MONITORING THE AQUATIC FOOD BASE IN THE COLORADO RIVER, ARIZONA DURING FISCAL YEARS 1998 AND 1999

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ABSTRACT

Discharge from Glen Canyon Dam (GCD) strongly influences the lower trophic levels (phytobenthos and macroinvertebrates) of the aquatic ecosystem in Grand Canyon National Park. The aquatic food base in the Colorado River is affected by the duration and timing of low releases from GCD, as well as the range of daily fluctuations. The overall objectives of this project are to seasonally monitor the effect of discharge characteristics below GCD, under modified low fluctuating flow criteria, on the distribution, standing mass and composition of primary and secondary producers in the benthos and drift, and to examine the linkages between lower and higher trophic levels. This information is critical because the lower aquatic trophic levels provide essential resources to both aquatic and terrestrial components of the fluvial ecosystem in Grand Canyon National Park.

Water quality monitoring of the Colorado River below GCD is within the typical seasonal ranges for temperature, conductivity, pH and dissolved oxygen. We have insufficient baseline data to determine if nutrient concentrations are within typical ranges. Secchi depths and light intensity data indicated seasonally turbid water typical of a wetter than normal year (1998) as evidenced by the Little Colorado and Paria River spates this past summer and winter.

Dam discharges were highly variable between months with the first use of the E.I. S. selected flows starting in September 1997. These included higher than normal monthly discharges including daily flow fluctuations over 560 $m^3 \cdot s^{-1}$. These flows have returned instability in the benthic community as indicated by high variability of detritus and Oscillatoria. All other biotic categories are either equal to or enhanced when compared to 1991 reference data.

Comparing June 1996 benthic biomass estimates to the 1991 reference data showed the biotic categories to either be maintained or enhanced. This collection trip generally had the highest biomass estimates documented through the study site and macroinvertebrate biomass was enhanced at all cobble sites. We can attribute this pattern to consistent flows with little daily fluctuations and possibly a result of clear water conditions from scour after the 1996 Spike Flow. These results indicate that if during normal dam operations, including management or research flow scenerios that the biotic factors comprising the food base were degraded at enough sites to warrant remediation then we recommend steady flows of at least 400 $m^3 \cdot s^{-1}$ for several months.

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Micheal Kersley, of NAU, provided us within near shore vegetation estimates through Grand Canyon. Owen Gorman and Dennis Stone of the U.S. Fish and Wildlife Service provided various fish specimens from Grand Canyon for our food web construction. Ted McKinney and Bill Persons of Arizona Game and Fish provided us with fish specimens from Glen Canyon for dual stable isotope analysis.

INTRODUCTION

Discharge from Glen Canyon Dam (GCD) strongly influences the lower trophic levels of the aquatic ecosystem in Grand Canyon National Park (Blinn et al. 1992, 1993, 1994, 1995a,b,1998, Shannon et al. 1996a). The aquatic food base in the Colorado River is affected by the duration and timing of low releases from GCD, as well as the range of daily fluctuations. The objectives of this project is to seasonally monitor the effects of discharge characteristics below GCD, under the Bureau of Reclamation's modified low fluctuating flow criteria, on the composition, distribution and standing mass of primary and secondary producers in the benthos and drift. This information is valuable because the lower aquatic trophic levels provide essential resources for both aquatic and terrestrial components of the fluvial ecosystem.

Grand Canyon National Park's Colorado River Management Plan (NPS 1989) states that its resource management goals are "to preserve the natural resources and environmental processes of the Colorado River corridor and the associated riparian and river environments.... (and) to protect and preserve the river corridor environment (NPS 1989:9). Among its objectives are:
1) "establish.....a long-term monitoring program to assess changes in the status of natural....resources. This program will require definition of present resource status (NPS 1989:10)"; and 2) "advocate and support operational objectives for the Glen Canyon Dam (GCD) which are most compatible with protection of the intrinsic resources of the Colorado River within Grand Canyon National Park (NPS 1989:10). The aquatic food base is an integral part of the natural resources in Grand Canyon National Park.

The Secretary of the Interior authorized implementation of modified low fluctuating flow criteria from Glen Canyon Dam (GCD) in August 1996 based on the recommendations set forth by the Environmental Impact Statement (1990-1992). These flows are designed to mitigate impacts of dam operations on downstream riverine resources. The flows consist of low-, medium-, and high-volume months, with low flows during the spring and late fall, moderate flows in May and September, and high flows during mid-summer and mid-winter. These flows have a maximum discharge of 566 $m^3 s^{-1}$, a reduced range of daily fluctuation, and reduced up- and down-ramping rates.

The Environmental Impact Statement (USBR,1995) on the operation of GCD identified the aquatic food base as an "indicator resource" and important habitat for wildlife. Wildlife linked directly to the aquatic food base include native and non-native fish, insectivorous birds and bats, reptiles and waterfowl. Indirect links to the

aquatic food base include peregrine falcons feeding on waterfowl, swifts, swallows and bats, as well as king fishers, great blue herons, osprey and bald eagles preying on fish.

The National Park Service and the Bureau of Reclamation have both stated the importance of understanding the aquatic food base in the Colorado River below GCD through Grand Canyon National Park. This can only be accomplished through continued monitoring which will add to the established data base and provide the foundation for long-term adaptive management planning.

This report provides information on the following objectives;

Objective 1:	Monitor the effects of modified low fluctuating flows from Glen Canyon Dam (GCD) on the benthic community in the Colorado River between Glen Canyon Dam and Diamond Creek.
Objective 2:	Monitor the effects of modified low fluctuating flows from GCD on the organic drift in the Colorado River between GCD and Diamond Creek.
Objective 3.	Assess the benthos and drift of major tributaries in Grand Canyon National Park.
Objective 4.	Construction of an aquatic/riparian food web using stable isotope analysis.

METHODS

Objective 1: Monitor the effects of modified low fluctuating flows from Glen Canyon Dam (GCD) on the benthic community in the Colorado River between Glen Canyon Dam and Diamond Creek.

Biomass, composition, and habitat requirements of primary and secondary producers were monitored during the low to moderate flow months of March, June and October of each year within the mainstem of the Colorado River (n = 3 sampling trips per year). Seven sites will be monitored at the start and end of three major sections below GCD, including Glen, Marble, and Grand Canyons (n = 6 sites), and in Middle Granite Gorge (n = 1 site, Table 1).

These locations generally correspond with the monitoring sites used by Blinn et al. (1993, 1994, 1995b) and Shannon et al. (1996b) with the following modifications. The collection sites at the USGS gauging stations previously used by the NAU Aquatic Food Base Program will be dropped because they provide limited information (Blinn et al. 1992, 1993, 1994, Shannon et al. 1996b). These sites were originally selected for continuous flow across the channel and uniform depth in a pool above a rapid for gauging purposes, whereas pools below debris fans provide slow water velocities that collect fine organic and sediment particles. The previous gauging sites will be replaced with a site in the Middle Granite Gorge (RKM 203) to assess the food base of the largest mainstem humpback chub aggregation (Valdez and Ryel 1995). Sites were selected that will provide the most amount of information about the food base and for fish monitoring programs based on the past seven years of collecting in the Grand Canyon, and the allowed budget while decreasing river user days (Table 1).

Table 1. Collection Sites, River Kilometer (RKM), Elevation (m), Orientation, Reach Type, and Habitat in the Colorado River Below Glen Canyon Dam for Cobble Riffles, Pools and Tributaries within Glen, Marble and Grand Canyons, Arizona. Habitat describes area of collecting activity.

GLEN CANYON

Name	RKM	Elevation	Orientation	Reach Type	Habitat
1. Glen Canyon Gauge	-23.2	953	Southwest	Narrow	Drift
2. Lees Ferry	0.0L	947	Southwest	Wide	Pool/Drift
Lees Ferry Cobble	0.8R				Cobble
Paria River	1.0R				Tributary

MARBLE CANYON

Name	RKM	Elevation	Orientation	Reach Type	Habitat
3. Two-mile Wash	2.9R	876	South	Wide	Drift
Two-mile Cobble	3.1R				Cobble
Two-mile Pool	3.1L				Pool
Cathedral Island	4.0L				Shore
Vasey's Paradise	50.8R				Tributary
4. 60 Mile Gauge	95.7L	831	South	Wide	Pool
Gauge above LCR	98.4L				Shore/Drift
LCR Island	98.6C				Cobble
LCR	98.6L	826			Tributary

GRAND CANYON

Name	RKM	Elevation	Orientation	Reach Type	Habitat
Lava Chuar	104.0R	815			Tributary
5. Tanner Canyon	108.8R	808	Southwest	Wide	Pool/Shore
Tanner Cobble	109.6L				Cobble/Drift
Bright Angel Creek	140.8R	739			Tributary
6. 127 Mile Rapid	202.9R	616	Northeast	Narrow	Cobble/Drift
Middle Granite Gorge	203.2L				Pool/Shore
Tapeats Creek	214.8R	610			Tributary
Kanab Creek	231.2R	572			Tributary
Havasu Creek	249.6L	544			Tributary
7. Spring Canyon	326.4R	454	South	Wide	Pool/Shore
205 Mile Rapid	328.8R				Cobble/Drift
Spring Canyon Creek	327.2L	454			Tributary
Diamond Creek	361.6L	424			Tributary

Three habitat types (pools, riffles, and near shore habitats) were monitored at each site. Sampling was conducted along three transects 30 m apart in each habitat type. Petersen or Petit Ponar dredges will be used in the fine sediment and Hess substrate samplers will be utilized on cobble bar riffles. Pool habitats were sampled at five locations along the three transects; thalweg, <28 m^3/s, baseflow (142 m^3/s), lower-varial (~280 m^3/s), and upper-varial (~500 m^3/s). Cobble riffle collections were taken at the greatest depth possible with three paired samples along with lower and upper-varial samples.

Samples were processed live within 48 h and sorted into five biotic categories: C. glomerata, Oscillatoria spp., detritus, miscellaneous algae and macrophytes, and macroinvertebrates which were numerated into Gammarus lacustris, chironomid larvae, simuliid larvae, and miscellaneous invertebrates. Miscellaneous invertebrates included lumbriculids, tubificids, physids, trichopterans, terrestrial insects and unidentifiable animals. Detritus was composed of both autochthonous (algal/bryophyte/macrophyte fragments) and allochthonous (tributary upland and riparian vegetation) flotsam. Each biotic category was oven-dried at 60°C and weighed to determine dry weight biomass. Samples were then ashed (500°C, 1 h), and reweighed for ash free dry mass estimates. Preservatives alter biomass estimates and accurate dry weights are required for building an energetics model. Adult and pharate specimens will be collected with sweep nets, white and UV lights, spot samples, and Thienemann collections for taxonomic verification. Specimens are housed at NAU and logged according to NPS requirements.

Water temperature, dissolved oxygen, pH, specific conductance, substratum type, microhabitat conditions, Secchi depth, water velocity or stage, depth, date, site, and time of day will be recorded at each sample site. Depth integrated light intensity data loggers will be deployed at five collection sites. These sites corresponded with those initiated in the FY97 Steady High Flow Program. Benthic biomass estimates were compared between clear and turbid water sites with light as a predictor variable. The protocol outlined above will help depict the relationship between benthic biomass, discharge and light variability.

Nutrient levels were monitored at each collection site. Th efollowing nutreints were analyzed; ammonia, phosphate and nitrate-nitrogen. Triplicate samples were collected, acidified and analyzed within one month of collecting on a Technicon Auto Analyzer II™ after digestion.

Shoreline habitats were sampled for invertebrates in emergent vegetation, fine sediments and tychoplankton. These nearshore habitats have become quite extensive throughout the river corridor due to steady, low fluctuating flows

and preliminary observations indicate they provide an important habitat for fish; similar to return current channels, but with greater stability. These low velocity near-shore habitats, composed primarily of Equisetum (horsetails) may provide similar habitat to backwaters, but are more abundant and readily available for invertebrate and small fish colonization. Presently, only minimal data exists for these abundant shoreline habitats. The following collections will be made from kayaks in an effort to reduce damage to this fragile and potentially critical habitat.

1)Triplicate harvests of shoreline emergent vegetation were taken in circular stovepipe samplers (0.02 m^2 area) just above the sediment, and screened as it's pulled through the water column in an effort to capture macroinvertebrates associated with the vegetation.

2)Triplicate plankton collections (156 μm,) were taken along the outer interface of shoreline vegetation. Samples were preserved in 70% EtOH and sorted in the lab with a dissecting scope into the following categories: Copepoda (Calanoida, Cyclopoida, Harpacticoida), Cladocera, Ostracoda, and miscellaneous zooplankton which include small chironomids, Gammarus lacustris, planaria, hydra, etc. Large samples were split with either 1 ml, 5 ml or 10 ml sub-samples sorted from a 100 ml dilution. Zooplankton densities of each category, general condition, reproductive state and presence of nauplii were recorded. Samples were processed for dry mass estimates and converted to ash-free dry mass using regression equations (Shannon et al, 1996b). The remaining organic material was filtered through a 1 mm sieve to remove CPOM and then filtered onto glass fiber filter (Whatman® GF/A, μm mesh) with a Millipore Swinex® system. These filters were dried at 60°C and combusted for 1 h at 500° C. Volumetric estimates (mass/m^3/s) were estimated from hand-pumping 15L of river water at eacg transect.

3)Triplicate sediment samples were taken with a Petite Ponar (0.02 m^2 area) and sieved for benthic macroinvertebrates. Macroinvertebrates were processed with the same protocol as emergent vegetation collections. Sediment was dried and sieved for clast fractioning.

4) Six minnow traps (0.48 m x 0.22 m) were set at an adjacent near-shore habitat for 12 h overnight to determine if fish utilize the habitat. Size, weight, total length, and standard length of each fish species were determined. General condition factors were determined for each fish population with the following equation:

$$K = \frac{W \cdot X}{L^3},$$

where W = weight in grams, L = total length in millimeters and constant $X = 10^5$ (Moyle and Cech, 1988). This information will determine the importance of the

shoreline habitat for fish as a food resource and refuge. Appropriate permits were obtained from Arizona Game and Fish and US Fish and Wildlife Service.

Multivariate statistical analysis (MANOVA) using abiotic predictor variables and biotic response variables were used to determine significant patterns in composition, distribution and biomass of the benthic community. Also, relationships between AFDM of biotic components and the physical, chemical, spatial and temporal variables were examined with multivariate canonical correspondence analysis (CANOCO, Ter Braak 1992, Palmer 1993). The SYSTAT computer software package (Version 5.1, Wilkenson 1989) and/or the NAU mainframe was used for all calculations.

Objective 2: Monitor the effects of modified low fluctuating flows from GCD on the organic drift in the Colorado River between GCD and Diamond Creek.

Drift was collected on each river trip from sites at or near the above sampling stations (Table 1). Two components of stream drift were assessed:
1) Coarse Particulate Organic Matter (CPOM).
Near-shore surface drift samples (0-0.5 m deep) were collected at each pool site for CPOM during each collection trip. Collections were taken in triplicate between 1000 h and 1500 h at each site to establish the affects of discharge on drift. Collections were made with a circular tow net (48 cm diameter opening with 500 μm mesh) held in place behind a moored pontoon raft or secured to the river bank. Samples were sorted and processed live for biota as outlined for the shoreline emergent vegetation in Objective 1. Current velocity was measured with a Marsh-McBirney electronic flow meter and collection duration were measured for volumetric calculations (mass/m^3/s).

2) Fine Particulate Organic Matter (FPOM). FPOM drift was collected at the same time and with the same general protocol as CPOM ($n = 3$). The net has a 30 cm diameter opening with 153 μm mesh. Samples were preserved in 70% EtOH and sorted in the lab with a dissecting scope according to procedures outlined in Objective 1 for plankton tows near shoreline vegetation.

Multivariate statistical analysis as outlined in Objective 1 was employed to determine significant patterns in the composition, distribution and biomass of drift along the river corridor.

OBJECTIVE 3. Assess the benthos and drift of major tributaries in Grand Canyon National Park.

Benthic Collections: Aquatic macroinvertebrates, phytobenthos and detritus were collected during January 1999 from 11 major tributaries of the Colorado River through Grand Canyon. At each tributary, two Hess samples were taken along three transects, 30 m apart ($n = 6$). All tributary transects were located above the influence of the mainstem ($\geq$2,265 m^3/s), starting at least 10 m above the mesquite line at the old high water zone. Biomass samples were sorted into the five biotic categories as outline for benthic collections (Objective 1) in the Colorado River for comparison with benthos in the mainstem.

Taxonomic samples were collected, preserved in 70% alcohol, identified to the lowest taxonomic level possible, counted, and measured for total length. Water temperature, dissolved oxygen (DO), pH, specific conductance, and time of day were measured at each sampling site. Current velocity and depth was measured at each sample location.

Drift Collections: Both CPOM ($n = 3$) and FPOM ($n = 3$) collections were made following the same protocol as used in the mainstem (Objective 2). Tributary discharge were estimated by measuring the channel geometry and water velocity along a transect perpendicular to flow.

Past collections in the tributaries in Grand Canyon were made bimonthly in 1991 and annually in June 1992, 1993, 1994, 1996, and 1997 by the NAU Aquatic Food Base Monitoring Program. Shannon et al. (1996b) reported the month of January had the highest biomass and biodiversity which supported findings by Hofknecht (1981). This may result from low hydrologic disturbance during this period. June tributary discharge changes were dependant on the timing and amount of snow-melt. Changing the collection period is not only sound science but is also wise from a river ethic standpoint. June is the peak commercial river running season and all of the perennial streams in Grand Canyon are attraction sites.

This work is proposed to provide additional abiotic and biotic information on 11 major tributaries of the Colorado River through Grand Canyon National Park. These tributaries vary widely in physicochemical characteristics and yield a variety of different biotic communities, all of which may potentially invade the mainstem under favorable conditions (Shannon et al. 1996b). The information collected in this study will help characterize the seasonal abiotic conditions of tributaries in the Canyon and will provide information on their suitability as a habitat for native and exotic fishes. It will also provide knowledge on the diversity and biomass of macroinvertebrates that serve as food for native and exotic fishes in Grand Canyon. Some tributaries are

highly susceptible to flash floods and periods of reduced or no flow which are common to many southwestern desert streams, while other spring-fed tributaries have more stable flow conditions. A comparison of these widely disparate systems will not only provide distributional information on aquatic macroinvertebrates within the Grand Canyon but will offer valuable information on abiotic variables that might determine their distribution.

Monitoring these tributaries is also a valuable management tool for assessing biota that are sensitive to changes within a given watershed. Therefore changes in land practices both within and outside the boundaries of Grand Canyon National Park may be monitored. Management decisions such as seasonally adjusted steady flows, as described in the U.S. Fish and Wildlife Service Biological Opinion, or the installation of multiple withdrawal structures on GCD will have an impact on the aquatic food base in the Colorado River. Both of these management options will have an influence on at least the temperature range within the mainstem. Recruitment of aquatic macroinvertebrates into the mainstem will initially be from the tributaries. Understanding these tributaries may help resource managers determine reasonable ranges for important variables such as seasonal water temperature and discharge regimes.

OBJECTIVE 4. Construction of an aquatic/riparian food web using stable isotope analysis.

A valuable management tool, for critical ecosystem level decisions, is the development of a comprehensive food web. To date, a data based food web has not been constructed for the Colorado River ecosystem through the Grand Canyon aquatic/riparian community. Blinn et al. (1994) presented an aquatic energetics model for the tailwaters, Angradi (1994) developed a dual isotope model for Glen and Marble Canyons, and the NAU Aquatic Food Base Program used dual stable isotopes to track the origin and composition of organic drift (Shannon et al. 1996b). The above data sets were used to further expand our understanding of a system-wide food web with an emphasis on native fish food habitats. Schell and Ziemann (1993) used $\delta^{13}C$ natural isotope abundances to derive a food web in the Arctic coastal plain, which is similar to the Colorado River ecosystem in terms of simplicity.

Food web construction using stable isotopes has the advantage of defining the source of organic drift, which is visually uniform and a critical carbon source in lotic ecosystems. Stable isotope analysis also depicts what is assimilated by an organism, which eliminates "last meal" bias, and the complication of digestion rates that gut-analysis alone can lead to (Rosenfield and Roff 1992).

All potential major carbon sources within the aquatic/riparian communities of the Colorado River ecosystem were collected and analyzed for $\delta^{13}C$ $\delta^{15}N$, and $\delta^{33}S$ natural isotopic ratios. Triplicate samples of major plant and animal components of the riparian and upland vegetation, the benthic community in the river, and plankton from Lake Powell were taken throughout the Colorado River ecosystem. Fish and bird samples were obtained from incidental deaths from the projects monitoring these animals. The technique does not require the release of radioactive tracers to follow the path of ^{13}C, ^{15}N and ^{33}S through a food web. Samples were air-dried immediately after collection in the field or laboratory and ground to a powder (<0.05 mm particle size) with a Whir-L-Bug™. Samples were analyzed with a mass-spectrophotometer. We obtained muscle plugs or fin clips from endangered fish for non-lethal data in cooperation with the fish monitoring projects.

Establishing Reference Data

Although the management objective for the aquatic food base states that it should be "maintained or enhanced" the exact levels were not defined. After analysis of our data from 30-plus river trips and 80-plus collections in the Lees Ferry area we have concluded that the data defining the aquatic food base in 1991 was the most degraded. It was during this time period of the GCES/ BOR sand movement research flows that included two 3-d steady 142 $m^3 \cdot s^{-1}$ flows each month, highly variable ramping rates and flows up to 934 $m^3 \cdot s^{-1}$. We have since learned that these flow regimes are the worst possible for the aquatic food base. It was also during this time period that the trout in Glen Canyon were in poor condition and native fish down river were also in poor health from an increase in parasitism. Therefore a reference data set has been developed for both pool and cobble habitats for each site which should be maintained or enhanced.

A reference data set was developed by using the mean biomass for each bi-monthly collection trip in 1991 for Cladophora, Oscillatoria, detritus and macroinvertebrates at each site. Miscellaneous algae, macrophytes and bryophytes or MAMB were not a separate category in 1991 so the means from the 1992 seasonal collections from each site were used. This procedure results in six data points being used at each site for each biotic category. MANOVA was then run for each monitoring collection trip against the 1991 reference data.

Results of the MANOVA for each biotic factor or univariate probability would indicate if the biotic resource was enhanced significantly, maintained or nonsignificant change and if the biotic resource was degraded significantly. This

determination was made after comparing the mean data for the monitoring trip and the 1991 reference data. **An increase in biomass of the biotic categories during monitoring would indicate an enhanced resource. Oscillatoria is the only biotic factor that is does not enhance the aquatic food base with increasing biomass, therefore reduced Oscillatoria biomass over 1991 reference data would be an enhanced resource.**

RESULTS

Objective 1: Monitor the effects of modified low fluctuating flows from Glen Canyon Dam (GCD) on the benthic community in the Colorado River between Glen Canyon Dam and Diamond Creek.

Water quality patterns were consistent with past collections; however, a decrease in conductivity and an increase in dissolved oxygen concentrations were detected at all collection sites (pages a2-a55)

Pool and cobble habitat biomass estimates either were maintained or enhanced during October 1997. March and June 1998 in comparison to 1991 reference data (Table 2, Appendices). Most of the variability was in the Oscillatoria and detrital estimates. These two biotic categories are sensitive to dam operations, with Oscillatoria increasing in the varial zone due to daily fluctuating flows and detritus also moving through the study site.

Comparing June 1996 benthic biomass estimates to the 1991 reference data showed the biotic categories to either be maintained or enhanced (Table 3). This collection trip generally had the highest biomass estimates documented through the study site and macroinvertebrate biomass was enhanced at allcobble sites. We can attribute this pattern to consistent flows with little daily fluctuations and possibly a result of clear water conditions from scour after the 1996 Spike Flow. These results indicate that if during normal dam operations, including management or research flow scenerios, the biotic factors comprising the food base were degraded at enough to warrant remediation then we recommend steady flows of at least 400 $m^3 \cdot s^{-1}$ for several months

Table 2. Comparison of benthic biomass from 1991 reference data to monitoring data collected from October 1997 through June 1998 at nine sites in the Colorado River through Grand Canyon. Results of MANOVA are depicted as follows for each biotic factor; (+) resource enhanced significantly, (=) resource maintained, nonsignificant change and (-) resource degraded significantly. Miscellaneous algae, macrophytes and bryophytes are depicted by MAMB. Pool habitats are indicated by P and cobble habitats by C. **Oscillatoria is the only biotic factor that is does not enhance the aquatic food base with increasing biomass, therefore a + means reduced Oscillatoria biomass over 1991 reference data.**

Site/Date	Cladophora	Oscillatoria	Detritus	MAMB	Macroinvertebrates
October 1997					
Rkm 0.0 P	=	+	=	=	+
Rkm 0.8 C	+	=	+	+	+
Rkm 3.1 C	+	=	=	+	=
Rkm 95.7 P	-	=	+	=	=
Rkm 98.6 C	=	-	+	=	+
Rkm 108.8 P	=	+	+	=	=
Rkm 109.6 C	=	-	=	+	=
Rkm 326.4 P	+	+	+	=	=
Rkm 328.8 C	=	=	+	=	=
March 1998					
Rkm 0.0 P	+	+	=	=	=
Rkm 0.8 C	=	=	=	+	=
Rkm 3.1 C	+	=	=	+	+
Rkm 95.7 P	-	+	+	=	=
Rkm 98.6 C	=	-	+	=	+
Rkm 108.8 P	=	+	+	=	=
Rkm 109.6 C	=	=	+	+	=
Rkm 326.4 P	=	+	+	=	=
Rkm 328.8 C	=	-	+	=	=

Table 2 continued

Site/Date	Cladophora	Oscillatoria	Detritus	MAMB	Macroinvertebrates
June 1998					
Rkm 0.0 P	=	_	=	=	=
Rkm 0.8 C	=	=	=	+	+
Rkm 3.1 C	=	_	=	+	+
Rkm 95.7 P	=	+	+	=	=
Rkm 98.6 C	=	+	=	=	=
Rkm 108.8 P	=	+	+	=	=
Rkm 109.6 C	=	=	=	=	=
Rkm 326.4 P	=	_	+	+	=
Rkm 328.8 C	=	+	+	=	=

Table 3. Comparison of benthic biomass from 1991 reference data to data collected in June 1996 at nine sites in the Colorado River through Grand Canyon. This analysis demonstrates how reduced daily flow fluctuations can enhance the aquatic food base. Results of MANOVA are depicted as follows for each biotic factor; (+) resource enhanced significantly, (=) resource maintained, nonsignificant change and (-) resource degraded significantly. Miscellaneous algae, macrophytes and bryophytes are depicted by MAMB. Pool habitats are indicated by P and cobble habitats by C. **Oscillatoria is the only biotic factor that is does not enhance the aquatic food base with increasing biomass, therefore a + means reduced Oscillatoria biomass over 1991 reference data.**

Site/Date	Cladophora	Oscillatoria	Detritus	MAMB	Macroinvertebrates
June 1996					
Rkm 0.0 P	-	=	=	-	=
Rkm 0.8 C	=	=	=	=	+
Rkm 3.1 C	=	=	+	+	+
Rkm 95.7 P	=	=	=	=	=
Rkm 98.6 C	=	=	+	+	+
Rkm 108.8 P	=	=	=	=	=
Rkm 109.6 C	=	=	+	+	+
Rkm 326.4 P	=	+	=	=	=
Rkm 328.8 C	=	=	+	=	+

Objective 2: Monitor the effects of modified low fluctuating flows from GCD on the organic drift in the Colorado River between GCD and Diamond Creek.

Organic drift in the Colorado River below Glen Canyon Dam reflects both the productivity of the benthos and scouring effects of daily fluctuating flows (Shannon et al. 1996; A118-A158). Drift patterns for CPOM show an increase in biomass during the seasonal benthic high for June with lower amounts before and after at all sites in 1997 and 1998. Annual amount of CPOM from all sites was estimated to be 0.13 $g \cdot m^3 \cdot s^{-1}$ (SE$\pm$ 0.04) in 1997. This estimate is above the annual average from 1993-1996 of 0.09 $g \cdot m^3 \cdot s^{-1}$ ($\pm$ 0.04).

Detritus constitutes ~95% of the CPOM drift averaged for all sites in 1997, which is typical. Primary producer biomass in CPOM drift is composed of about 55% Cladophora and 45% MAMB for all sites in 1997. This is a reflection of more phytobenthic diversity in the past couple of years (Shannon et al 1997). Aquatic diptera larvae, pharate pupae and adults comprised the majority of the CPOM drift of macroinvertebrates. In 1997 miscellaneous macroinvertebrates contributed more to macroinvertebrate portion of CPOM since 1995.

Objective 3. Assess the benthos and drift of major tributaries in Grand Canyon National Park.

Collection trip has not yet occurred.

Objective 4. Construction of an aquatic/riparian food web using stable isotope analysis.

Dual stable isotope analysis of 672 samples are currently being processed by the University of Georgia, Institute of Ecology. This batch of samples also includes Little Colorado River spring samples for construction of a food web in this area with USFWS.

CHAPTER TWO Comparison of processing rates between exotic and native riparian vegetation in the regulated Colorado River, Arizona

ABSTRACT.

Processing rates for leaves of the native willow (Salix exigua) and cottonwood (Populus Fremontii) were compared to those of the exotic Tamarix pentandra in the regulated Colorado River. Leaf packs of each species were incubated in the river at Lees Ferry, Arizona, approximately 26 km below Glen Canyon Dam. There was a significant difference in processing rate between species, with cottonwood showing the fastest breakdown. After 140 d, only 20% of the cottonwood leaf mass remained with 30% of the tamarisk and >50% of the willow leaf mass remaining.

The k_d value for cottonwood was 0.0062 compared to 0.0049 and 0.0038 for tamarisk and willow, respectively. Macroinvertebrate colonization was not significantly different between native and exotic plant species throughout the breakdown period. Oligochaetes were the most abundant macroinvertebrate colonizing the leaf packs. Processing rates for all leaves were generally slower in the tailwaters of Glen Canyon Dam compared to unregulated lotic systems reported in the literature. This is likely due to the absence of common insect shredders, cold and constant water temperatures (≤10°C), and slower metabolic rates by microbes.

INTRODUCTION

There have been numerous studies on the decomposition rates of natural riparian vegetation in unregulated streams, but only a few have examined this process in regulated systems (Boulton and Boon 1991). In addition, few studies have compared processing rates of native vegetation to that of Tamarix (salt cedar), a plant species that has invaded many regulated lotic streams in southwestern USA (Stevens 1989). Microbes and macroinvertebrates play a key role in the processing of leaf litter that enter stream ecosystems (Petersen and Cummins 1974, Boulton and Boon 1991, Allan 1996). Any changes in the composition of these processing agents and/or the type of leaf material being processed may have dramatic implications on the overall foodweb in lotic ecosystems.

Terrestrial inputs (allochthonous materials) are critical energy sources to stream foodwebs (Fisher and Likens 1973, Boulton and Boon 1991, Gregory et al. 1991, Allan 1996). For example, Fisher and Likens (1973) reported that nearly 99% of the

energy in Bear Brook, New Hampshire was derived from riparian vegetation. Although considerably less energy is supplied to rivers in arid biomes (Minshall 1978), riparian and upland vegetation are still important potential energy sources in the impounded Colorado River ecosystem.

Impoundments, disturbances from cattle grazing, and agriculture can enhance the ability for exotics to outcompete native species. For example, growth of native species under a Tamarix canopy is inhibited by hydrophobization of the soil which is caused by the leaching of resins or sugars from tamarisk leaves or by duff accumulation which can be up to 150 cm deep compared to willow which averages 1 cm deep (Stevens 1989). Tamarix also releases salt from its leaf glands, creating a harsh local environment for native plants. Salt cedar has invaded nearly every creek, stream and river in the American Southwest (Correll and Correll 1972), and therefore it is important that we understand the impact this invasion will have on lotic communities.

We compared the rate of leaf decomposition of two native riparian species with the exotic Tamarix pentandra Pall. in a leaf pack study in the regulated Colorado River below Glen Canyon Dam, AZ. All three plant species are present in the riparian community along sections of the Colorado River between Lees Ferry and Lake Mead. We also compared the colonization rates of the macroinvertebrate community associated with each leaf pack treatment. Comparisons are made between leaf decomposition in the regulated Colorado River and other studies in unregulated rivers.

METHODS

During November of 1997, abscised leaves of cottonwood (Populus Fremontii Watts), sandbar willow (Salix exigua Nutt.) and tamarisk (Tamarix pentandra Pall.) were collected from the ground at Lees Ferry, Arizona and brought to Northern Arizona University to air dry. Leaves were stored until the experiment was initiated in February 1998. In order to reduce fragmentation due to handling, air-dried leaves of each species were soaked in distilled water for 5 min (Benfield 1996). Leaves (4 g) of each species were then placed into separate packs (15 x 15 cm) constructed from plastic window screen with 1 mm mesh openings. The screen material for each pack was cut 30 cm long and 15 cm wide and folded in half and bound along two edges with packaging tape and staples. In order to determine the mass of the initial leaf pack, five, 4 g samples of each leaf type were weighed from a separate batch of

leaves. Leaves were then ashed at 500° C for 1 h and ash-free dry mass (AFDM) was calculated (APHA 1989).

Twenty-five leaf packs of each leaf type were placed into the Colorado River at Lees Ferry, AZ, approximately 26 km below Glen Canyon Dam, on February 14, 1998. Leaf packs were fastened to a chain in moderate current velocity (~0.02 m s^{-1}), weighted with concrete filled buckets, and lowered into the river bed to a depth of 2 m. Two types of control treatment packs were established by filling mesh packs with plastic cut to the approximate area of leaves to mimic leaf pack treatments. In addition, empty leaf packs were employed to test for macroinvertebrate colonization on the plastic screen. Three packs of each of the two controls were placed into the river.

Five replicate packs of each plant species were retrieved from the river at the following time intervals starting on 16 February 1998: 2, 21, 46, 84 and 142 d. Macroinvertebrates on the outside of the packs were removed to prevent counting animals which were only using the screen mesh as a substrate. Each pack was placed into a separate plastic tuperware container with additional filtered river water, packed on ice, and brought to the laboratory for processing. Macroinvertebrates in each leaf pack were sorted within 24 h to the lowest taxonomic category, enumerated, dried in an oven at 60-70°C, and weighed. AFDM for total invertebrates was then calculated using a regression equation (Shannon et al. 1996). Leaf material was then placed into a preweighed crucible and allowed to oven dry at 60° C. Dry weight of the leaves was calculated and AFDM determined by ashing material at 500° C for 1 h.

On day 84, diatom AFDM was measured on each leaf pack to determine the relationship between diatom (food) and macroinvertebrate mass. Once all macroinvertebrates and leaf material were removed, the sorting water for each sample was allowed to settle in separate beakers. Diatoms from each sample were filtered onto glass fiber membranes (GFC). Diatom AFDM was then calculated for each sample as described above.

River discharge ranged from 19,600 to 19,800 cfs during the study. Water velocity was measured with a Marsh McBirney electronic flow meter and specific conductance, dissolved oxygen, pH, and water temperature were determined with a Hydro-Lab Scout II during each visit to the study site ($\underline{n}$ = 6 measurements).

The percentage of leaf mass remaining at each time interval was log transformed and regressed against time. The slope of the regression line is the decay rate coefficient

(k_d) and is the fractional loss rate per day. For example, the k_d value for cottonwood (0.0062 d^{-1}, Fig. 1) indicates a loss rate of 0.62% per day.

Multivariate analysis of variance (MANOVA) was used to analyzed predictor variables (species densities) for each leaf pack treatment. Analysis of variance was performed for testing differences in macroinvertebrate mass between leaf treatments. All calculations were performed with SYSTAT computer software (Version 5.1, Wilkinson 1989).

RESULTS

There were significant differences ($p < 0.05$) in decomposition rates for the three leaf types over the 142 d incubation period in the Colorado River below Glen Canyon Dam. Decomposition rates were fastest for cottonwood (k_d = 0.0062 or 0.62% d^{-1}) and slowest for willow (k_d = 0.0038 or 0.38% d^{-1}) with intermediate values for tamarisk (K_d = 0.0049 or 0.49% d^{-1}; Fig. 1). After 142 d, cottonwood lost nearly 80% of its initial mass compared to 70% for tamarisk and 48% for willow. The greatest loss in mass occurred after 2 d incubation for all taxa with cottonwood losing 48%, tamarisk 28% and willow 18% of their respective initial masses (Fig. 1). The rate of loss in leaf mass declined after 46 d.

Water temperature was low and constant (8.9°C±0.1) during the study period. In addition, dissolved oxygen averaged 10.8 mg/L and pH averaged 7.4, with an average value of 0.73 mS for specific conductivity during the incubation period.

Multivariate analysis showed that densities of all macroinvertebrate taxa were significantly ($p < 0.001$) different between collection dates, but only gastropods were significantly different ($p = 0.002$) between leaf types (Table 1). Gastropods preferred willow over tamarisk and cottonwood by a factor of 2:1. Oligochaetes dominated all leaf packs (≥80%) for all intervals except after 2 d when chironomids were dominant (Fig. 2). After 2 d, leaf packs averaged only 1.0, 0.4 and 0.8 total animals per leaf pack for cottonwood, willow and tamarisk, respectively (Fig. 2). Average macroinvertebrate densities increased by nearly 20-fold by day 21 and continued to increase until day 84 afterwhich there was nearly a 6-fold decrease in average macroinvertebrate density at 142 d (Fig. 2). Tamarisk showed overall slower colonization by macroinvertebrates than the other two leaf types, with little increase from 21 d to 46 d. However, after 84 d tamarisk slightly exceeded cottonwood in

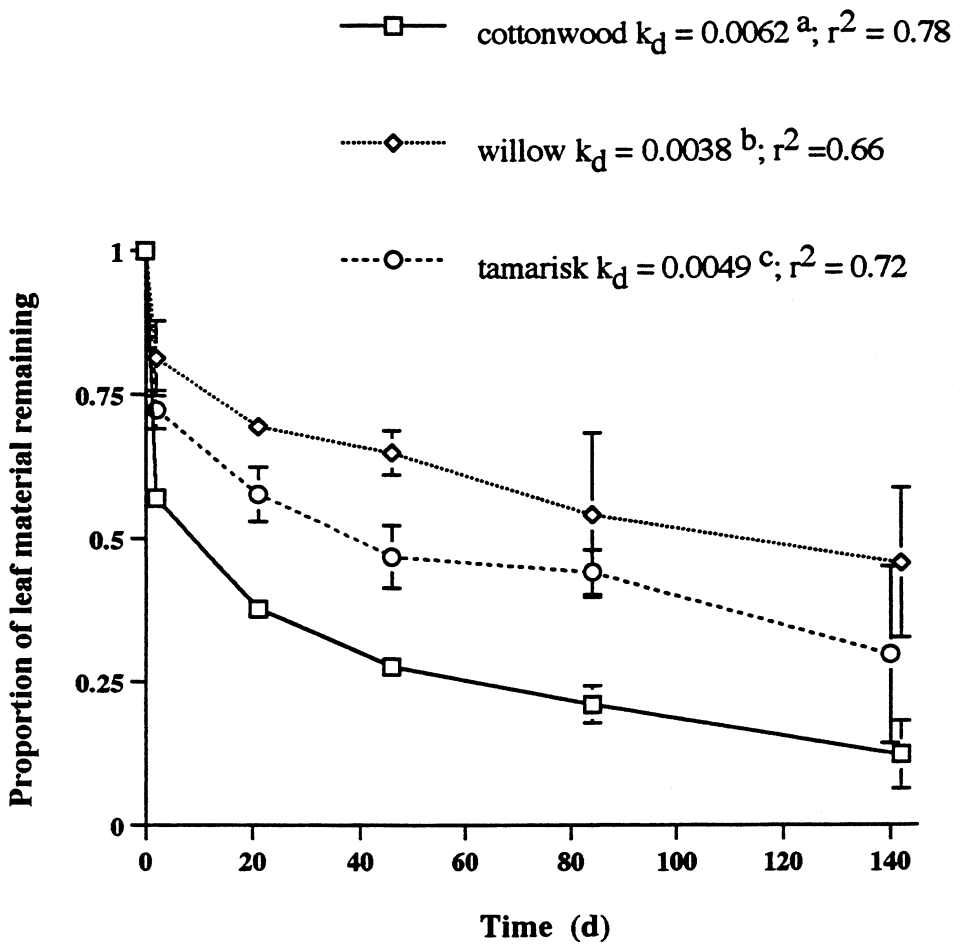

Figure 1. Proportion of remaining leaf mass (mean ±SE) as a function of decomposition rates for three leaf types in the Colorado River below Glen Canyon Dam, AZ. Superscript on kd values in legend indicates significantly different values.

Table 1. Multivariate analysis comparing macroinvertebrate densities on three leaf types during a 142-d leaf decomposition experiment in the Colorado River below Glen Canyon Dam, Arizona. Biotic response variables were lumbriculids (l), Gammarus (g), tubificids (t), simuliids (b), chironomids (c), snails (s), miscellaneous macroinvertebrates (m). All significant response variables listed are for probabilities <0.04. (n = 75).

Source	Wilks' Lambda	df	F-ratio	p	Significant Response Variable
Collection Date	0.10	28,196	6.25	<0.001	l,g,t,b,c,s
Leaf Type	0.55	14,108	2.67	0.002	s
Collection Date x Leaf Type	0.32	56,296	1.23	0.142	s

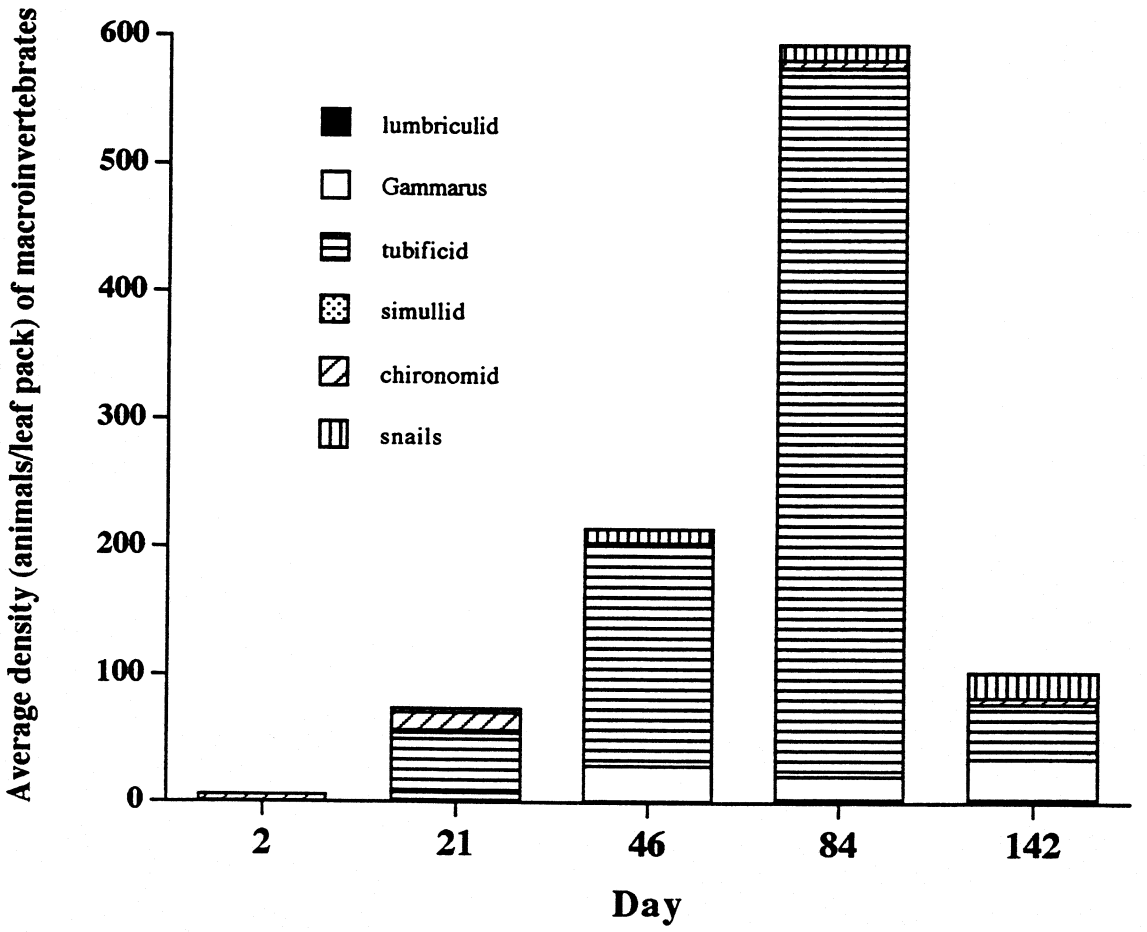

Figure 2. Average densities (animals/leaf pack) of macroinvertebrate taxa for all leaf packs combined during incubation in the Colorado River below Glen Canyon Dam, AZ.

macroinvertebrate density, while macroinvertebrate density on willow was nearly three times higher than for either tamarisk or cottonwood.

There was a significant (<0.001) difference in macroinvertebrate AFDM between collection dates (Fig. 3), but no significant difference occurred between leaf treatments (Table 2). Overall mass increased through time for each collection interval, except for day 142 when mass decreased.

At day 84, cottonwood leaves had the lowest diatom mass (0.049 g/leaf pack; ±0.01), willow leaves had an intermediate mass (0.046 g/leaf pack; ±0.001), and tamarisk leaves had the highest diatom mass (0.032 g/leaf pack; ±0.001).

DISCUSSION

Leaf pack decomposition rates were considerably slower in the regulated Colorado River below Glen Canyon Dam than breakdown rates recorded in other unregulated southwestern USA streams. Daily breakdown rates for cottonwood, tamarisk and willow in the Colorado River at Lees Ferry were 0.62%, 0.49%, and 0.38% per day, respectively. In contrast, Schade and Fisher (1997) reported that cottonwood and willow had decomposition rates of 1.25% and 0.77% per day, respectively, in Sycamore Creek, AZ. Although both lotic environments had similar macroinvertebrate assemblages (dipterans and oligochaetes), water temperatures were considerably different between the two systems. Water temperatures in the constantly cold tailwaters of Glen Canyon Dam were considerably lower (<10°C) than those in Sycamore Creek (>20°C). These reduced temperatures in the regulated Colorado River likely reduced microbial activity and slowed decomposition rates of all leaf packs (Peterson and Cummins 1974).

Furthermore, the absence of insect shredders such as caddisflies and stoneflies, further reduced decomposition rates of all leaf packs in the Colorado River. The invertebrate assemblage below Glen Canyon Dam has been primarily reduced to chironomid larvae, oligochaetes, gastropods, and the amphipod, Gammarus lacustris (Stevens et al. 1997). All of these organisms are relatively weak shredders of allochthonous materials compared to stoneflies and caddisflies (Merritt and Cummins 1996).

As reported in other decomposition studies (Peterson and Cummins 1974, Boulton and Boon 1991, Benfield 1996), the greatest loss in leaf mass for all packs occurred within the first two days of incubation. This loss is likely due to leachates leaving the leaf material (Peterson and Cummins 1974). The slower breakdown rates for willow and

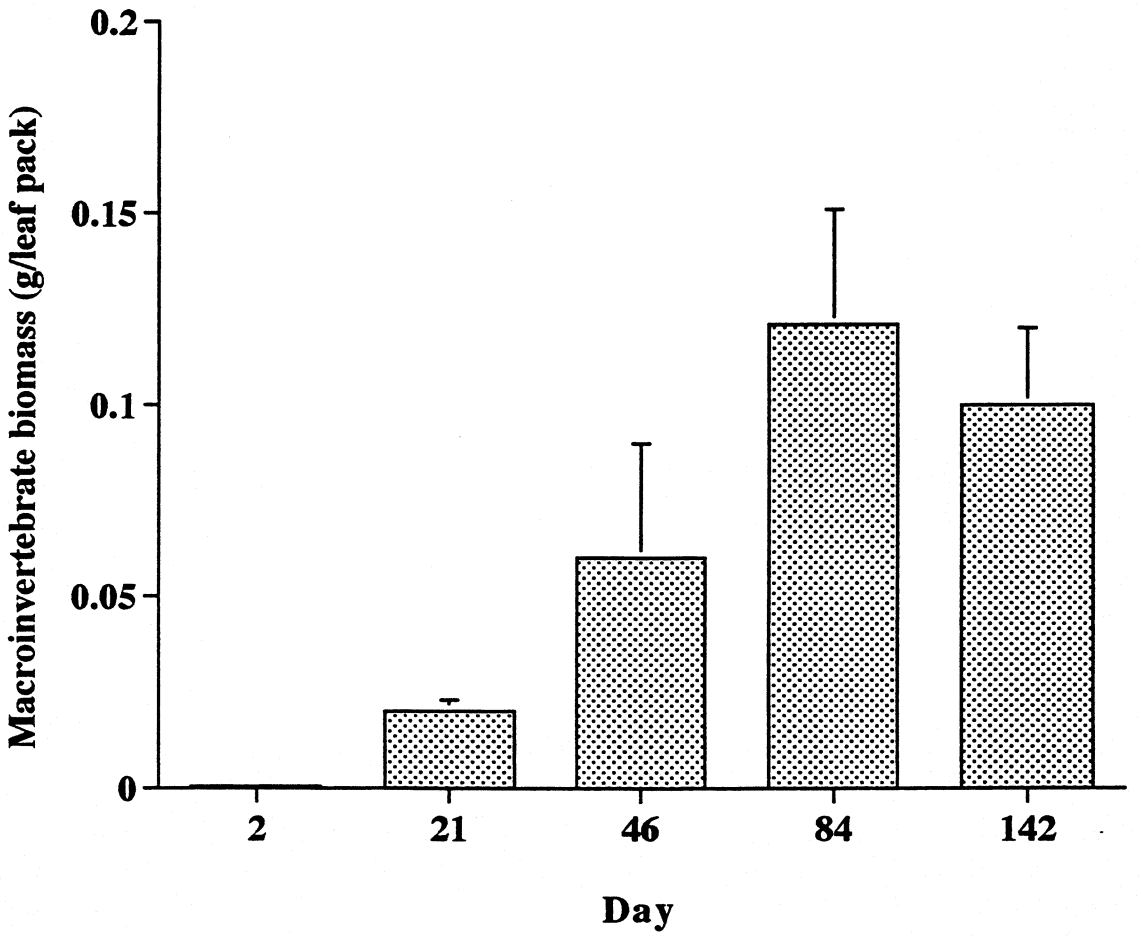

Figure 3. Average macroinvertebrate biomass (g/leaf pack, ±SE) on all leaf types (cottonwood, willow, and tamarisk) during a 142 d incubation period in the Colorado River below Glen Canyon Dam, AZ.

Table 2. Analysis of variance comparing macroinvertebrate biomass (AFDM) estimates colonizing leaf packs during a 142-d leaf decomposition experiment in the Colorado River, below Glen Canyon Dam, Arizona. (n = 75)

Source	Sum of Squares	df	F-ratio	p
Collection Date	0.15	4	11.73	<0.000
Leaf Type	0.01	2	1.49	0.232
Collection Date x Leaf Type	0.01	8	0.23	0.983

tamarisk compared to cottonwood may have resulted from the presence of secondary compounds in the two former plants which inhibited both microbial breakdown or macroinvertebrate processing (Rosenthal and Berenbaum 1979, Driebe 1997). However, there was no significant difference in macroinvertebrates densities and composition between leaf types. Additional work is needed on the roles of secondary compounds on microbial processing, especially in regulated systems. Schade and Fisher (1997) also found that cottonwood decayed the fastest in Sycamore Creek, AZ.

Conditions such as varied discharges, constantly low temperatures and modified food sources cause a decrease in biological diversity below tailwaters (Munn and Brusven 1991, Blinn and Cole 1991). Without key invertebrates shredders, allochthonous energy contributions may be decreased due to reduced rate of decomposition. Further leaf decomposition studies are needed in streams located in disturbed, arid biomes to understand the role of invading riparian species on trophic webs in aquatic systems.

Chapter Three: Management Considerations

During the past year of monitoring and research we continue to develop a large data base to compare patterns which will add to our general knowledge of the aquatic community structure in the Colorado River through Grand Canyon. This knowledge can then be used in assisting the adaptive management process in determining how best to operate Glen Canyon Dam to conserve resources.

Discharge In 1997, we got our first exposure to the full use of the E.I.S. mandated flow regimes. These flows included a minimum flow of 142 $m^3 \cdot s^{-1}$ (5K cfs) with a maximum flow of 708 $m^3 \cdot s^{-1}$ (25K cfs) with an up-ramp of no more than 114 $m^3 \cdot s^{-1}$ (4K cfs) per hour and a down ramp of no more than 43 $m^3 \cdot s^{-1}$ (1.5K cfs) per hour. Total daily fluctuations were to not exceed 142, 170, 227 $m^3 \cdot s^{-1}$ (5K, 6K, or 8K cfs) for low, moderate or high delivery months respsectively (Figure 1)

Three types of flows occured in 1997 and 1998; flows to lower the level of Lake Powell, flows to raise the level of Lake Powell, and research flows. Flows until September 1997 were of high discharge with minimal daily fluctuations in order to reduce the water level of Lake Powell. In September a three day low flow occured for overflight purposes. These discharges were followed by daily fluctuation flows, within EIS criteria, which averaged about 550 $m^3 \cdot s^{-1}$ (20K cfs) that continued through April. This was the first attempt at lowering Lake Powell using daily fluctuating flows >550 $m^3 \cdot s^{-1}$ (20K cfs). These types of flows have not occurred since 1991 during research flows of GCES Phase II. Also in November there was small spike

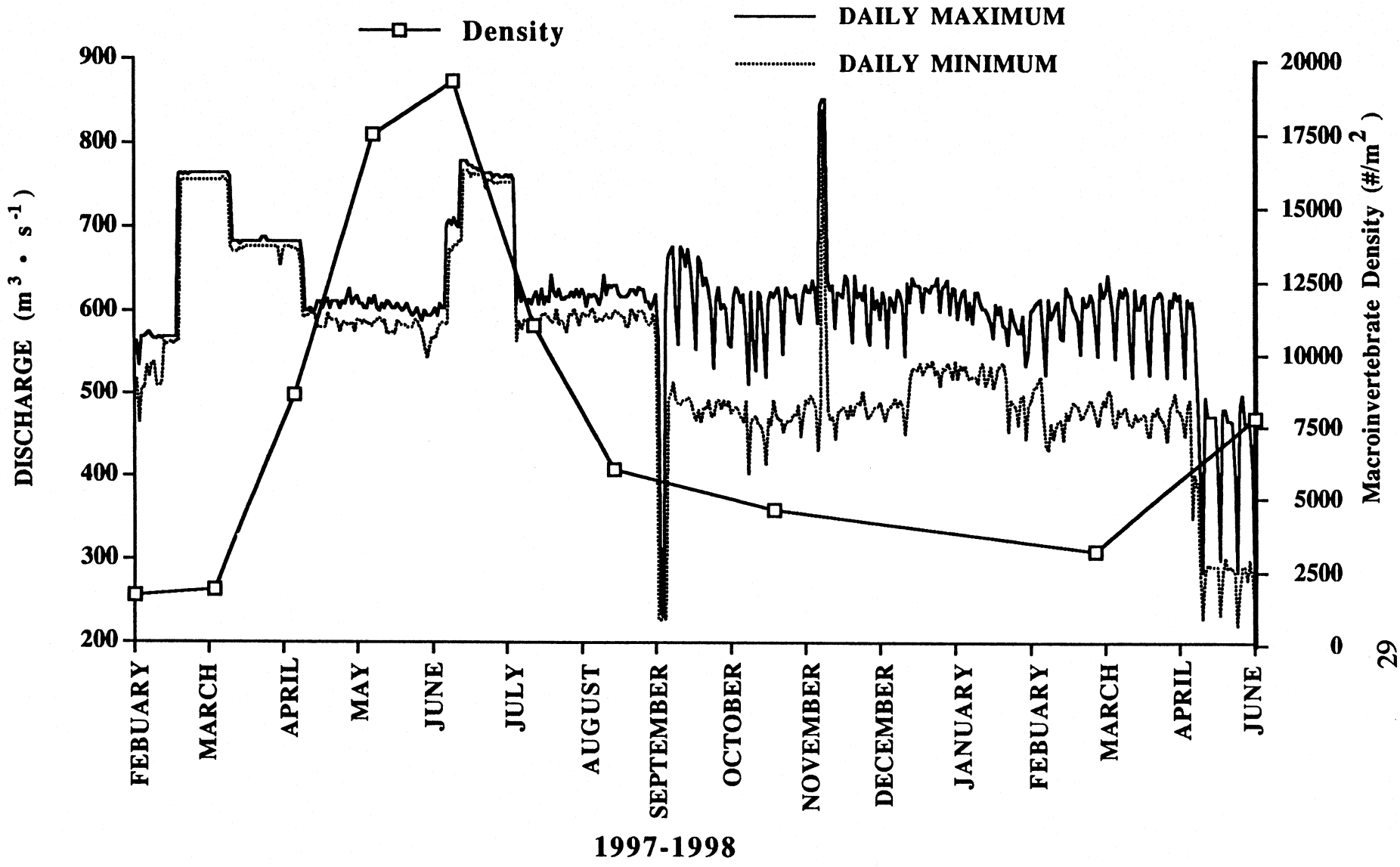

Figure 1. Comparison of daily flows ($m^3 \cdot s^{-1}$) from Febuary 1997 through June 1998 and average macroinvertebrate density (#/m) from cobble habitats from Rkm 0.8 and 3.1.

flow of ~ 800 $m^3 \cdot s^{-1}$ (~30K cfs) to conserve sand after unusally high flows from the Paria River during August. In April, it was decided by dam operators that Lake Powell was now low enough and conserving water was in order so the previous minimum flows became the daily maximum flows.

In order to examine these flows we compared macroinvertebrate density (#/m^2) on cobble habitats and the discharge pattern from February 1997 through June 1998 (Figure 1). Data from Lees Ferry (Rkm 0.8) and Two-Mile Wash (Rkm 3.1) was averaged for each collection trip. These two sites give a fair indication of the study site because of contrasting suspended sediment loads. This general comparison of density to discharge indicates that the seasonal peaks, typically in June, are apparent however the June 1998 peak is about 50% of the 1997 peak. Winter lows are higher in 1998.

Two conclusions can be drawn from this comparison of flows and macroinvertebrate density; the less variable flows in early 1997 were more conducive to aquatic food base growth than the flows after September 1997. Furthermore, with such dramatic changes in flow regime it was hard to determine the cause and effect for aquatic resources. Shannon et al (1998) more fully describes the impacts of the September 1997 draw down and daily fluctuating flows on the aquatic food base.

Biotic response times need to be considered when flow patterns are modified. When the GCMRC staff and stakeholders design a research or management flow, ability to discern cause and effect on a resource needs to be a top priority. The 1996 Spike Flow was difficult to accurately assess because there were really five flow regimes examined; 1) late winter flows of 1996, 2) steady pre-spike, 3) the spike flow itself, 4) steady post-spike flow, and 5) high relatively steady flows through the summer which were not like the late winter flows. Perhaps the projects that study abiotic variables such as sand can define these variables through discreet sampling techniques, but the biologists have more difficulty with variable flow regimes. This is very important when designing Beach Building Flows and the trigger criteria for initiatings these flow regimes . According to draft documents on the GCMRC web-page as of 12/07/98, a two-day Spike Flow could be followed by daily fluctuations of between 550 and 800 $m^3 \cdot s^{-1}$ (~20K - 30K cfs). This was stated as good way to reduce the level of Lake Powell following a spike flow because the trigger criteria would mean > 140% run-off. The sedimentologists have decided that large daily fluctuations may not adversly impact the newly formed beaches any more than steady flows. Also, these flows would allow for maximum hydro-electric production. These post flows will then be much different than the pre-spike flow and radically different than any flows seen in the past 9 years which will create sampling and analysis problems for understanding aquatic food base patterns.

Benthic recovery from the scouring that will occur with the Spike Flow will take a protracted period of recovery under this proposed flow regime. Blinn et al. (1995) concurred with Angardi and Kubly (1993) that daily fluctuating flows will require over one year for complete recovery in the Glen Canyon reach and even longer in Marble and Glen Canyons due to suspended sediments. The aquatic food base is not in the same robust condition in 1998 as it was in 1996 (see chapter 1). Therefore it is not hard argue that the native fish and trout that depend on this resource are also not in the same condition as in 1996. It is vastly easier to examine and research sand movement than aquatic life; this does not justify a single resource management plan (Frissell and Bayless 1996; Stanford et al. 1996).

Temperature Investigation of leaf processing rates in the Colorado River has provided critical information on carbon cycling and the microbial community. Decay rates are very similar to other cold water rivers indicating that the microbes below Glen Canyon Dam are cold water tolerant. The consistently cold water of the Colorado River has selected for cold stenothermic midges and amphipods, the microbes are no different. Short term increases in water temperature will probably negatively impact these taxa and uncouple carbon processing. The environmental analysis of thermal modification of Glen Canyon Dam by the Bureau of Reclamation, GCMRC and it's stakeholders is not adequate. Again single resource mangement, in this case humpback chub, is not a sound mangement style (Frissell and Bayless 1996, Stanford et al. 1996). We ask that GCMRC through it's adaptive management process to initiate a complete environmental impact statement on the thermal modification of Glen Canyon Dam so that adequate baseline data can be gathered on the most fundamental aspects of the aquatic food base. The problem of not knowing what the pre-Glen Canyon Dam conditions were (baseline data) will always haunt resource managers, knowingly repeating the same mistake is not an astute decision.

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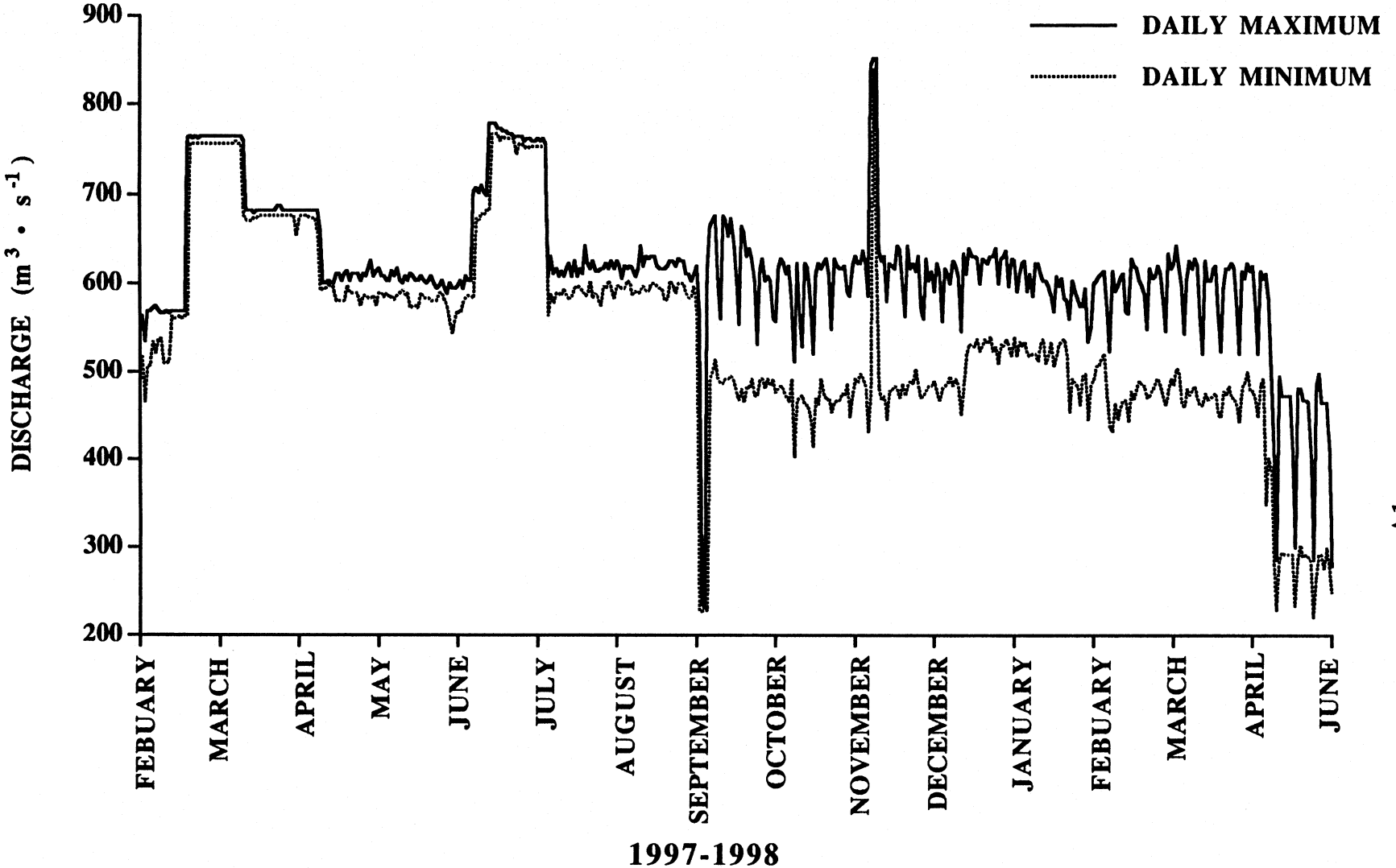

Figure 1. Daily maximum and daily minimum discharge ($m^3 \cdot s^{-1}$) from Lees Ferry Gauging Station Rkm 0.0 from Febuary 1997 to June 1998, obtained from the USGS ADAPS program.

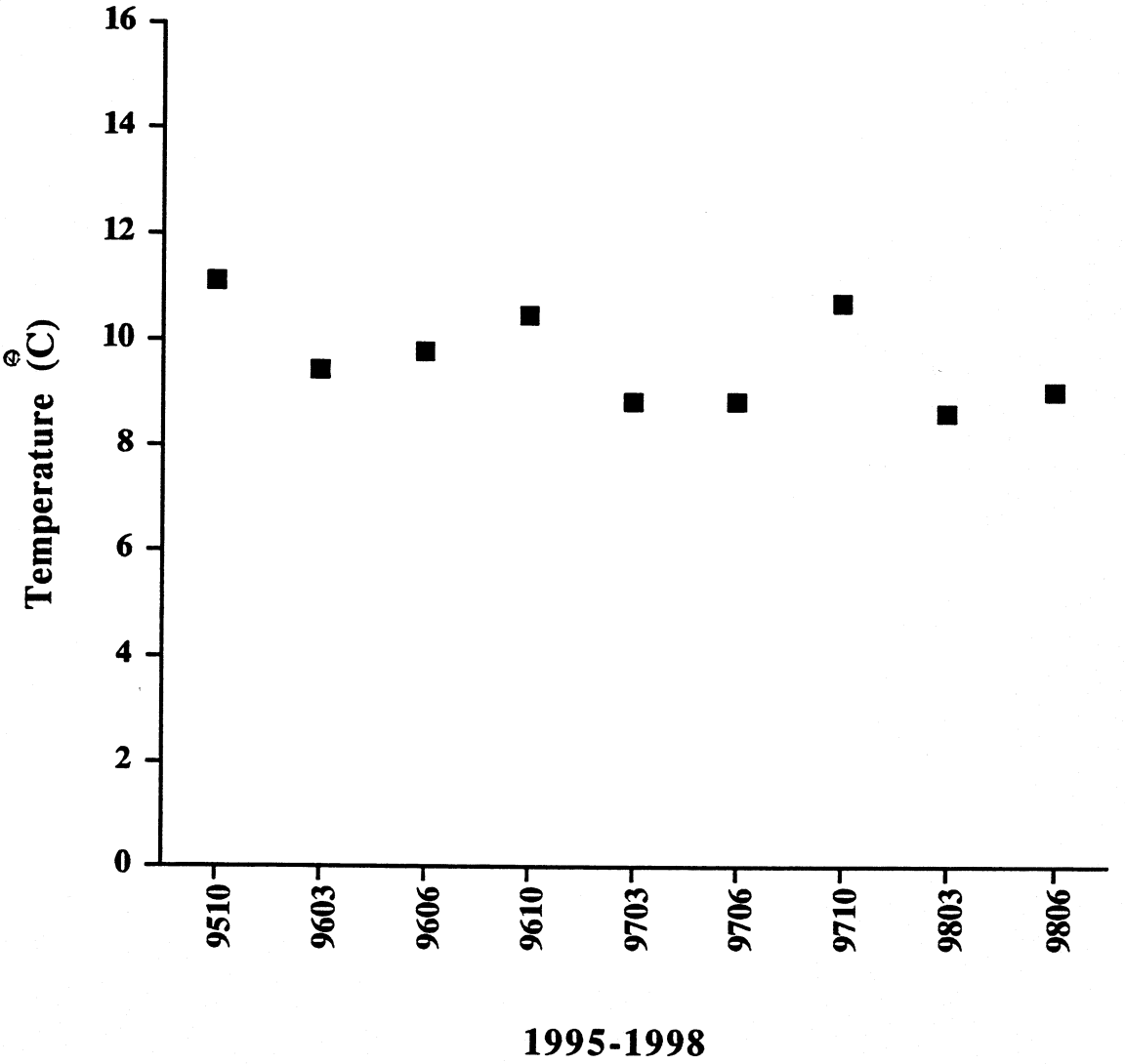

Figure 2. Water temperature (°C) collected at Glen Canyon Gauge Rkm -23.2 from October 1995 to June 1998.

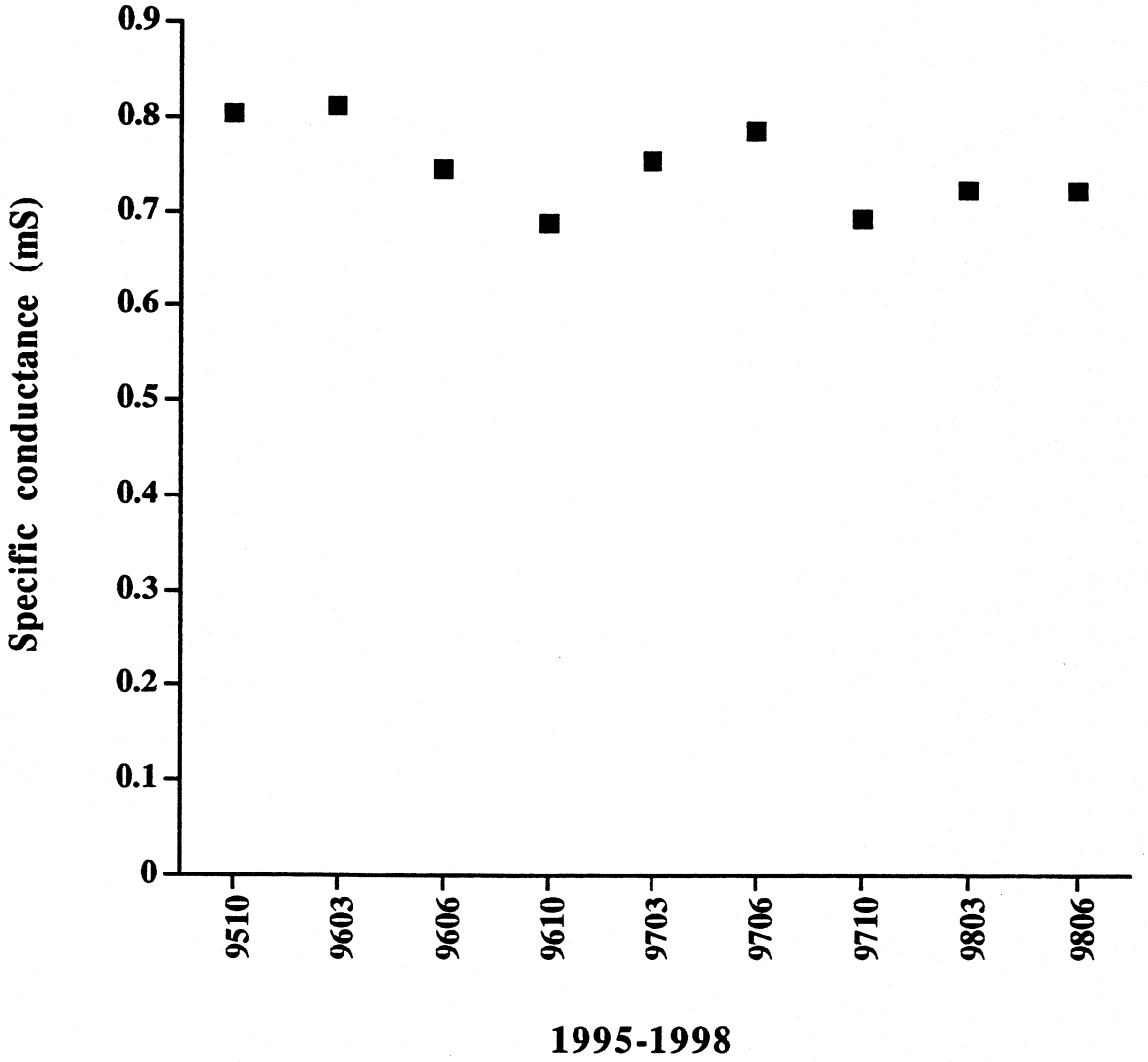

Figure 3. Specific conductance (mS) collected at Glen Canyon Gauge Rkm -23.2 from October 1995 to June 1998.

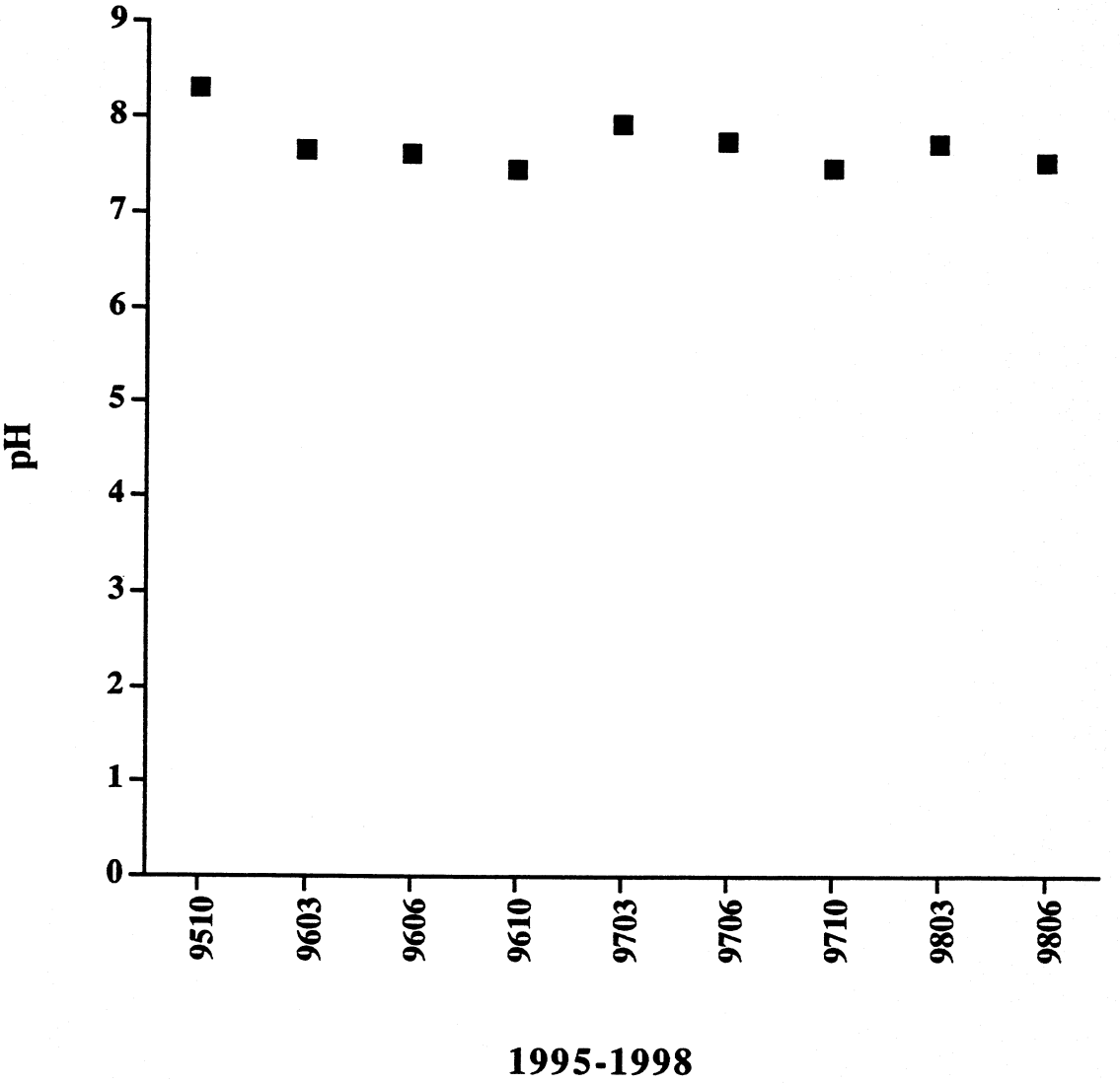

Figure 4. pH collected at Glen Canyon Gauge Rkm -23.2 from October 1995 to June 1998.

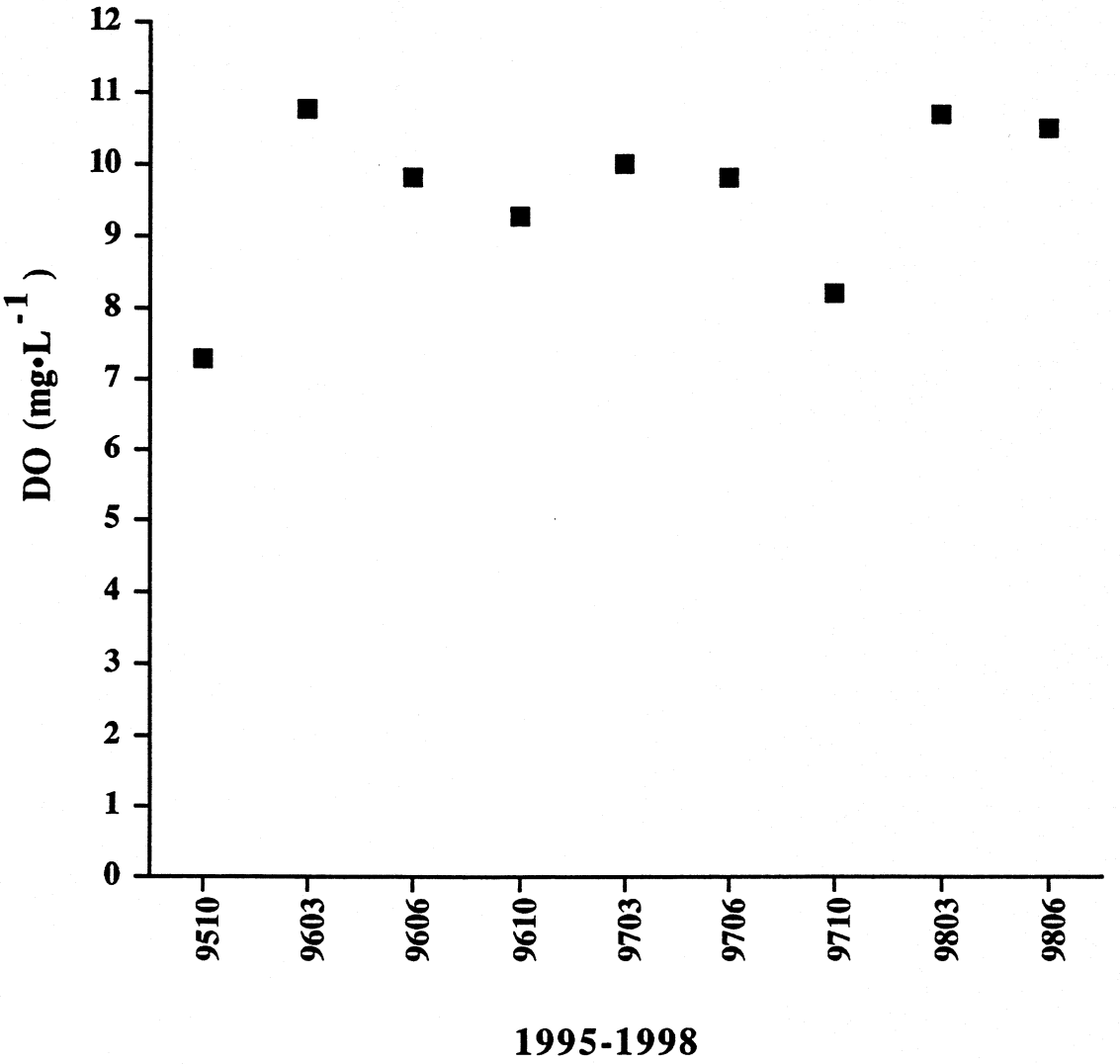

Figure 5. Dissolved oxygen ($mg \cdot L^{-1}$) collected at Glen Canyon Gauge Rkm -23.2 from October 1995 to June 1998.

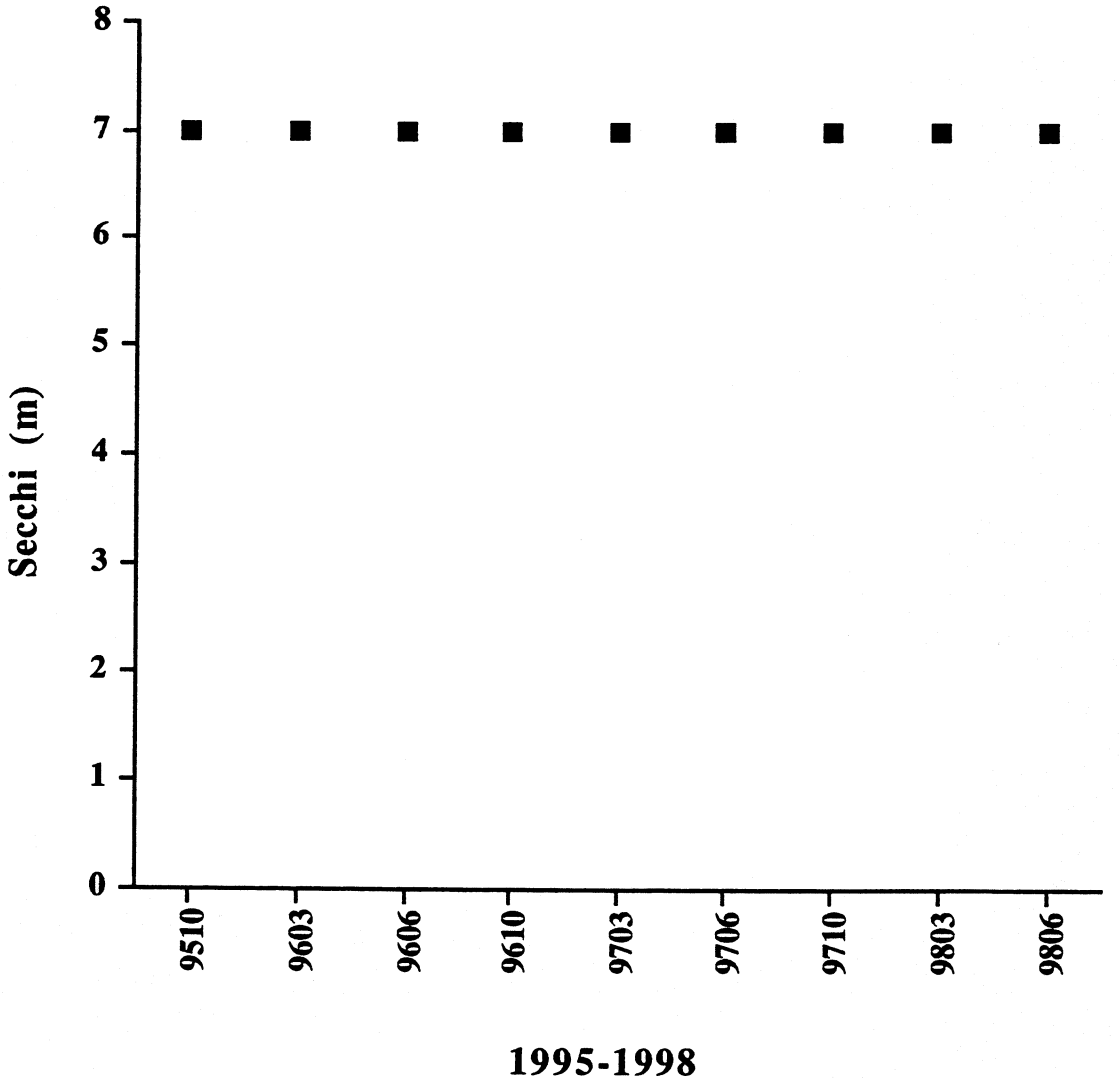

Figure 6. Secchi depth (m) collected at Glen Canyon Gauge Rkm -23.2 from October 1995 to June 1998.

