Riparian Vegetation Monitoring in the Colorado River Corridor 1998 to 1999

Final Report

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Table of Contents

List of Figures

List of Tables

PREFACE

Although the United States is officially on a course towards joining the rest of the world in embracing the metric (SI) system of measurement, old habits and ways of thinking change slowly in a place where 100,000 years is considered an eye blink. Because most references to places, and indeed many of the place names, in the Colorado River corridor of Grand Canyon are based on river mileage downstream of Lees Ferry, we use miles for designation of distance along the river corridor (river miles, or RM in this report). Similarly, because dam managers and those who have worked in the river corridor have historically referred to discharge from Glen Canyon Dam in terms of thousands of cubic feet per second (kcfs), we will do so in this report. Conversions to SI equivalents are straightforward: cubic meters per second (CMS) = kcfs * 0.0283 and river kilometers (rKm)= river mile * 1.61. All other measurements of plants and habitats are in SI units.

There are many sources for determining plant identities in the Southwest. Often, these give conflicting names, depending on differences of opinions of the authors of the floras, taxonomic revisions more recent than the source, and so on. We follow the naming conventions in Kartesz and Kartesz (1990) in this report. We commonly use acronyms for plant names to speed up our data collection. To make the interpretation of the data files appended to this report possible, we have provided a glossary of these abbreviations in Appendix B which also serves as a list of species encountered in this report.

THIS IS A DRAFT REPORT. The analyses and conclusions contained in this report are provisional and therefore subject to revision. Neither should not be relied on or cited until the final version of this report is produced.

ABSTRACT

We monitored riparian vegetation at 11 sites in the Colorado River corridor of Grand Canyon between Lake Powell and Lake Mead and along the shoreline throughout the section between Lees Ferry and Diamond Creek. We based our sampling and analysis on vegetation polygon maps developed from data and analyses in those previous years. We had sampled nine of these sites in between 1994 and 1997.

We measured vegetation changes in a number of different ways. For the whole corridor measurements, we compared the number of, total area of, and percent of shoreline occupied by vegetation below the 33,000 cubic feet per second (33 kcfs) stage elevation in 1999 to that in 1998. For site-based data, we first classified polygons as being wetland, mixed riparian scrubland, dense tamarisk scrubland, bar top or "other" vegetation. We then compared mean total foliar cover per polygon , mean species richness per polygon, and mean Shannon diversity per polygon by vegetation type in the eleven sites in 1997 and 1996 to check for overall patterns of loss or gain. Second, we used polygon areas from our GIS vegetation maps changes in the areal extent of the four main vegetation types. Third, we compared the areal extent of important species found in riparian wetlands: *Typha domingensis, Phragmites australis*, combined taxa of *Juncus*, *Scirpus*, and *Carex*. Finally, we compared the 1997, 1998 and 1999 whole-site complements of species the eleven sites to assess the level of species turnover.

Our results showed a complex pattern of stasis with some fluctuations which did not lend itself to summarization by a single measure. We found no change in the total area or length of vegetated shoreline between 1998 and 1999. However, this vegetation was divided among more patches in 1999 vs. 1998 reflecting, we believe, the effects of higher winter flows in early 1998 which "homogenized" patches by removing most of the cover and, consequently, most of the apparent differences between patches.

We were unable to detect a change between 1997 and 1999 in total foliar cover for wetland, dense tamarisk, and bar top polygons. Gains in cover in 1998 in wetland polygons were offset by losses in 1999. We did find an overall decrease in cover in mixed riparian scrubland polygons in 1999. However, this disagreed with more reliable data from the total vegetation volume data set, and we do not believe that the losses represent more than differences between observers in 1999 and 1998.

Nor were there consistent or significant changes in the areal extent of riparian patch types. Patches in some sites increased slightly in size, but others either decreased in size or were unaffected. The total areal extent, measured as percent cover multiplied by polygon areas, covered by the three important wetland species did not change significantly either. There was a considerable amount of species turnover between 1997, 1998 and 1999. Overall species richness within for each of the four patch types did not change between years, despite changes in the composition measured by species turnover. Nor did Shannon diversity (H') differ between years; measures in the two years differed by less than half a percent indicating that patterns of species abundance did not change.

We also measured changes in the physical structure of vegetation by comparing the total vegetation volume (T.V.V.) of live leaves and branches in 1997 to that in 1998 in each of the four vegetation types. Using a repeated measures analysis of variance with polygons of each type pooled within sites, we showed significant time effects in all the vegetation types. All types

had a significant rise in the total number of decimeters "occupied" in 1998, and wetland, bar top and mixed scrubland polygons had non-significant decreases in 1999. TVV measures in dense tamarisk scrubland patches increased in 1999.

We measured the adequacy of our sampling intensity in a number of ways, depending on which measure was being considered. For vegetated shoreline, we used a bootstrap method to examine the effect of sample size on the standard deviation of our estimates of number and total area of vegetated patches and percent vegetated shoreline. This analysis showed that the variance stabilizes at a level of intensity about half to three-quarters of our current effort. For vegetation sampling in our polygons, we used a bootstrap compositional dissimilarity analog of the species area curve on polygons we had over-sampled to find the number of samples at which the mean dissimilarity of remaining samples to the composite sample was below 0.30 as measured by the Bray-Curtis distance measure. The data showed that for wetland and both scrubland patch types, three samples was more than adequate. Bar top patches were sparsely and patchily vegetated and required four or five samples to achieve the same results, but these are all within the current level of sampling. For total vegetation volume, we used an method similar to that used with the vegetated shoreline data, and found that ten samples were all that were required to stabilize the variance of the TVV estimate.

We also measured or noted the distribution of target species in the river corridor. No federally listed or candidate species appeared in our sites, in the channel margin habitats we examined, or in any of the other areas we visited during our field work. Nor was there a detectible change in the distribution of target invasive exotic plant species, *Eragrostis curvula*, *Erianthus ravennae*, *Alhagi camelorum*, or *Lepidium latifolium*. They did not appear in any of our study sites in which they had not been previously found, nor were there significant within-site changes in their distribution.

We measured the elevations of transect thalwegs in return-current channels and on low sandbars in 1998 and 1999 and compared them to elevations in 1997. These transects, through marshes which had been studied in previous studies since 1991 showed a small amount of change in most cases, Most of this activity occurred at the upstream end of sandbars, and near the mouths of return-current channels.

Because of the complex nature of the changes in the riparian vegetation during the past year, we recommend that monitoring continue at a level at least as detailed as that described in this report. As patches of vegetation change in size and species composition, reflecting their continued adjustment to conditions created by the experimental flood in 1996 and other dam operations, managers should have access to information whose scale can be adjusted from the species level to a site-wide perspective. We believe that the whole-site approach of the vegetation mapping we describe here is a better method than one involving permanent quadrats, in that it is sensitive enough to detect species turnover, yet robust enough to describe changes in the distribution of multi-species vegetation types. We further recommend the exploration of methods which will allow calibration of cover estimates between years, as this is one area which can cast doubt on results of monitoring studies.

RIPARIAN VEGETATION MONITORING IN THE
COLORADO RIVER CORRIDOR 1998 TO 1999

INTRODUCTION

The effects of dam construction and flow regulation on riparian plant communities are complex, variable, often indirect, but generally negative. Stabilization of annual floods can allow the establishment of diverse plant communities in areas previously scoured (Carothers and Aitchison 1976, Turner and Karpiscak 1980, Stevens et al. 1995), although many species which enter newly available riparian habitats are exotics (Hunter 1988, Décamps et al. 1995). The dewatering of habitats by flow reduction, on the other hand, decreases the growth rates of and causes mortality in mature plants and prevents the germination seeds of many riparian species (Smith et al. 1991, Stromberg 1993, Stromberg and Patten 1996). And although higher river flows are generally associated with increased vigor and survival of plants in riparian habitats (Stromberg and Patten 1990, Smith et al. 1991, Auble et al. 1994) the effects of flow augmentation on riparian species are not always positive (e.g., Stromberg and Patten 1992). Further, the loss of nutrient-rich and moisture-retaining fine soils due to fluctuating flow levels can lead to a loss of species diversity through changes in germination sites(Nilsson et al. 1997).

Although riparian plant species are assumed to be adapted to the effects of flooding, the responses to high flows and recovery of riparian habitats after flooding vary significantly with flood intensity and duration and the recent history of the habitats examined. Degraded habitats which are subject to large flooding events generally lose most of their vegetation, and do not recover quickly (Platts et al. 1985). More intact habitats subject to high flows show initial losses of vegetation, but tend to recover more quickly (Pucherelli 1986, Stevens and Waring 1986).

Additionally, the degree of response of riparian habitats to either flow regulation or disturbance will vary considerably with the type of measurement taken. For example, flooding nearly always causes an initial loss of foliar cover in riparian habitats (Pucherelli 1986, Stevens and Waring 1986, Stromberg et al. 1993, Kearsley, M. J. C. and Ayers 1996b). However, when simultaneous measurements are taken of other important indices of habitat integrity, there is often no consistent effect of either flooding or flow regulation on species diversity (Kearsley, M. J. C. and Ayers 1996b, Nilsson et al. 1997), leaf litter accumulation (Malanson 1993, Molles et al. 1995, Kearsley, M. J. C. and Ayers 1996b) or the extent of wetland patches (Kearsley, M. J. C. and Ayers 1996b).

Problem Statement

The development of riparian communities along the Colorado River in Grand Canyon below the 125,000 cubic feet per second (125 kcfs) stage elevation was the direct result of flow stabilization by Glen Canyon Dam (Carothers and Aitchison 1976). Within this zone, a variety of vegetation types have developed in response to the interaction of biotic and abiotic factors including geomorphology, hydrology, biogeography, and the vagaries of history and human dam management (Carothers and Aitchison 1976, Schmidt and Graf 1988, Stevens et al. 1995, Kearsley, M. J. C. and Ayers 1996a, Webb 1996). In the Colorado River corridor, habitats are generally divided between the old high water zone, above the 125,000 cubic foot per second (125 kcfs) stage elevation and the new high water zone (between ca. 5 kcfs and 125 kcfs). New high water zone habitats are further divided, depending on elevation and geomorphic setting, into

wetland / marsh, low riparian areas of channel margin habitats, and upper riparian zone areas of debris fans and sand bar tops (Kearsley, M. J. C. and Ayers 1996a, p. 5).

Riparian vegetation and associated wetlands have been identified as one of the principal resources of concern along the Colorado River corridor downstream of Glen Canyon Dam (Bureau of Reclamation 1995, National Research Council 1996, pp. 96-110). The areal extent, as well as floristic composition, of riparian vegetation has therefore been of special concern to the agencies managing these areas and water releases from Glen Canyon Dam (e.g., National Park Service, Hualapai Tribe, Bureau of Reclamation). As in other parts of the arid and semi-arid southwestern U.S., riparian areas support a disproportionately large proportion of plant and wildlife biomass and diversity (Brown et al. 1983, Knopf 1985, Stevens et al. 1995). The large losses of riparian habitats in this area due to human activity (Kusler 1985) makes this habitat type even more important.

The operation of Glen Canyon Dam has direct and indirect impacts on riparian and wetland vegetation in the river corridor. The reduction of the upper flow limits since 1991 has caused wetland vegetation patches to contract and reduced the input of fine soils to areas formerly dominated by wetland vegetation (Kearsley, M. J. C. and Ayers 1996a, Kearsley, M. J. C. et al. 1996). In addition, the scour and burial of riparian habitats during the 1996 Beach / Habitat Building Flows further modified these important areas.

