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**NUTRIENT AND HYDROLOGIC LOADING TO UPPER KLAMATH LAKE,  
OREGON, 1991-1998**

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## INTRODUCTION

### **Relationship between algal biomass, water quality, and fish.**

Upper Klamath and Agency Lakes, a large, shallow, hypereutrophic lake system located in south-central Oregon (Fig. 1), exhibits many water quality problems typically associated with excessive algal production. These include extended periods of low dissolved oxygen, elevated pH, and toxic levels of un-ionized ammonia (Kann and Smith 1999; Perkins et al. 1999). These water quality problems are of great concern because of their potential impact on native fish populations in the lake, including the shortnose sucker (*Chasmistes brevirostris*), Lost River sucker (*Deltistes luxatus*), and interior redband trout (*Oncorhynchus mykiss* ssp.). Both sucker species were listed as endangered under the Endangered Species Act in 1988, and water quality degradation resulting from algal blooms had been identified as a probable major factor in their declines (Williams 1988). Moreover, based on harmful levels of dissolved oxygen, pH, and chlorophyll (algal biomass), both lakes have been designated as water quality limited for resident fish and aquatic life (ODEQ 303(d) List 1998).

All three of the above species, as well as native blue and tui chubs, were found in substantial numbers during fish kills occurring in 1995, 1996, and 1997 (BRD 1996; Perkins et al. 1999). Harmful water quality conditions associated with a period of high algal biomass ( $>150 \mu\text{g L}^{-1}$  chlorophyll a) preceding the kills included extended periods (30-90 days) of high pH (9-10) and un-ionized ammonia ( $200\text{-}2000 \mu\text{g L}^{-1} \text{NH}_3$ ). This was followed by a period of sharply decreased algal biomass characterized by low dissolved oxygen ( $<4 \text{mg L}^{-1}$ ) throughout the water column that persisted for up to several days (Perkins et al. 1999). Repeated exposure to these water quality stressors has also been implicated in immune system suppression and subsequent mortality due to diseases and pathogens (e.g., *Flexibacter columnaris*; Perkins et al. 1999).

There is clearly a link between high algal biomass (blooms) and harmful water quality in Upper Klamath and Agency Lakes, and such algal blooms, dominated by the blue-green alga *Aphanizomenon flos-aquae* now occur annually from June through October (Kann 1998). The degraded water quality that results from these blooms is thus a significant threat to the long-term persistence of the endangered suckers and other aquatic life, not only because of catastrophic mortality events, but also because of reduced fitness and survival as a result of chronic stress. Overall reduction of algal biomass is then a critical element of any management program designed to allow recovery of fish populations.

### **Relationship between nutrients and algal biomass.**

The general relationship between water quality dynamics and subsequent fish success in Upper Klamath Lake is as follows: physical-chemical factors (e.g., flushing rate, volume/depth, phosphorus and nitrogen concentration and supply rate, light dynamics, temperature) → algal biomass → habitat quality parameter of concern (e.g., pH, dissolved oxygen, un-ionized ammonia) → physiological tolerance → fish success (fish kills or chronic stress). Under this scenario, knowledge of mechanisms affecting the various linkages within the above relationship is key to management of ecosystem processes that will allow recovery of fish populations. As stated, one essential management goal is the reduction of algal biomass, and because increased

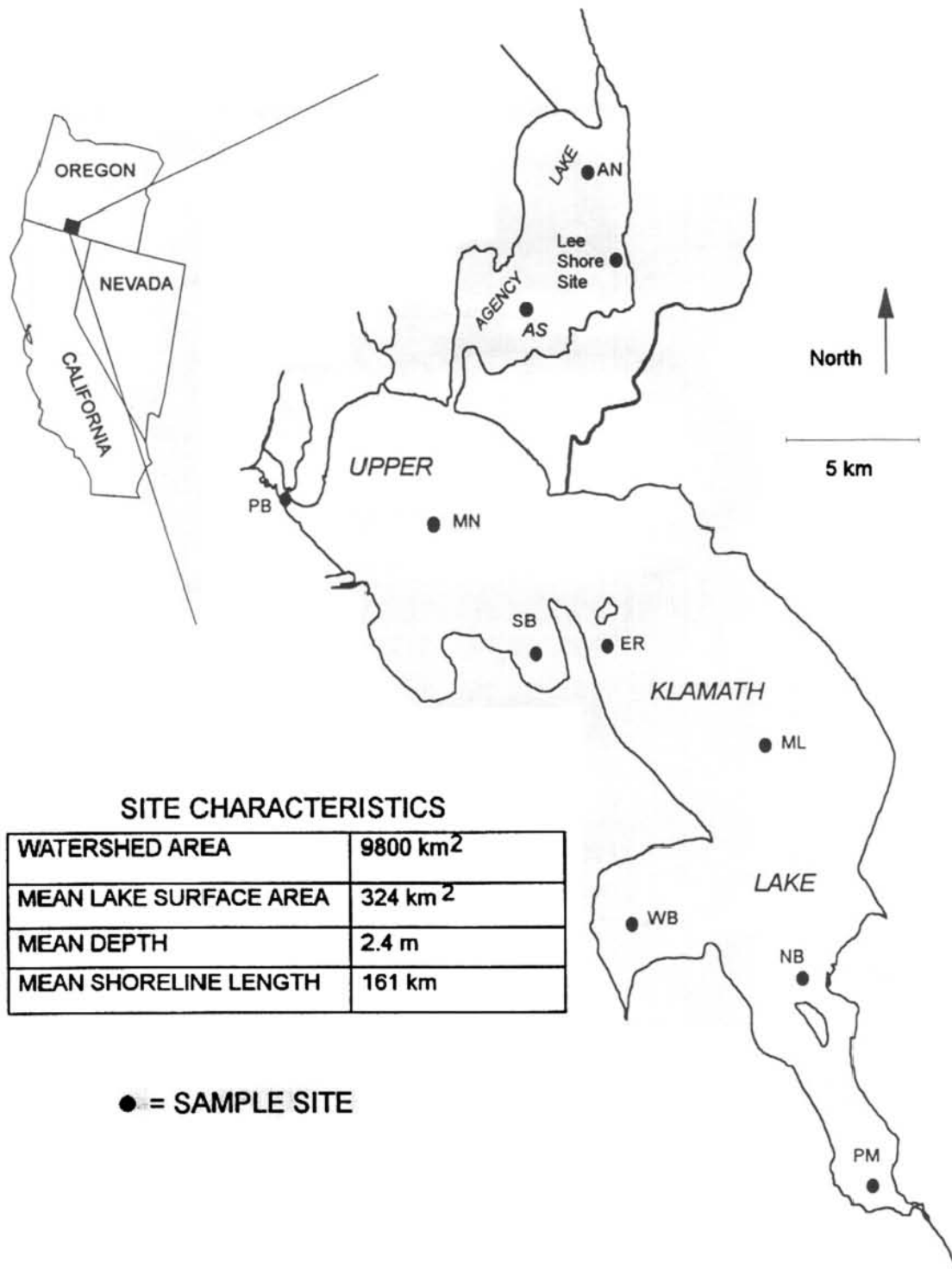


Fig. 1. Upper Klamath and Agency Lakes with in-lake sample site locations.

algal biomass is most often caused by increased nutrient enrichment by nitrogen (N) and phosphorus (P) (Carpenter et al. 1998, Cooke et al. 1993), knowledge of the in-lake concentration and source of these nutrients is crucial.

Although nitrogen can be important in structuring algal communities and determining biomass, P reduction has been shown to be the most effective long-term nutrient management option to control algal biomass (Sas et al. 1989). This is especially true for nitrogen-fixing species such as *Aphanizomenon*, which can augment their nitrogen needs in what may otherwise be a nitrogen limiting system. The chlorophyll-phosphorus relationships described by Kann (1993; 1998) and Walker (1995) also support phosphorus reduction as the management goal. In addition, in shallow hypereutrophic lakes, algal biomass in general and blue-green algae in particular show substantial reductions in response to reduction in P loading, even when P concentrations remain in the hypereutrophic range ( $>100 \mu\text{g L}^{-1}$  total P) after restoration (Jeppesen et al. 1990; Seip et al. 1992).