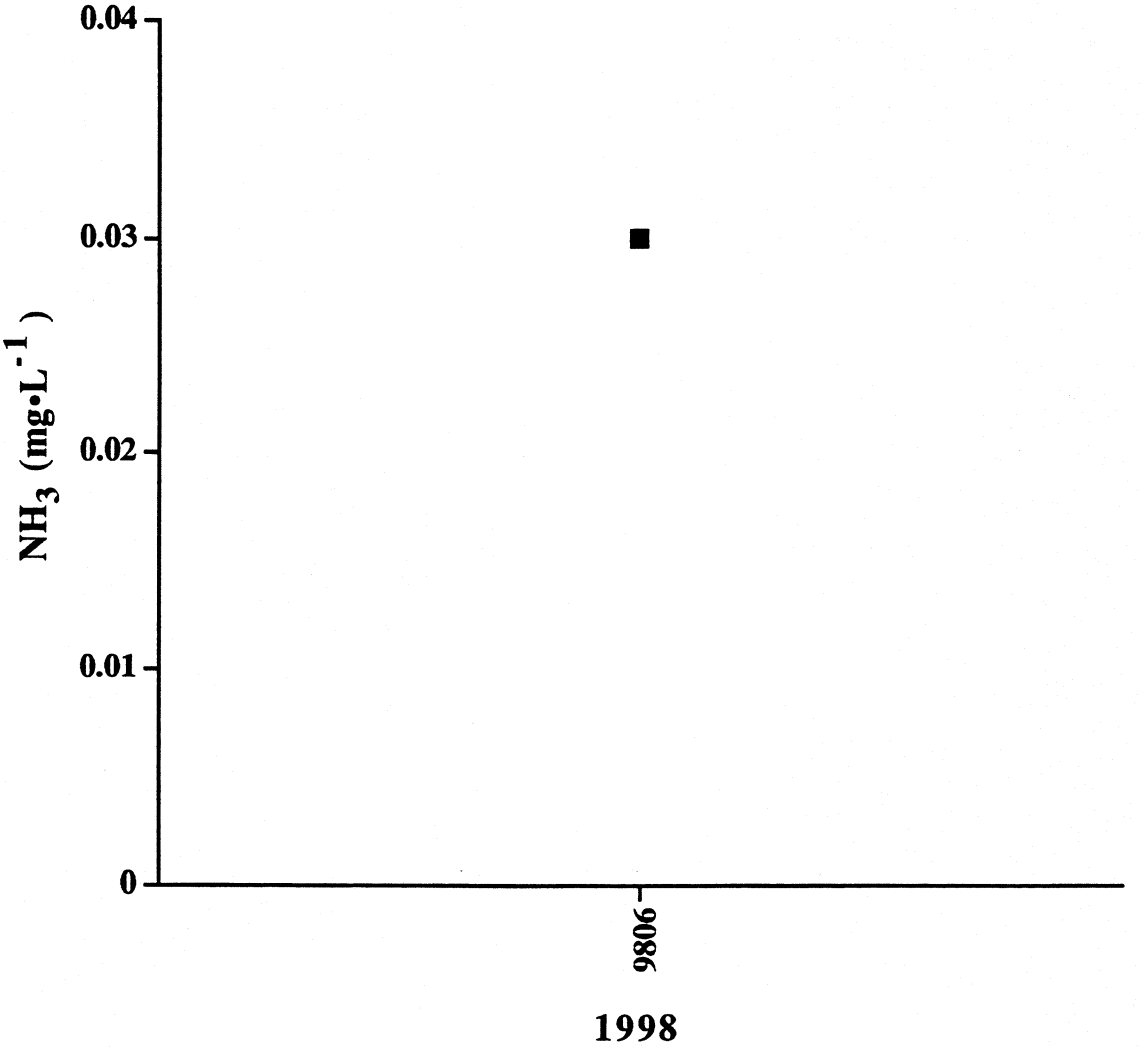

Figure 7. Ammonia (NH_3 mg•L^{-1}) collected at Glen Canyon Gauge Rkm -23.2 in June 1998.

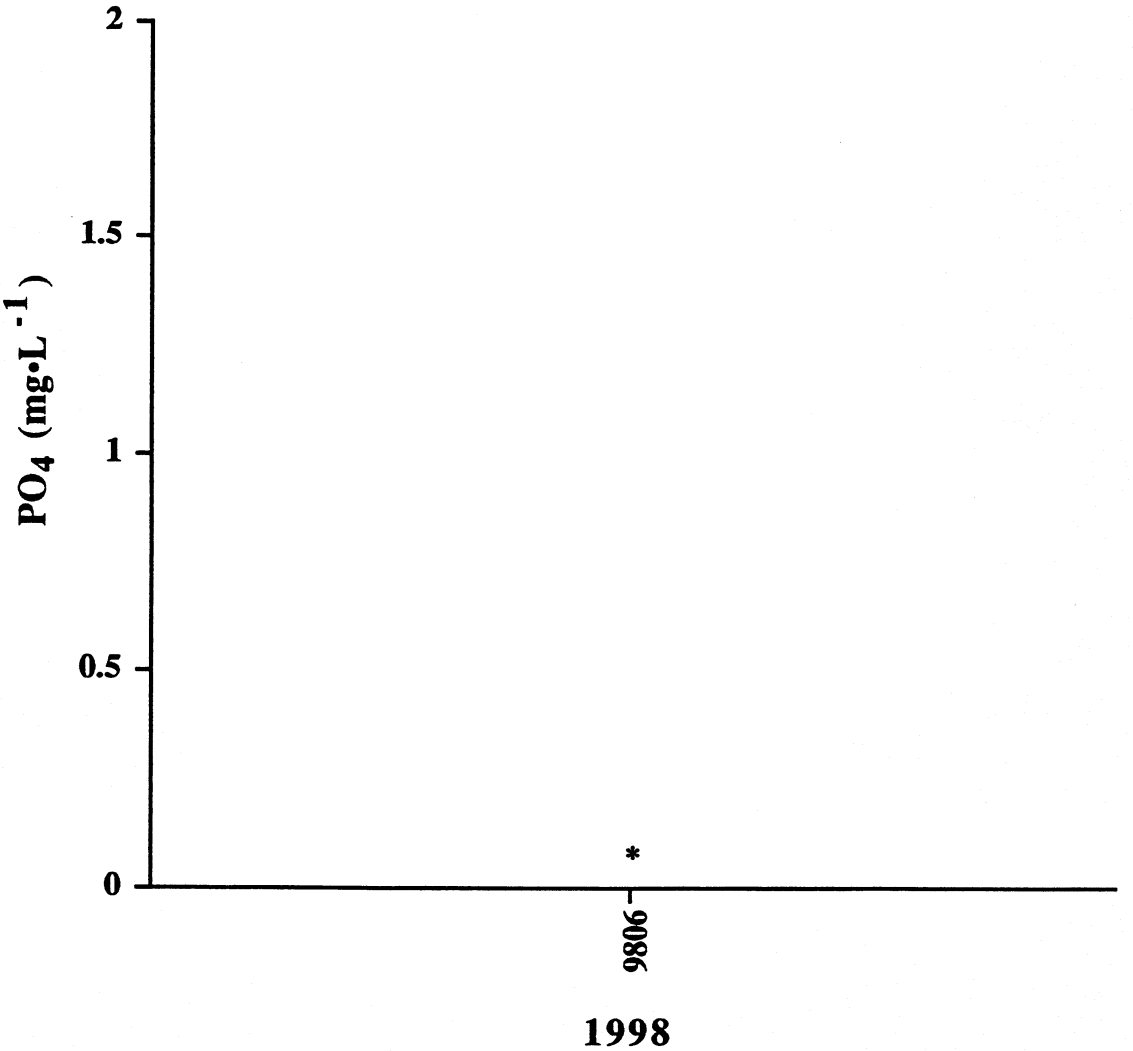

Figure 8. Phosphate (PO_4 mg•L^{-1}) collected at Glen Canyon Gauge Rkm -23.2 in June 1998. Samples below detectable levels are represented by (*).

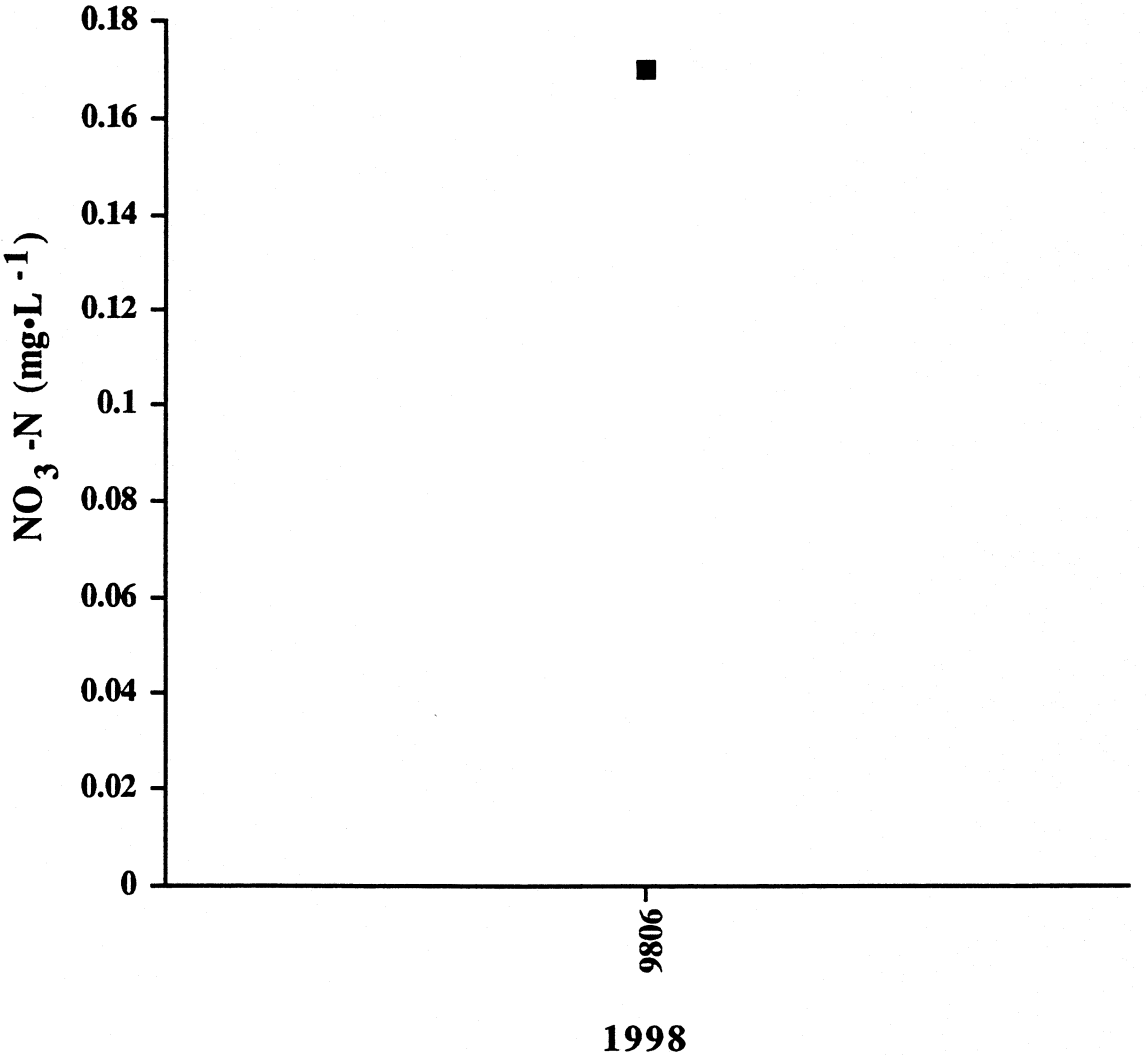

Figure 9. Nitrate-nitrogen (NO_3-N mg•L^{-1}) collected at Glen Canyon Gauge Rkm -23.2 in June 1998.

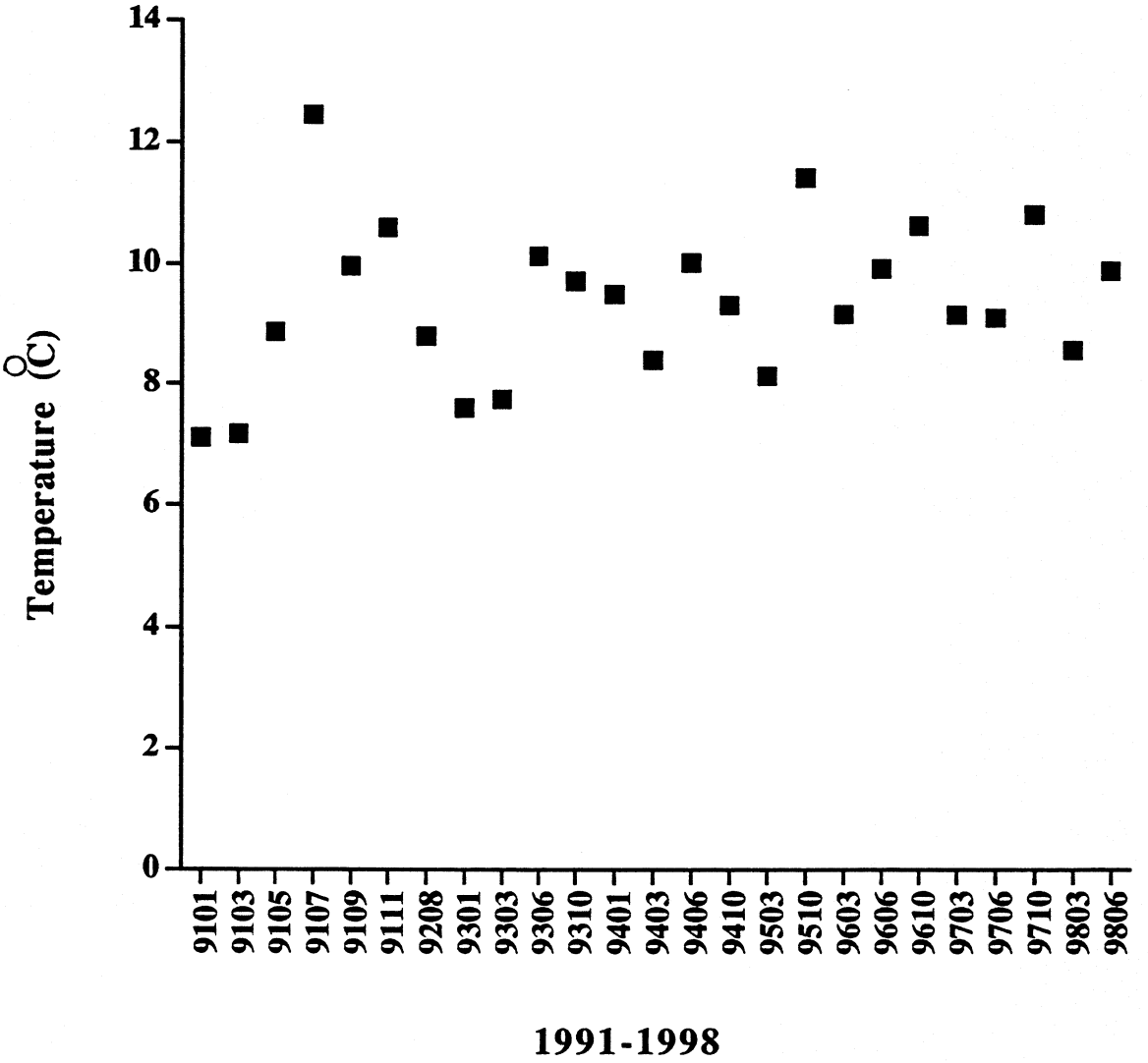

Figure 10. Water temperature (°C) collected at Lees Ferry Rkm 0.0 from January 1991 to June 1998.

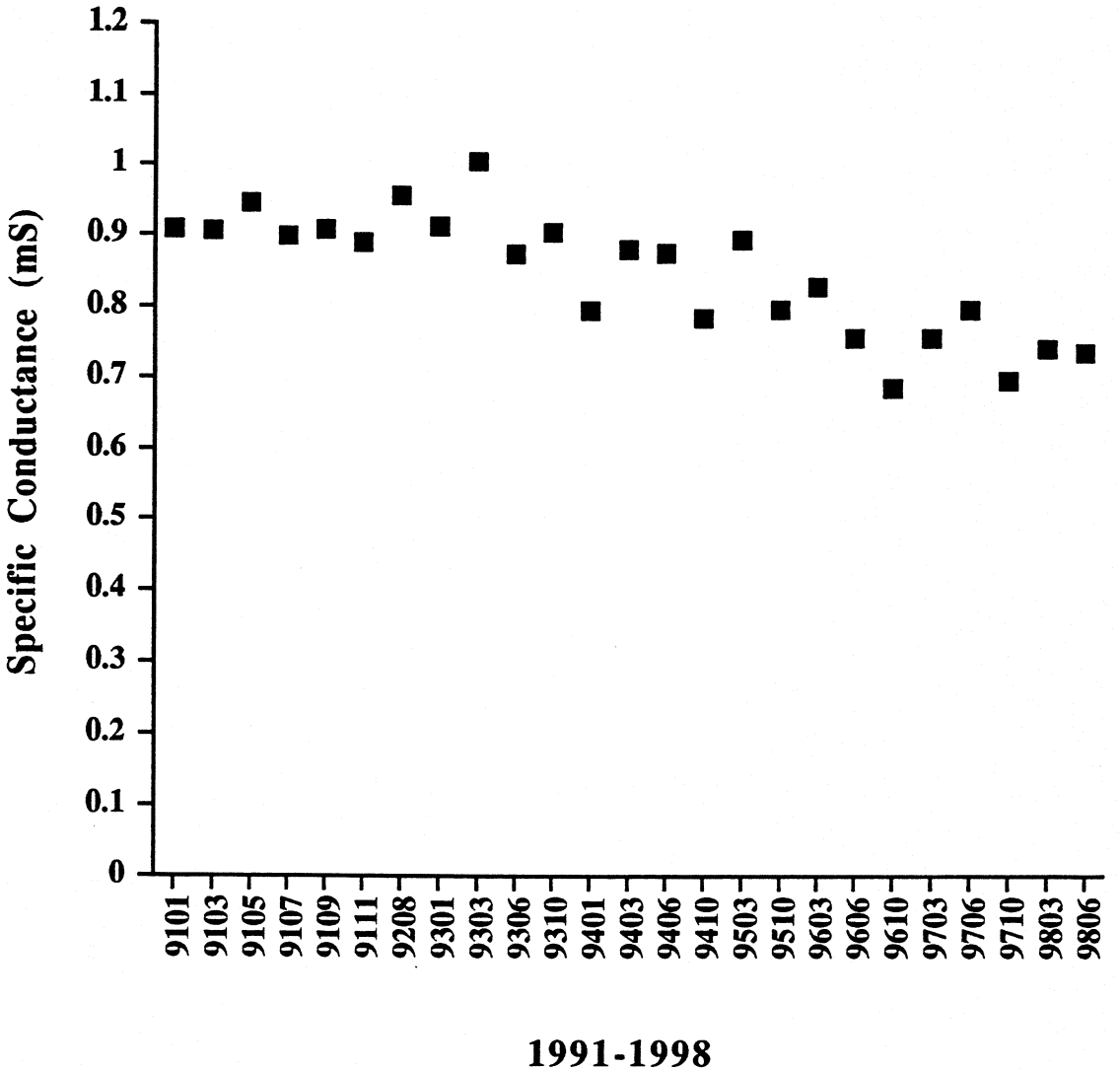

Figure 11. Specific conductance (mS) collected at Lees Ferry Rkm 0.0 from January 1991 to June 1998.

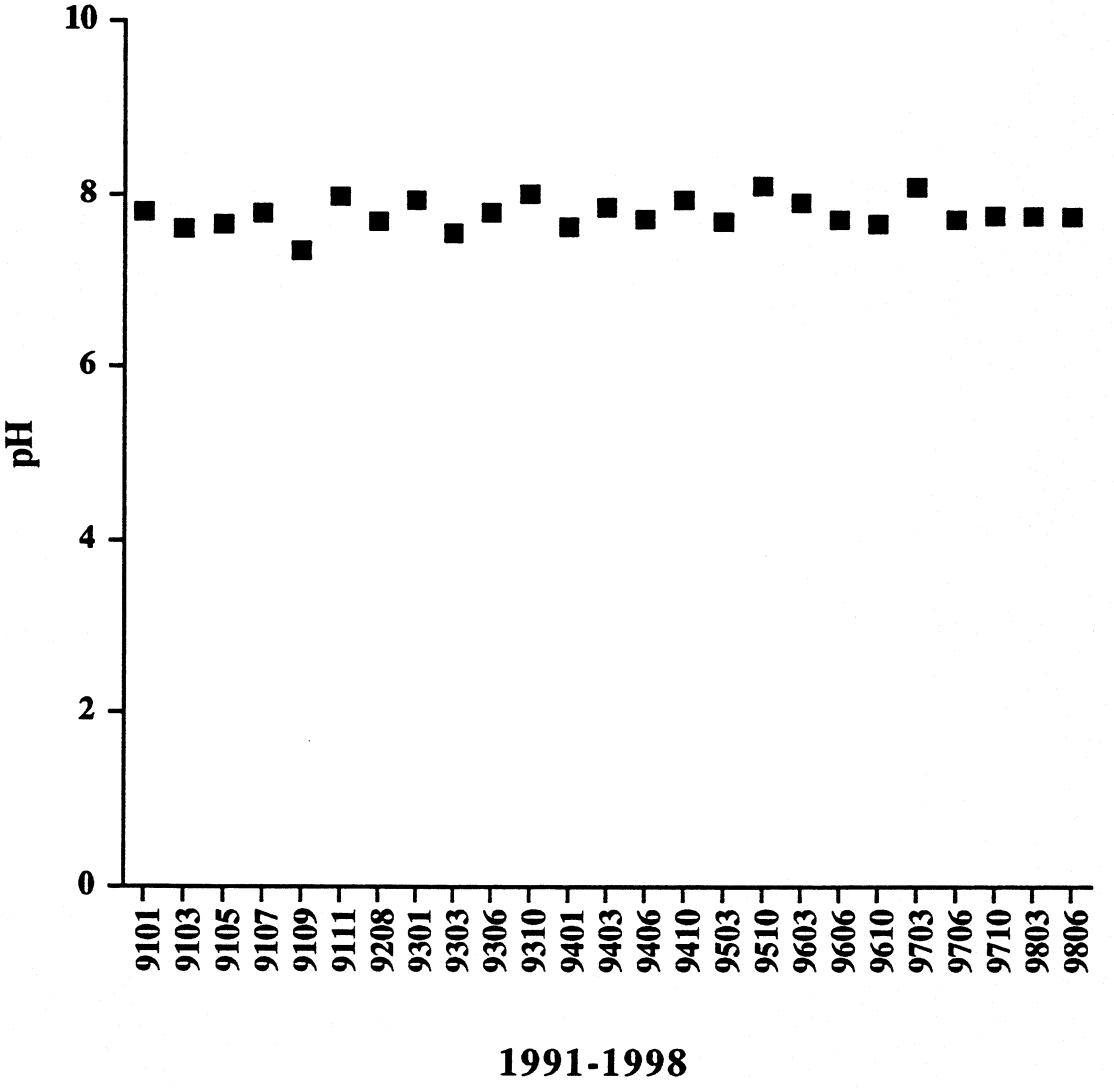

Figure 12. pH collected at Lees Ferry Rkm 0.0 from January 1991 to June 1998.

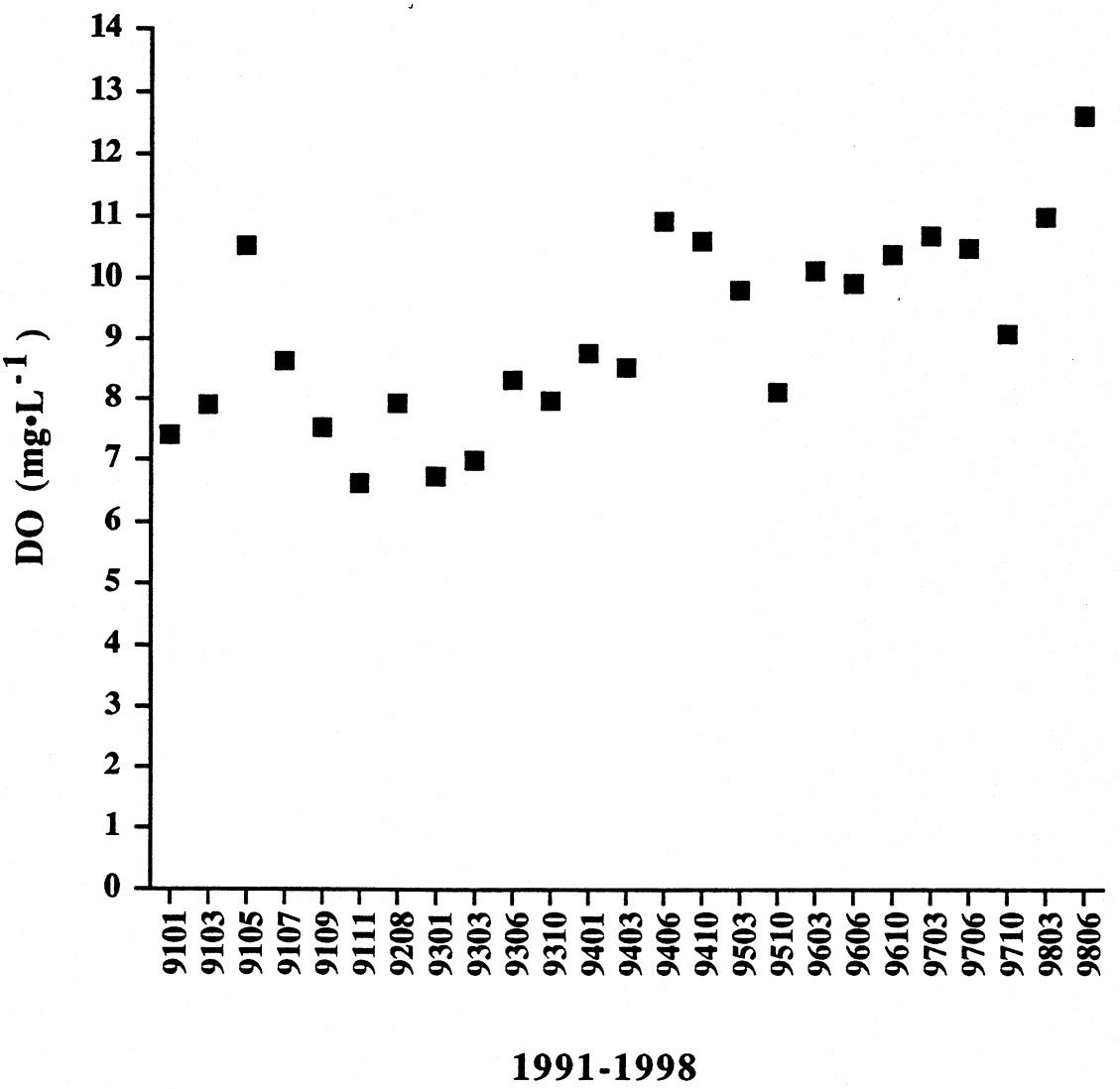

Figure 13. Dissolved oxygen ($mg \cdot L^{-1}$) collected at Lees Ferry Rkm 0.0 from January 1991 to June 1998.

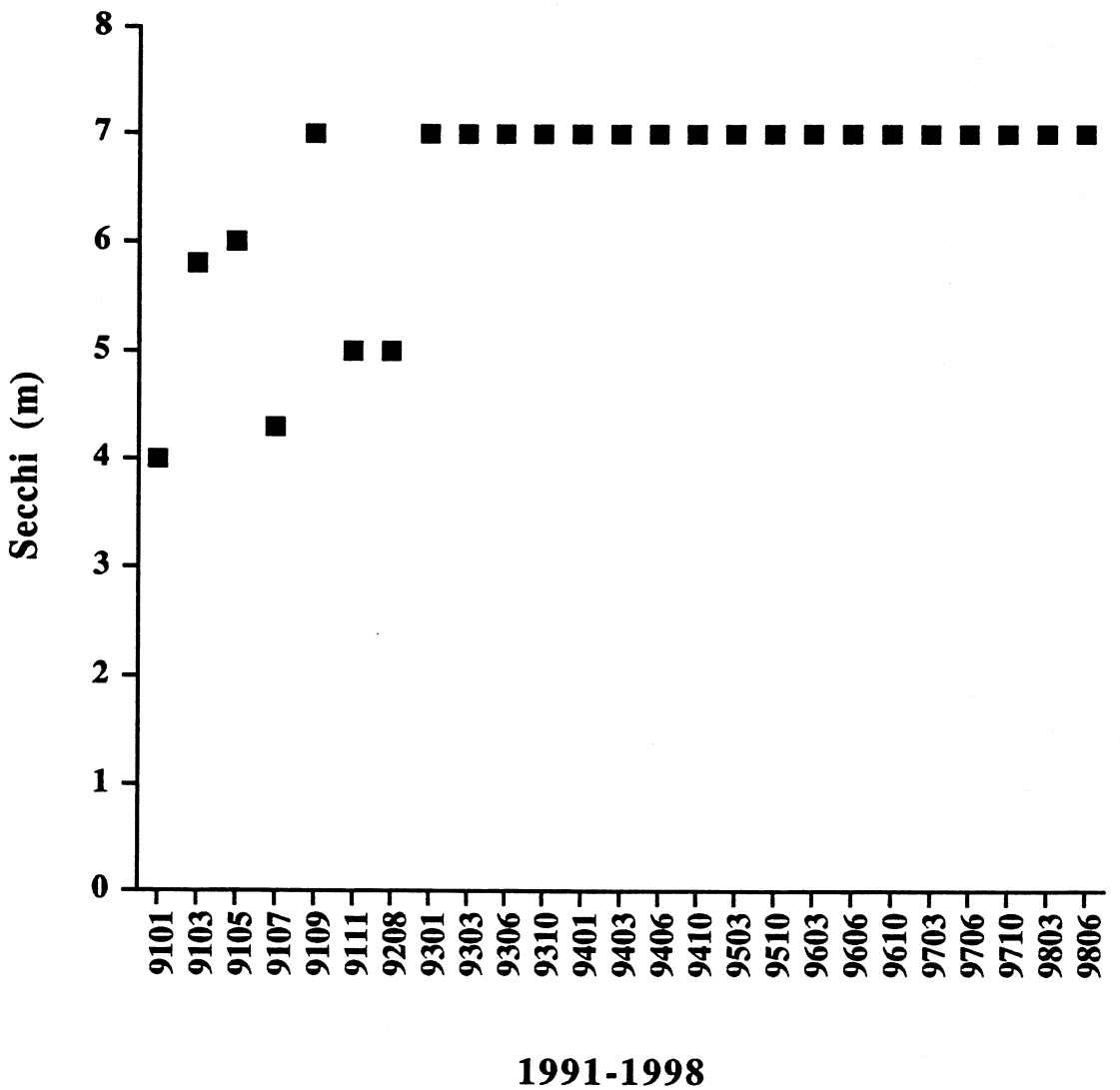

Figure 14. Secchi depth (m) collected at Lees Ferry Rkm 0.0 from January 1991 to June 1998. Lees Ferry clear water conditions allow Secchi depth readings to bottom of channel. Prior to 1992, high fluctuating flows decreased this depth to below 7 meters.

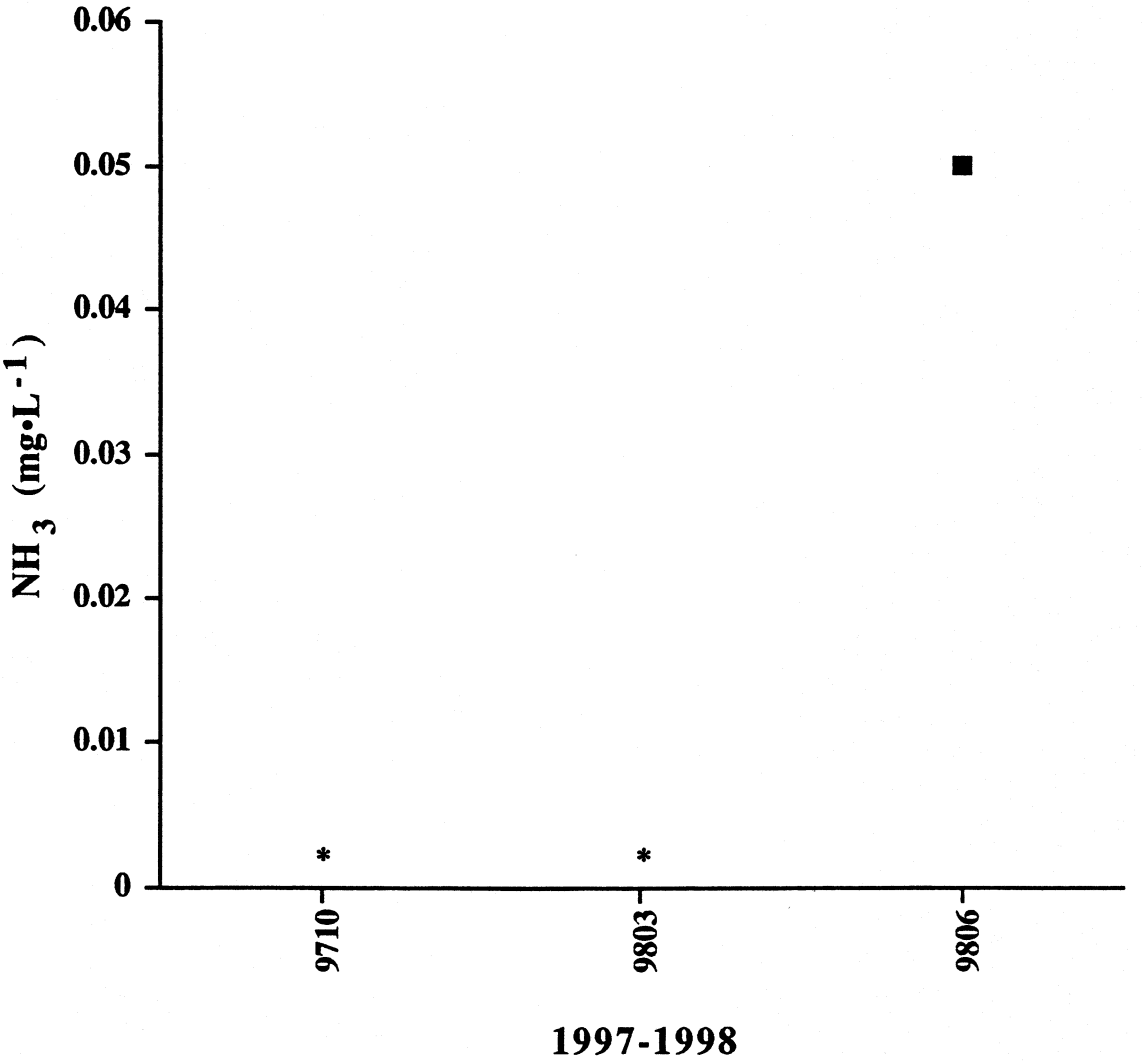

Figure 15. Ammonia (NH_3 mg•L^{-1}) collected at Lees Ferry Rkm 0.0 from October 1997 to June 1998. Samples below detectable levels are represented by a (*).

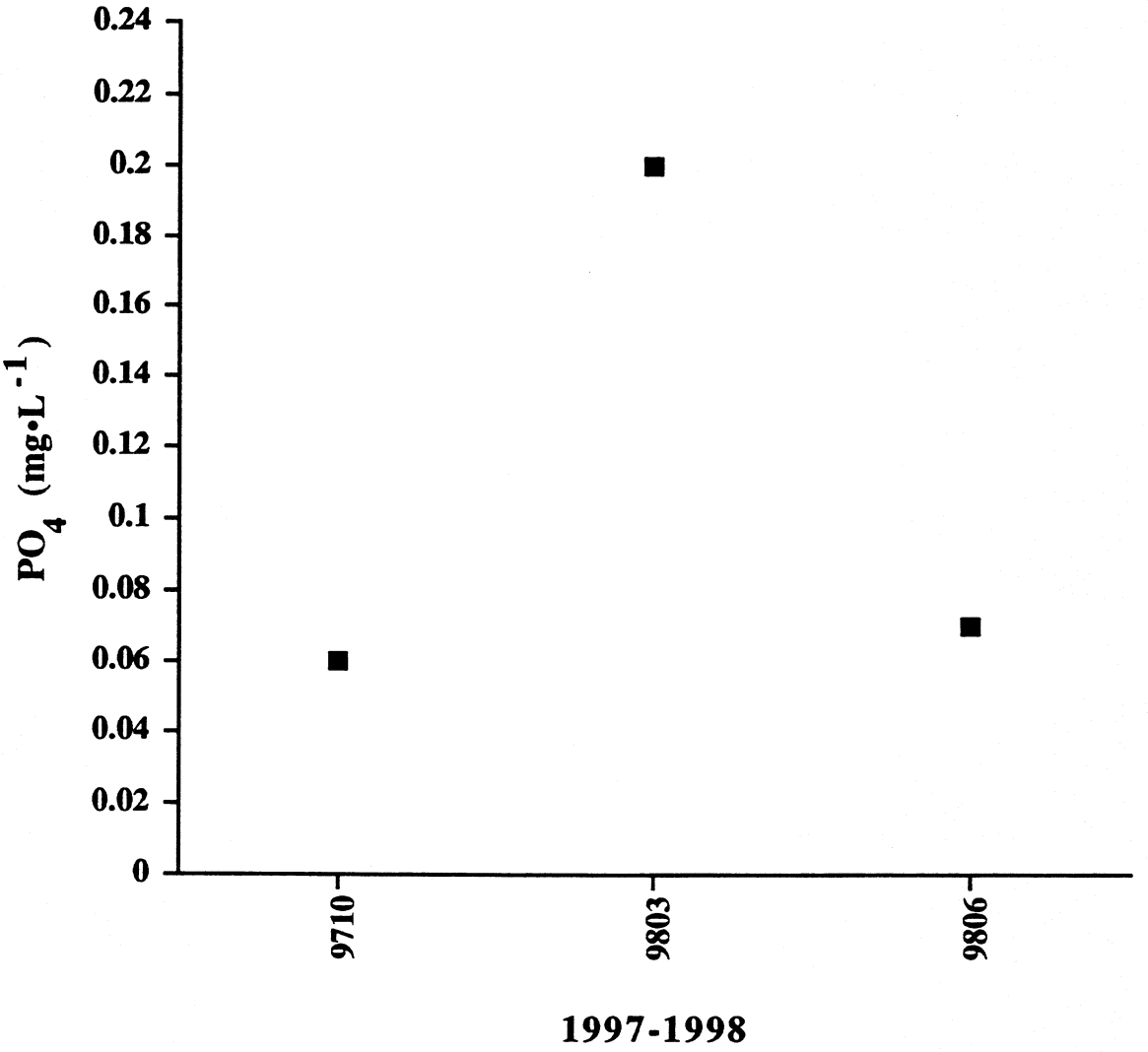

Figure 16. Phosphate (PO_4 mg•L $^{-1}$) collected at Lees Ferry Rkm 0.0 from October 1997 to June 1998.

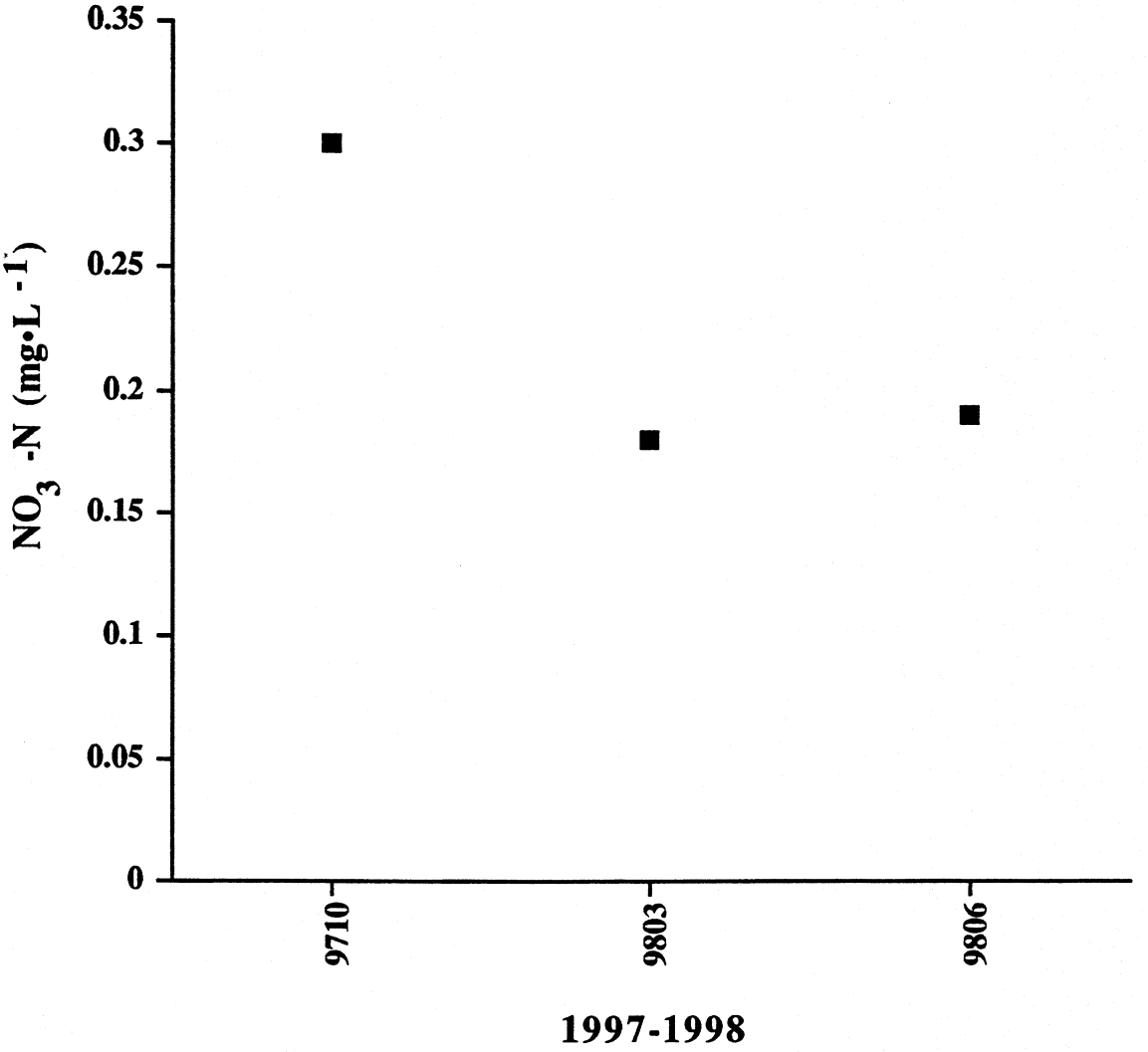

Figure 17. Nitrate-nitrogen (NO_3-N mg•L^{-1}) collected at Lees Ferry Rkm 0.0 from October 1997 to June 1998.

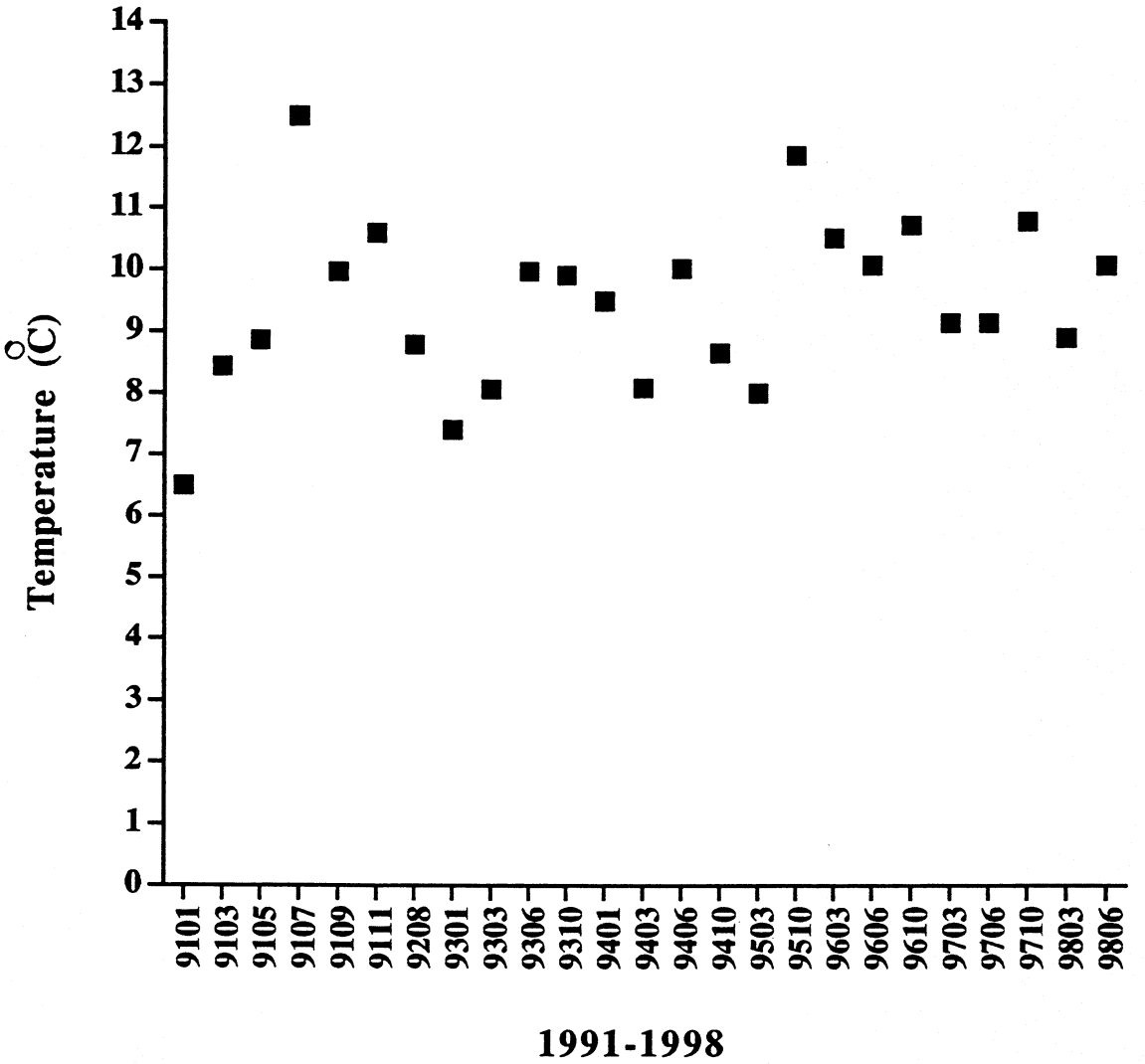

Figure 18. Water temperature (°C) collected at Two-Mile Wash Rkm 3.1 from January 1991 to June 1998.

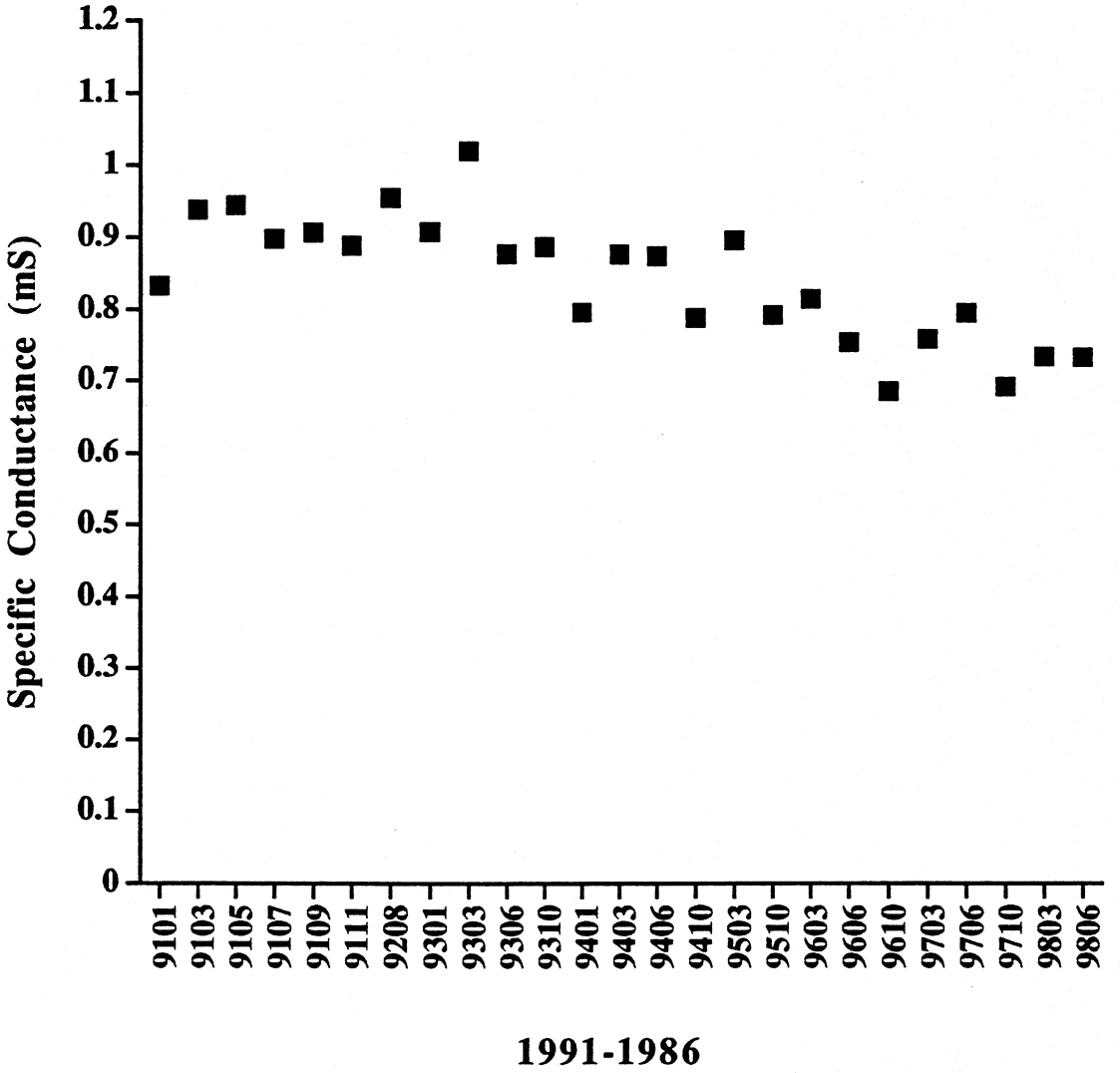

Figure 19. Specific conductance (mS) collected at Two-Mile Wash Rkm 3.1 from January 1991 to June 1998.

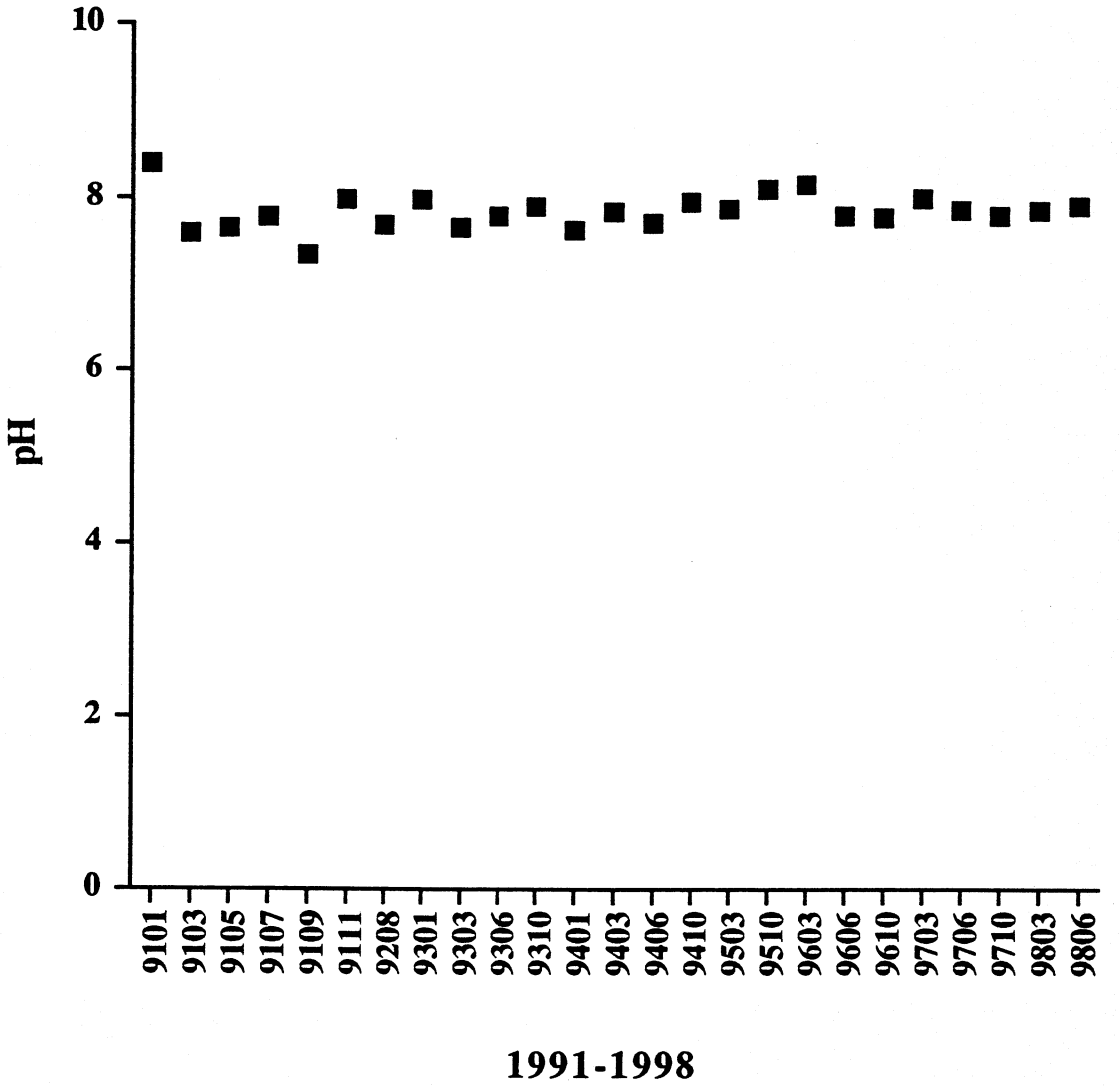

Figure 20. pH collected at Two-Mile Wash Rkm 3.1 from January 1991 to June 1998.

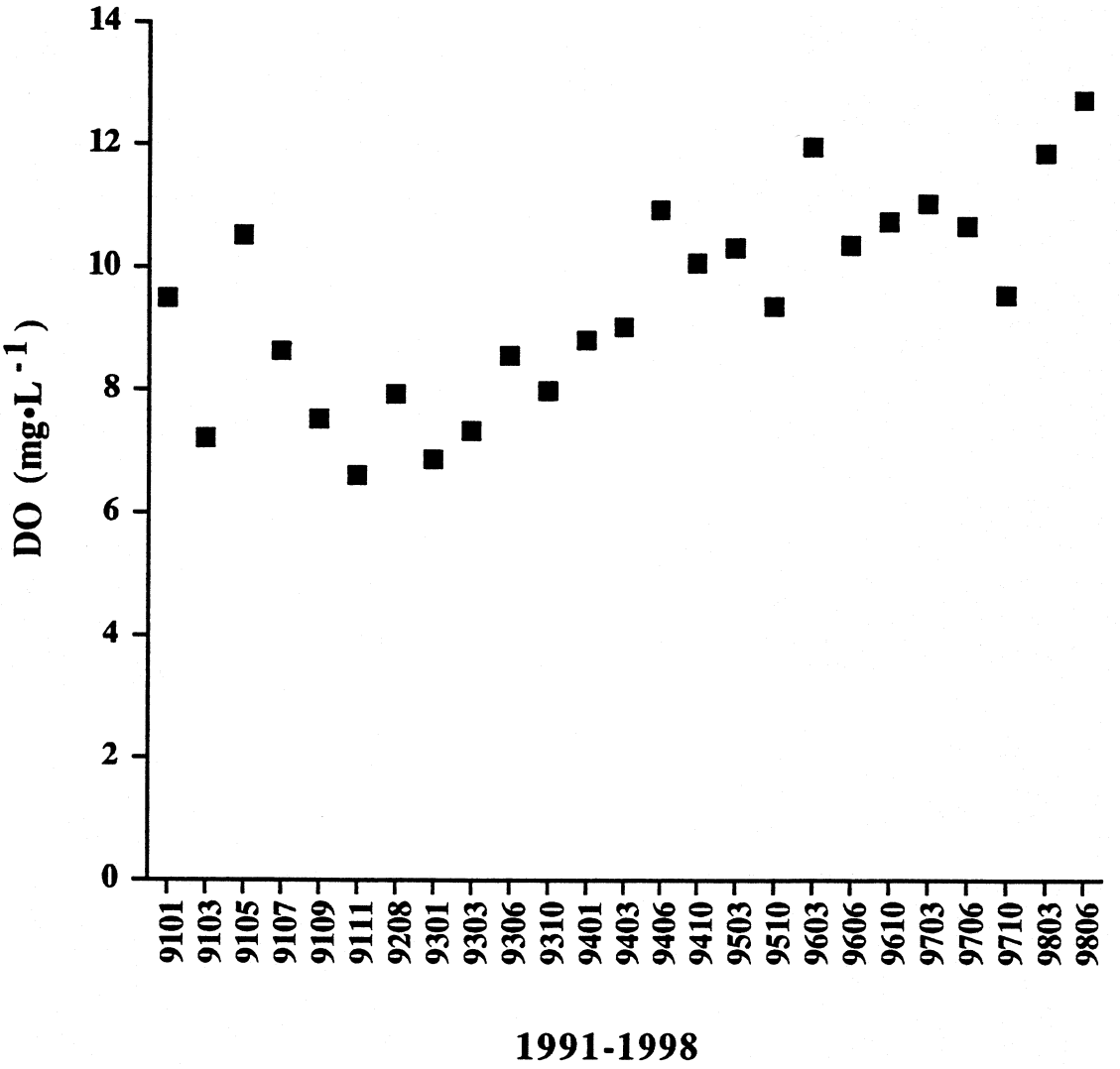

Figure 21. Dissolved oxygen ($mg \cdot L^{-1}$) collected at Two-Mile Wash Rkm 3.1 from January 1991 to June 1998.

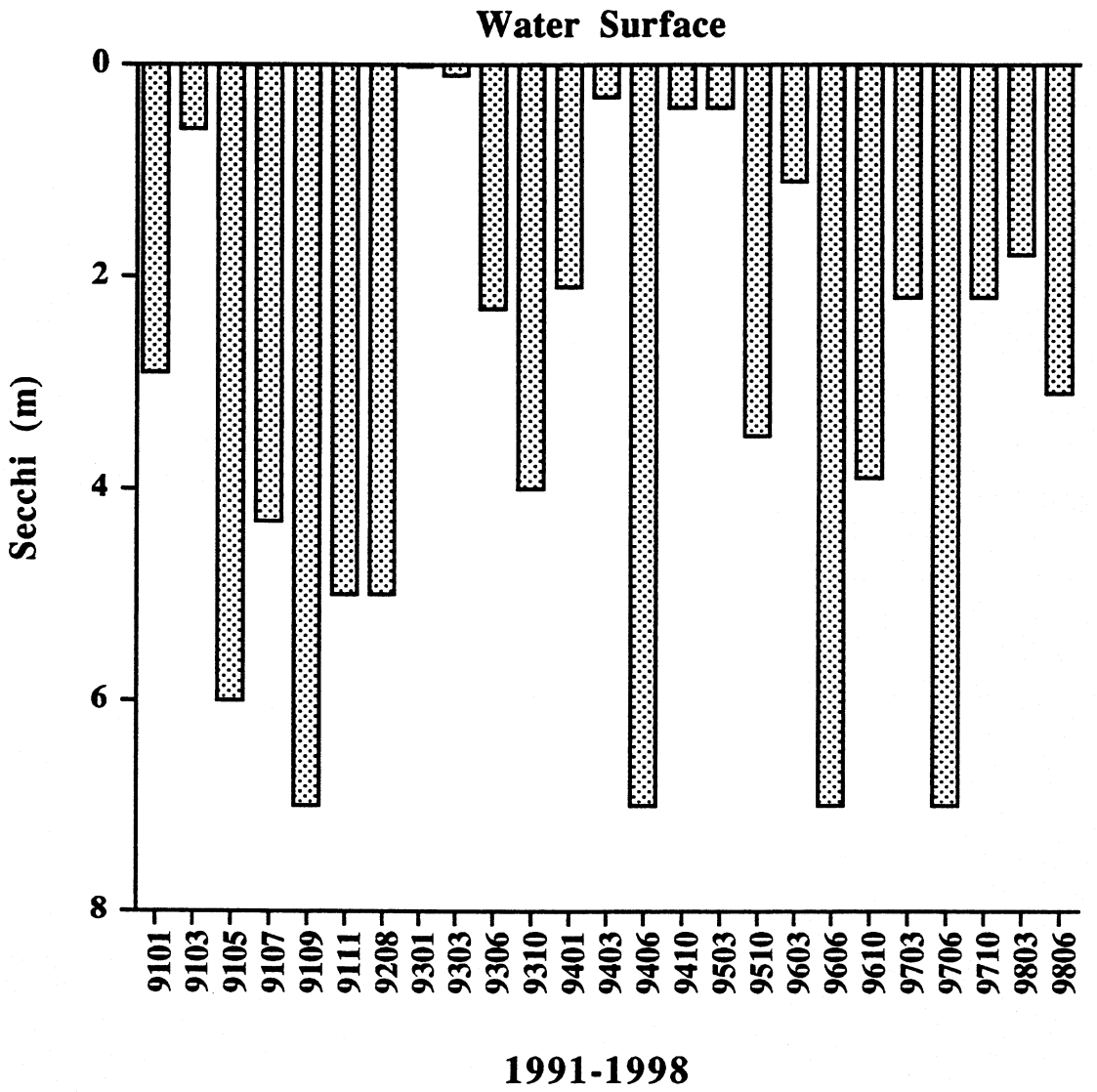

Figure 22. Secchi depth (m) collected at Two-Mile Wash Rkm 3.1 from January 1991 to June 1998.

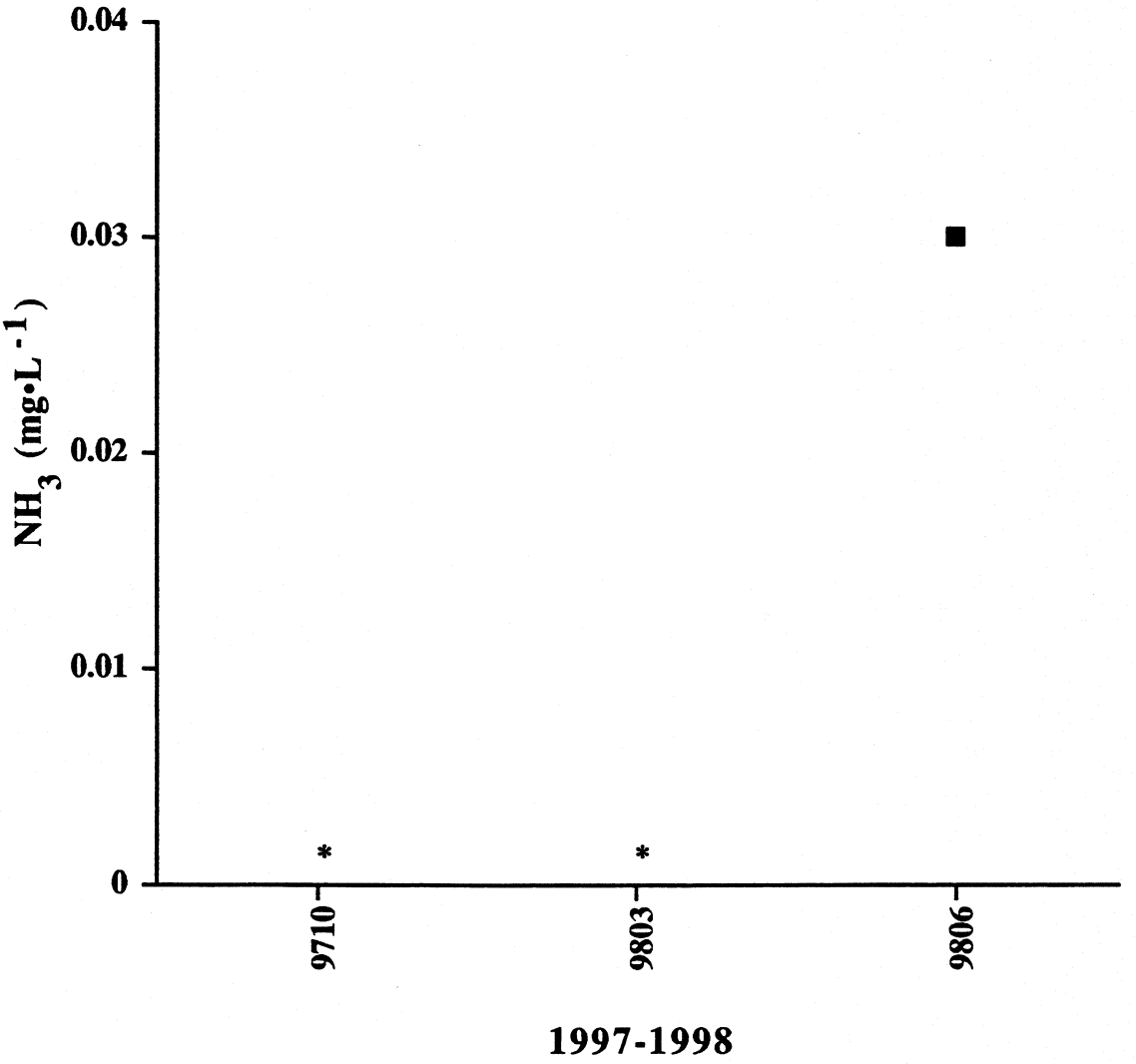

Figure 23. Ammonia (NH_3 mg•L^{-1}) collected at Two-Mile Wash Rkm 3.1 from October 1997 to June 1998. Samples below detectable levels are represented by a (*).

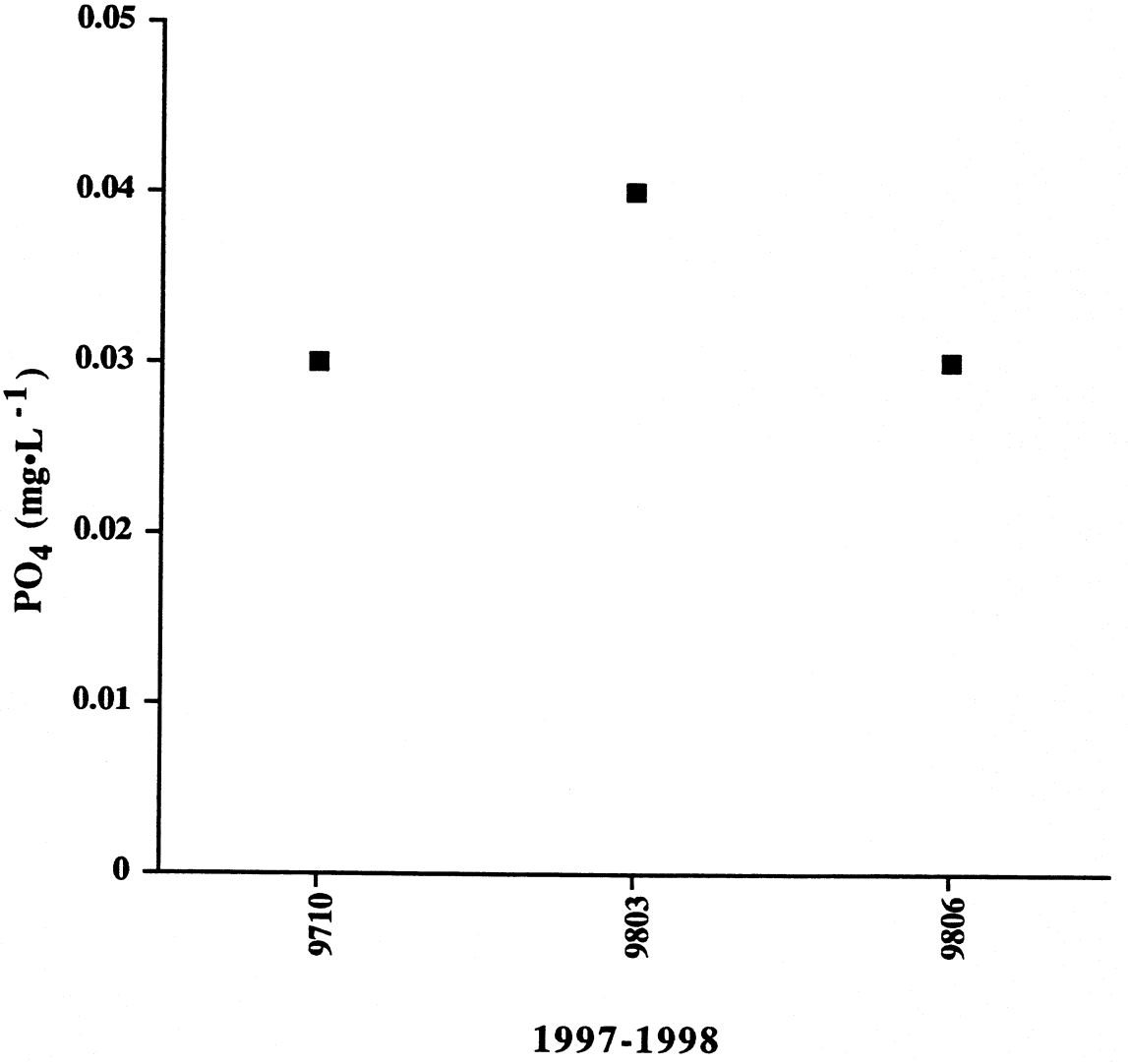

Figure 24. Phosphate (PO_4 mg•L^{-1}) collected at Two-Mile Wash Rkm 3.1 from October 1997 to June 1998.

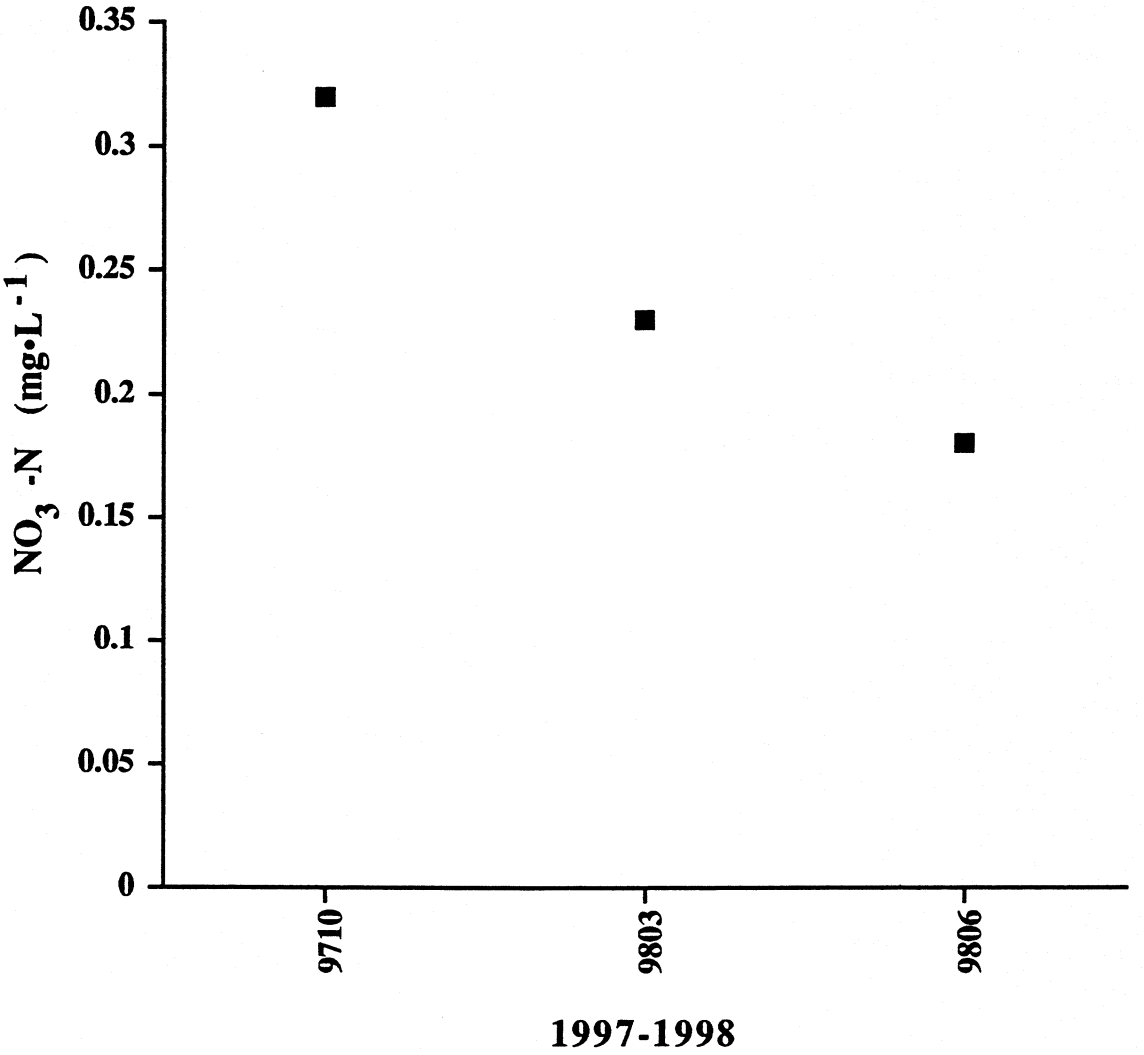

Figure 25. Nitrate-nitrogen (NO_3-N mg•L^{-1}) collected at Two-Mile Wash Rkm 3.1 from October 1997 to June 1998.

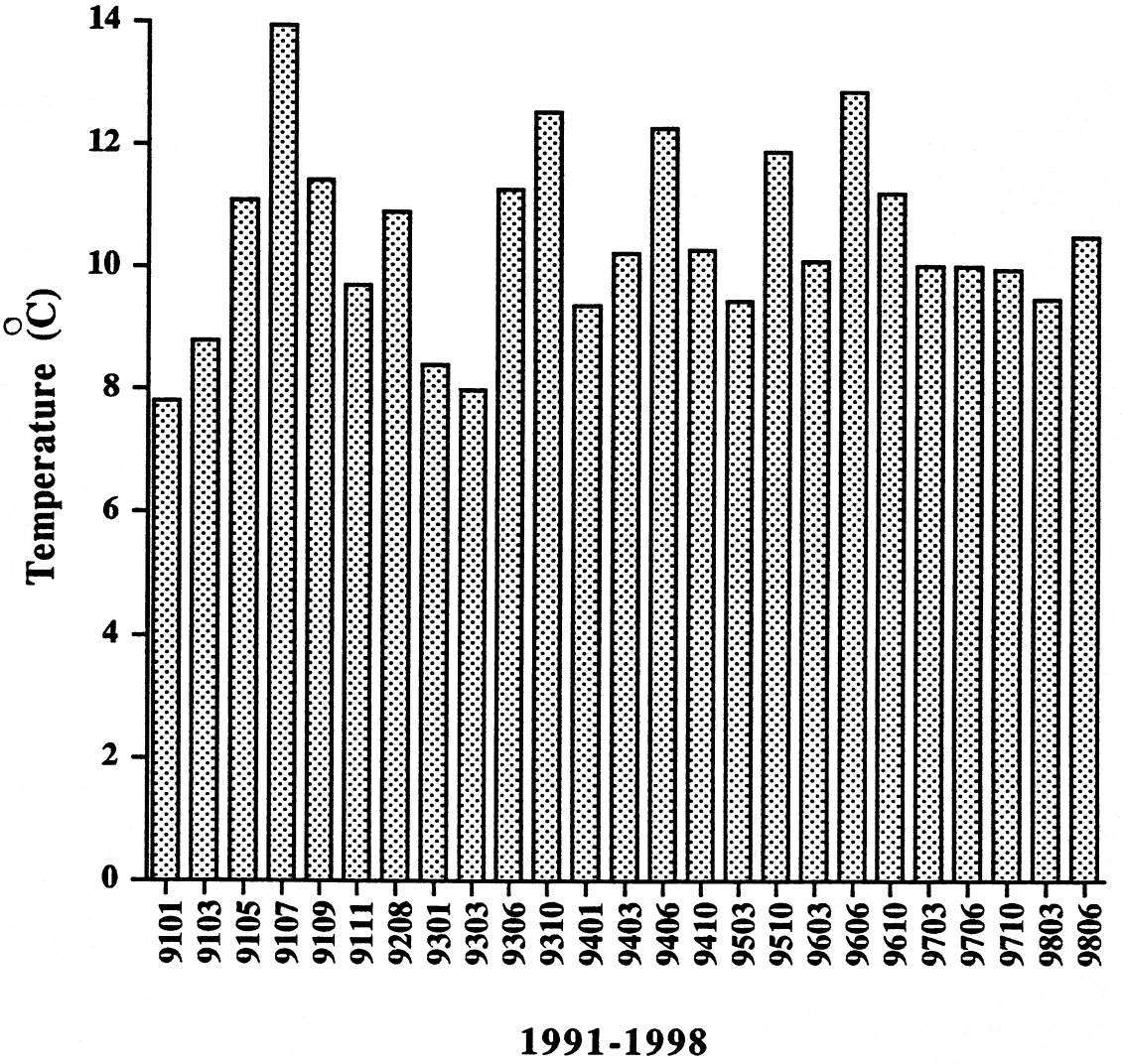

Figure 26. Water temperature (C) collected at the Gauge Above LCR Rkm 98.4 from January 1991 to June 1998.

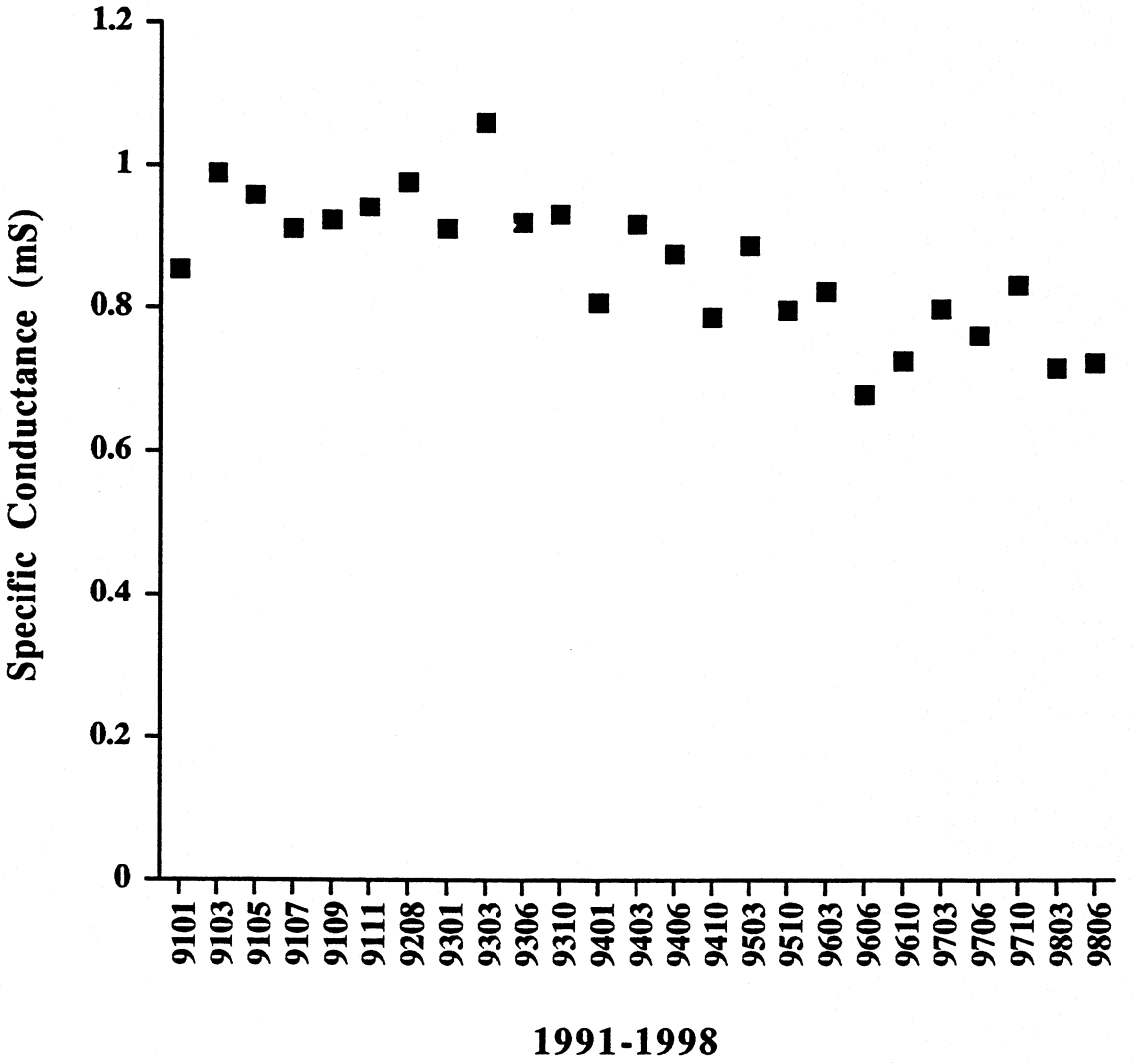

Figure 27. Specific conductance (mS) collected at the Gauge Above LCR Rkm 98.4 from January 1991 to June 1998.