Monitoring Questions

The work described in this report was aimed at monitoring the riparian vegetation of the Colorado River corridor of Grand Canyon. The specific areas of interest fell under one of three general areas.

1. Floristics / Vegetation: Did riparian plant assemblages in the new high water zone change during the year? This question was broken down into four specific questions.

    A. Were there changes in the distribution and abundance of near-shore vegetated patches which are preferred habitats of juvenile native fishes (Converse et al. 1998). We tested null hypotheses of no change in the number, total area, and percent of shoreline occupied by vegetation in 11 geomorphic reaches between Lees Ferry and Diamond Creek.

    A. Were there changes in the areal extent of certain assemblage types (e.g. wetland or riparian woodland / scrubland patches) during the transition year? We tested a null hypothesis of no change.

    B. Were there changes in the areal extent of individual obligate wetland species? We tested a null hypothesis of no change in three major wetland taxa : *Typha domingensis*, *Scirpus* spp. and *Juncus* spp.

    C. Within a site, was there a significant species turnover during the transition year? Using complete species lists generated during censuses, we looked for evidence of species loss and colonization on a site-wide basis.

2. Structure: Did the physical structure of riparian habitats changed during the transition year? We looked for the signs of succession in three ways.

    A. Were there significant changes in the vertical foliage distribution of an assemblage type? We tested a null hypothesis of no change within polygons.

    B. Did transition year flow patterns cause changes in the physical layout of return current channels? We tested a hypothesis of no changes in the surface elevation of the thalweg of permanent, georeferenced transects through the return current channels.

3. Target Species: Did the distribution of target species change in our study sites during the

transition year? We worked with significant exotics which we have been following, and looked for the appearance of Federally listed or candidate natives.

A. Were the ranges of exotic species expanding? We counted the number of patches where four significant species were found, and the number of our 11 monitoring sites in which these species occurred.

B. Did listed or candidate species appear in the riparian zone in our study sites? We looked for any of these listed species both in our sites and in areas visited incidentally during random stops, lunch stops, and so on.

METHODS

Study Sites and Field Trips

Site-specific data on vegetation were collected from 11 long-term study sites. Sites were selected in an earlier study (Stevens and Ayers 1995) on the basis of several criteria, including long-term stability, diversity of habitats, and integration with other studies. Table 1 lists the study sites and other relevant information.

Table 1. Vegetation monitoring sites.				
River Mile and Side	Site Name	Geomorphic Reach[+]	Special GIS Study Site[++]	Other Studies[+++]
-6.5 R	Hidden Slough	N/A	14	M,V,S,A
43.1 L	Anasazi Bridge Camp	Lower Marble Canyon	3	M,V,G,S,F
51.2 L	Unnamed Camp	Lower Marble Canyon	4	M,V,G,S,F
55.5 R	Kwagunt Marsh	Lower Marble Canyon	4	M,V,G,C,F
68.2 R	Tanner Beach	Furnace Flats	5	M,V,S,F
71.4 L	Cardenas Marsh	Furnace Flats	5	M,V,A,F
93.9 L	Granite Camp	Upper Granite Gorge	6	M,V,S,F
122.8 L	Forster Camp	Aisles	7	M,V,S,F
194.1 L	Hualapai Acres	Lower Canyon	10	M,V,G,S,F
209.0 L	Granite Park	Lower Granite Gorge	11	M,V,F
249.5 L	Below Lost Creek	N/A	N/A	M,V,A,F

[+]Geomorphic Reaches as per Schmidt and Graf 1990
[++]Special GIS study sites designated by Bureau of Reclamation / GCES
[+++]Codes for other studies: M = Vegetation mapping, V = Interim Flows vegetation study plots, G = NAU groundwater studies, S = NAU sandbar studies, C = ASU climatology / productivity studies, A = Avifauna studies, F = Beach / habitat building flow vegetation studies

Vegetation and plant measurement data were collected on a series of field trips in 1998 and 1999. Fish habitat (near shore vegetation) and springtime total vegetation volume data were collected on down-river trips from April 12 - 24, 1998 and April 15 - 27, 1999, and upriver trips to the Glen Canyon site (-6.5 R) on May 17, 1998 and May 18, 1999. Polygon vegetation data and fall total vegetation volume data were collected on down-river trips on September 3 - 17, 1998 and September 2 - 16, 1999, upriver trips to the Glen Canyon site on October 7, 1998 and September 30, 1999, and one-day trips from Pearce Ferry to the Lower Canyon site (249.5 L) on October 19, 1998 and October 14, 1999. Figure 1 shows the timing of the trips relative to the hydrographs of water years 1998 and 1999. Personnel included professional botanists from
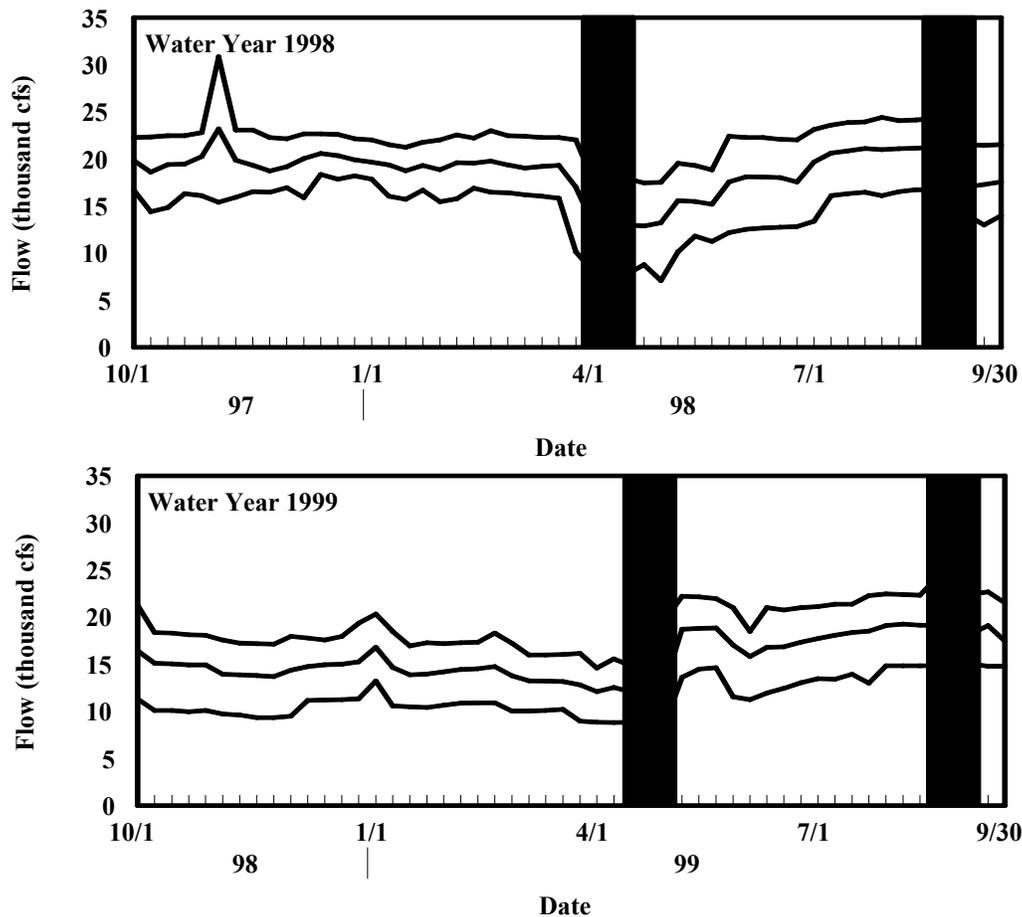
1. Hydrograph of water years 1998 and 1999 at Lees Ferry, Arizona. Lines indicate weekly minimum, mean, and maximum flows. Shaded bars indicate the timing of major data collecting trips.

Northern Arizona University, the National Park Service, the University of Washington, and a local botanical consulting firm.

Data on beach topography used in calculating marsh transect elevation change data were collected by the N.A.U. Sandbar Studies Group on trips in April 1998 and May 1999. Digital elevation models were created in the Sandbar Studies office over the three months following each trip. Transect elevation data were pulled off the digital elevation models in August of 1998 and 1999.

Near-Shore Habitat Assessment

In order to measure the abundance of near-shore, vegetated habitat for fish, we collected data within the geomorphic reaches of Schmidt and Graf (1988). This system divides the Colorado River corridor into units defined by the bedrock unit through which the river is cutting. Softer rock layers produce wider canyon sections in which sandbars are larger (Schmidt and Graf 1988) and both terrestrial and aquatic components are more productive (Stevens et al. 1995, 1997).

Within each geomorphic reach we randomly selected 20 shoreline points to be measured. We used the mileage indicators and local topography in a commercially available river guide (Belknap and Evans 1989) to estimate the location of each point while we were on the river. At each point, we created a transect running downstream from the random starting point by laying

out a 50 meter measuring tape within 2 meters of the water's edge. We then determined the elevation of the upper end of current power plant discharges (30 kcfs) by noting the position of flotsam from recent high flows and the lower extent of woody vegetation, and by using the judgement of a highly experienced river guide.   We measured the elevation of the lower and upper limits of the vegetation in this zone above the water's edge with a survey rod and a clinometer. Finally, we measured the length, to the nearest meter, and width, to the nearest half meter, of all patches of vegetation between the water's edge and the 30 kcfs stage elevation.

After the 1998 trip, we analyzed the data on the mean number, total area, and total length per sample for sample size adequacy using a bootstrap routine. Using a resampling program (Resampling Stats; Simon 1995), we calculated standard errors of samples of size 2 to 20 for all 3 measures of near shore habitat abundance. We graphed plots of standard deviation versus sample size to determine the point at which we would gain a minimal amount of precision for increases in sample size.

To compare 1998 and 1999 data on the number, total area, and total length of near shore habitat by reach, we calculated means per 50-meter transect for each measure in each reach. We then compared estimates for the two years with the non-parametric Wilcoxon T test. Because samples were combined within each reach, our sample size was 11 for each measure.

Marsh Transect Elevations

To document changes in the topography in patches of marsh vegetation, we measured the surface elevations of transects in return-current channel and low-elevation bar marshes in 1998 and 1999 and compared them to 1997 elevations. These marshes had been censussed and surveyed twice per year from 1992 to 1996 (Stevens and Ayers 1995, Kearsley, M. J. C. et al. 1996). Sites had been selected as part of studies from GCES Phase 1 and Interim Flows studies. Within these sites, marshes had formed since 1987 either in return current channels or on top of low platform sandbars which had been periodically inundated. In return current channel settings, between four and 15 transects had been laid across the channel at approximately 10 m intervals from the head of the return channel to the mouth. On the low bar marsh (Kwagunt Marsh, RM 55.5 R), 1 m^2 plots were arranged at 10 m intervals along transects perpendicular to the river channel, with transects were laid out every 10 meters,.

Members of the GCES Survey Department or the N.A.U. Sandbar Study Group, using standard GCES Survey Department protocols, surveyed the entire sites, including these transects. In the office, a digital surface model of the entire site was generated using SDRmap software (Datacomm Software Research Ltd. 1997). We then used the coordinates of the end of the transects as endpoints of a "road" across the surface, and pulled surface elevations off at one meter increments on this road. In this way we were certain of having measurements from standard surface points which could be compared between years. We designated the lowest point on each transect as the thalweg.

To determine whether flow regimes and tributary inputs had created environments where significant deposition or erosion in marsh patches in return current channels, we compared thalweg elevations in 1997 with those in 1998 and 1999. We analyzed thalweg data for transects in all sites with a repeated measures ANOVA. When a significant time x site interaction was found, we analyzed changes in the thalweg elevation in each site separately using a Wilcoxon T with transect as the unit of measure.