Given the linkage between P and algal biomass, and between algal biomass and water quality deterioration, the role of P in driving primary production is of central importance. It is therefore desirable to make management decisions regarding phosphorus reduction and subsequent algal biomass reduction as they relate to water quality improvement. Phosphorus-based models form the core of current lake management models (e.g. Vollenweider 1975; OECD 1982; Reckhow and Chapra 1983; Walker 1987; Ryding and Rast 1989; Sas 1989), and such models first require nutrient loading studies and detailed lake phosphorus budgets.

### **Factors affecting nutrient dynamics**

Although historic accounts indicate the lake would have been considered eutrophic even 100 years ago, there have since been numerous land- and water-use alterations within the lake basin which have the potential to impact hydrologic regimes and nutrient export characteristics. The hydrology of the lake has been profoundly changed by water diversions in tributaries entering the lakes, by diversion of water out of the lakes, and by the construction of a dam at the lake's outlet in 1921. As a result, both the timing and quantity of lake flushing flows and nutrient retention dynamics have been altered, and lake surface elevation and volume are reduced below historic levels. As importantly, there have been major changes in the watershed including forest clear-cutting, grazing of  $>75,000$  head of cattle in upstream flood plains, the degradation of riparian corridors, and the conversion of 35,000 acres of wetlands to pasture and agriculture on the lake periphery itself (Gearheart et al. 1995; Risley and Laenen 1999). The Environmental Protection Agency (EPA Index of Watershed Indicators 1998) indicates that at least 110,000 acres of the watershed have been converted to irrigated pasture or other agricultural activities, and Risley and Laenen (1999) show an 11 fold increase in permitted irrigated land acreage between 1900 and the present. Due to the proximity of water sources and the ability to gravity convey water, most of these 110,000 acres occur in riparian and flood plain areas, with the majority being flood-irrigated. These watershed land-use changes are consistent with the types of activities that would cause altered hydrologic regimes (Poff et al. 1997) and increased nutrient loading (above historic background levels) both to tributaries and to lakes (Johnes et al. 1996; Carpenter and Cottingham 1997). An apparent shift in relative abundance from the eutrophic

green alga *Pediastrum* sp., to the nitrogen-fixing blue-green *Aphanizomenon flos-aquae* (S. Coleman and P. Bradbury, US Geological Survey, written communication) during the first 50 years of these land-use changes, is further indication of altered nutrient and hydrologic regimes in this system.

### **Past nutrient studies**

Despite high background P levels in Upper Klamath Basin tributaries, data exists from numerous studies to indicate that P loading and concentrations are elevated substantially above these background levels (Miller and Tash 1967; USACE 1982; Campbell et al. 1993; USGS Water Resources Data 1992-1997, EPA Storet Data 1959-1997). One of the earliest nutrient loading studies (Miller and Tash 1967; updated by USACE 1982) indicates that despite accounting for only 12.4% of the water inflow, direct agricultural input from pumps and canals account for 31% of the annual external total phosphorus (TP) budget. Other studies show that drained and diked wetlands consistently pump effluent containing 3-10X the phosphorus (P) concentration of tributary inflows (Klamath Tribes 1994), and that nitrogen and phosphorus are liberated from drained wetland areas, leach into adjacent ditches, and are subsequently pumped to the lake or its tributaries (Snyder and Morace 1997). Coupled with the considerable but diffuse non-point contribution stemming from wetland loss, flood plain grazing, flood irrigation, and channel degradation, the TP input from anthropogenic sources likely accounts for a far greater percentage than that indicated by the 31% contributed due to direct pumping alone. Gearheart et al. (1995) estimated that over 50% of the annual TP load to the watershed could be reduced with management practices, and Anderson (1998) likewise estimates that in-lake TP concentration can be reduced by utilizing watershed management strategies. Walker (1985) also estimates that an increase in Agency Lake inflow concentration from 81 to 144  $\mu\text{g L}^{-1}$  P is an estimate of the anthropogenic impact.

In addition, there is evidence indicating that sediment regenerated P (internal loading) is also a large source of P in Upper Klamath Lake (Barbiero and Kann 1994; Laenen and LeTourneau 1996; Kann 1998). An important mechanism for release of phosphorus in shallow productive polymictic lakes is photosynthetically elevated pH (Welch 1992; Sondergaard 1988; Jacoby et al. 1982). Elevated pH increases phosphorus flux to the water column by solubilizing iron-bound phosphorus in both bottom and resuspended sediments as high pH causes increased competition between hydroxyl ions and phosphate ions decreasing the sorption of phosphate on iron. Evidence for this exists in Upper Klamath Lake where it was shown that the phosphorus associated with hydrated iron oxides in the sediment was the principal source of P to the overlying water, and that iron-phosphorus fractions decreased from May to June and July (Wildung et al. 1977). In addition the probability of achieving increased internal loading rates increases with pH, and it appears that ~9.3 is the pH level at which the probability of internal loading sharply increases (Kann 1998). Empirical evidence from Upper Klamath Lake along with supportive evidence from other lakes indicates that as the bloom progresses and elevated pH increases the flux of phosphorus to the water column, increased water column phosphorus concentration further elevates algal biomass and pH, setting up a positive feedback loop ( $\uparrow\text{P} \rightarrow \uparrow\text{Algal Biomass} \rightarrow \uparrow\text{pH} \rightarrow \uparrow\text{P}$ ).

## **Study goals**

While it is clear that enough data exist to indicate that P loading and subsequent P concentration are both elevated considerably over background levels, the data have not been detailed enough to develop a dynamic lake nutrient budget that can be used to model in-lake water quality dynamics. The earliest attempt at a phosphorus and hydrologic budget that relied on rigorous inflow loading data was completed over 30 years ago (Miller and Tash 1967), and was only partially revised since then (USACE 1982). In addition, Miller and Tash (1967) did not provide a true mass-balance budget (in-lake mass was not included), include internal loading, or determine seasonal loading and retention patterns. USACE (1982) provided an internal loading estimate based on laboratory experiments, but also did not incorporate in-lake changes in mass, and relied extensively on Miller and Tash (1967) inflow data. Although useful for evaluating watershed restoration strategies, Gearheart et al. (1995) and Anderson (1998) phosphorus models are annual average steady-state models and are based only partly on detailed in-lake and inflow nutrient data. Previous work by Walker (1995) only included the Agency Lake subbasin.

Between 1991 and 1998 detailed in-lake and inflow nutrient loading data have been collected by the Klamath Tribes and US Bureau of Reclamation as part of a long-term water quality monitoring program in the Upper Klamath Basin. Utilizing this data, it is our goal here to develop detailed mass balances for phosphorus and nitrogen at both annual and seasonal time scales. These mass balance budgets will then form the basis for further modeling efforts to evaluate management decisions regarding phosphorus reduction and subsequent algal biomass reduction as they relate to water quality improvement.

## **DEVELOPMENT OF MASS BALANCE FOR WATER AND NUTRIENTS**

### **Lake and inflow nutrient data**

#### ***Lake data***

Regular limnological measurements were made in Upper Klamath and Agency Lakes from January 1990 through September 1998 in order to monitor seasonal water quality, nutrient, phytoplankton, and zooplankton dynamics (Kann 1998). To coincide with collection of inflow nutrient loading data, the lake nutrient data collected between April 1991 and September 1998 were extracted from this larger data set. Samples were taken biweekly or occasionally weekly during the June-September growing period, and were taken approximately monthly during the winter-early spring period except when prevented by ice cover or dangerous storms. In general, 10 sites were sampled on each date, 2 in Agency Lake, and 8 in Upper Klamath Lake (Fig. 1). The exceptions to this were in 1990 when a subset of sites (ER, SB, MN, PB, and AS) in the northern portion of the lakes were sampled on some dates, and in 1996 when only 6 (PM, ML, ER, SB, MN, and AS) of the 10 sites were regularly sampled. Because the lake is polymictic and only undergoes weak and intermittent stratification, a depth-integrated water sample of the entire water column was taken coincidentally with the water quality profiles. This was accomplished by combining a minimum of three replicate hauls from a weighted 5-cm diameter plastic tube at each site. This composite sample was then mixed and portioned off to appropriate collection bottles for the analysis of total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate+nitrite nitrogen ( $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$ ), ammonia nitrogen ( $\text{NH}_4\text{-N}$ ), and total nitrogen (TN). Total inorganic nitrogen (TIN) was taken as the sum of  $\text{NO}_3\text{-NO}_2\text{-N}$  and  $\text{NH}_4\text{-N}$ . With the

exception of TN which followed D'Elia et al. (1977), APHA (1985) was followed for all analyses. All sampling trips included field duplicates and blanks, and laboratory analysis included split samples as well as samples spiked with a known concentration of the parameter to be analyzed.