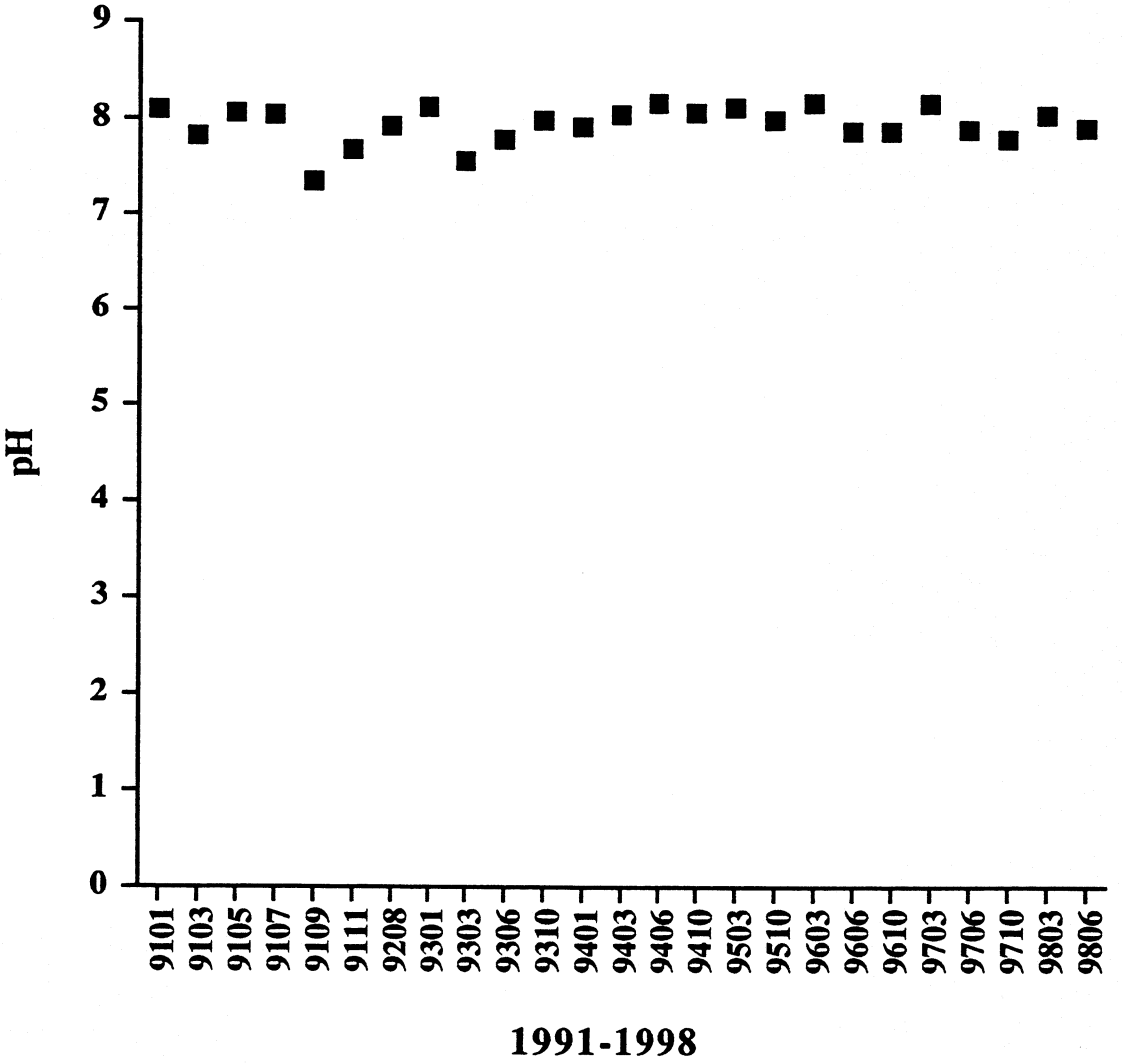

Figure 28. pH collected at the Gauge Above LCR Rkm 98.4 from January 1991 to June 1998.

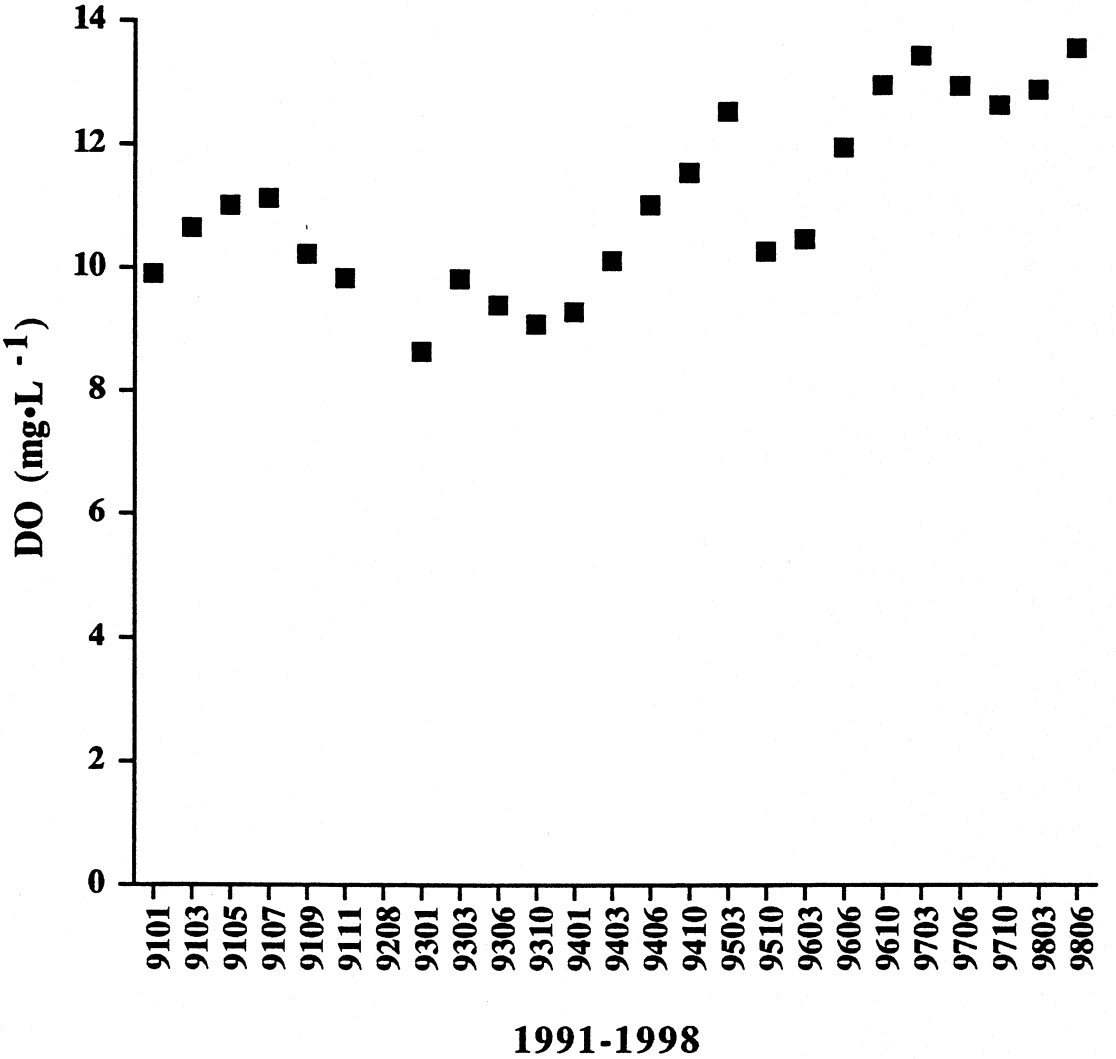

Figure 29. Dissolved oxygen (DO mg•L^{-1}) collected at the Gauge Above LCR Rkm 98.4 from January 1991 to June 1998.

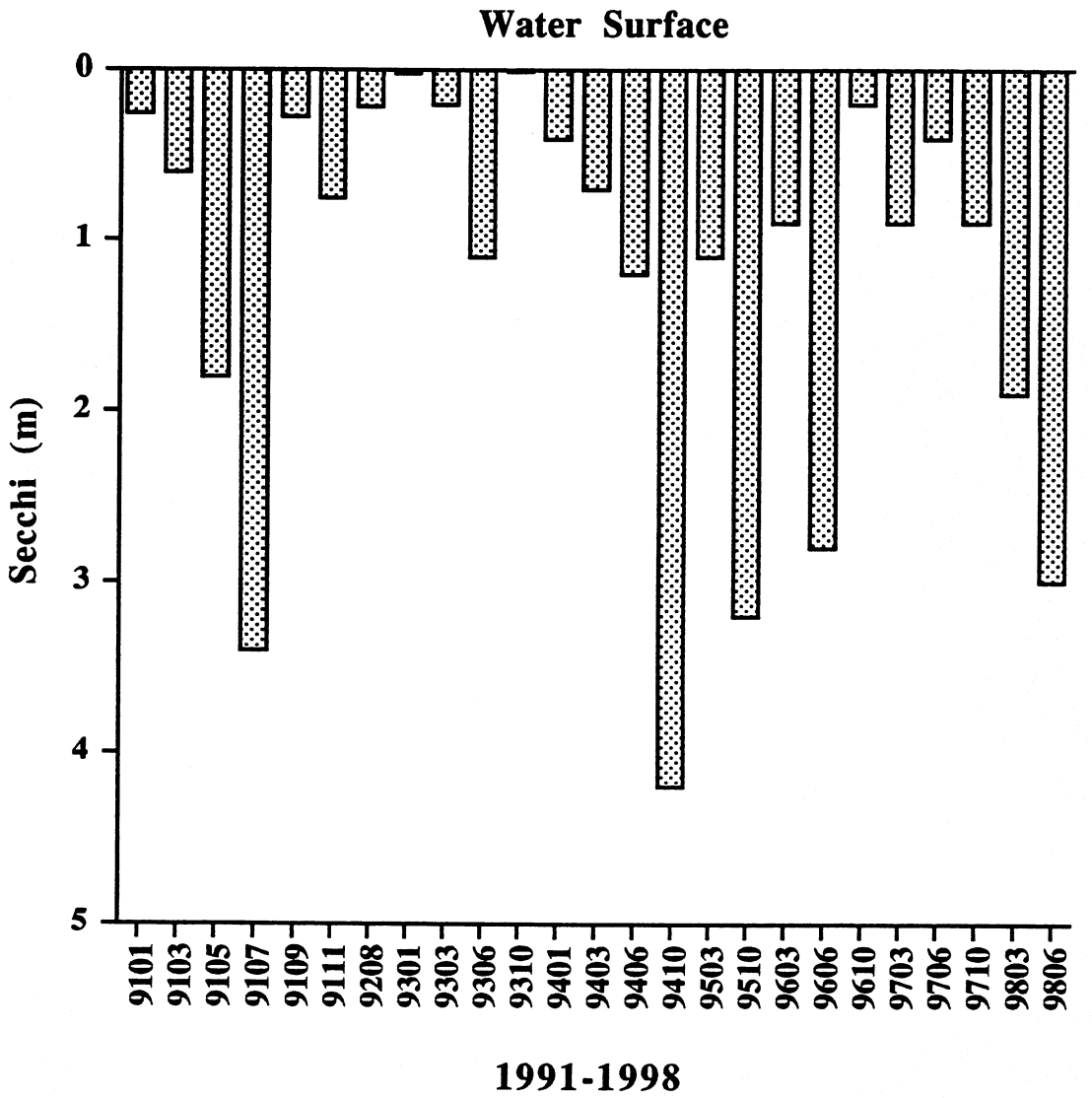

Figure 30. Secchi depth (m) collected at the Gauge Above LCR Rkm 98.4 from January 1991 to June 1998.

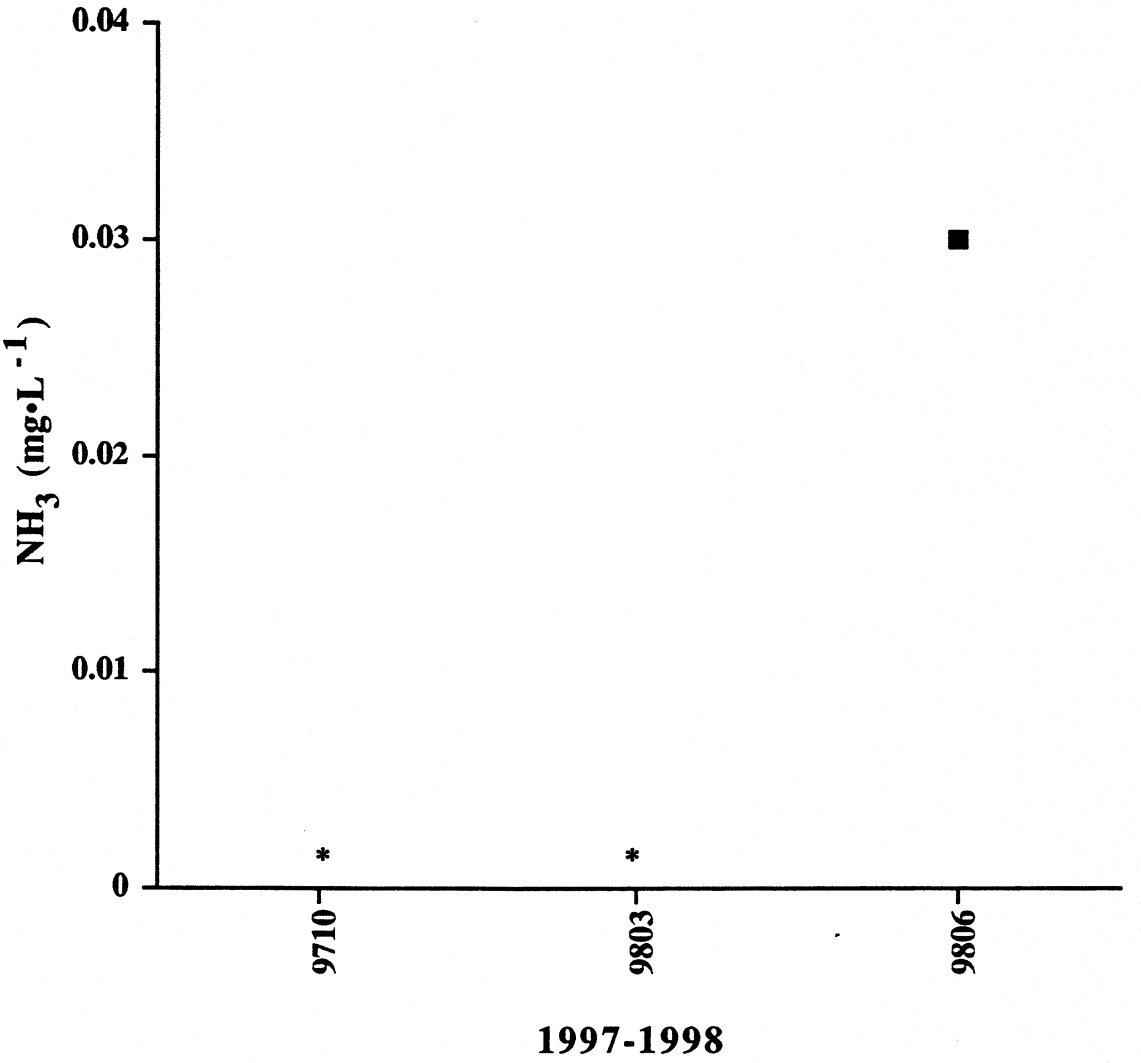

Figure 31. Ammonia (NH_3 mg•L^{-1}) collected at the Gauge above LCR Rkm 98.4 from October 1997 to June 1998. Samples below detection levels are represented by a (*).

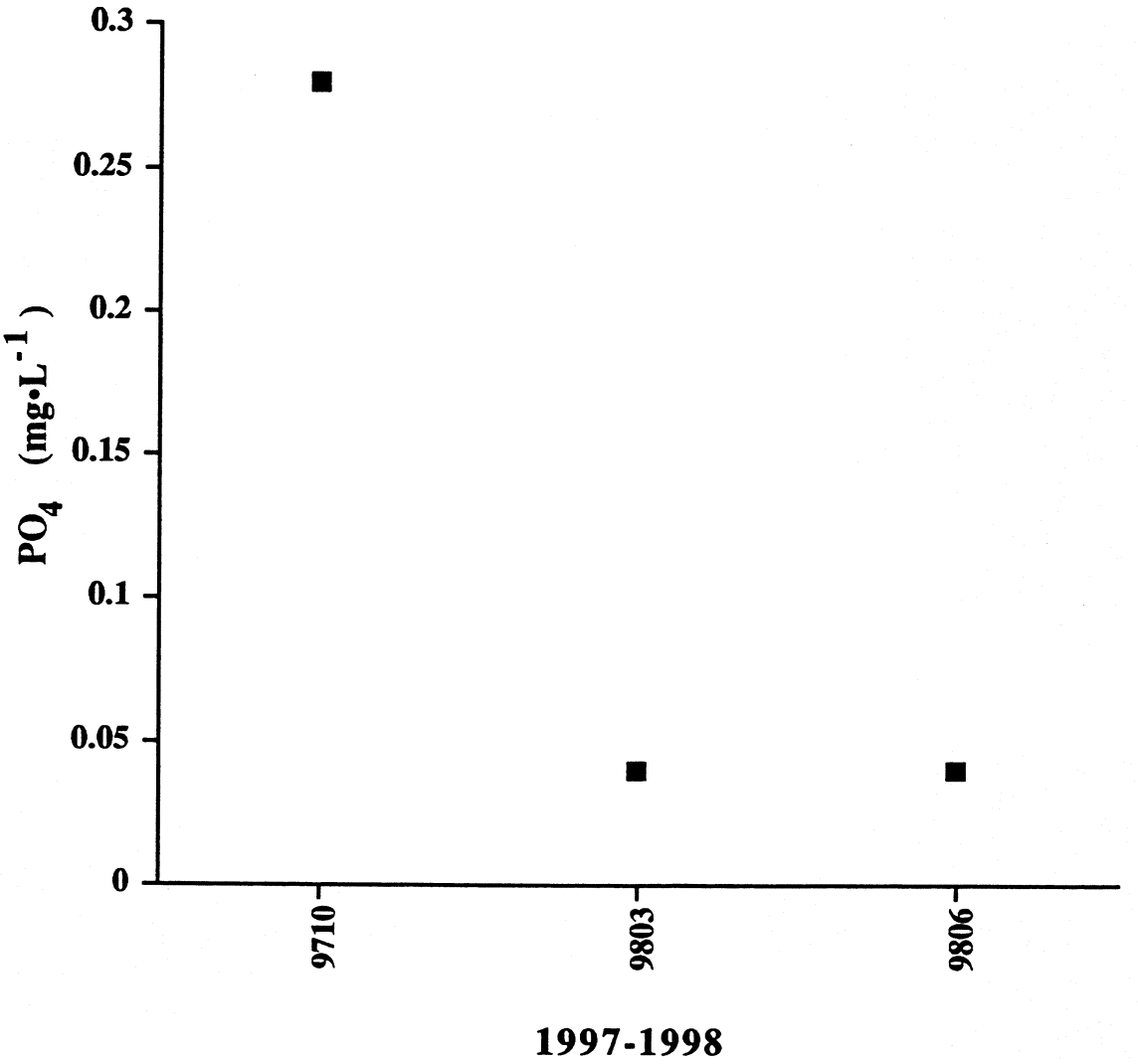

Figure 32. Phosphate (PO_4 mg•L^{-1}) collected at the Gauge above LCR Rkm 98.4 from October 1997 to June 1998.

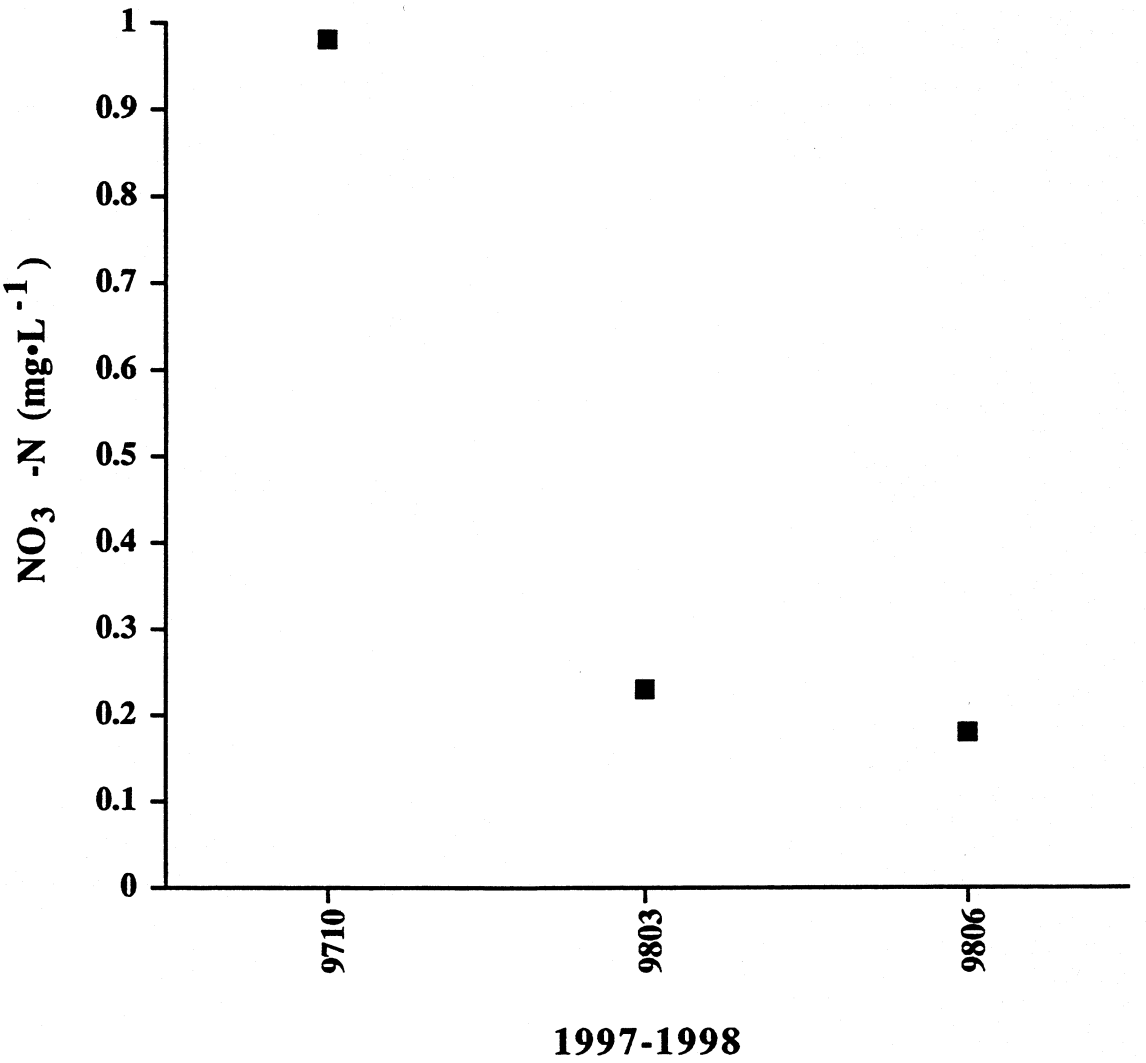

Figure 33. Nitrate-nitrogen (NO_3 -N mg•L^{-1}) collected at the Gauge above LCR Rkm 98.4 from October 1997 to June 1998.

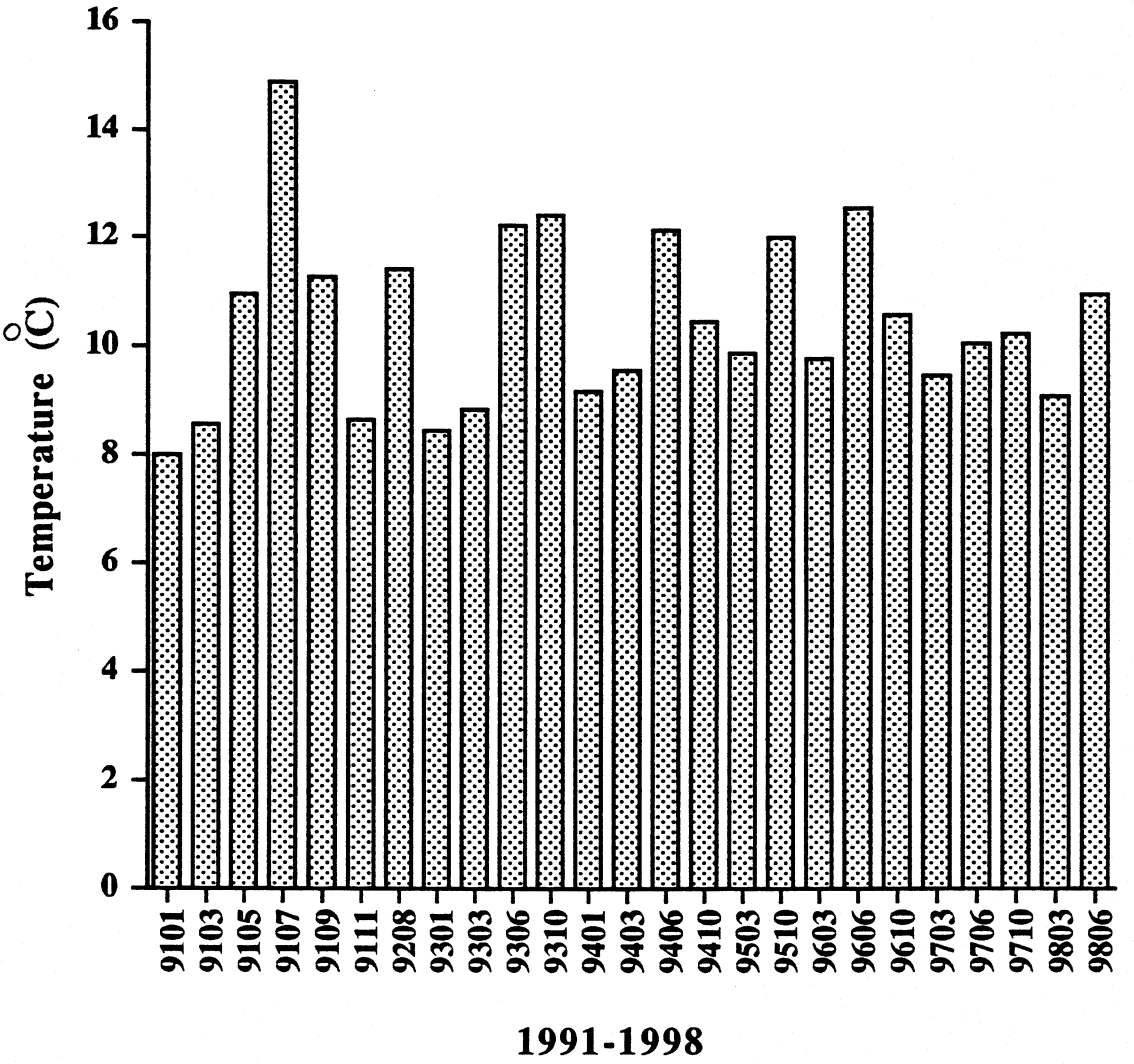

Figure 34. Water temperature (°C) collected at Tanner Canyon Rkm 108.8 from January 1991 to June 1998.

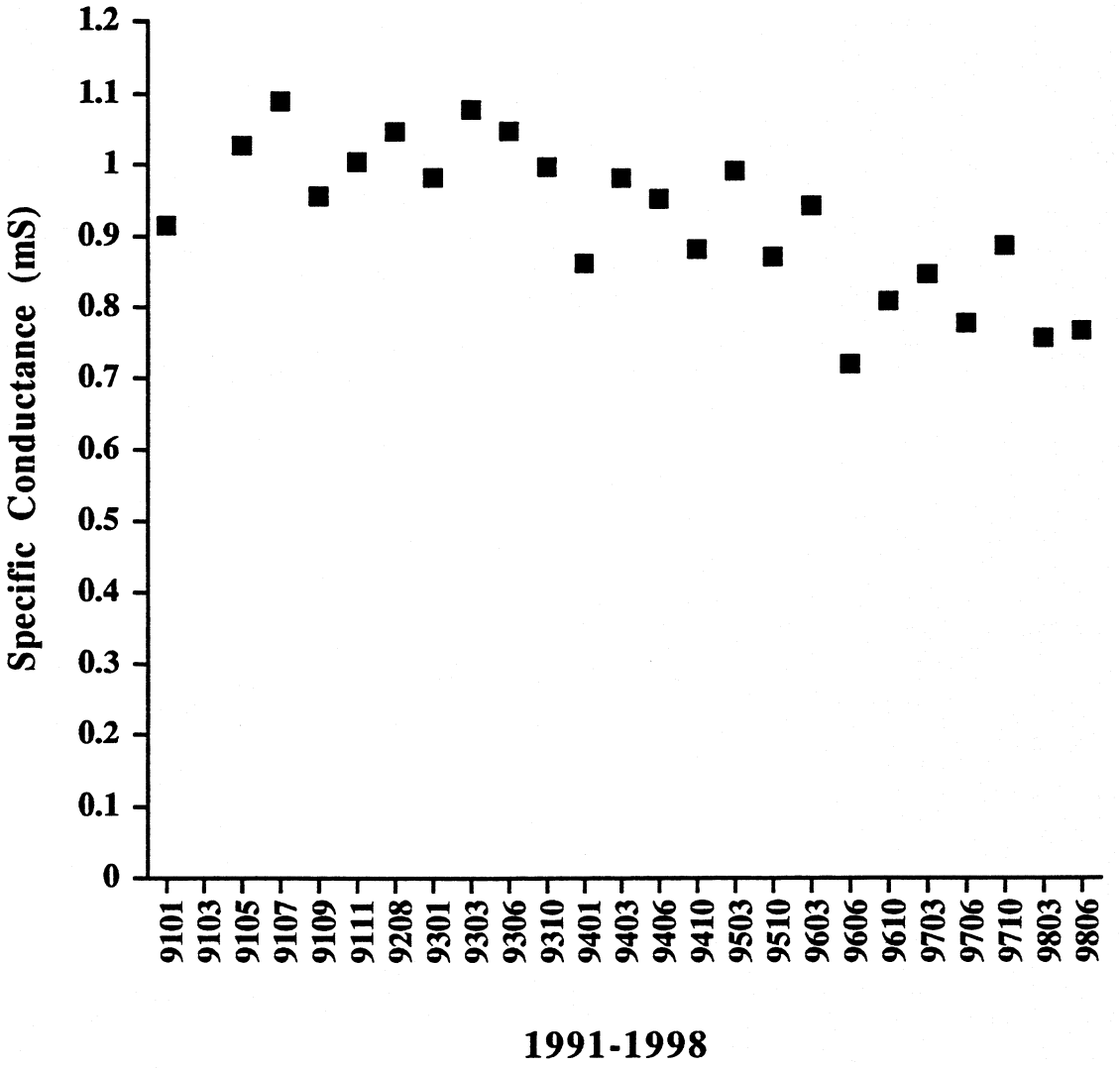

Figure 35. Specific conductance (mS) collected at Tanner Canyon Rkm 108.8 from January 1991 to June 1998.

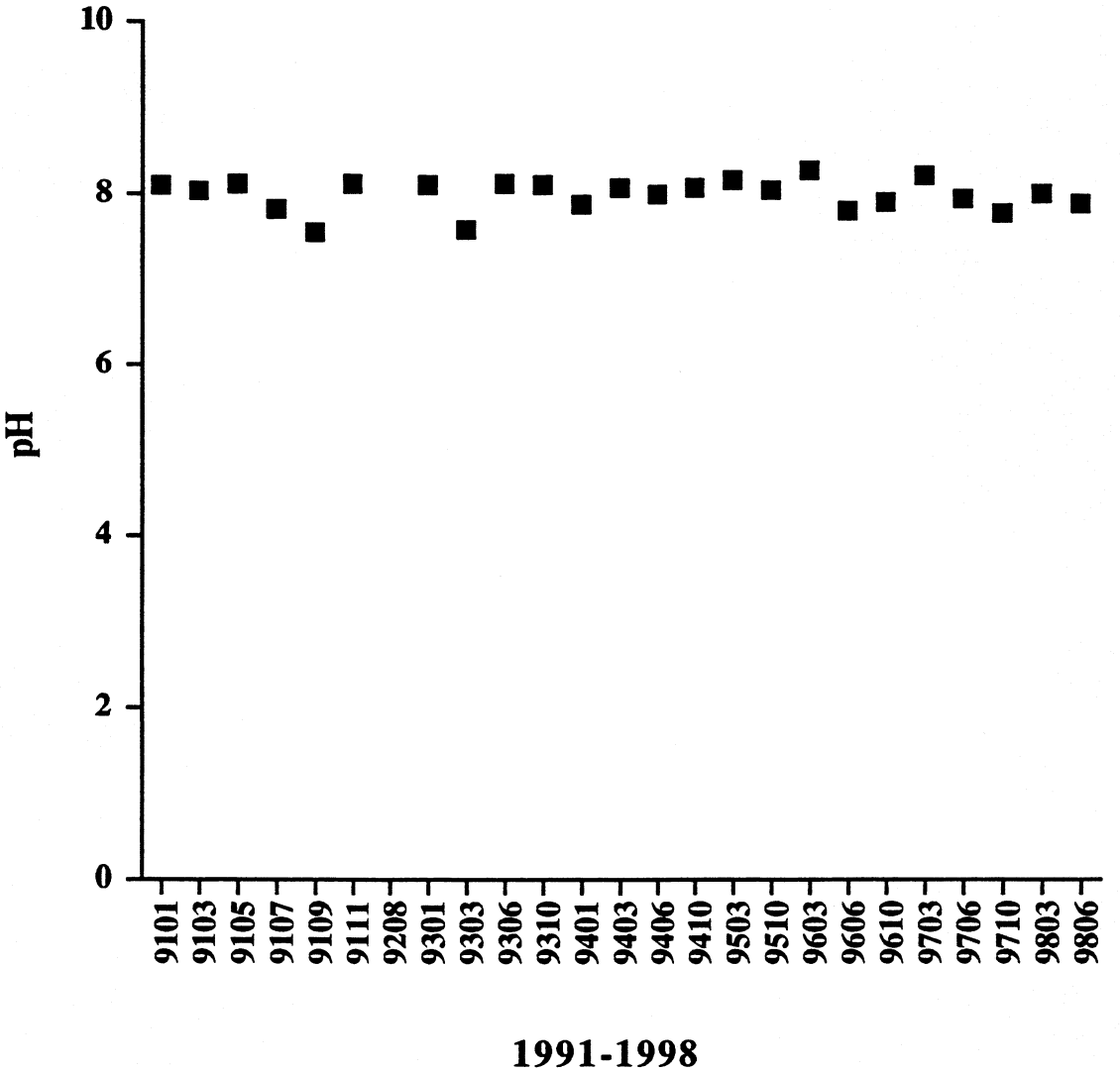

Figure 36. pH collected at Tanner Canyon Rkm 108.8 from January 1991 to June 1998.

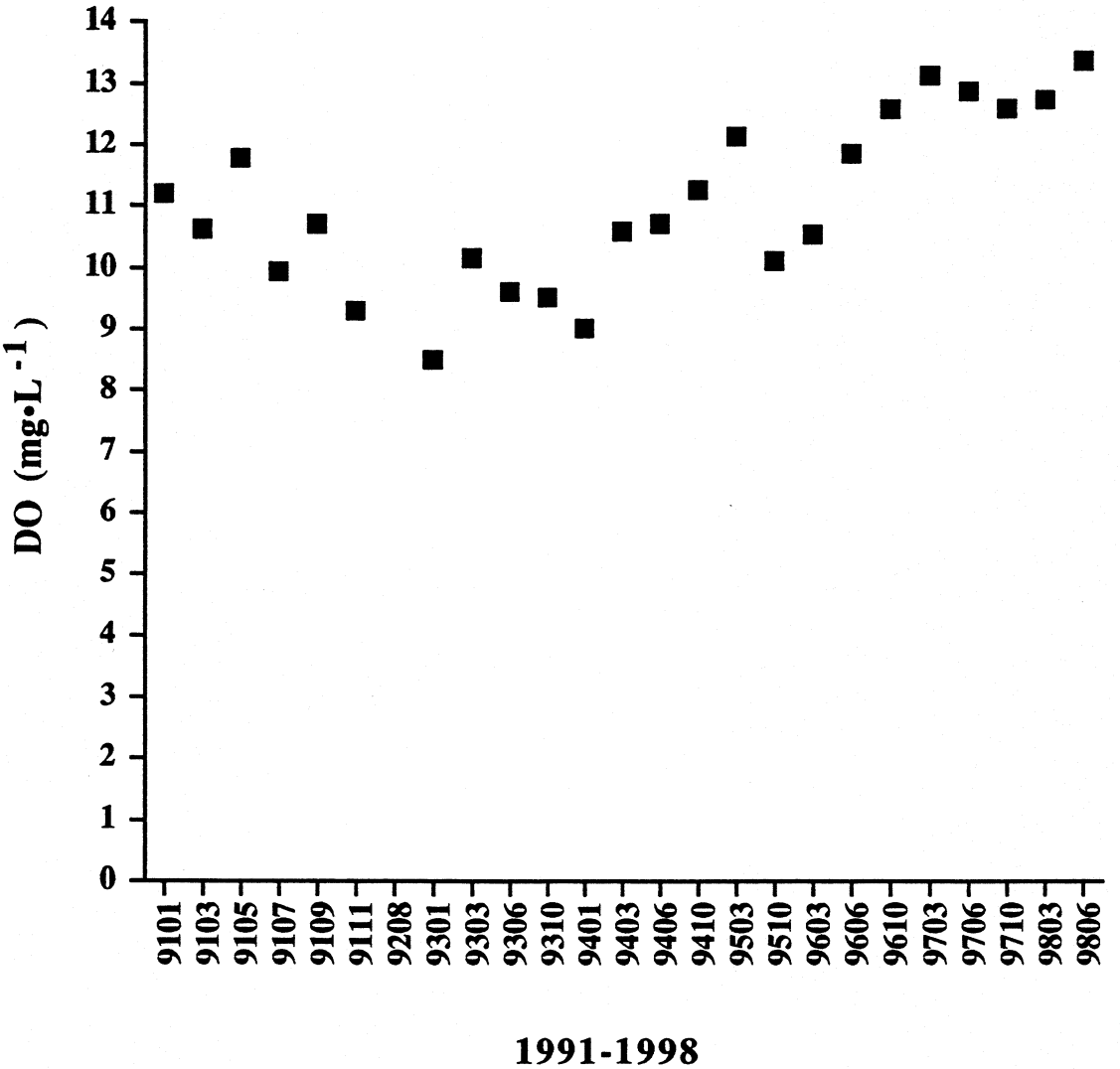

Figure 37. Dissolved oxygen ($mg \cdot L^{-1}$) collected at Tanner Canyon Rkm 108.8 from January 1991 to June 1998.

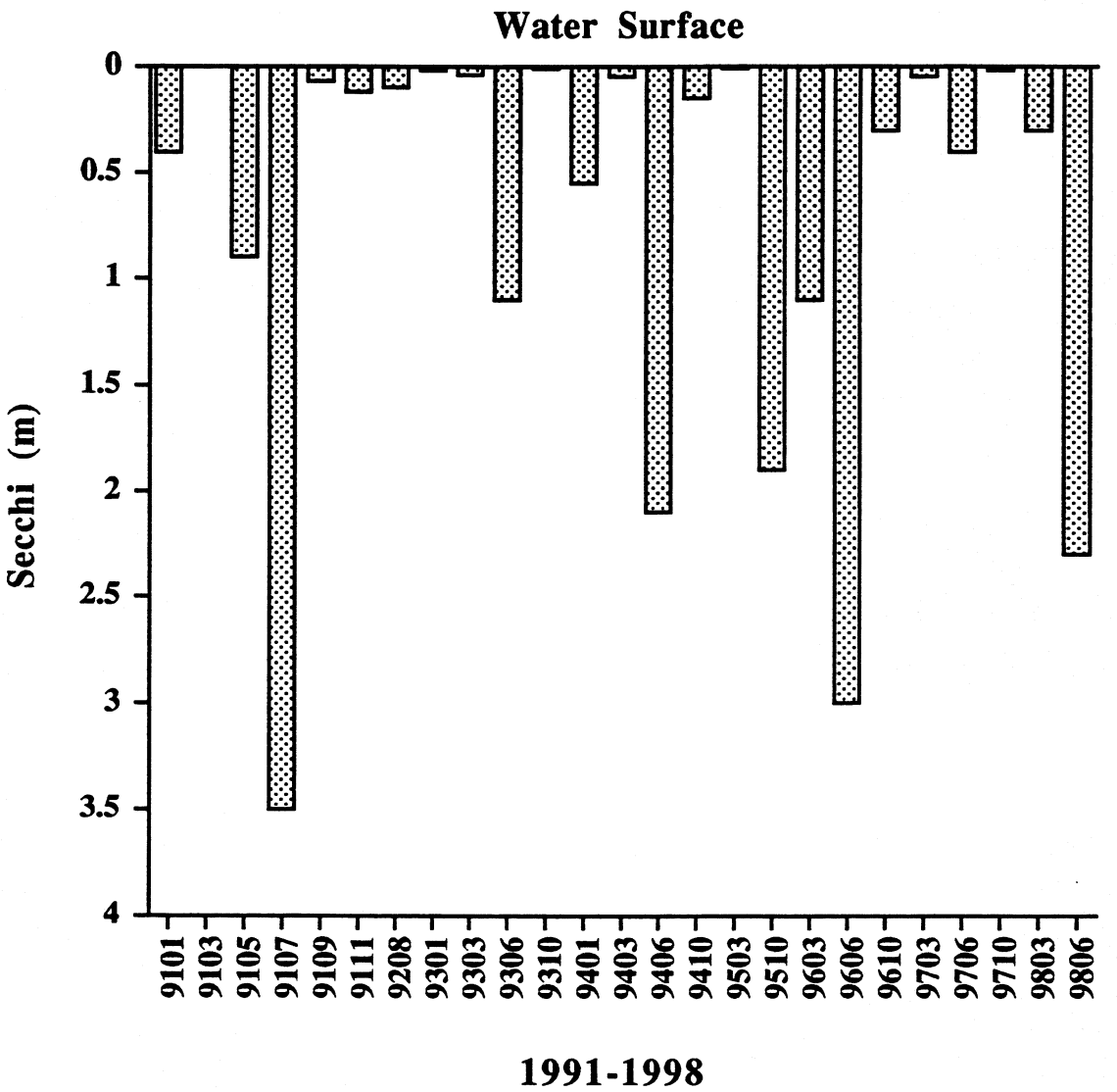

Figure 38. Secchi depth (m) collected at Tanner Canyon Rkm 108.8 from January 1991 to June 1998.

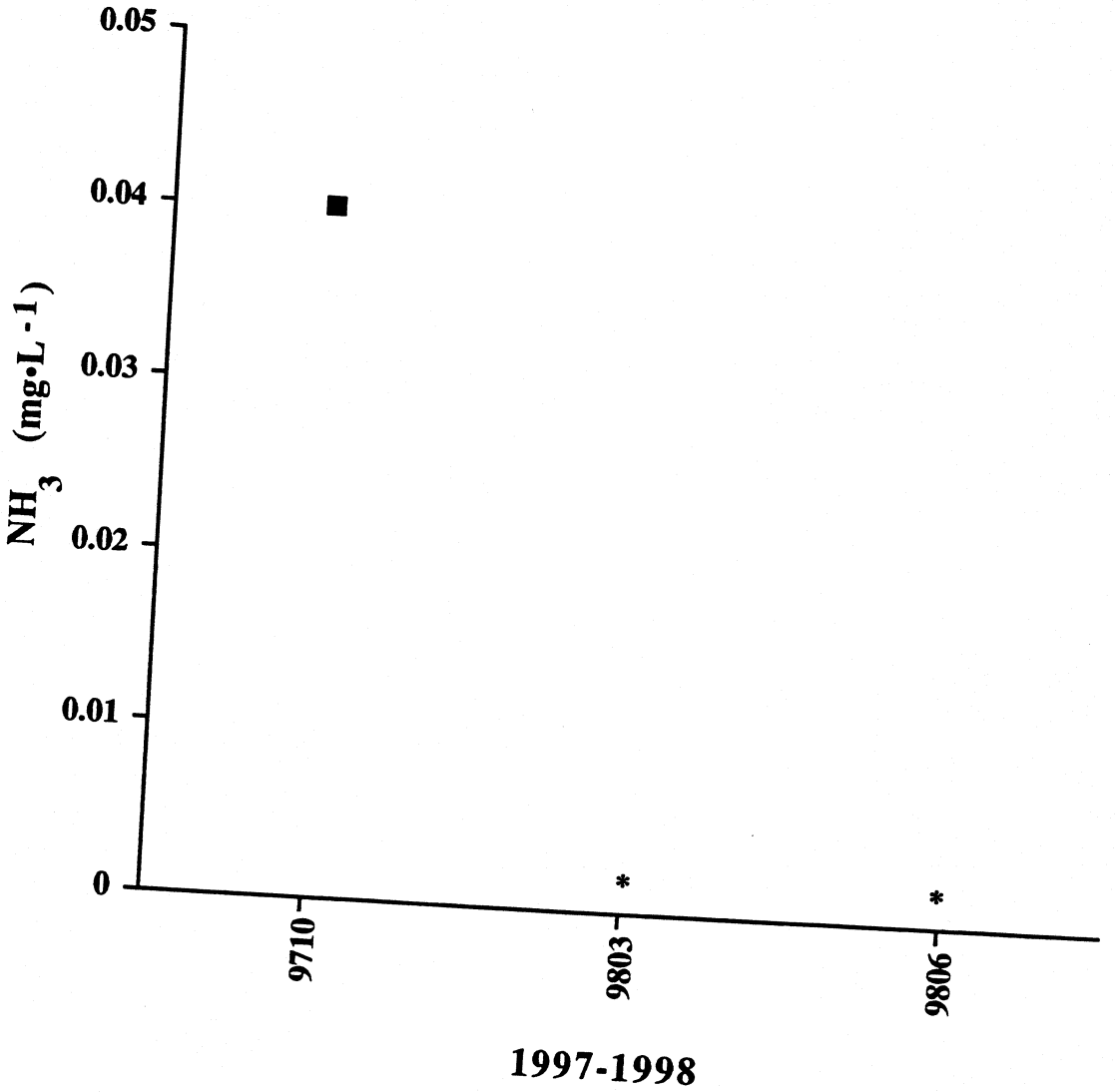

Figure 39. Ammonia (NH_3 mg•L^{-1}) collected at Tanner Canyon Rkm 108.8 from October 1997 to June 1998. Samples below detectable levels are represented by a (*).

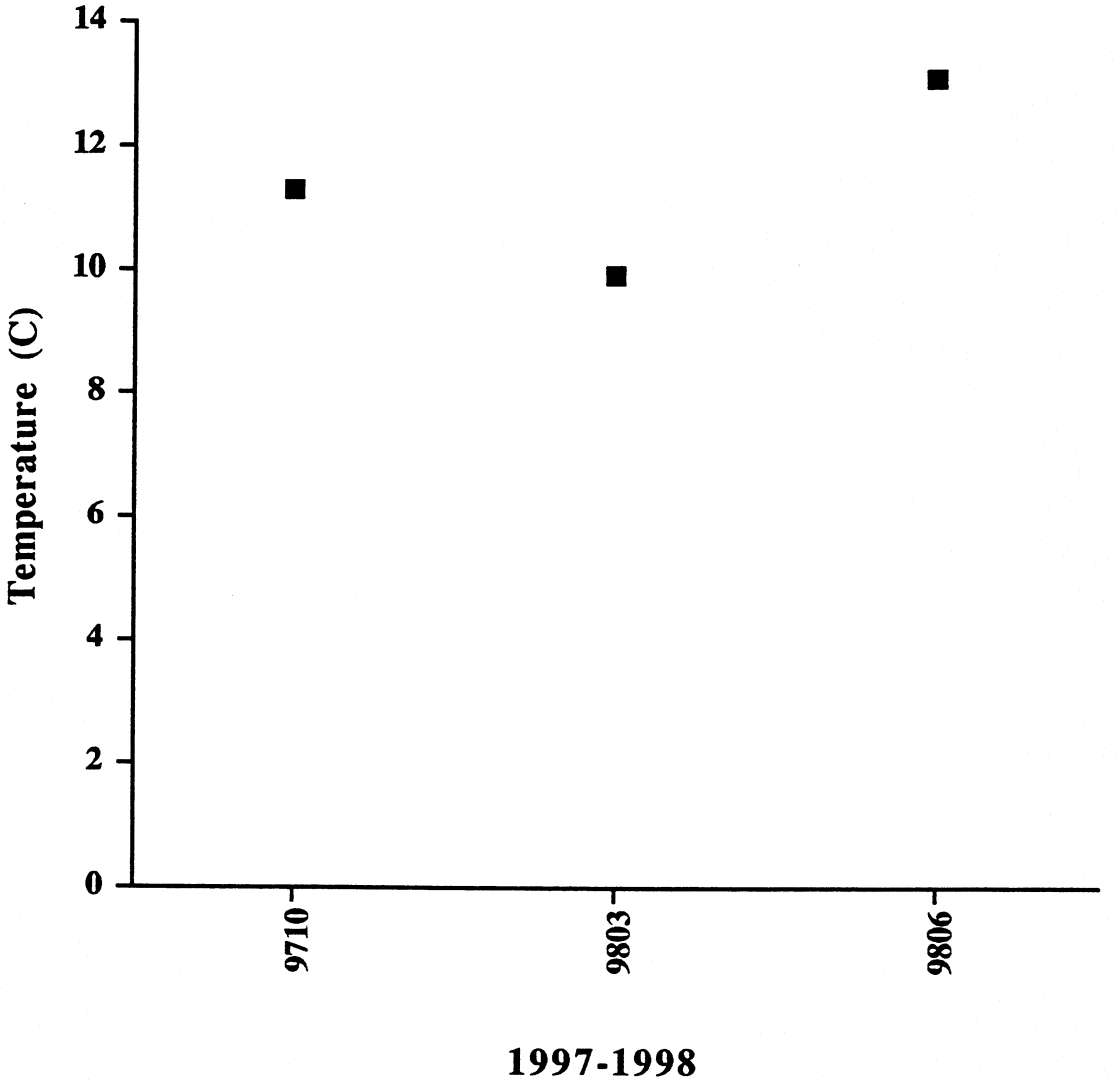

Figure 42. Water temperature (C) collected at 127 Mile Rkm 202.9 from October 1997 to June 1998.

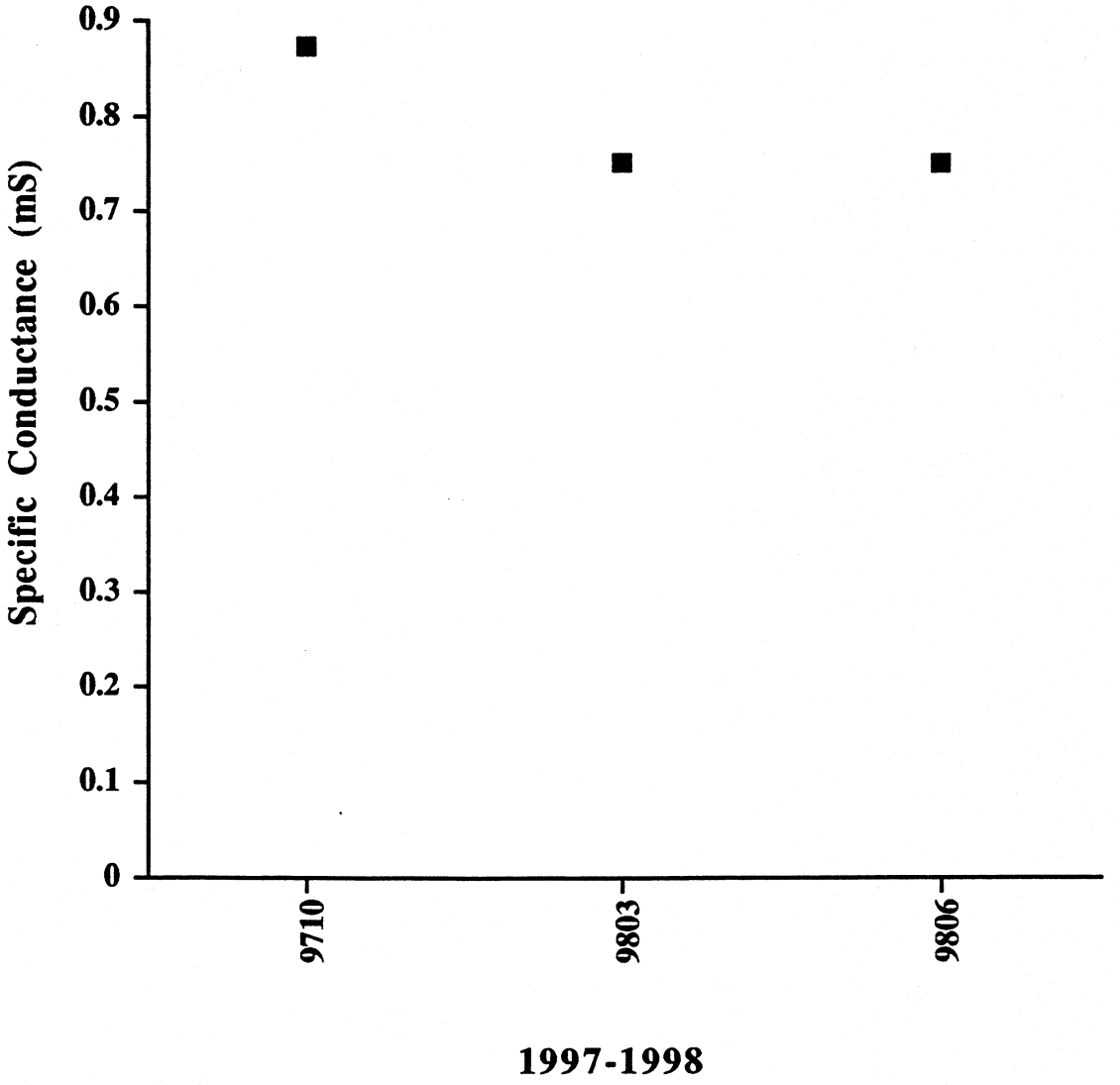

Figure 43. Specific conductance (mS) collected at 127 Mile Rkm 202.9 from October 1997 to June 1998.

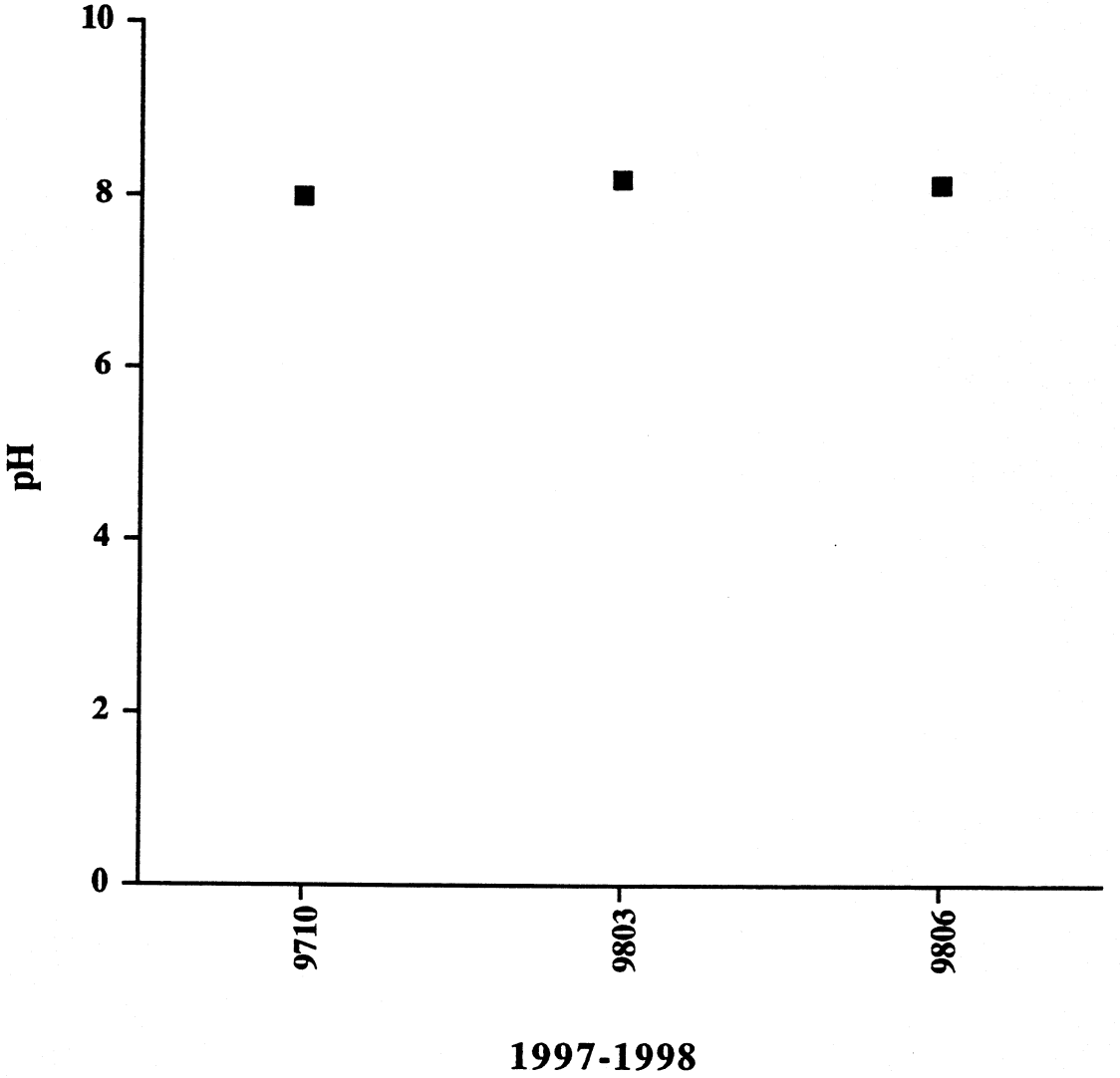

Figure 44. pH collected at 127 Mile Rkm 202.9 from October 1997 to June 1998.

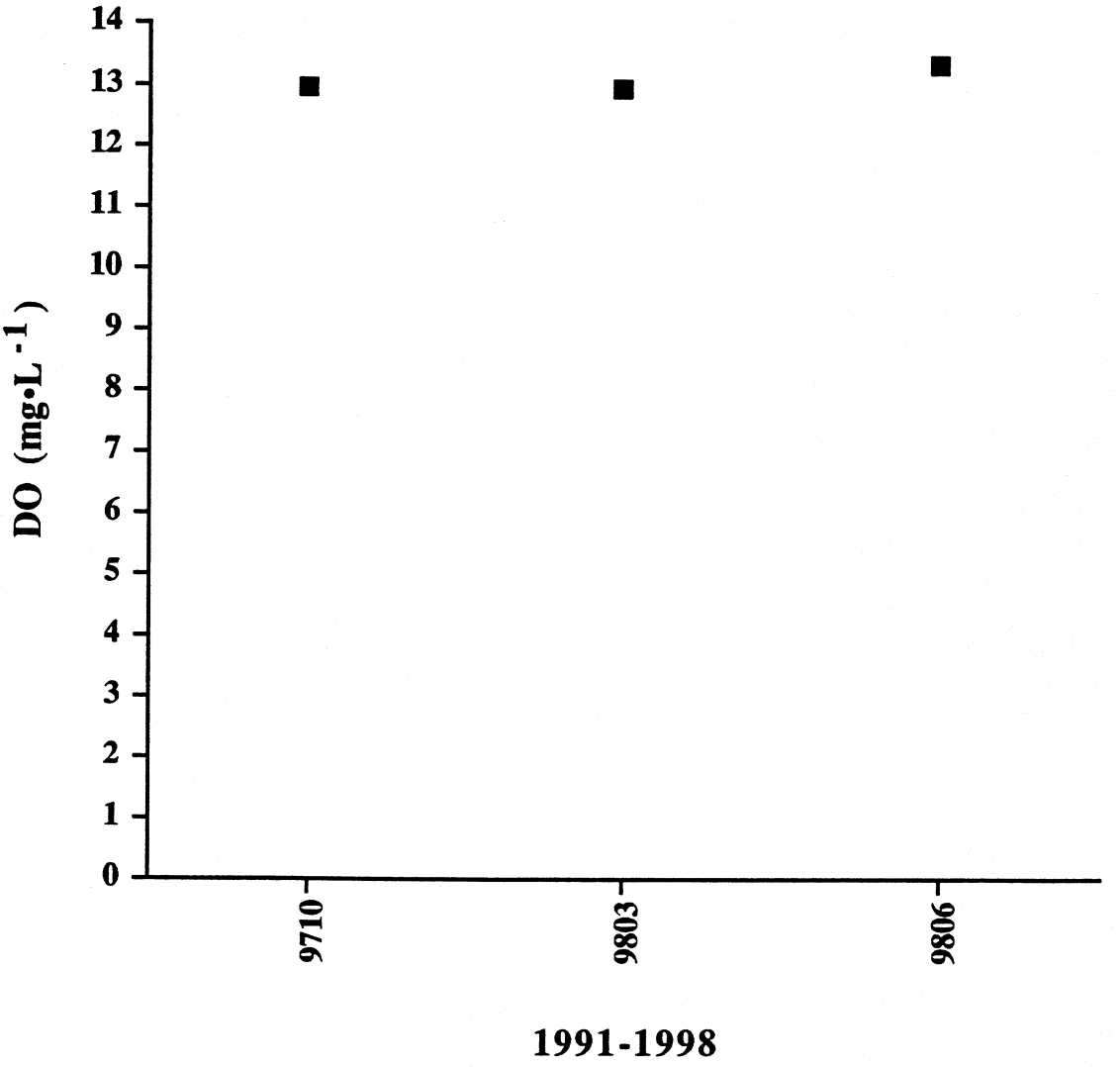

Figure 45. Dissolved oxygen ($mg \cdot L^{-1}$) collected at 127 Mile Rkm 202.9 from October 1997 to June 1998.

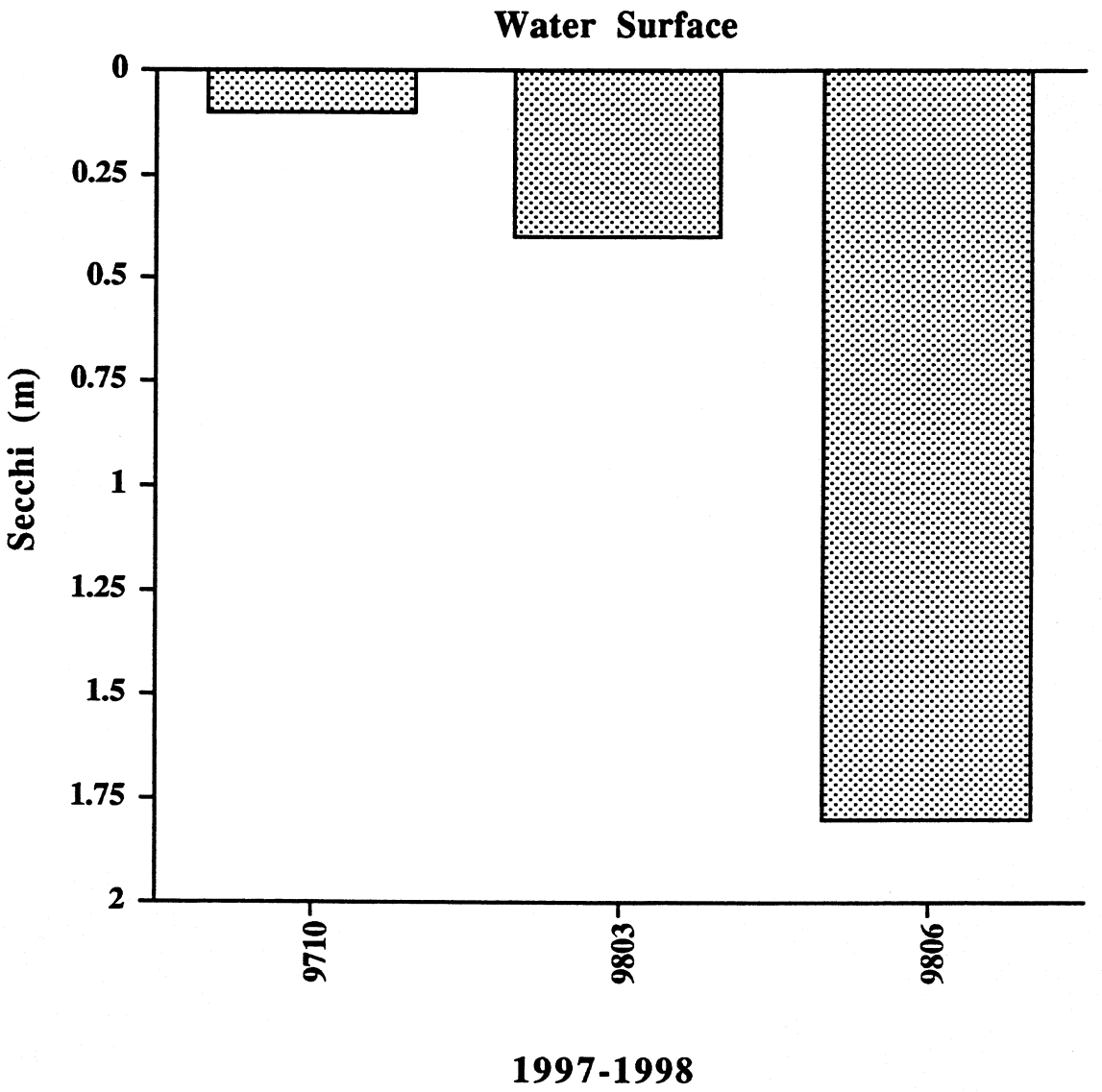

Figure 46. Secchi depth (m) collected at 127 Mile Rkm 202.9 from October 1997 to June 1998.

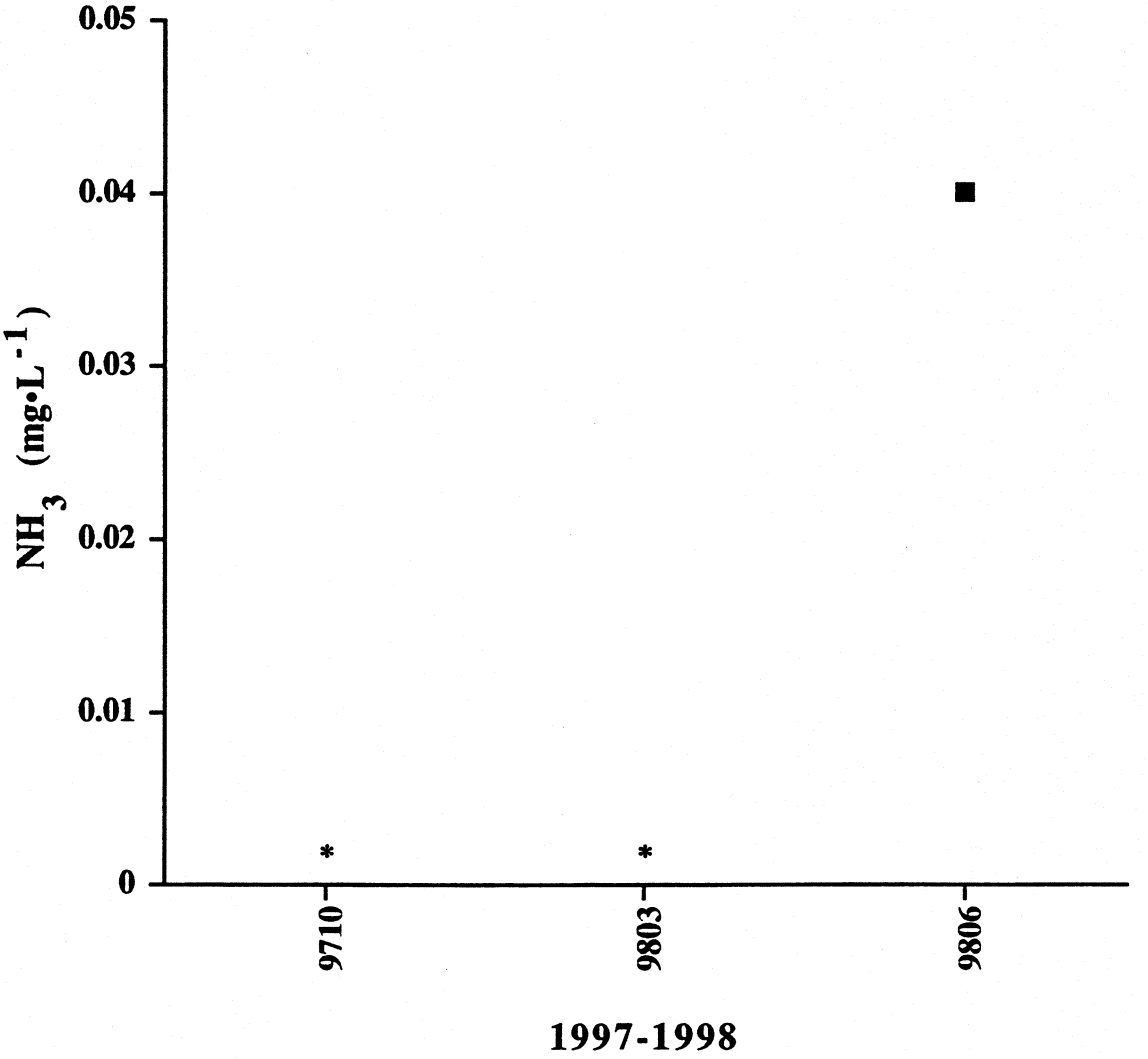

Figure 47. Ammonia (NH_3 mg•L^{-1}) collected at 127 Mile Rkm 202.9 from October 1997 to June 1998. Samples below detectable levels are represented by a (*).

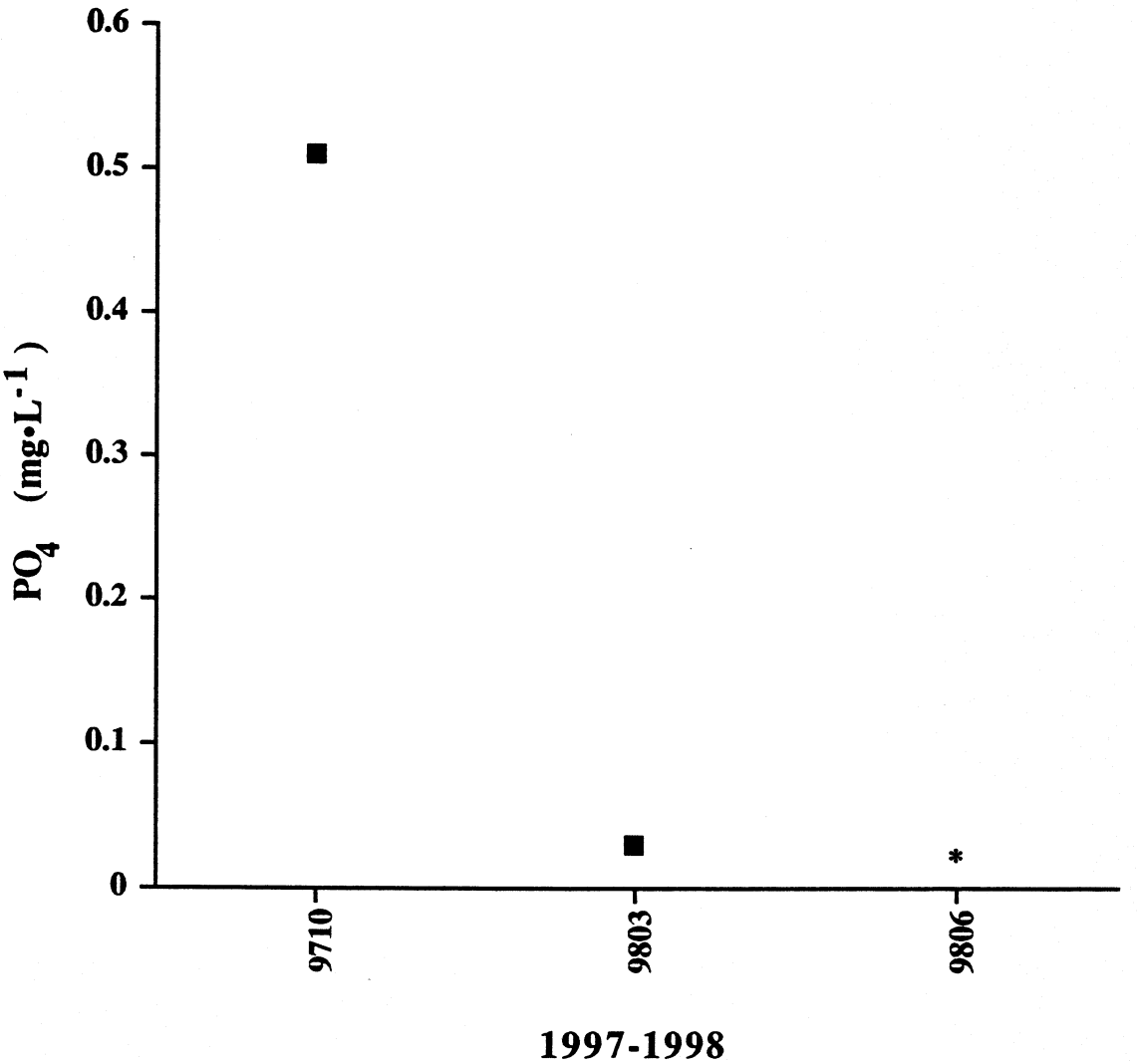

Figure 48. Phosphate (PO_4 mg•L^{-1}) collected at 127 Mile Rkm 202.9 from October 1997 to June 1998. Samples below detectable levels are represented by a (*).

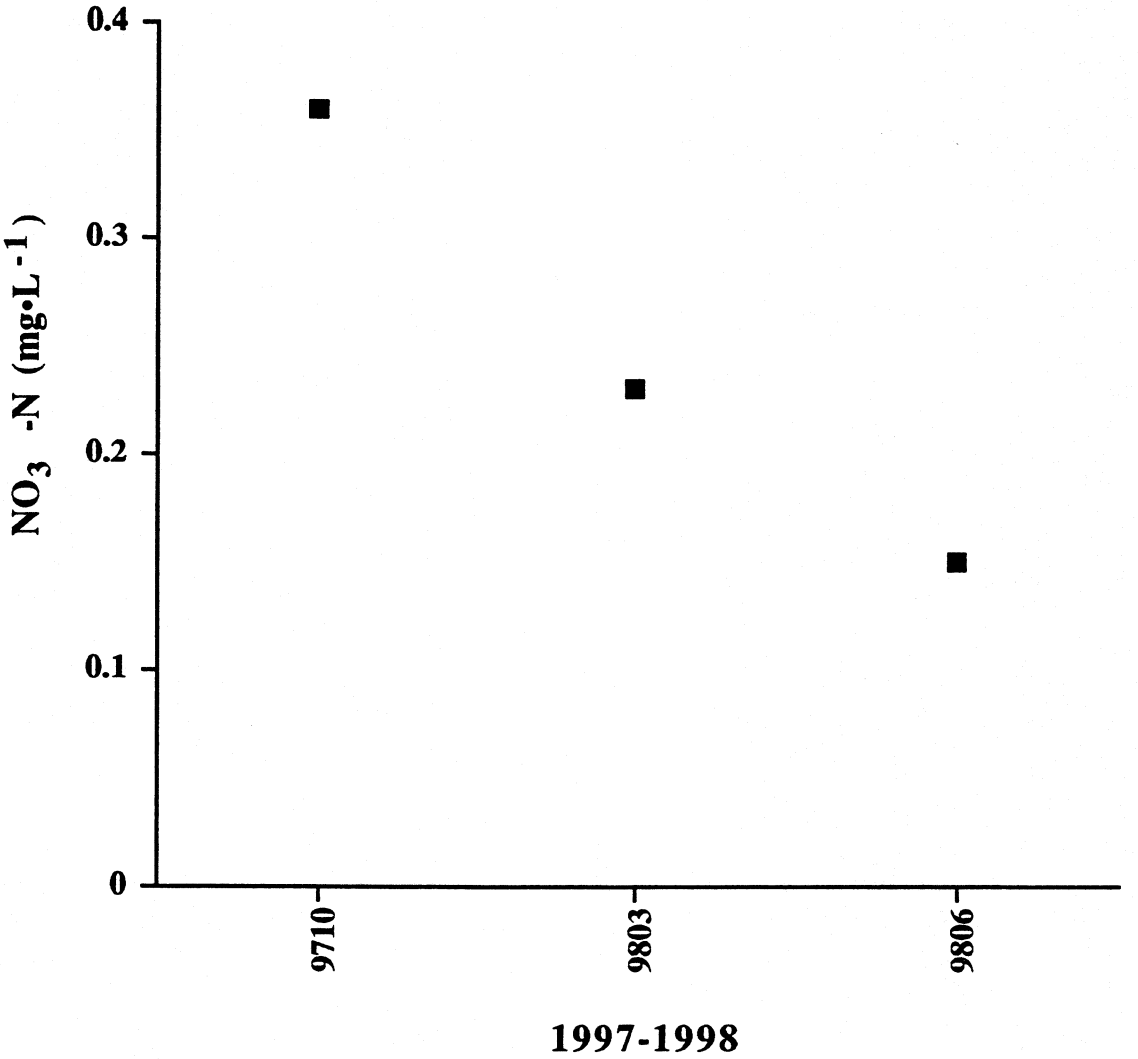

Figure 49. Nitrate-nitrogen (NO_3 -N mg•L^{-1}) collected at 127 Mile Rkm 202.9 from October 1997 to June 1998.

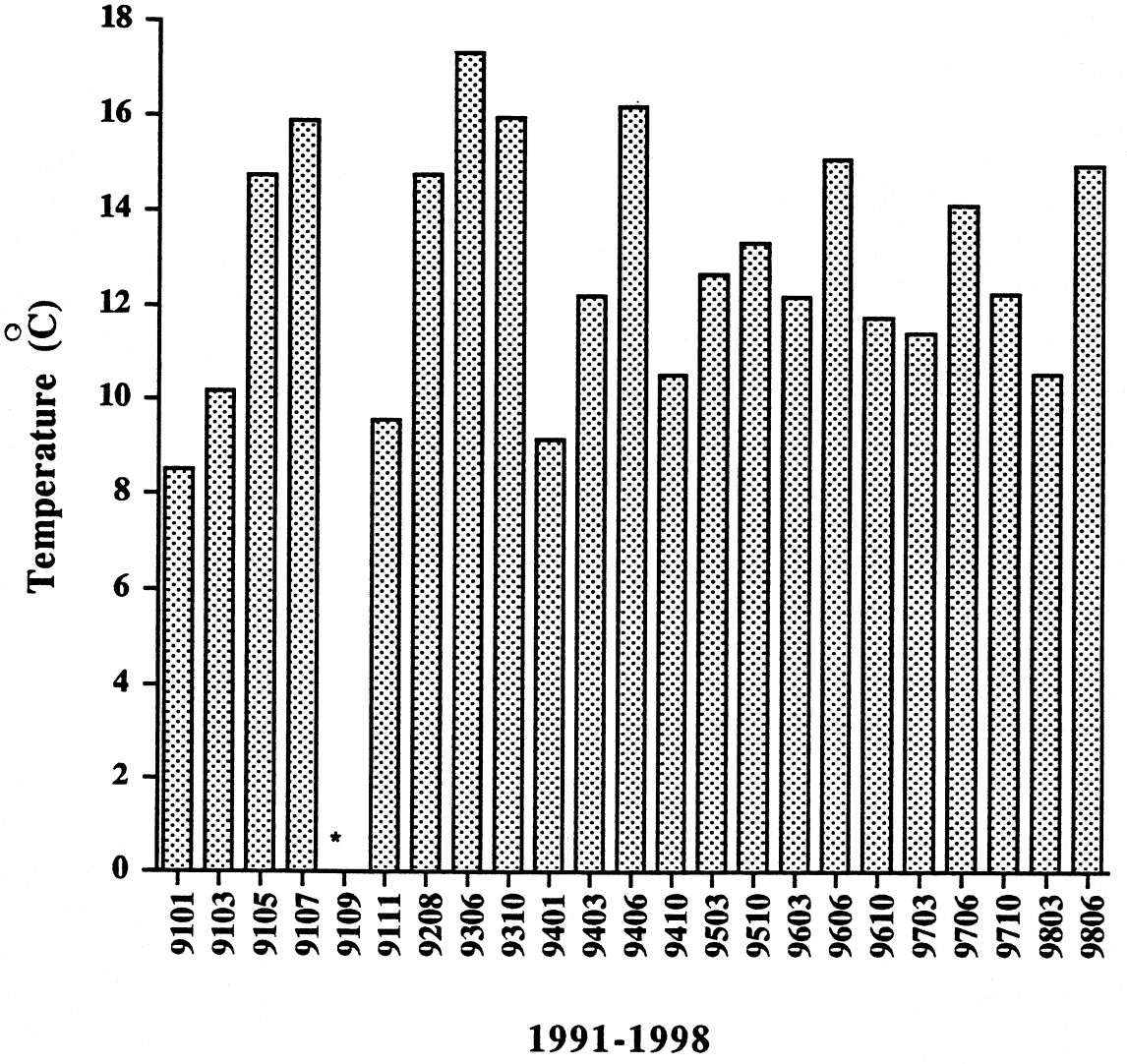

Figure 50. Water temperature (°C) collected at 205 Mile Rkm 328.8 from January 1991 to June 1998. Asterisk (*) represents missing data point.

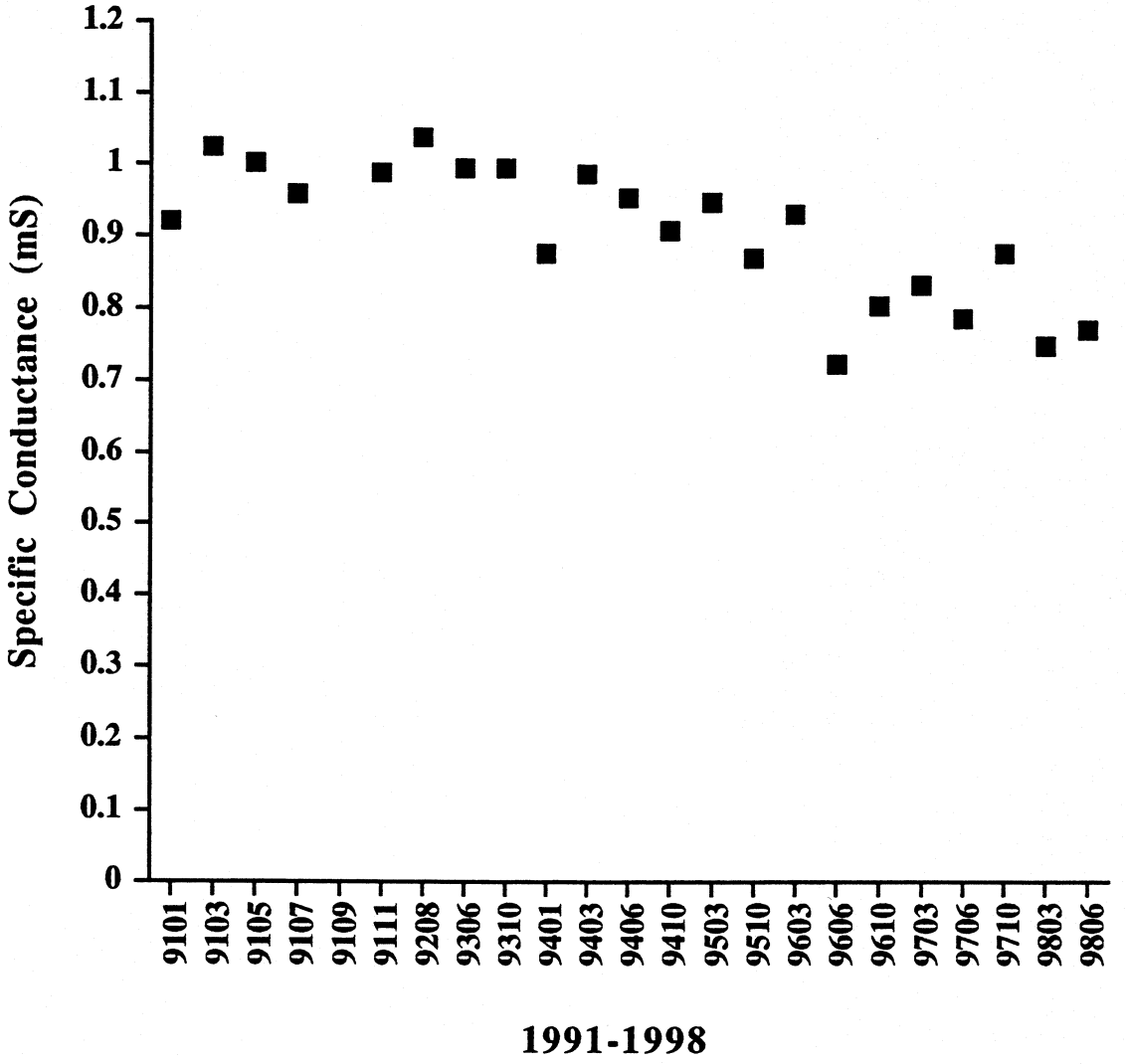

Figure 51. Specific conductance (mS) collected at 205 Mile Rkm 328.8 from January 1991 to June 1998.

