discriminant functions or neural network algorithms which use pixels of known identity to classify the rest.

Bohnenstiel and Weber (1999) mapped riparian vegetation in Blue Canyon on the Hopi reservation using multispectral data from the airborne terrestrial applications sensor (ATLAS) combined with some satellite data. Airborne data had resolution of ca. 2.5 m, which was adequate for a supervised classification of the vegetation categories they worked with. Digitized color IR photographs were used in an extensive mapping project in the Everglades which eventually covered 1.2 million hectares of wetland, scrubland, and forest types (Doren et al. 1999, Madden et al. 1999, McCormick et al. 1999, Welch et al. 1999).

There are several strengths of the color-IR / multispectral method. First, they are visually interpretable – it is an easy thing to relate texture and pattern in hard-copy or digital images to individual plants in the field. With more experience comes a greater ability to interpret images. Furthermore, relatively untrained observers can be given visual "keys" to interpret new images (e.g., Madden et al. 1999). For digital data, classification of each pixel is possible, based on field-derived "signatures" of each species (Stow et al. 2000). Therefore many years worth of images could be classified based on libraries of images or individual pixel signatures, regardless of personnel turnover.

The weaknesses of color-IR and multispectral methods arise from rectification routines and variability. Rectification, even when based on upwards of 25 ground control points, results in positional uncertainty of 5 - 10 meters when checked against actual coordinates in 1:10,000 images (Welch et al. 1999). Spectral variability within species, and even within individuals (Cochrane 2000) caused by differential illumination, cloud cover, and microclimate can result in automatic classification schemes giving different classifications of pixels within an individual (Stow et al. 2000). The sensitivity of these methods to cloud cover and shadowing (Lillesand and Kiefer 1987) further reduces the utility of color-IR methods in Grand Canyon, given that the maximum leaf-on period, which has been suggested for the time of image acquisition by the review panel (Berlin et al. 1998), coincides with the period of maximum rainfall in Arizona.

Hyperspectral Imaging Systems

Hyperspectral imaging systems can be thought of as an extension of multispectral sensors.

Positional information from KGPS and INS units are combined with spectral reflectance data so that rectification is part of the data. Rather than collecting data at discrete wavelengths which are known or suspected to be of use in differentiating among species, hyperspectral sensors collect reflectance information across a continuous section of the electromagnetic spectrum. Usually reflectance data are collected between 300 and 2000nm, although below 450nm and above 1400nm, returns are often so low that these are discarded.

Interpretation of species or vegetation type abundance is based on either many wavelengths or the "shape" of the reflectance curve. Investigators have the luxury of examining the entire spectrum for bands which might allow the best separation of species or vegetation types (Cochrane 2000). Alternatively, the first or second derivative of the reflectance curve can be used to describe different reflectance responses for different species across sensitive areas of the curve, especially near the "red edge" at which reflectance goes from low values in the far red (high absorption) to high values in the near infrared (high reflectance)(Chen et al. 1998). As with multispectral methods, either linear discriminant methods or neural network algorithms can be used to classify pixels after a suitable training data set is created (Gong et al. 1997, Martin et al. 1998).

In Grand Canyon, hyperspectral data from low-altitude AVIRIS (airborne visible / infra-red imaging system) flight were used for classifying 8 types of terrestrial vegetation in the Glen Canyon reach along with some aquatic and geomorphic setting types (Floyd and Anderson 1982, Sutherland 1996). Data from other AVIRIS missions were used to detect seasonal dynamics in arid shrub- and grassland areas near Mono Lake by taking advantage of the sensitivity of the "red edge" such that vegetation with cover as low as 5% could be monitored (Chen et al. 1998). Hyperspectral data from another AVIRIS mission at the Jornada LTER site was part of a satellite data prototype validation (PROVE; Privette et al. 2000), although the results of that study are not yet available. Cochrane (2000) and Gong et al. (1997) collected hyperspectral data under more controlled conditions (hand held units with either intact trees or leaves from tall tree canopies) to show that it was possible to discriminate among tree species based on hyperspectral signatures. Sandmeir and Deering (1999) applied a bi-directional reflectance function to hyperspectral data to describe the physical structure of boreal forests with varying compositions.