Utilization of these nutrient data in mass balance analyses required computing the lake-wide mean for each sample date. The Pelican Bay site PB (Fig. 1) was not included due to the direct influence of spring-water and wetland vegetation on the nutrient dynamics at this site. With the exception of utilizing the outlet site Fremont Bridge (FB) to represent lake nutrient concentration during the late fall through early spring period, only those dates when 3 or more sites were measured ( $n \geq 3$ ) were included in the analyses. Based on a comparison of both log transformed ( $\log_{10}$ ) and non-transformed data with the normal distribution using Kolmogorov-Smirnov one-sample tests (*cf.* Zar 1984), it was apparent that site specific nutrient data tended to be lognormally distributed within a sample date. Thus the geometric lake-wide mean provided the best estimate of lake-wide central tendency. Sensitivity of the mean to utilizing a subset of sample sites (e.g., such as in 1996 when only 6 sites were regularly sampled) in the computation of these sample date means was negligible based on a comparison of means computed from a reduced number of sample sites on a given date to means computed using all sites on that same date (data not shown). Lake wide mean nutrient concentrations are provided in Appendix 1.

### ***Inflow data***

Nutrient measurements were made at all major inflows to Upper Klamath Lake from April 1991 through September 1998 (Fig. 2; see data in Appendix 2). However, many of the minor and ephemeral inflows, as well as springs shown in Fig. 2 were sampled less frequently. Moreover, due to the lack of a direct connection to the lake many of these less frequently sampled sites were discontinued after the 1992 water year (WY). Additional samples were collected from agricultural pumps discharging to the lake from adjacent diked and drained wetlands or tributaries (Fig. 3). Pumps were visited on approximately a weekly schedule during the pumping period (late winter-mid spring), but because pumping was sporadic due to float-driven operation, samples could not always be collected.

Nutrient concentration data were collected by both the US Bureau of Reclamation (USBR) and the Klamath Tribes in the Wood River drainage (Campbell and Ehinger 1993, Campbell et al. 1993). Three of the major inflow stations, Sevenmile Canal (7MCA), Wood River Dike Rd. (WODR), and Wood River Weed Rd. (WOWR) (Fig. 2; sites 4, 41, and 42), were sampled monthly by USBR from 1991-1993. More frequent sampling at these same sites was initiated by the Klamath tribes in 1992, when samples were collected approximately weekly or biweekly during high runoff periods, and monthly during low runoff periods. When USBR and Klamath Tribes data coincided, an average value was used to characterize that date. Location information for these sites can be found in Campbell and Ehinger (1993)

As above, an approximately weekly or biweekly sampling frequency during high runoff periods, and monthly during low runoff periods, was initiated for the remaining major inflow, the Williamson River (WRST), its major tributary the Sprague River (SRKB), and the lake outflow

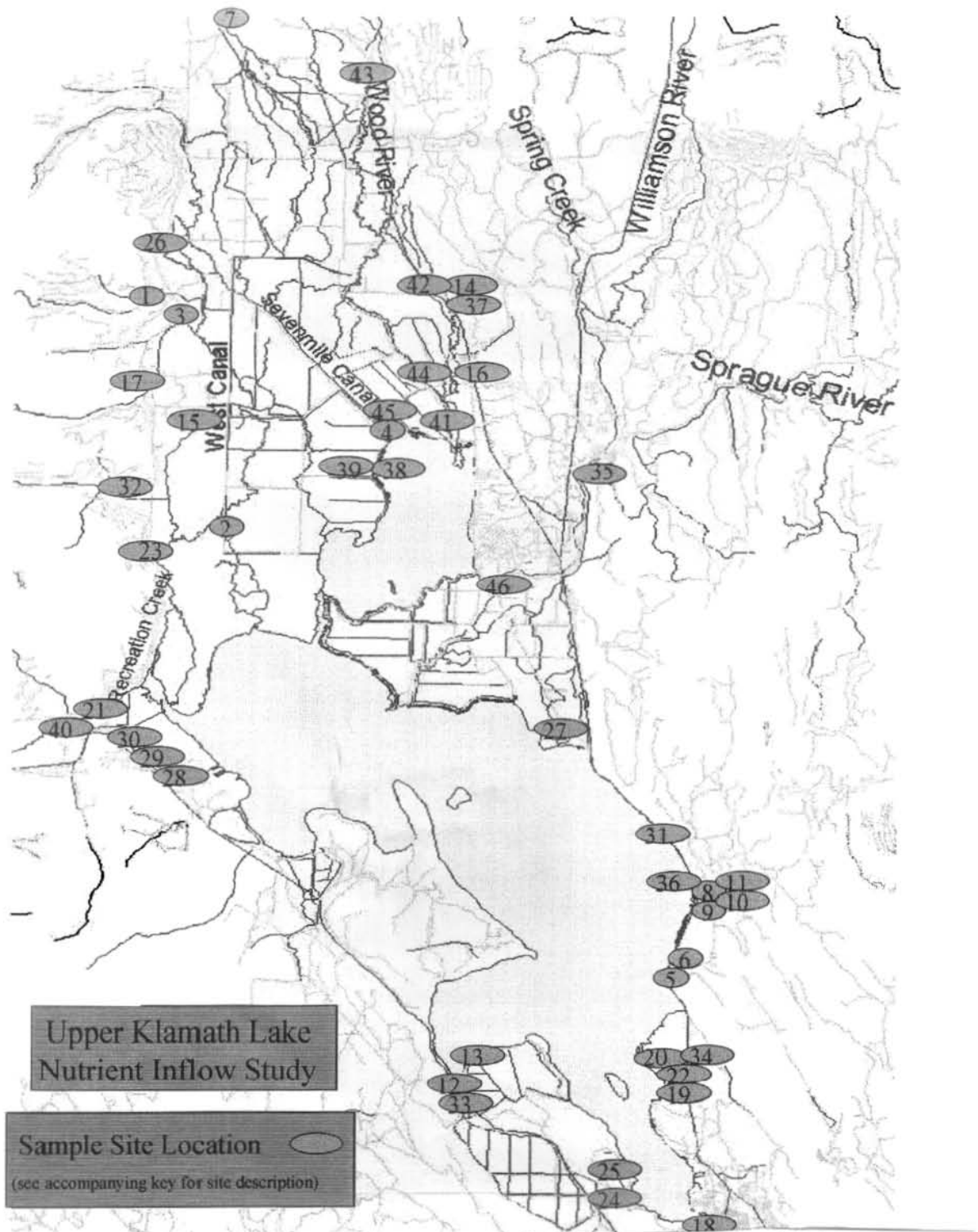


Fig. 2. Location of discharge measurements and nutrient sample sites for Upper Klamath Lake inflows.

Site #	Site Code	Description
1	3MCR	Threemile Creek
2	4MCA	Four Mile Canal
3	4MSP	Four Mile Spring
4	7MCA	Seven Mile Canal
5	ALGOMA BRIDG	Hwy 97 where Alogoma drains to UKL
6	ALGPU	Rattlesnake Point Pump (ALGOMA PUMP)
7	ANCR	Annie Creek at USFS Bridge near Snow park
8	BSCA	Barkely Canal
9	BSFD	Barkley Field
10	BSPU	Barkley Pump
11	BSSP	Barkley Springs @ Hagelstein Park
12	CACA	Caledonia Canal (Lakeside of Hwy 140)
13	CAPU	Caledonia Pump - Draining Caledonia marsh
14	CC62	Crooked Creek AT HWY 62 BRIDGE
15	CCLO	Crystal Creek LODGE CHANNEL
16	CCPA	Crooked Creek AT PAIGE'S
17	CHCR	Cherry Creek.
18	FRBR	Freemont Bridge at Outlet of Klamath Lake
19	HAFD	Hanks Field
20	HAMA	Hanks Marsh (in marsh east of 97)
21	HASP	Harriman Spring
22	HMPU	Hanks Marsh Pump ( draining ag area below marsh)
23	MASP	Malone Spring
24	MCPS	McCornack Pump South
25	MCPU	McCornack Pump
26	MESP	Mares Egg Spring
27	MOCA	Modoc Canal
28	ODCR	Odessa Creek
29	ODESSA CANAL	Odessa Canal draining to pump
30	ODPU	Odessa Pump
31	OXSP	Ouxy Springs
32	ROCR	Rock Creek (Draining to Crystal Creek)
33	RYPU	Wocus, Caledonia Canal South Pmp (RUNNING Y PUMP SOUTH)
34	SPPU	Shady Pine Pump (discharging into the marsh)
35	SRKB	Sprague @ Kirchers
36	SUSP	Sucker Springs
37	TESP	Tecumseh SPRINGS
38	TULANA PUMP	TULANA PUMP (LAKESIDE)
39	TUPU	Tulana Ranch Pump (into Agency Lake south of sevenmile)
40	VACR	Varney Creek
41	WODR	Wood River @ AGENCY LAKE (Dike Rd.)
42	WOWR	Wood River (WEED ROAD)
43	WODX	Wood River Dixon Road
44	WRRPE	Wood River Ranch PUMP EAST (INTO WOOD RIVER)
45	WRRPW	Wood River Ranch PUMP WEST (INTO 7-MILE CANAL)
46	WRST	Williamsom River @ Store (Bridge Crossing on Modoc Pt. Rd.