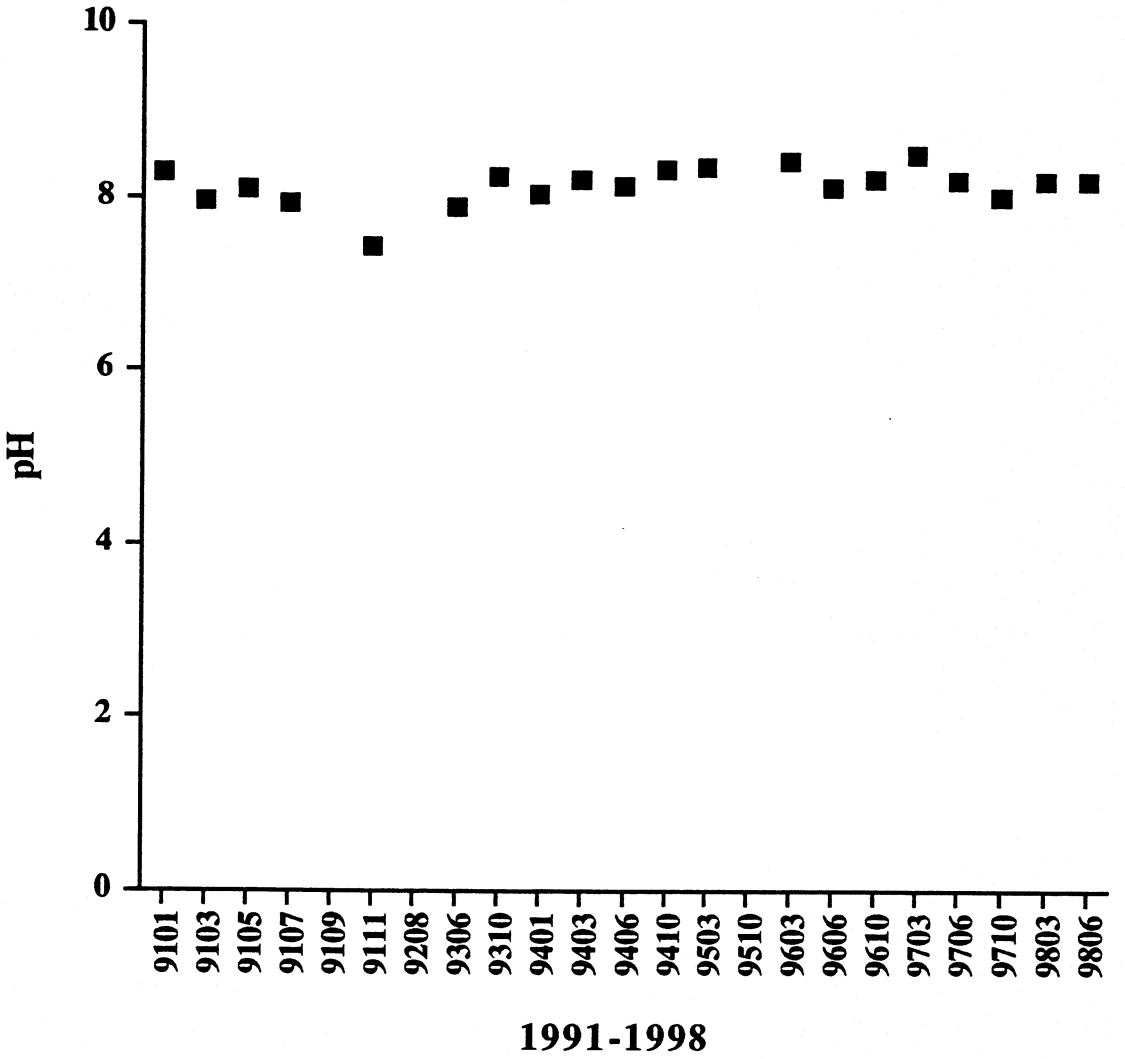

Figure 52. pH collected at 205 Mile Rkm 328.8 from January 1991 to June 1998.

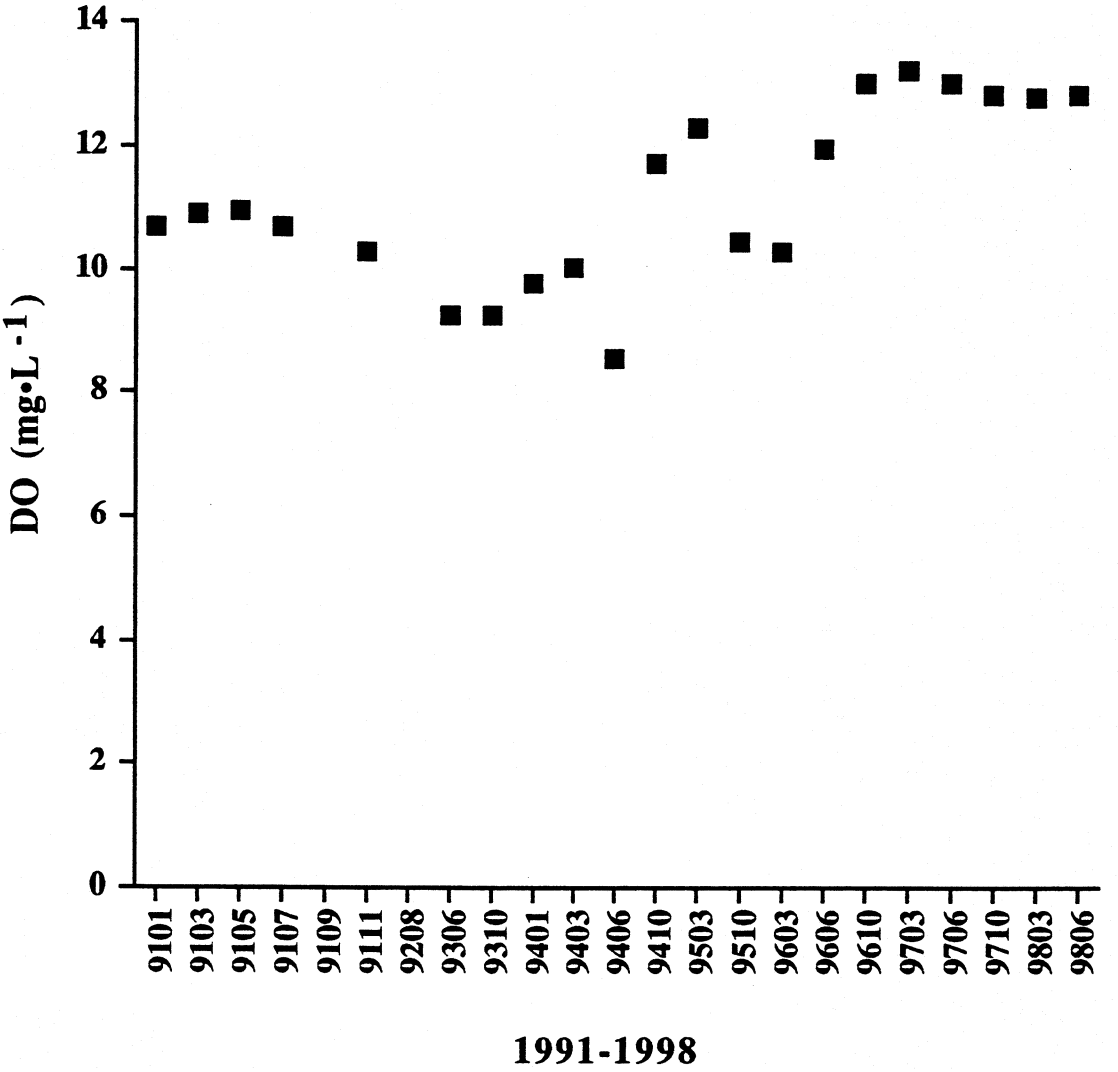

Figure 53. Dissolved oxygen ($mg \cdot L^{-1}$) collected at 205 Mile Rkm 328.8 from January 1991 to June 1998.

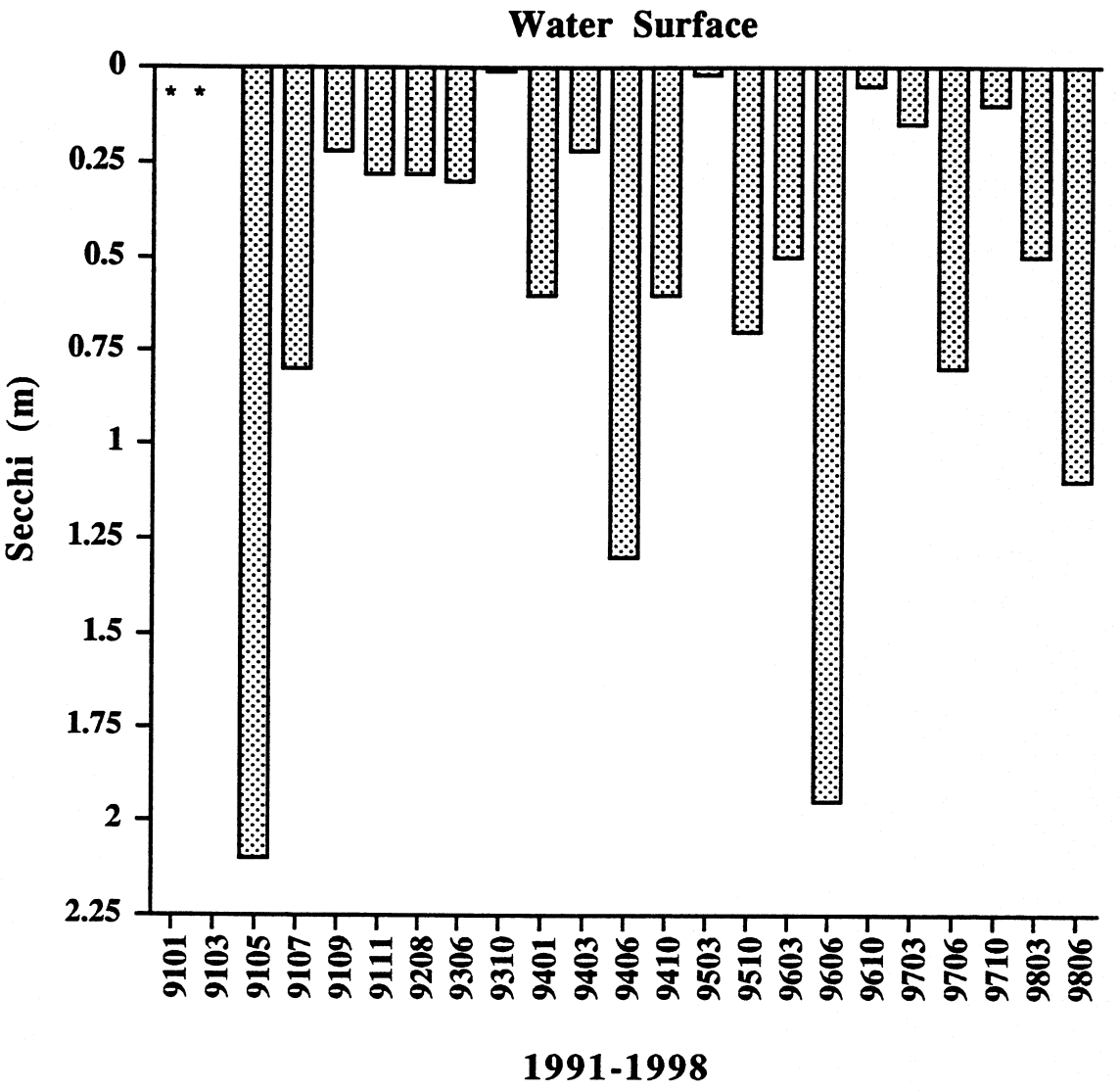

Figure 54. Secchi depth (m) collected at 205 Mile Rkm 328.8 from January 1991 to June 1998. Asterisk (*) represents missing data point.

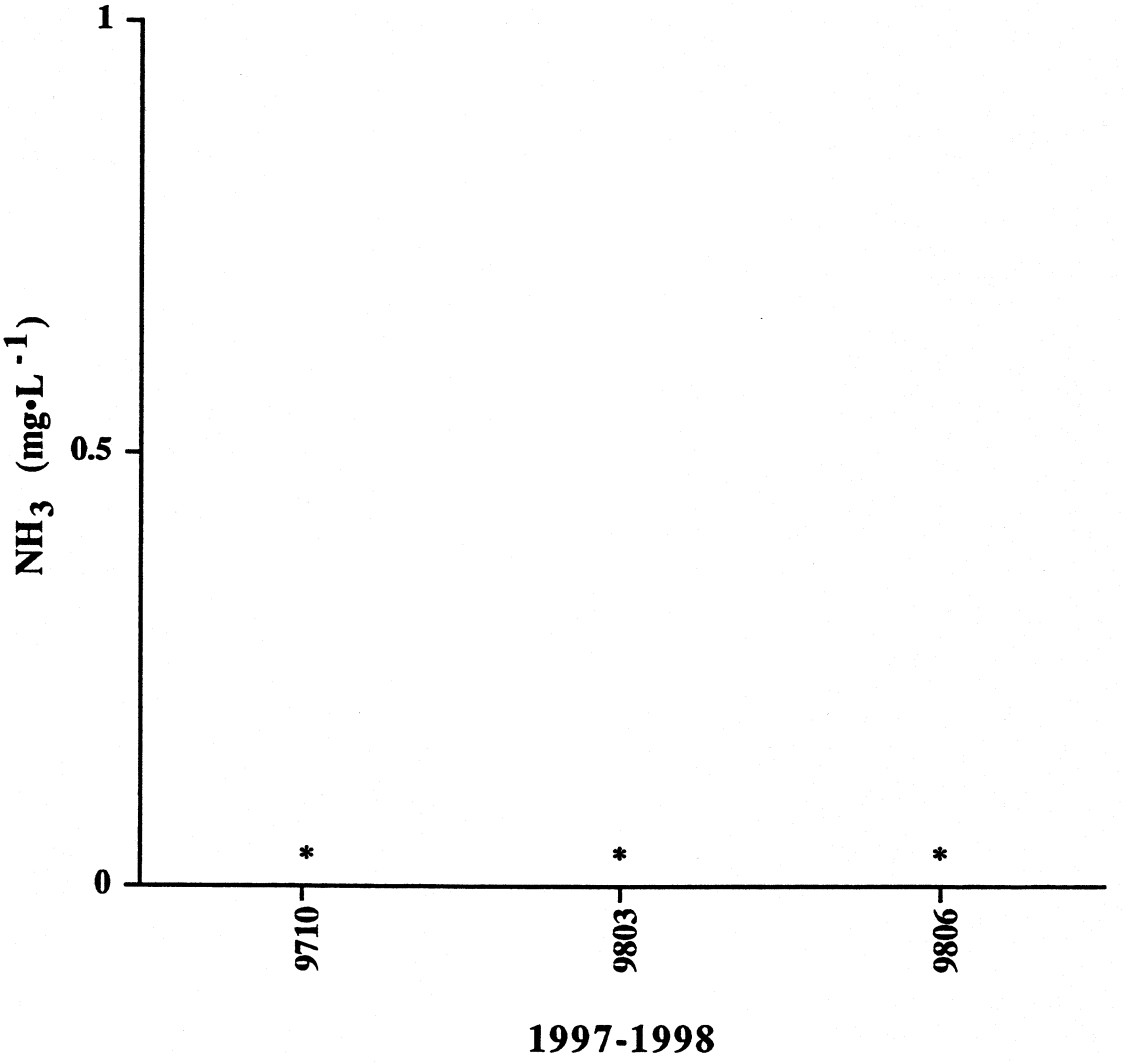

Figure 55. Ammonia (NH_3 mg•L^{-1}) collected at 205 Mile Rkm 328.8 from October 1997 to June 1998. Samples below detectable levels are represented by (*).

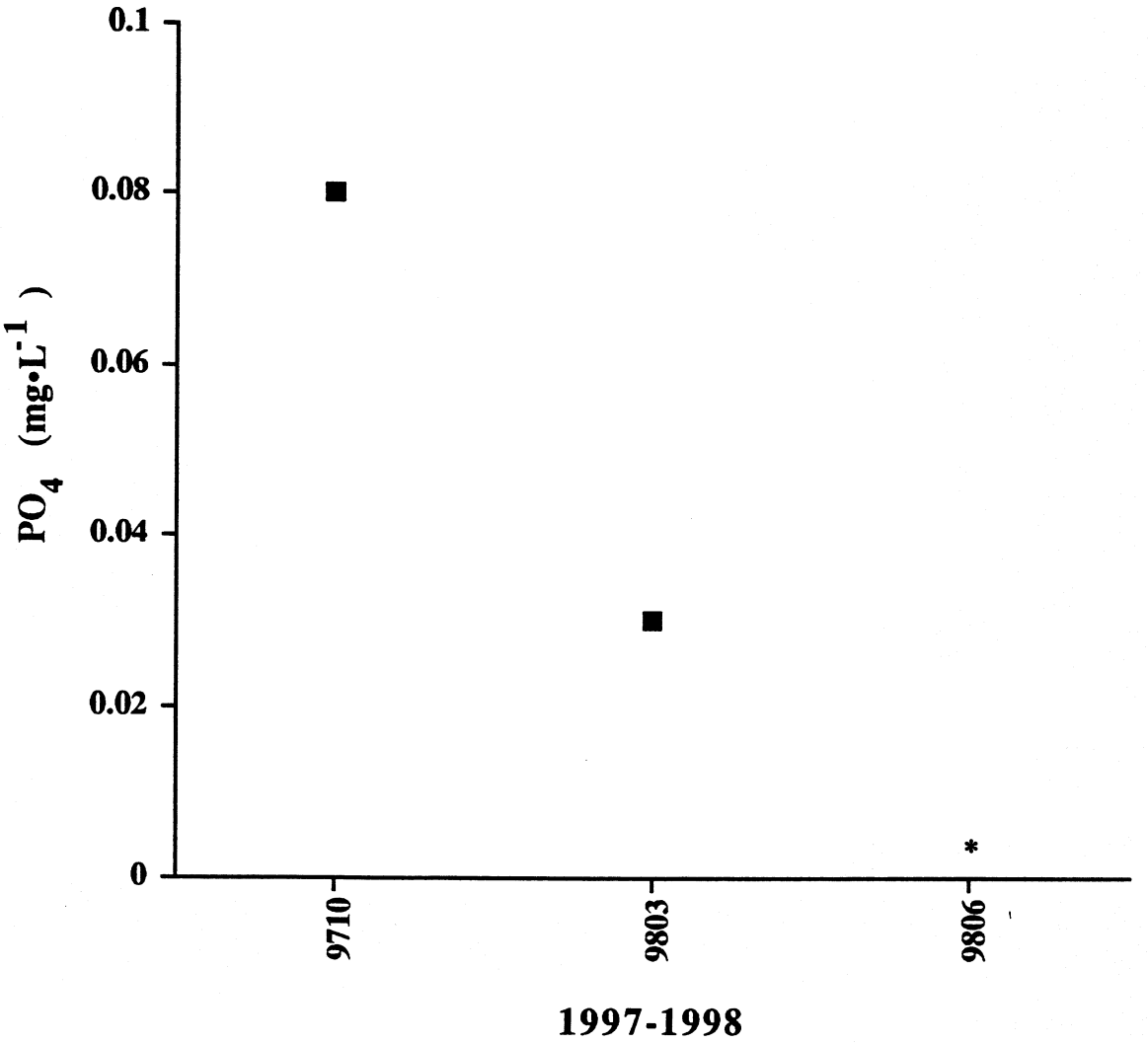

Figure 56. Phosphate (PO_4 mg•L^{-1}) collected at 205 Mile Rkm 328.8 from October 1997 to June 1998. Samples below detectable levels are represented by a (*).

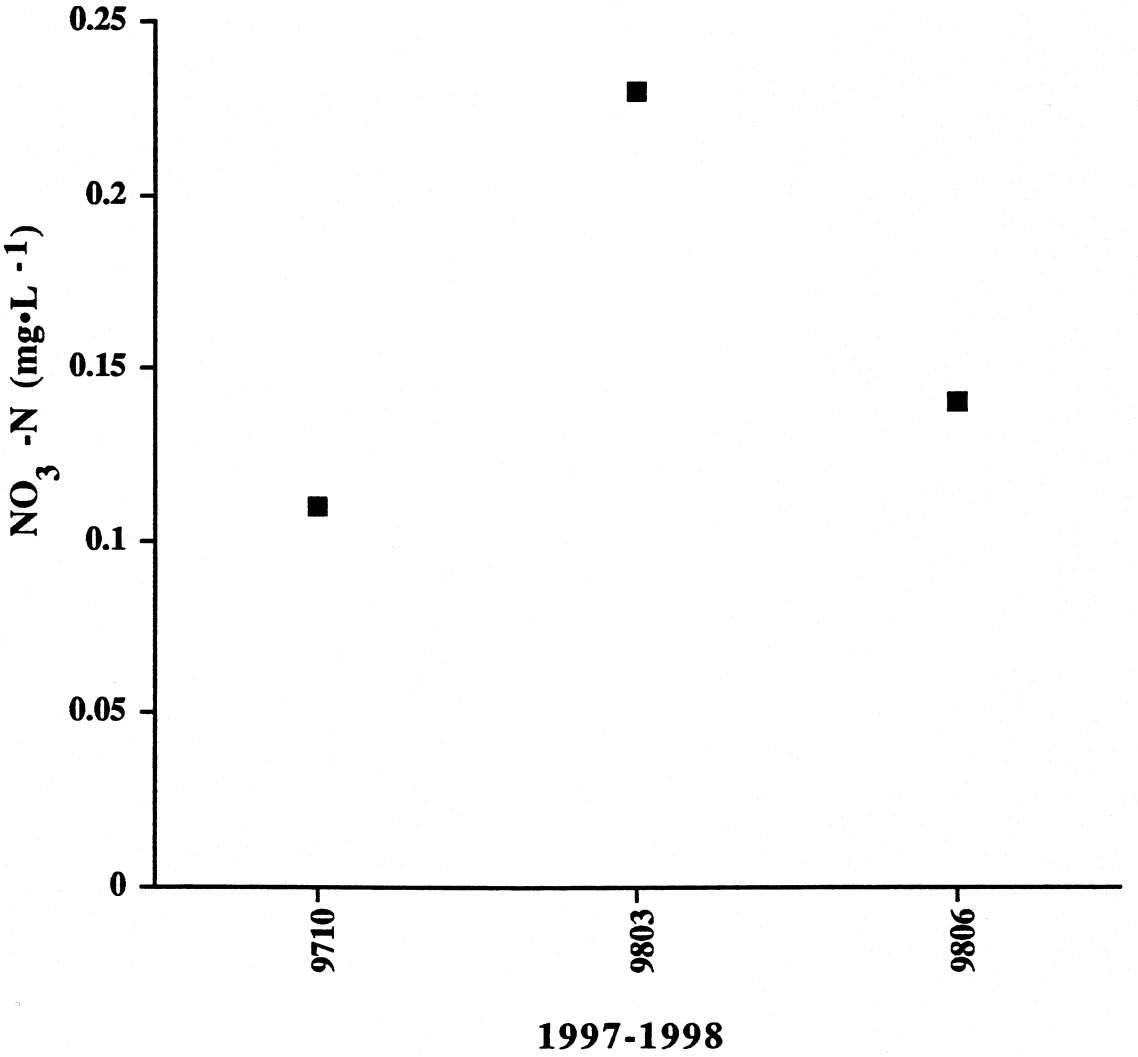

Figure 57. Nitrate-nitrogen (NO_3 -N mg•L^{-1}) collected at 205 Mile Rkm 328.8 from October 1997 to June 1998.

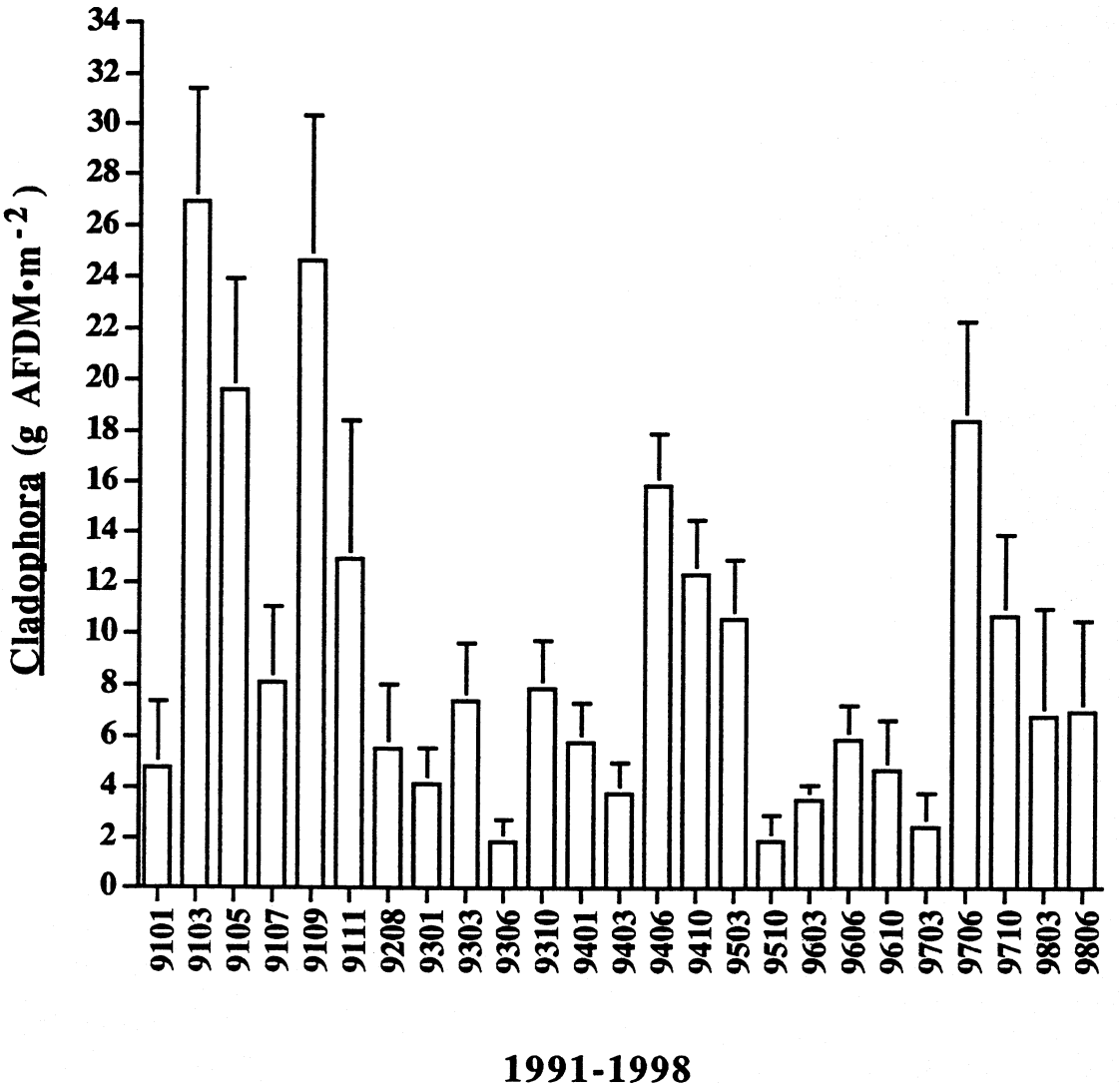

Figure 58. Cladophora biomass estimates (g AFDM•m^{-2}) at Lees Ferry cobble Rkm 0.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

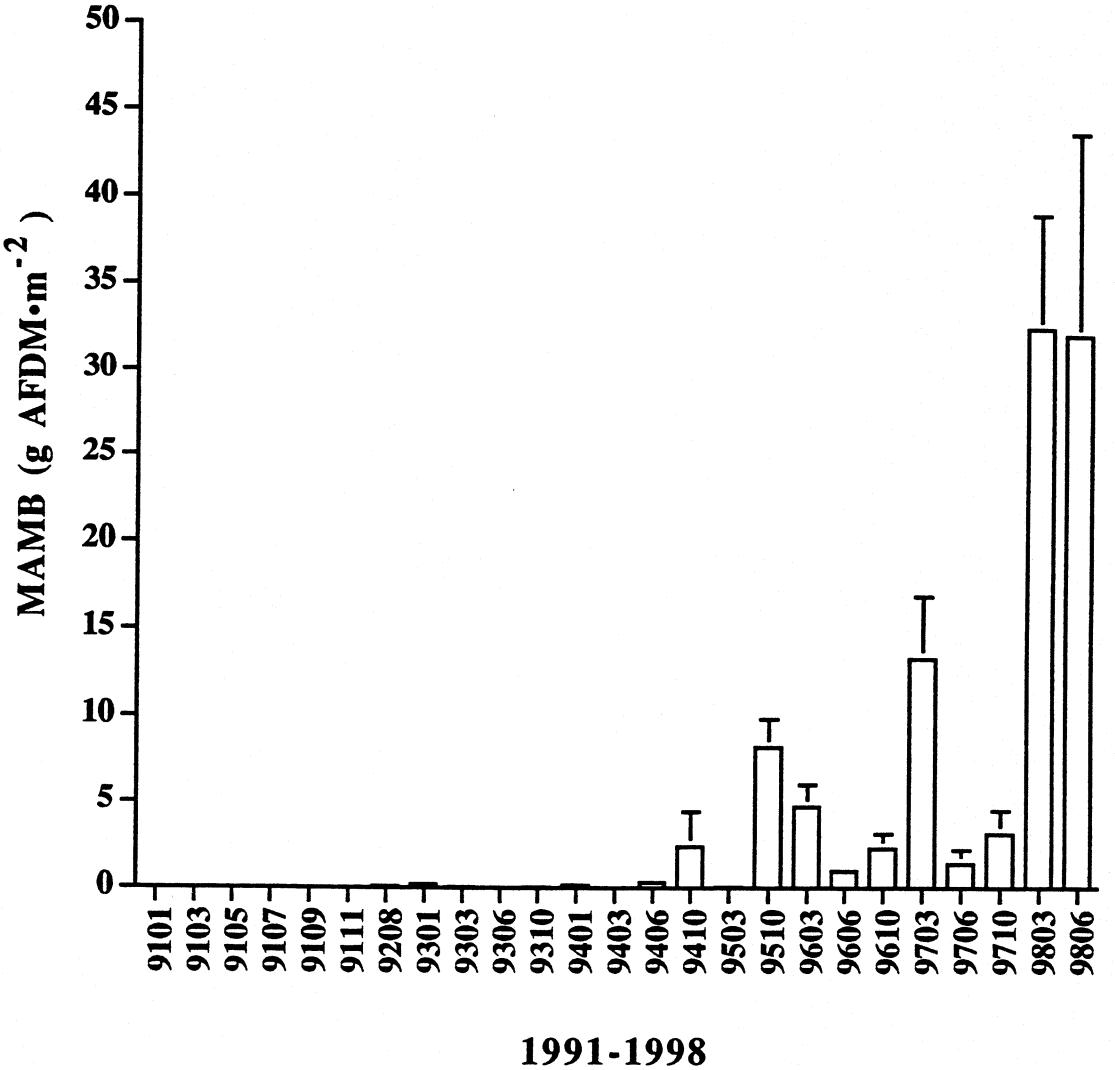

Figure 59. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^2) at Lees Ferry cobble Rkm 0.8 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=6).

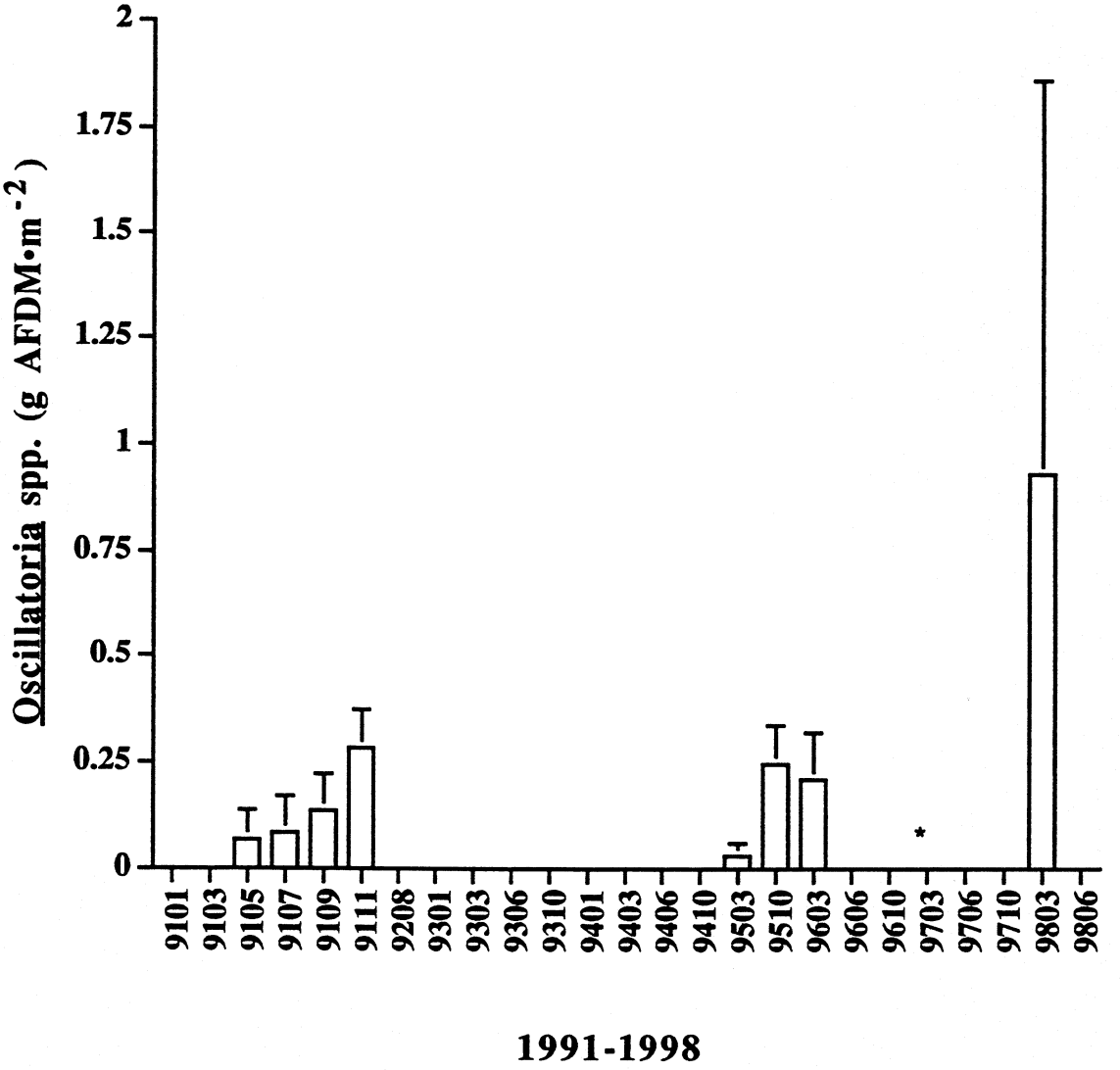

Figure 60. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at Lees Ferry cobble Rkm 0.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6). Asterisk (*) represents 7 g AFDM•m^{-2}(± 4).

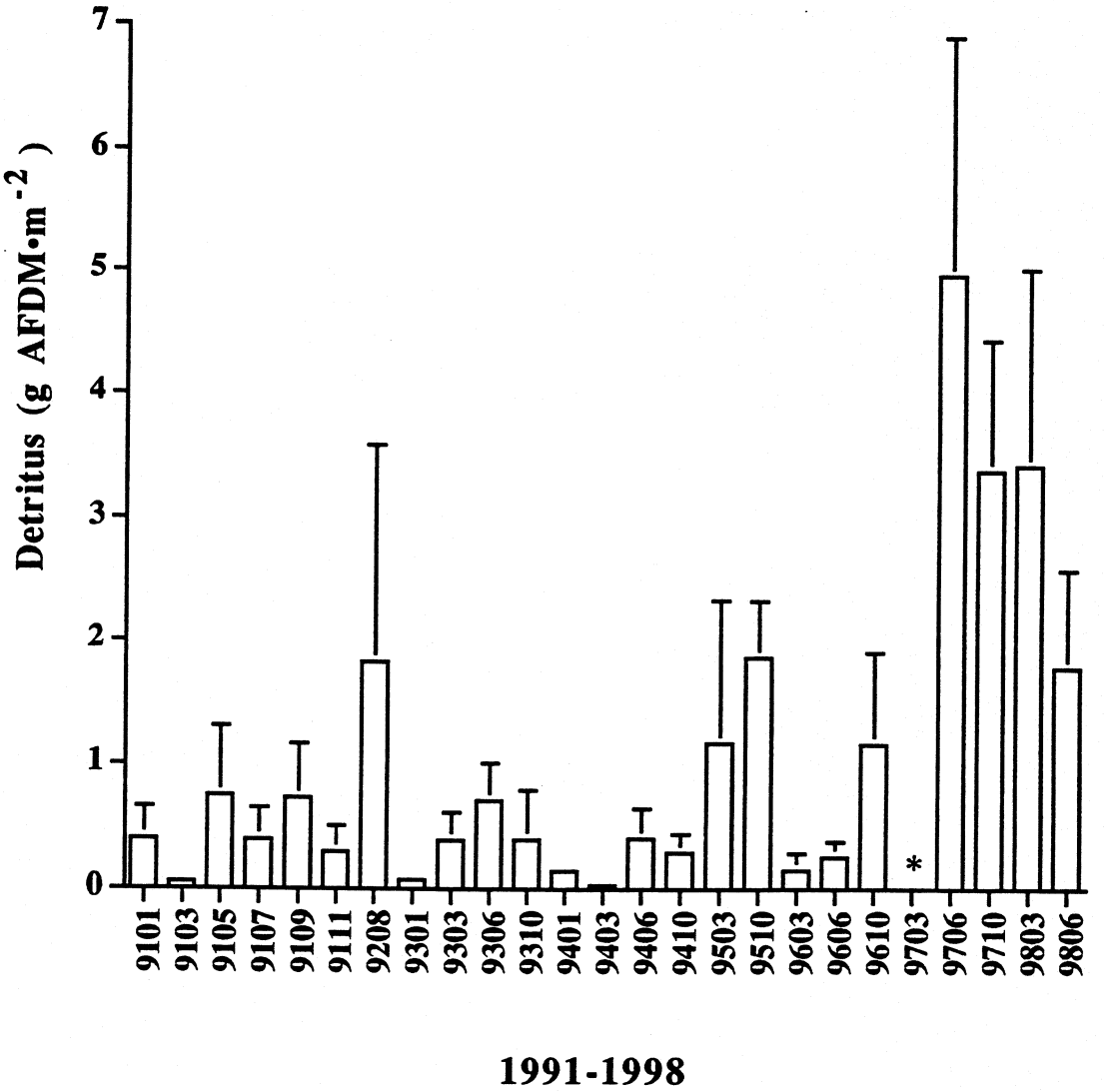

Figure 61. Detritus biomass estimates (g AFDM•m^{-2}) at Lees Ferry cobble Rkm 0.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).
* represents 15 g AFDM•m^{-2} (± 8).

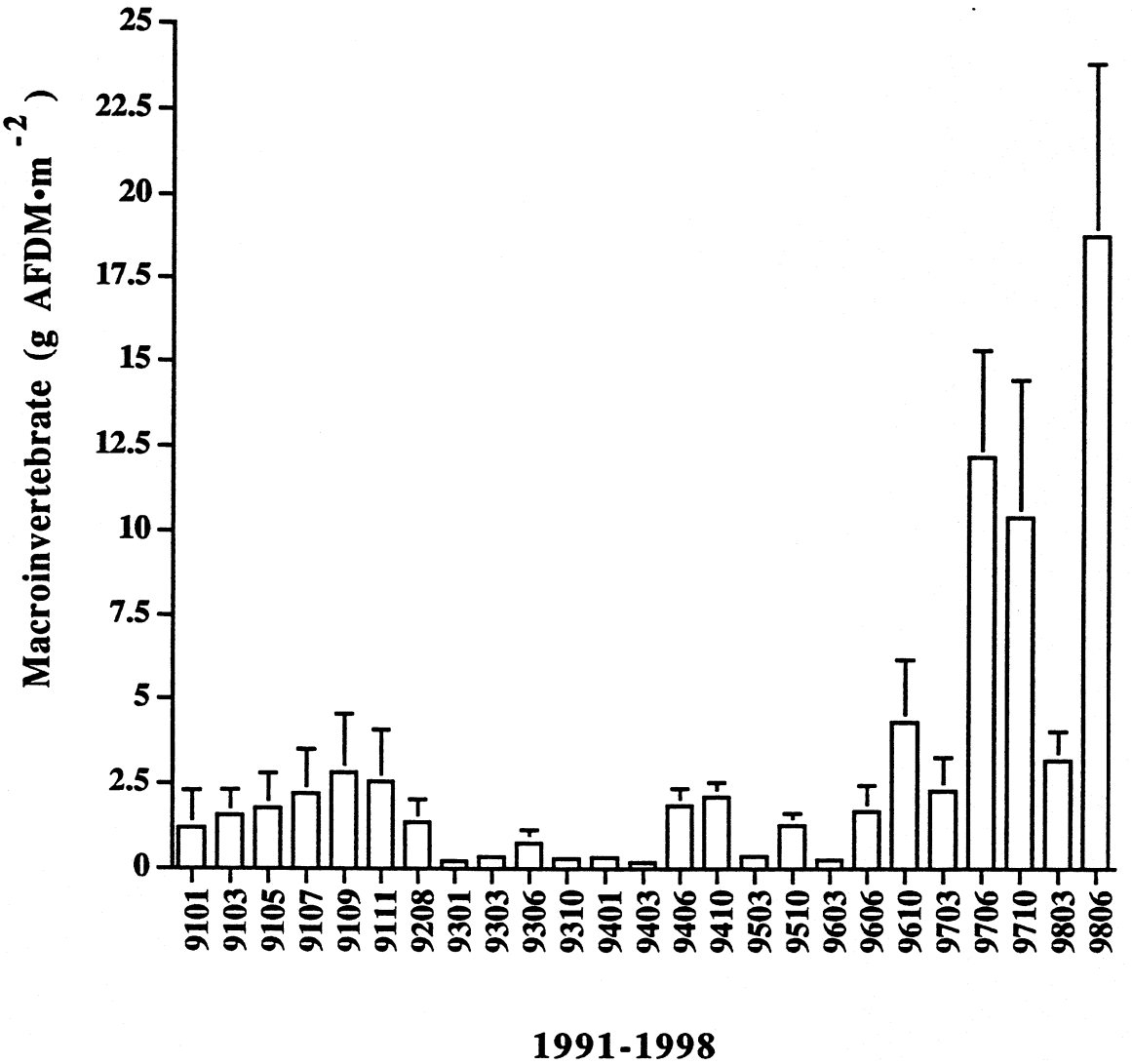

Figure 62. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at Lees Ferry cobble Rkm 0.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

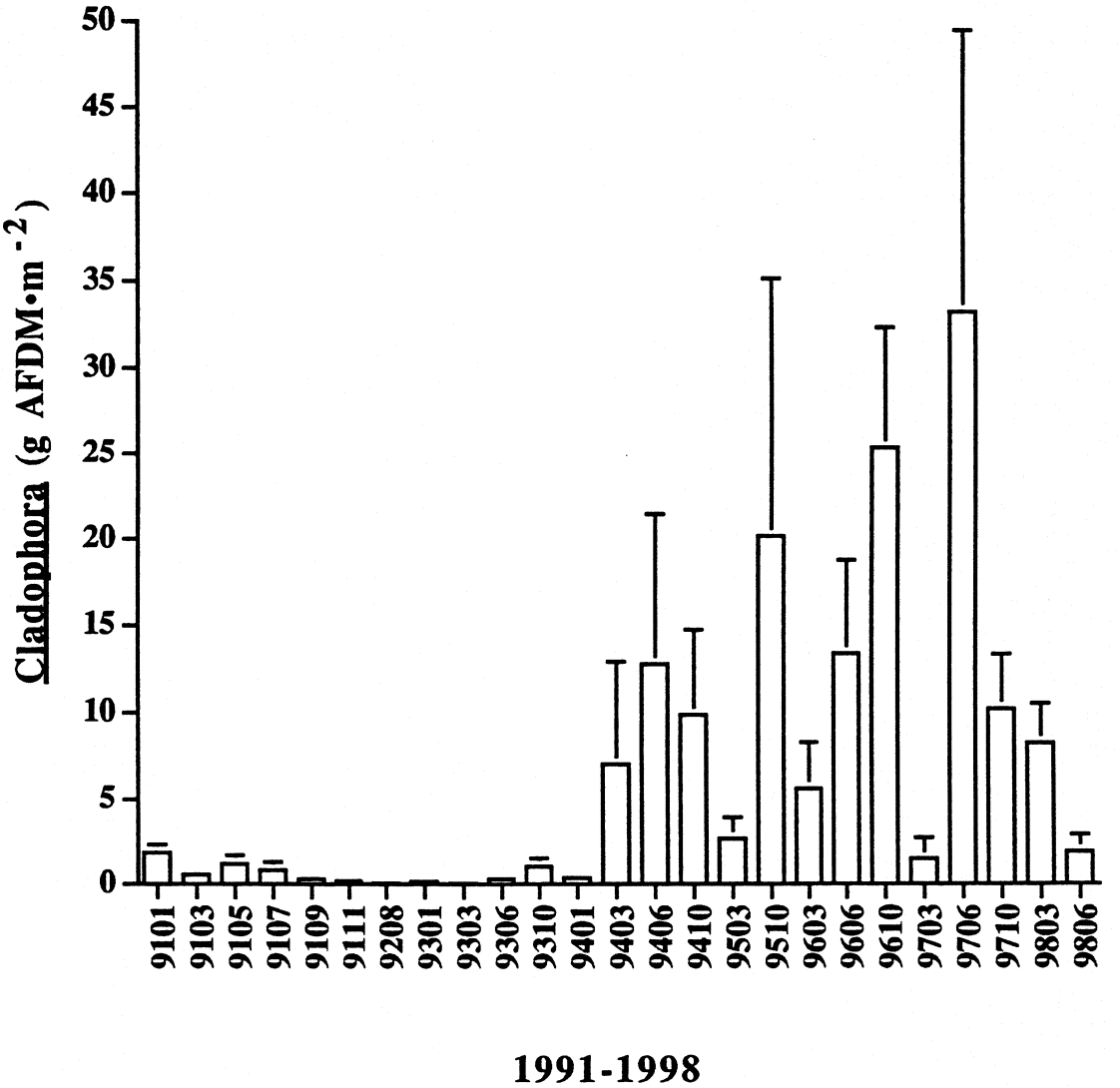

Figure 63. Cladophora biomass estimates (g AFDM•m^{-2}) at Two-Mile Wash cobble Rkm 3.1 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

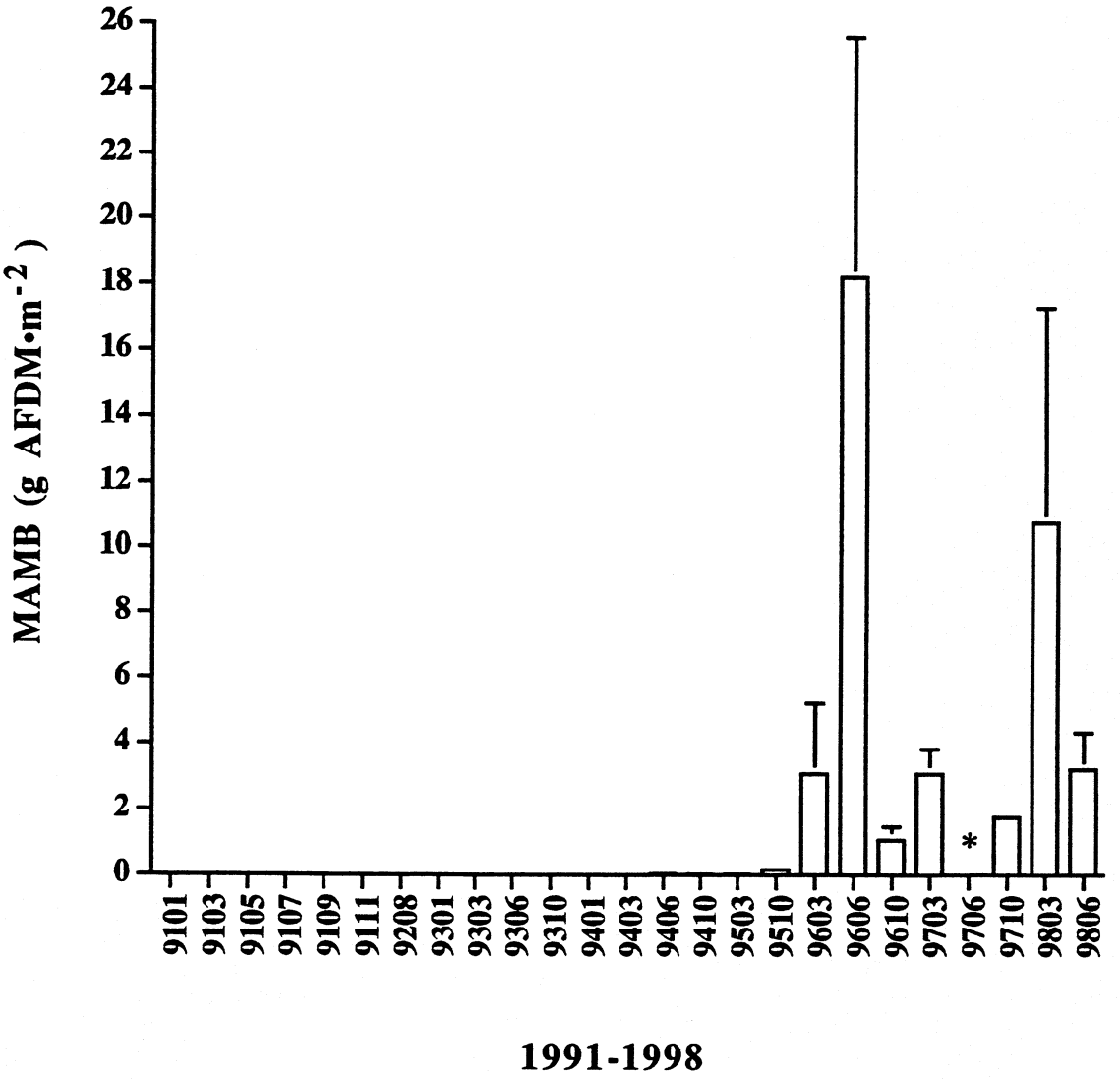

Figure 64. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at Two-Mile Wash cobble Rkm 3.1 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=6).
* represents 65 g AFDM•m^{-2}(± 23).

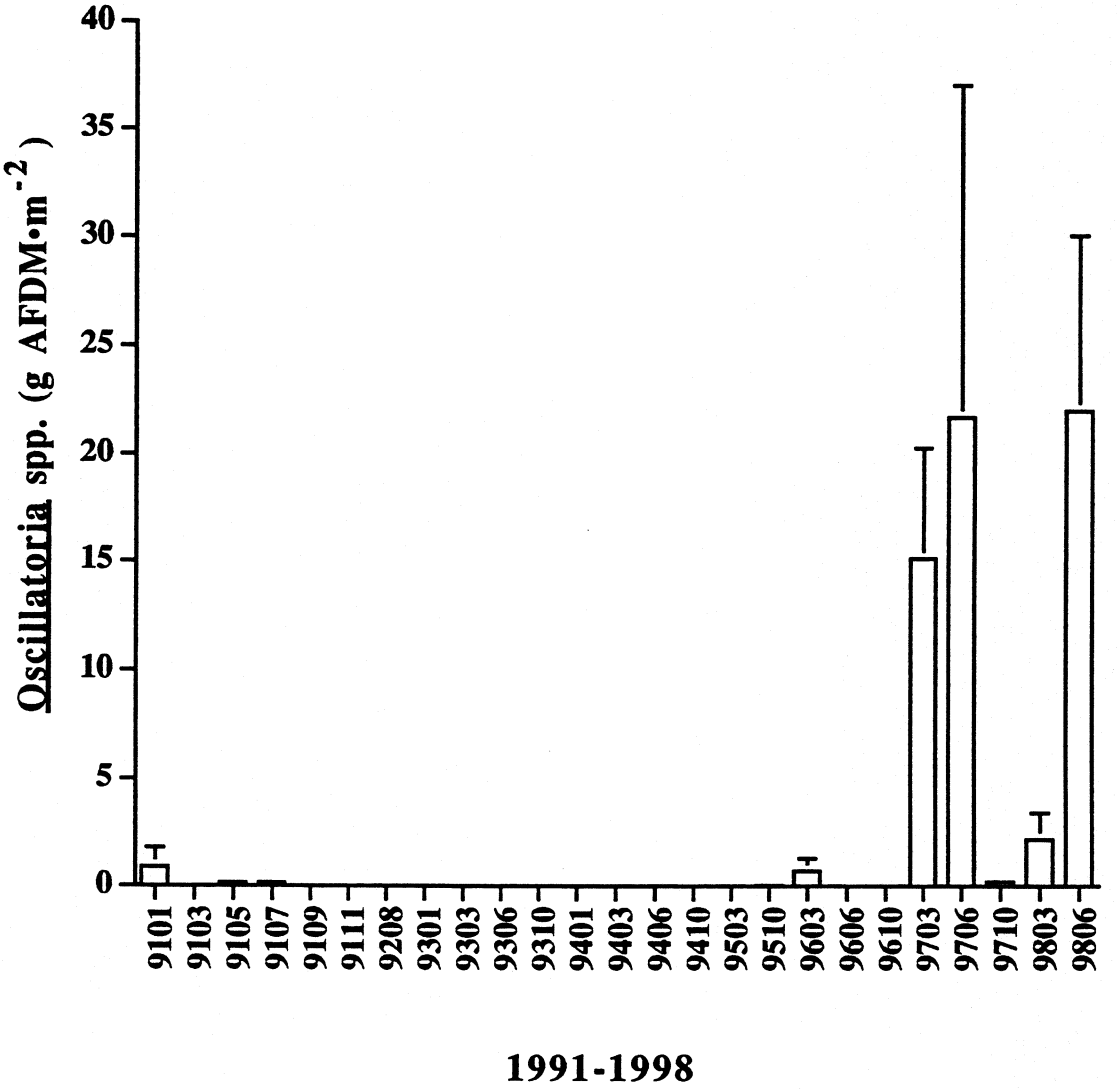

Figure 65. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at Two-Mile Wash cobble Rkm 3.1 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

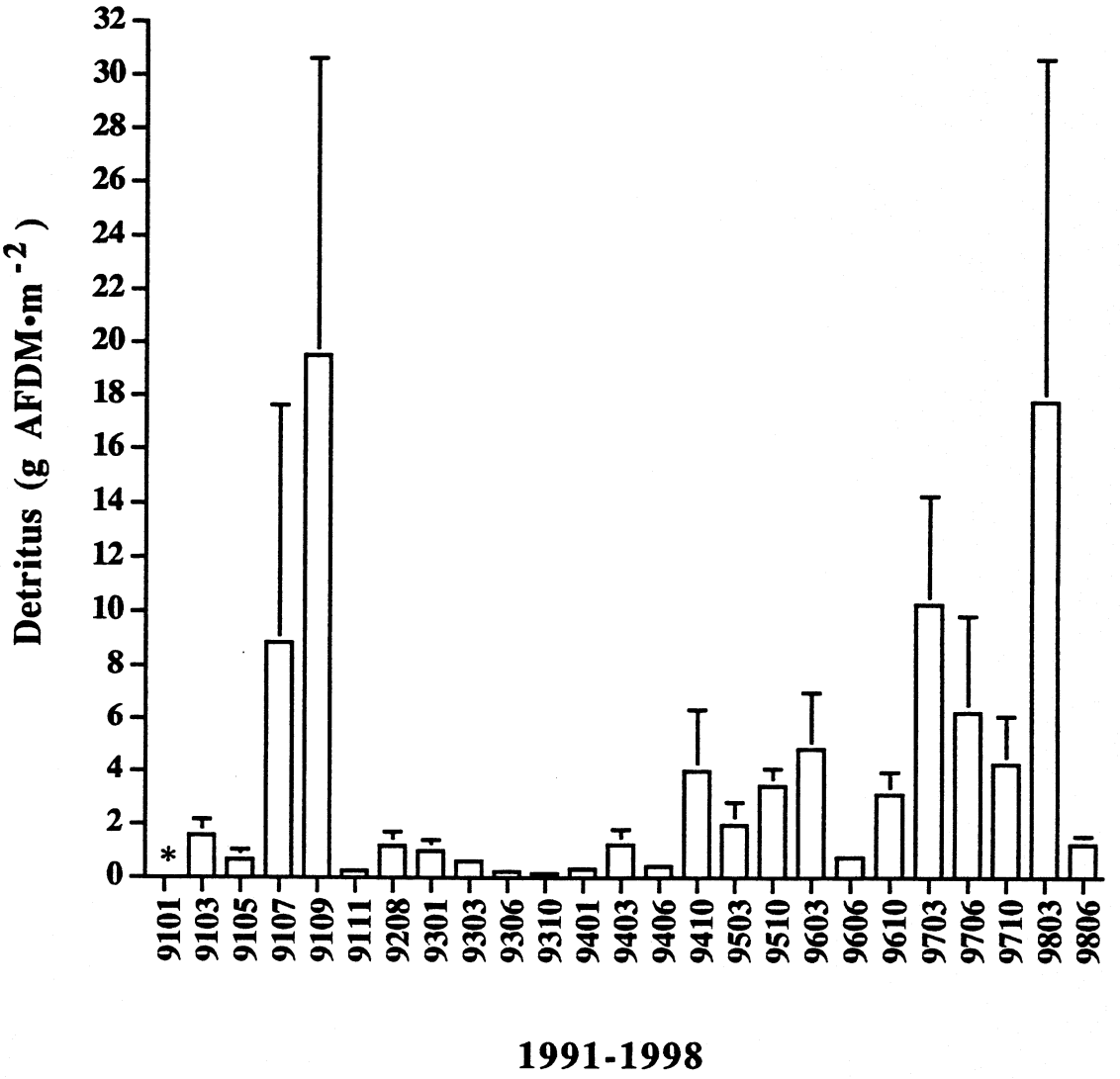

Figure 66. Detritus biomass estimates (g AFDM•m^{-2}) at Two-Mile Wash cobble Rkm 3.1 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).
* represents 29 g AFDM•m^{-2} (± 25).

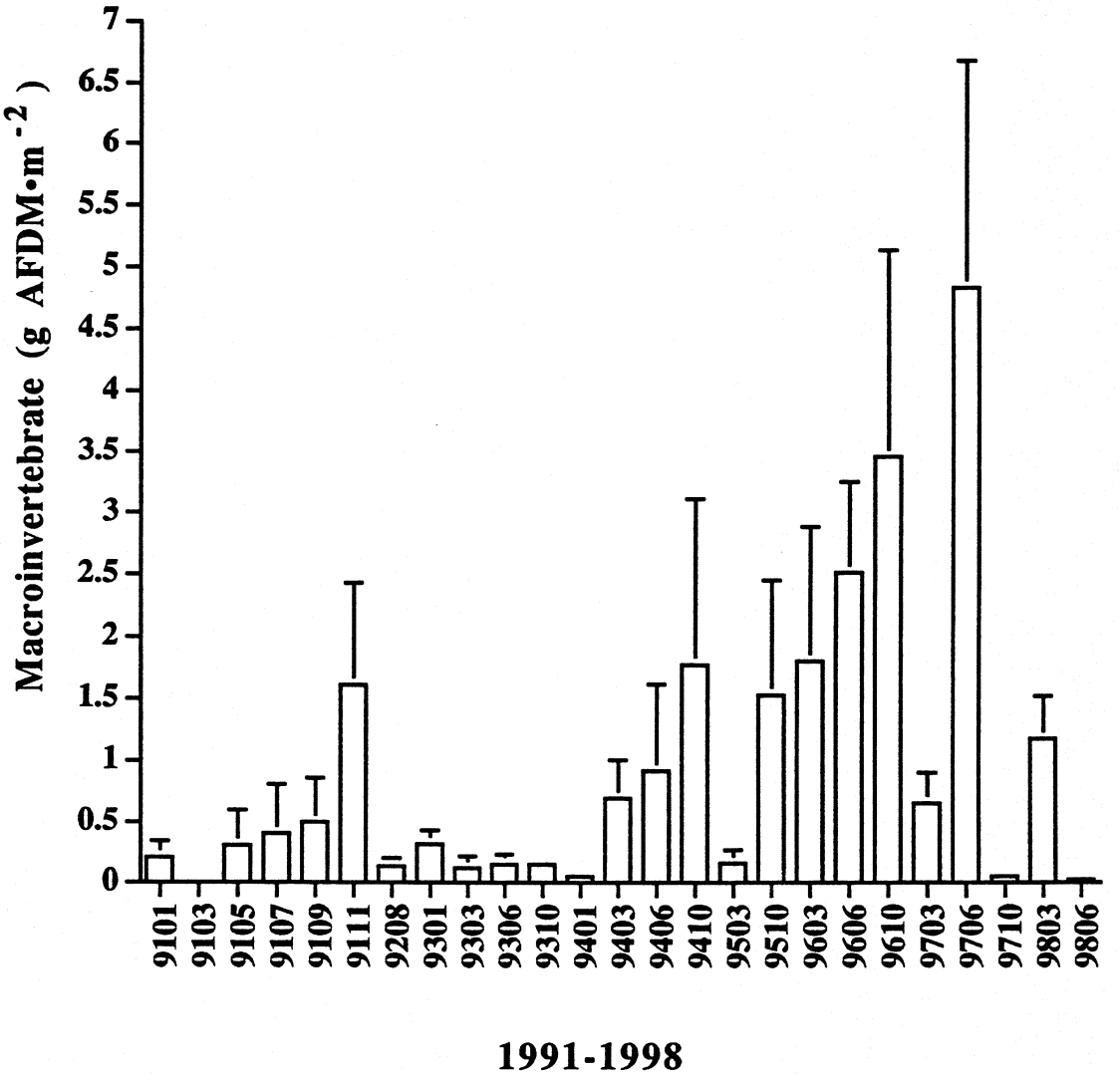

Figure 67. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at Two-Mile Wash cobble Rkm 3.1 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

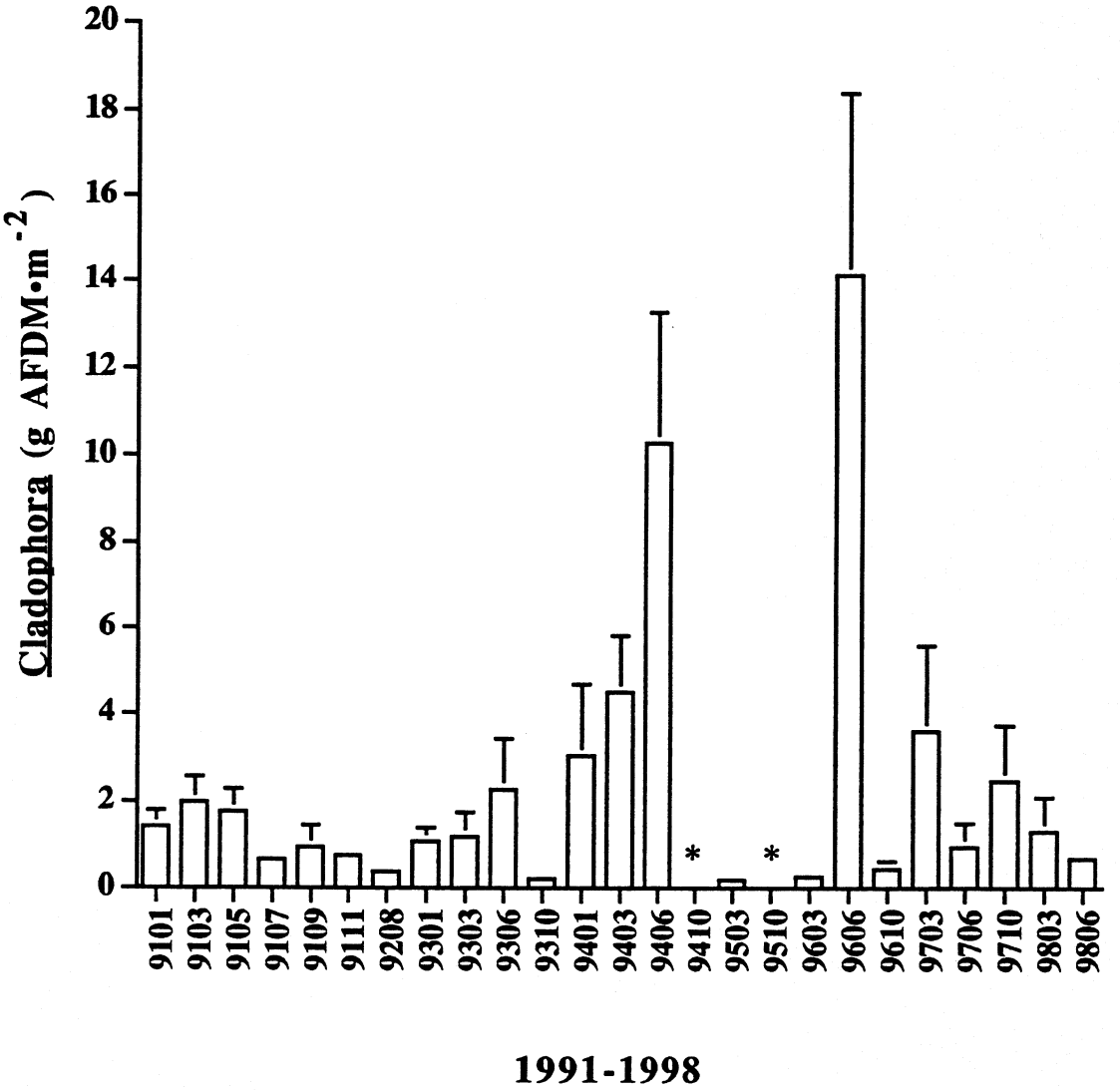

Figure 68. Cladophora biomass estimates (g AFDM•m^{-2}) at LCR Island cobble Rkm 98.6 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).
* at 9410 represents 32 g AFDM•m^{-2} (± 11).
* at 9510 represents 96 g AFDM•m^{-2} (± 38).

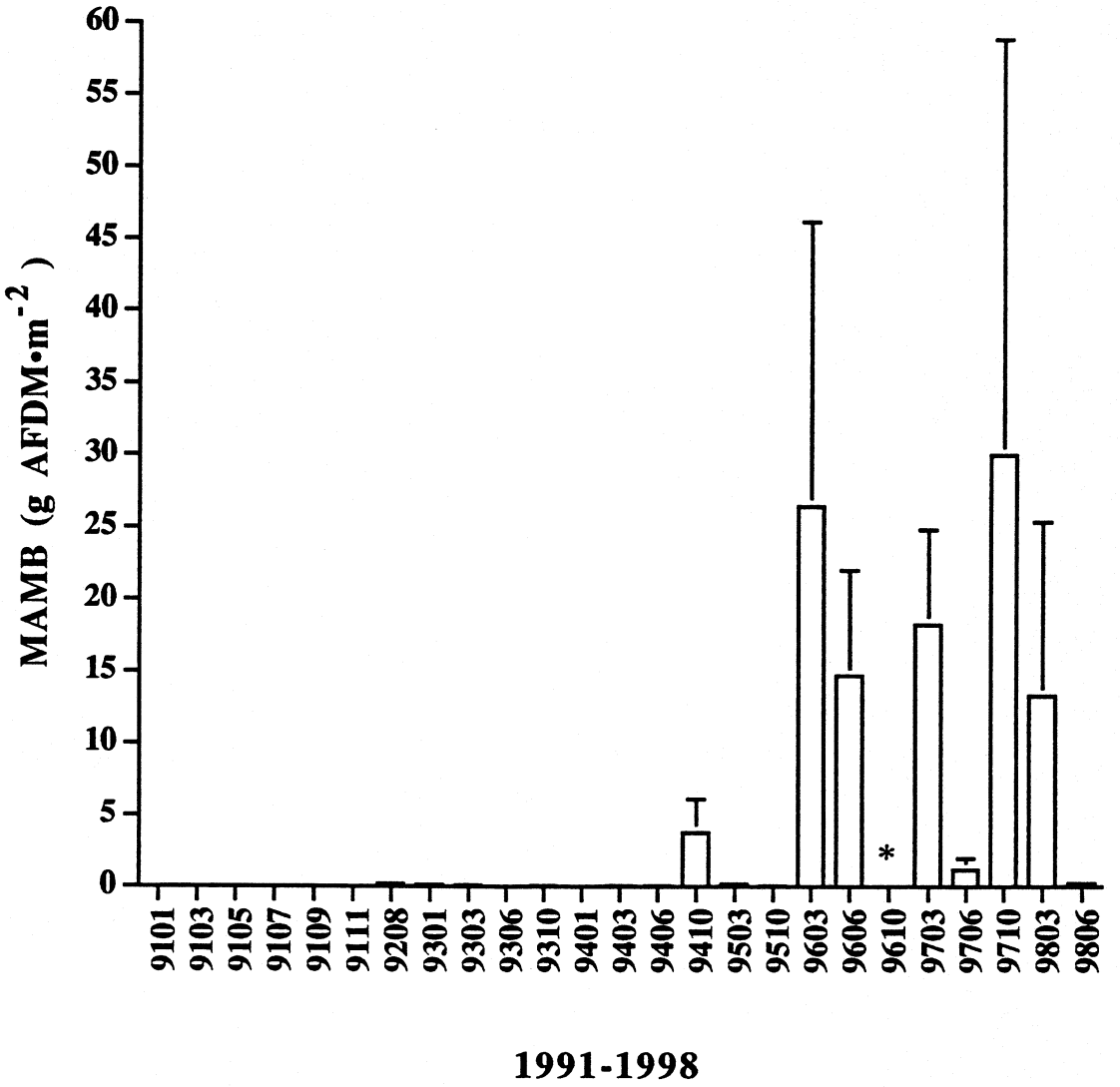

Figure 69. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at LCR Island cobble Rkm 98.6 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=6).
* represents 91 g AFDM•m^{-2} (± 26).

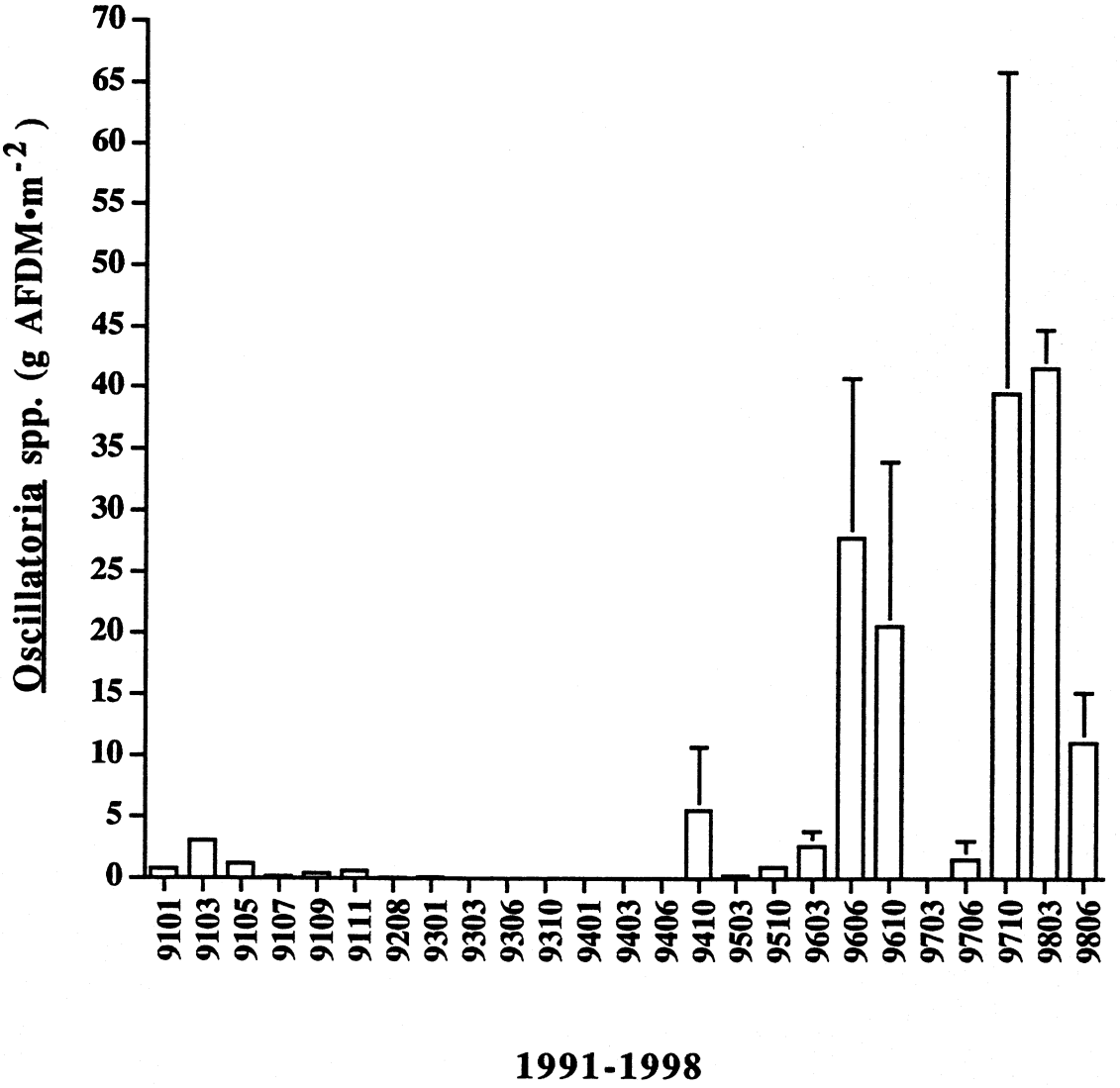

Figure 70. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at LCR Island cobble Rkm 98.6 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

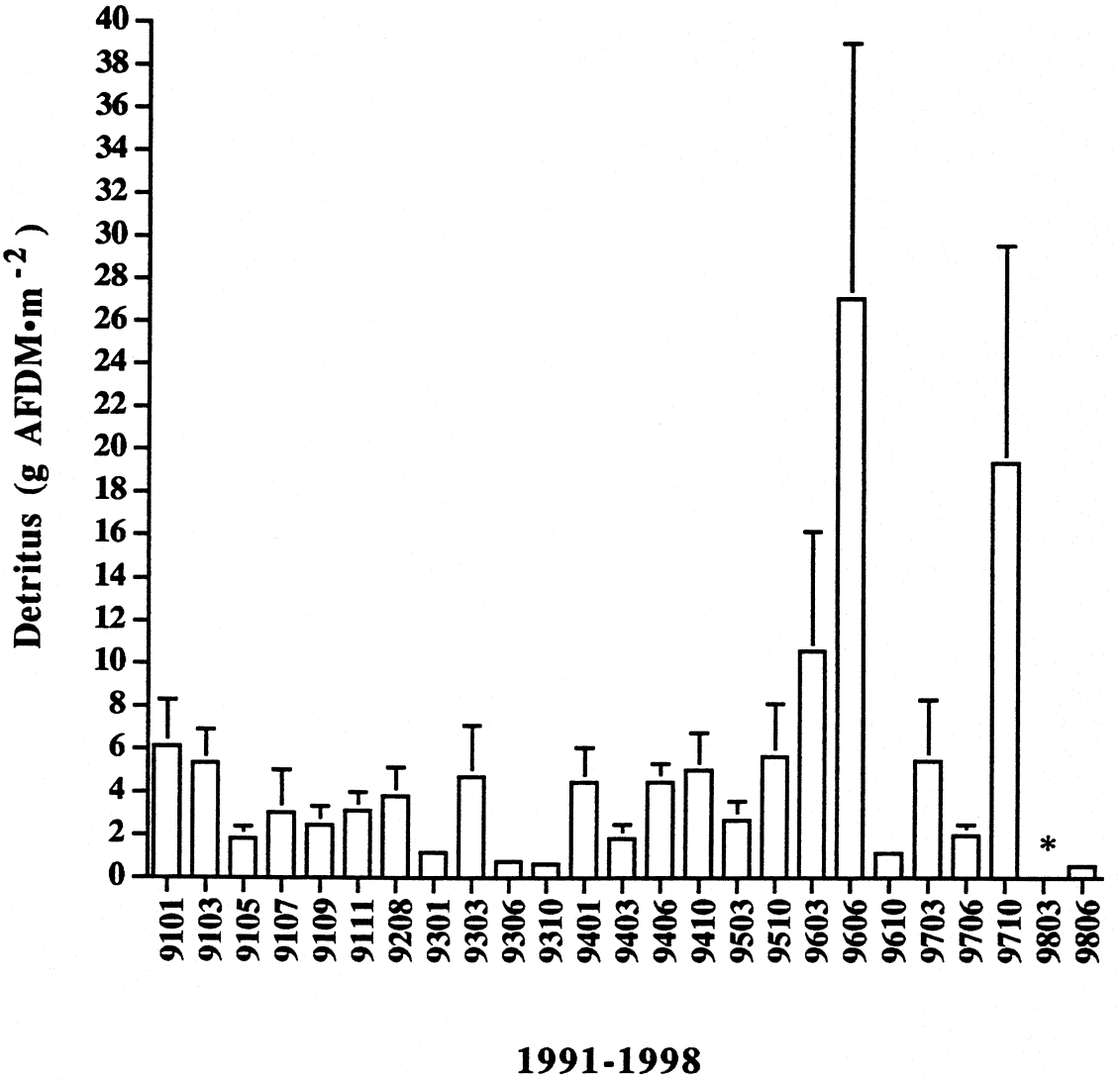

Figure 71. Detritus biomass estimates (g AFDM•m^{-2}) at LCR Island cobble Rkm 98.6 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).
* represents 39 g AFDM•m^{-2} (± 31).

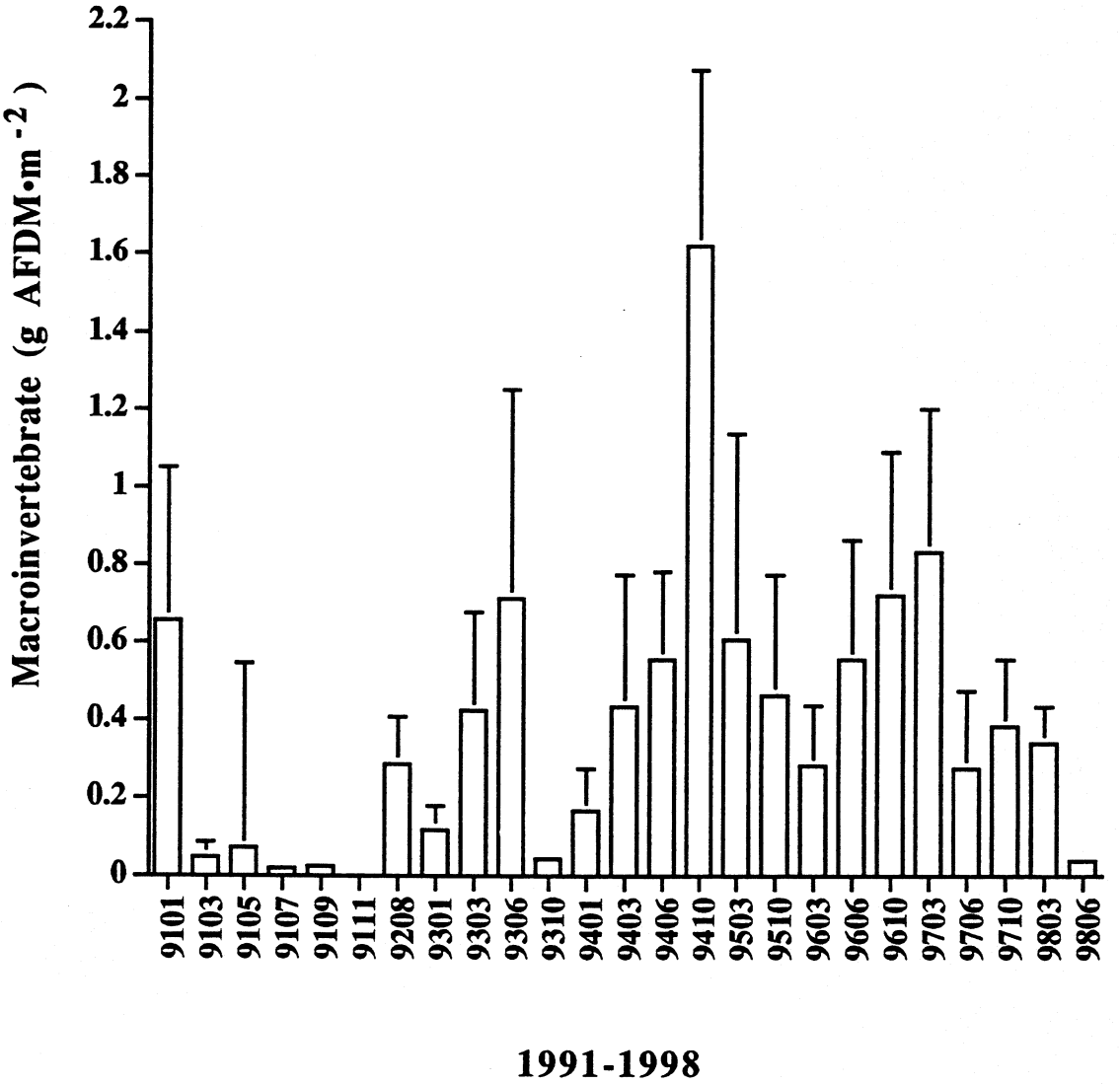

Figure 72. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at LCR Island cobble Rkm 98.6 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

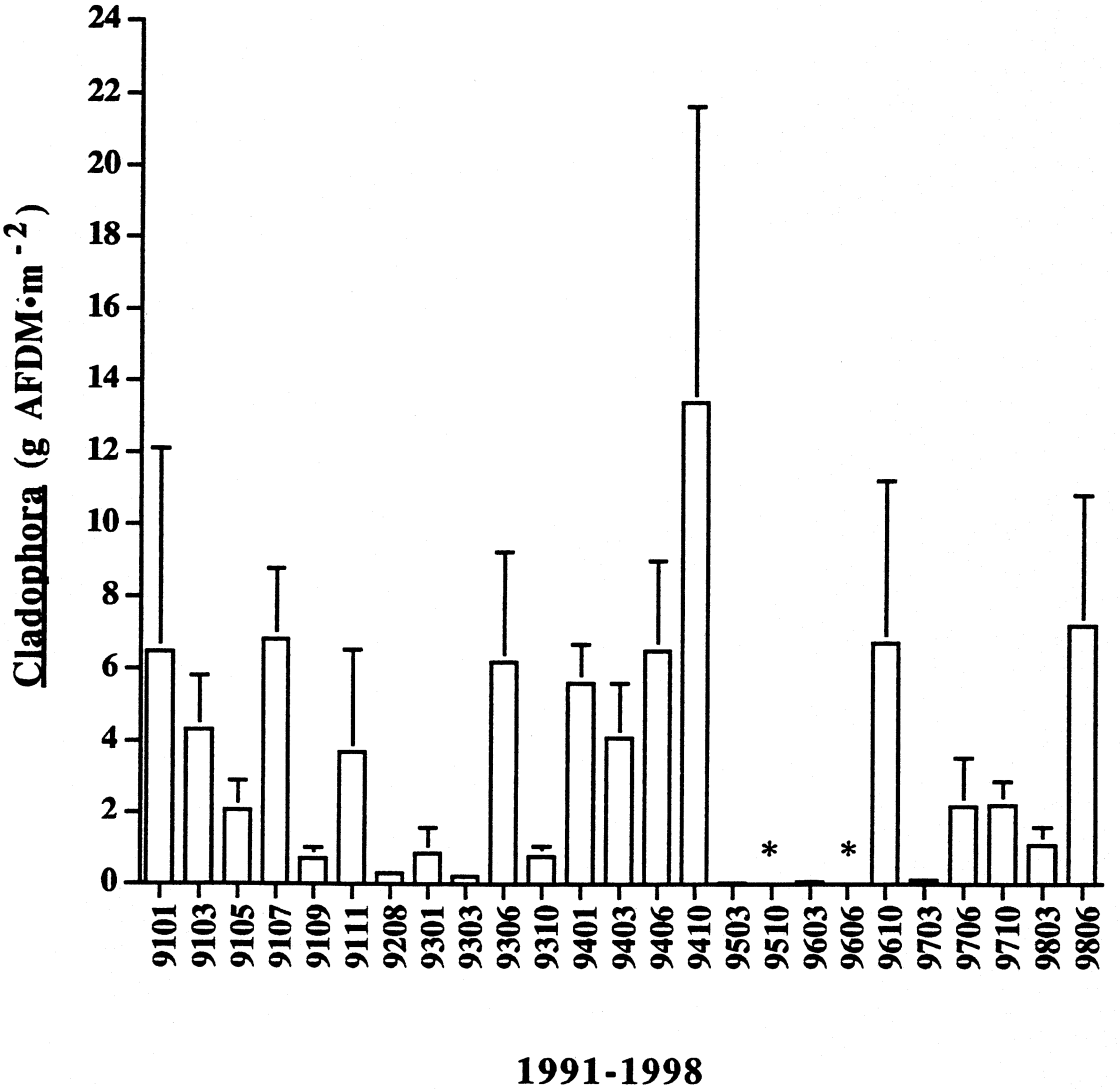

Figure 73. Cladophora biomass estimates (g AFDM•m^{-2}) at Tanner cobble Rkm 109.6 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).
* at 9510 represents 53 g AFDM•m^{-2} (± 17).
* at 9606 represents 75 g AFDM•m^{-2} (± 59).