Hyperspectral sensor systems have several advantage over multispectral systems. First, by using the entire spectrum, investigators have access to more information in the form of more bands which can be selected on the basis of a visual inspection or some automatic / numerical criteria (Cochrane 2000). Theoretically, differences among more band should yield better separation of species or vegetation types. Second, by using the"shape" of the reflectance curve, especially in areas like the "red edge", the sensitivity of the sensors can be extended (Chen et al. 1998).

However, these systems suffer from the same weaknesses as multispectral systems. Positional errors from the KGPS and INS will still be present. Furthermore, variation is an integral part of natural systems, and increases in resolution and detail will only yield more variation so that no species will have all its members within a unique "shape space" which can be differentiated from all others (Cochrane 2000). For example, when Cochrane (2000) attempted to adjust his classification criteria so that he included all spectra from individuals of mahogany, he failed to exclude 7 of 31 non-mahogany tree spectra. By excluding nearly all non-mahogany trees (only 2 of 31 inclusion mistakes), he misclassified roughly 25% of the mahogany individuals as non-mahogany (Cochrane 2000). Furthermore, the inclusion of bands which are not based on photosynthetic pigments or other plant characteristics allows noise from background sources such as soil, litter and water to affect spectral signals and the impacts of shadowing and scatter from deeply incised canyon walls on hyperspectral data is not known (Merényi et al. 2000a, b).

Lidar systems

Active laser imaging systems, or lidar (light detection and ranging) systems, used for vegetation studies come in two basic varieties. First, the fluorescence of leaves when excited by particular wavelengths of light gives an indication of the health of the plant (Cecchi et al. 1994, Méthy et al. 1994). Second, by taking extremely accurate measures of the return time of pulsed lasers which emit light in the near-IR region, distances to objects can be calculated. This is the basis of laser altimeters and ranging devices (Mamon et al. 1978).

The altimeter model has been used for documenting canopy structure and biophysical properties. Because some of the light returns from reflection from the top of the canopy and other amounts make it through varying levels of the canopy or to the ground before being reflected, the difference between the first and peak return can be used to determine the relative distances to the top of the canopy and the ground. The difference between these is the canopy height. Early systems used relatively small "footprints" between 0.3 and 3.0 meters (Mamon et al. 1978, Leckie 1990) and relied on data from multiple footprints to create a picture of the entire canopy. More recently, large-footprint (10 - 25m) systems, such as SLICER (scanning lidar imager for canopies by echo recovery) and LVIS (laser vegetation imaging system), have tested the ability of the sensor to recover the complete return so that information on the canopy top, forest floor, and layers within the canopy are available within the same footprint (Lefsky et al. 1999a, Weishampel et al. 2000). Within-canopy structure can be determined by the proportion of the total above-ground return from each level of the canopy, usually divided into 30 - 100 cm strata. As with other sensor types, canopy information is typically coupled with positional information from KGPS and INS equipment.

Most large footprint lidar applications have been done in forest settings. Quadratic mean canopy height data from SLICER predicted 70% of the variability in basal area and 80% of the variability in above-ground biomass in deciduous forests of eastern Maryland (Lefsky et al. 1999b). The same data was able to account for 90% and 75% of the variability in biomass and leaf area index (m^2 of leaf area per m^2 of ground surface) in Douglas fir and hemlock stands in northwest forests (Lefsky et al. 1999a). In the western Cascades of Oregon, a SLICER unit predicted height, above ground biomass, foliar biomass, and canopy cover with R^2 values in excess of 0.84. With a very low elevation test of a scanning lidar unit, Nilsson (1996), underestimated canopy heights of an even-aged stand of Scots pine by 2 - 3 meters, although this may have been a positional problem rather than a scanning problem. In tropical rainforests of Costa Rica, where leaf area index values run in excess of 10, LVIS data on canopy height, gap area, and mean foliage height was in good agreement with published values from studies done in the same area (Weishampel et al. 2000). In a slightly more esoteric application of SLICER data, Drake et al. (2000) showed that spatial patterns of height variation demonstrate both fractal and multifractal properties.

The main strength of lidar data is in its ability to document structural information from canopies across a broad area. This habitat structure, the physical arrangement of plant parts in space (Bell et al. 1991), is often more important to community patterns than the species present

(Mills et al. 1991, Noon 1999). Structure is often very difficult to measure in species-rich and productive habitats, or where canopy heights can exceed 50 meters which is precisely where lidar data would be most useful (Nilsson 1996, Lefsky et al. 1999a). That this can be done across a broad area in a relatively short time with little ground work makes it potentially even more valuable.