Fig. 2 (continued). Sample site names and descriptions for location numbers on previous page.

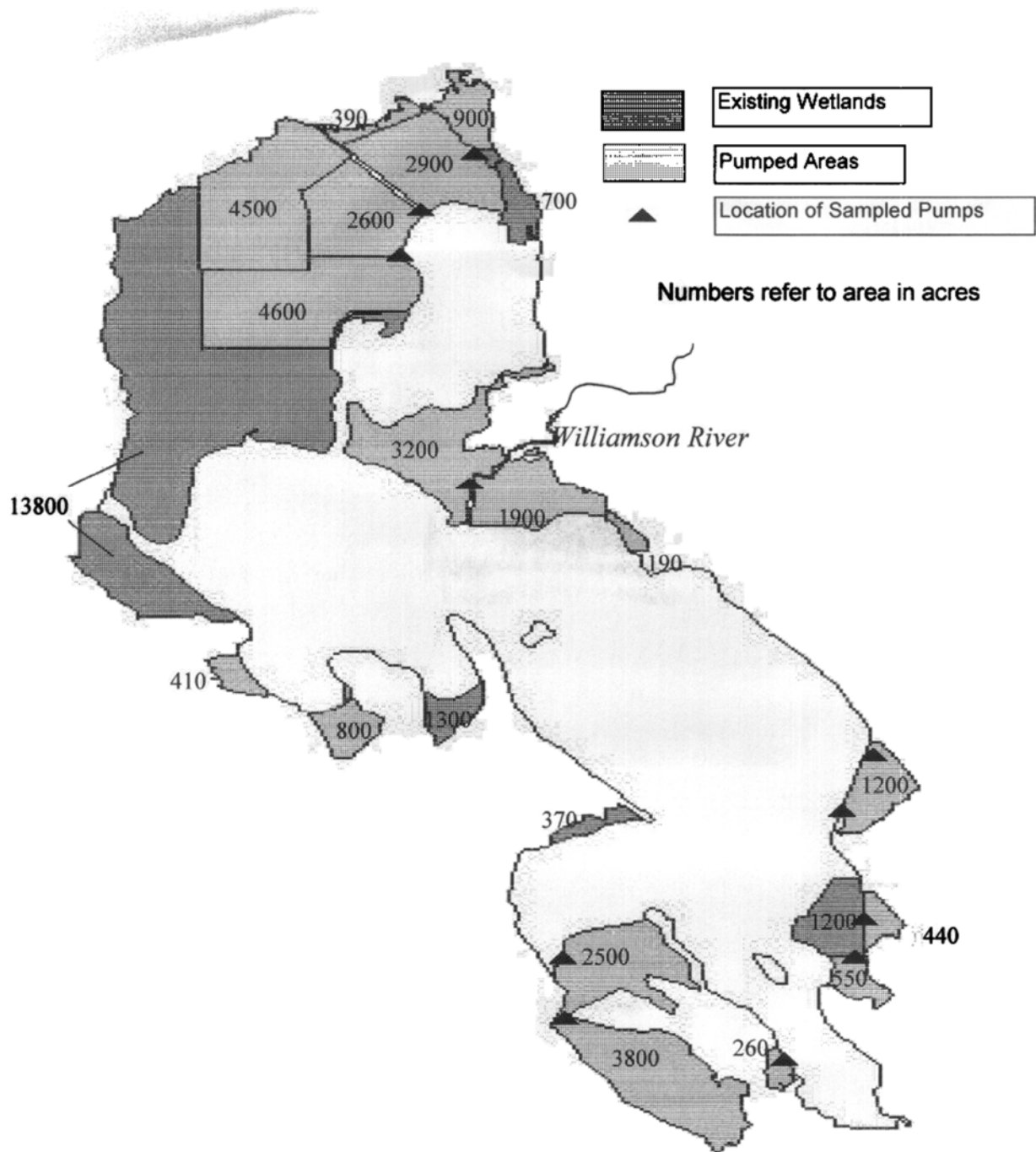


Fig. 3. Location of drained and diked agricultural areas adjacent to Upper Klamath and Agency Lakes.

(FB) (Fig. 2; sites 46, 5, and 18). Because the lake outflow site FB is near the lake sampling site PM, these sites were used interchangeably to represent lake outflow nutrient concentration (earlier statistical analyses showed no significant difference between the two sites). WRST is located at the Modoc Point Rd. Bridge at RM ~5.5; SRKB is located at Kirchers Bridge at RM ~0.5; and FB is located at Fremont Bridge at RM 0 of the Link River.

All inflow and outflow nutrient concentration data were derived from samples that were composited both cross-sectionally and with depth. This was accomplished at the major inflow sites by combining multiple hauls of a Van Dorn horizontal-style water sampler; with the combined sample mixed and portioned off to appropriate collection bottles for analytical analysis as above, and as outlined in Campbell and Ehinger (1993) and Campbell et al. (1993). Both lake and tributary concentration data were interpolated between adjacent sample dates to generate a daily record for input to the mass balance model.

## **Lake and inflow hydrologic data**

### ***Lake data***

Daily lake elevation data for the period of this study (1991-1998) were obtained from the USBR Klamath Falls Project Office. Daily lake volume was then computed from the elevation-volume relationship developed from new bathymetric data collected in 1996 (Upper Klamath Lake/Agency Lake Mapping – 1997 Revised Area Capacity, Leon Bastekas note to files, PRJ1310). Lake volume used here was computed from the USBR elevation-capacity relationship as follows:

$$\text{lake volume} = (\text{active storage}) + (\text{dead storage}) - (\text{marsh storage}).$$

Daily precipitation records were obtained from the Oregon Climate Service, Corvallis Oregon for Station 4506 in Klamath Falls and Station 1571 in Chiloquin. Precipitation volume entering the lake was then computed from the mean of the two precipitation gaging sites multiplied by lake surface area (lake surface area was computed from elevation-area curves supplied by USBR).

Daily class A pan evaporation data obtained from the OSU Experiment Station located in Klamath Falls were corrected to approximate open-water evaporation by multiplying by 0.7 (Merritt 1968; Kadlec 1989). Long-term mean monthly evaporation values were used to fill in missing observations during the months of March, April, May and October. Evaporative loss from the lake surface was computed by multiplying daily open-water evaporation estimates by lake surface area to obtain total volume lost.

Daily lake outflow volume was computed from continuous daily stream-flow gaging stations located on the “A” Canal and Link River near the outlet of the lake. The total volume of lake outflow is equal to the sum of the “A” Canal and Link River gages (data obtained from USBR).

### ***Inflow data – tributary flows***

Daily inflow volume for the Williamson and Sprague Rivers was computed from continuous daily discharge data obtained from U.S. Geological Survey stream-flow gaging stations (USGS Gages 11502500 and 11501000, respectively). Continuous daily stage-height measurements were made by USBR at the Wood River Weed Rd. (WOWR) site from 1991-1996, and by Matthews (1999) thereafter. Daily inflow volume for WOWR was then computed from the stage-discharge relationship generated from instantaneous discharge measurements (measured at the same frequency as nutrient concentration data) and stage height readings (detailed methods are contained in Campbell and Ehinger 1993 and Matthews 1999).