For the platform bar marsh (55.5 R), elevations of grid points were taken from the digital surface model. We tested for changes between years using a repeated measures ANOVA in

which we blocked for transect effects. We also tested for interaction effects between time and transects.

Vegetation Maps

Since 1995, our overall approach to monitoring and measuring changes in riparian vegetation has been based on the behavior of vegetation patches at 11 long-term study sites in the river corridor. If the vegetation in a patch is determined primarily by the physical parameters affecting it, then measurable changes in vegetation should be reflecting changes in these factors. Furthermore, if patches of similar vegetation at a large number of sites change in the same way, it can be assumed that the factors which govern the distribution of that patch type are changing throughout the system. In the riparian habitats of the Colorado River corridor, the unifying factors are weather and dam-driven hydrology, with the latter being a more important one, given the patchy nature of precipitation in the area and the large geographic separation among our sites.

In 1998 and 1999, we delineated vegetation patches on 400% enlargements of the Labor Day 1:4800 color aerial photographs of our sites. Patches were delineated based on the color and density of vegetation present. Patch boundaries were checked during field work and were accepted, moved or deleted according to conditions we encountered. To determine patch areas and include the maps in the GCMRC GIS, the finalized maps were digitized and orthorectified using standard protocols. In this way, all patch areas were calculated in true plan view, and between-year comparisons of patch area could be made without the distortions caused by slight variations in photographic equipment or conditions.

Vegetation Sampling

To sample vegetation in patches during 1998 and 1999, we used the same methods employed between 1995 and 1997. We generated a complete species list for each patch during an initial set of traverses and boundary checks. We then selected three to five random points per polygon for sampling with a table of random numbers. At each point, we centered a 3-meter circle, and estimated percent foliar cover for all species within the sample. Species which were in the polygon but which were not picked up in any samples were assigned a trace cover value of 0.001%. As with previous data collections, current year annuals were included even if they were senescent. In situations where polygon shapes and sizes did not allow the standard sample, we either reduced the circle's radius or took half-circle samples.

We attempted to control sampling consistency among vegetation sampling teams in two ways. First, to reduce variation among sampling groups in cover estimates, all sampling groups sampled one or two polygons together and come to a common set of cover estimates for species within the polygons. Because overestimation of cover at low levels (< 10%) is generally the greatest source of among-group variation (Kennedy and Addison 1987), we used a 53 cm x 53 cm square frame as an approximate "standard" for 1% cover during these group estimates (0.281 m2 / 28.3 m2 . 0.99 %). We also rotated crews, so that all recorders worked with all readers at least once during the trip, to control for the effects of among-crew differences.

To measure foliage vertical structure, we used a modification of the total vegetation volume (TVV) method of Mills et al.(1991). At 10 to 20 locations per patch, we recorded the number of 1–cm. Increments in each vertical meter in which live vegetation contacted or was within 10 cm. of a survey rod. To avoid problems with the lack of independence, we used random locations for each point, rather than running transects which the original authors had (Mills et al. 1991).

Vegetation Analysis

Underlying most of our analyses was the classification of polygons into one of four basic

types: wetland, tamarisk scrubland, mixed/willow scrubland, and open bar-top. These types represent important ecological and management elements in the river corridor. Wetland patches are important centers of productivity and biodiversity in Grand Canyon (Carothers 1977, Stevens et al. 1995). Dense tamarisk and mixed / willow scrubland patches provide important nesting habitats for neotropical migrants (Brown et al. 1983, Brown 1992, Sogge et al. 1998). Bar top habitats are important for camping in the river corridor (Kearsley, L. H. et al. 1994, 1999).

To classify vegetation polygons, we analyzed data from 1995, through 1999 simultaneously within each site. We classified the vegetation in each polygon in each year using two-way indicator species analysis (TWINSPAN; Hill et al. 1975, Hill 1979) and a post-classification adjustment. TWINSPAN is often used for the classification of vegetation data (see reviews in Gauch 1982 and Legendre and Legendre 1998) and has been used in this system in previous studies (Kearsley and Ayers 1996a, Stevens et al. 1996). We examined the TWINSPAN divisions after six levels, and ignored those which did not make biological sense (e.g., 10 % versus 20 % cover of annual bromes). Because some data structures can lead to problems with clustering by TWINSPAN (Belbin 1987, Belbin and McDonald 1993) we then examined the classification with a similarity-based ordination of the samples (NMDS, non-metric multidimensional scaling ; (Kruskal and Wish 1978, Faith et al. 1987). Within the framework generated by the TWINSPAN analysis, samples were moved among groups based on vegetation affinities which the original analysis failed to uncover. The result was the classification of polygons in each site into four or five distinct types within each site. New high water zone types were used in the analyses which follow. These types were then assigned to one of the four basic new high water zone patch types. We wish to point out that the classification of vegetation types in this report is entirely *ad hoc,* and was done solely for the analysis in this report. Because the scope of our project is limited to the lower-elevation areas which are more strongly influenced by dam operations, we did not include polygons dominated by obligate- or facultative upland species (per Reed 1988) in analyses (e.g., desert scrub or old high water zone areas).

To make our vegetation data more readily applicable to other areas, we had our classification cross-walked to a standard regional vegetation scheme. The SRFR Classification is specific to the Colorado Plateau (Spence et al. 1995). Floristic units are based solely on the species present at the time of classification, with no influence of geomorphology or potential vegetation. The primary author of the scheme (J. Spence, National Park Service, Glen Canyon N.R.A.) provided the cross-walk for us, based on indicator species (sensu Dufrêne and Legendre 1997) for the groups and the examination of a table of samples by major species we provided.

We first needed to ensure that we were sampling the polygons adequately. We tested this by oversampling a series of patches of the four basic types and computing an analog of the species-area curve. We took between 7 and 20 samples in selected wetland, bar top, tamarisk scrubland and willow scrubland patches. We then used a resampling routine in PC-ORD (McCune and Mefford 1999c) to randomly select one plot, two plots, three plots and so on. For each sample size, the average quantitative Si rensen distance between the composite sample to any potential other sample was calculated. When this "marginal dissimilarity" reached 0.30, a degree of dissimilarity expected among replicate samples of the scale at which we were working (Kortekaas et al. 1976, Neuhäusl 1977, Lausi and Feoli 1979, Komárková 1980), we assumed that further sampling would not yield useful information. We compared the number of samples required to bring the marginal dissimilarity to 0.30 with our practice of censussing from 3 to 5 plots per polygon.

In order to measure gross changes in 1998 and 1999, we first compared total foliar cover, species richness (S), and vegetation diversity (Shannon H') of each polygon in to values in 1997. We used the Data Summary function of PC-ORD (McCune and Mefford 1999a) to generate polygon total cover, S, and H' estimates for each year. Patches which were totally eroded or under water in a year were assigned total cover, S, and H' values of 0 %. To test for overall effects on each of these three measures across two years of censussing, we used a repeated measures ANOVA. In cases where the sphericity criterion was not met, we used the Geisser-Greenhouse adjustment to the degrees of freedom for the F-test (Muller and Barton 1989). Finally, because we were testing for changes in three measures which can be intercorrelated (Magurran 1988, Buzas and Hayek 1996), we used a Bonferroni corrected alpha of 0.017 (= 0.05/3; (Sokal and Rohlf 1995, p. 240).

We also compared the areal extent of four general types of polygons: those dominated by wetland species, two types dominated by woody riparian species, *Salix exiguua, Tamarix chinensis*, and polygons with open bar-top vegetation (sparse cover with *Dicoria* and *Corispermum*). Our first focus was on polygons dominated by obligate wetland species such as *Typha, Phragmites, Juncus*, and *Scirpus*. Within each site we computed the total area of polygons classified as "wetland" in 1997, 1998 and 1999. If a polygon was classified as wetland in all years, we used area data from each year in the analysis. If a polygon changed to another type in 1998 or 1999 or changed to a wetland type polygon from something else in 1997, or if it was eroded or under water in a given year, it was assigned a area of "0" in those years. To avoid problems with spatial dependence, we summed areas for all polygons within sites before the analysis. Because the data to be used were matched by site, and because we could not make assumptions about the distribution of the data we used a non-parametric paired comparison, the Wilcoxon T, to compare the 1998 to 1997 data, and 1999 to 1998 data.

We used a similar method for comparing the areal extent of both woody riparian vegetation patch types and the open bar top patches in our sites. Using the same classification from above, we would compare data on polygons which were clustered together based on their dominated by willows, tamarisk, or the grasses and annuals common in the "beach" habitats . Again, polygons which were classified as a different type were given an area of "0" in the appropriate year. We pooled data within sites and used a Wilcoxon T to compare data from 1997 to 1998 and 1999.

Because univariate comparisons can yield misleading results when actual changes are multivariate in nature, we also compared site compositions across the three years. In order to test for shifts in the relative abundance of the four vegetation patch types, we summed areas of patches of each type within sites to get site totals by type. We then used blocked MRPP (MRBP: Mielke and Berry 1982, McCune and Mefford 1999b) to test for compositional changes across years while blocking for site effects.

As a further check on changes in wetland patches, we compared the distribution of six important wetland taxa, *Carex, Equisetum x ferrisii, Juncus, Phragmites australis, Scirpus,* and *Typha domingensis*. Species of *Carex, Juncus* and *Scirpus* were lumped within genera for this analysis. We calculated the total areal extent of each of these by multiplying the area of each polygon in which they occurred by the percent foliar cover estimate for that species in that polygon. Again, to avoid problems with spatial dependence, we summed across all such polygons within sites and used these pooled area estimates for testing. We used the paired, non-parametric Wilcoxon T because of the small sample size and the extreme non-normality of the data.

We measured species turnover by compiling a complete species list for the years 1997, 1998 and 1999 in each site. We then calculated the similarity between the species composition of the two years with the qualitative (presence / absence) version of the Sorensen similarity index. The index is relatively robust to changes in sample sizes (Faith et al. 1987) and has been used in this system in previous years for monitoring and to test for the effects of the experimental flood (Kearsley, M. J. C. and Ayers 1998, 1999). To address the question of how important these transient species were, we noted the maximum percent cover value for each transient species in the year in which it was present. No statistics were applied to these numbers. Rather, we used the data to generally assess patterns of species turnover.

To test for growth and successional types of vegetation changes, we compared the vegetation vertical structure of polygons in 1997 to that in 1998 and 1999. Although we collected data in the manner described by Mills et al. (1991) as the TVV method during this study, we had to compare the data to numbers collected in different sets of height increments. Where the TVV method collects data from each meter increment from 0 - 1 up through 6 - 7 meters and 7 meters and above, the 1995 - 1997 data were collected in just five increments: 0 - 0.3 m, 0.3 - 1.0 m, 1.0.- 2.0 m, 2.0 - 4.0 m, and above 4 m. Therefore we could directly compare the <u>total</u> density of vegetation as the sum of all contacts, which we did using a repeated measures ANOVA. Because there was a discrepancy between T.V.V.-based and total vegetative cover based conclusions regarding density changes between 1997 and 1999 in mixed riparian scrub patches, we analyzed each height increment separately. First, the TVV data on contacts in the 2 - 3m and 3 - 4m increments had to be combined into a single value, as did all meter increments above 4 meters. Similarly, the 1997 data on contacts within the 0 - 0.3 and 0.3 - 1 m sections had to be combined into a single interval between 0 and 1 meter. Once this was done, we analyzed data from 1999 and 1997 with an ANOVA in which polygon was the experimental unit. Because this involved making four univariate ANOVA tests (one for each height increment), we used a Bonferroni-adjusted significance level of 0.0125 (= 0.05 / 4 levels; Sokal and Rohlf 1995 p. 240) to keep the experiment-wide error rate low.