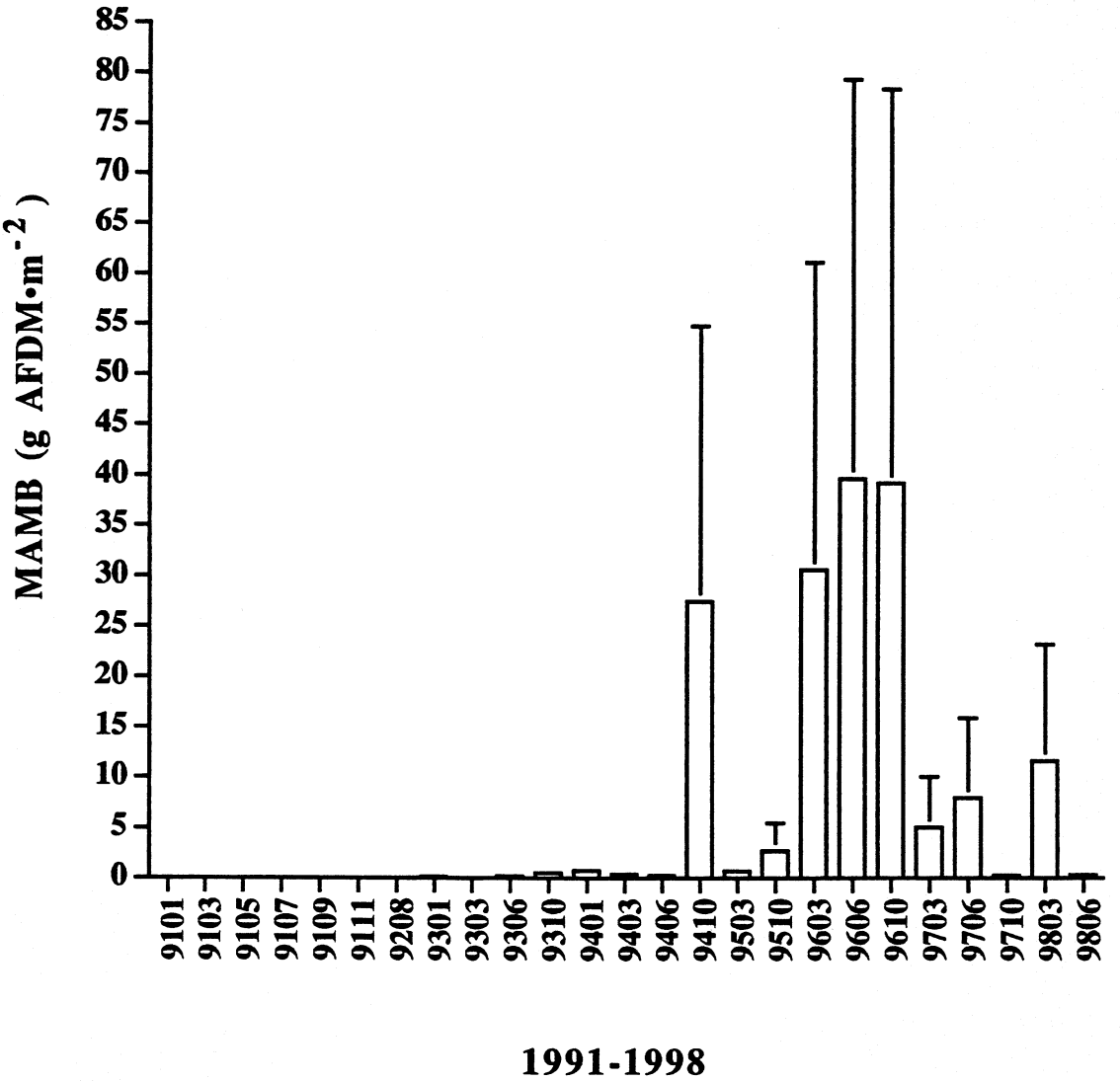

Figure 74. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at Tanner cobble Rkm 109.6 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=6).

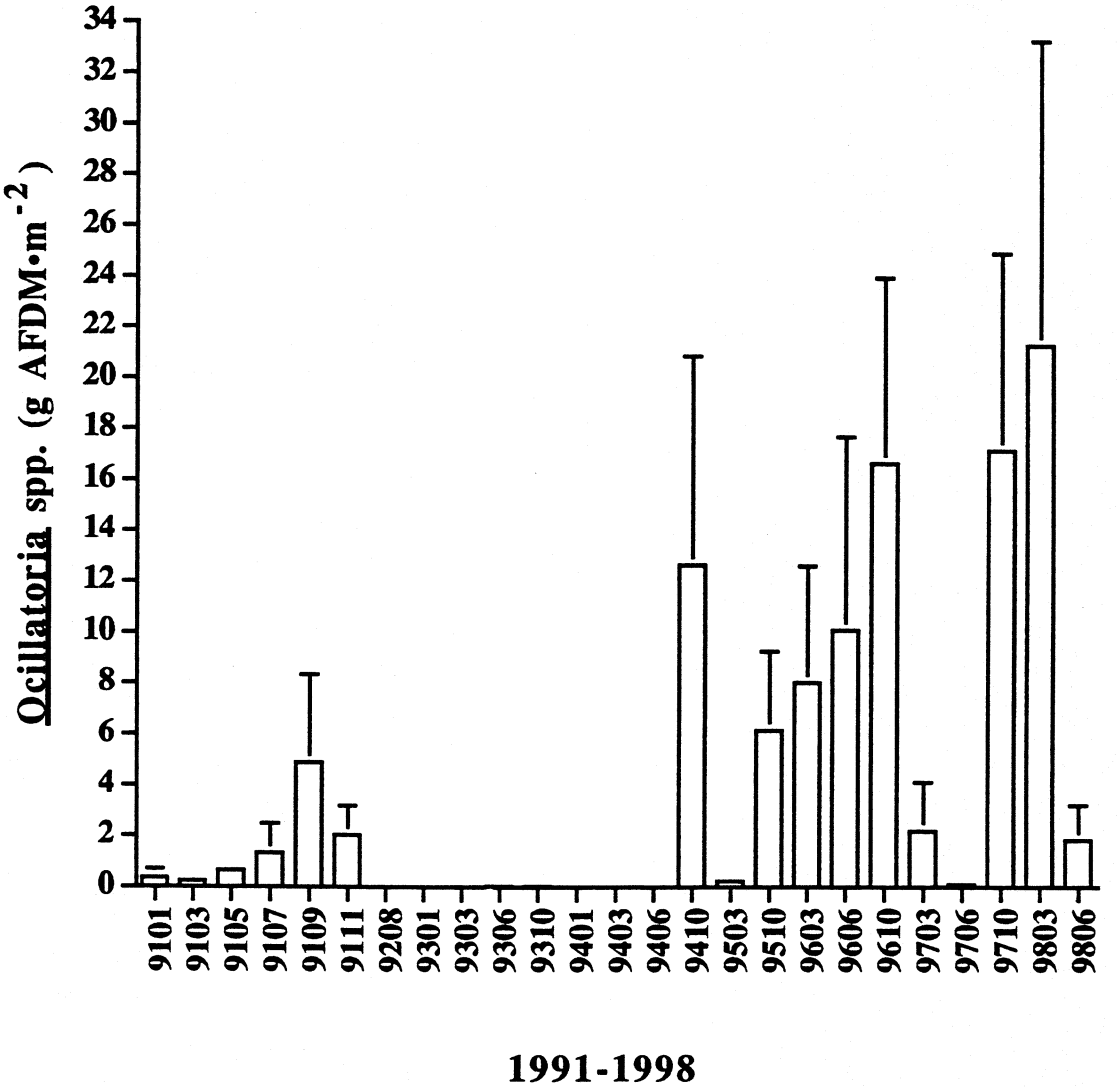

Figure 75. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at Tanner cobble Rkm 109.6 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

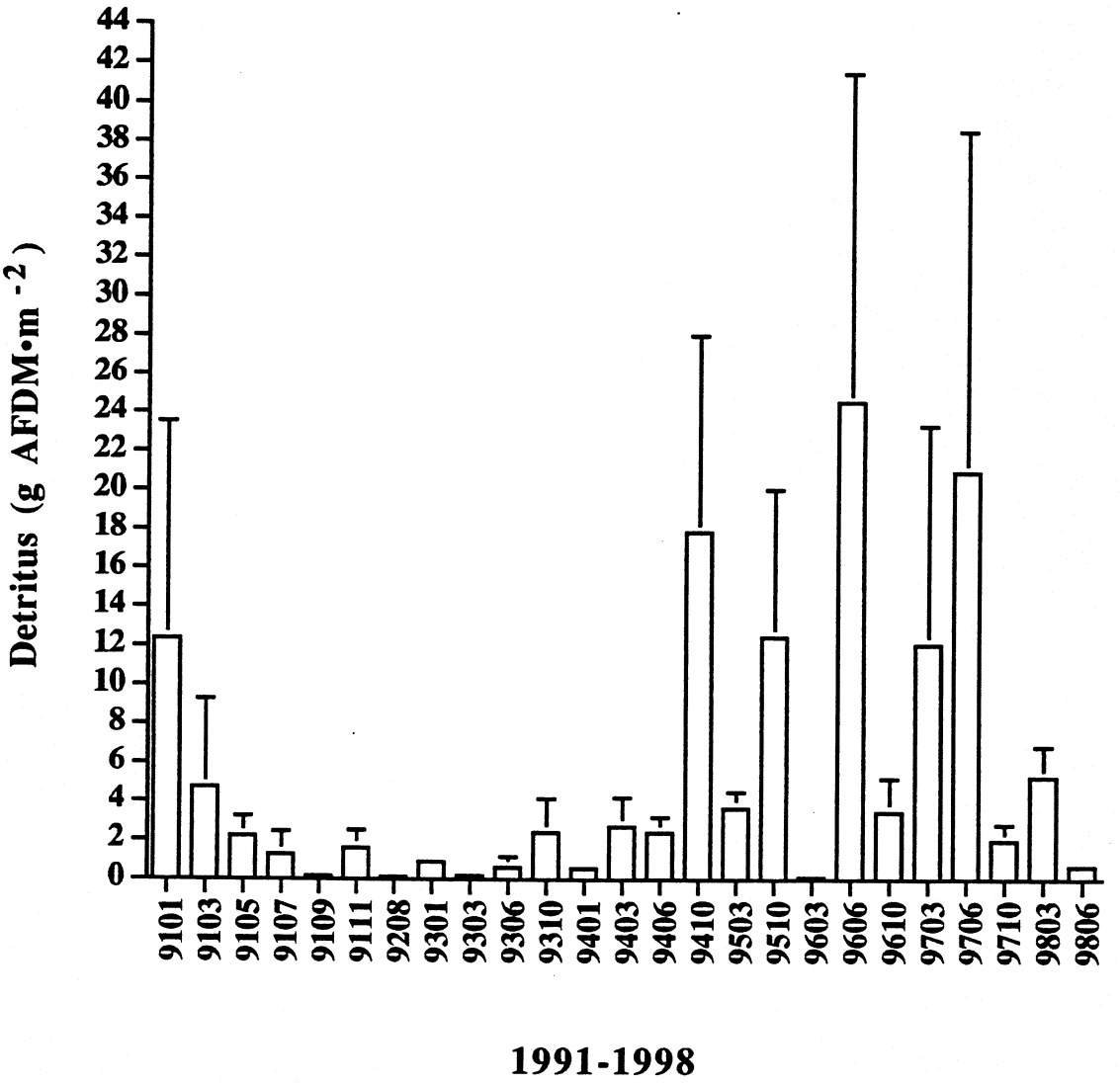

Figure 76. Detritus biomass estimates (g AFDM•m^{-2}) at Tanner cobble Rkm 109.6 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

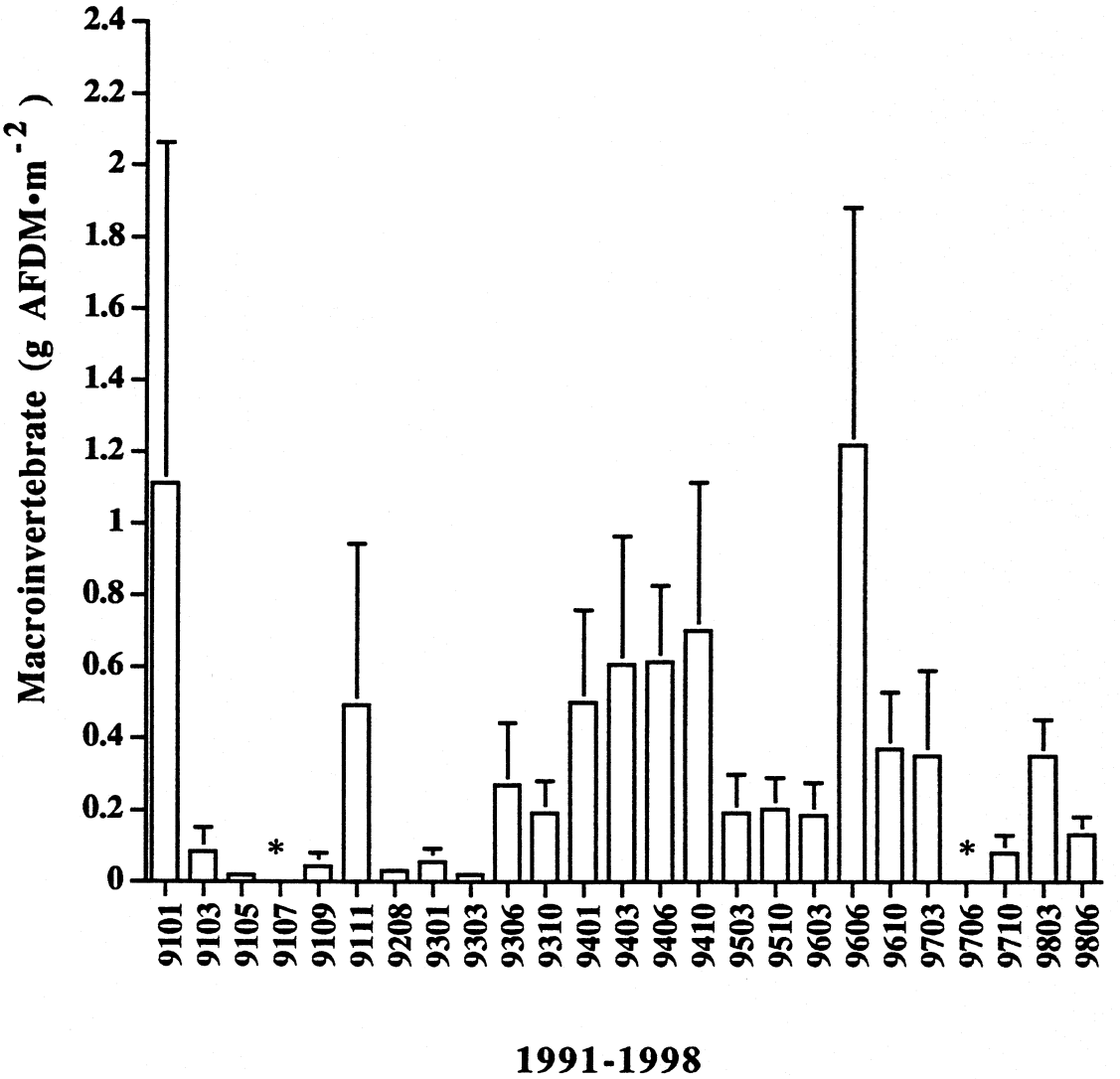

Figure 77. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at Tanner cobble Rkm 109.6 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).
* at 9107 represents 7 g AFDM•m^{-2} (± 2).
* at 9706 represents 2.5 g AFDM•m^{-2} (± 2).

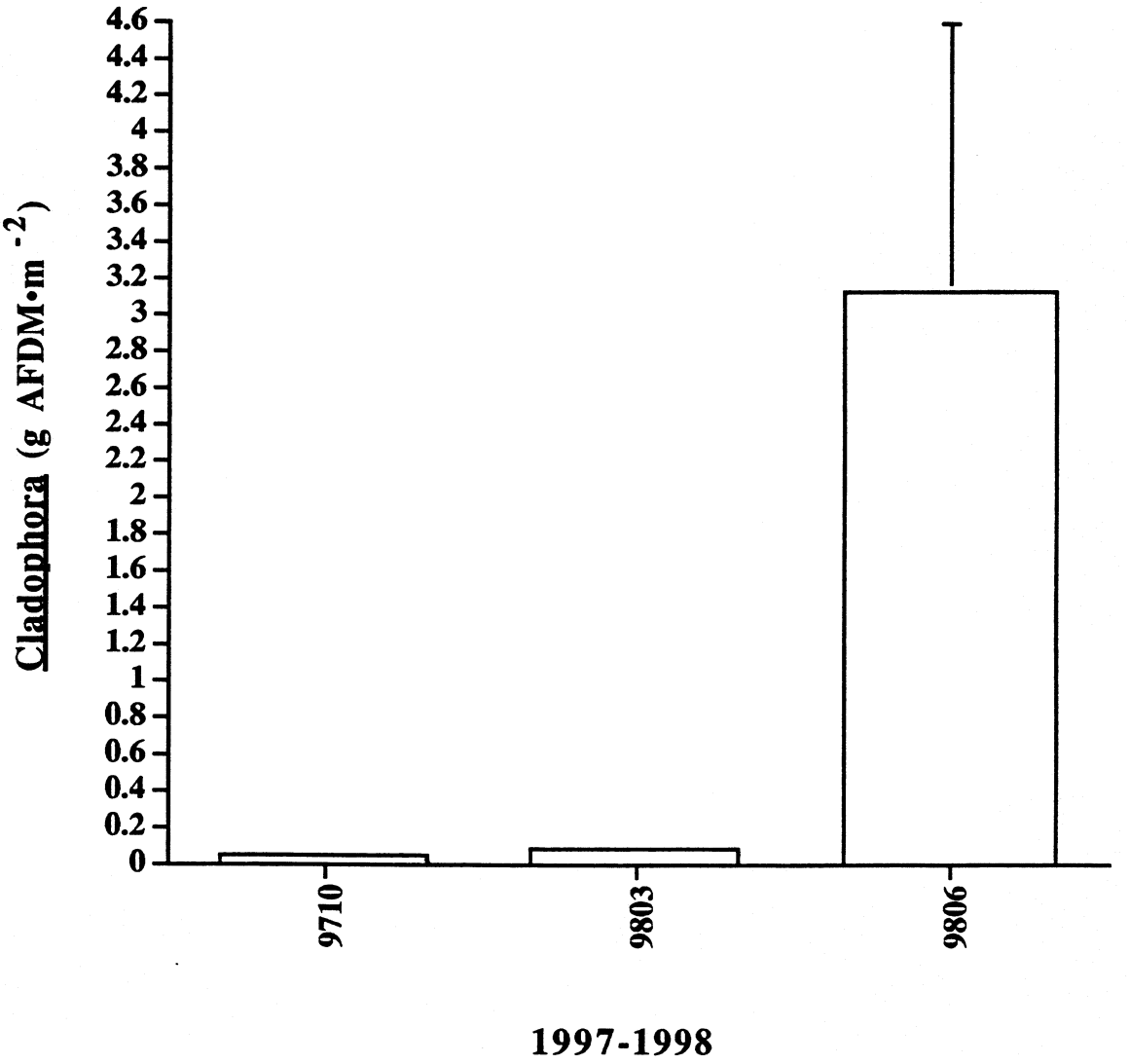

Figure 78. Cladophora biomass estimates (g AFDM•m^{-2}) at 127 Mile Rapid cobble Rkm 202.9 from October 1997 to June 1998. Error bars represent (± 1 SE, n=6).

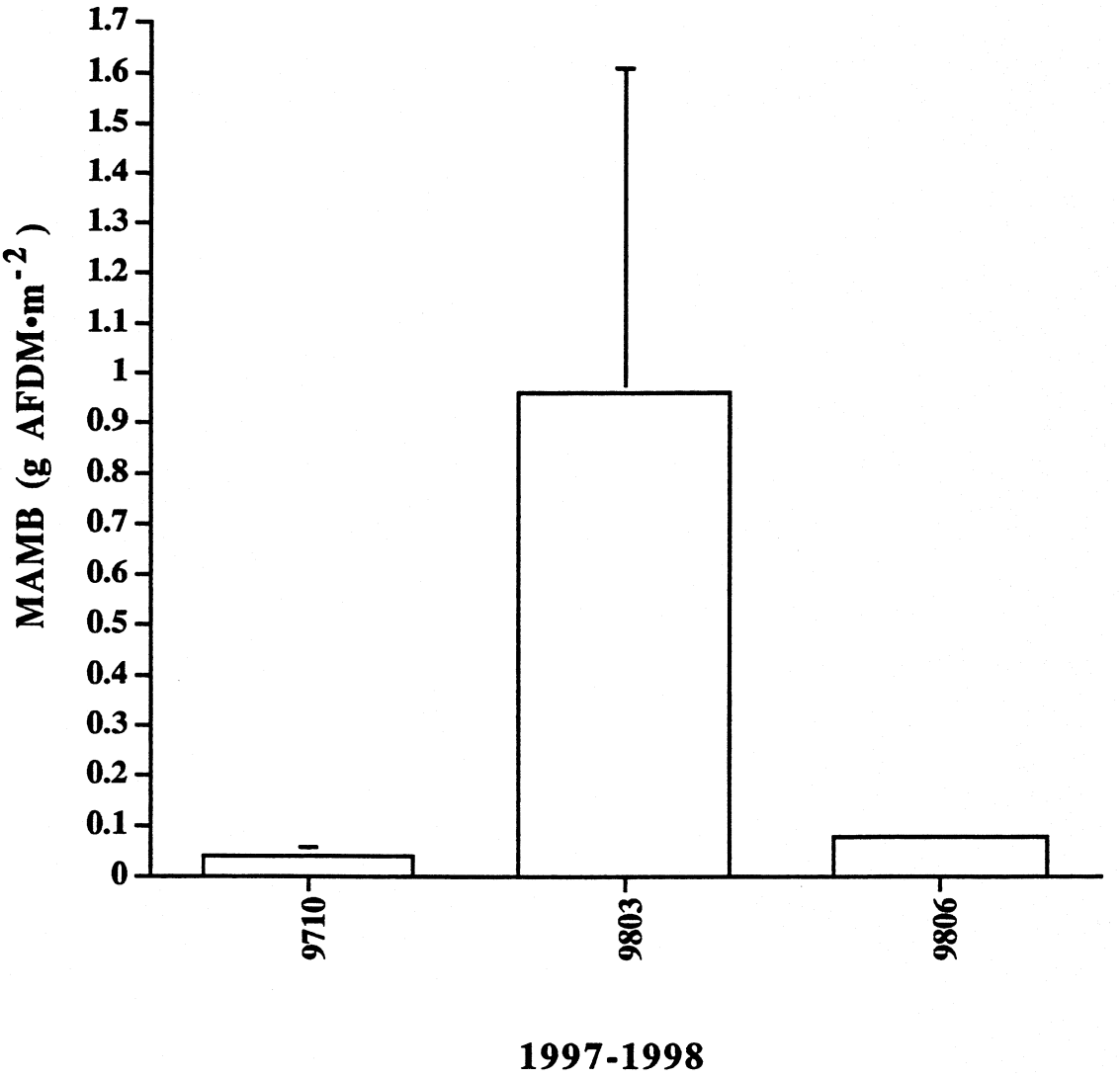

Figure 79. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at 127 Mile Rapid cobble Rkm 202.9 from October 1997 to June 1998. Error bars represent (± 1 SE, n=6).

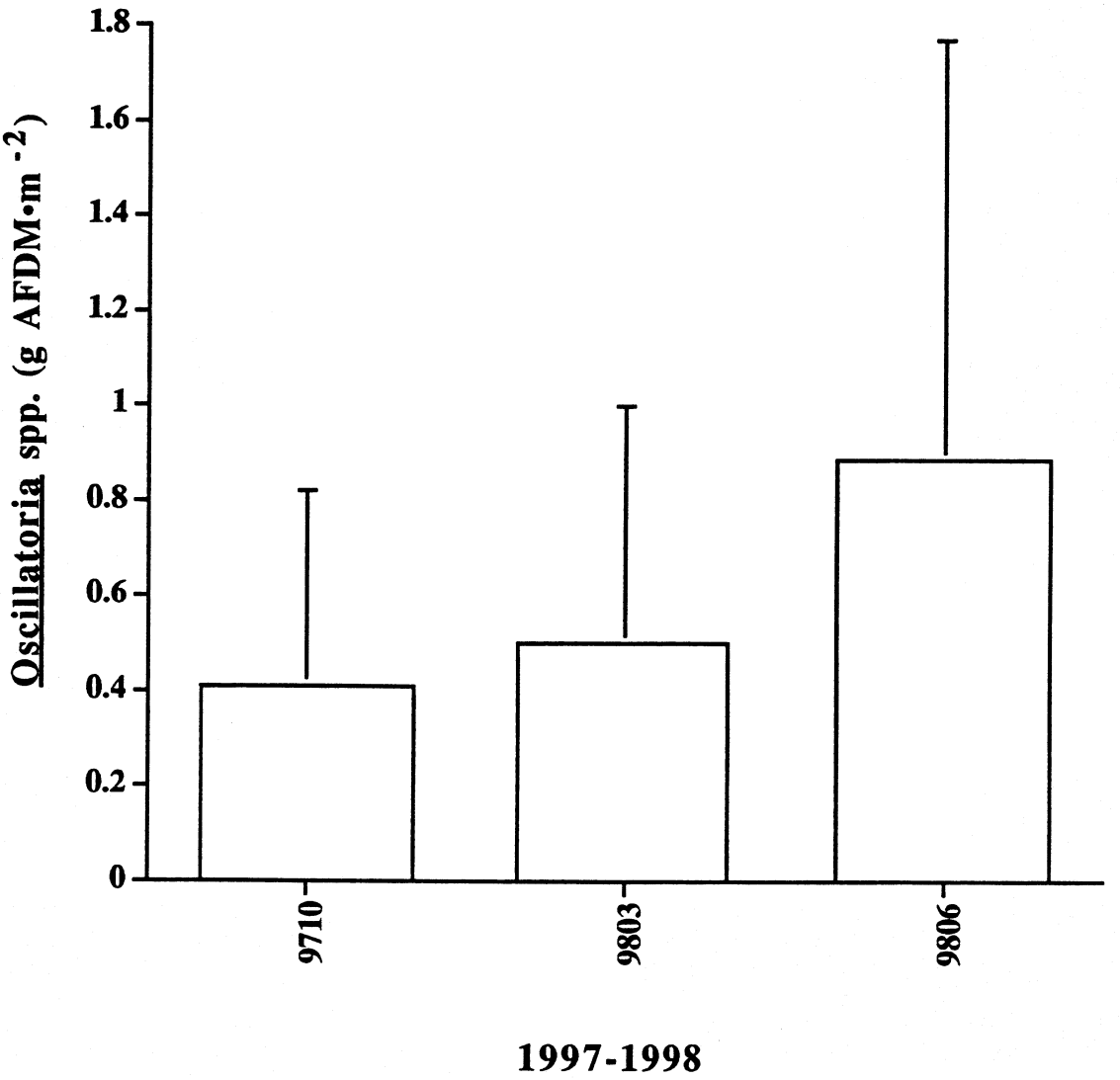

Figure 80. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at 127 Mile Rapid cobble Rkm 202.9 from October 1997 to June 1998. Error bars represent (± 1 SE, n=6).

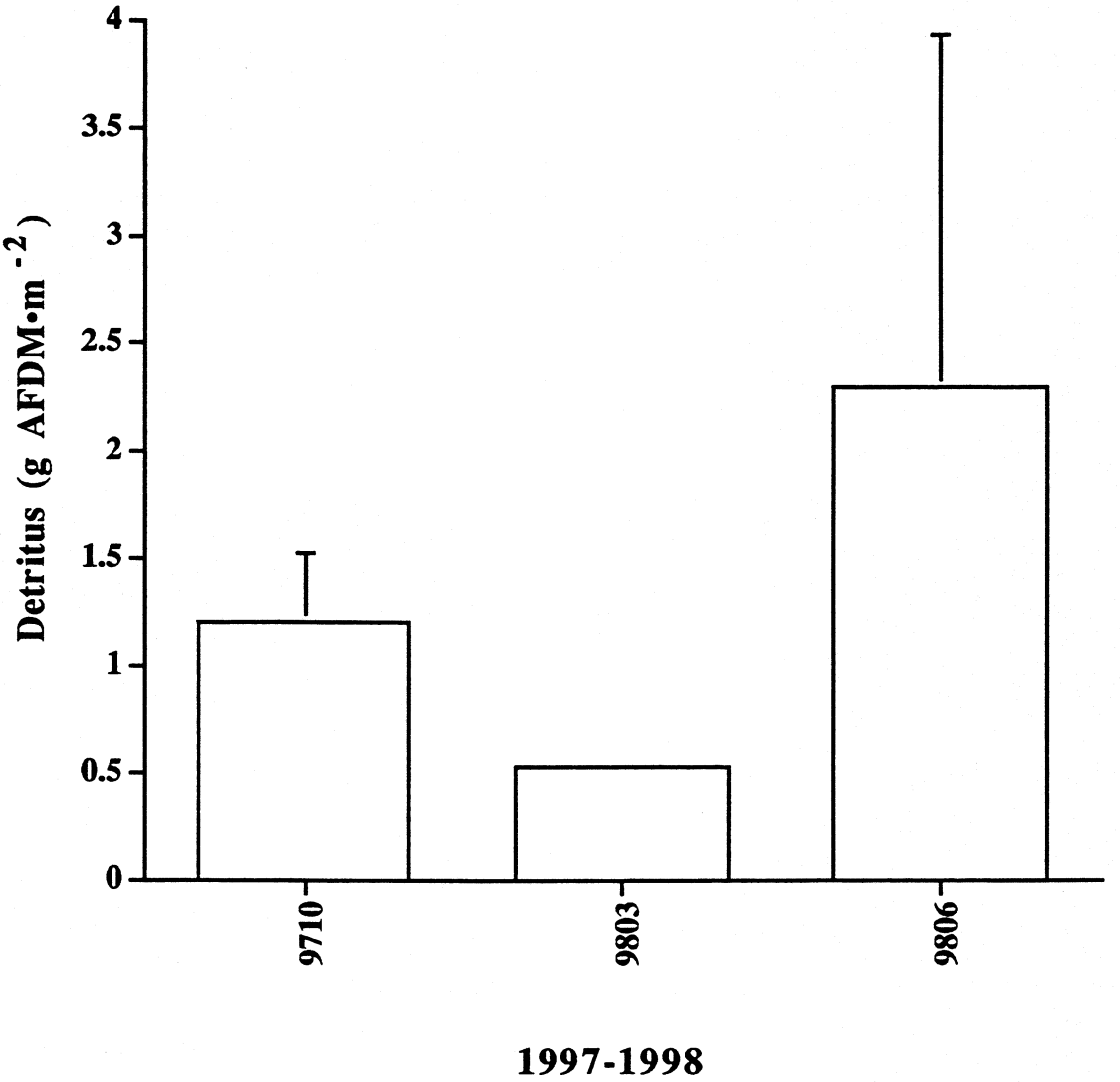

Figure 81. Detritus biomass estimates (g AFDM•m^{-2}) at 127 Mile Rapid cobble Rkm 202.9 from October 1997 to June 1998. Error bars represent (± 1 SE, n=6).

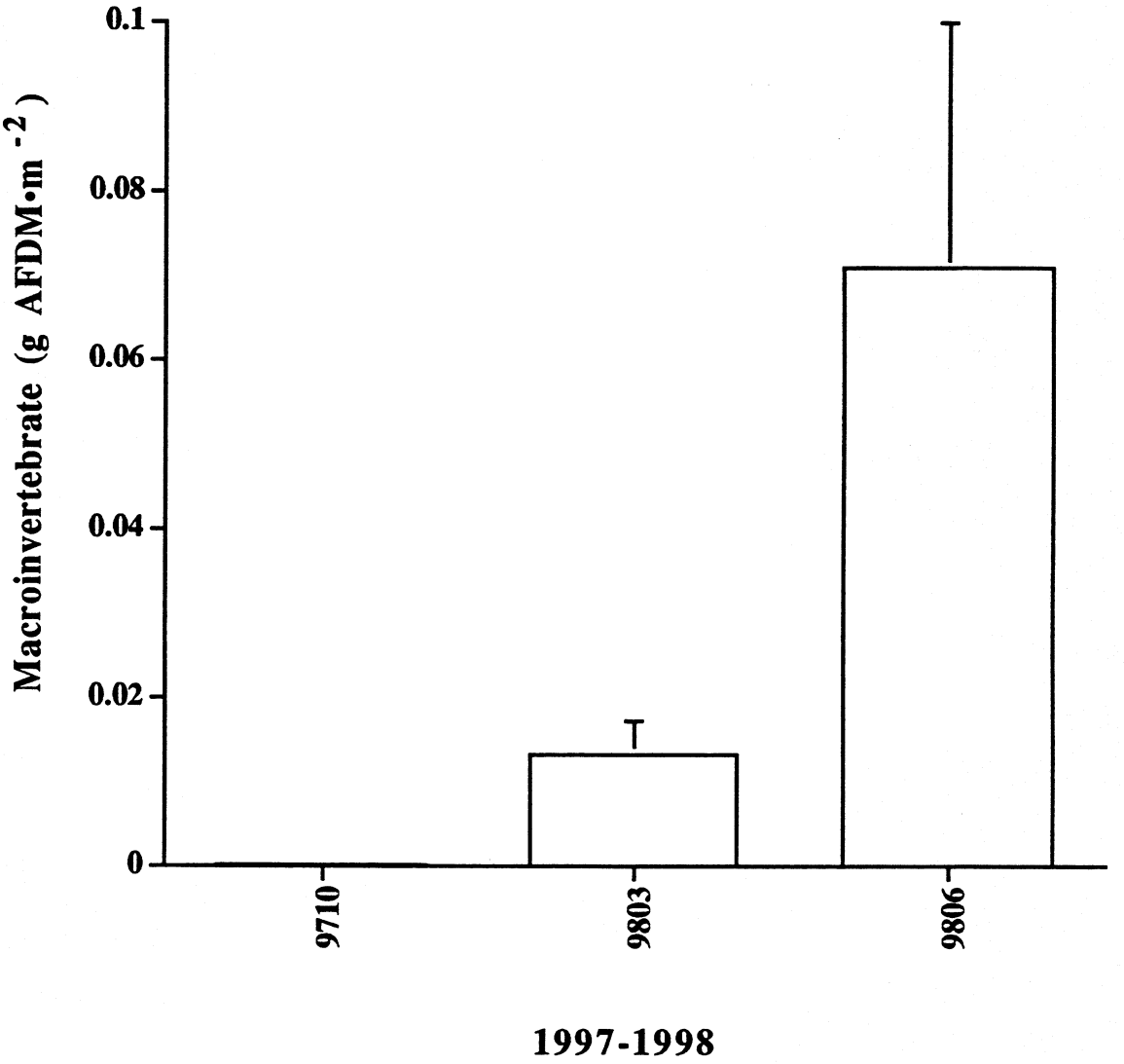

Figure 82. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at 127 Mile Rapid cobble Rkm 202.9 from October 1997 to June 1998. Error bars represent (± 1 SE, n=6).

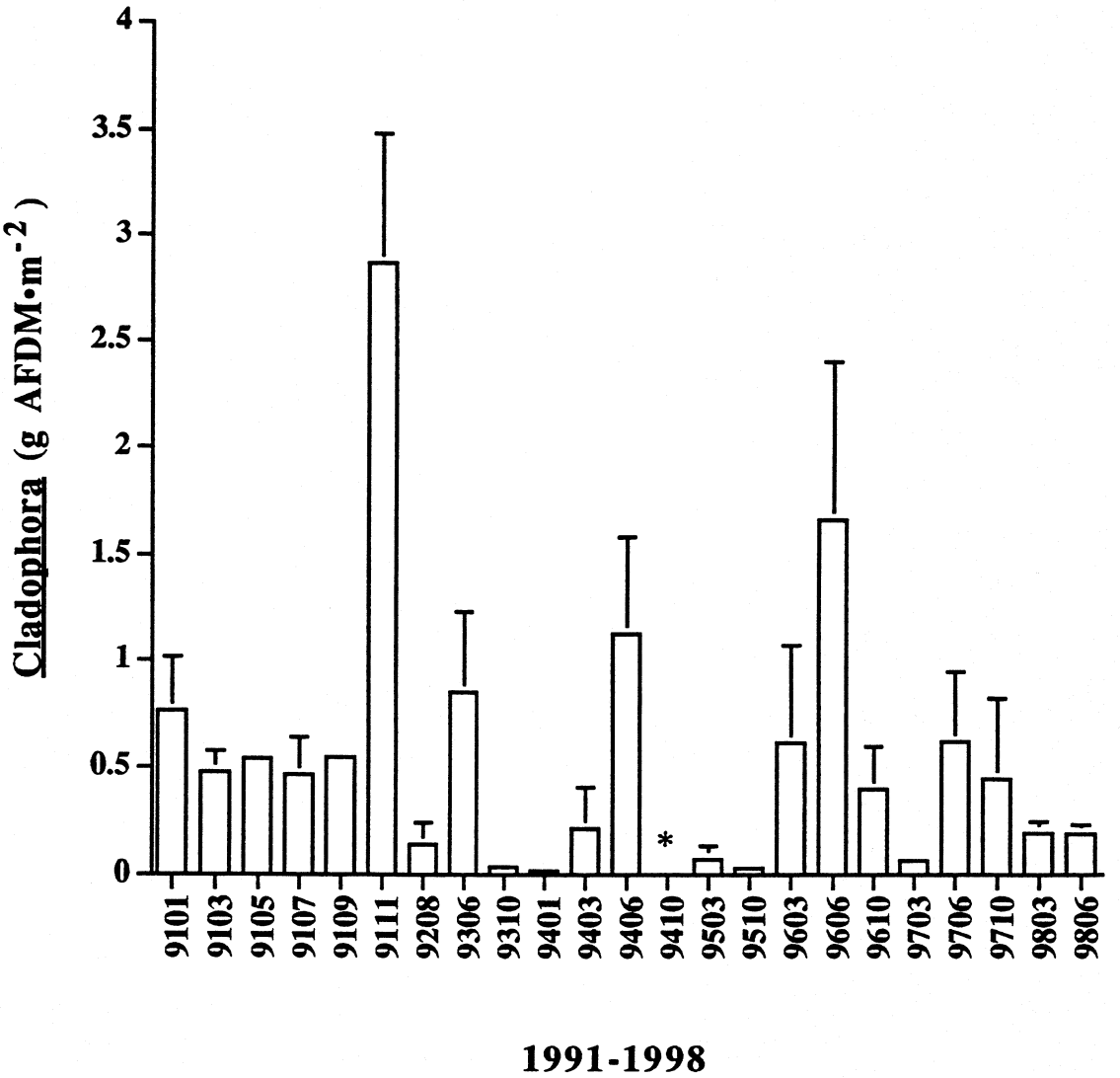

Figure 83. Cladophora biomass estimates (g AFDM•m^{-2}) at 205 Mile rapid Rkm 328.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).
* represents 8 g AFDM•m^{-2} (± 4).

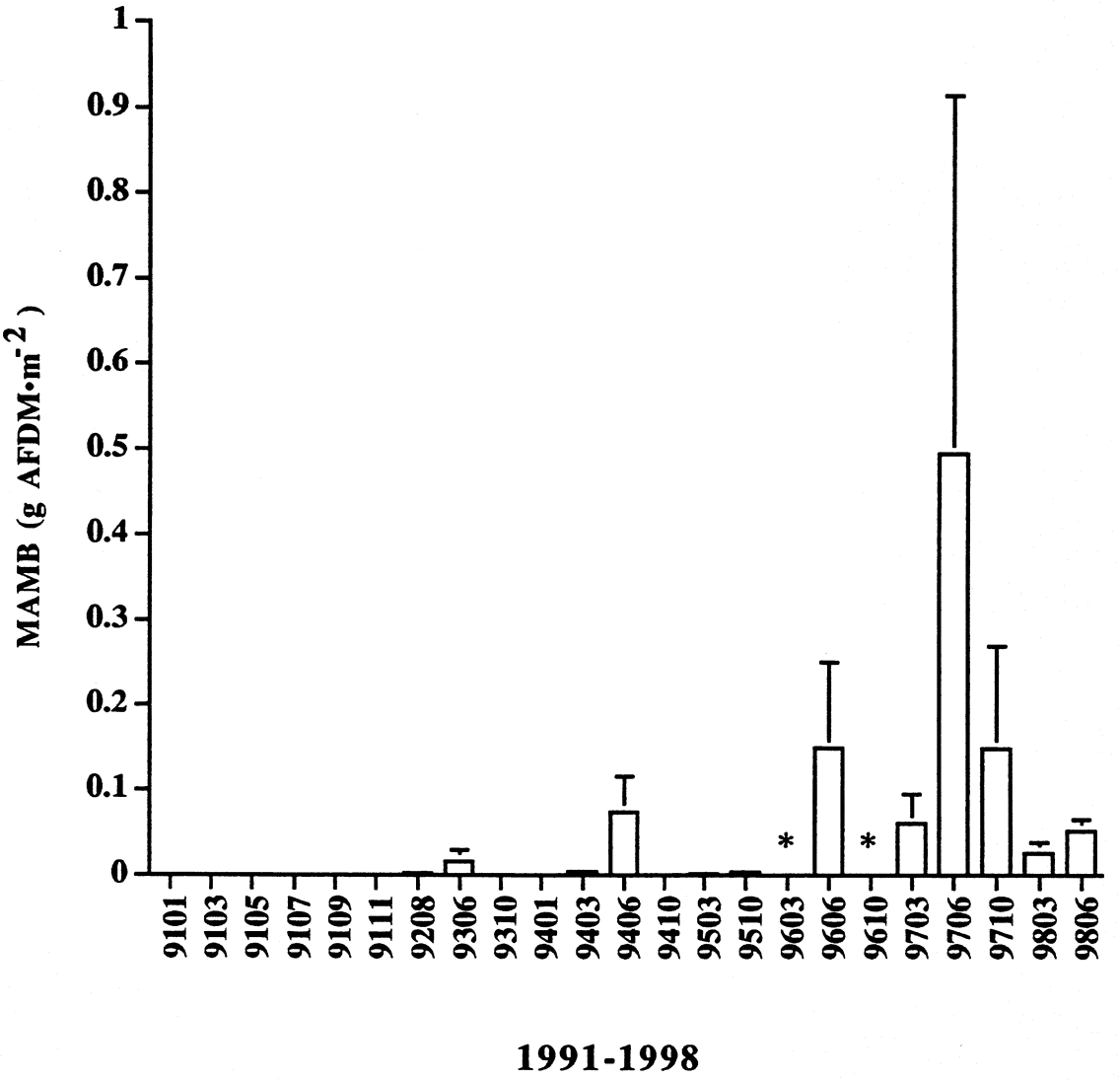

Figure 84. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at 205 Mile rapid Rkm 328.8 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=6).
* at 9603 represents 8 g AFDM•m (± 7).
* at 9610 represents 4 g AFDM•m (± 2).

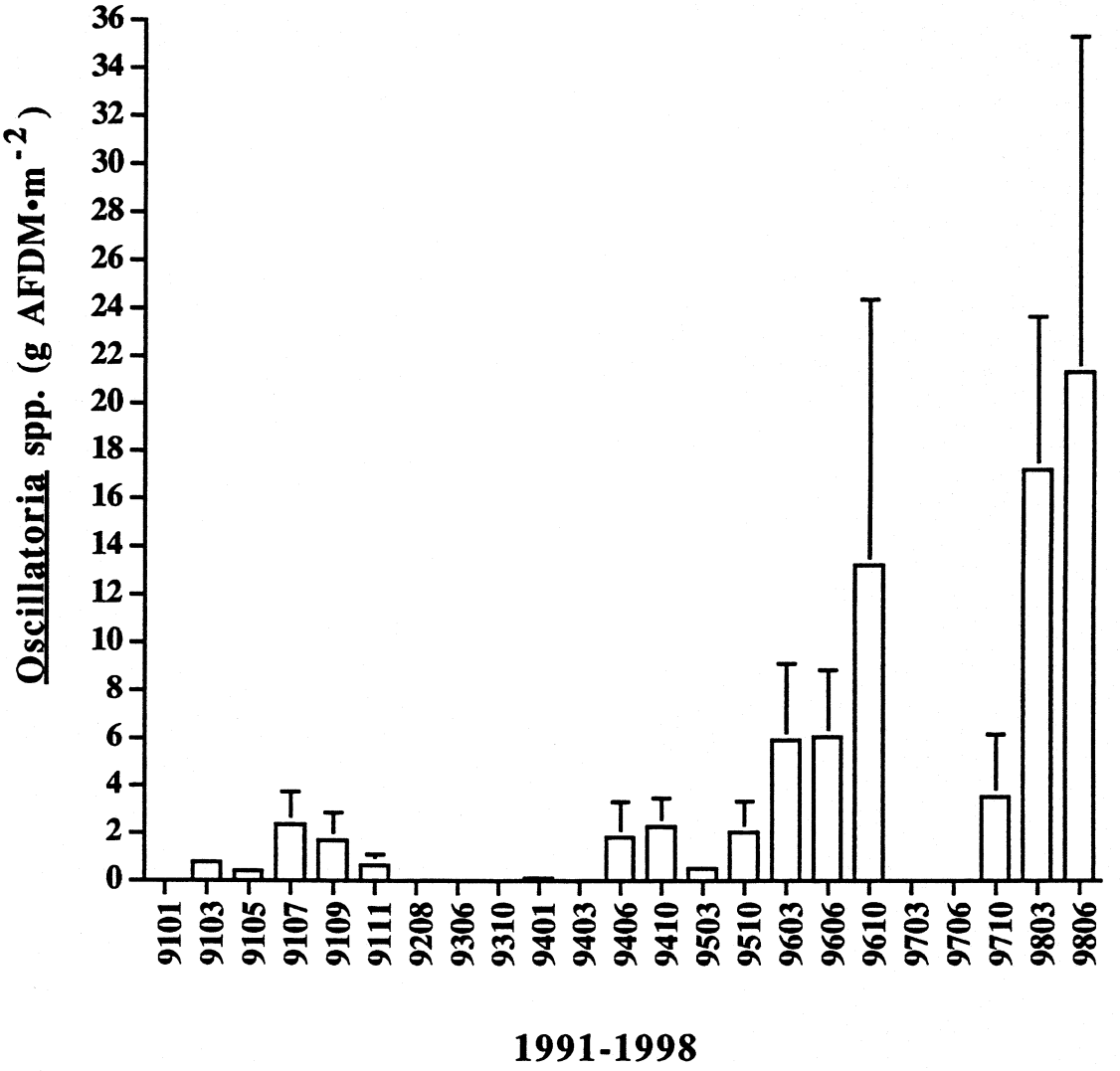

Figure 85. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at 205 Mile rapid Rkm 328.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

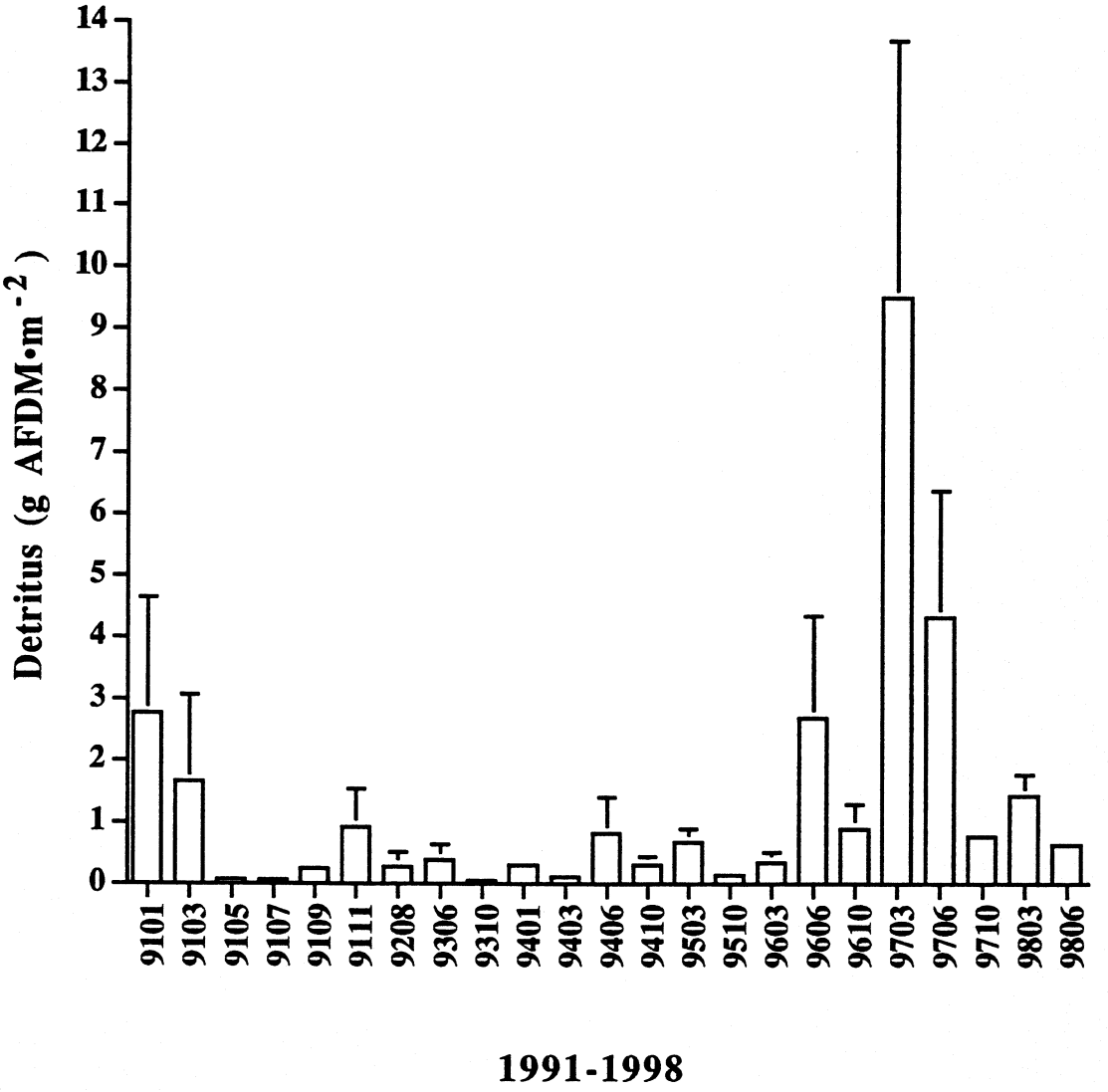

Figure 86. Detritus biomass estimates (g AFDM•m^{-2}) at 205 Mile rapid Rkm 328.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

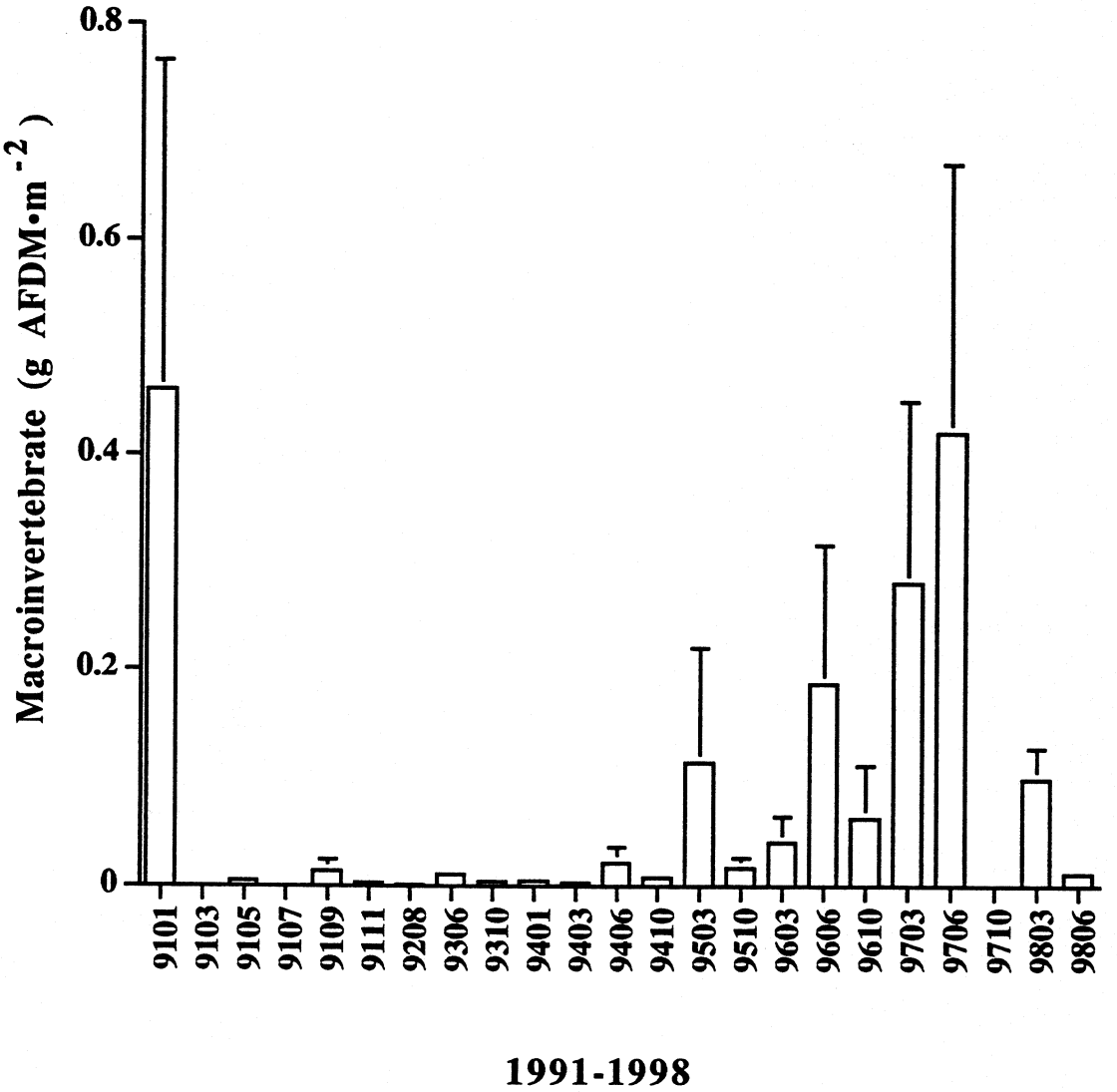

Figure 87. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at 205 Mile rapid Rkm 328.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).

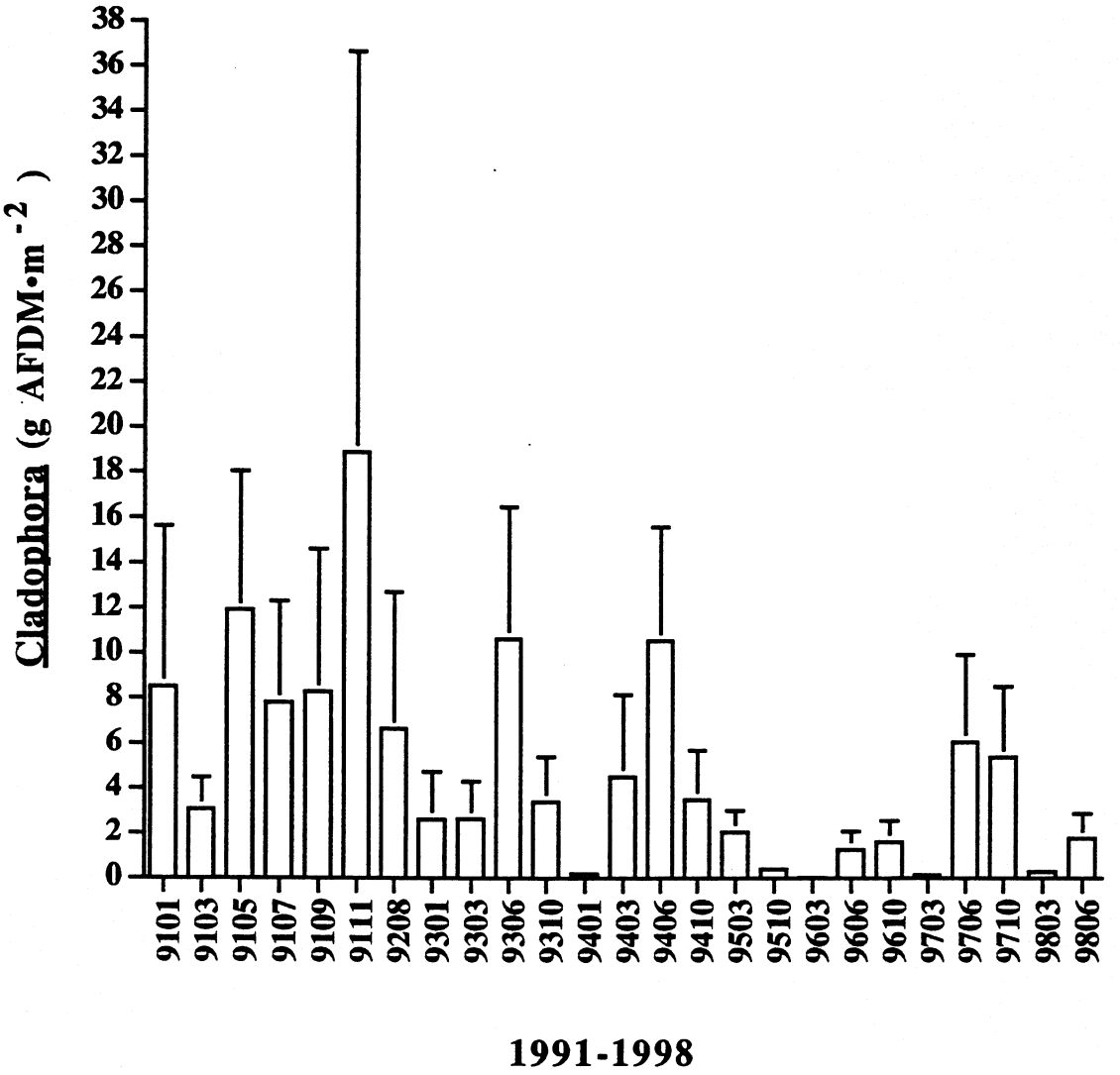

Figure 88. Cladophora biomass estimates (g AFDM•m^{-2}) at Lees Ferry pool Rkm 0.0 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

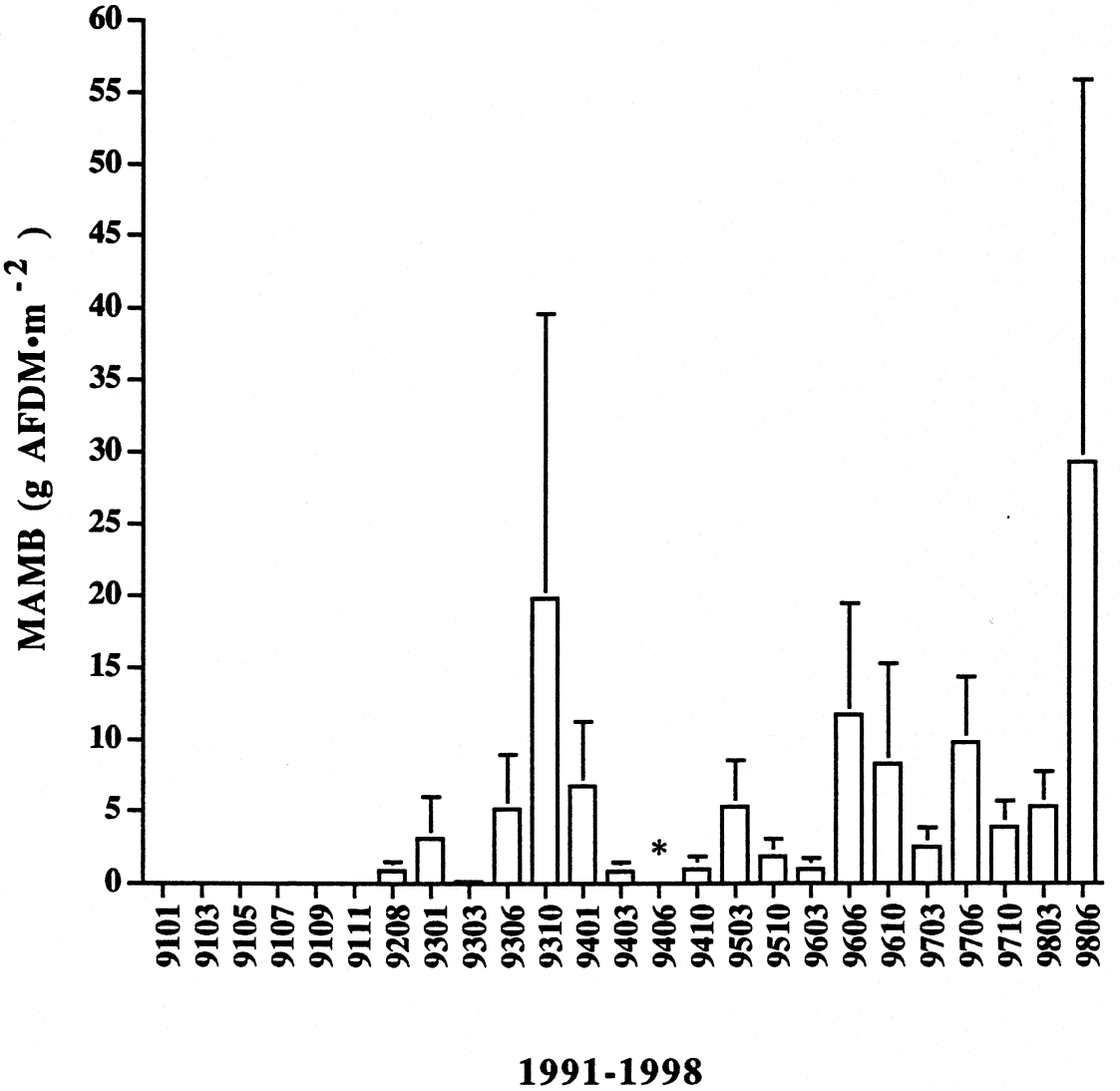

Figure 89. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at Lees Ferry pool Rkm 0.0 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=12).
* represents 50 g AFDM•m^{-2} (± 39).

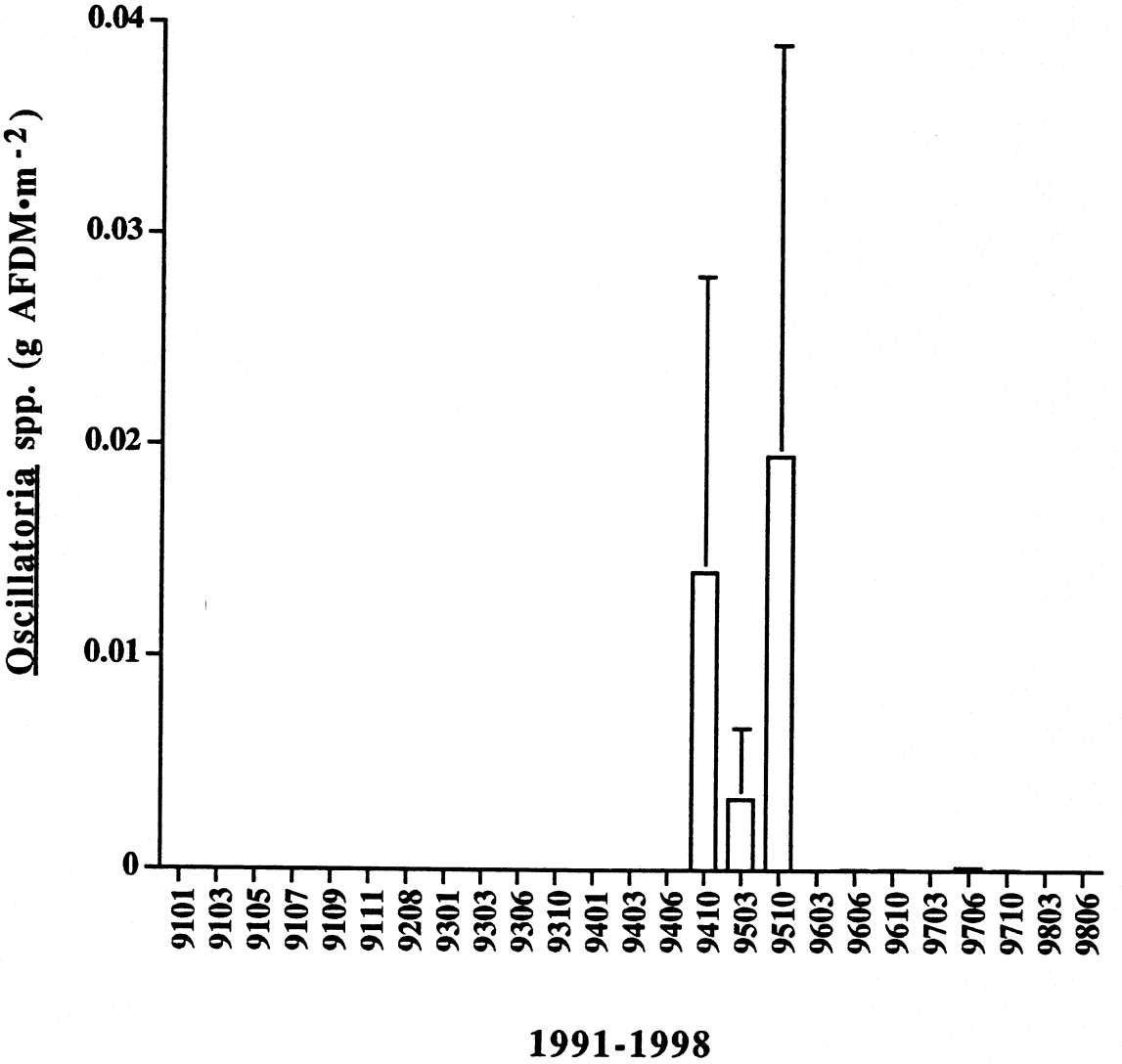

Figure 90. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at Lees Ferry pool Rkm 0.0 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

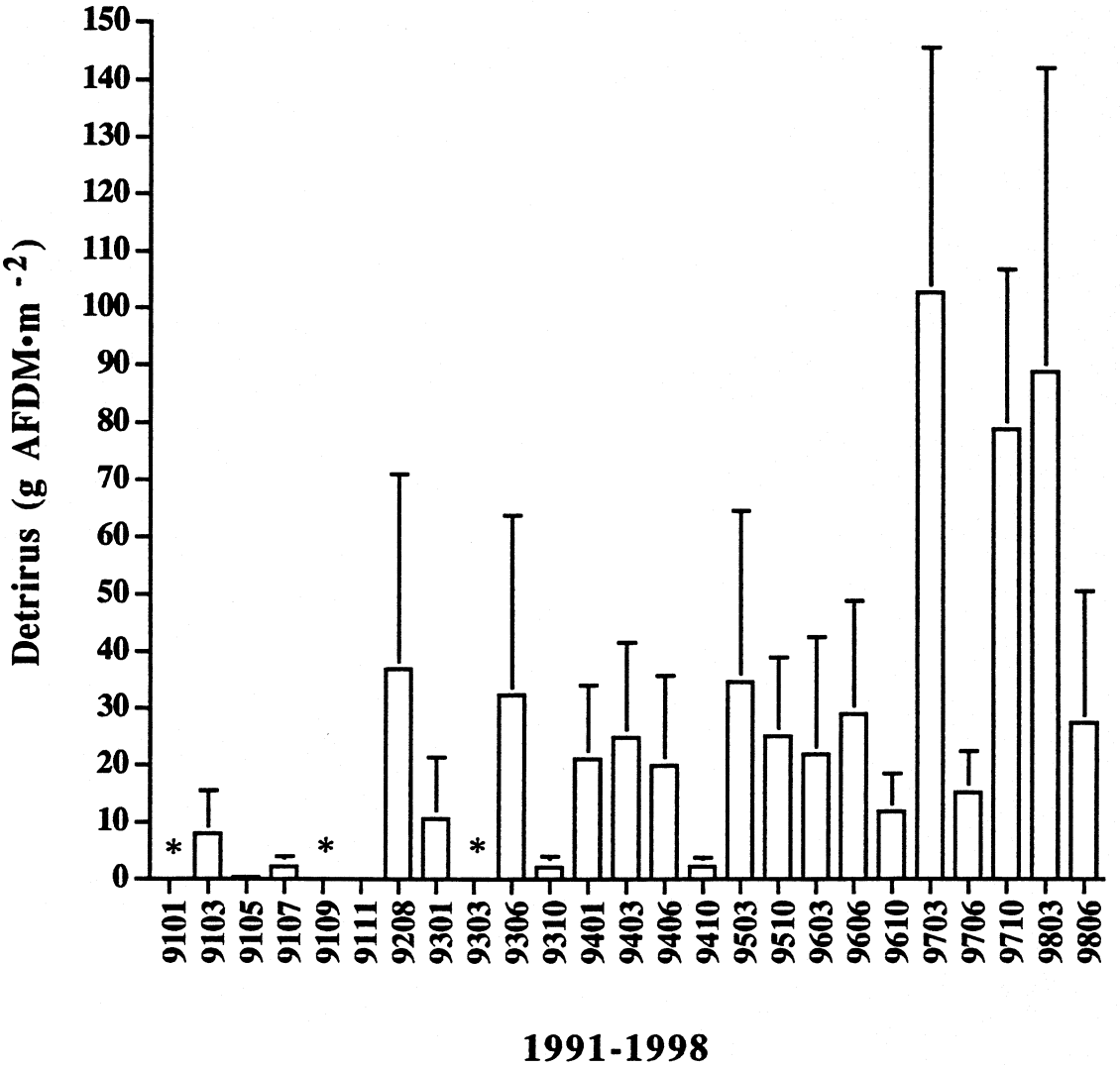

Figure 91. Detritus biomass estimates (g AFDM•m^{-2}) at Lees Ferry pool Rkm 0.0 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).
* at 9101 represents 1143 g AFDM•m^{-2} (± 1143 SE).
* at 9109 represents 523 g AFDM•m^{-2} (± 521 SE).
* at 9303 represents 213 g AFDM•m^{-2} (± 175 SE).

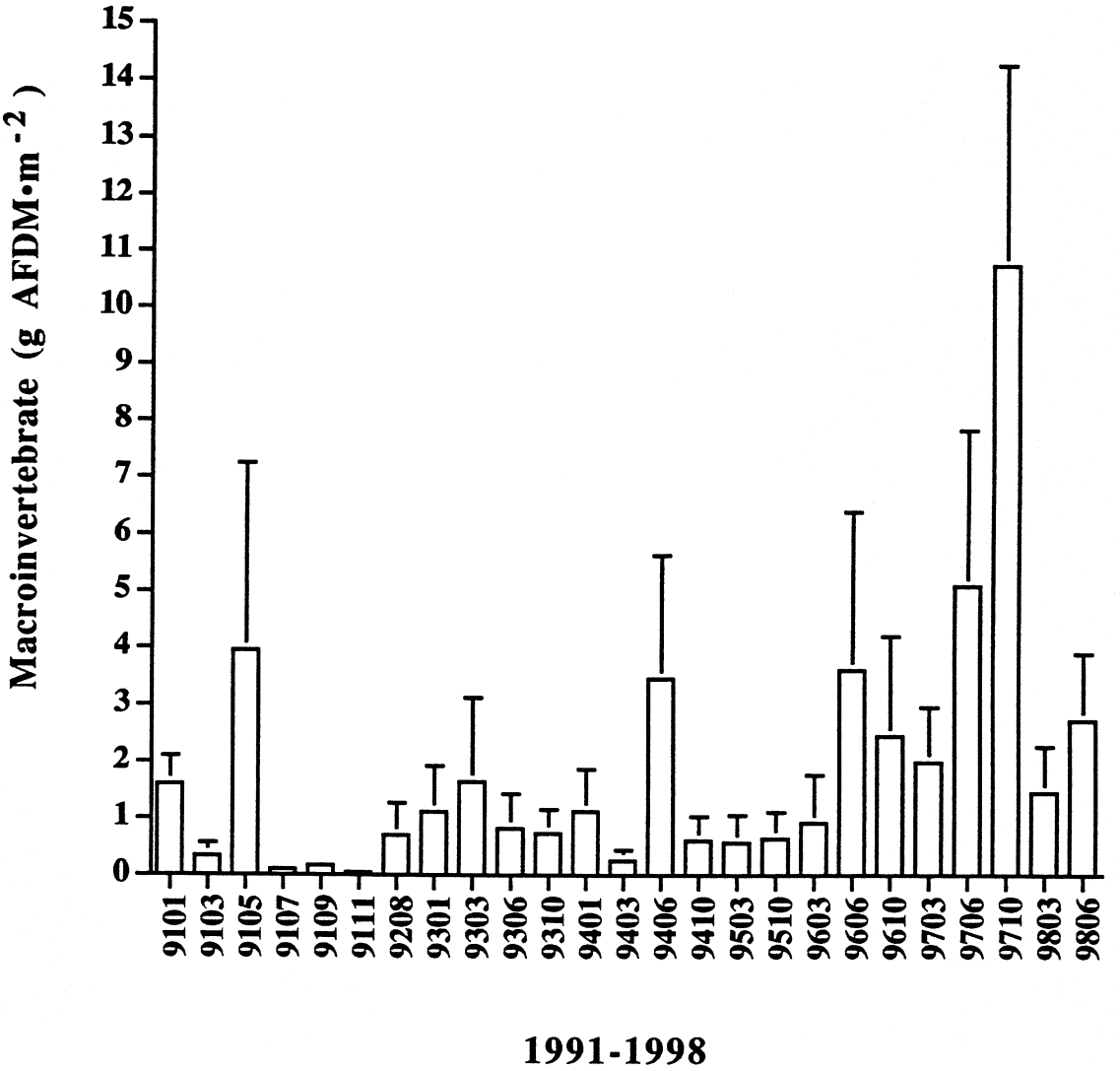

Figure 92. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at Lees Ferry pool Rkm 0.0 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

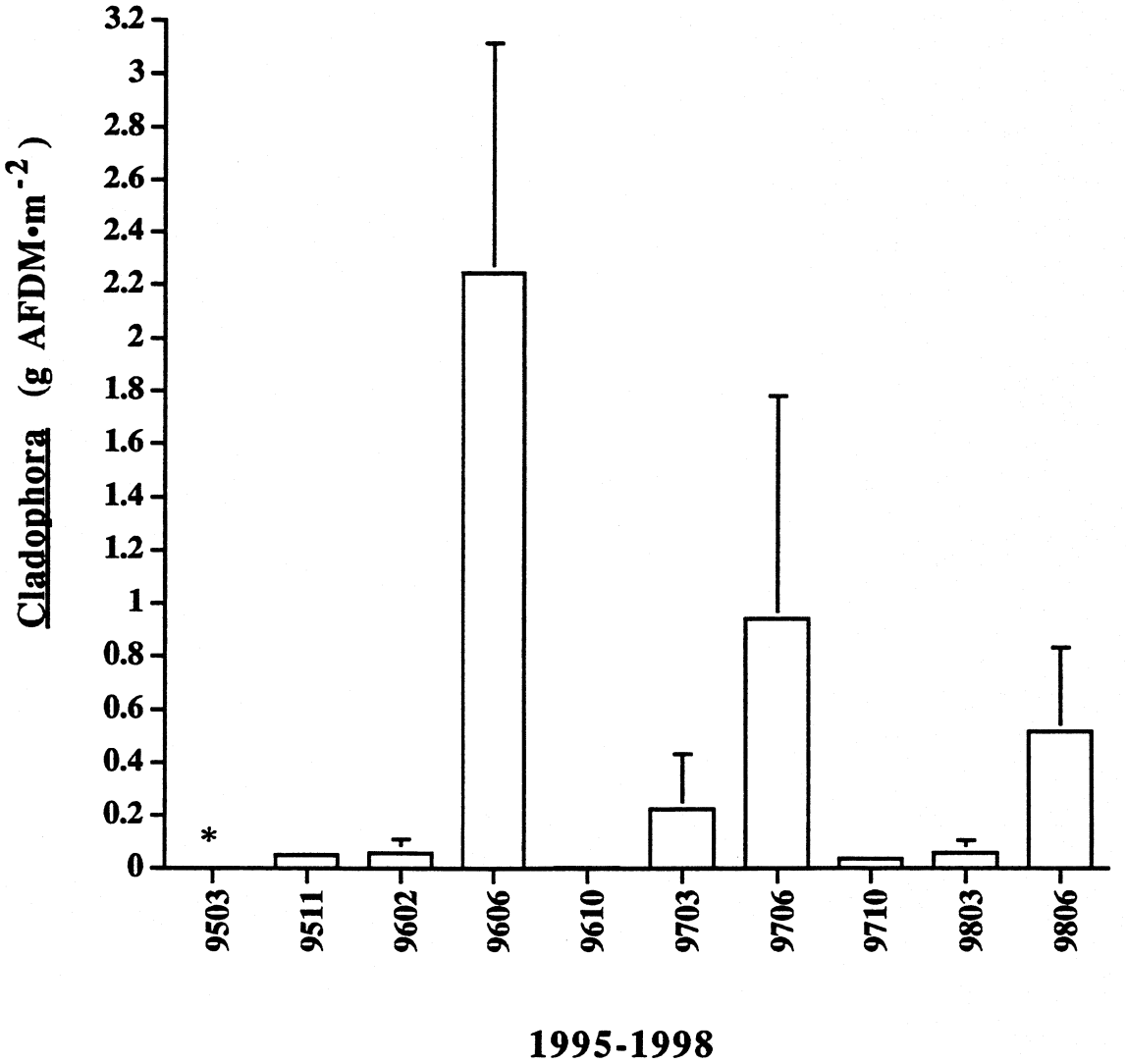

Figure 93. Cladophora biomass estimates (g AFDM•m^{-2}) at Two-Mile pool Rkm 3.1 from March 1995 to June 1998. Error bars represent (± 1 SE, n=12).
* represents 22 g AFDM•m^{-2} (± 22 SE).

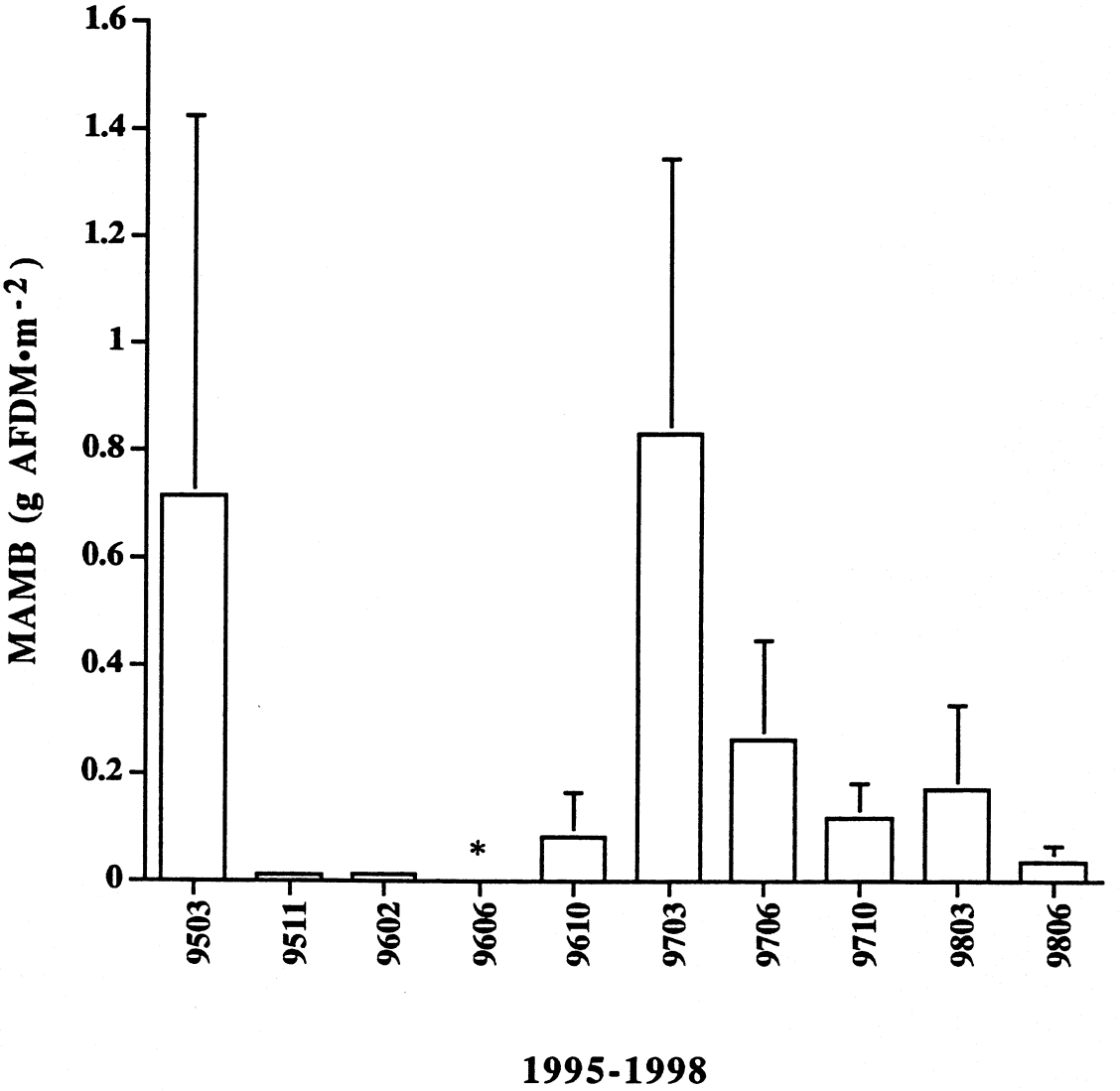

Figure 94. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at Two-Mile pool Rkm 3.1 from March 1995 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=12). * represents 5 g AFDM•m^{-2} (± 2).