Attempts to generate daily stage-discharge relationships at the Wood River Dike Rd. (WODR) and 7-Mile Canal (7MCA) stations failed due to the seasonal back-water affect of lake elevation on the recorders deployed at these stations. However, it was possible to measure instantaneous discharge at both locations, and these were also generally taken at the same frequency as the nutrient concentration data. The exception to this was at 7MCA, where, due to lower flows and close proximity to the lake, there were no detectable velocities on some sample dates. The nutrient concentration was then assumed to be influenced by the lake, and zero loading was assigned for these dates. At all other miscellaneous sites, instantaneous discharge was measured at the same time as concentration data were collected.

To generate complete daily flow records for the WODR and 7MCA sites the following procedure was used:

1. Each instantaneous discharge measurement was paired with the corresponding daily-mean flow at the WOWR station.
2. A regression equation was developed relating the station discharge to the WOWR discharge.
3. The regression equation was applied to generate a predicted discharge for each day of the record.
4. The residual was calculated (observed-predicted) for the days having instantaneous measurements.
5. The residuals were interpolated over time to generate a residual for each day in the record.
6. The daily discharge was calculated by adding the predicted flow (3) and the interpolated residual (5).

All regression analyses used in these procedures are shown in Appendix 3. In situations where the correlation between WOWR flow and the station flow is high (e.g., WODR), this procedure tends to track the Weed Road flow (with an appropriate adjustment in scale). In situations where the correlation is weak (e.g., 7MCA), this procedure approaches a direct interpolation of the instantaneous flows over time. A similar procedure was used to fill in missing discharge values for WOWR when the automatic stage recorder was temporarily inoperable. However, in this case the regression equation calculated using Williamson River daily data was not able to predict

WOWR flows greater than 300 cfs. This is because the Williamson River tends to have higher peaks relative to its mean during spring runoff than does the Wood River. Missing values occurring during periods when the correlation is poor (i.e., when Wood River flow is >300 cfs) then approach a directly interpolated value (see Appendix 3).

*Inflow data – direct agricultural pumping*

The volume of water pumped from adjacent diked and drained agricultural areas was estimated from Hubbard (1970) and Snyder and Morace (1997) by developing a regression between annual Williamson River volume and the annual agricultural unit area flow (UAF; annual pumped volume/area drained) for the WY's of 1965-1967, and 1985. Both studies provided detailed estimates of the volume pumped during the respective years; with Hubbard (1970) utilizing meters on all irrigation pumps, and Snyder and Morace (1997) using records of electricity use and then applying a rating curve that was developed between kilowatt/hours and volume for each measured pump. The calculations for estimation of pumped agricultural drainage for this study (WY's 1992-1998) are contained in Appendix 4, with the general procedure as follows:

1. Total annual agricultural volume discharged to the lakes was obtained for WY's 1965-1967 (Hubbard 1970). Because some pumped areas measured in Hubbard (1970) were no longer managed as pumped agricultural areas at the time of the Snyder and Morace (1997) study, the UAF for each year was calculated in order to standardize the values so that the data from both studies could be combined.
2. Pumped volume data were obtained for WY's 1993-1995; however, only enough data existed for WY 1995 to provide an estimate of total annual volume for the 4 areas represented by the pumping stations (data provided by J. Morace, USGS, Portland, Oregon). The average UA flow of the 4 measured areas was then used to estimate the volume pumped from remaining unmeasured agricultural areas (see Appendix 4). The total volume of measured and unmeasured areas combined was then divided by the total area to obtain the UAF for WY 1995.
3. A regression equation was developed relating UAF for each of the four WY's to total annual Williamson River discharge. The regression equation was then applied to total annual Williamson River volume for WY's 1992-1998 to generate a new predicted agricultural UAF for each WY.
4. The predicted UAF was then multiplied by the area of agricultural pumping to obtain total volume discharged to the lake for each WY. The agricultural area used to predict total volume was exclusive of the Wood and Thomas properties because the volume contributed by these areas is already included in the WODR and 7MCA station estimates.
5. This total annual volume was then apportioned to each month based on the average monthly fraction pumped from Hubbard (1970).

### ***Inflow data – ground water***

Groundwater inflow volume (including springs and seeps) was calculated as the residual of the lake water balance as follows:

$$\text{GW} = \text{outflow} + \text{evaporation} + \Delta \text{ lake storage} - \text{tributary inflow} - \text{agricultural inflow- precipitation}$$

Due to the difficulty in obtaining accurate discharge time series for some of the minor tributaries discharging directly to the lake (e.g., 4-Mile Canal) these were not estimated separately, but are included in the residual GW term here. In the summary tables and figures this term is denoted as “Springs”. This term also includes all other error in measured discharge and lake hydrologic characteristics.

### **Nutrient budget construction**

The above estimates of nutrient concentration and water volume were used in all subsequent determinations of nutrient mass. The nutrient mass from each surface inflow and outflow was computed as the product of daily estimated nutrient concentration and discharge. The nutrient mass contained within the lake was computed as the product of daily lake volume and daily estimated lake-wide mean nutrient concentration.

The nutrient mass from direct agricultural input from adjacent diked and drained wetlands was estimated at monthly intervals as the product of the weighted-average nutrient concentration of measured pumps (weighted by the area representing each pump having measured nutrient concentration data; Table 1) and the estimate of monthly water volume pumped from the contributing areas (derived as above).

The nutrient mass entering the lake from groundwater (springs, seeps, and small surface flow) was determined as the product of the GW volume estimate from the above equation and mean nutrient concentration of measured springs in proximity to the lake (Fig. 2; Table 2).

Atmospheric inputs (the sum of wetfall and dryfall) were estimated at fixed areal rates of 18 kg/km<sup>2</sup> yr<sup>-1</sup> for phosphorus, and 1080 kg/km<sup>2</sup> yr<sup>-1</sup> for nitrogen (USEPA 1975).

Estimates of the background and anthropogenic loads were made as follows:

1. Background load was estimated as the total inflow volume from all sources (tributary+precipitation+agric. pumping + springs) multiplied by the mean nutrient concentrations from the spring sources in Table 2 (63 µg L<sup>-1</sup> TP, etc.). Because many of the tributaries originate from springs (e.g., Wood River, Williamson River, etc), this provides an estimate of loading in the absence of anthropogenic inputs.
2. Anthropogenic load was then estimated as the difference between total inflow nutrient load and background load from (1) above.

### ***Internal Loading***

Net nutrient retention was calculated as the residual of the phosphorus mass balance equation as follows:

$$\text{Net Retention} = \text{tributary inputs} + \text{atmospheric inputs} - \text{outputs} - \Delta \text{ lake storage}$$

Net retention reflects net losses from the water column resulting from sedimentation, atmospheric fixation (nitrogen), nutrient releases from bottom sediments, and the cumulative effects of errors in the other mass-balance terms. Negative retention values denote a source from within the lake. Net internal TP load ( $TP_{\text{int}(\text{net})}$ ) then equals  $-1 \times (\text{net retention})$  and is defined as the difference between TP release from the sediments and specific sedimentation. The total or gross internal TP loading ( $TP_{\text{int}}$ ) is then defined as  $TP_{\text{int}(\text{net})} + \text{sedimentation}$ . Assuming deposition of TP to the sediments is a function of phosphorus mass in the lake (Vollenweider 1969), specific sedimentation rate ( $s$ ) can be empirically estimated from the mass balance equation using the time period when internal loading is assumed to be negligible (March-May). Although we did not intend to provide detailed estimates of internal loading in this report, a sedimentation rate of 0.01/day (.07/week) was calculated from positive retention values during the March through May period. The following equation was then used to approximate  $TP_{\text{int}}$ :

$$TP_{\text{int}} = TP_{\text{int}(\text{net})} + (s \times TP_{\text{lake}}); \text{ where } s=0.01 \text{ and } TP_{\text{lake}}=\text{TP mass in lake}$$

The total annual or time period estimate of  $TP_{\text{int}}$  is taken as the sum of all positive  $TP_{\text{int}}$  values over the time period. This allows for an approximate estimate of internal TP loading for comparison to external sources.

Due to the generally biweekly time step for collection of nutrient concentration data, the water and nutrient balance time series have been formulated at biweekly intervals (Appendices 5 and 6).