We assessed the adequacy of our sampling using a resampling routine which we applied to the first set of data collected in 1998. In each polygon, we had taken 20 randomly located TVV measurements. For the analysis, we used Resampling Stats (Simon 1995) to make 1000 random samples of each sample size between two and twenty measurements and calculated a mean and standard deviation of the TVV estimates for that sample size. We then visually examined the plots of the coefficient of variation versus sample size to judge when further sampling would not yield greater precision.

We assessed the changes in the distribution and abundance of the three significant exotic in two ways. First, within sites, we calculated the number of polygons containing the species in the two years. Second, we compared the total areal extent of each species within sites by multiplying mean percent cover estimates by polygon areas, and summing across all polygons. No statistical analyses were used in either of these comparisons.

Nor did we use statistics in our monitoring for the presence of federally listed and candidate species. We simply used the data from our censuses and observations from encounters during rest stops while on the water. The presence and location of any of these species would simply be noted.

Quality control

As with any long-term data set, we were concerned about quality control for data files. We

did not expect to completely do away with errors, but we attempted to minimize them in the field in the following ways. First, data collectors and recorders who performed our field work were all professional botanists or people who were very familiar with the vegetation of Grand Canyon. Second, at the end of each day's censussing, we reviewed data sheets to check for errors of identification or species name abbreviation.

We also minimized the data entry errors by checking data at several steps. First, at the end of entering a plot's worth of data, a quick scan of the entries were performed to check for mistyped or really unusual entries. Second, one subplot was selected haphazardly, and entries were checked line by line for missed entries. Finally, before analyses were begun, we lined up all data from 1995, 1996, and 1997 by polygon for a species concordance check. In this way we could check for common misidentifications of a more subtle nature (e.g. *Muhlenbergia asperifolia* for *Cynodon dactylon* in their sterile state). This also allowed us to find obvious errors in cover estimate entries which had escaped us earlier. The hard copy and electronic versions of the data files submitted with this report reflect these quality checks.

RESULTS

Near Shore Habitats

Resampling analysis of the 1998 data showed that we sampled adequately for near-shore habitat. In all 11 reaches, the means of our estimates were fairly stable and standard deviations of estimates for the numbers, total area, and percent vegetated shoreline all stabilized well before the total of 20 samples was reached (Figures 2,3, and 4). In most cases, the standard deviation of the estimate had stabilized within the first 10 samples to where increasing sample size did not bring a large increase in the precision of our estimate. Data from 1998 and 1999 are included in

2. Results of resampling tests showing stabilization of variance in length of vegetated shoreline per 50 meters takes place withing 10 - 15 samples in each reach.

Number of Samples

Appendix A.

Mean Area / 50 m (y-axis label, left)

S.D. Area / 50 m (y-axis label, right)

Number of Samples (x-axis label)

Reach 1, Reach 2, Reach 3, Reach 4, Reach 5, Reach 6, Reach 7, Reach 8, Reach 9, Reach 10, Reach 11
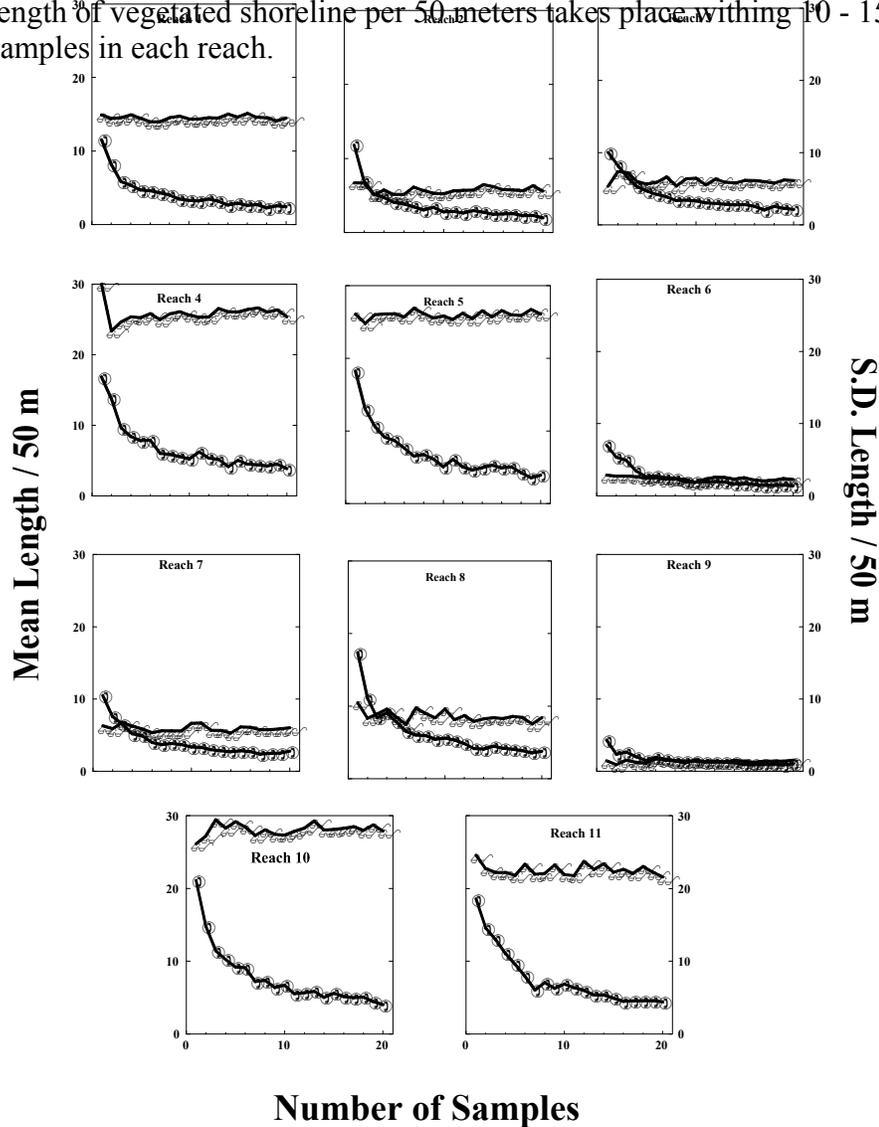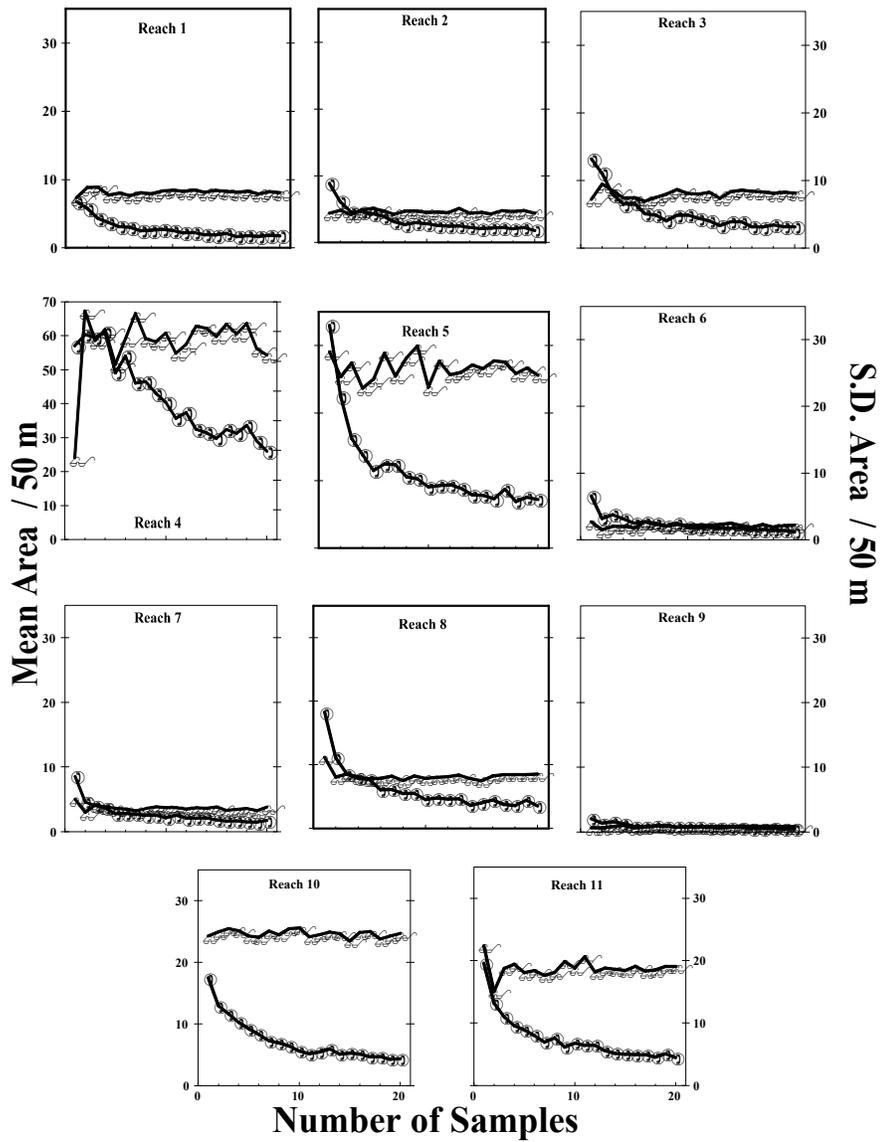
3. Results of resampling tests showing stabilization of variance in area of vegetated shoreline per 50 meters takes place withing 10 - 15 samples in each reach.

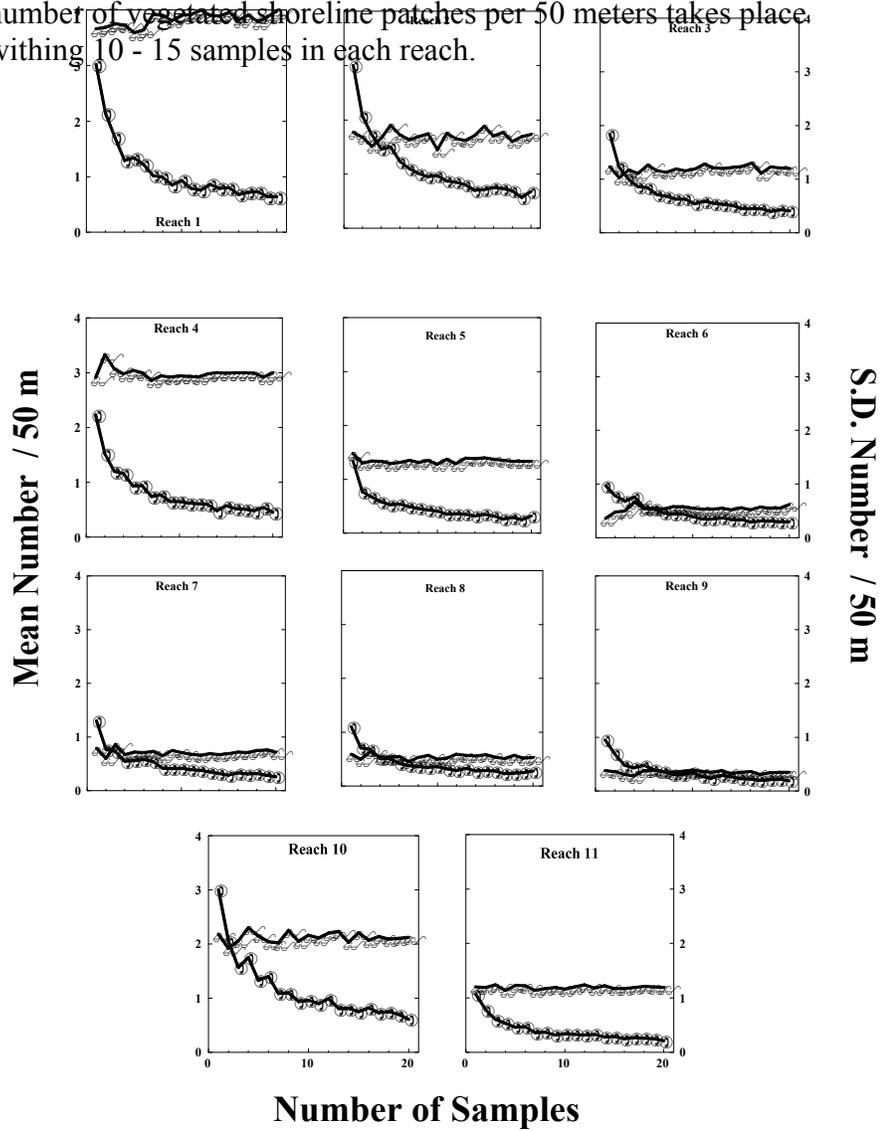4. Results of resampling tests showing stabilization of variance in number of vegetated shoreline patches per 50 meters takes place withing 10 - 15 samples in each reach.