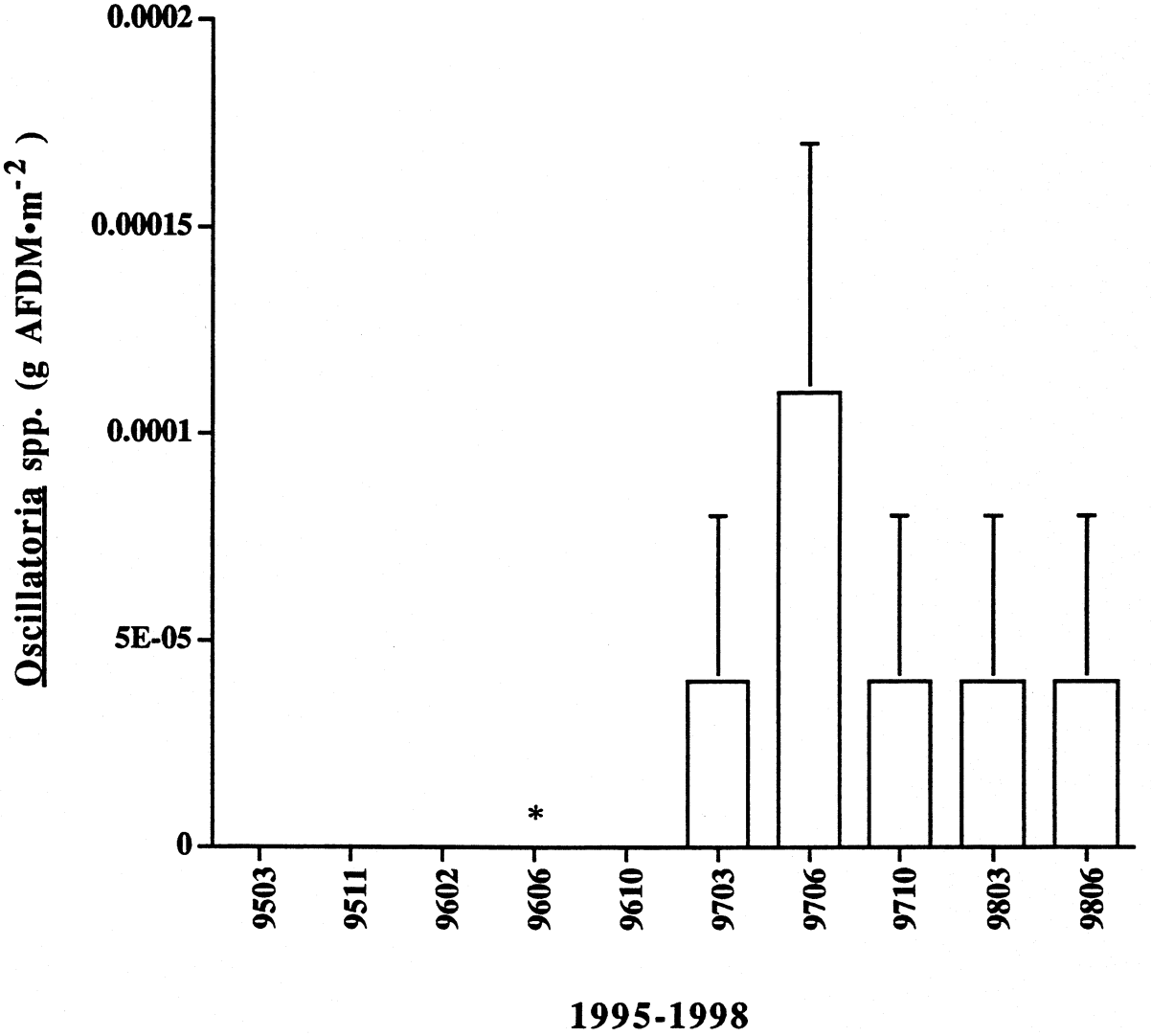

Figure 95. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at Two-Mile pool Rkm 3.1 from March 1995 to June 1998. Error bars represent (± 1 SE, n=12).
* represents 13 g AFDM•m^{-2} (± 11.5).

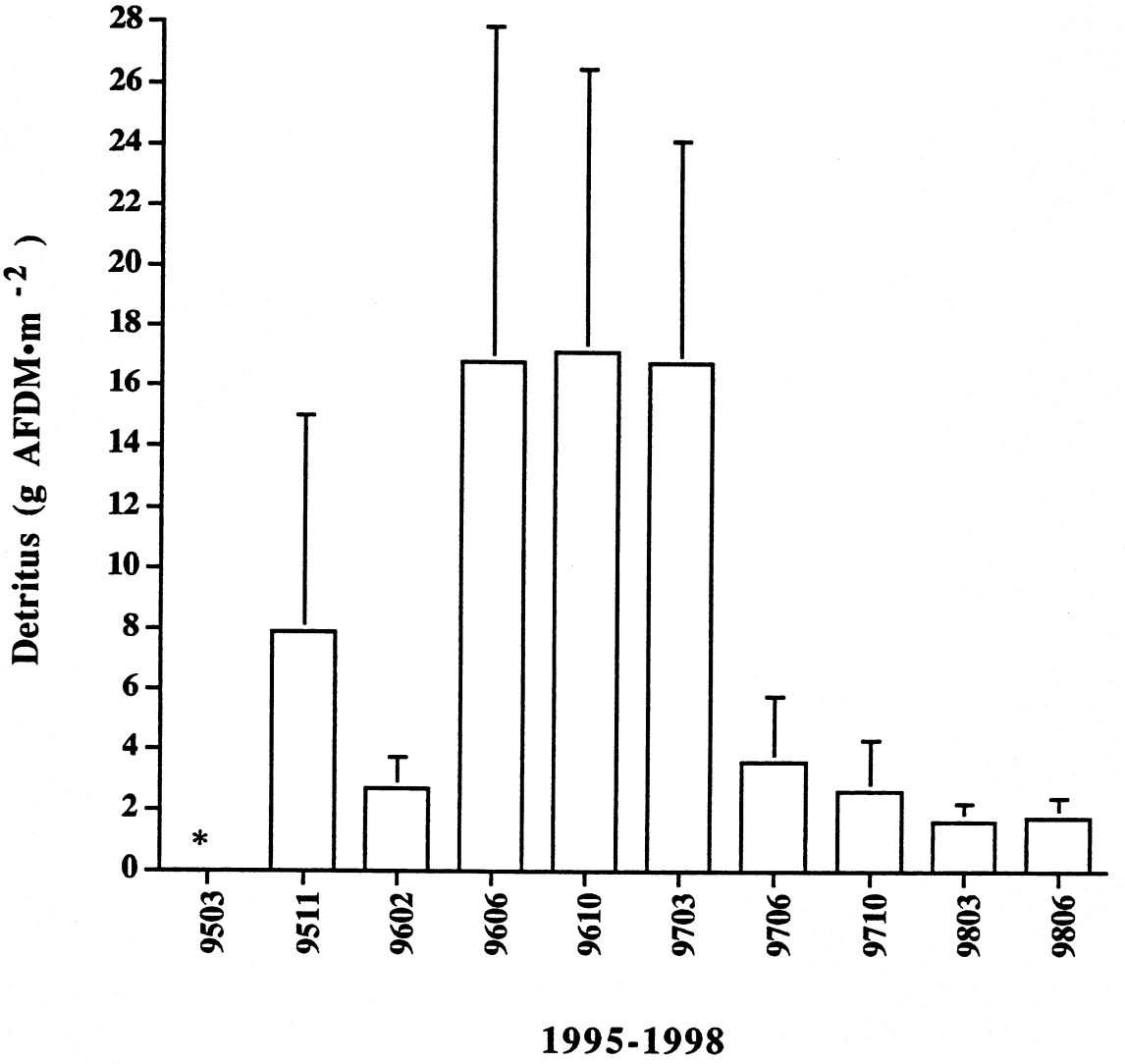

Figure 96. Detritus biomass estimates (g AFDM•m^{-2}) at Two-Mile pool Rkm 3.1 from March 1995 to June 1998. Error bars represent (± 1 SE, n=12).
* represents 822 g AFDM•m^{-2} (± 821).

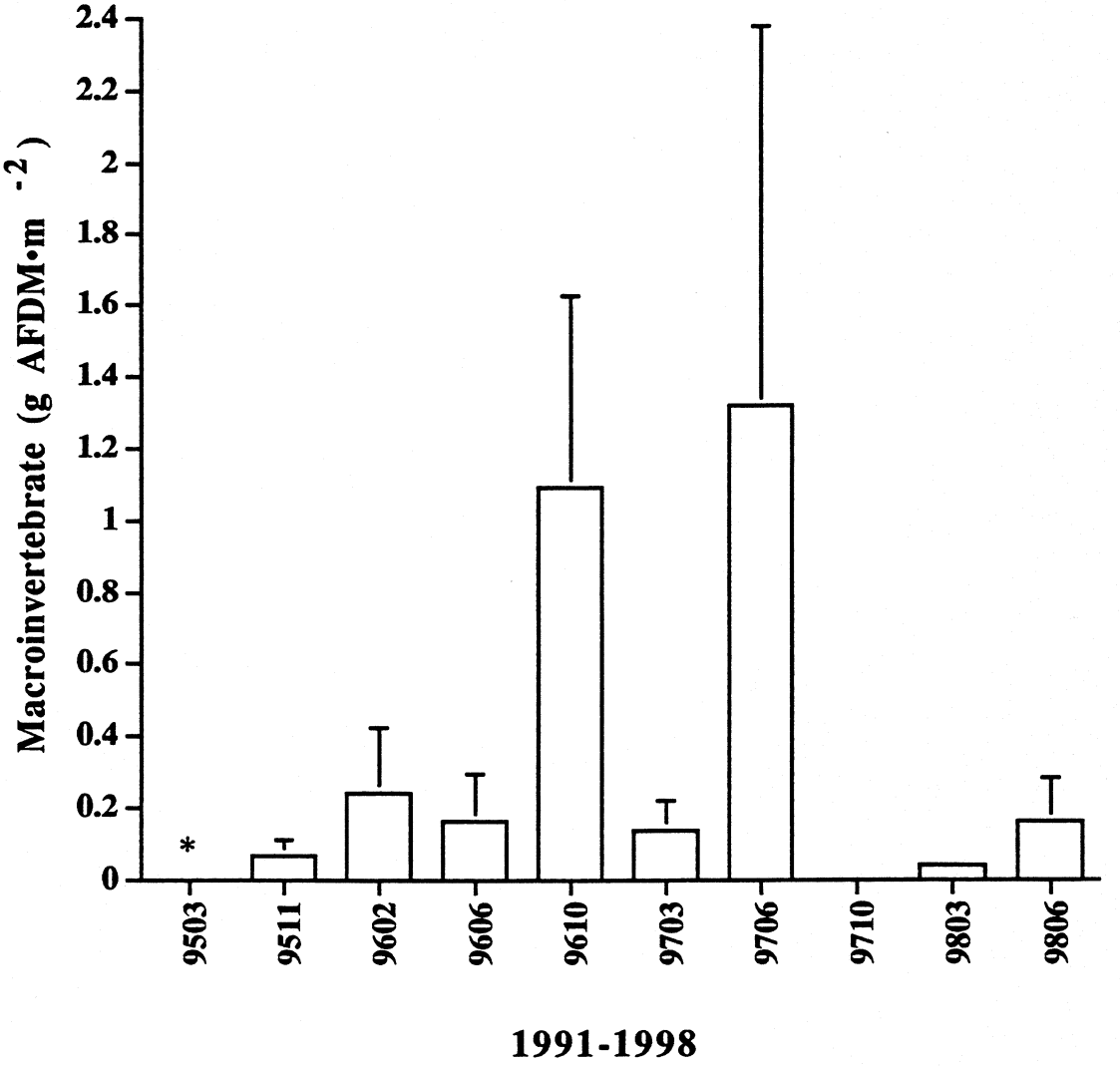

Figure 97. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at Two-Mile pool Rkm 3.1 from January 1991 to June 1998. Error bars represent (± 1 SE, n=6).
* represents 2 g AFDM•m^{-2} (± 2).

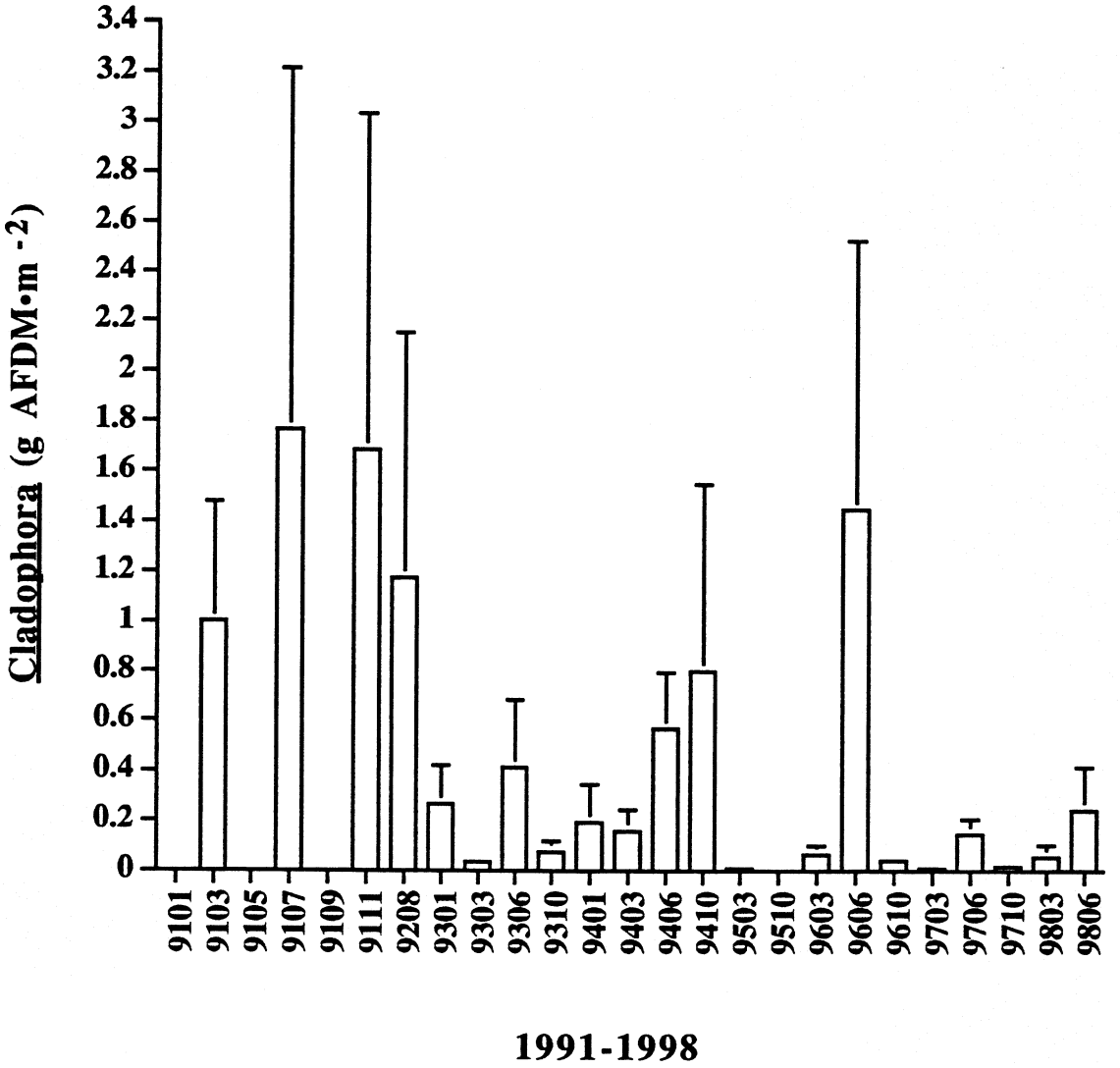

Figure 98. Cladophora biomass estimates (g AFDM•m^{-2}) at 60 Mile rapid pool Rkm 95.7 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

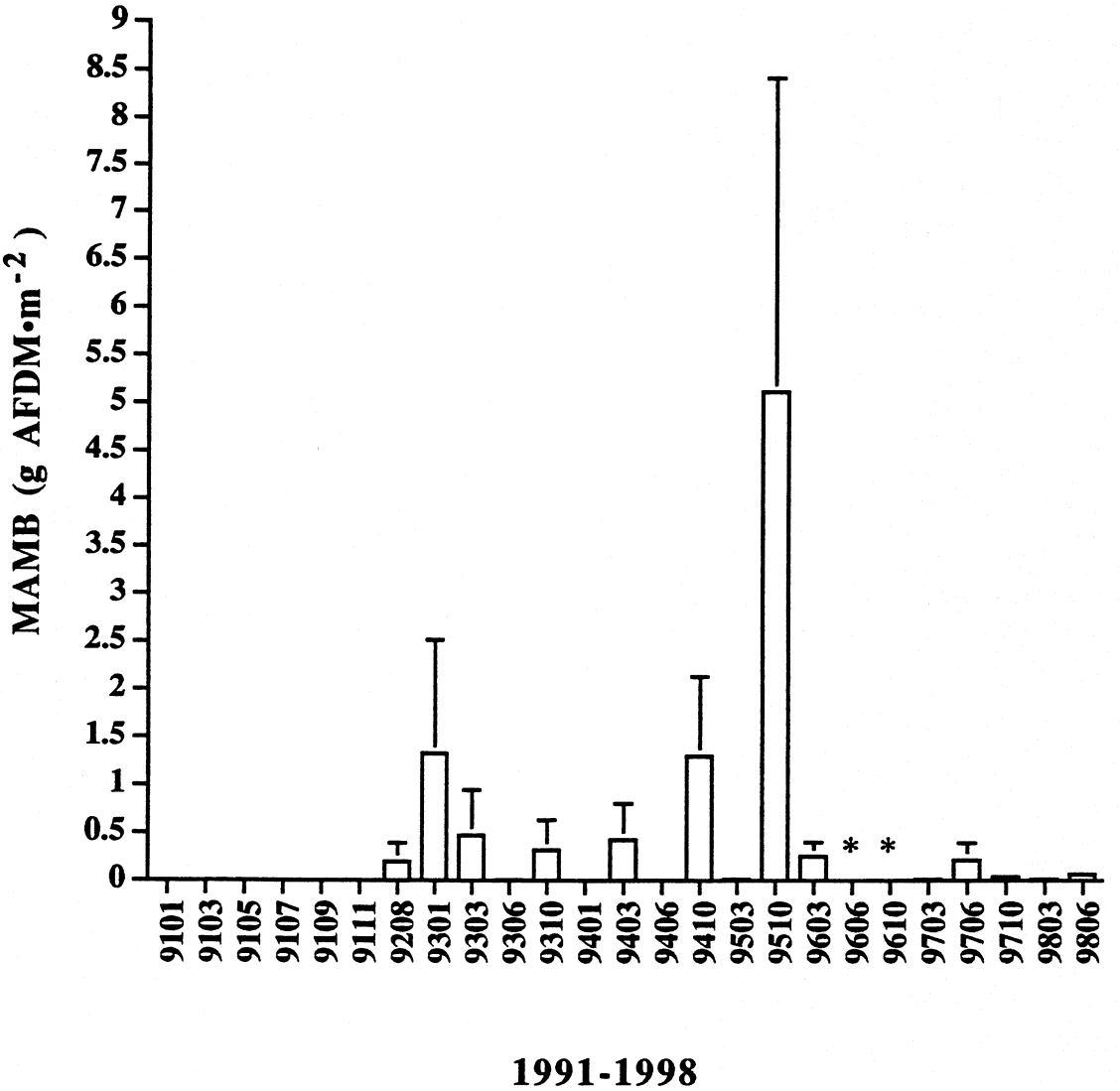

Figure 99. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at 60 Mile rapid pool Rkm 95.7 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=12).
* at 9606 represents 12 g AFDM•m^{-2} (± 10).
* at 9610 represents 9 g AFDM•m^{-2} (± 6).

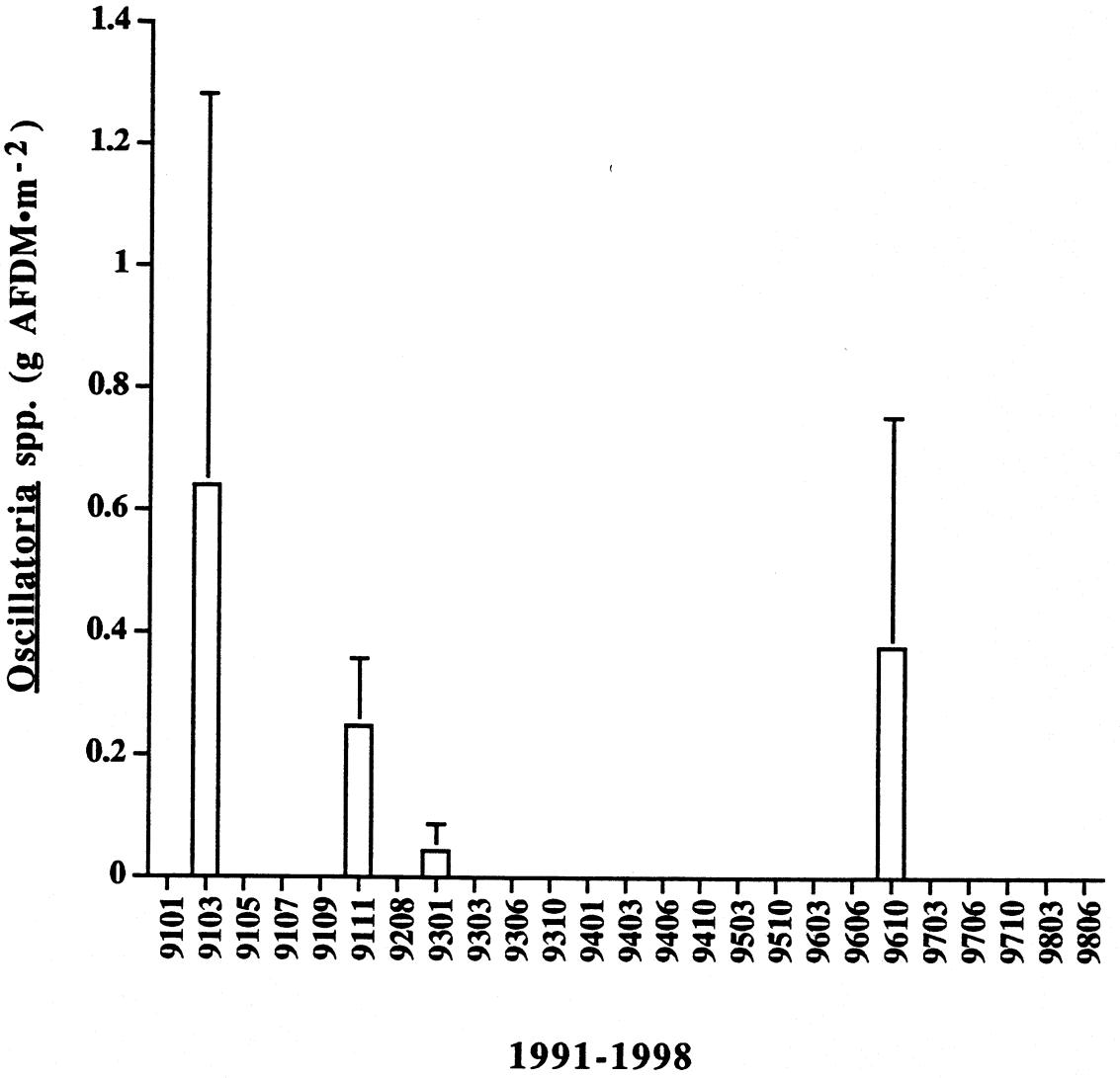

Figure 100. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at 60 Mile rapid pool Rkm 95.7 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

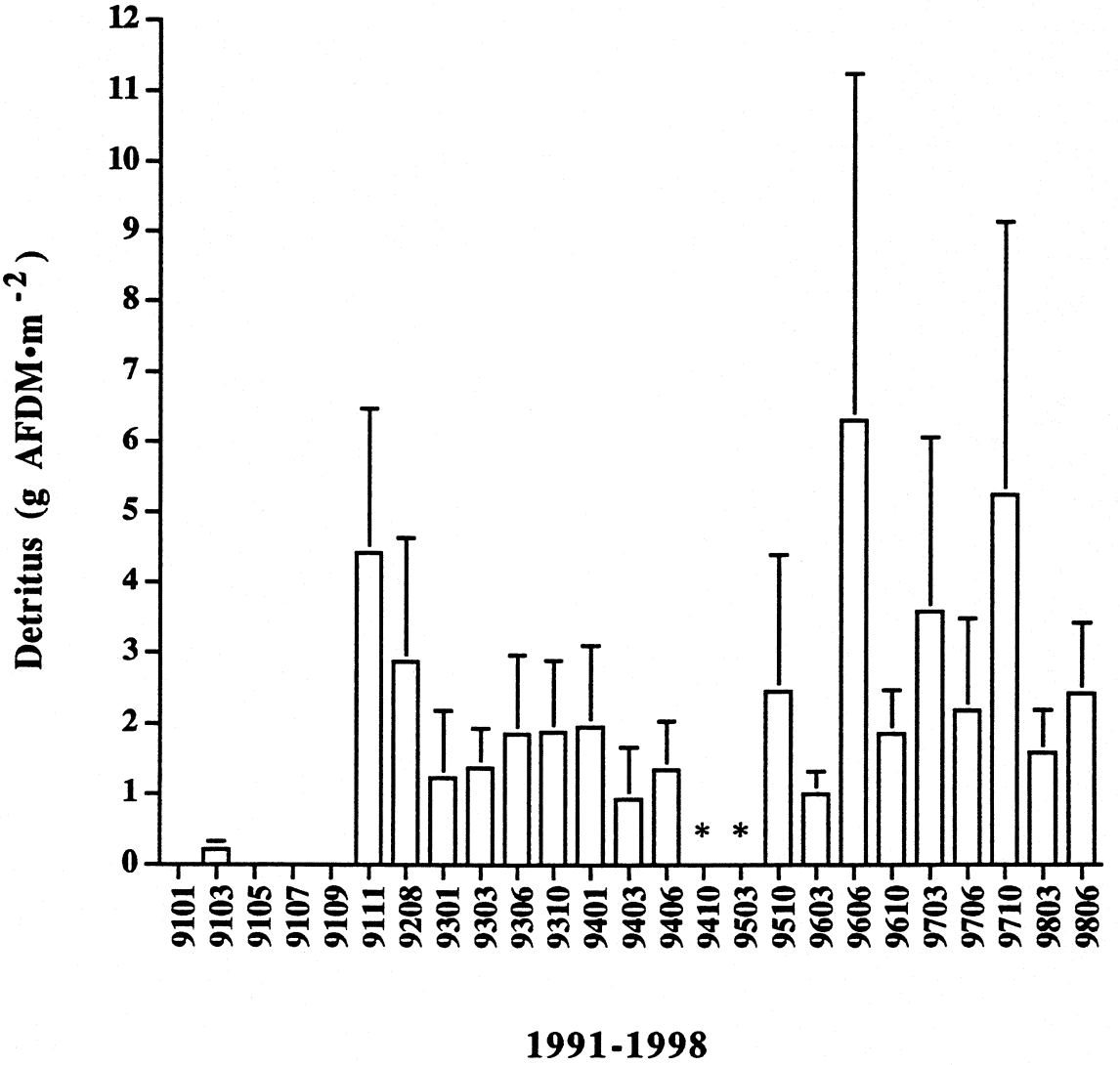

Figure 101. Detritus biomass estimates (g AFDM•m^{-2}) at 60 Mile rapid pool Rkm 95.7 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).
* at 9410 represents 19 g AFDM•m^{-2} (± 14).
* at 9503 represents 49 g AFDM•m^{-2} (± 41).

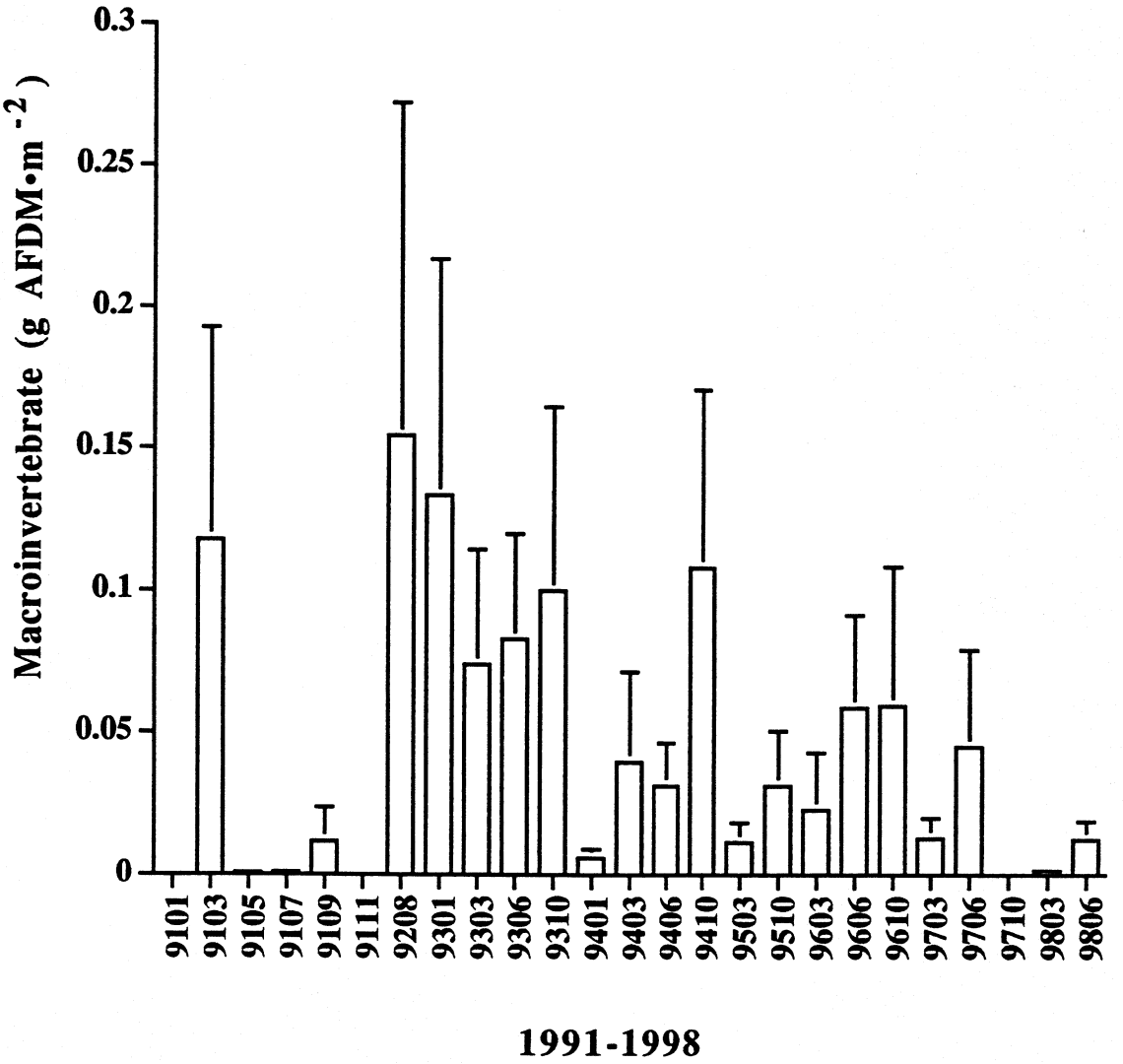

Figure 102. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at 60 Mile rapid pool Rkm 95.7 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

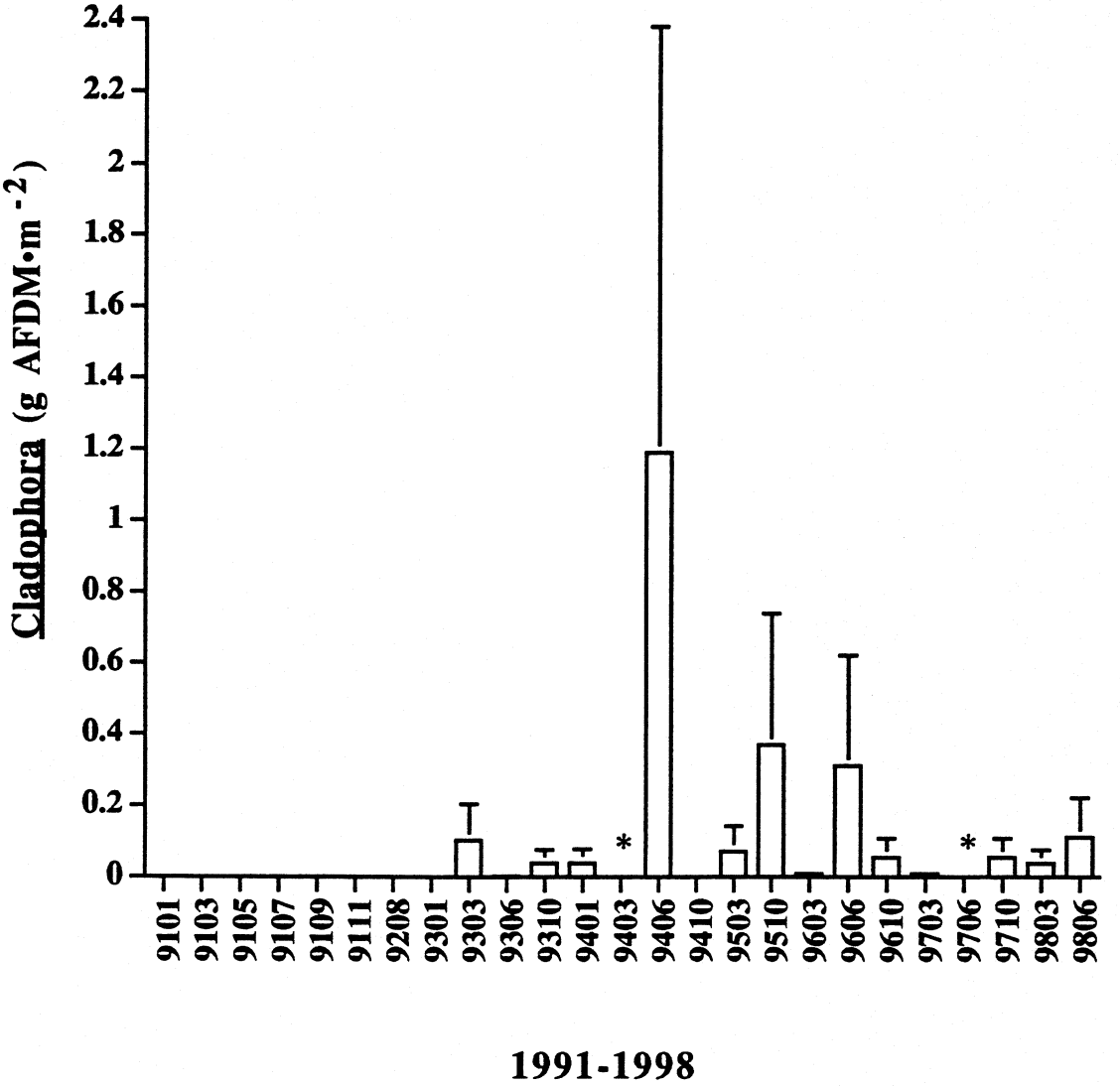

Figure 103. Cladophora biomass estimates (g AFDM•m^{-2}) at Tanner Canyon pool Rkm 108.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).
* at 9403 represents 5.8 g AFDM•m^{-2} (± 5.8).
* at 9706 represents 2 g AFDM•m^{-2} (± 2).

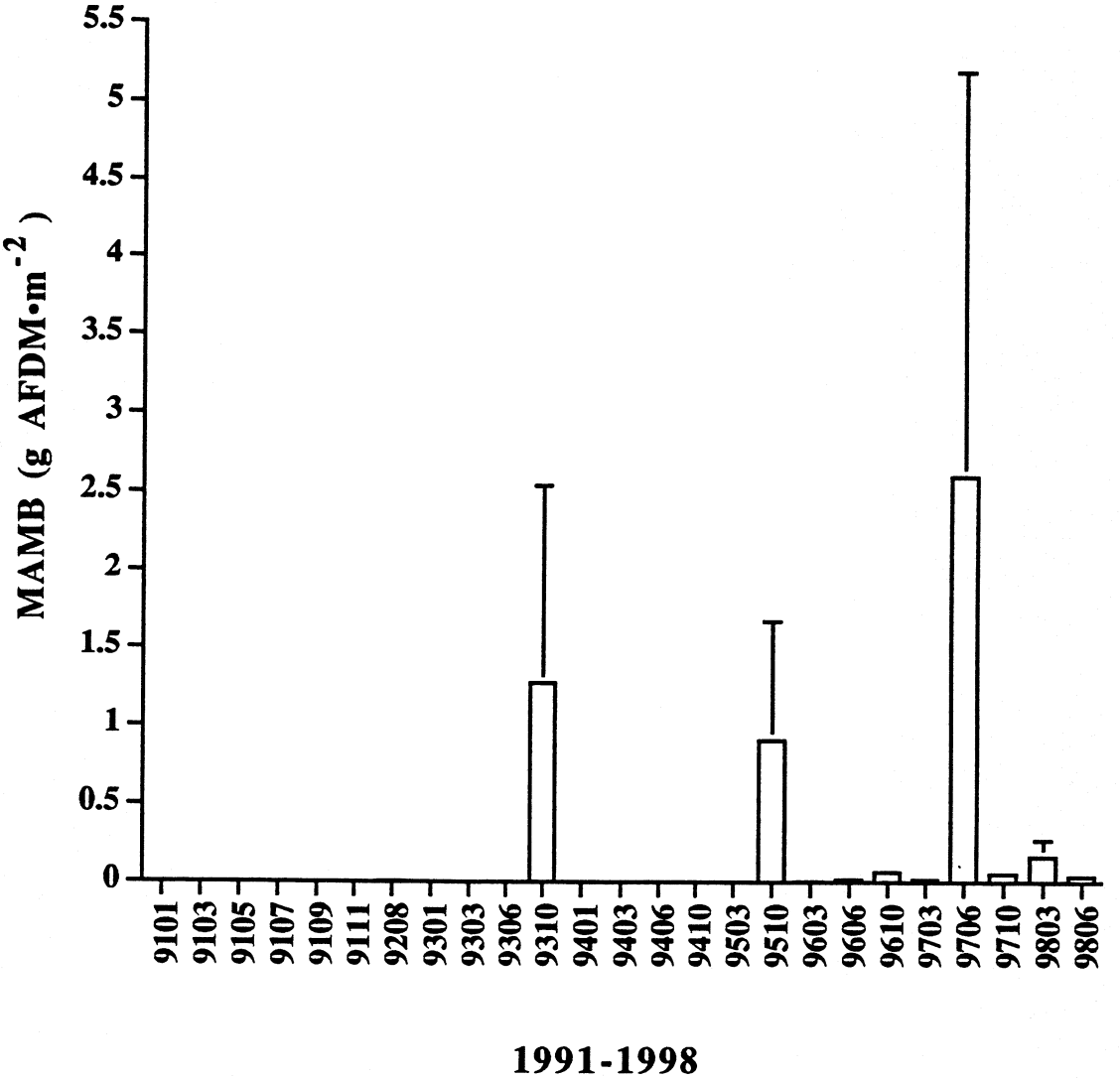

Figure 104. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at Tanner Canyon pool Rkm 108.8 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=12).

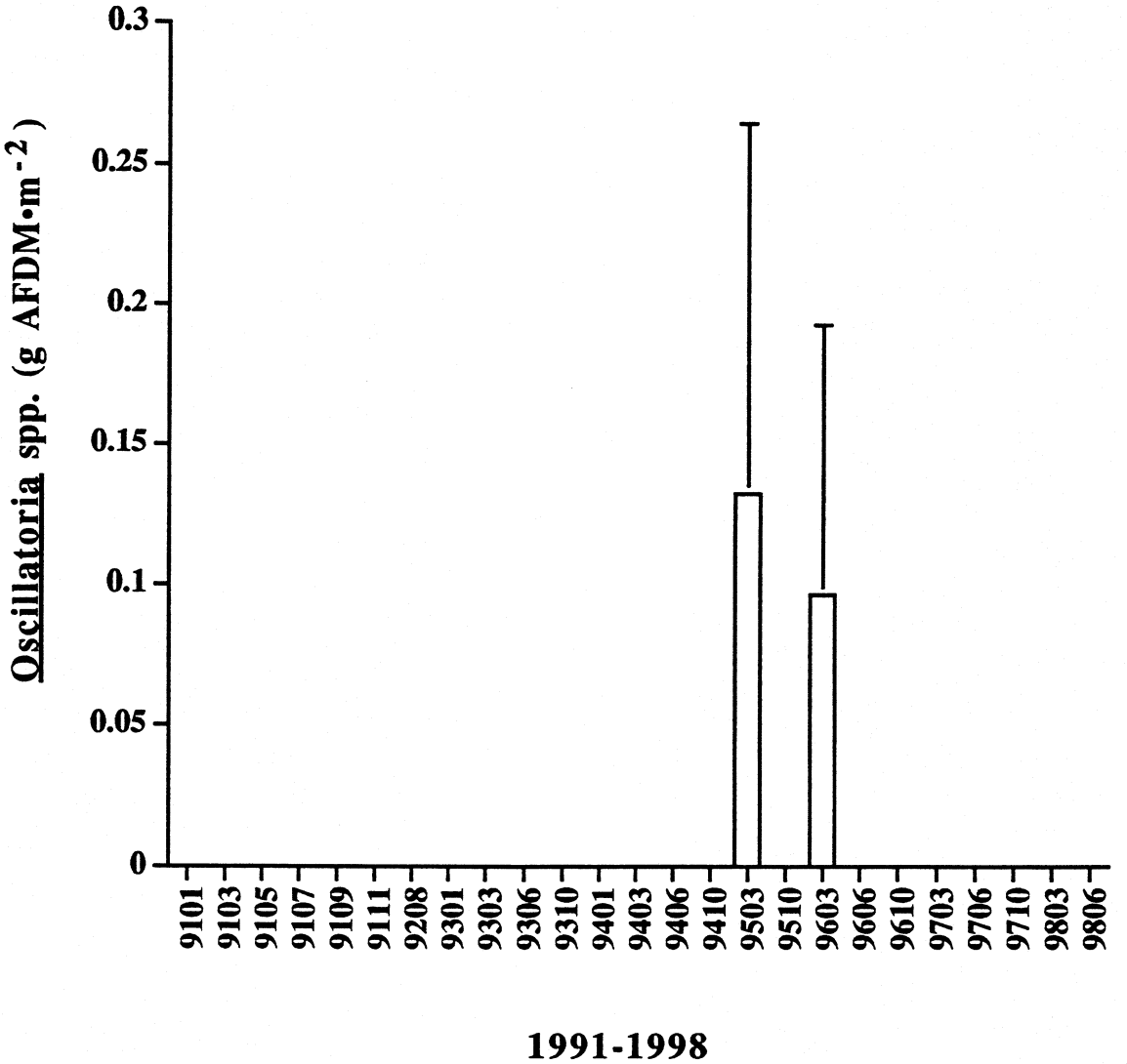

Figure 105. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at Tanner Canyon pool Rkm 108.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

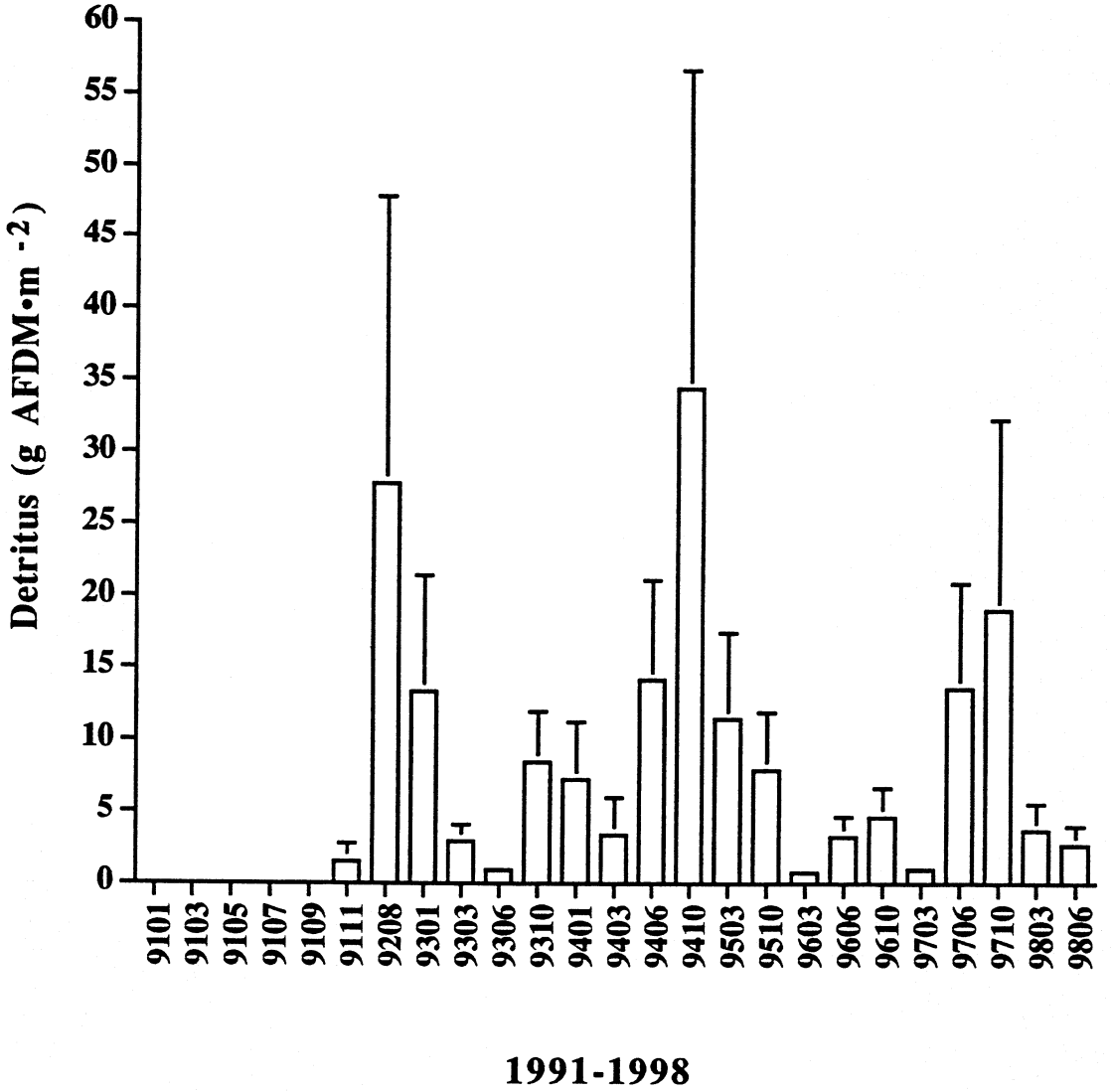

Figure 106. Detritus biomass estimates (g AFDM•m^{-2}) at Tanner Canyon pool Rkm 108.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

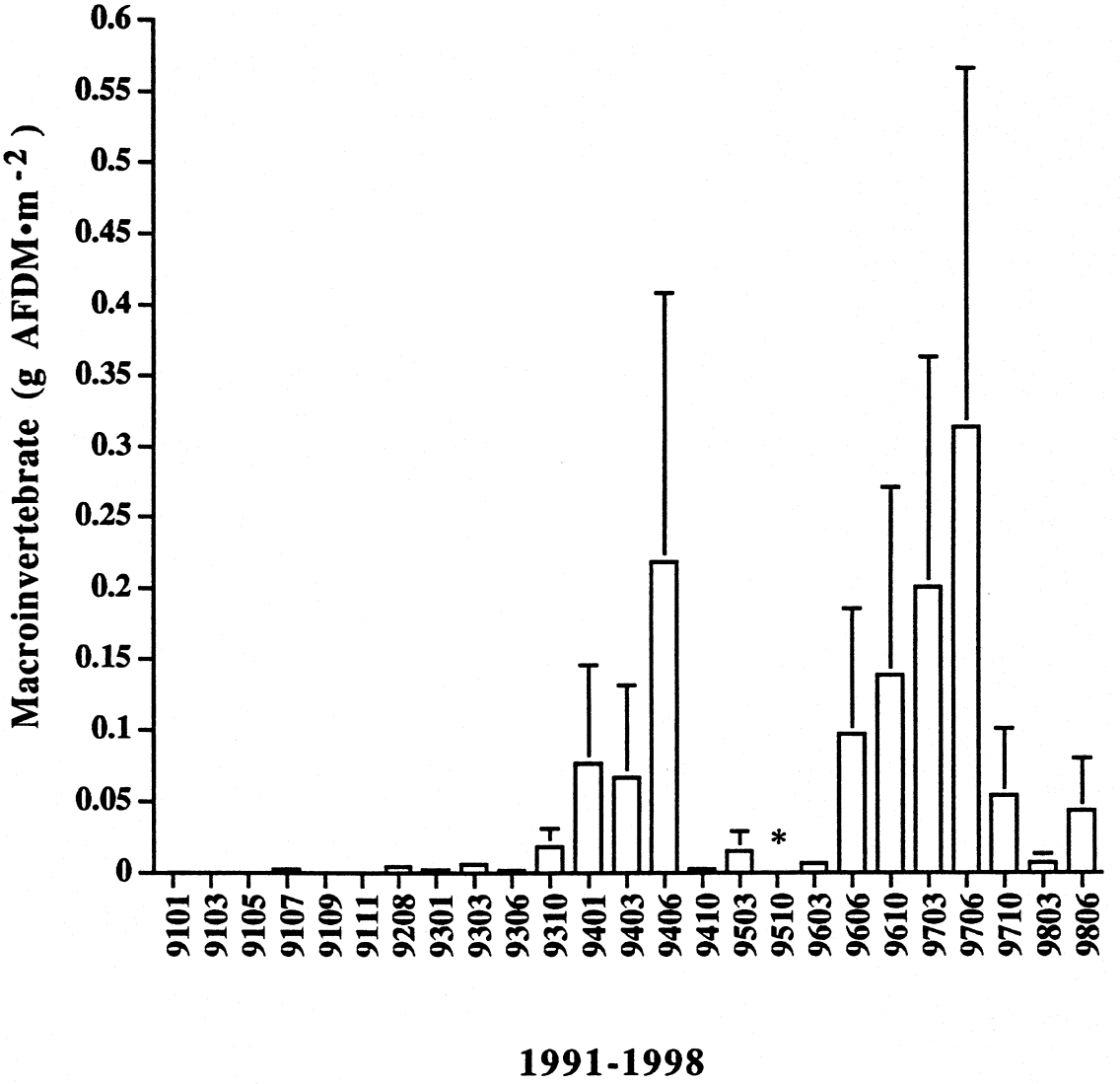

Figure 107. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at Tanner Canyon pool Rkm 108.8 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).
* represents 0.5 g AFDM•m^{-2} (± 0.5).

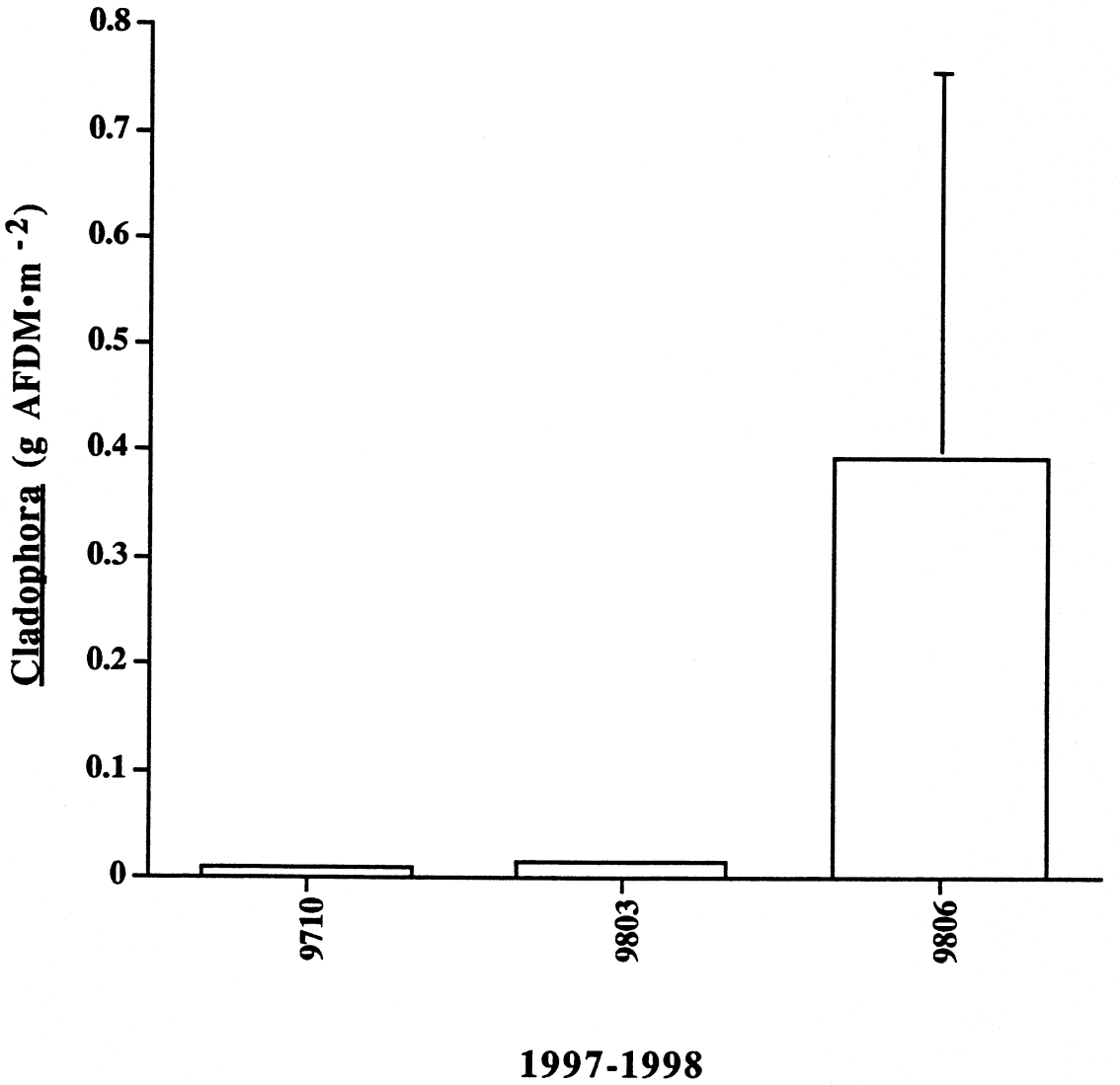

Figure 108. Cladophora biomass estimates (g AFDM•m^{-2}) at Middle Granite Gorge pool Rkm 203.2 from October 1997 to June 1998. Error bars represent (± 1 SE, n=12).

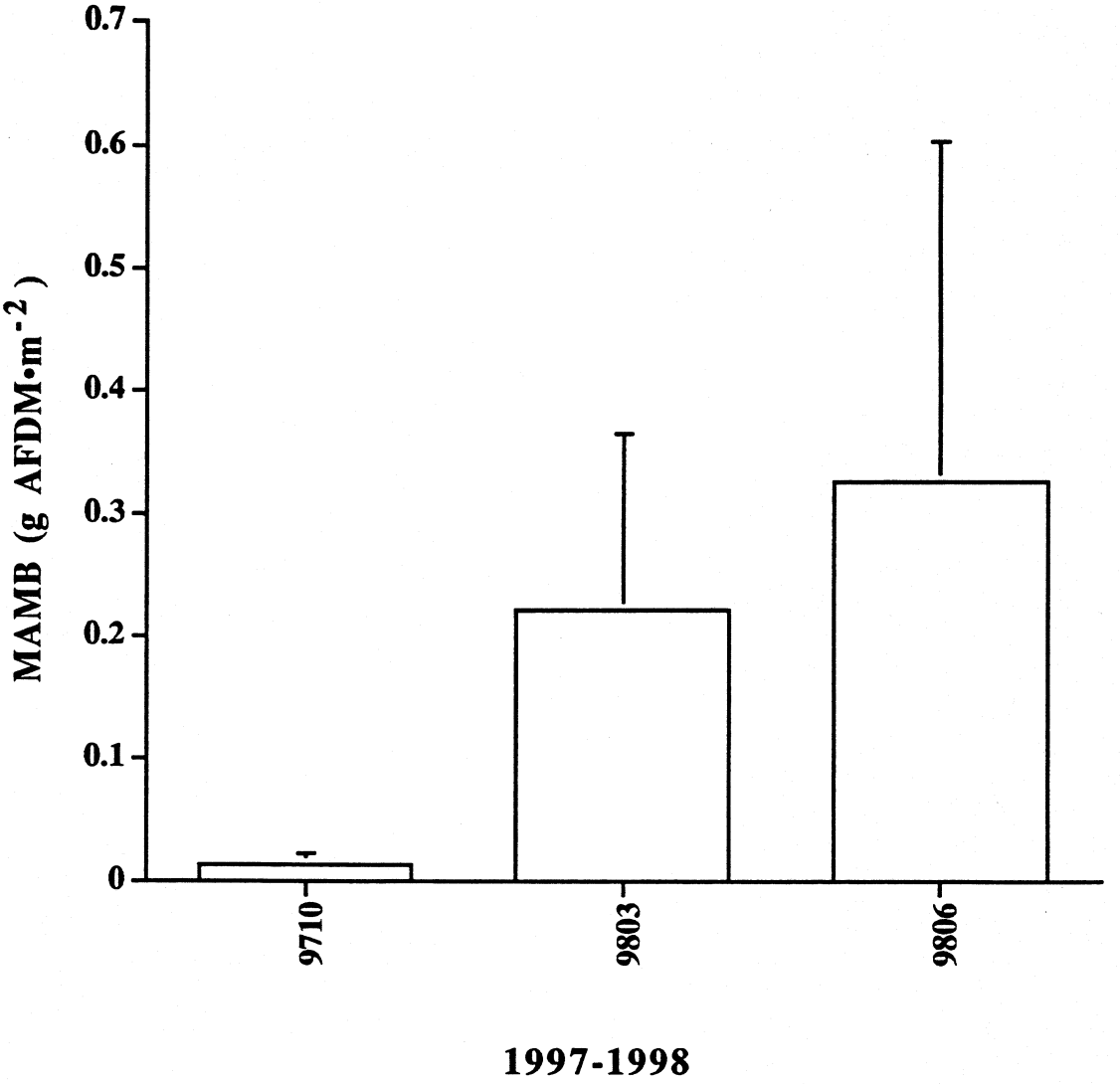

Figure 109. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at Middle Granite Gorge pool Rkm 203.2 from October 1997 to June 1998. Error bars represent (± 1 SE, n=12).

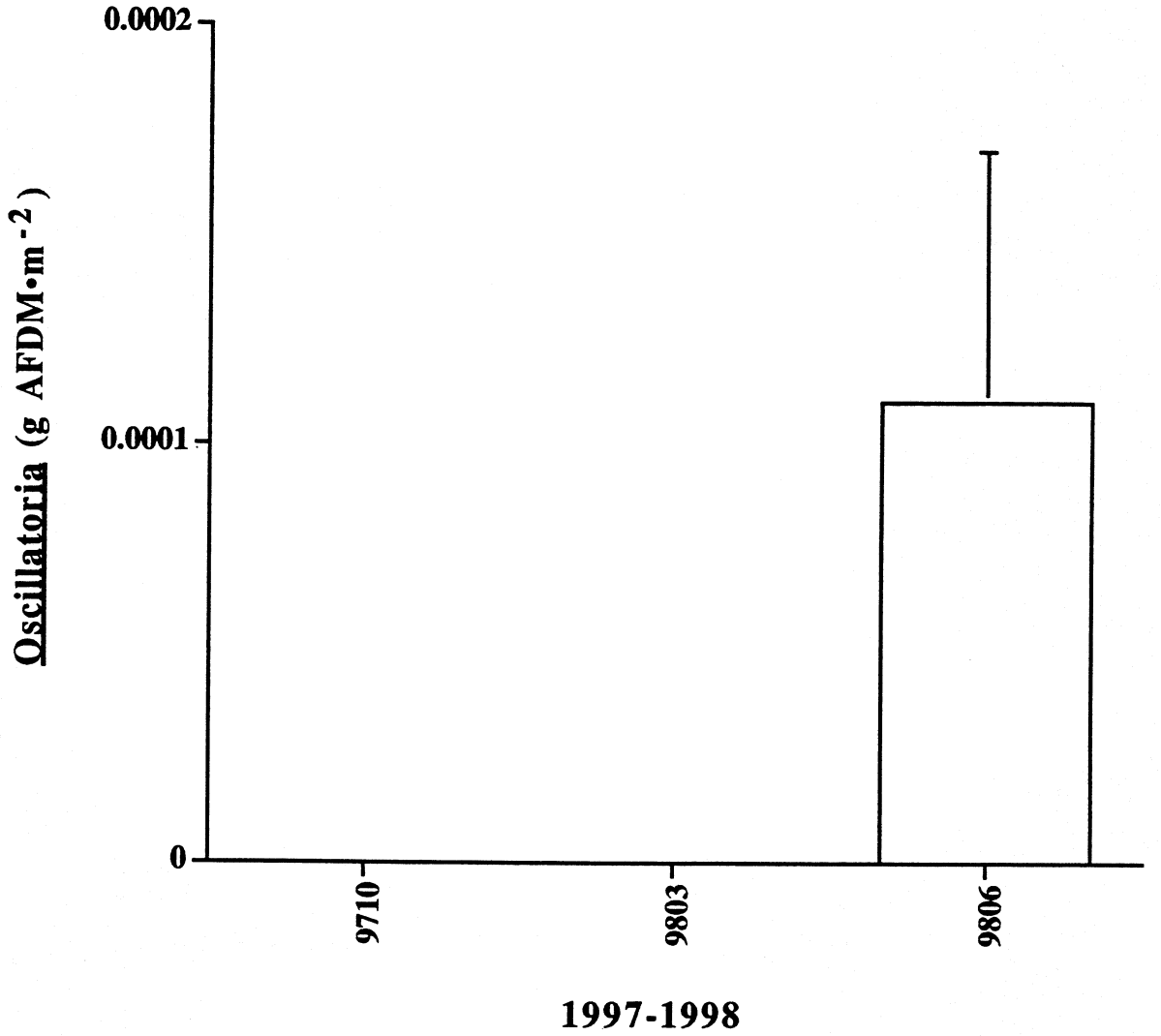

Figure 110. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at Middle Granite Gorge pool Rkm 203.2 from October 1997 to June 1998. Error bars represent (± 1 SE, n=12).

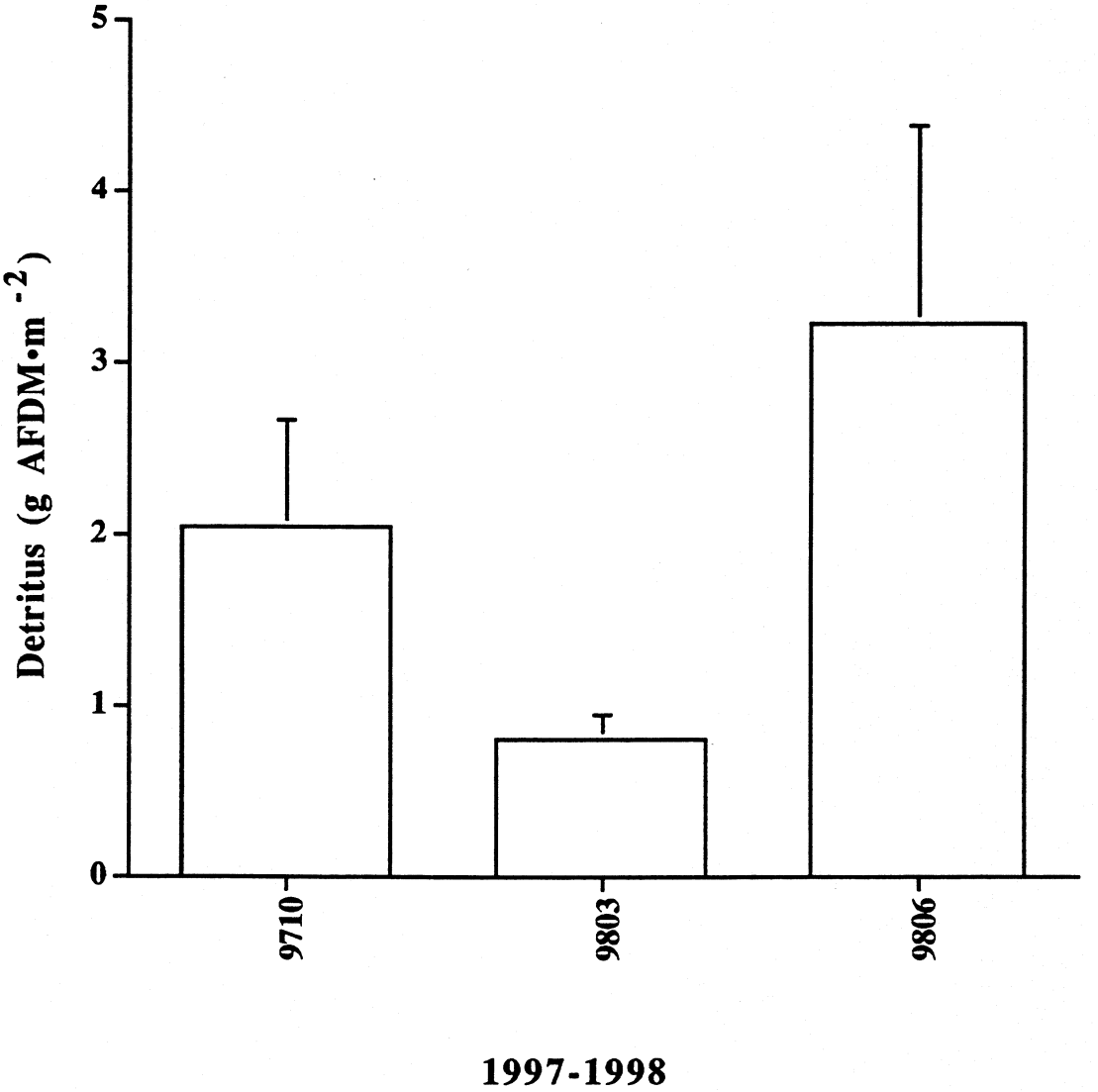

Figure 111. Detritus biomass estimates (g AFDM•m^{-2}) at Middle Granite Gorge pool Rkm 203.2 from October 1997 to June 1998. Error bars represent (± 1 SE, n=12).

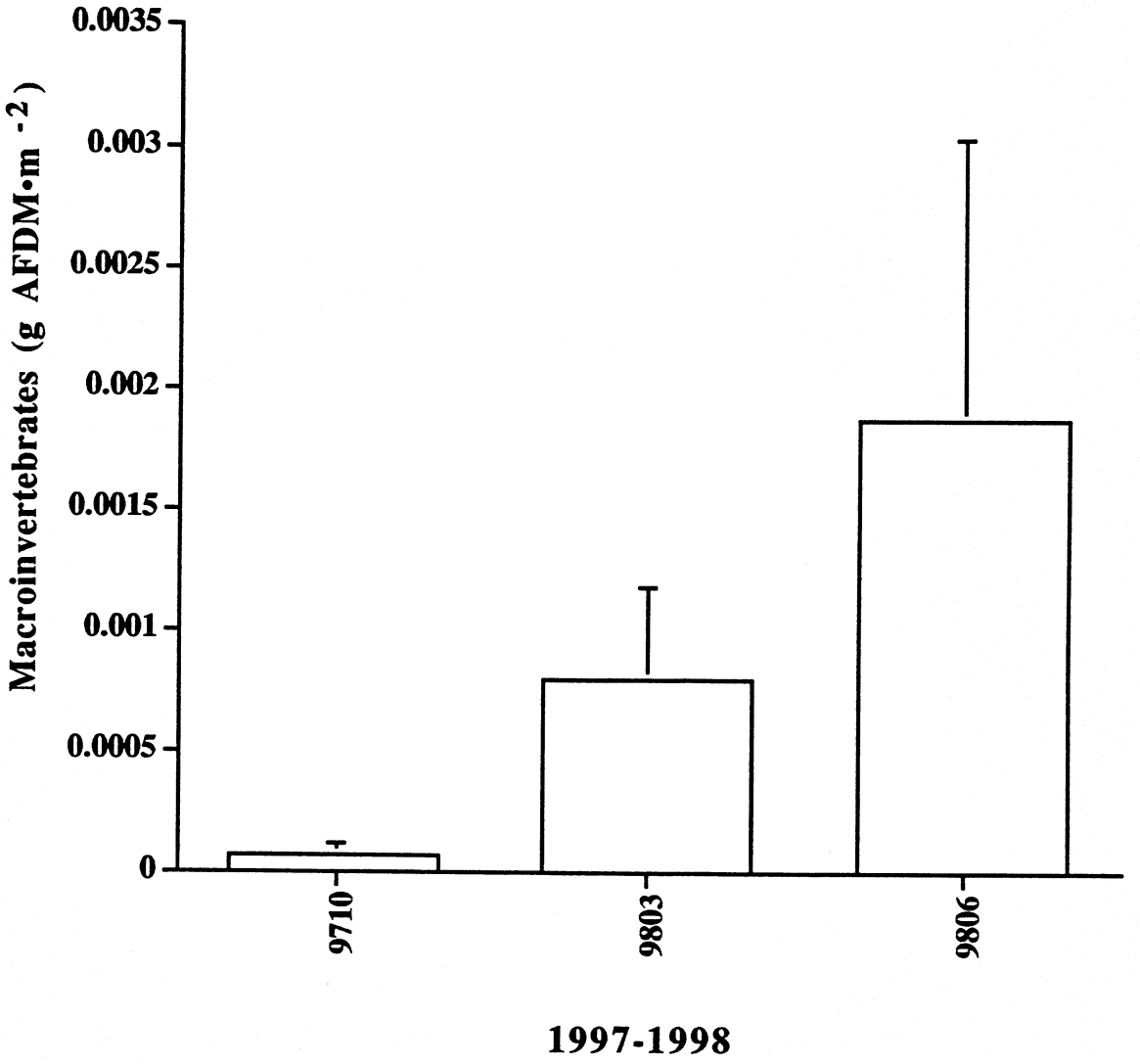

Figure 112. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at Middle Granite Gorge pool Rkm 203.2 from October 1997 to June 1998. Error bars represent (± 1 SE, n=12).

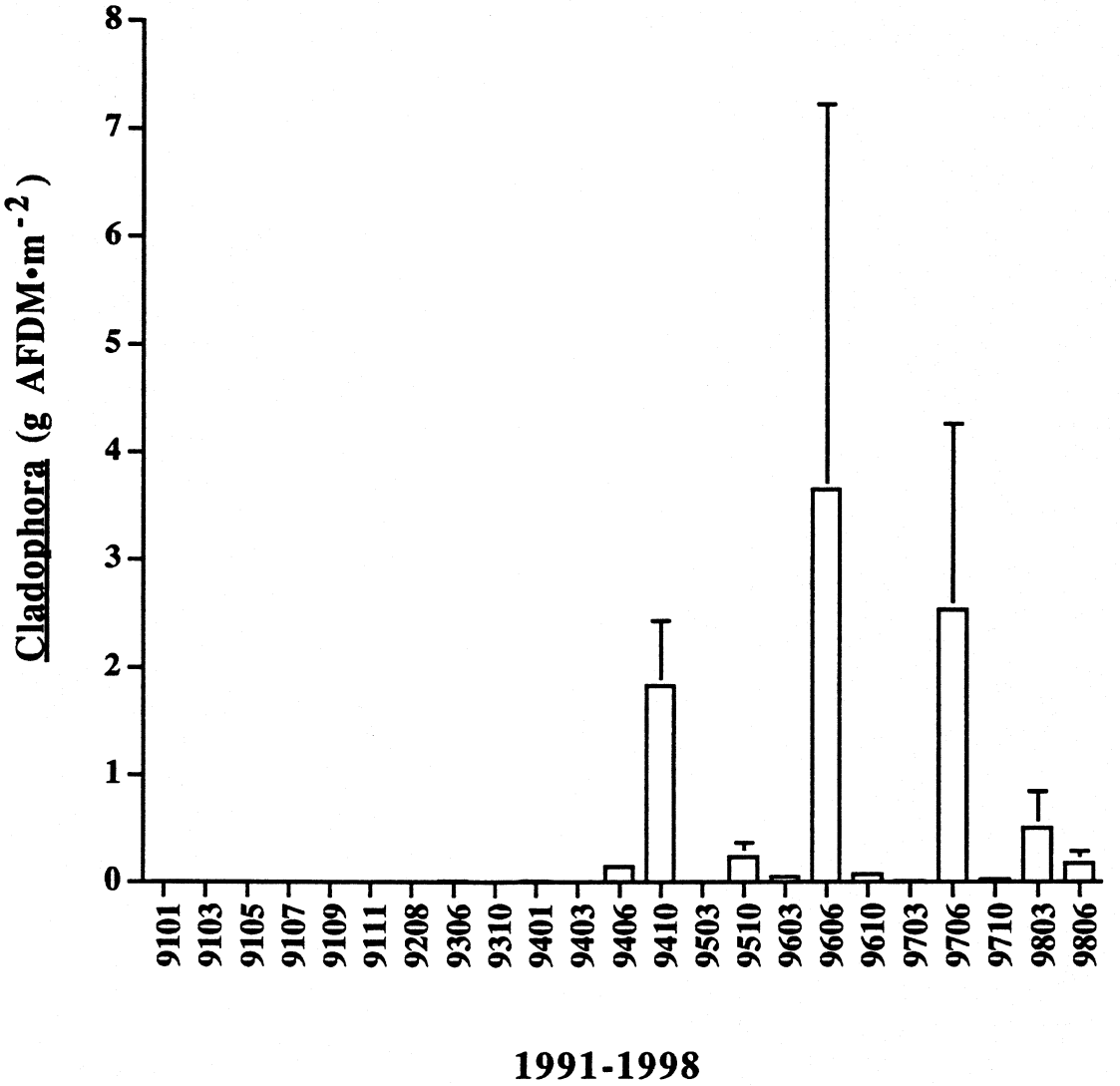

Figure 113. Cladophora biomass estimates (g AFDM•m^{-2}) at Spring Canyon pool Rkm 326.4 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

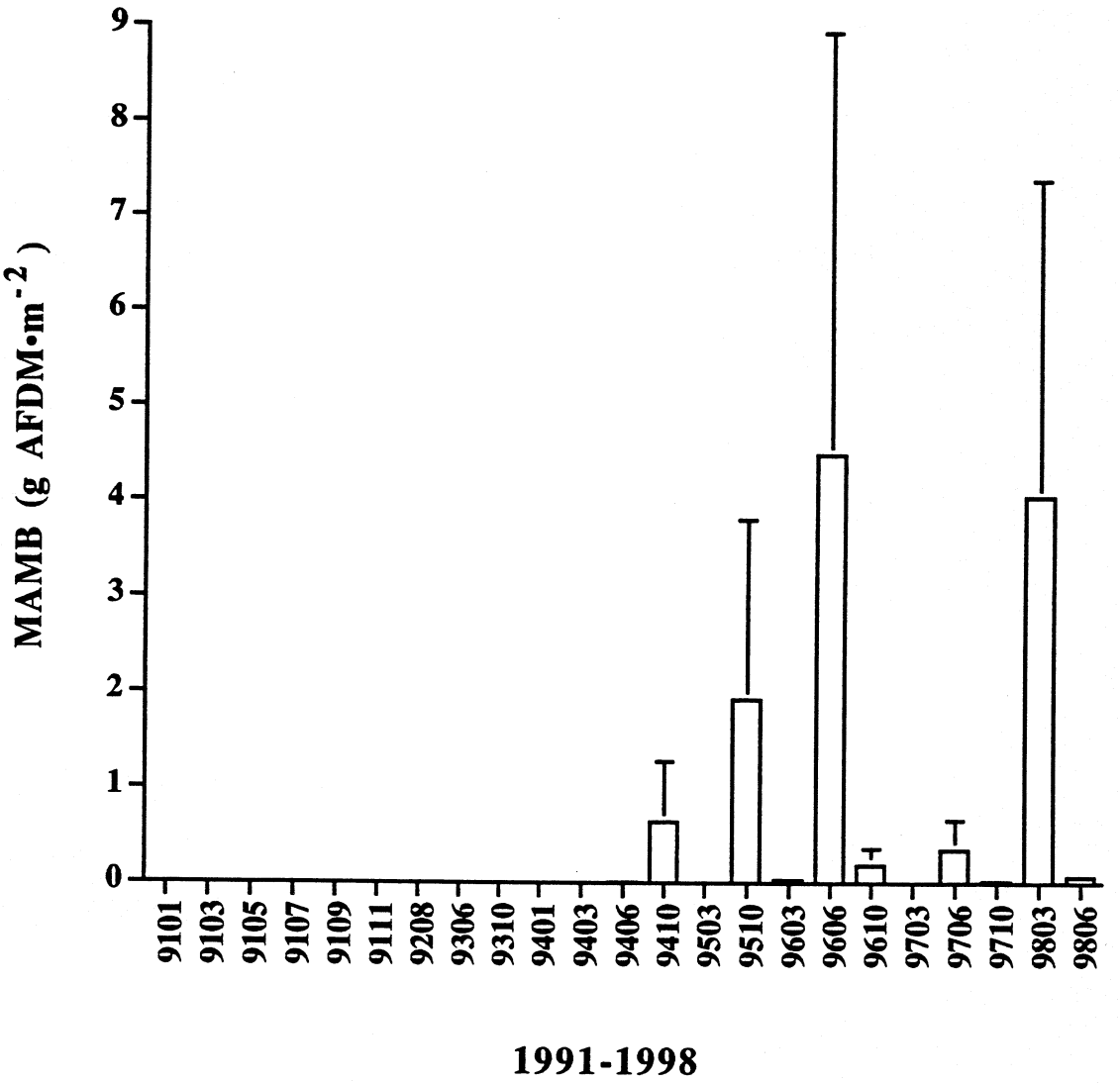

Figure 114. Miscellaneous algae, macrophytes and bryophytes (MAMB) biomass estimates (g AFDM•m^{-2}) at Spring Canyon pool Rkm 326.4 from January 1991 to June 1998. MAMB was not collected prior to August 1992. Error bars represent (± 1 SE, n=12).