## **WATER AND NUTRIENT BUDGETS**

### ***Hydrologic Budget***

The Williamson River and Wood River together accounted for 67% (51 and 16%, respectively) of the mean 1992-1998 (WY's) total inflow; with springs and small ungaged tributaries contributing another 16% (Fig. 4; Table 3). 7-Mile canal, precipitation, and agricultural pumping accounted for the remaining 17% (Fig. 4; Table 3). Relative contributions from the various inflow sources were remarkably similar to Hubbard 1970 and USACE 1982. As expected given relatively small watershed size (drainage area; Table 3) and high percent of flow contributed by springs, the unit area flow is substantially higher for the Wood River and 7-Mile Canal (Fig. 4). Indicative of input from adjacent irrigated areas with water not originating from within the defined drainage area, as well as inflow from another spring-dominated yet small drainage area (Crooked Cr.), unit area flow is also substantially higher for the area below the Weed Rd. station on the Wood R (Fig. 4). A similar phenomenon affects 7-Mile Canal, where

### Water Balance - Annual Summary (1992-98)

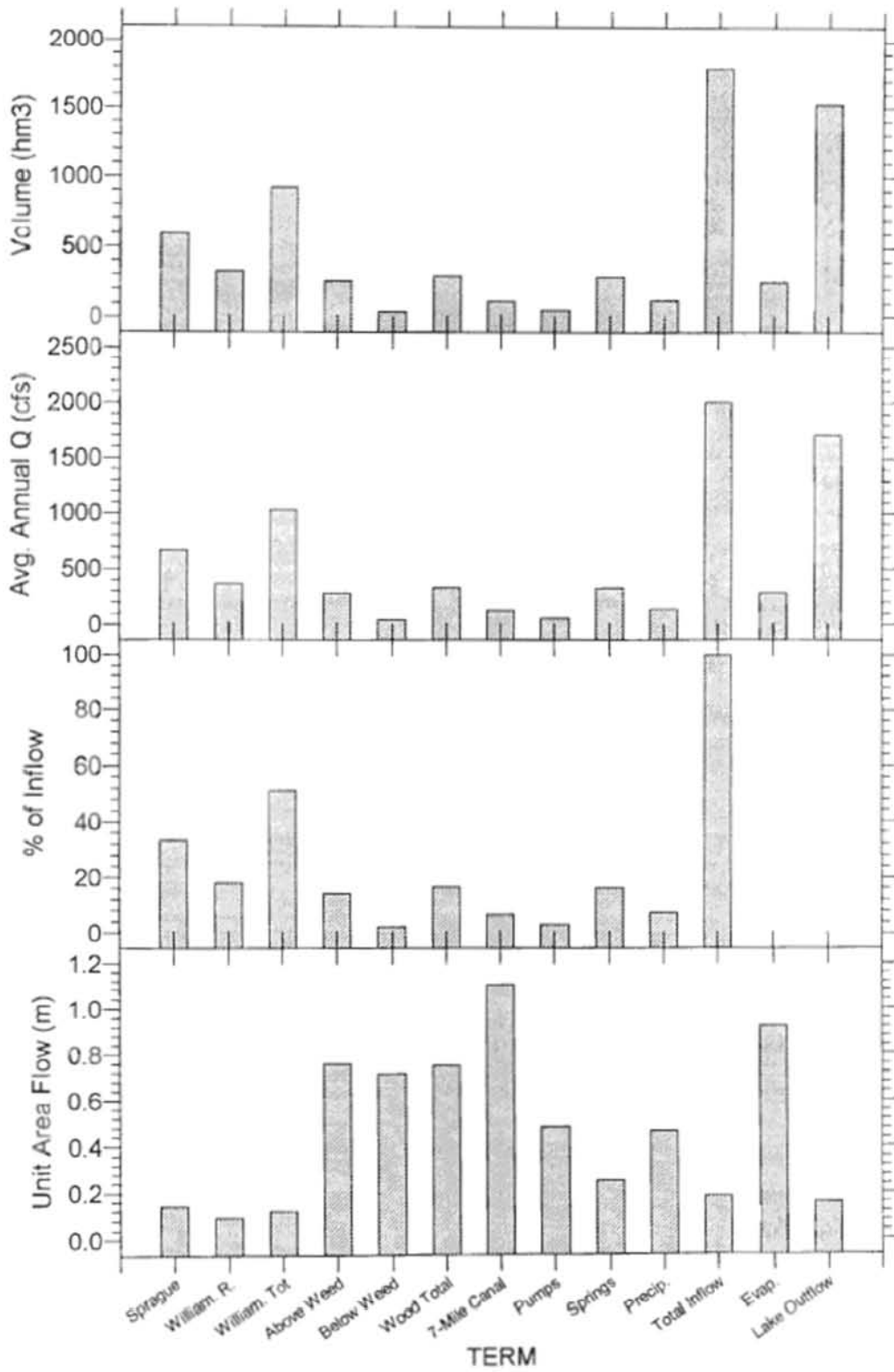


Fig. 4. Average annual water balance for water years 1992-1998.

drainage from the Wood River (e.g., Central Canal) is returned to 7-Mile, thus inflating the unit area water load.

Below average precipitation in 1992 and 1994 (the 1992 April snow pack was 26% of average) caused below average inflows to Upper Klamath Lake, with total inflows 55% and 61% of the 1992-1998 average (Table 3). The water years 1996, 1997, and 1998 had substantially higher inflow than the 7-year average from this study (126%, 131%, and 123%, respectively, while 1993 and 1995 were near the average (108% and 96%, respectively).

As expected when runoff is dominated by springs (versus snowmelt dominated), or by return flows originating from other drainages, the relative water contribution from the Wood River, 7-Mile Canal, and the Williamson R. proper (Williamson R. Total minus Sprague R.) are inflated during low flow years (Table 3). For example, in 1992 Wood River flow was nearly 1/2 that of the Williamson River, while during the next high flow year, 1993, it was only 1/4 of the Williamson River flow.

On an annual basis inflow volume only slightly exceeded outflow volume including evaporation (Table 3), but during the May-September period outflow exceeded inflow by ~1.3X in high flow years and ~2X in low flow years (e.g. 1994; Table 4).

Biweekly time series for major water balance terms are presented in Fig. 5 and Appendix 5. Significant seasonal inflow patterns are evident for all major sources, with the Williamson R. (due to the influence of snowmelt driven runoff from the Sprague River), showing the most pronounced springtime peaks in 1993 and 1995-1998 (Fig. 5). Substantial differences in timing and duration of the seasonal peak are also evident (e.g., as in 1998 when the bulk of the runoff occurred in late spring, as opposed to early spring in other years). The Wood R. and 7-Mile Canal also show seasonal peaks coinciding with snowmelt, but they are less pronounced than those occurring on the Sprague R. (Fig 5). Summertime declines in flow on all systems are influenced by irrigation withdrawal, and it is particularly evident in the spring-dominated Wood R. (which should have relatively constant flow once snowmelt declines) in the 1992 and 1994 water years.

Lake outflow during the snowmelt runoff period is determined by flood control needs and downstream anadromous fishery needs, and in summer months primarily by the irrigation withdrawal schedule on the "A" Canal. This usually causes two outflow peaks, one in late-winter/spring and the other during the irrigation season; the irrigation season outflow increase is accompanied by a decrease in lake volume (Fig. 5). Water residence time (lake volume/inflow) also follows a season pattern of low values during high runoff periods (ca. 50 days) and high values during low runoff periods (up to 300 days; Fig. 5 and Appendix 5). Residence time computed using net inflow (inflow-evaporation) yields significantly higher residence times during the summer months, particularly in 1992 and 1994 when residence time exceeded several thousand days (Appendix 5).