Mean Number / 50 m

S.D. Number / 50 m

Number of Samples

14

5. Two years of data on the percent shoreline vegetated, total vegetated area, and number of vegetated patches per kilometer in each of the geomorphic reaches.
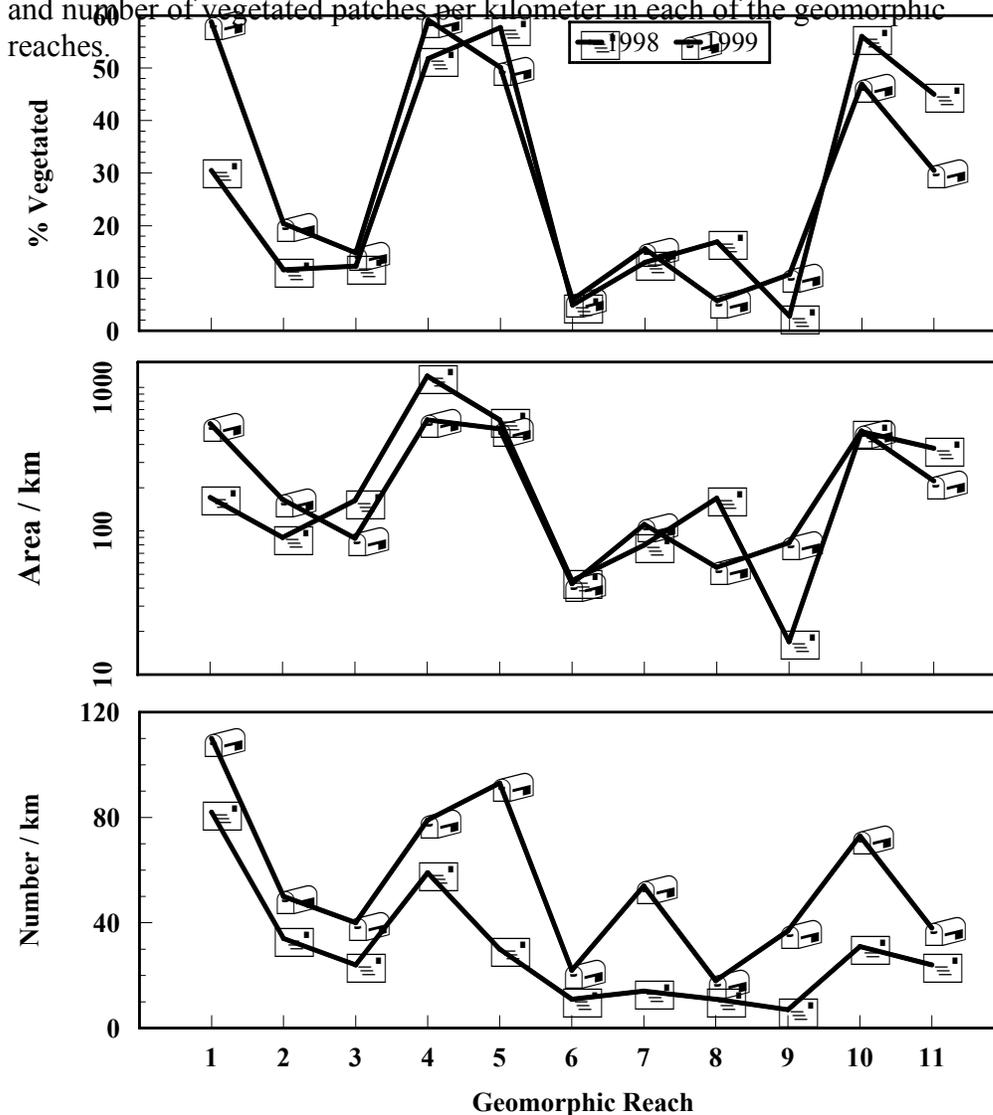
Our comparison of the 1998 and 1999 data showed only marginal changes in near-shore habitats during that year. There was no significant difference in the mean length of vegetated shoreline per sample in 1998 and 1999 (Figure 5; $F_{(1,395)} = 0.2733$, n.s.). Nor was there a change in the area of vegetated shoreline in the reaches (Figure 5; $F_{(1,395)} = 0.3308$, n.s.). The total amount of shoreline was distributed in a greater number of patches per sample, however (Figure 5; $F_{(1,395)} = 33.654$, $p < 0.001$), according to our estimates.

Marsh Transect Elevations

Survey data on some transects had to be excluded because of poor survey coverage in the areas surrounding them which made the digital surface elevation model suspect in those areas. We excluded transect A from 43 L, transect G at 172L, and transects N and O at 194 L. Survey point density in all other areas was considered adequate to produce a reliable surface model. Data on transect point elevations in 1998 and 1999 are attached as Appendix B.

6. Changes in mean thalweg elevations from return and current
cha

The repeated measures ANOVA on transect thalweg elevations showed no significant effect of time (Figure 6; $F_{(2,41)} = 1.158$, n.s.). Because there was a significant interaction between time and site, we analyzed the thalweg data within sites for the periods 1997 - 1998 and 1997 - 1999
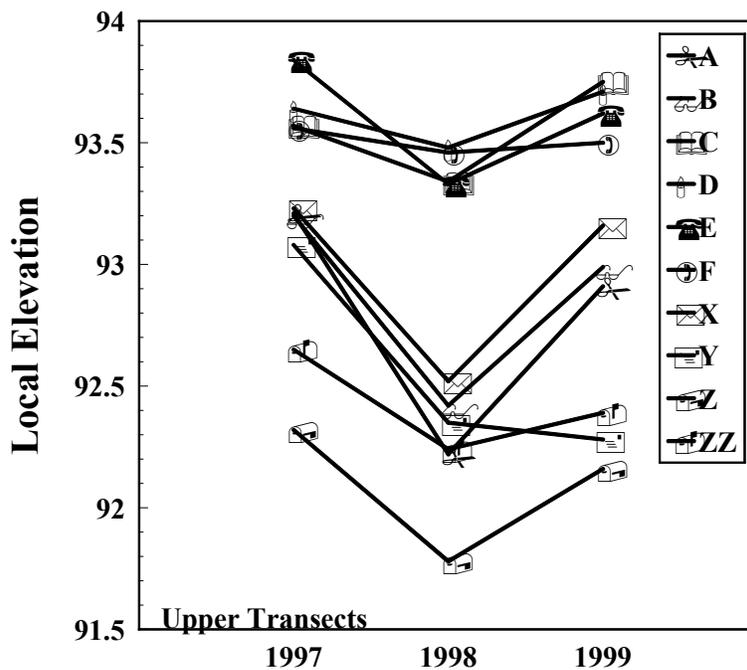
7. Change in transect elevations at the upper end of the
marsh at 55 R (Kwagunt).

with the non-parametric Wilcoxon T. Figure 6 shows that there was significant deposition at 43 L in 1998 and overall, and at 194 L in 1998 and overall. However, all of the significant changes were below 15 centimeters, which is near the limit of detection of topographic change (10 cm) used by the sandbar survey group.

The data from the low platform marsh at 55.5 R showed a significant time effect ($F_{(2,148)}$ = 5.624, p < 0.01; Figures 7 and 8). In general there was a slight loss of elevation at the marsh overall in 1998, and a slight increase in 1999 (1997 = 93.34, 1998 = 93.14, 1999 = 93.28). As with the return current channel marsh data, these data were within a +/- 15 cm range. There was no significant interaction between time and transect.

Vegetation Maps

Vegetation maps were created from the 1998 and 1999 Labor Day color aerial photographs. In 1999, portions of two sites (55 R, 94 L) were in deep shadow in the color aerials, so we created our maps with the black and whites aerials which were taken at about the same time. The maps were digitized and rectified by the GCMRC GIS group and are now part of the GIS database. Copies of these maps are available from the GCMRC I.T. department. Data on the 1998 and 1999 area of polygons in these sites is attached as Appendix C.

Vegetation Analysis

Raw vegetation data, as species abundances in polygons, were extensively proofed, and are attached to this report as Appendix D. Vegetation data were used to classify polygons into four to six basic types per site. Appendix E contains each sites type descriptors, including indicator species, along with their SRFR classification. The assignment of polygons to types for each site follows the type descriptions. The sampling intensity versus sample dissimilarity analysis showed that we have been adequately sampling the vegetation polygons. Marginal dissimilarities in oversampled polygons in all four patch types dropped below the 0.30 level within the first three or four samples (Figure 9). Wetland and dense tamarisk scrubland patches had very homogeneous samples, willow / mixed scrubland patches tended to have slightly more between-sample variability, and samples from the sparely vegetated and patchy bar-top were the

9. Similarity analysis showing that within 3 - 5 samples, the marginal benefit of adding new samples drops below what is required for gaining significant amounts of information.

most dissimilar.