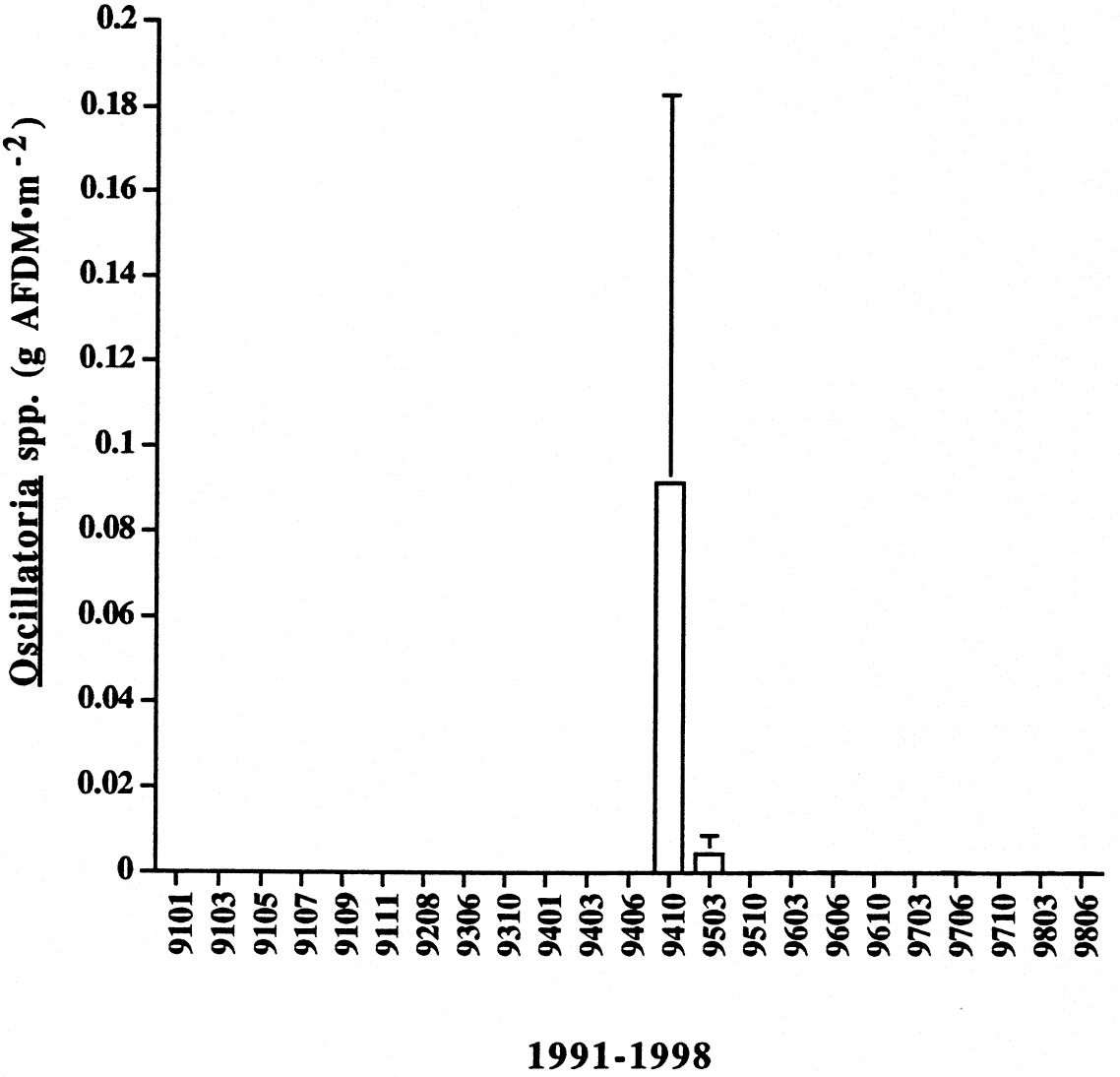

Figure 115. Oscillatoria spp. biomass estimates (g AFDM•m^{-2}) at Spring Canyon pool Rkm 326.4 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

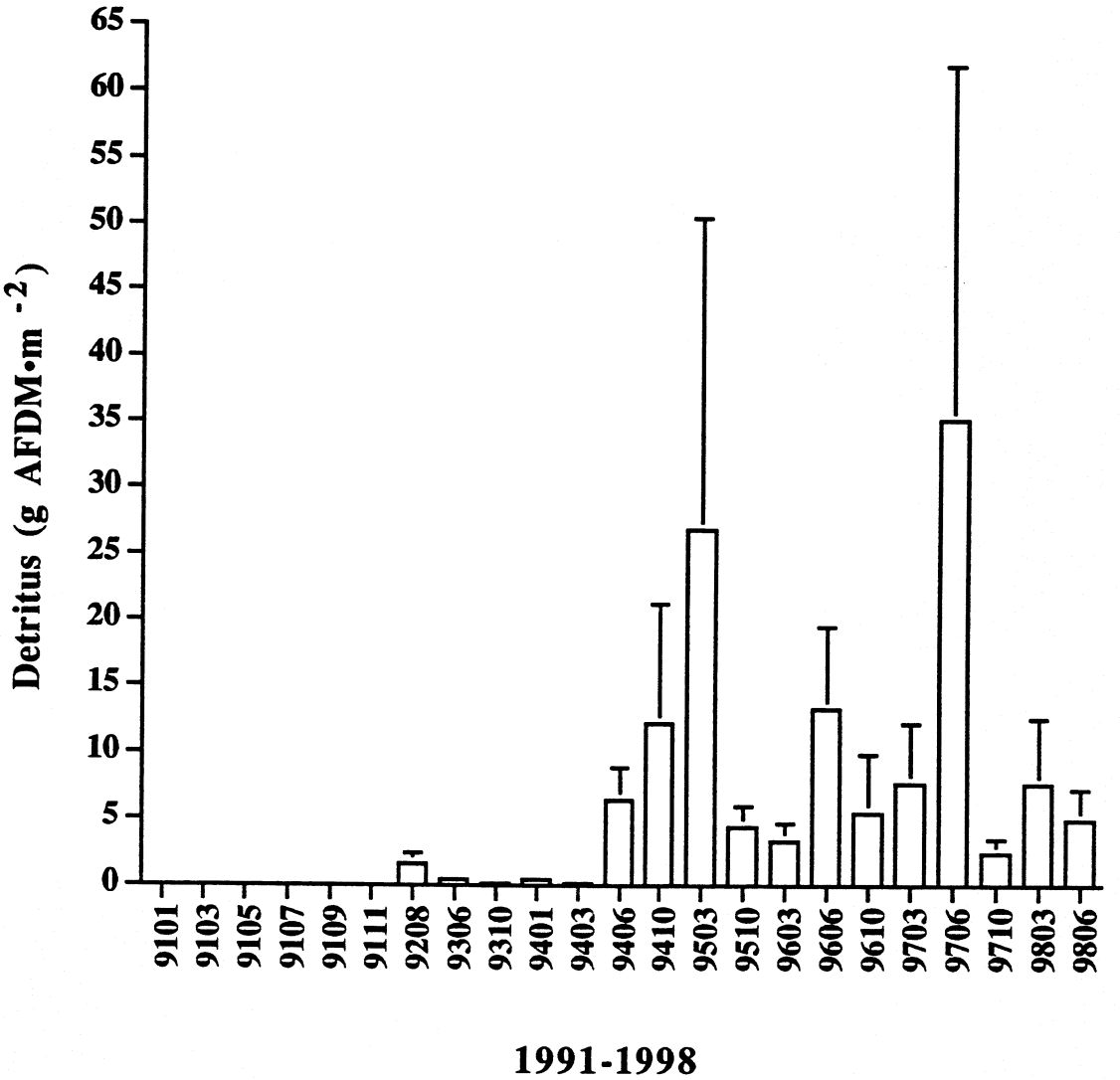

Figure 116. Detritus biomass estimates (g AFDM•m^{-2}) at Spring Canyon pool Rkm 326.4 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

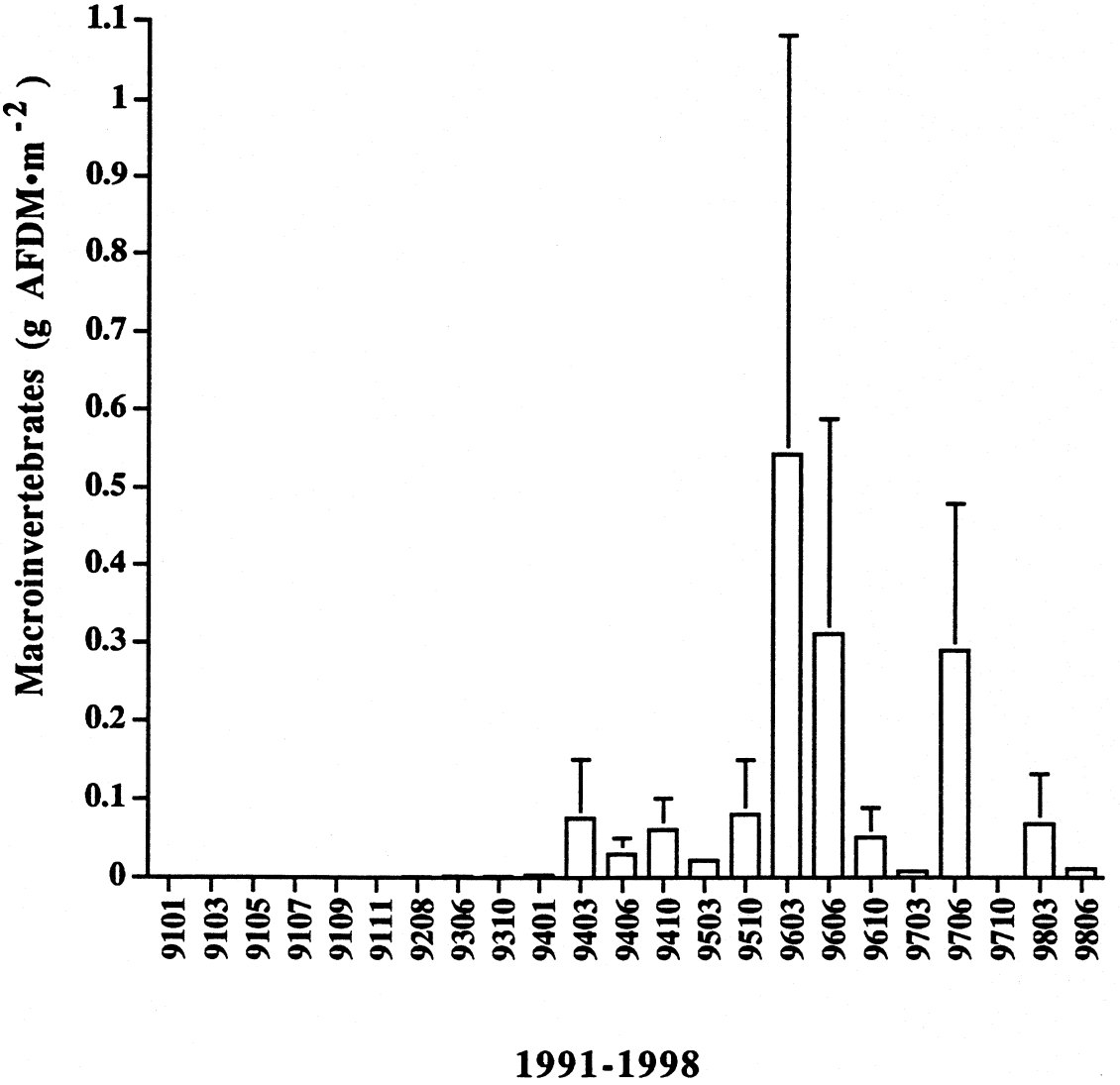

Figure 117. Macroinvertebrate biomass estimates (g AFDM•m^{-2}) at Spring Canyon pool Rkm 326.4 from January 1991 to June 1998. Error bars represent (± 1 SE, n=12).

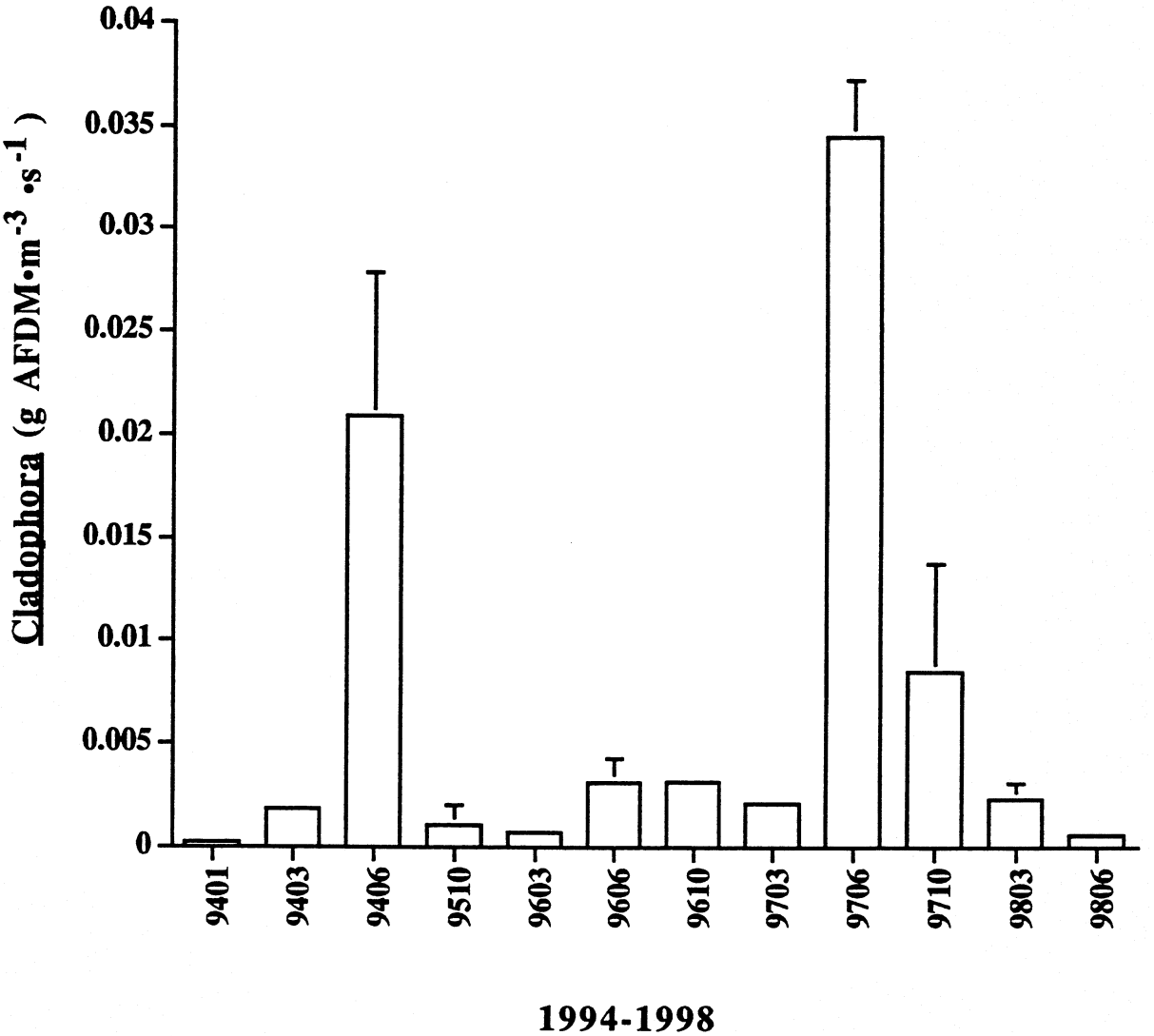

Figure 118. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) of Cladophora collected at Glen Canyon Gauge Rkm -23.2 from January 1994 to June 1998. Error Bars represent (± 1 SE).

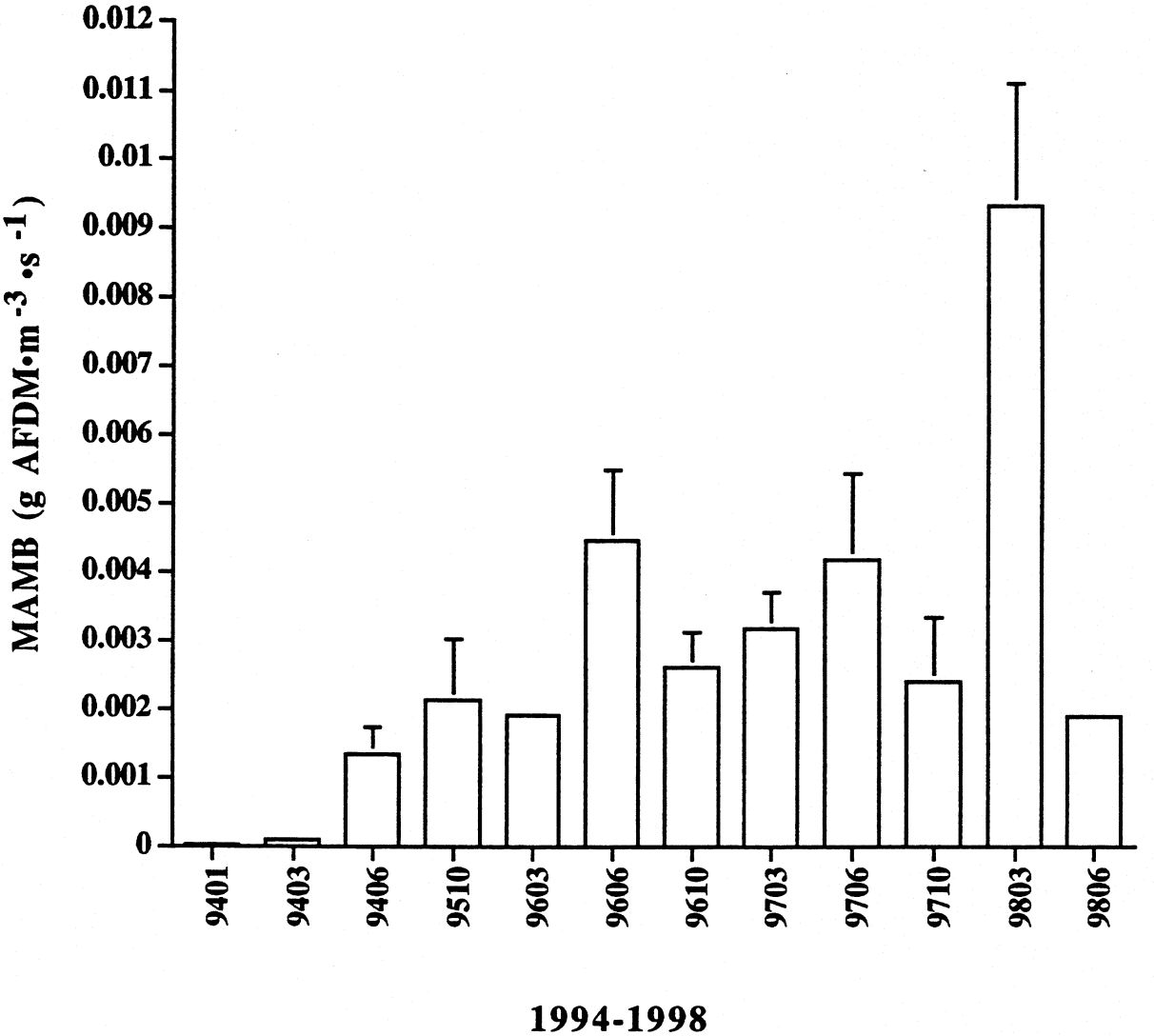

Figure 119. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) of miscellaneous algae, macrophytes, and bryophytes (MAMB) collected at Glen Canyon Gauge Rkm -23.2 from January 1994 to June 1998. Error Bars represent (± 1 SE).

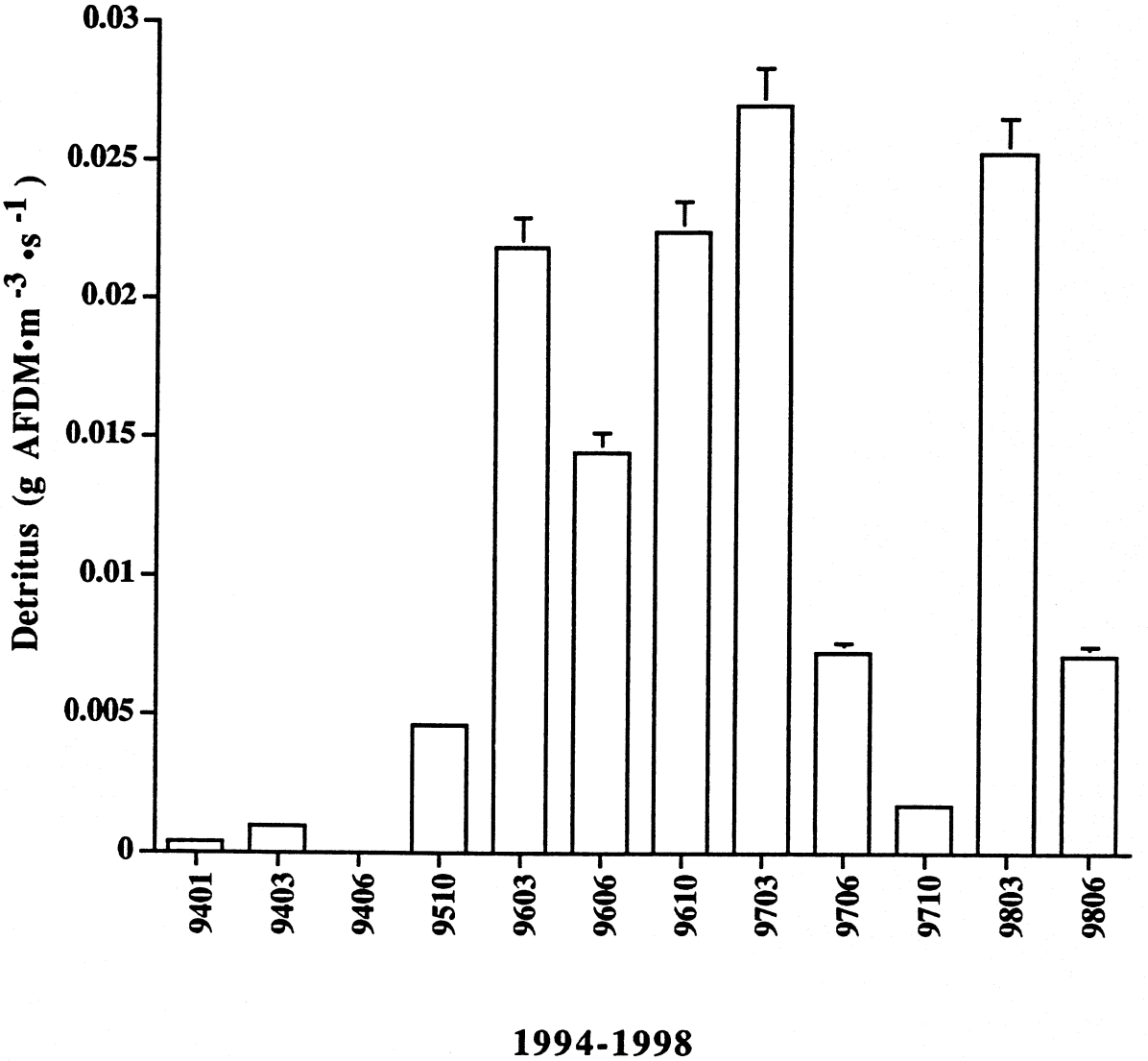

Figure 120. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for detritus collected at Glen Canyon Gauge Rkm -23.2 from January 1994 to June 1998. Error Bars represent (± 1 SE).

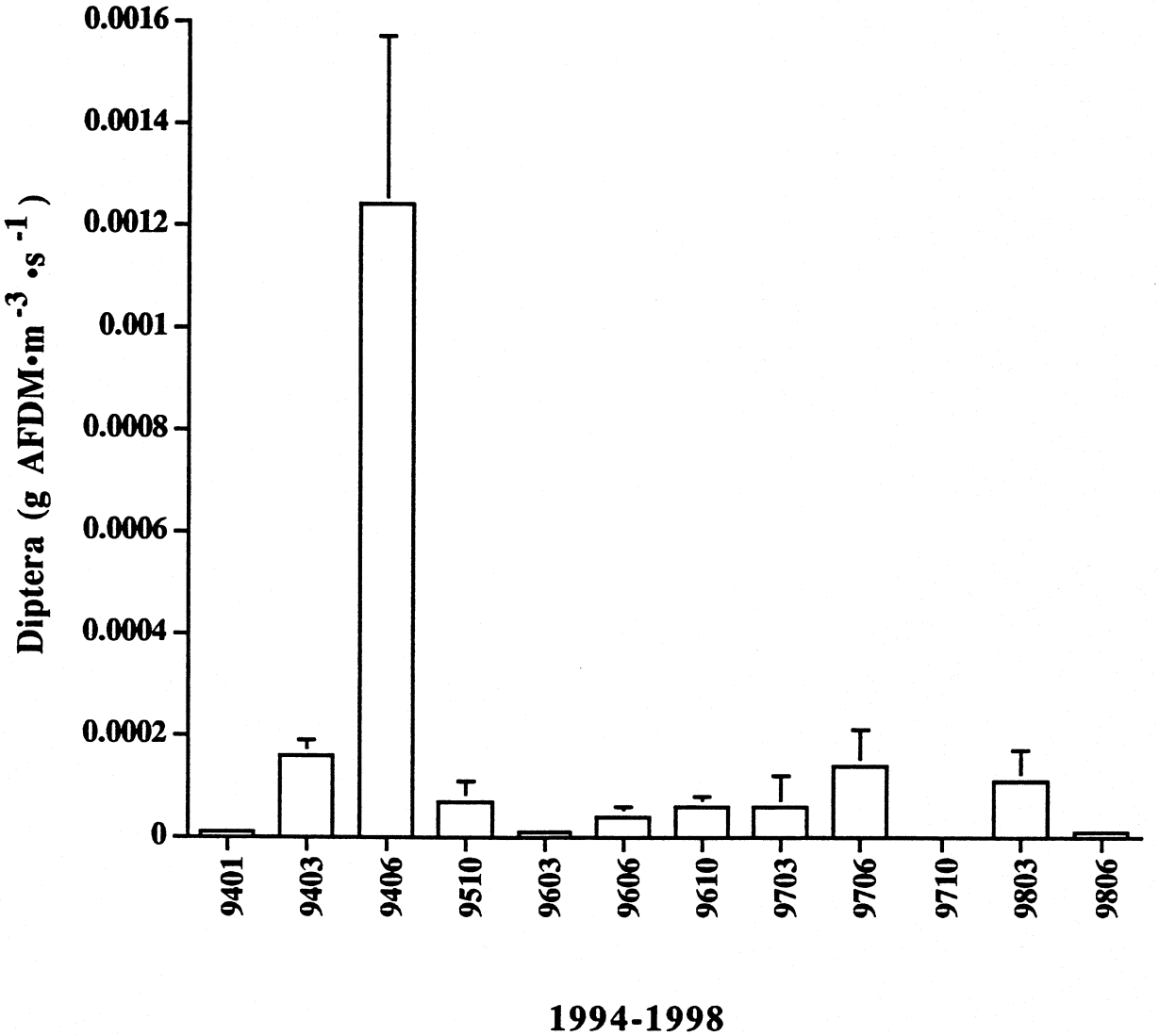

Figure 121. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for diptera collected at Glen Canyon Gauge Rkm -23.2 from January 1994 to June 1998. Error Bars represent (± 1 SE).

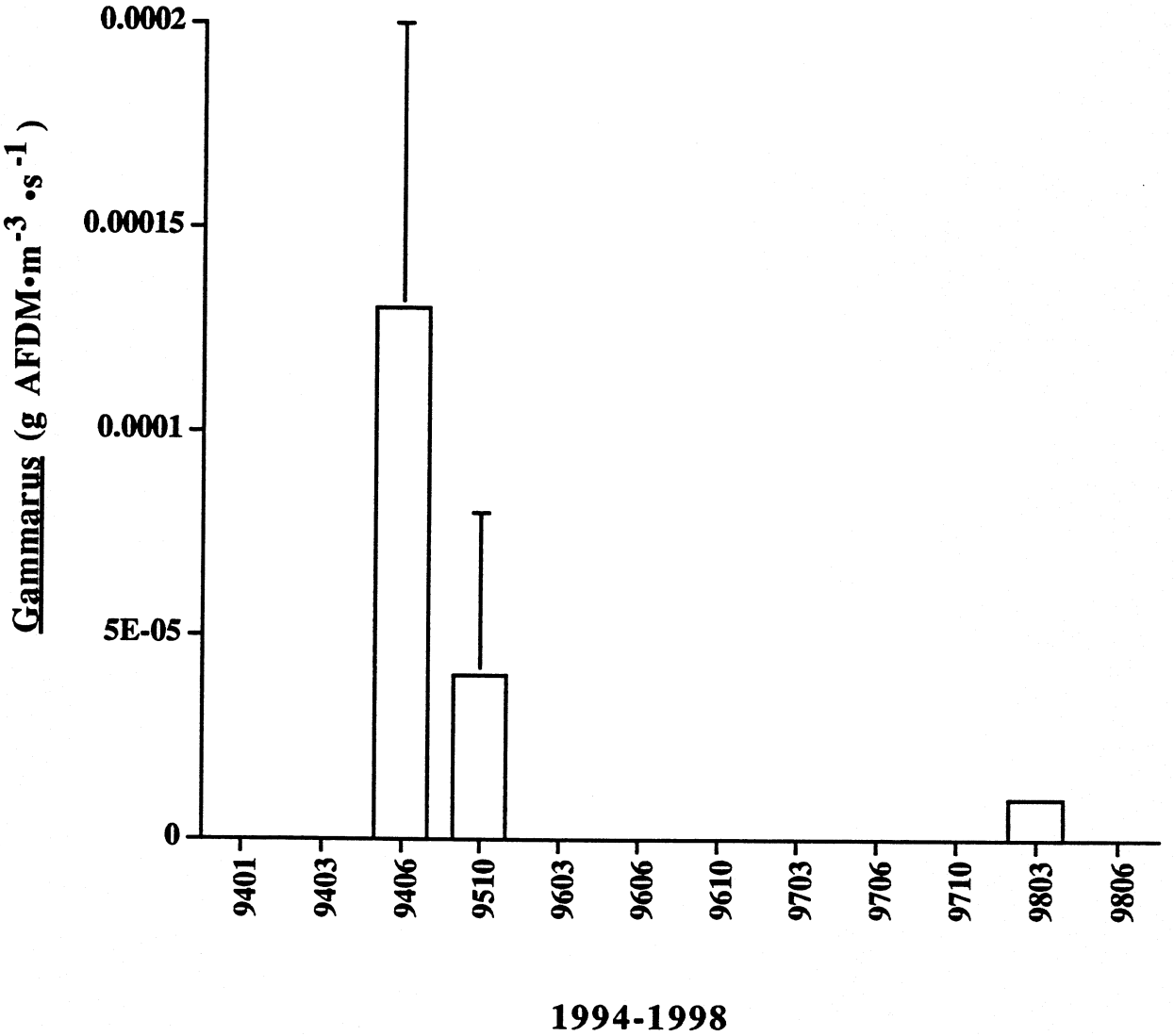

Figure 122. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Gammarus collected at Glen Canyon Gauge Rkm -23.2 from January 1994 to June 1998. Error Bars represent (± 1 SE).

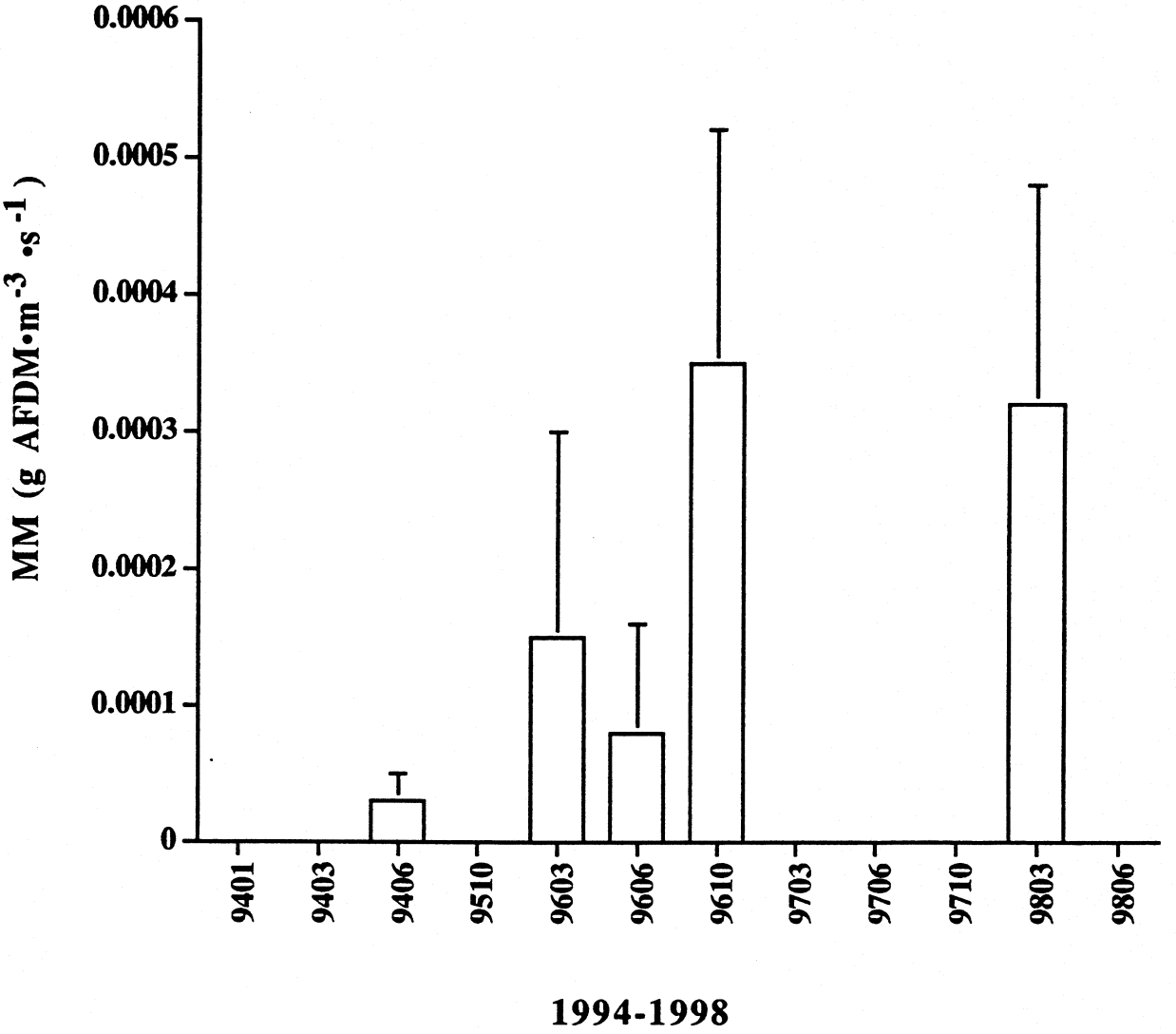

Figure 123. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous macroinvertebrates (MM) collected at Glen Canyon Gauge Rkm -23.2 from January 1994 to June 1998. Error Bars represent (± 1 SE).

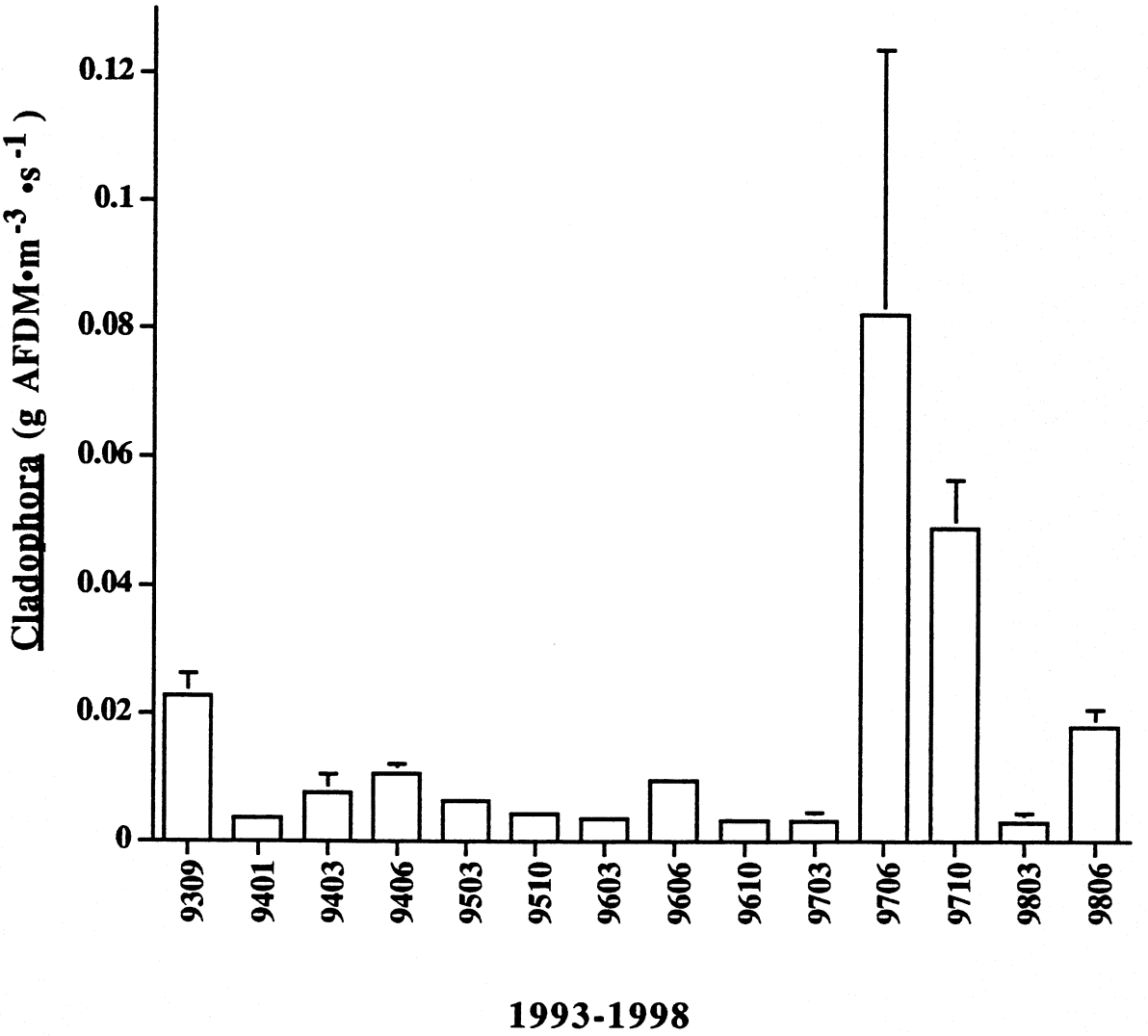

Figure 124. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Cladophora collected at Lees Ferry Rkm 0.0 from September 1993 to June 1998. Error Bars represent (± 1 SE).

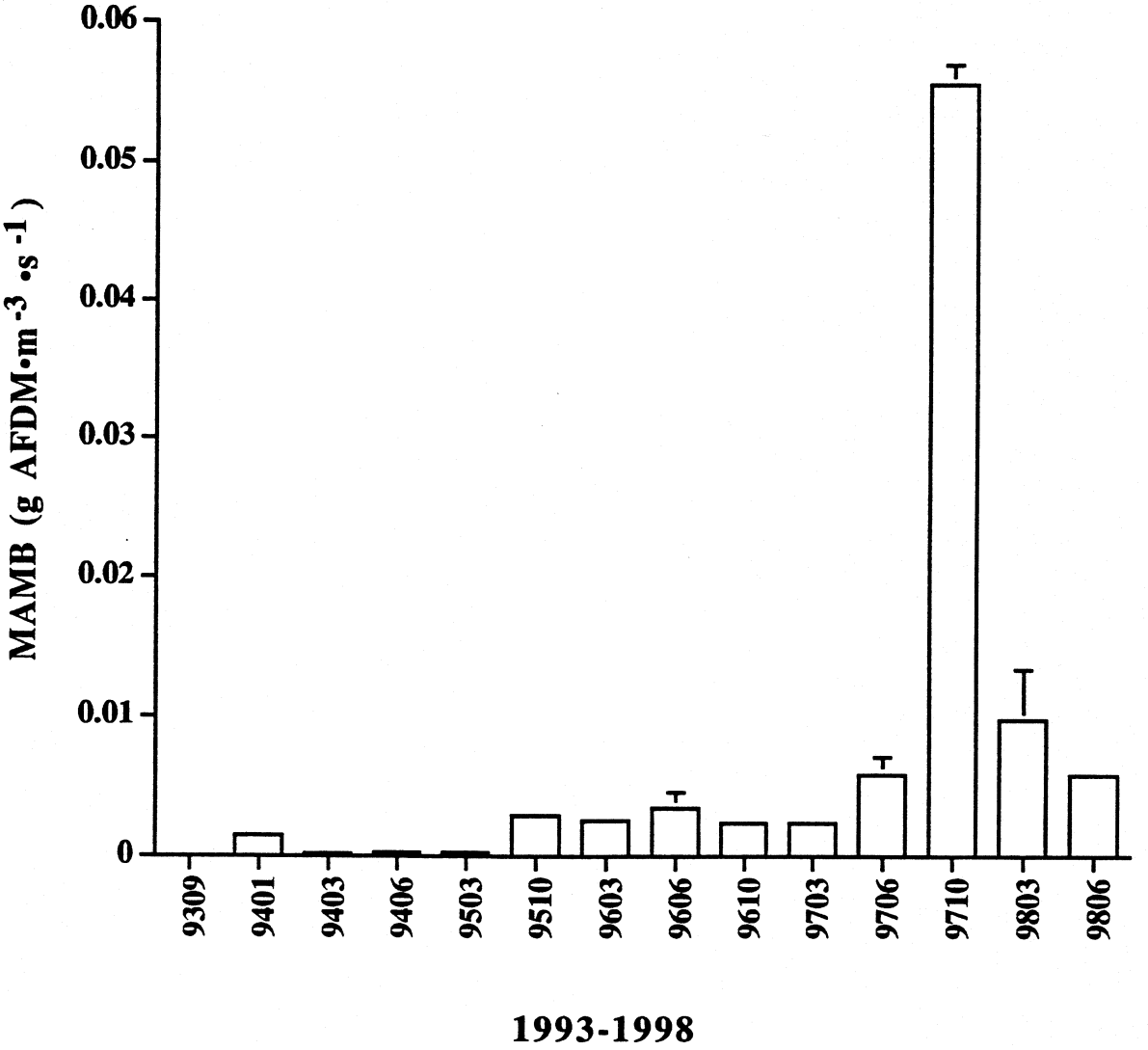

Figure 125. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous algae, macrophytes and bryophytes (MAMB) collected at Lees Ferry Rkm 0.0 from September 1993 to June 1998. Error Bars represent (± 1 SE).

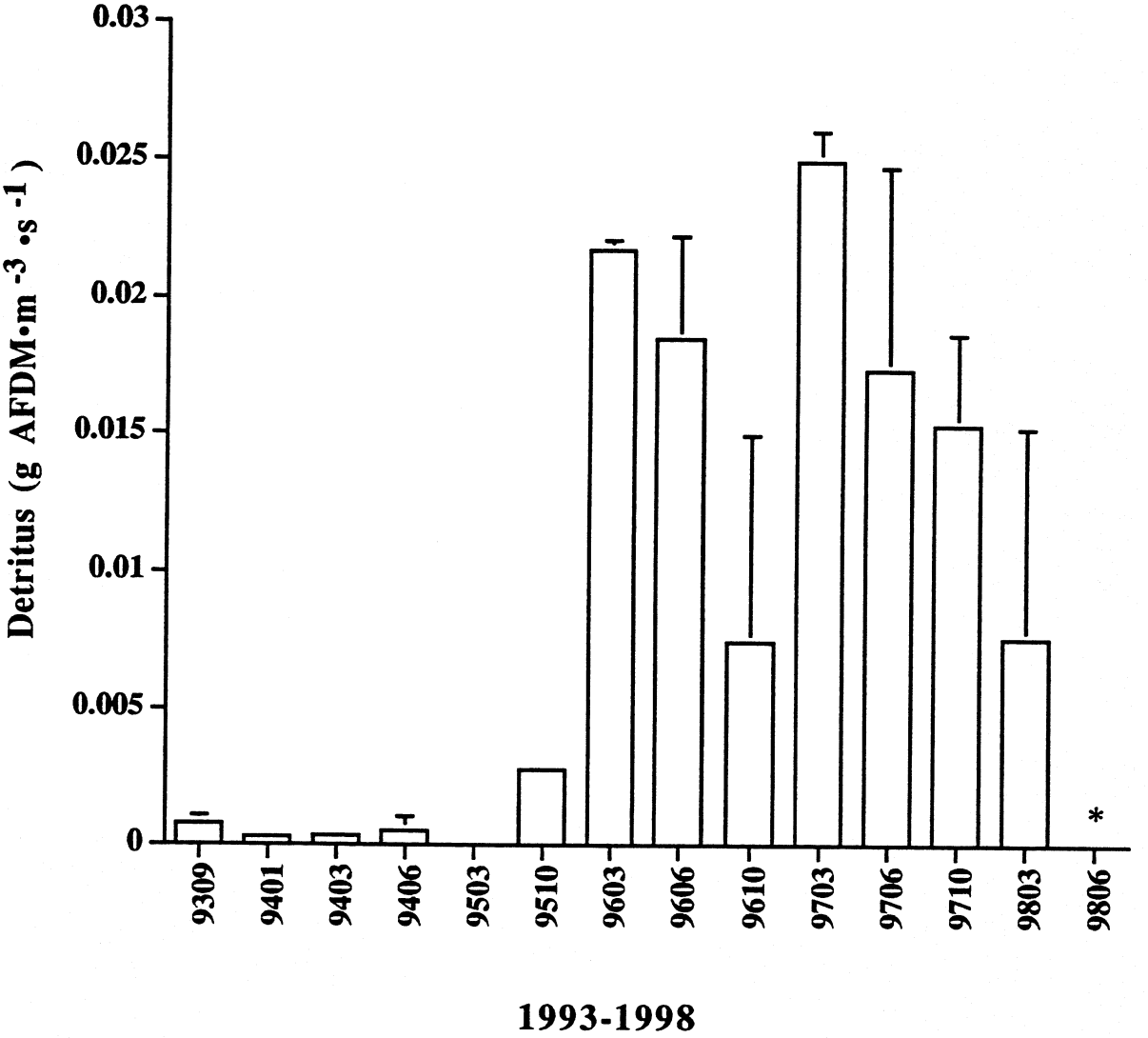

Figure 126. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for detritus collected at Lees Ferry Rkm 0.0 from September 1993 to June 1998. Error Bars represent (± 1 SE).
* represents 0.15 g AFDM•m^{-3}•s^{-1} (± 0.12).

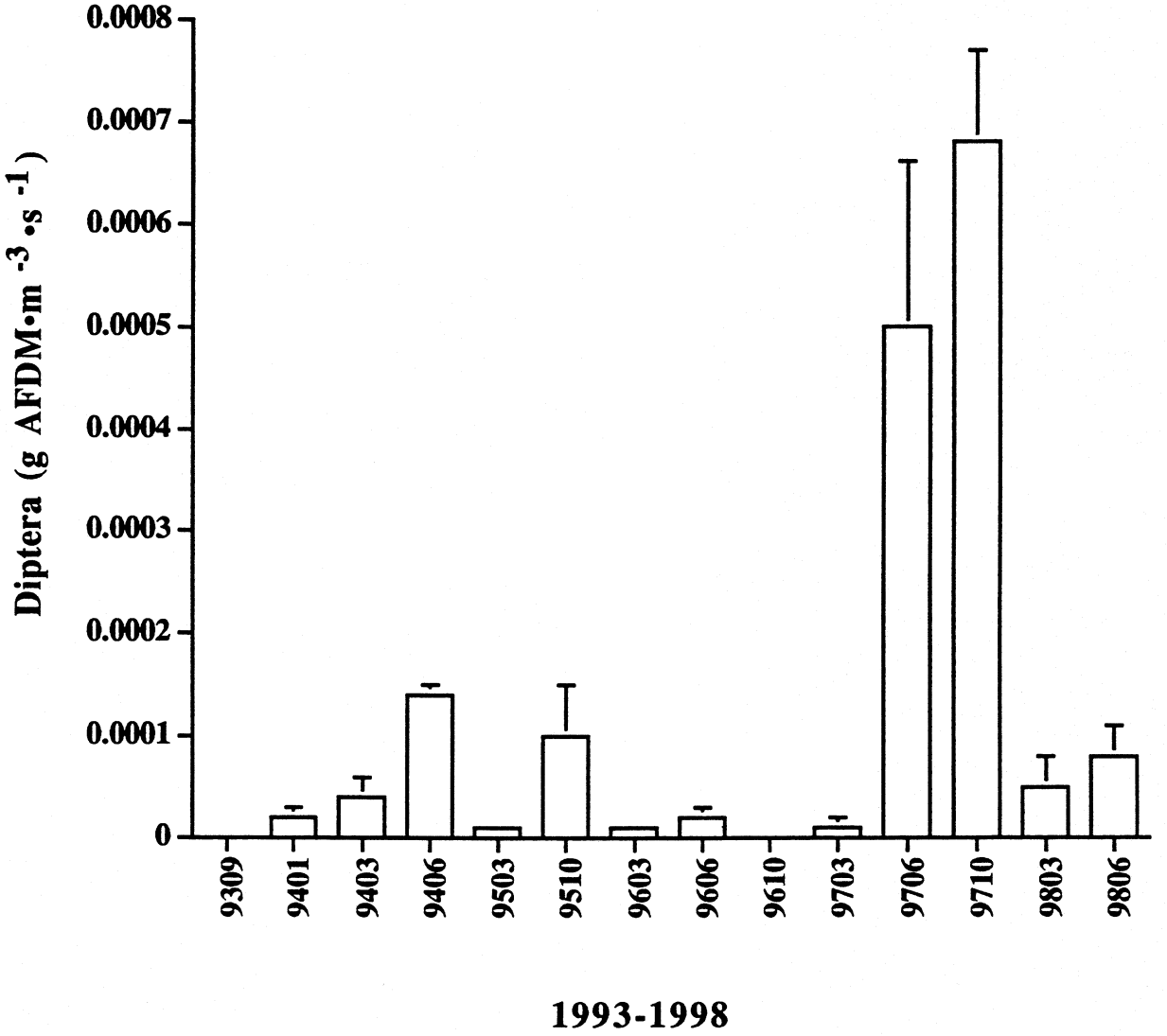

Figure 127. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for diptera collected at Lees Ferry Rkm 0.0 from September 1993 to June 1998. Error Bars represent (± 1 SE).

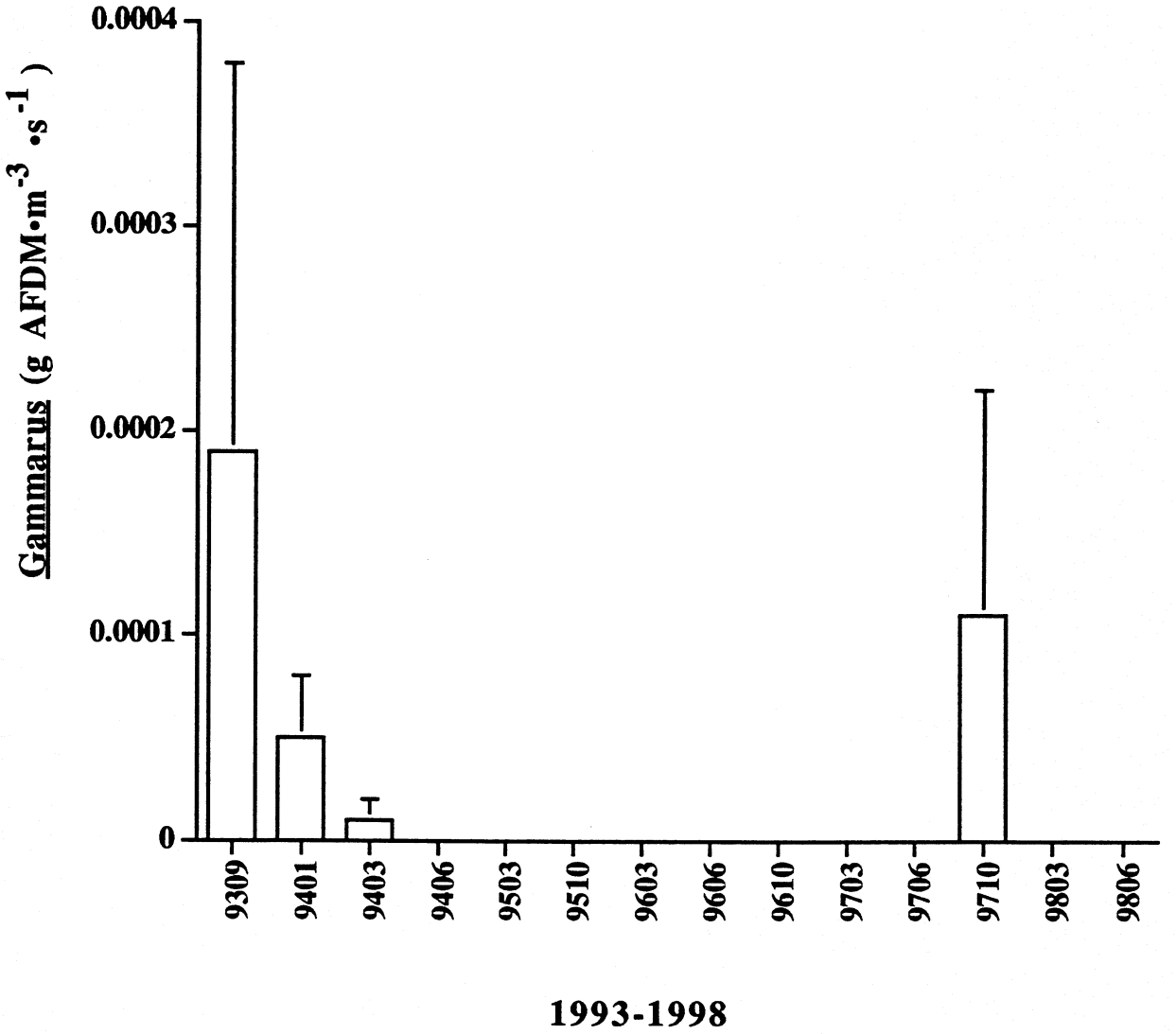

Figure 128. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Gammarus collected at Lees Ferry Rkm 0.0 from September 1993 to June 1998. Error Bars represent (± 1 SE).

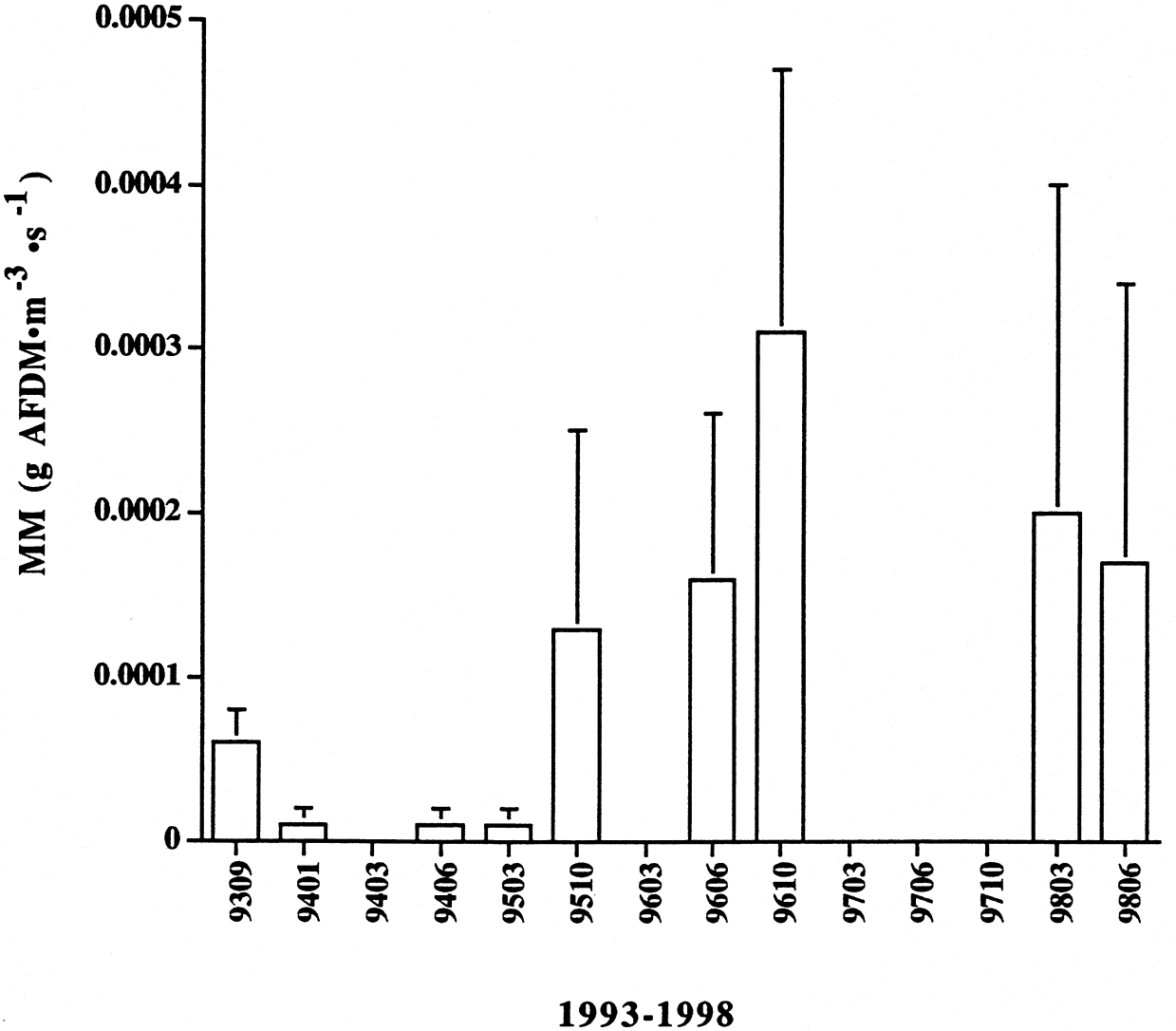

Figure 129. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous macroinvertebrates (MM) collected at Lees Ferry Rkm 0.0 from September 1993 to June 1998. Error Bars represent (± 1 SE).

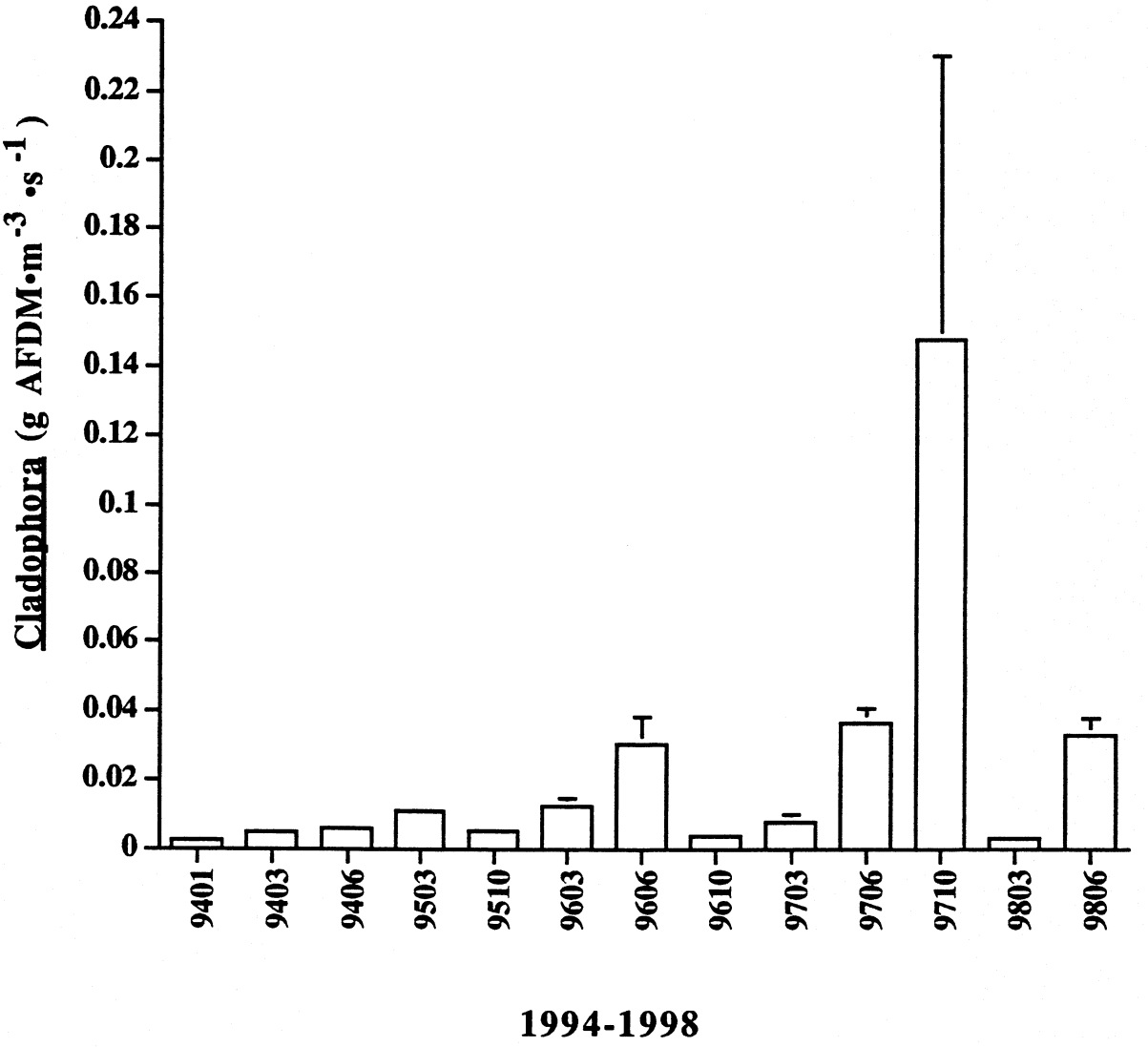

Figure 130. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Cladophora collected at Two-Mile Wash Rkm 2.9 from January 1994 to June 1998. Error Bars represent (± 1 SE).

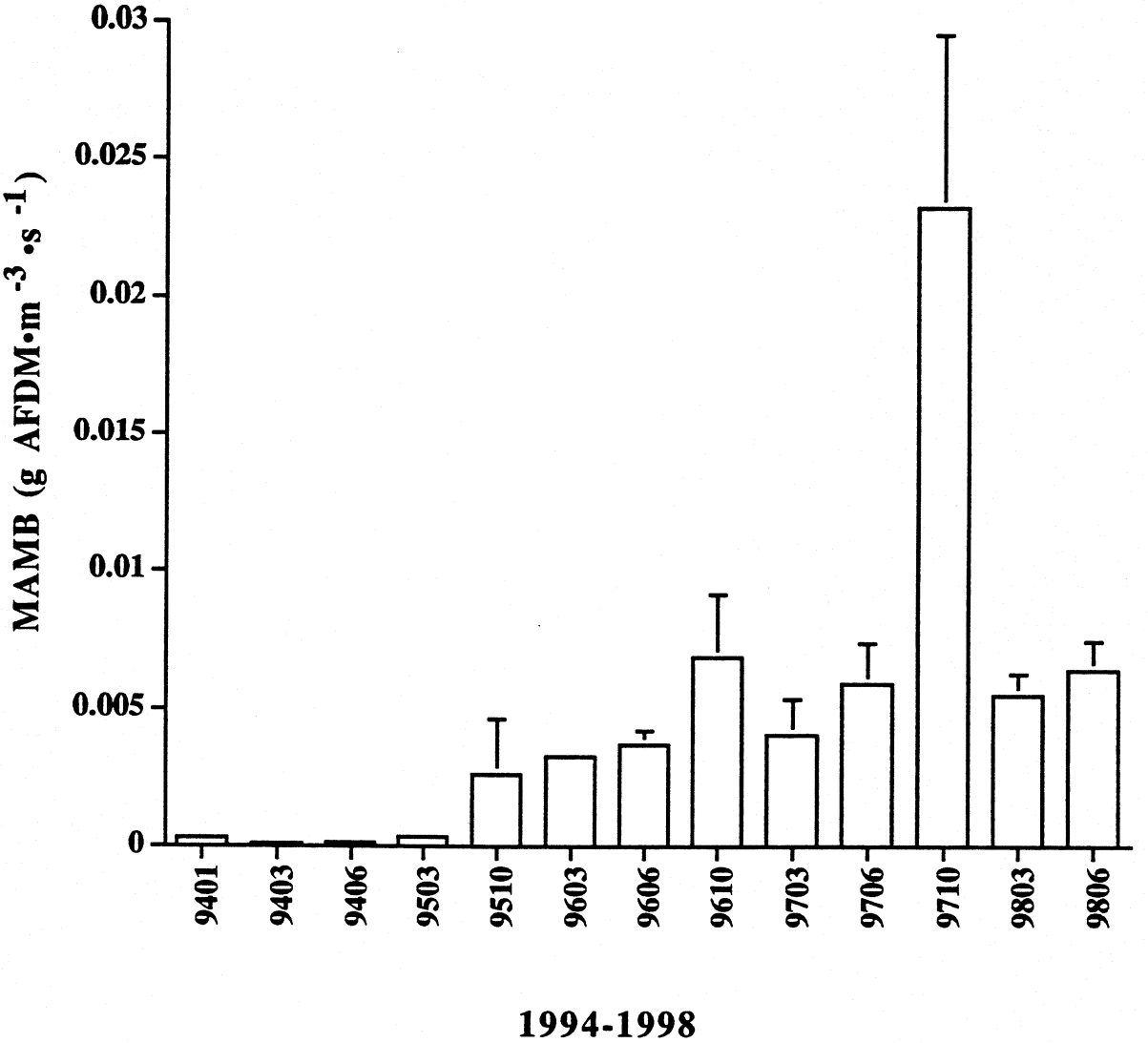

Figure 131. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous algae, macrophytes and bryophytes (MAMB) collected at Two-Mile Wash Rkm 2.9 from January 1994 to June 1998. Error Bars represent (± 1 SE).

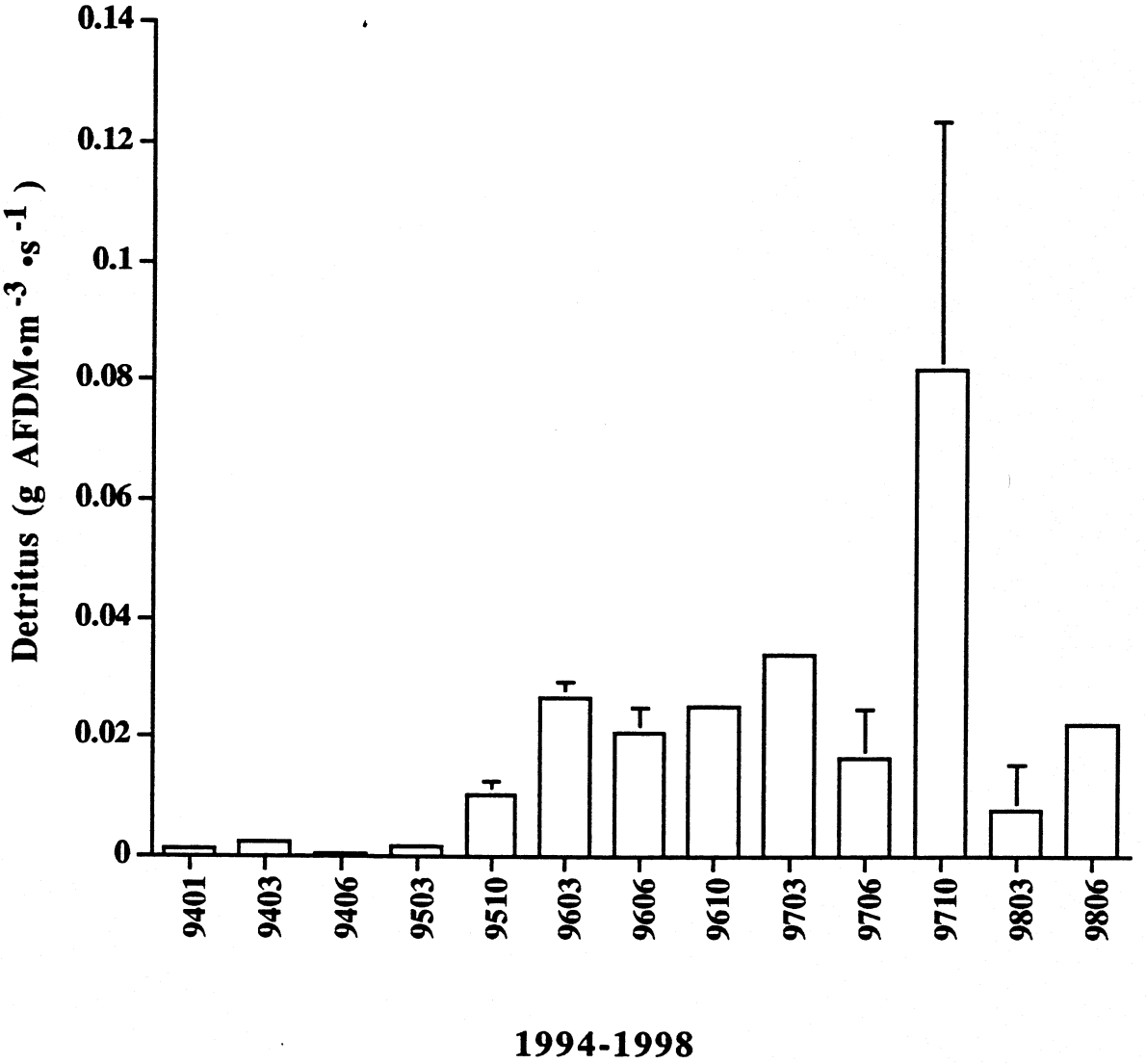

Figure 132. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for detritus collected at Two-Mile Wash Rkm 2.9 from January 1994 to June 1998. Error Bars represent (± 1 SE).

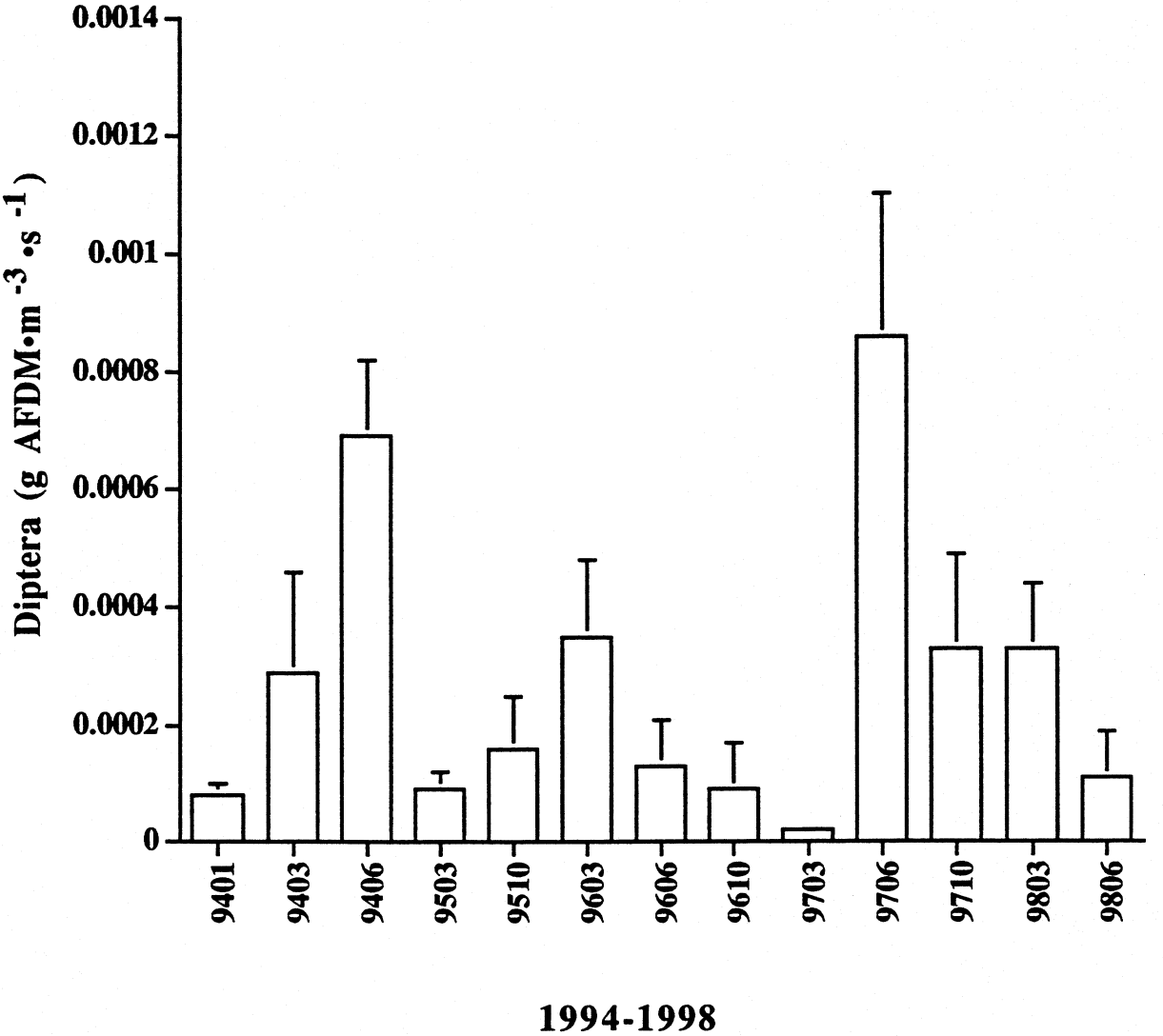

Figure 133. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for diptera collected at Two-Mile Wash Rkm 2.9 from January 1994 to June 1998. Error Bars represent (± 1 SE).

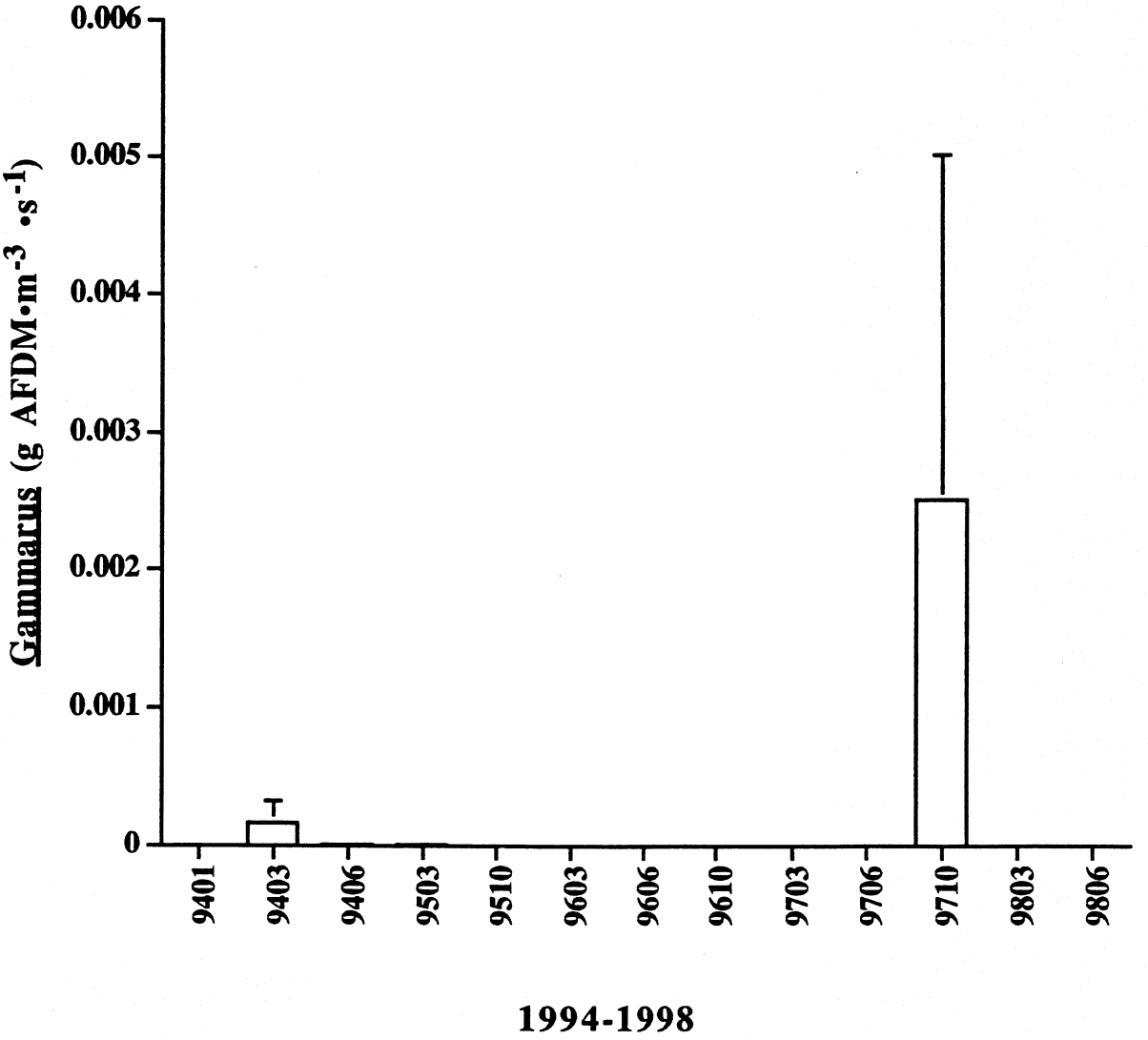

Figure 134. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Gammarus collected at Two-Mile Wash Rkm 2.9 from January 1994 to June 1998. Error Bars represent (± 1 SE).

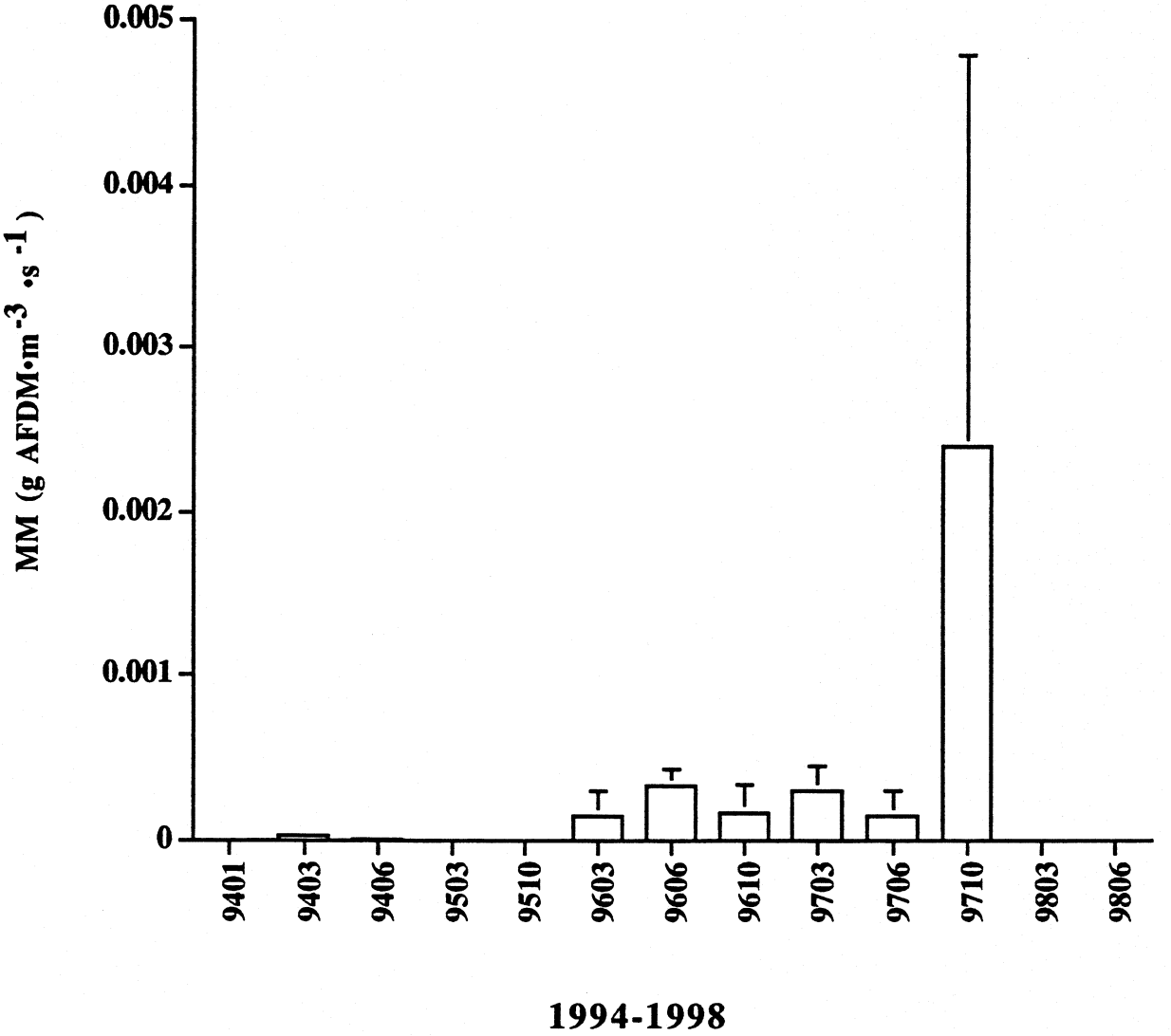

Figure 135. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous macroinvertebrates (MM) collected at Two-Mile Wash Rkm 2.9 from January 1994 to June 1998. Error Bars represent (± 1 SE).

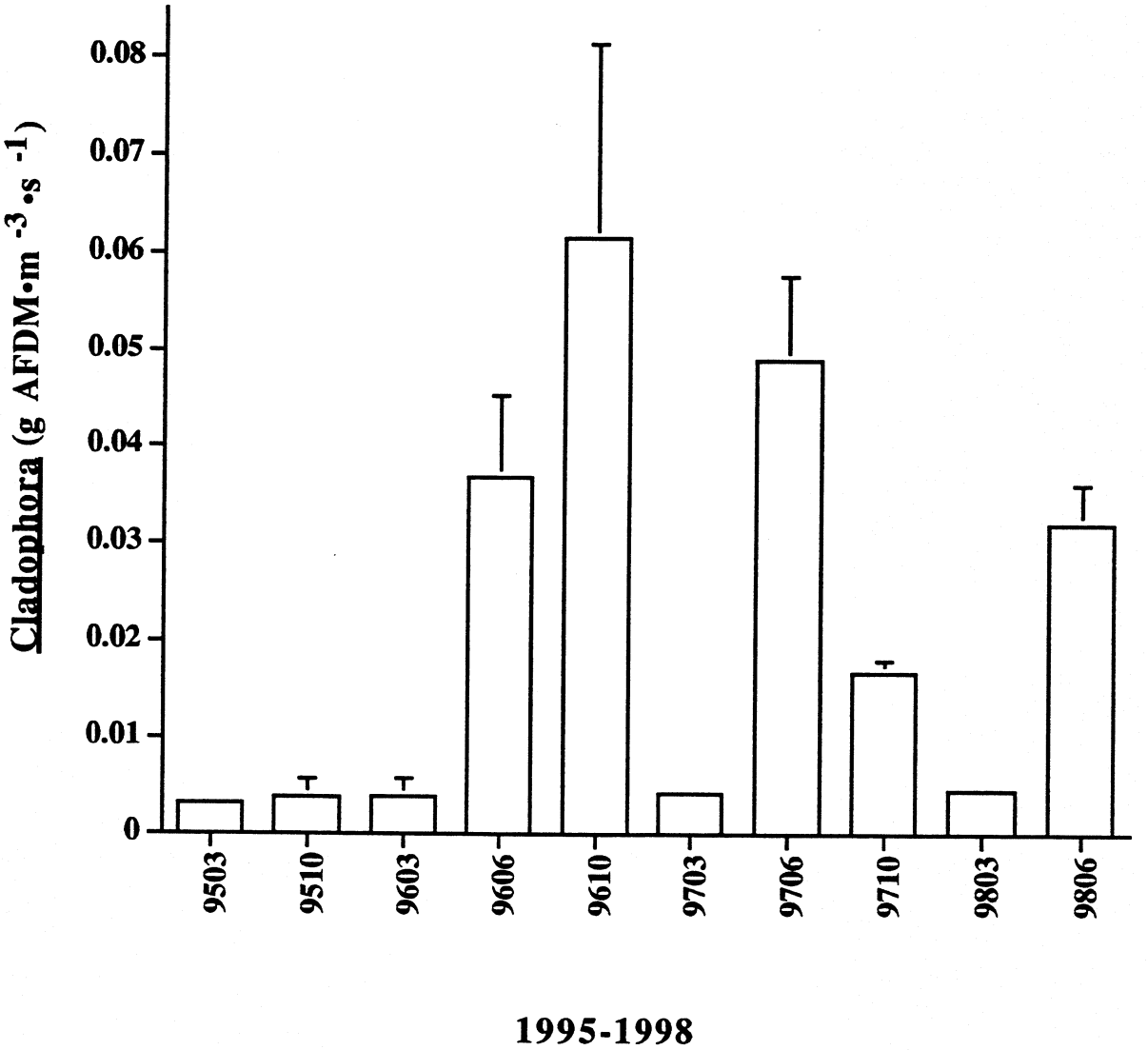

Figure 136. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Cladophora collected at the Guage above LCR Rkm 98.4 from March 1995 to June 1998. Error Bars represent (± 1 SE).

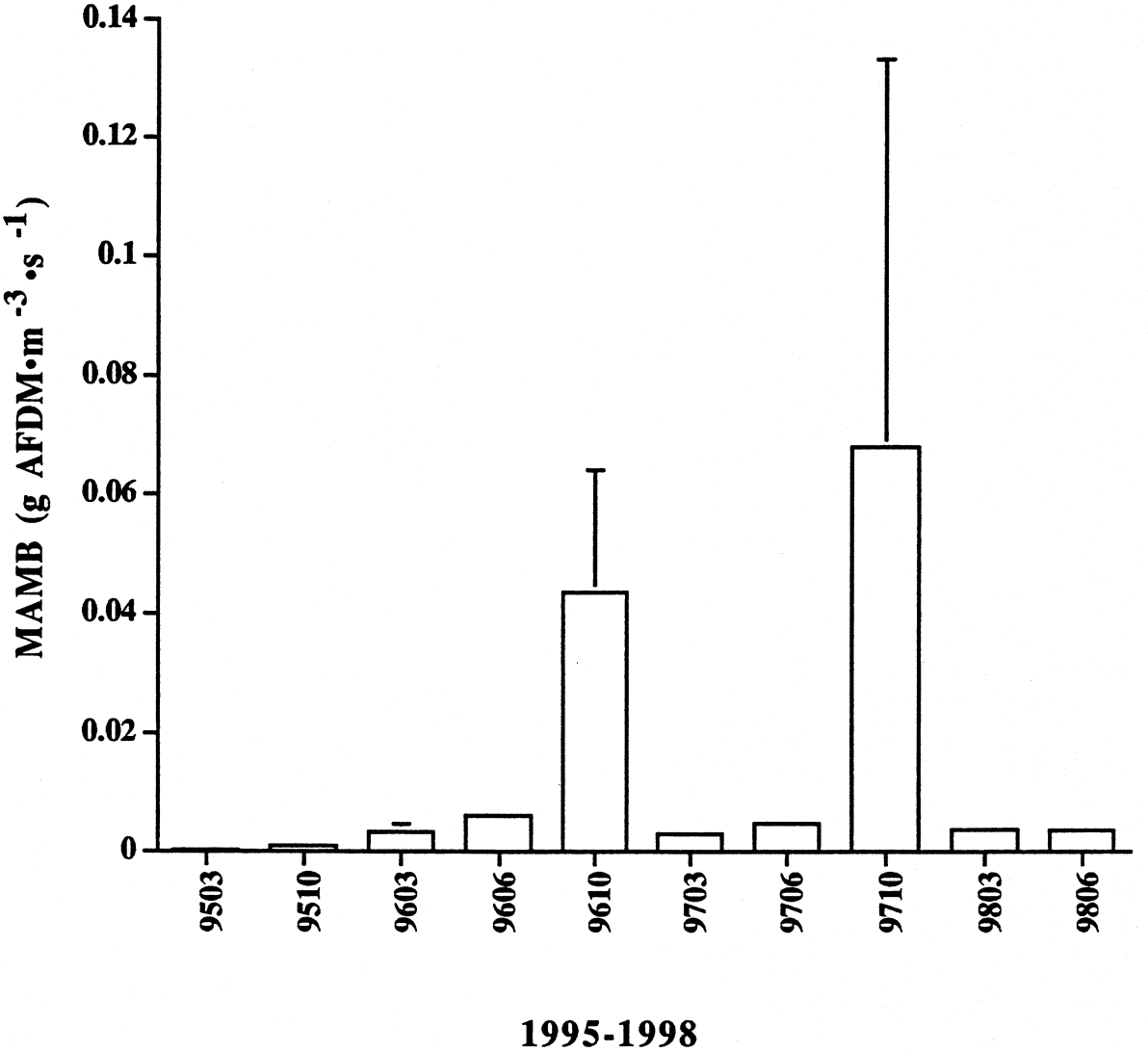

Figure 137. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous algae, macrophytes and bryophytes (MAMB) collected at Guage above LCR Rkm 98.4 from March 1995 to June 1998. Error Bars represent (± 1 SE).