### ***Nutrient Budget***

The Williamson River and Wood River together accounted for 67% (48% and 19%, respectively) of the mean 1992-1998 (WY's) total phosphorus load; with springs and small

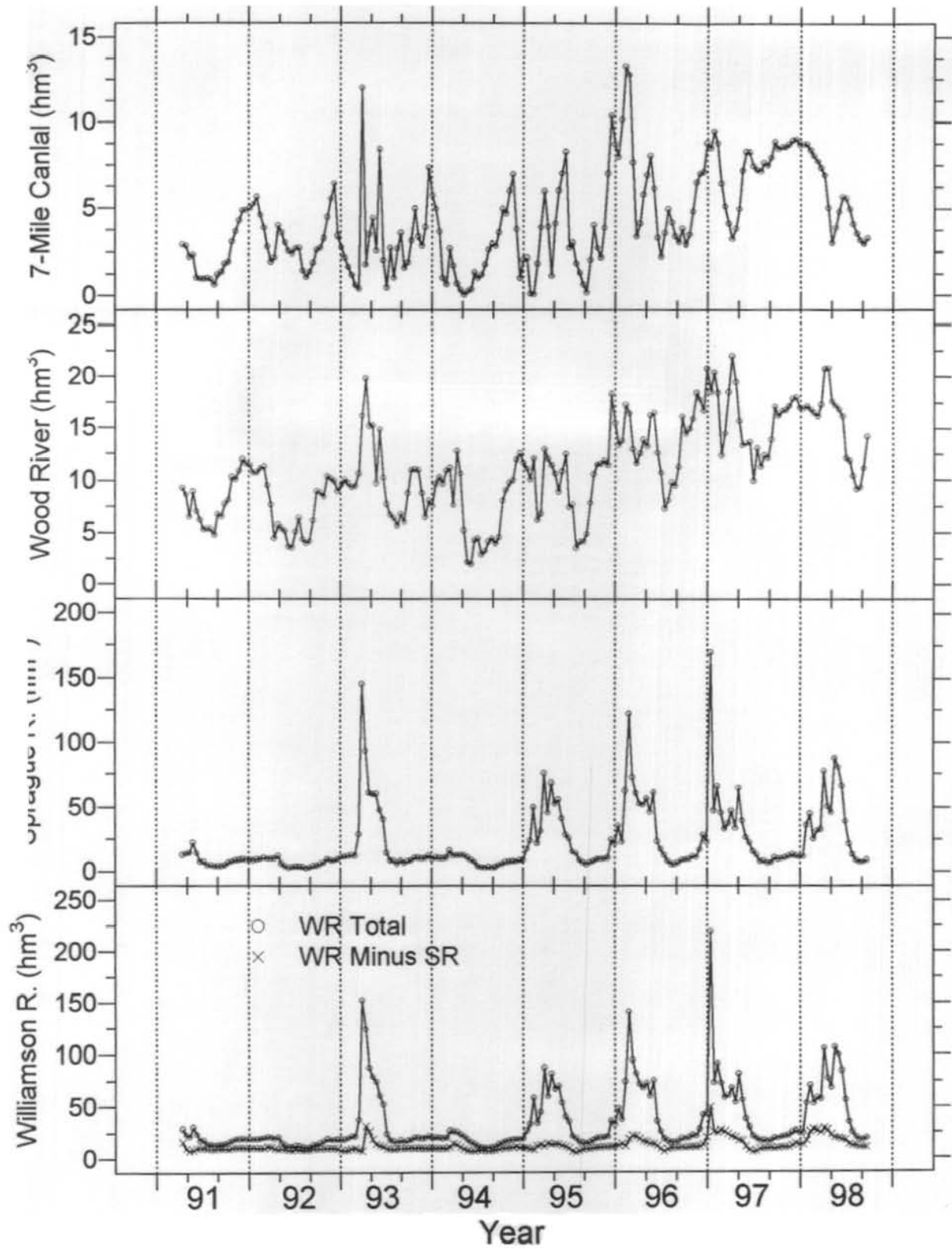


Fig. 5. Biweekly time series of water balance terms, 1991-1998.

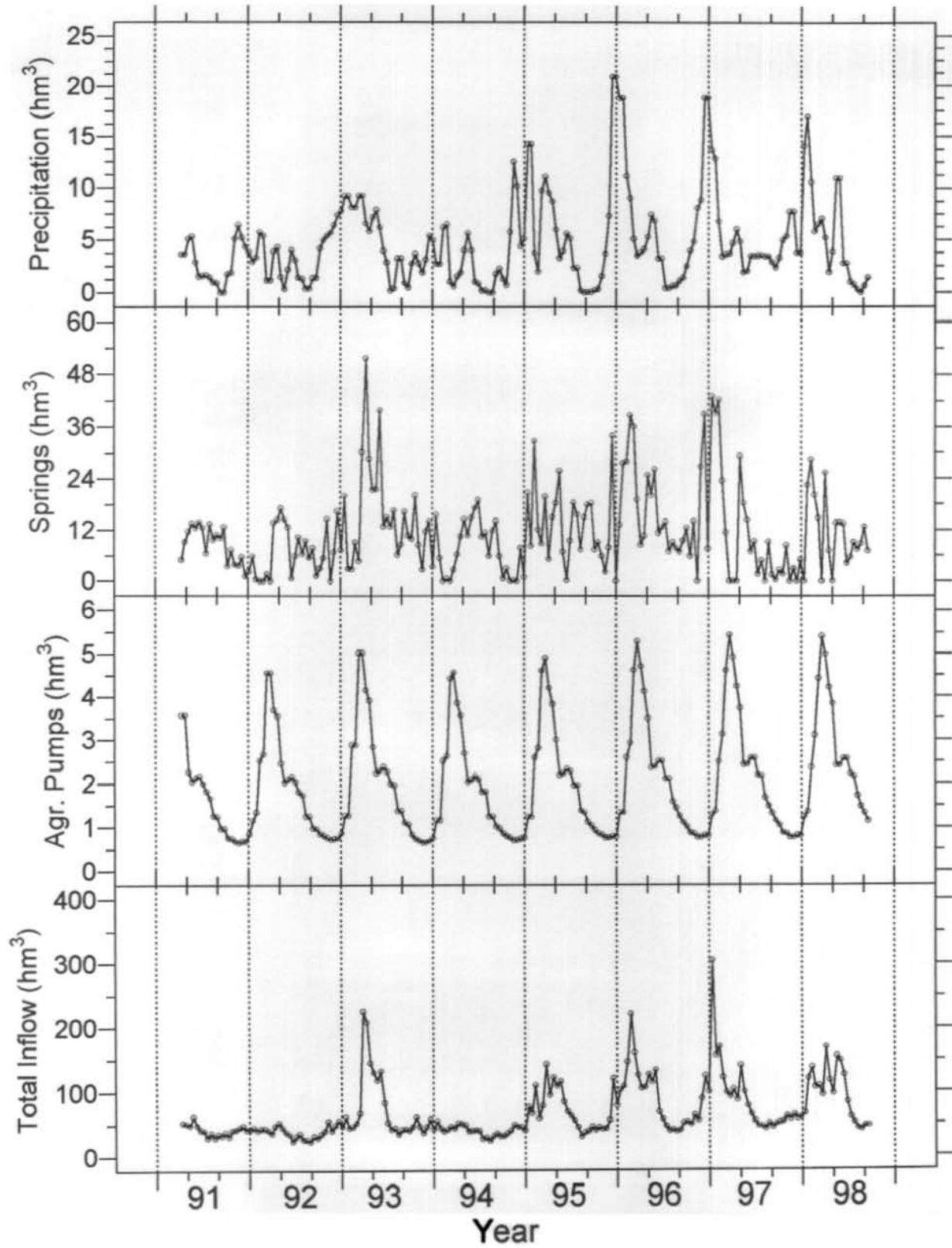


Fig. 5. (continued).