18
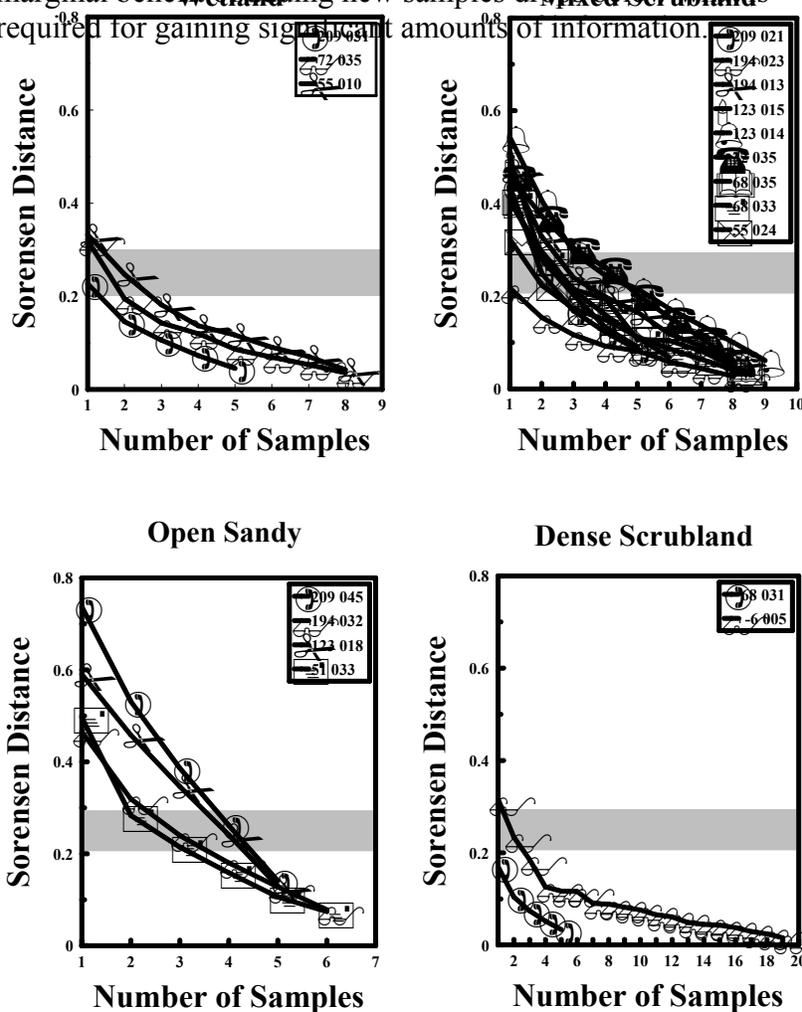
Table 2. Repeated measures ANOVA analysis of total cover, species richness (S), and Shannon diversity (H') of polygons in the four major new high water zone vegetation types. Asterisks indicate significant results with Bonferroni adjustment of the significance test.				
Foliar Cover	1997	1998	1999	Time effects
Wetland	41.0	56.5	41.4	$F_{(2,16)} = 4.07$
Tamarisk Scrub	62.9	63.1	39.2	$F_{(2,16)} = 16.59*$
Willow / Mixed Scrub	62.2	59.0	36.4	$F_{(2,17)} = 20.30*$
Open / Bar Top	12.1	9.9	5.9	$F_{(2,9)} = 12.84*$

Species Richness				
Wetland	14.5	15.7	15.2	$F_{(2,16)} = 0.17$
Tamarisk Scrub	12.5	11.0	9.9	$F_{(2,16)} = 4.92$
Willow / Mixed Scrub	14.7	12.9	13.3	$F_{(2,17)} = 2.24$
Open / Bar Top	7.9	8.0	8.5	$F_{(2,9)} = 0.19$

Diversity				
Wetland	1.090	1.221	1.339	$F_{(2,16)} = 0.51$
Tamarisk Scrub	0.827	0.943	0.807	$F_{(2,16)} = 4.62$
Willow / Mixed Scrub	1.170	1.164	1.202	$F_{(2,17)} = 0.40$
Open / Bar Top	0.665	0.857	1.060	$F_{(2,9)} = 1.21$

The analysis of wetland foliar cover data collected in the three years indicated that there was a significant increase in total cover between 1997 and 1998, but a loss of cover in 1999 resulted in there being no overall change over the two year period. However, cover in the other three types changed significantly during the period 1997 - 1999 (Table 2). Both scrub polygon types had cover measures approximately 20% lower in 1999 than in 1998, and bar top patches showeda steady decline over the two year period. None of the four patch types showed significant changes in either species richness or Shannon diversity during the study period (Table 2).

The polygon area data generated from the vegetation maps showed only a small amount of change in the total areal extent of the four basic polygon types (Figure 10, Table 3). There was a significant loss of wetland area (18.26 to 13.49 ha total) across all sites between 1997 and 1998, but an increase in 1999 led to an outcome of no significant difference between 1997 and 1999.

10. Change in areas of the fur basic patch habitat types during the study period.
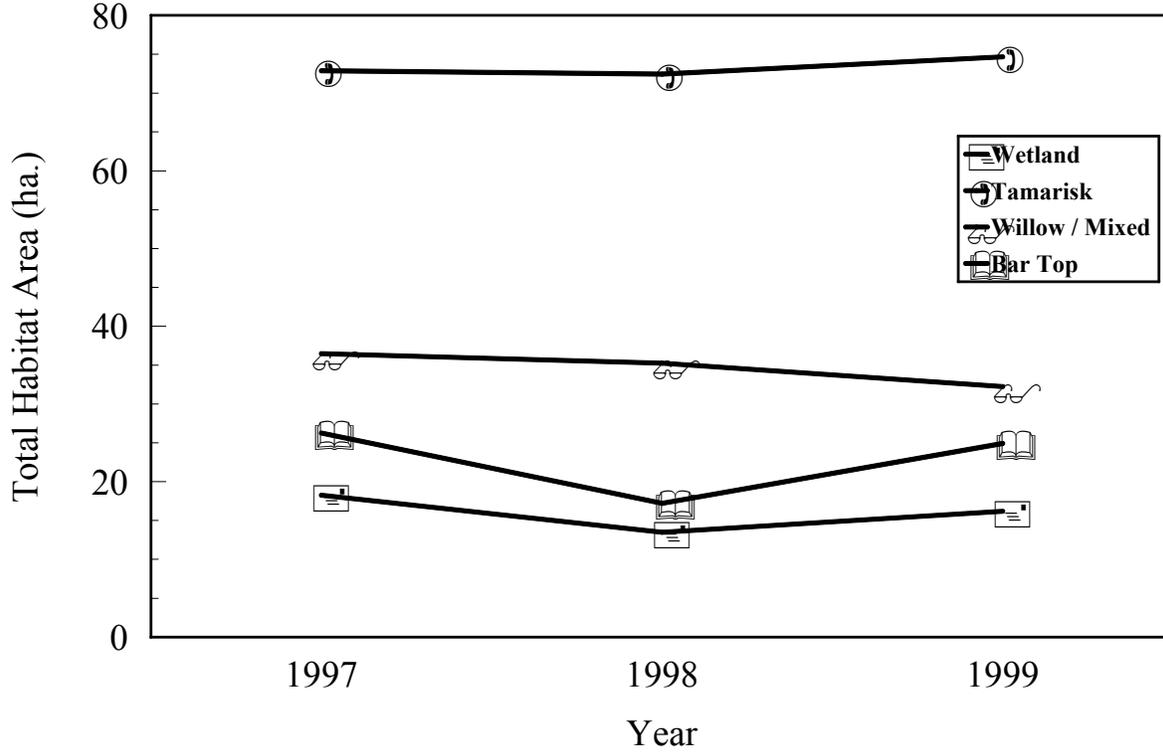
Table 3. Total area (ha) of the four basic habitat types in 1997, 1998, and 1999 in the 11 monitoring sites. T values in the last two columns are Wilcoxon T statistics comparing 1997 with 1998 and 1999. Asterisk indicates a significant change at $p < 0.05$.

	1997	1998	1999	T (1998 vs. 1997)	T (1999 vs. 1997)
Wetland	18.26	13.49	16.18	-21.5 *	-5.5
Tamarisk Scrub	72.89	72.46	74.68	-0.5	4.5
Willow / Mixed Scrub	36.48	35.24	32.25	-3.0	-2.0
Bar Top	26.26	27.20	24.92	1.0	2.0

	1997	1998	1999	T $_{(1998\ vs.\ 1997)}$	T $_{(1999\ vs.\ 1997)}$
Table 4. Total area (m^2) of five important wetland species in 1997, 1998, and 1999 in the 11 monitoring sites. T values in the last two columns are Wilcoxon T statistics comparing 1997 with 1998 and 1999. Asterisk indicates a significant change at $p < 0.05$.

	1997	1998	1999	T $_{(1998\ vs.\ 1997)}$	T $_{(1999\ vs.\ 1997)}$
Carex spp.	468.3	987.9	611.6	7.5	-2.0
Equisetum x *ferrissii*	4329.1	3109.1	2971.8	-9.5	-7.5
Juncus spp.	452.2	515.6	303.8	-3.0	-15.5
Phragmites australis	3768.7	3901.6	3879.8	7.0	2.0

|					
Scirpus spp.	185.7	242.3	142.8	-4.5	-2.5
Typha domingensis	881.7	1205.4	758.4	19.5 *	1.5

Table 5. Species turnover in the 11 study sites between 1997 and 1999. $CC_{(S)}$ column indicates the value of the qualitative Si rensen distance (coefficient of community) and the numbers in parentheses are the number of species found only in the first year / found in both years / found only in the later year.

	1997 - 1998		1998 - 1999		1997 - 1999	
Site	$CC_{(S)}$	Turnover	$CC_{(S)}$	Turnover	$CC_{(S)}$	Turnover
-6.5 R	**0.826**	(6 / 45 / 13)	**0.800**	(12 / 46 / 11)	**0.833**	(6 / 45 / 12)
43 L	**0.781**	(15 / 59 / 18)	**0.812**	(12 / 65 / 19)	**0.752**	(15 / 59 / 24)
5l L	**0.809**	(20 / 76 / 16)	**0.822**	(11 / 81 / 24)	**0.796**	(16 / 80 / 25)
55 R	**0.807**	(14 / 46 / 8)	**0.726**	(13 / 41 / 18)	**0.773**	(14 / 46 / 13)
68 R	**0.736**	(22 / 39 / 6)	**0.769**	(10 / 35 / 11)	**0.729**	(22 / 39 / 7)
72 L	**0.661**	(20 / 37 / 18)	**0.650**	(16 / 39 / 26)	**0.574**	(22 / 35 / 30)
94 L	**0.768**	(13 / 53 / 19)	**0.671**	(25 / 47 / 21)	**0.687**	(20 / 46 / 22)
123 L	**0.737**	(25 / 59 / 17)	**0.764**	(13 / 63 / 26)	**0.717**	(22 / 62 / 27)
194 L	**0.746**	(16 / 50 / 18)	**0.741**	(18 / 50 / 17)	**0.767**	(15 / 51 / 16)
209 L	**0.772**	(14 / 49 / 15)	**0.781**	(14 / 50 / 14)	**0.787**	(13 / 50 / 14)
249 L	**0.468**	(20 / 11 / 5)	**0.686**	(4 / 12 / 7)	**0.440**	(20 / 11 / 8)

None of the other types varied significantly during the period covered by this study. Tamarisk-dominated polygon area varied by less than 3%, willow / mixed scrub polygon area changed by 7%, and bar top polygons fluctuated by less than 5% from their 1997 totals.