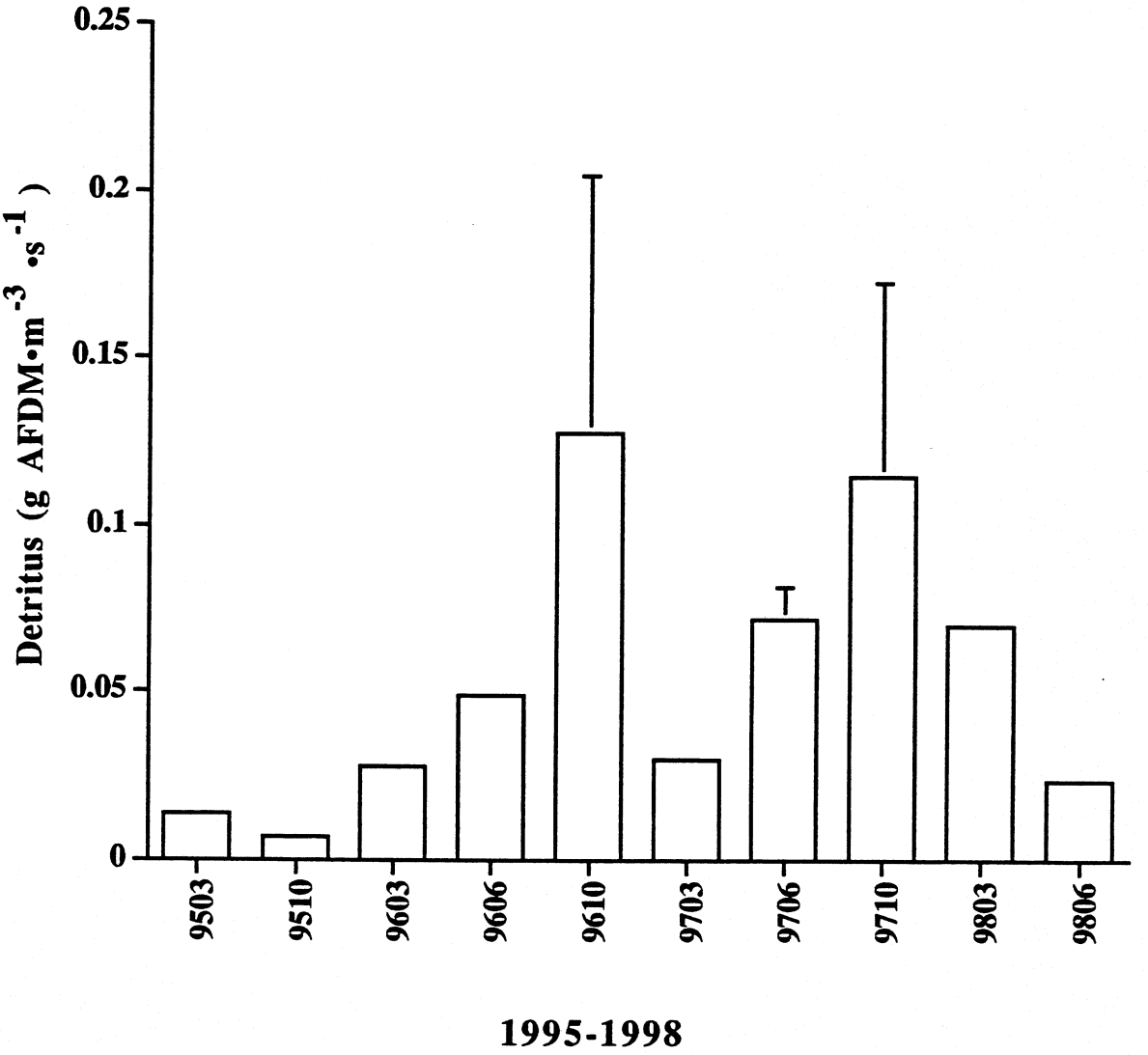

Figure 138. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for detritus collected at the Gauge above LCR Rkm 98.4 from March 1995 to June 1998. Error Bars represent (± 1 SE).

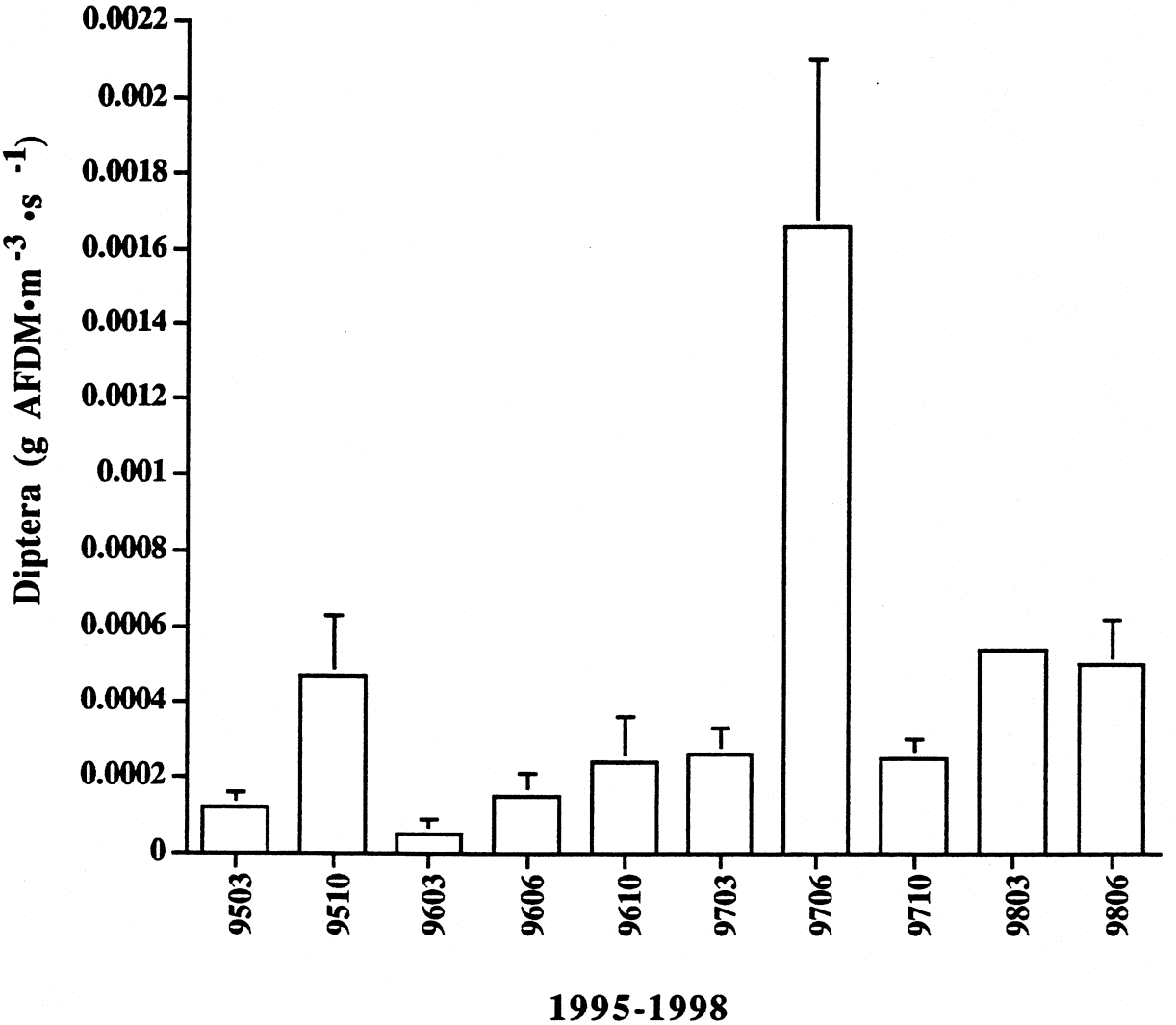

Figure 139. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for diptera collected at the Gauge above LCR Rkm 98.4 from March 1995 to June 1998. Error Bars represent (± 1 SE).

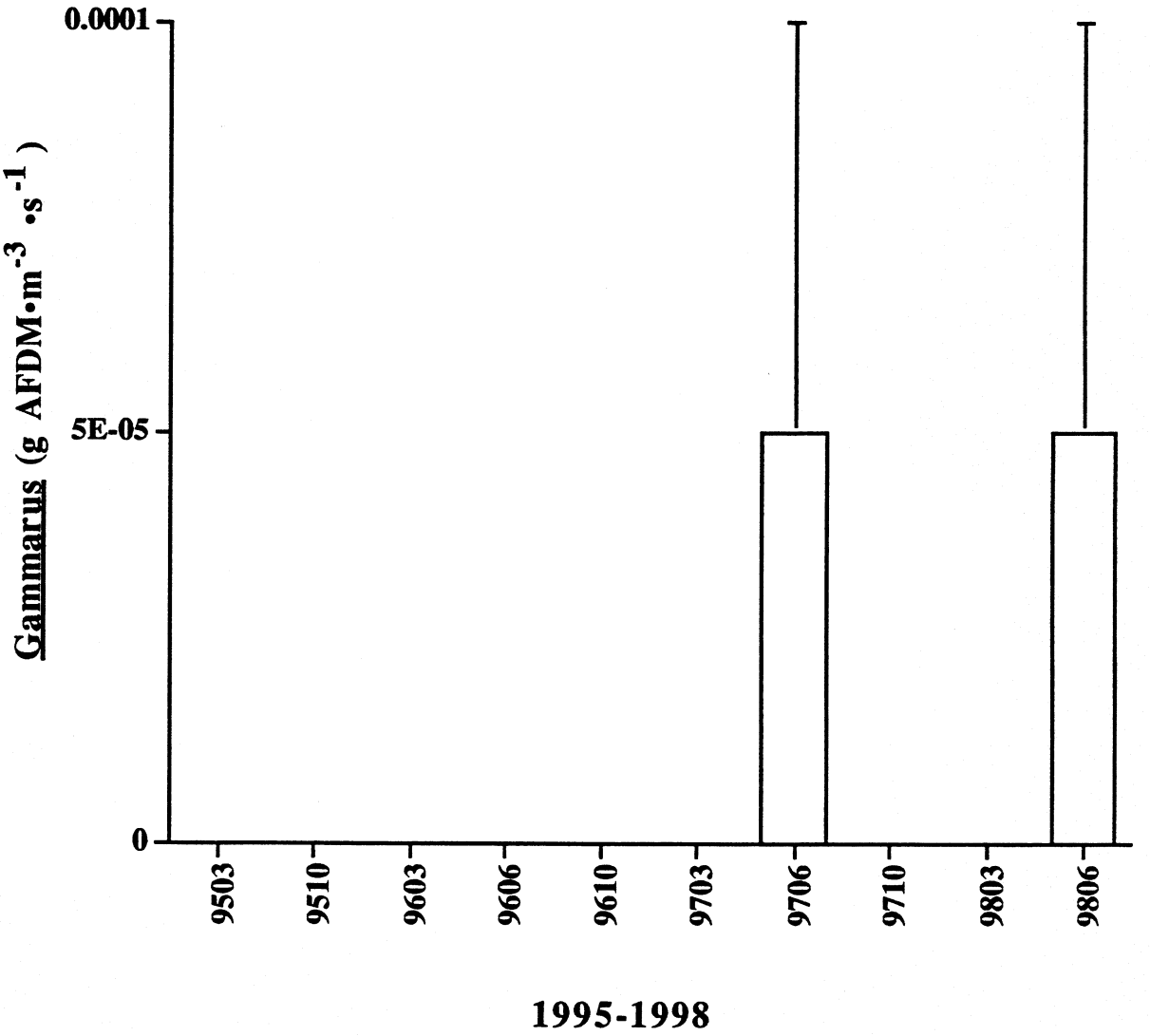

Figure 140. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Gammarus collected at the Gauge above LCR Rkm 98.4 from March 1995 to June 1998. Error Bars represent (± 1 SE).

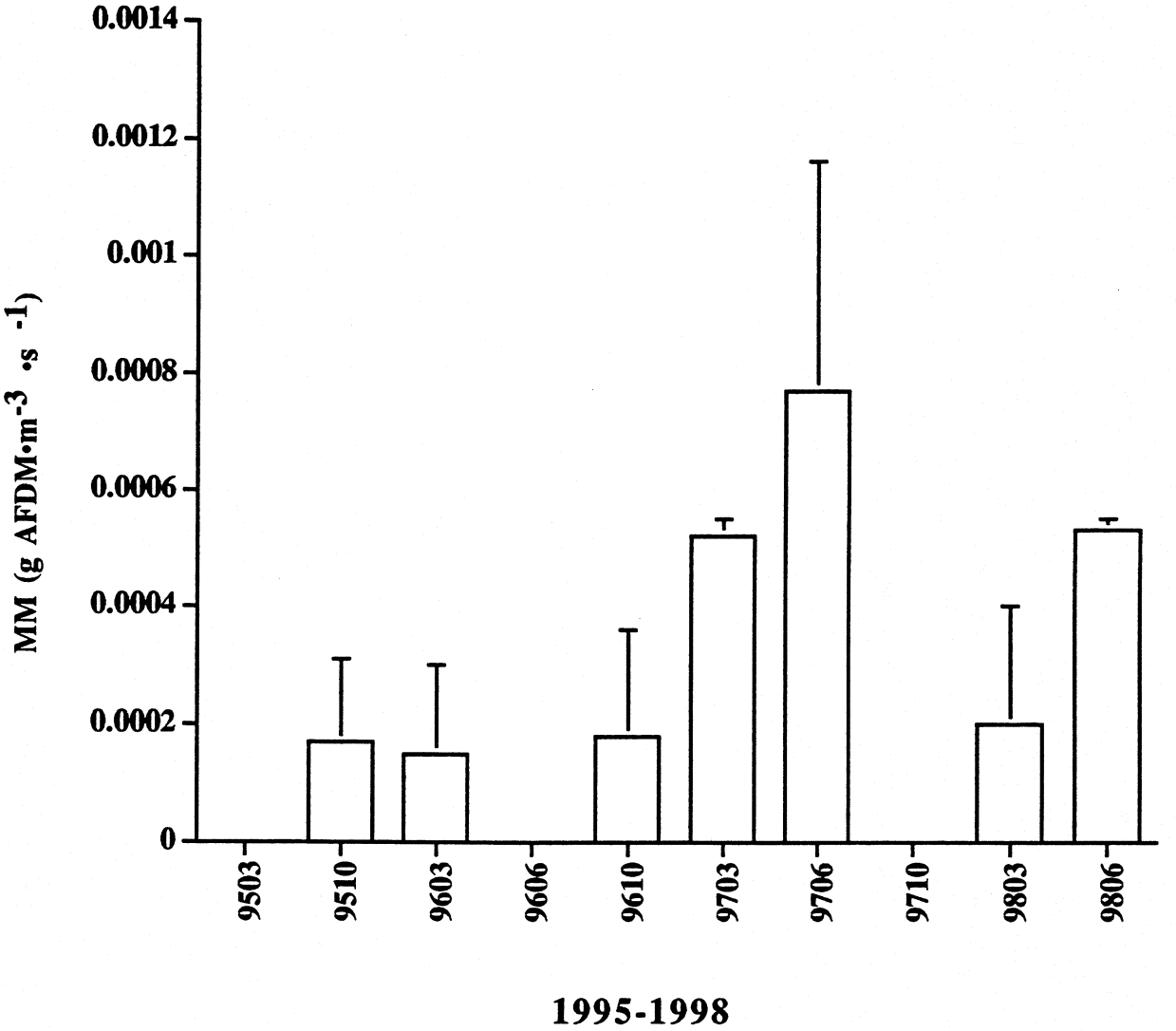

Figure 141. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous macroinvertebrates (MM) collected at the Gauge above LCR Rkm 98.4 from March 1995 to June 1998. Error Bars represent (± 1 SE).

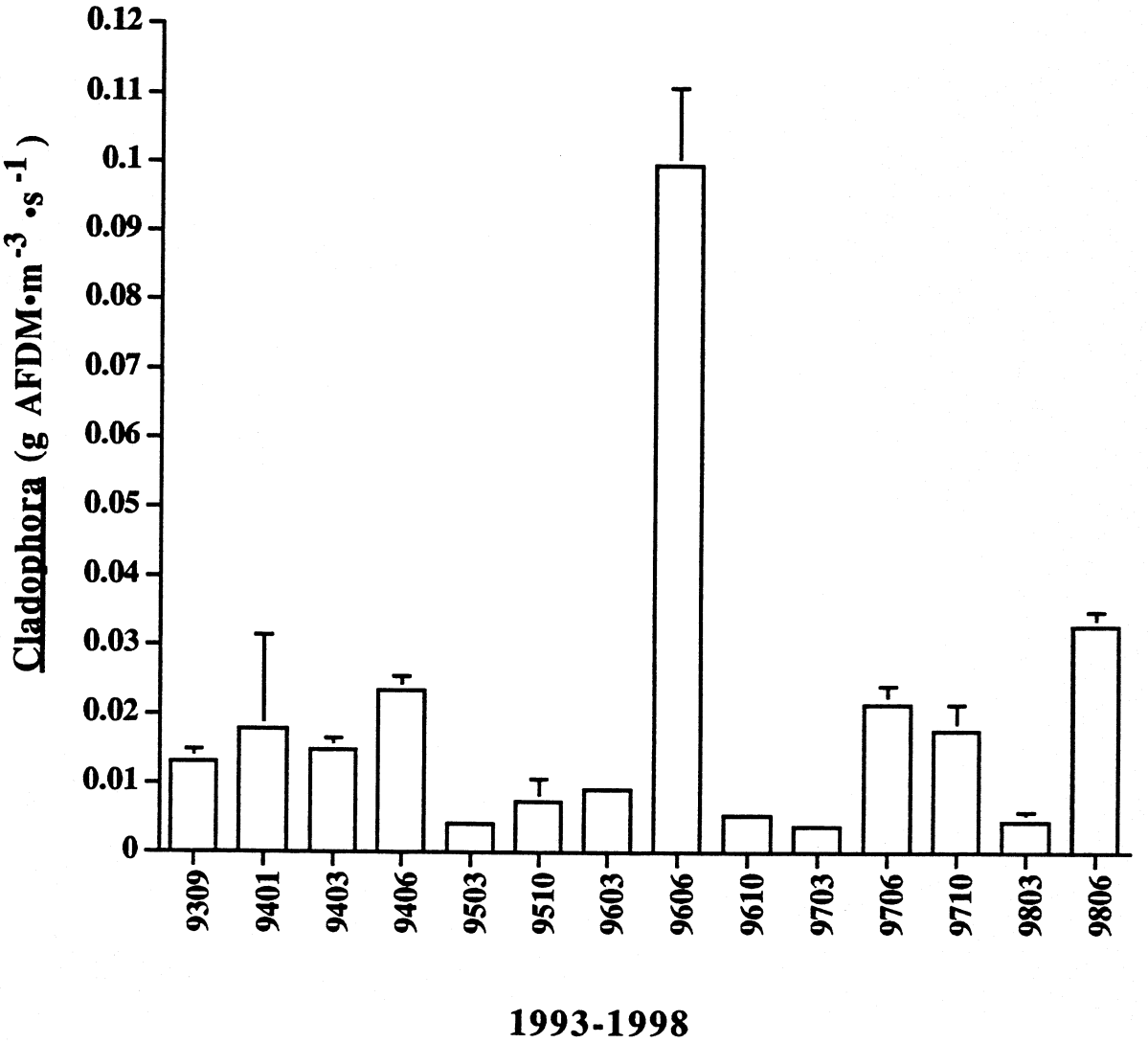

Figure 142. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Cladophora collected at Tanner cobble Rkm 109.6 from September 1993 to June 1998. Error Bars represent (± 1 SE).

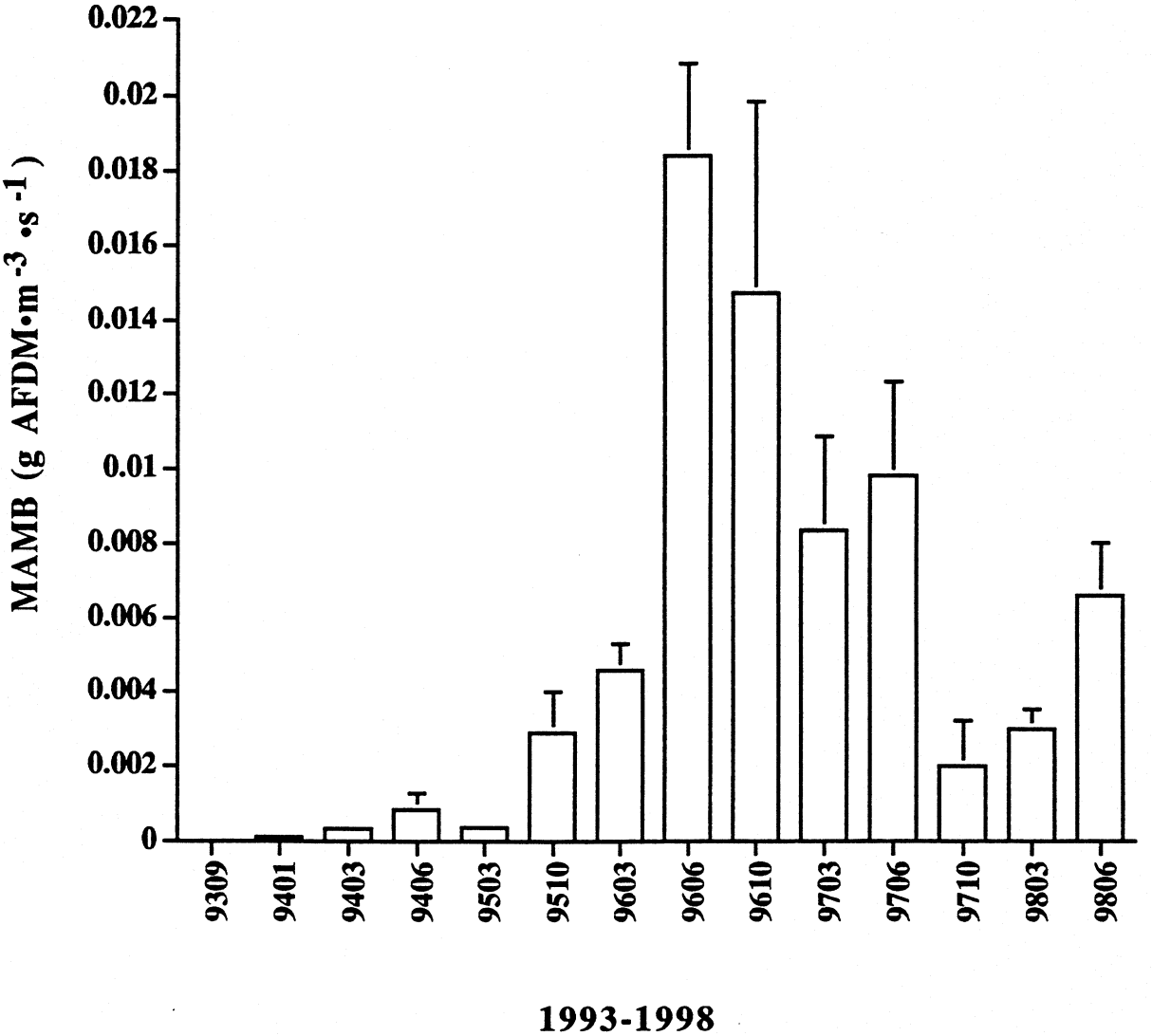

Figure 143. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous algae, macrophytes and bryophytes (MAMB) collected at Tanner cobble Rkm 109.6 from September 1993 to June 1998. Error Bars represent (± 1 SE).

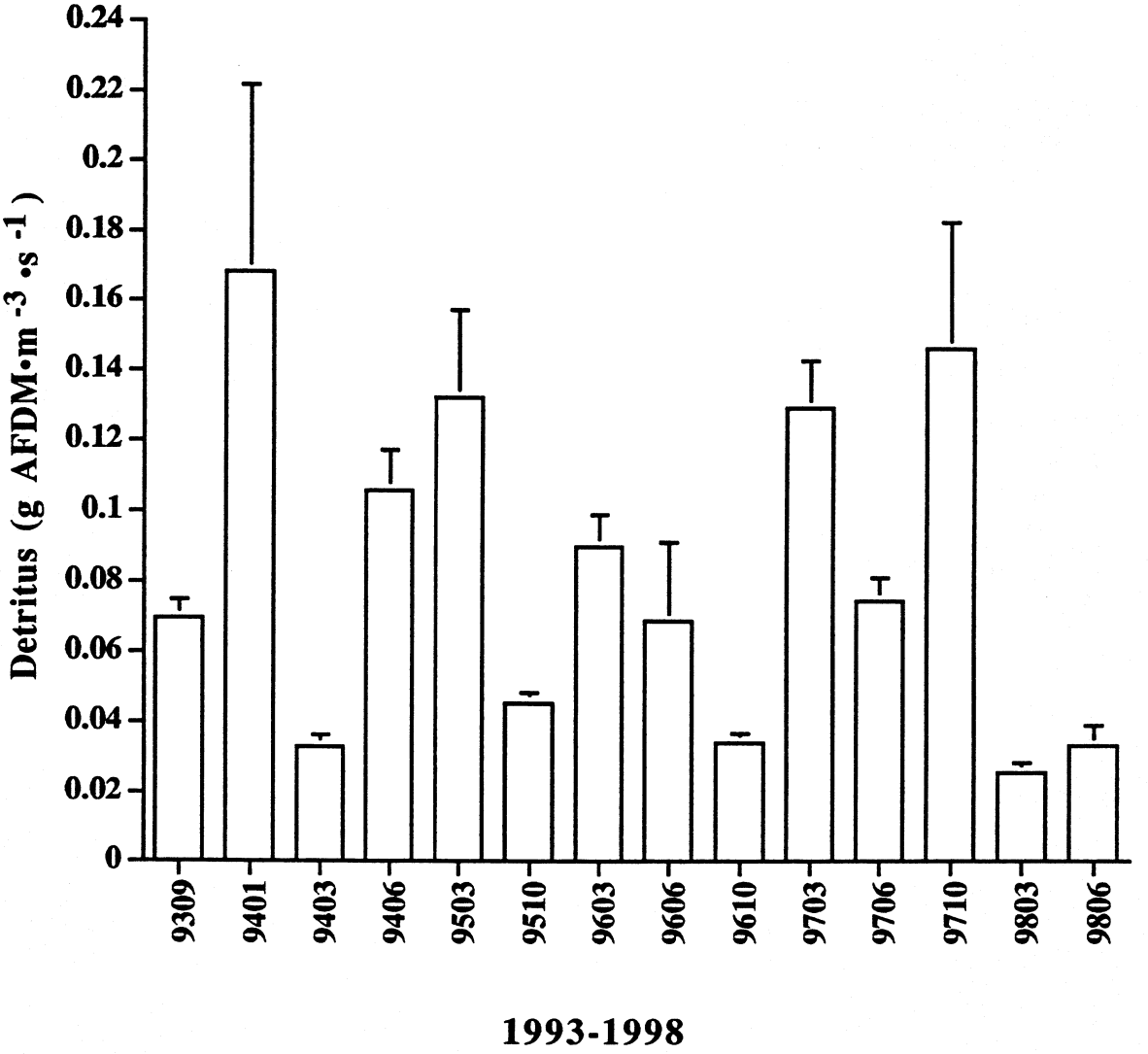

Figure 144. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for detritus collected at Tanner cobble Rkm 109.6 from September 1993 to June 1998. Error Bars represent (± 1 SE).

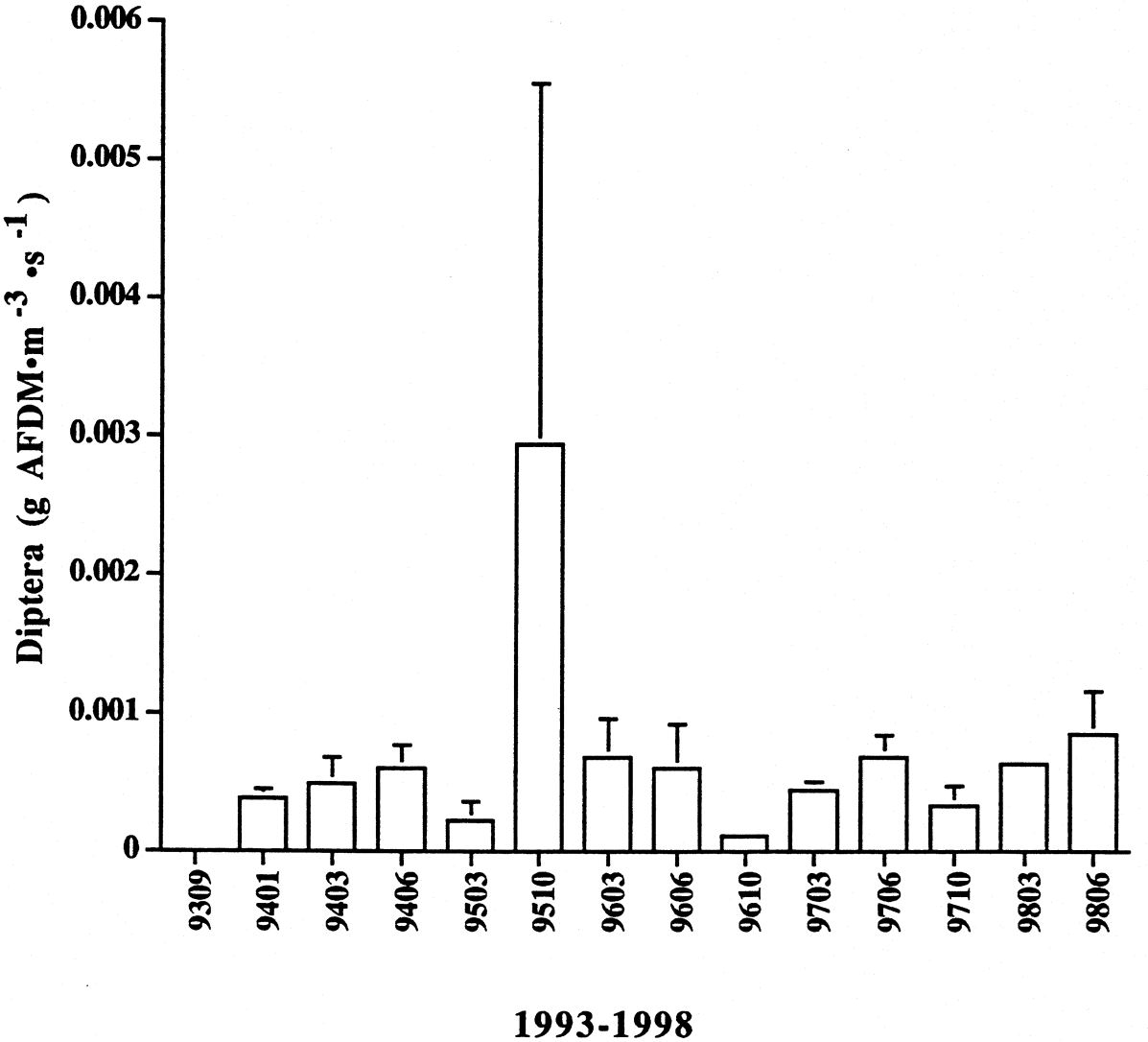

Figure 145. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for diptera collected at Tanner cobble Rkm 109.6 from September 1993 to June 1998. Error Bars represent (± 1 SE).

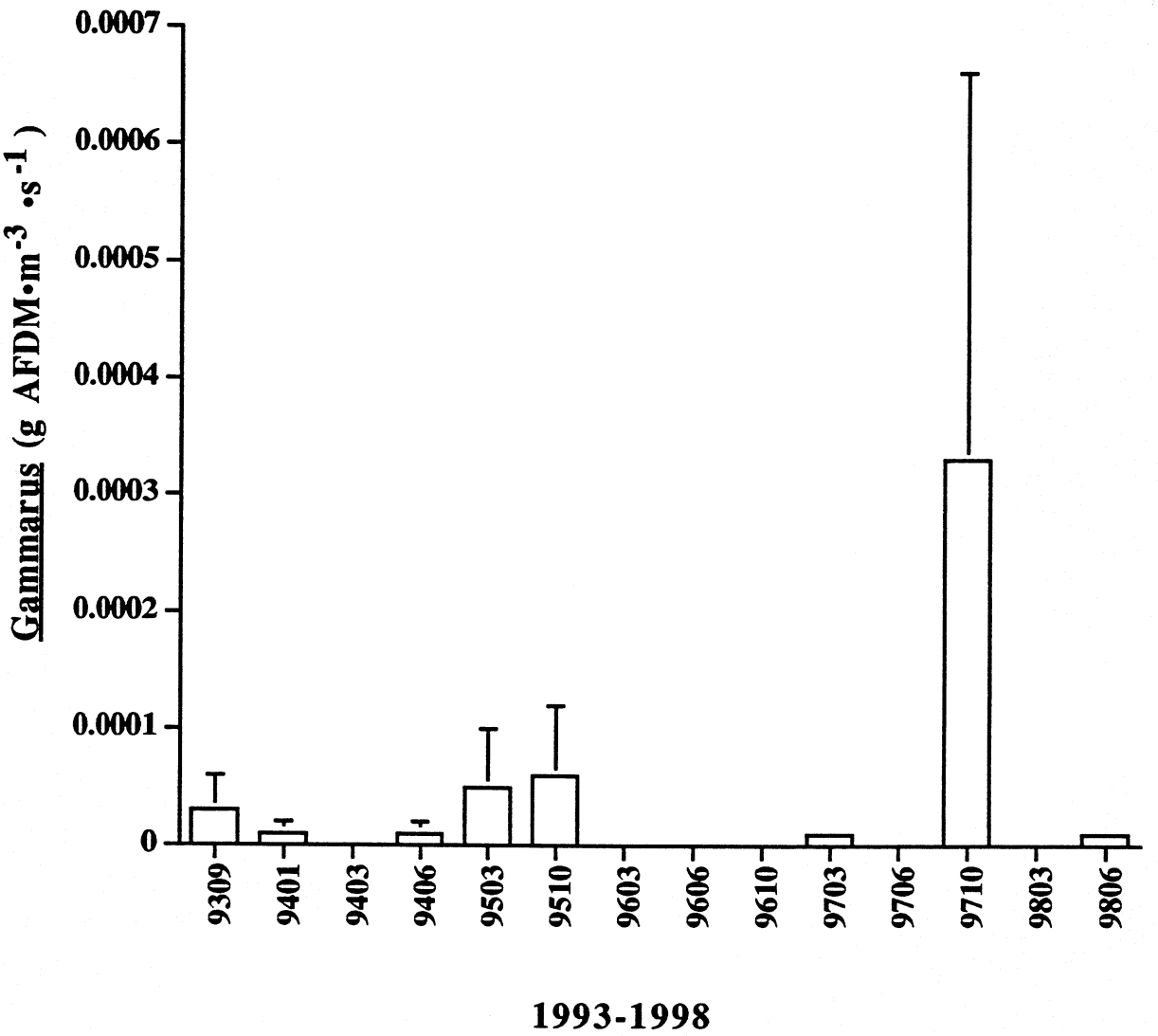

Figure 146. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Gammarus collected at Tanner cobble Rkm 109.6 from September 1993 to June 1998. Error Bars represent (± 1 SE).

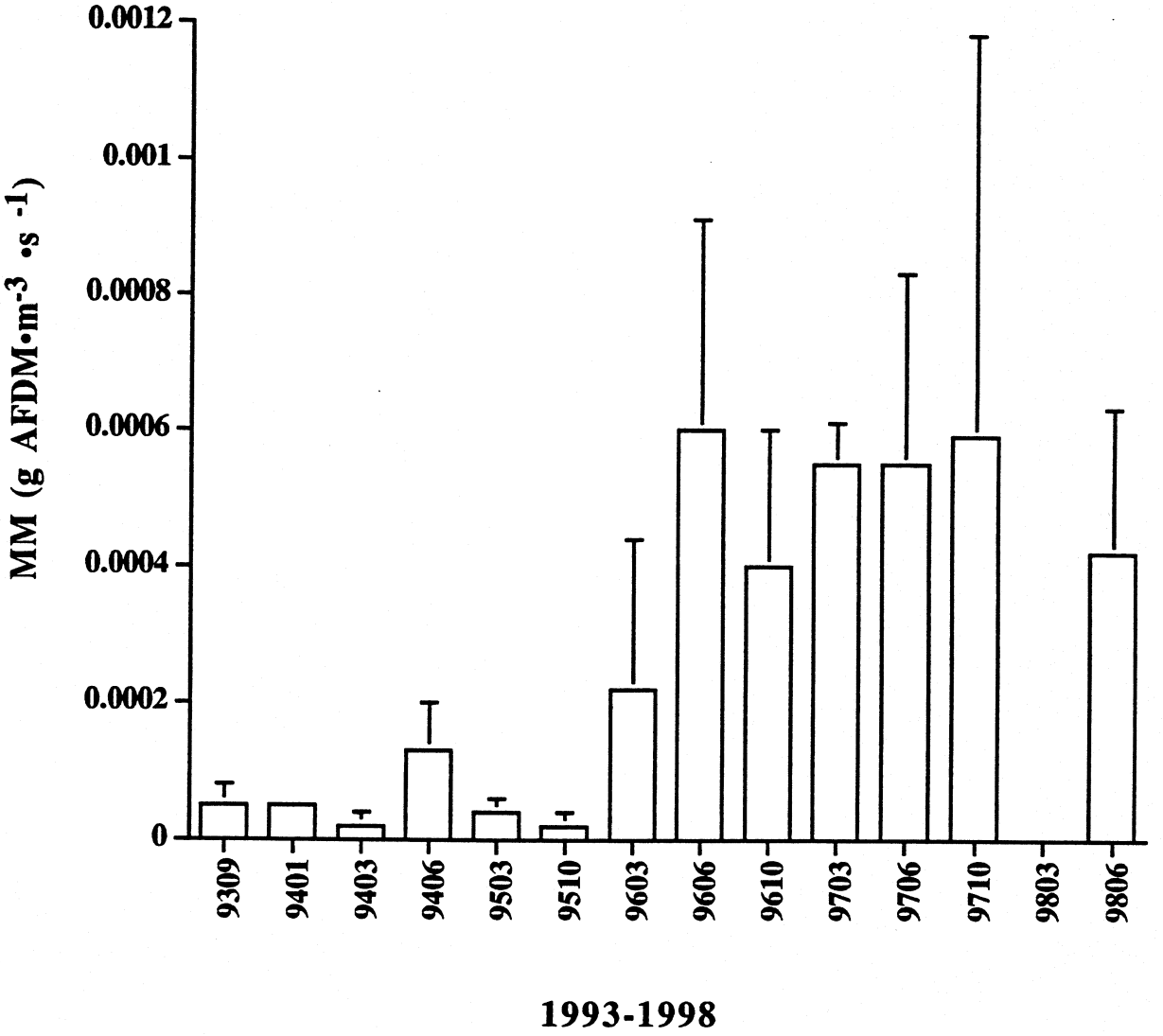

Figure 147. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous macroinvertebrates (MM) collected at Tanner cobble Rkm 109.6 from September 1993 to June 1998. Error Bars represent (± 1 SE).

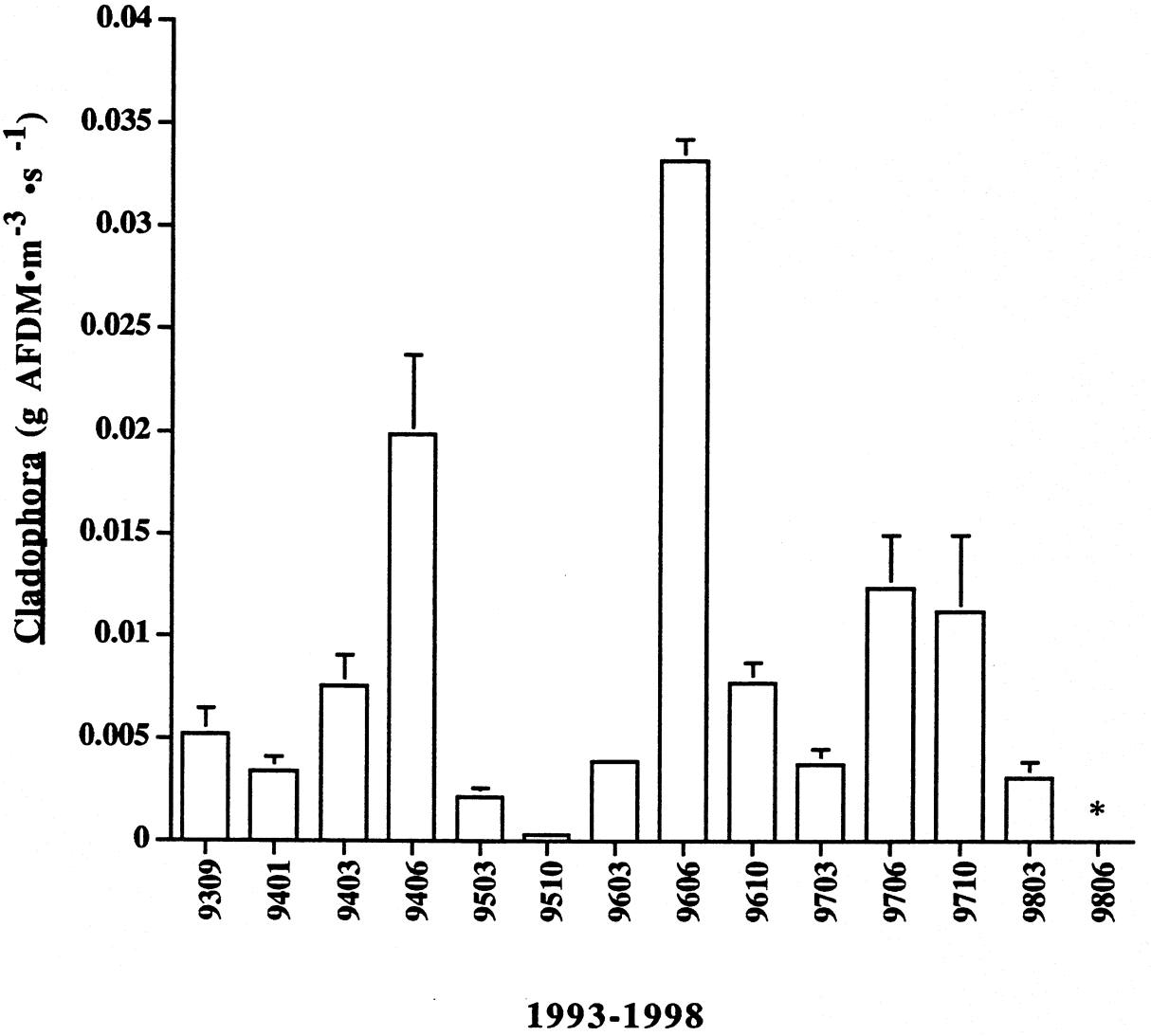

Figure 148. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Cladophora collected at 127 Mile rapid Rkm 202.9 from September 1993 to June 1998. Error Bars represent (± 1 SE). * represents 0.05 g AFDM•m^{-3}•s^{-1} (± 0.02).

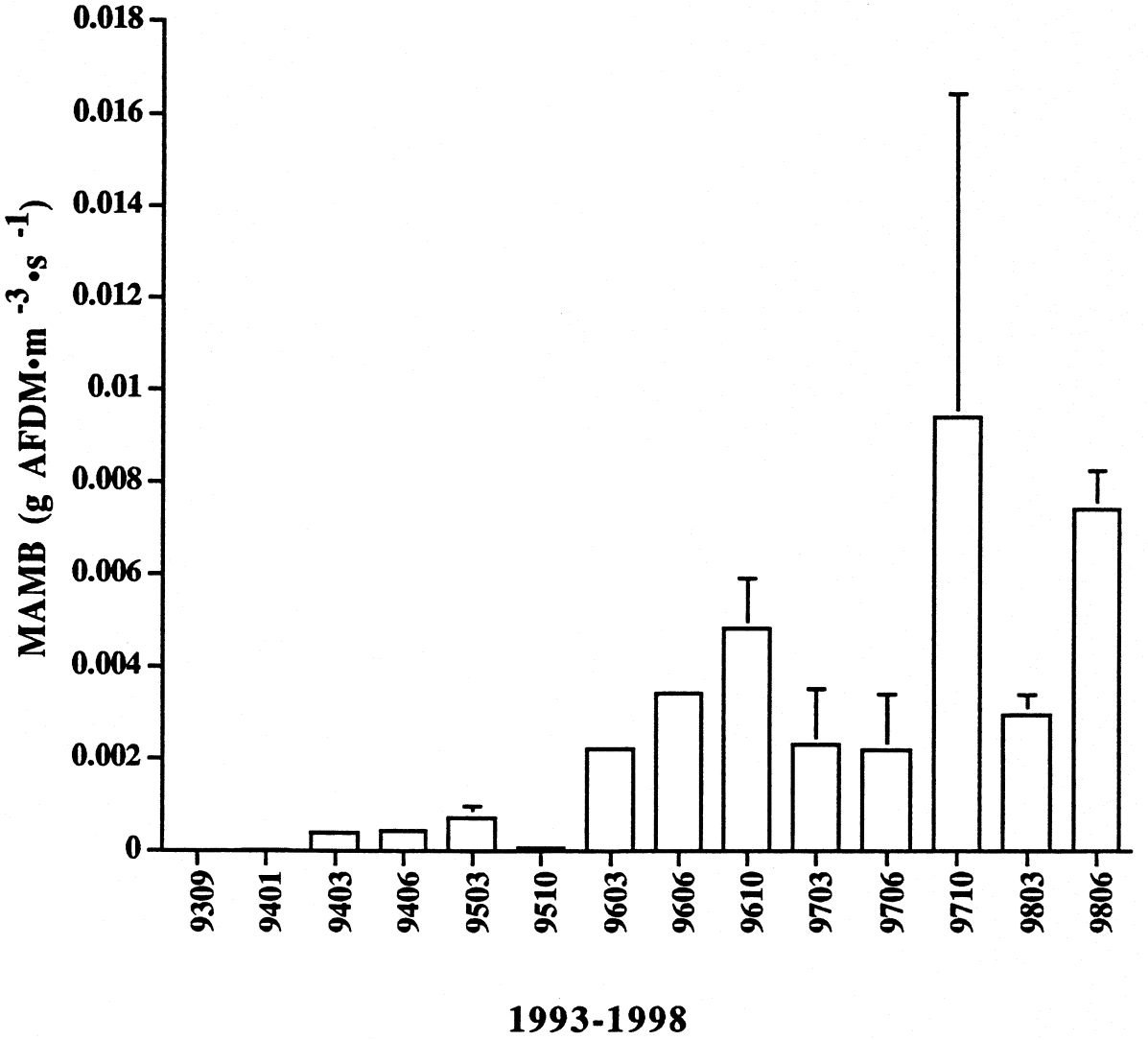

Figure 149. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous algae, macrophytes and bryophytes (MAMB) collected at 127 Mile rapid Rkm 202.9 from September 1993 to June 1998. Error Bars represent (± 1 SE).

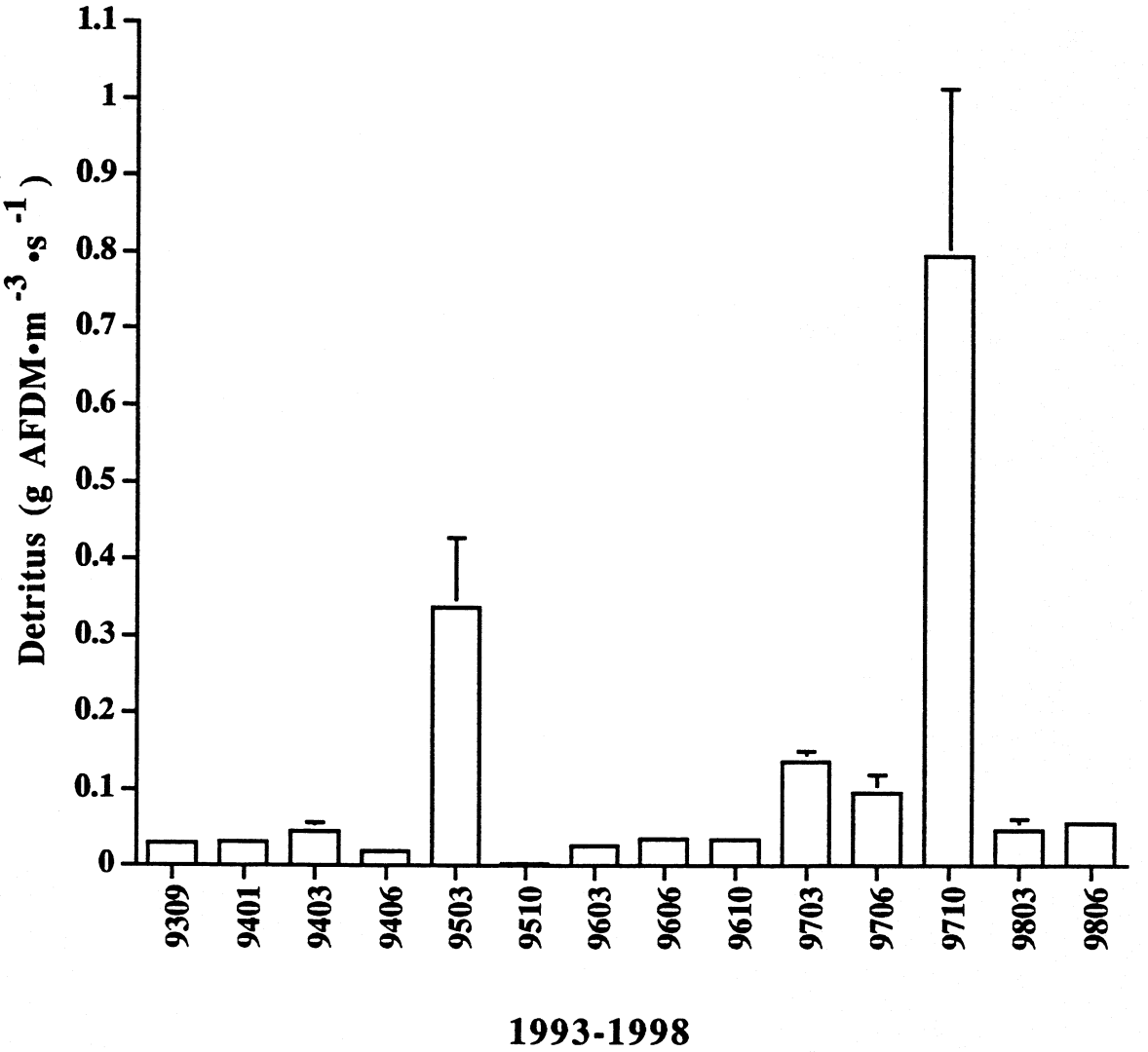

Figure 150. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for detritus collected at 127 Mile rapid Rkm 202.9 from September 1993 to June 1998. Error Bars represent (± 1 SE).

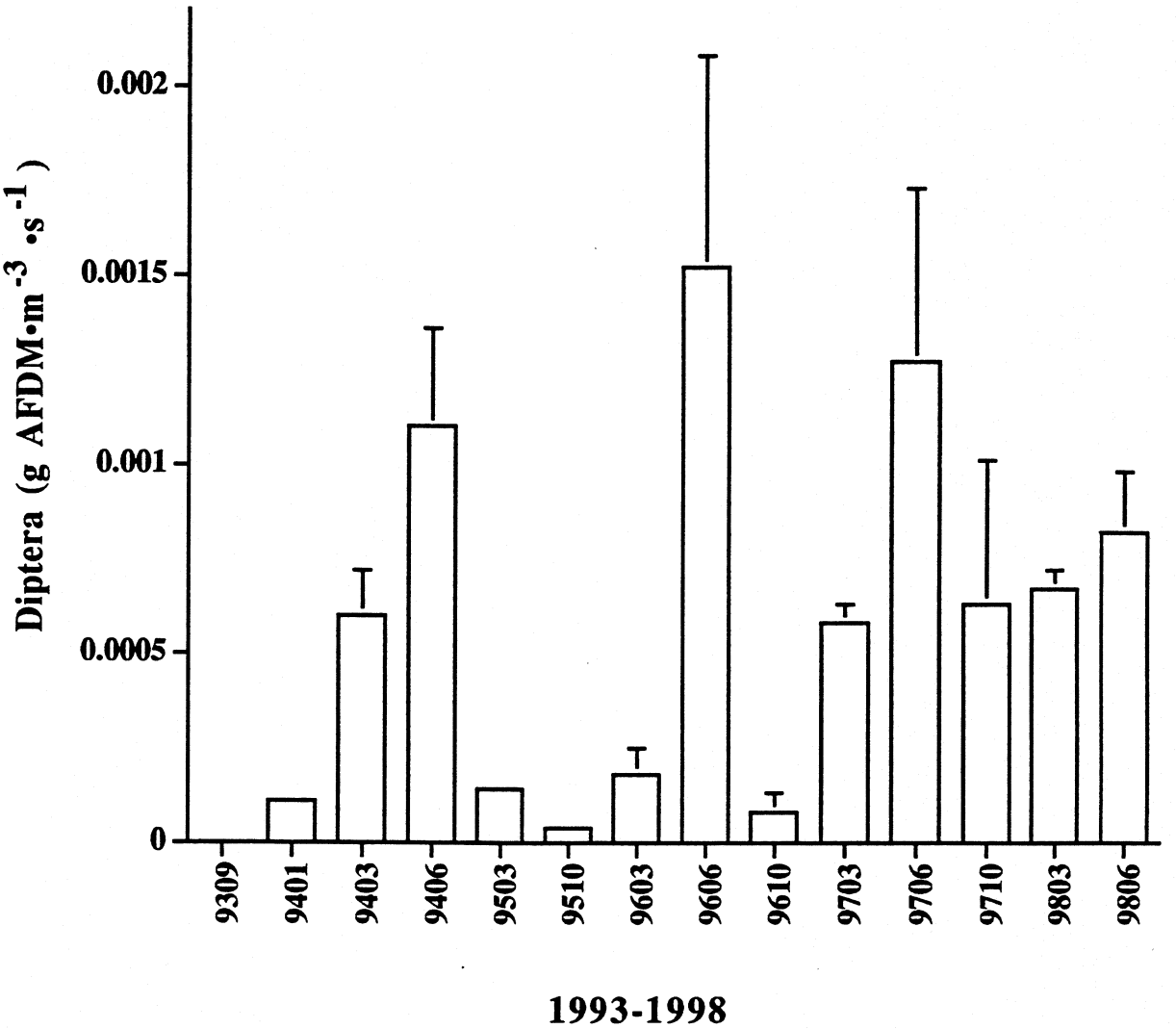

Figure 151. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for diptera collected at 127 Mile rapid Rkm 202.9 from September 1993 to June 1998. Error Bars represent (± 1 SE).

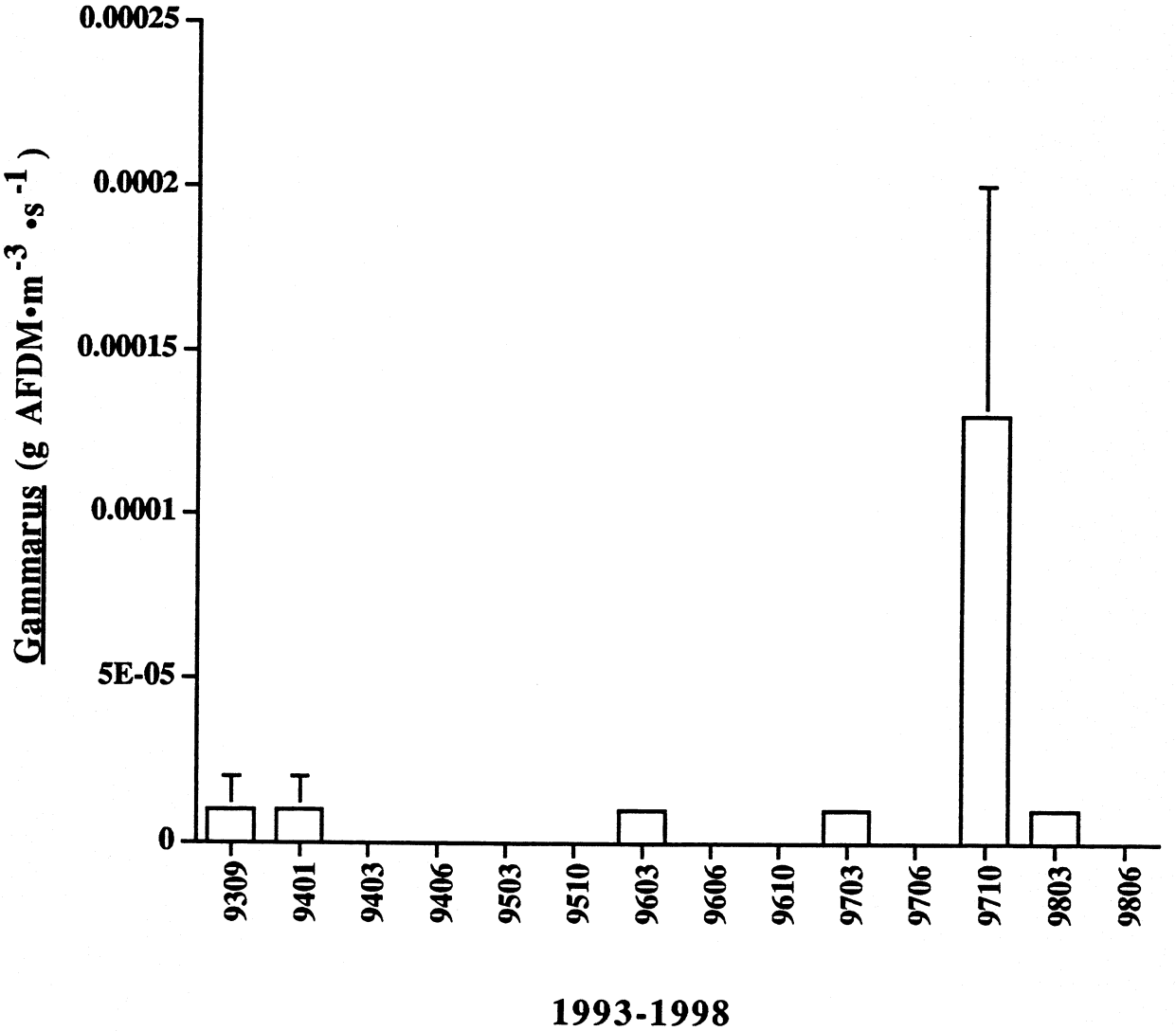

Figure 152. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Gammarus collected at 127 Mile rapid Rkm 202.9 from September 1993 to June 1998. Error Bars represent (± 1 SE).

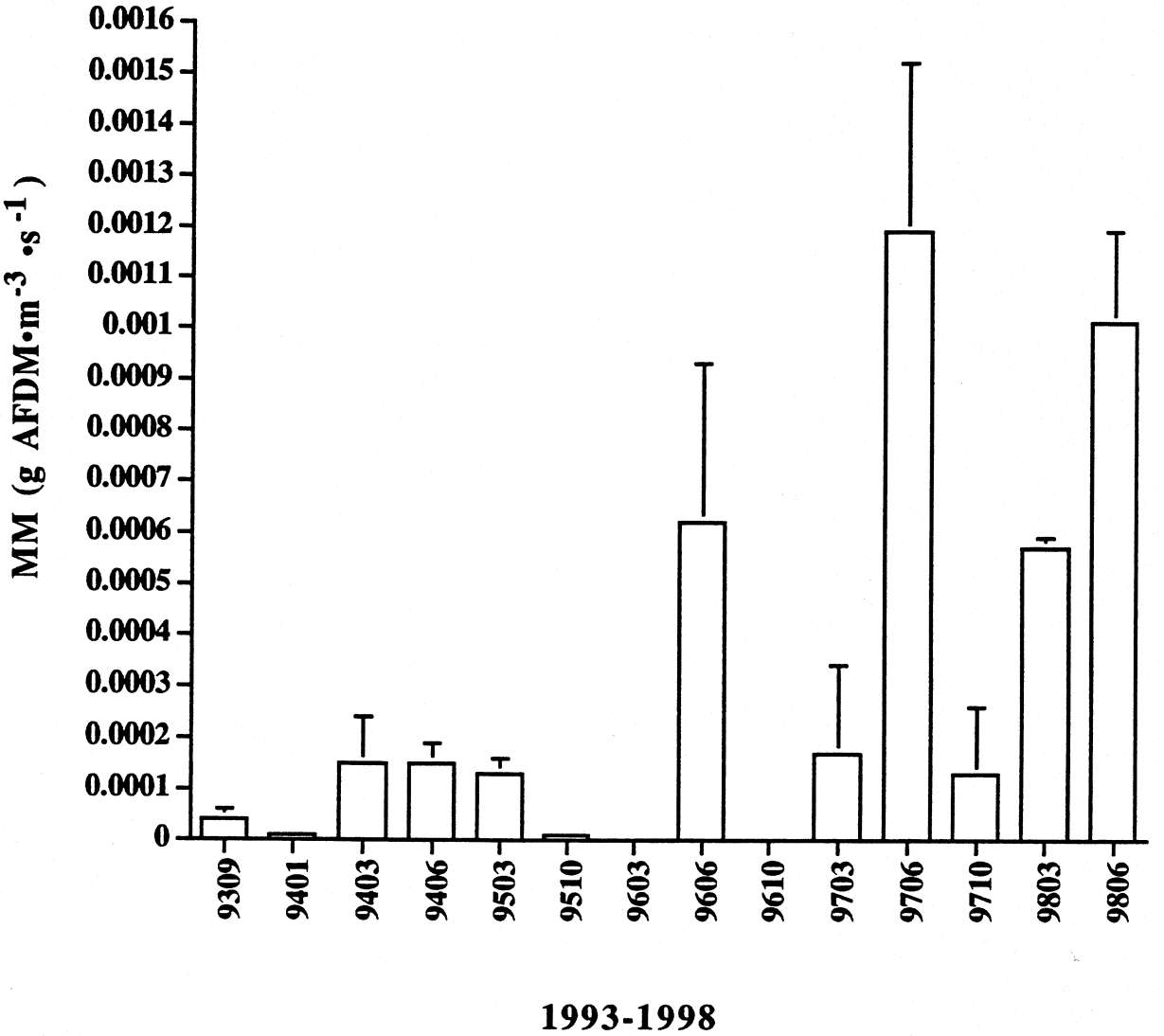

Figure 153. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous macroinvertebrates (MM) collected at 127 Mile rapid Rkm 202.9 from September 1993 to June 1998. Error Bars represent (± 1 SE).

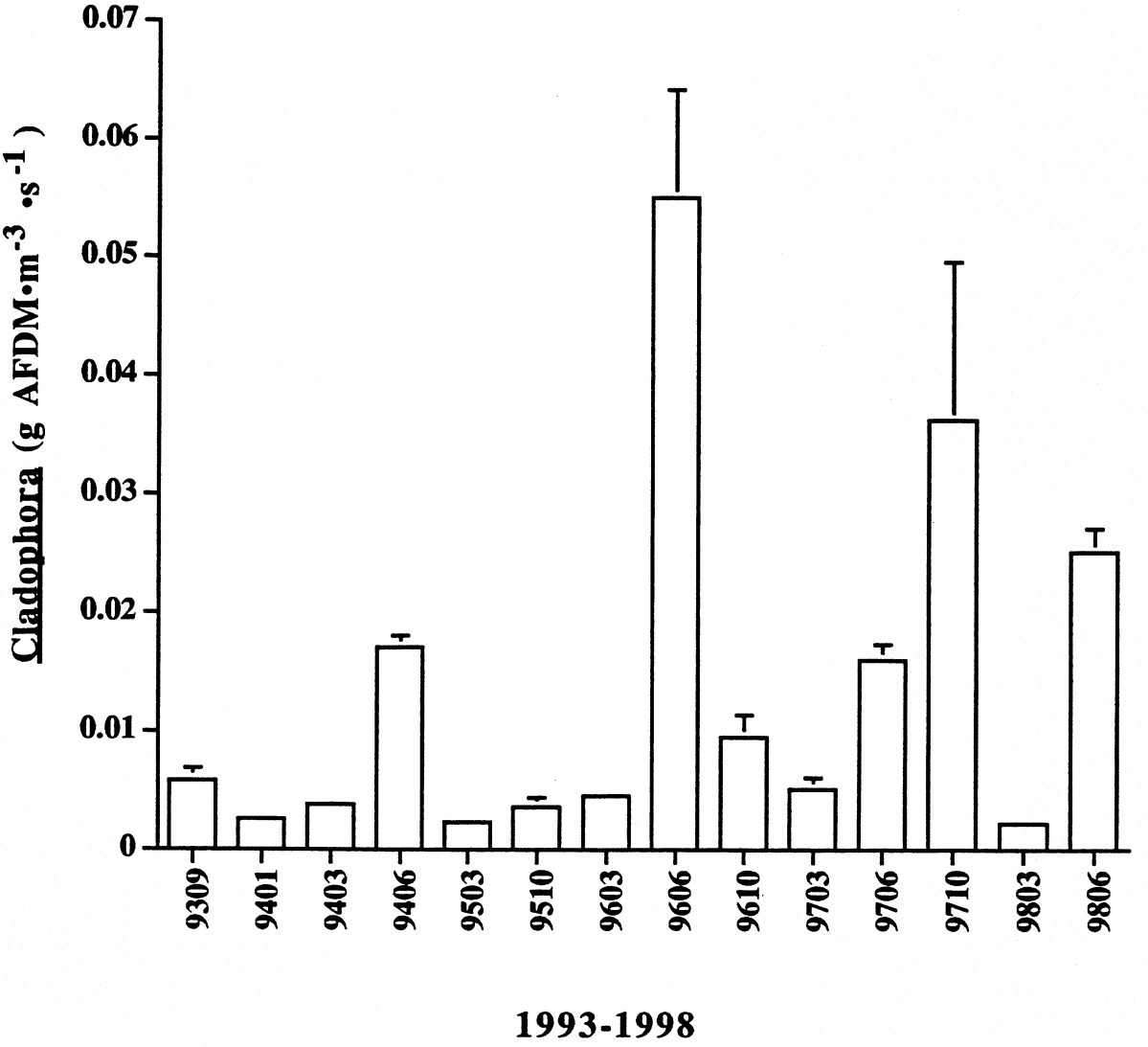

Figure 154. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for Cladophora collected at 205 Mile rapid Rkm 326.4 from September 1993 to June 1998. Error Bars represent (± 1 SE).

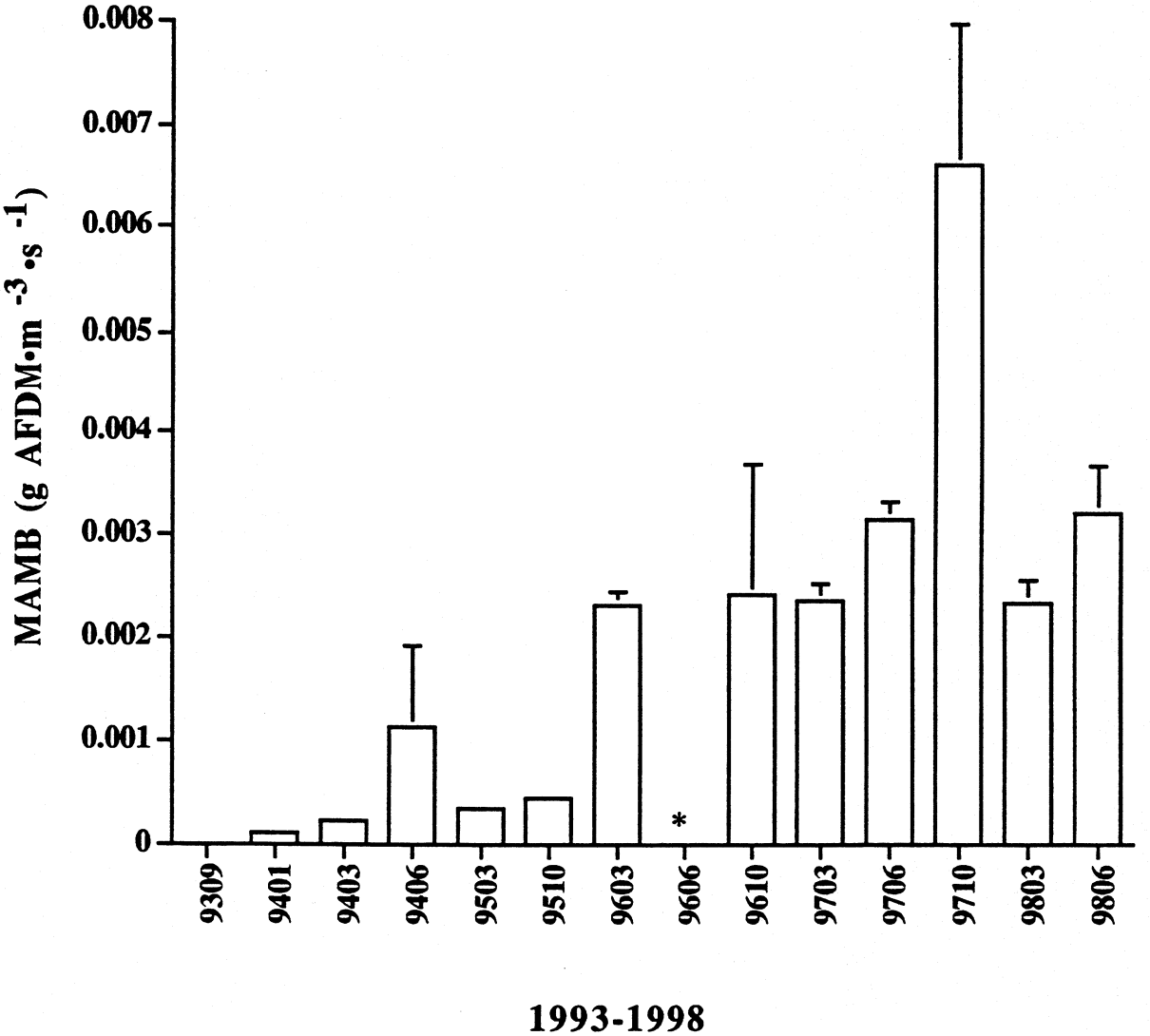

Figure 155. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous algae, macrophytes and bryophytes (MAMB) collected at 205 MIle rapid Rkm 326.4 from September 1993 to June 1998. Error Bars represent (± 1 SE).
* represents 0.018 g AFDM•m^{-3} •s^{-1} (± 0.011).

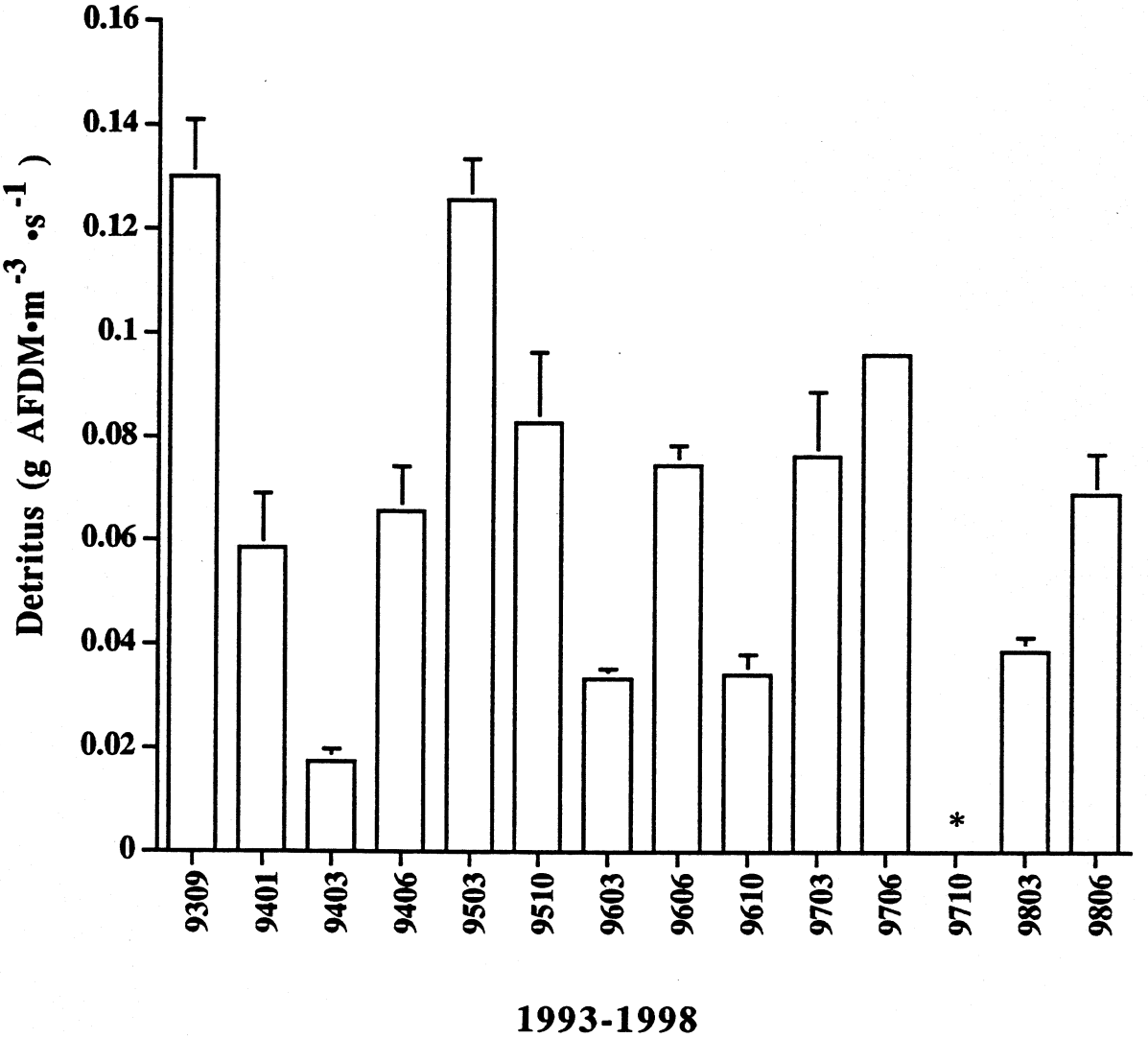

Figure 156. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for detritus collected at 205 Mile rapid Rkm 326.4 from September 1993 to June 1998. Error Bars represent (± 1 SE) * represents 0.5 g AFDM•m^{-3}•s^{-1} (± 0.2).

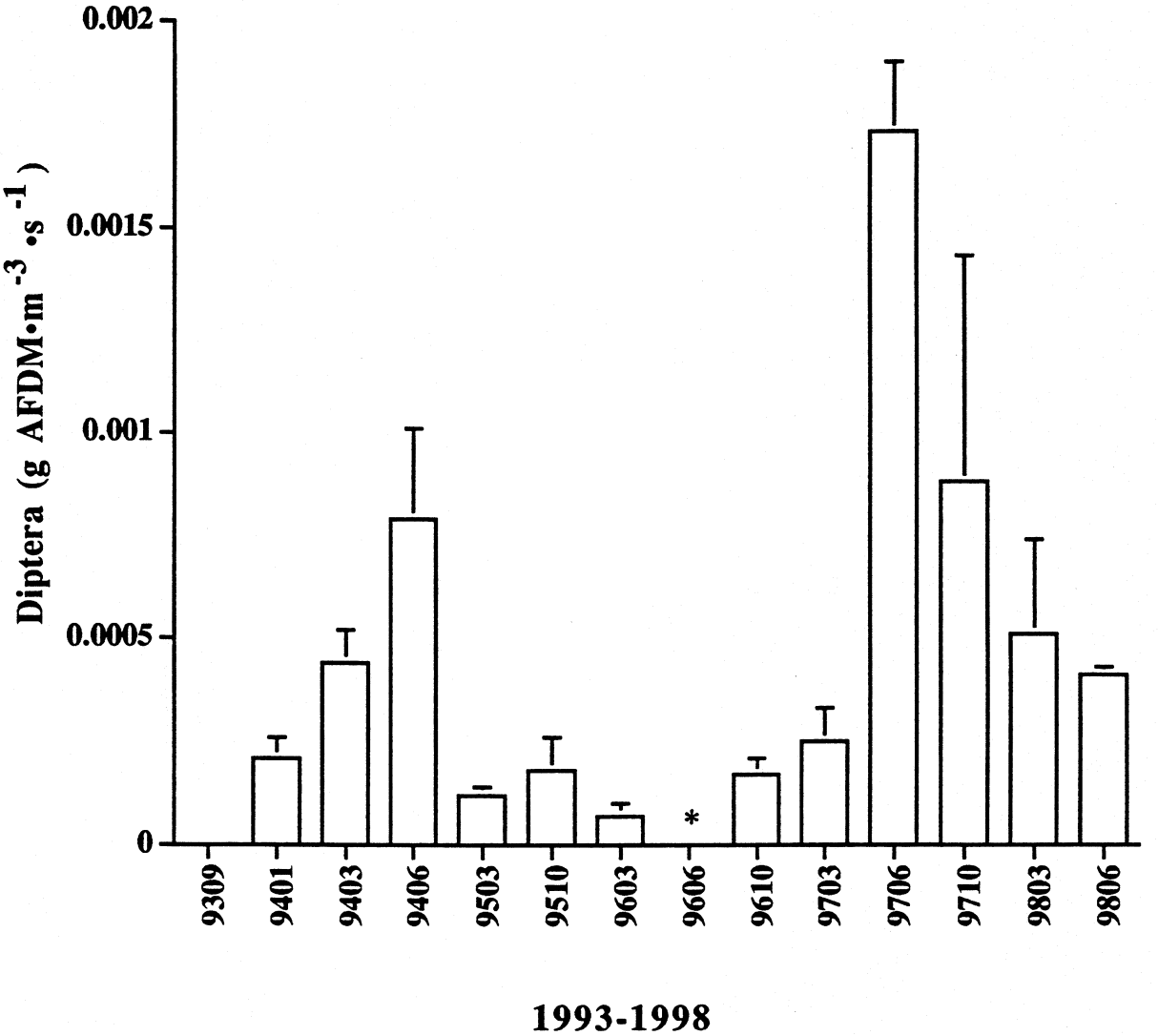

Figure 157. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for diptera collected at 205 Mile rapid Rkm 326.4 from September 1993 to June 1998. Error Bars represent (± 1 SE).
* represents 0.005 g AFDM•m^{-3}•s^{-1} (± 0.004).

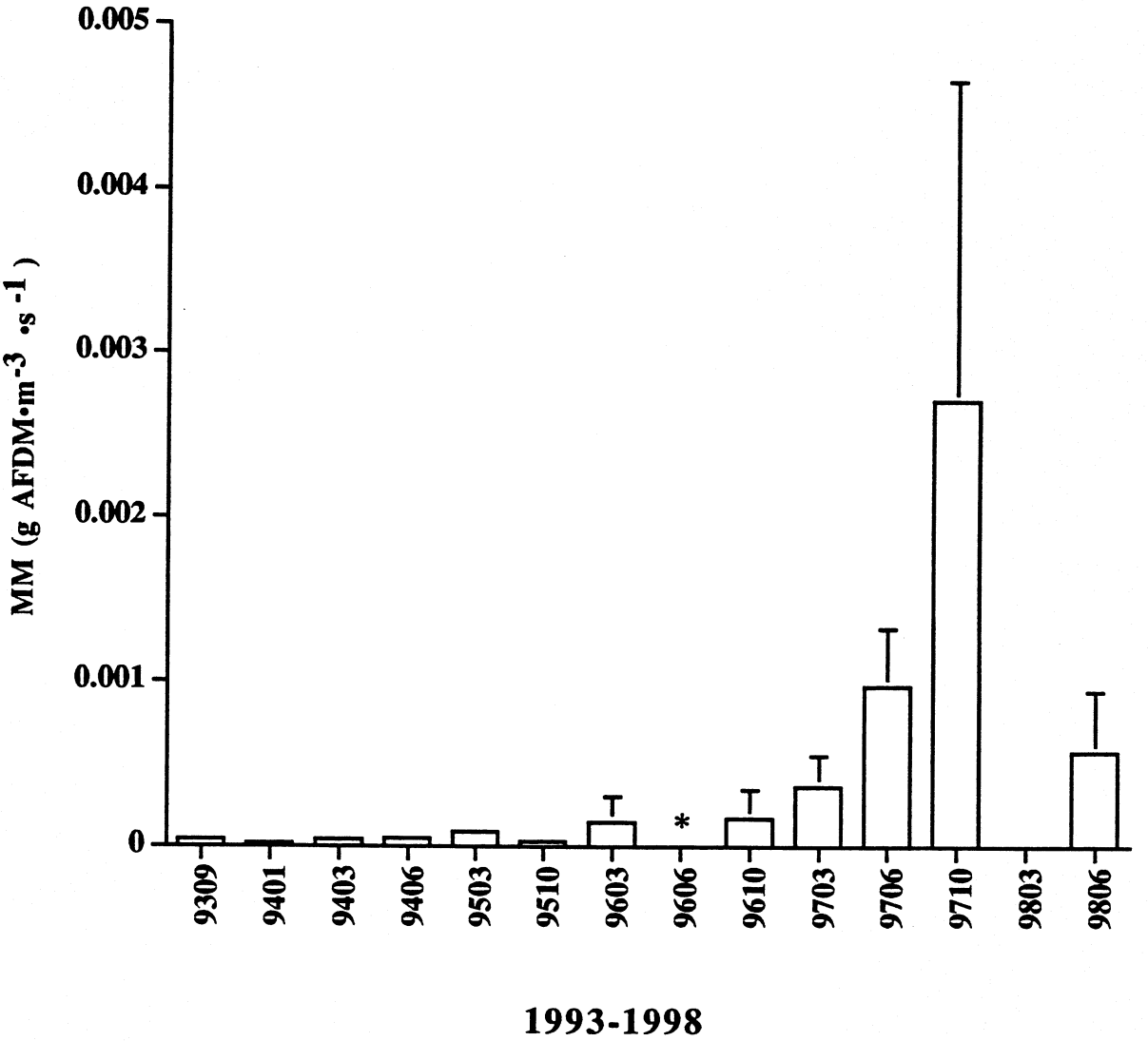

Figure 158. Average CPOM mass (g AFDM•m^{-3} •s^{-1}) for miscellaneous macroinvertebrates (MM) collected at 205 Mile rapid Rkm 326.4 from September 1993 to June 1998. Error Bars represent (± 1 SE).
* represents 0.014 g AFDM•m^{-3} •s^{-1} (± 0.007).