As with other sensors, the accuracy of the positional information is where the main problems arise. In none of the studies reviewed did the spatial accuracy have a better than 5m RMS error. In some cases, registration errors which arose when trying to tie ground GPS data to the positional data were as great as 15 meters. The other drawback with lidar data is that the return patterns from different canopy types (e.g., willow and tamarisk) will differ based on leaf type and canopy architecture, so that there must be calibrations done for a wide variety of vegetation types (Leckie 1990).

Common Threads

In reviewing the papers on remote sensing for this section and reports on vegetation monitoring which have used remotely sensed data for the earlier sections of this report, several common threads have emerged.

1. It is important to know what it is you want to measure before acquiring remotely sensed data. Collecting spectral information from throughout the river corridor will not answer any research or monitoring question unless it is couched in terms such that spectral data will answer it. Is it the abundance or spatial extent of a particular species which is in question, or is it more the distribution of a vegetation type which is important to some sort of wildlife? If it is the former, then some kind of hyperspectral data may be useful, although the thorough of within-species variability reviewed by Cochrane (2000) should caution against believing that any amount of spectral information will find a unique "space" occupied by a species. If it is the latter, it is critical that these vegetation types and the best way to collect spectral data on them be clearly defined before any data collecting begins. Further, there needs to be an acknowledgment that such decisions insert subjectivity into the analysis and will likely have impacts on the results of the study (i.e., pencil lines on a map are not necessarily any more subjective than decisions on the boundary of spectral signatures).

2. Know the limitation of the sensors you will employ and how these relate to the goals of the remote sensing program. For example, if remotely sensed data is to be collected at the"maximum leaf-on" period (Berlin et al. 1998), in Arizona this will coincide with the time of greatest precipitation which comes in the form of convective storms with attendant mid-day cloudiness. If the sensor being used, like color / color-IR multispectral sensors or hyperspectral sensors, is subject to errors which arise from shadows and shading, this may create timing problems which require short-notice changes to flight schedules (see Privette et al. 2000). Similarly, given that lidar return varies with canopy and leaf form (Leckie 1990), it would become prohibitively expensive to calibrate lidar returns for several dozen vegetation types. It would make more sense to target specific vegetation types which are important to wildlife and use lidar data within those only.

3. "[I]t is important to note that variability is an normal factor in vegetation reflectance

and ... no amount of wishful thinking or technical manipulation is going to remove it....higher resolution data will have greater levels of variability. (Cochrane 2000 p.2087). If reflectance varies within individuals based on shading and within species in different sites based on age, canopy structure, ground cover, soil type, stress and shade, then how much variation will there be at the broader scale of vegetation types and how much supervision will be required to achieve adequate levels of accuracy? Pixels within the canopy of individual trees will be classified as different species (Stow et al. 2000), trees within a stand will be classified differently based on shading (Gong et al. 1997), and stands of the same species will be classified differently based on ground cover and soils (Cochrane 2000) when rigid, numerically based classifications are used.

4. Know the scale at which questions are being asked, and whether the remotely sensed data are appropriate at that scale. Given that registration errors common to lidar studies (Nilson 1996, Lefsky et al. 1999a, Means et al. 1999) and hyperspectral sensors (Gong et al. 1997, Sandmeier and Deering 1999) but that these types of data have a certain internal coherence (e.g., GCMRC lidar data claim 15cm horizontal precision, but unknown accuracy), this limits the form in which questions can be asked. For example, they can be used to address questions regarding the total amount of dense tamarisk or mixed deciduous scrub in the entire river corridor, or in selected geomorphic reaches. However, the registration problems will prevent them from being used to address the degree to which pixels associated with specific plots, identified with x – y coordinates, has changed because there is no certainty that coordinates in two successive sensor runs will measure the same physical points on the ground.

5. Remotely sensed data and products such as maps should be used as an adjunct to, rather than a substitute for, process-based studies. A system-wide vegetation map for baseline information on the relative abundance of vegetation types is a worthy use of resources. A second map, produced several years later, should provide some indication of the degree of change over that time. It does not, however, constitute monitoring. Rather the map should be used to calibrate effects measured in process studies. For example, the vegetation monitoring program scheduled to begin in FY 2001 will provide data on changes in several different vegetation types by geomorphic reach and stage level. Data in the vegetation map should be comparable in some way to the data from the stage-based study so that system-wide manifestations of effects documented in the latter can be calibrated.

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