Similarly there was a small amount of change detected in the total areal extent of the six important wetland taxa in our study sites. Only the increase in *Typha domingensis* in 1998 was significant among all the changes documented (Table 4). The large drop in the total area of

11. Resampling analysis of total vegetation volume (TVV) sampling adequacy showing that the variation in TVV estimates stabilizes within the first 10 samples.
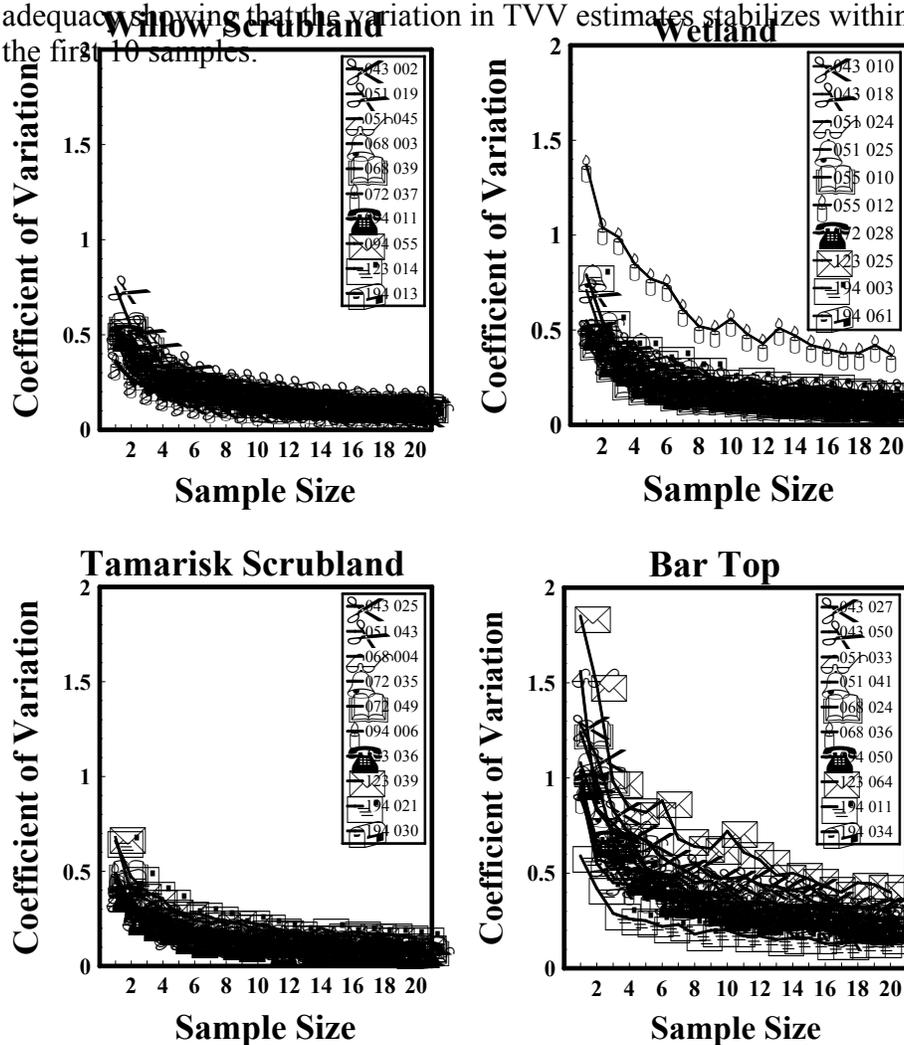
Equisetum and *Carex* between 1997 and 1999 (31% and 37%, respectively)was masked by the variability between sites.

We detected a moderate level of species turnover during the years of this study (Table 5). Between 10 and 30 percent of species found in a site in one year were not present in the same site in the following year. Likewise, over the two years covered by this study, we found up to 40 percent turnover of species in our sites. It should be noted that this did not represent a wholesale

change in the vegetation of the study sites. More than 85% of the transient species were annuals, and, with the exception polygons at one site, none of the transient species represented more than 5% cover in any of the polygons where they occurred. The exception occurred at 249 L where the rising levels of Lake Mead flooded several low-lying polygons in 1998 and 1999. These polygons contained nearly all of the sites wetland species in 1997.

The resampling analysis of the TVV data showed that the coefficient of variation stabilizes within approximately the first 10 samples in all habitat types (Figure 11). In both willow- and tamarisk scrubland patches, the standard deviation of the TVV estimates had dropped to a quarter of the mean when samples were made up of 10 measurements. With the exception of a flooded, patchily vegetated return channel (055 012), wetland polygon TVV estimates were even more stable than the scrubland patches. And although they were relatively more variable, estimates of TVV in the sparsely vegetated bar top patches were relatively stable after the first 10 measurements.

The TVV data showed that there were significant changes in all polygon types during the period of this study (Figure 12), with an increase in 1998 and variable amounts of change in 1999. Bar top, wetland, and mixed / willow scrubland polygons had an increase in vegetation density in 1998 and a decrease in 1999 (bar top: $F_{(2,493)} = 15.19$, $p < 0.01$; wetland: $F_{(2,645)} = 21.14$, $p < 0.01$; mixed: $F_{(2,656)} = 14.99$, $p < 0.01$). Dense tamarisk scrubland patches had
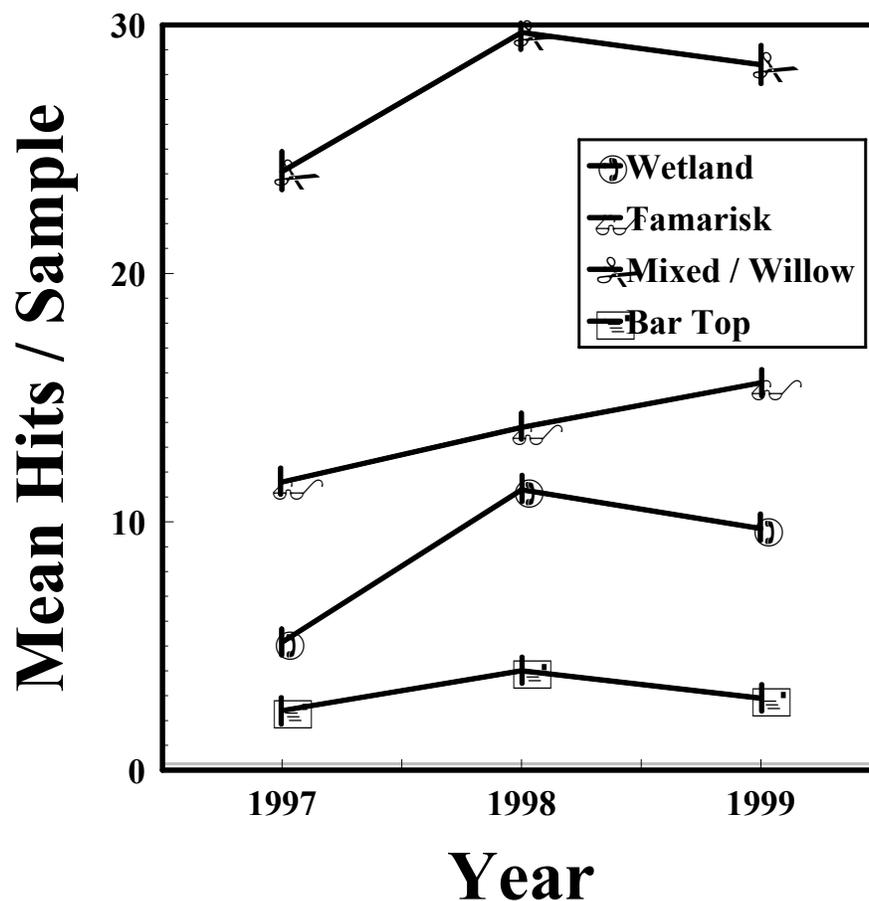
increase in vegetation density in both 1998 and 1999 ($F_{(2,673)} = 11.31$, $p < 0.01$).

Exotic species' abundances did not change consistently in 1998 or 1999 (Figure 13). The total amount of *Eragrostis curvula* increased significantly in 1998 (n = 5 sites, T = 7.5, p = 0.031), but a decrease in 1999 resulted in a final abundance not different from the 1997 total (T = 0.5, n.s.). The total abundance of *Alhagi camelorum* remained relatively constant in 1998, but decreased significantly in 1999 (n = 5 sites, T = -9.5, p = 0.031). *Lepidium latifolium* was not common in any of the three years considered here (only found in 4 sites) and did not change

13. Change in the total areal extent of three exotic species of concern: a) *Eragrosti curvula*, b) *Alhagi camelorum*, and c) *Lepidium latifolium*. Vertical bars represent +/- 1 s.e.

12. TVV data from the four basic patch types during the study showing significant gains in 1998 and variable amounts of change in 1999. Vertical bars represent +/- 1 s.e.
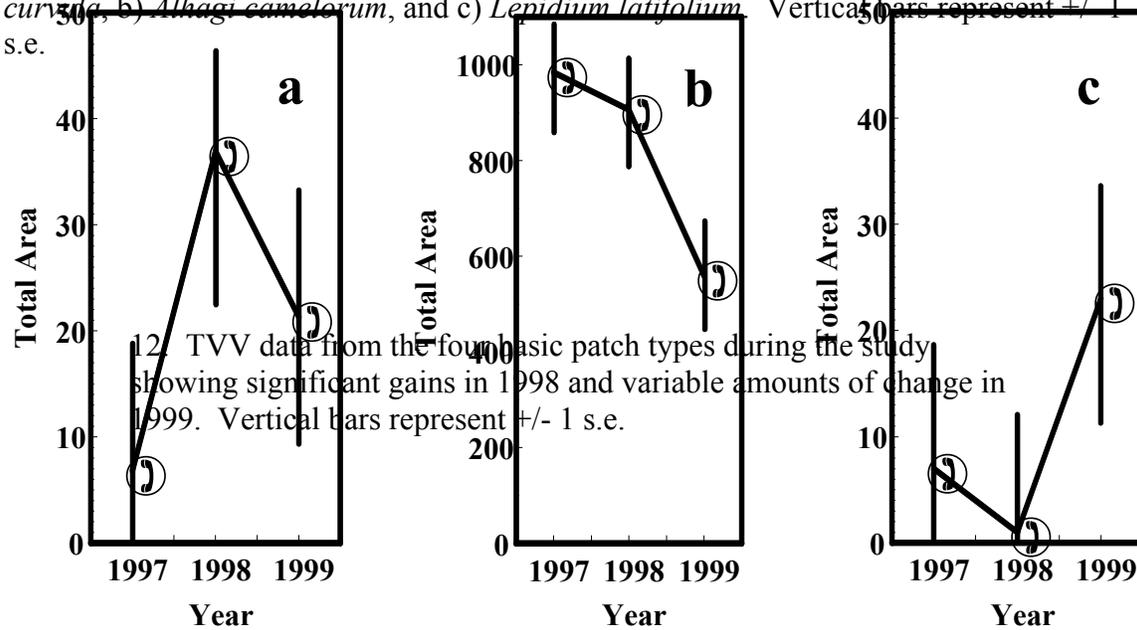
significantly in any of them.

DISCUSSION

Here we have reported on many aspects of riparian habitats in the Colorado River corridor. Although the system has remained more or less static, there are individual elements which have varied during one or both years. Below we discuss areas which we feel are the most important areas covered.

Sample Size Analysis

In all cases where we have examined the question of sample size adequacy, we have shown that the level of effort we are using is adequate to document the levels of, and variability in, the resources we are measuring. This has strengthened our confidence in our ability to detect meaningful changes in this system. For univariate measures such as the length, area, and number of vegetated shoreline patches, T.V.V. and so on, the sample variability stabilizes at about half the sampling intensity we have been using. Beyond that point, improving our sample precision would come at an increasingly large cost. For example, with a stable sample variance, we have decreased the standard error of our estimate by approximately 30% by doubling the minimum

sample size for an accurate variance estimate. To achieve an additional 30% improvement, we would have to triple our current effort. Thus, we feel that the current level is an acceptable compromise between efficiency and precision.

Likewise, the similarity versus sample size analysis shows that vegetation sampling in polygons of all types was more than adequate. In wetland and dense (tamarisk) scrubland patches, samples after the third were more or less superfluous, since they differed from the composite sample already collected by less than 15%. In more heterogeneous areas like the sparsely vegetated bar tops and areas with mixed riparian scrub (*Salix, Baccharis, Tessaria,* and *Tamarix*), four or five samples were required before dissimilarities moved consistently into the range of replicate samples.

Near Shore Vegetation Patches

Our data from 1999 indicate that there was a system-wide increase in the number of patches of near shore vegetation without a concomitant increase in either the total length or total area of vegetation in these habitats. Overall, our interpretation is that was not an actual change in the abundance of vegetation patches, only a different application of what defined the boundaries of a patch. Several factors, including inter-observer differences, the manner in which shoreline sections are divided into patches, and the impacts of different hydrographs in 1998 and 1999 likely contributed to these patterns. First, there were different observers on the two trips who may have had slightly different concepts of what constituted a distinct "patch" but similar enough definitions of what constituted "vegetated" vs. "non-vegetated" (both were covered in pre-trip meetings and protocol handouts). Thus, where one saw a half-meter break between two contiguous patches, the other saw a single continuous patch. Second, the hydrograph (Fig. 1) shows that prior to the 1998 spring trip, there had been relatively high flows from the late fall through the winter whereas in WY 1999, flows were approximately 5 kcfs lower, but during both trips, flows were more or less equivalent (ca. 8 - 15 kcfs). During the 1998 trip, we would have seen the perennial species, such as *Equisetum*, *Juncus*, *Scirpus*, *Phragmites*, and perennial grasses which dominate these habitats, but their above-ground parts would have been much smaller and sparser due to the effects of inundation and scouring during the fall and winter. In 1999, there would have been much less scouring during the fall and winter so that perennating parts (roots and rhizomes) would have covered roughly the same amount of area, but more densely. Thus where the observer in 1998 would have seen a more or less uniformly sparsely vegetated section of the shore, the 1999 observer would have seen more striking density changes and been more likely to read a break between contiguous patches rather than a single patch.

Vegetation Change

By using several different measures in our sites, we have been able to more strongly show patterns of vegetation change. For example, both total vegetation volume (TVV) and total foliar cover measure the density of vegetation in polygons and, for the most part, both showed similar changes between 1997 and 1999. In the dense tamarisk patches, both total foliar cover and TVV measures increased in 1998 and 1999, with an overall increase between 1997 and 1999. Likewise, in wetland patches, both total cover and TVV measured an increase in 1998 and a decrease in 1999, with an overall slight increase between 1997 and 1999. Although the foliar cover and TVV data from bar top patches disagree in the direction of change in 1998 (a slight increase in TVV, a slight decrease in cover), both showed decreases in 1999, and an overall

decrease between 1997 and 1999. We have disregarded the discrepancy between the two measures given that the differences are so small and the fact that there are no significant differences among any of the measures.

The only real discrepancy between the results from the TVV and total cover measurements was the disconnect between the mixed riparian scrub areas over time. The TVV data indicated an increase in density between 1997 and 1998, followed by a slight decrease in 1999 and an overall significant increase between 1997 and 1999 of approximately 15%. The cover data showed a non-significant decrease in 1998 followed by a large and significant decrease in 1999, and an overall significant decrease between 1997 and 1999 of approximately 20%. When the TVV data is broken down by height increments, it is clear that most of the between-year changes take place in the 1 - 2m section where large shrubs, such as willows and Baccharis show up (Fig 14). When we break down cover by species in these patches, it is obvious that the decreases in total cover are almost entirely due to changes in the cover of woody species, and most of that is due to decreases in the cover of willow and Baccharis (Fig. 14).

There are several possible explanations for this difference. First, increases in "hits" in the middle canopy layers could have come from stems and trunks which typically occupy several adjacent 10 cm increments in a TVV reading but do not contribute much to total foliar cover. The only other layer which showed an increase, albeit a non-significant one, was the area above 4 meters which is in keeping with a scenario of growth between years. However, if this is the case, it would require that plant foliage supported by these stems grow rapidly and end up in a more diffuse arrangement in a higher canopy layer. It is more likely that the discrepancy was caused by a lack of between-year calibration. Even though within years, we have very close

14. Changes in the foliar cover of common woody species in a) mixed riparian scrubland and b) dense tamarisk scrubland patches.
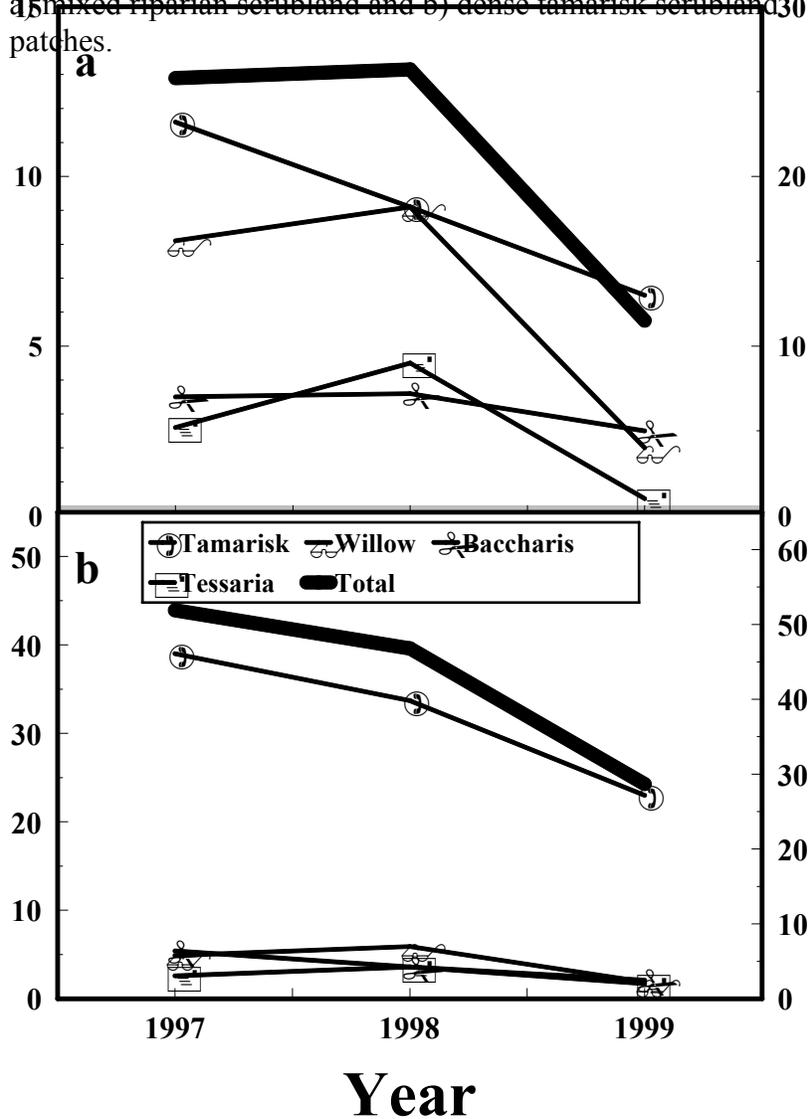
agreements among census groups in terms of species lists, cover estimates, and composition from these polygons, we have not been using any objective inter-observer standardization which would allow for between-year calibration.

Two methods have been suggested to us to rectify this if future monitoring involves cover data. First, all vegetation readers could participate in a standardization in which their own visual estimates were calibrated according to a series of point-contact measurements of cover for the dozen most common species of woody plants. This would standardize estimates of higher cover values in the same way that the 0.28m^2 frame standardizes the cover estimates for species at or below 1% cover. Second, readers could have their cover estimates calibrated individually by giving a series of cover estimates for the common species. These estimates would be judged against some kind of a point contact check (laying a meter tape in the shadow of the plant and running a point contact transect through the shadow). This would yield a factor by which that reader's estimates would be multiplied to give a "true" estimate for that observer. This could be done either as a single factor for all species, or as a series of factors dependent on growth form or leaf type.

In any case, for reasons related to collection methods and what we know about the two years' hydrographs and the relationship between hydrographs and soil water tables, we believe that the vegetation volume data are more reliable than the cover estimates. First, they were collected via a method which requires less individual judgement; there either is or is not live plant material within 10 cm. of the survey rod. Second, the hydrographs during both growing seasons shows high water levels (Fig. 1) with little fluctuations which translates into high water tables in the areas where willows and baccharis are found (Carpenter et al. 1994). Thus, the patter of higher numbers in 1998 and no difference in 1999 is probably the most representative of changes in these patches.

Exotic Species

Only one of the three exotic species of concern, camelthorn, showed a detectable change in our study sites. Our measures showed it covered approximately 40% less area in 1999 than it had two years earlier. For reasons outlined under the previous section, we cannot be certain that the difference represents a real change in the density of the species or if it resulted from between-year differences in observer judgements. None of the readers from 1999 questioned since the trip recall a dramatic change in camelthorn densities from the previous year (although some campsite clearing had occurred at 72L and 209L). We believe, therefore, that there was probably a reduction in camelthorn abundance, but not as dramatic as the one indicated by our data.

The lack of a detectable change in the abundance of exotic species of concern is a positive result. It may be related to the lack of large disturbance events in the years covered by the present study and to the time elapsed since the 1996 controlled flood. The data from the return current channels and low bar marsh survey transects show very little change in the physical environment of low elevation areas during the study period. Thus a lack of disturbance, which is often a prerequisite for invasion by exotic species (Grime 1974, Décamps et al. 1995), probably reduced the ability of exotics to establish.

We would like to insert two cautions to temper the interpretation of our data. First, our sites represent only a small fraction of the entire river corridor, comprising roughly half of one percent of the river shoreline between Glen Canyon Dam and Pearce Ferry. Second, these species are present at relatively low abundances and estimates of extent for rare species, even measures as robust as our broadly measured as our number of sites and number of polygons per site, are subject to large relative measurement error. Thus firm conclusions about the spread of exotics would have to include measures of changes in areas with large, well established populations as

well.

MANAGEMENT RECOMMENDATIONS

Other than the three-day spike of 31.5 kcfs, there were no significant flow events during the study period to comment on . The marginally higher flows during the fall and winter of 1997 / 1998 may have scoured off the above-ground parts of perennial species in near shore habitats, but it did not change their overall abundance measured by length of shoreline colonized or total area of shoreline vegetation in any of the geomorphic reaches. The late spring and summer hydrographs were very similar in the two years, with relatively high mean and maximum flows in both years.

The lack of disturbance-generating events coincided with a period of no significant change in the distribution and abundance of significant invasive exotics. For two reasons, we would caution against interpreting this as indicating that this is the best flow regime for minimizing the spread of exotics. First, the sites we measure do not contain significant amounts of exotic species, and therefore are unlikely to show rapid changes in abundance of exotics until numbers increase above a solid detection threshold. In addition, we did not detect changes in the distribution of invasive species as a result of the 1996 controlled flood, which would lead us to conclude that the Water Year 1996 hydrograph was an acceptable way to minimize the spread of exotics. What is needed is a study focused on the spread and population growth of these species in areas where they are present in larger, denser populations.

With regard to the monitoring methods themselves, we would recommend an examination of the between-observer calibration methods mentioned in the report. Given that the point of monitoring is to detect changes between years in the system, there should be some confidence in the consistency of observer standards between years. We have shown that we do have that for within-year comparisons, and that the level of sampling intensity adequately samples to account for within-year variability. However, without some objective measure of how closely observers adhere to a standard in different years, we cannot be entirely confident in our observed changes. It should be pointed out that while univariate measures are subject to large effects of observer bias, compositional data are much less so (McCune et al. 1997), so that tests based on similarities are more robust than those based on cover, richness, and diversity.

Finally, we would recommend that vegetation monitoring continue in 2000 and beyond. Although we have encountered some difficulties with elements of the monitoring methods, these are not considered fatal to the overall aim of detecting change through time. Furthermor, we believe that the importance of riparian vegetation itself and its position at the interface between terrestrial and aquatic systems requires that there be some level of monitoring.

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