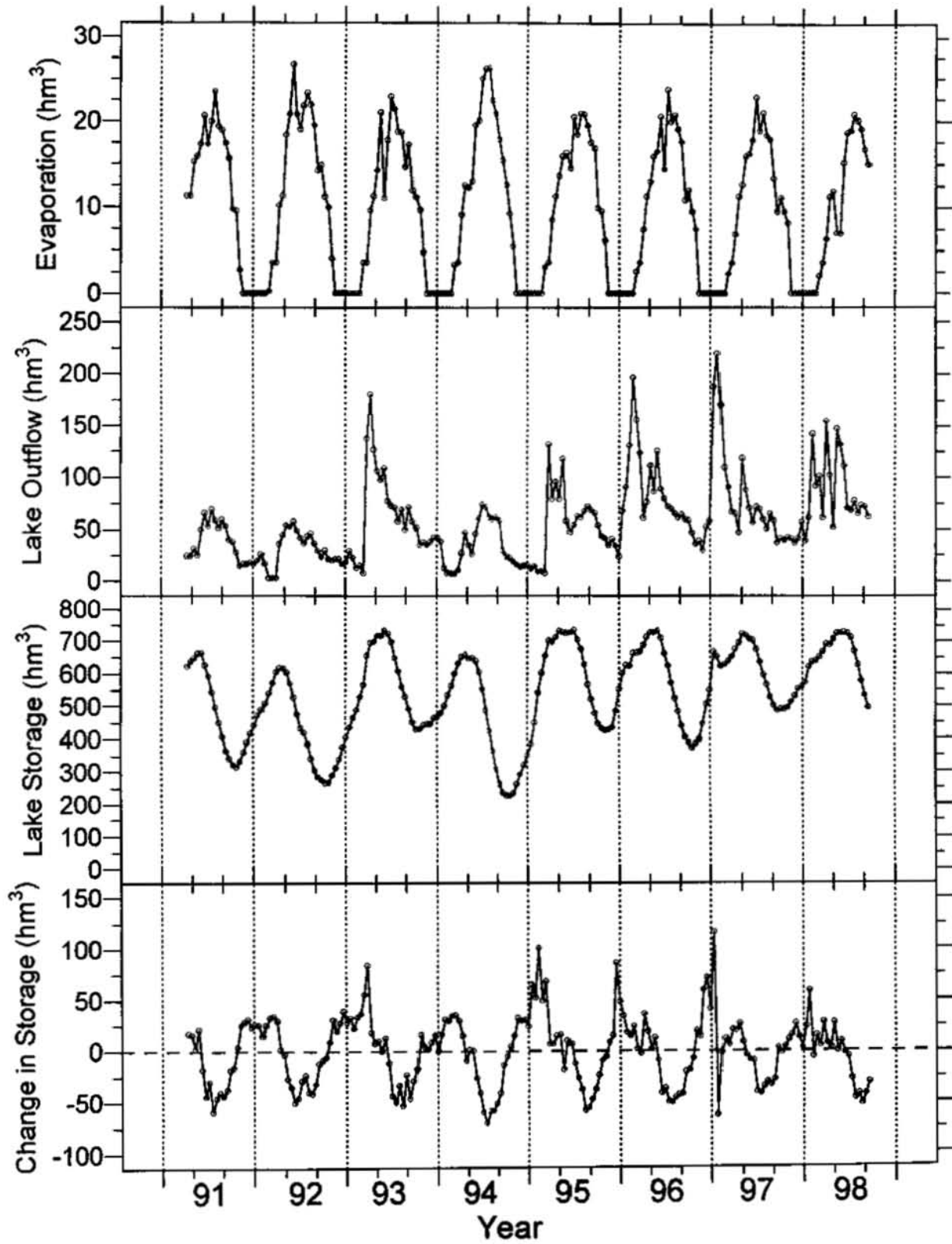


Fig. 5. (continued).

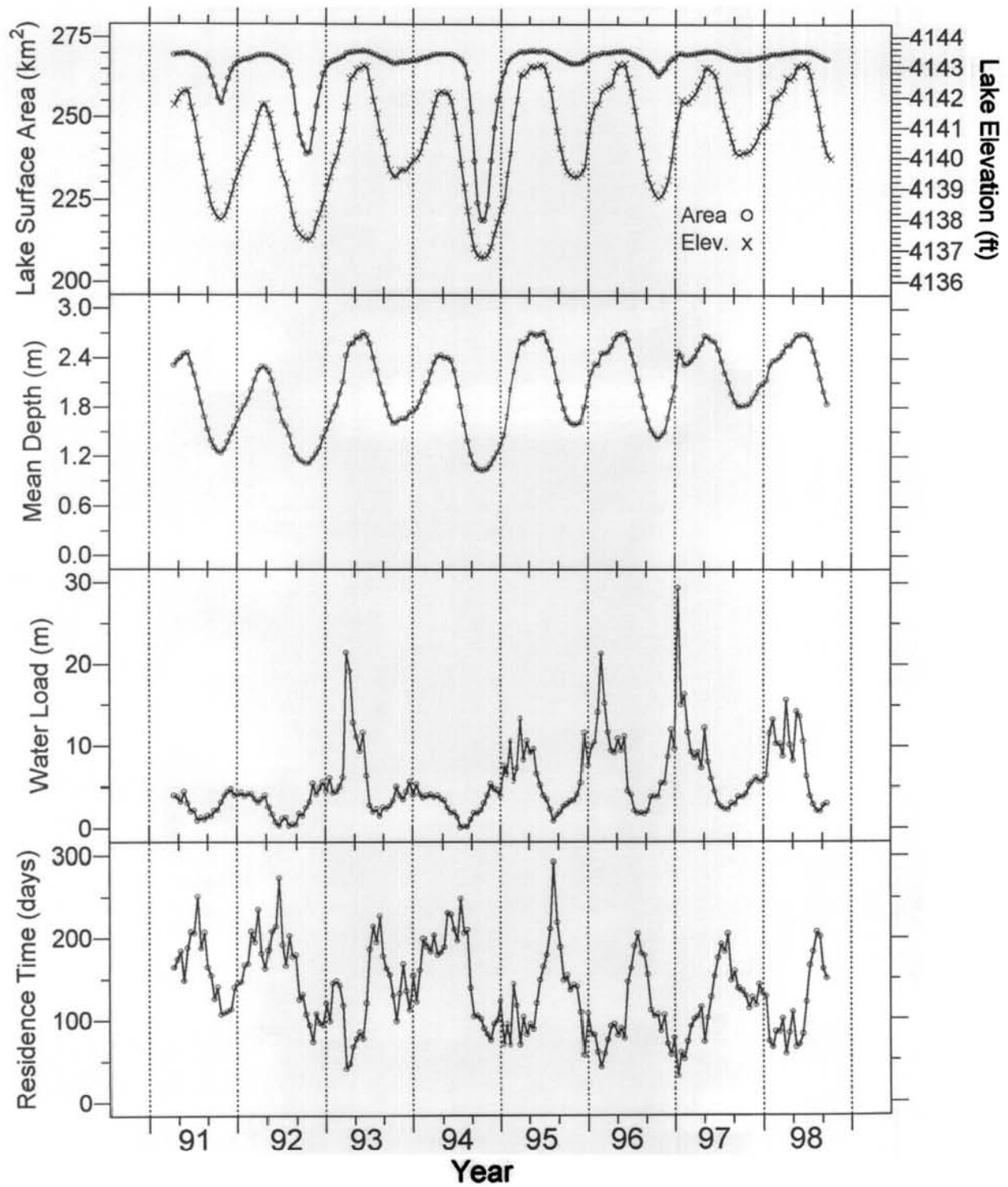


Fig. 5. (continued).

ungaged tributaries contributing another 10% (Fig. 6; Table 3). 7-Mile canal, precipitation, and agricultural pumping accounted for the remaining 23% (Fig. 6; Table 3). Unlike water contribution, where the Wood R., 7-Mile Canal, and Pumps contributed 25% of the water load, these same sources contributed 39% of the average annual TP load (Table 3). In contrast, springs contributed 16% of the water input, but contributed only 10% of the TP input. This appears to be partially due to the consistently higher volume weighted TP concentration occurring in the pump effluent, and Wood R. and 7-Mile Canal systems (Table 3; Fig 7). As stated above, these systems have small drainage areas and are influenced by return flows originating outside their respective drainages. This, combined with higher volume weighted concentrations, leads to substantially higher unit area export of TP (Fig. 6).

The average contribution of TP load from pumping was 11% and ranged between 9 and 17% over all years (Table 3). This occurred despite contributing an average of only 3% of the water load. During the peak pumping period occurring Feb-May, pumping can contribute as much as 32% of the total TP load (Appendix 6). It is important to note that these pumping estimates are not representative of the total pumping contribution because the load from pumps draining the Wood River Ranch is already included in the 7-Mile Canal and Wood R. loading estimates.

The disproportionate loadings from pumped areas as well as from the smaller 7-Mile Canal and Wood R. drainages illustrates the management importance of these areas. For example, these drainages combined are 606 km<sup>2</sup> in area, which is nearly 13 times smaller than the 7739 km<sup>2</sup> Williamson River watershed, yet they provide a TP load only 15% lower than the Williamson TP load (73 metric tons vs. 86 metric tons – average annual all years; Table 3).

The estimate of anthropogenic contribution to TP loading averaged for all 7 water years is 40%, with a range of 36 to 45% for individual years (Table 3). These values are very similar to the 40% anthropogenic TP contribution estimated by Walker (1985).

TP loads during the 1992 and 1994 drought years were 62% of the 1992-1998 average (Table 3); the water years 1996, 1997, and 1998 were 133%, 121%, and 115% of the 7-year average, while 1993 and 1995 were 114% and 93% of the average, respectively. The 1993 WY is of note because while flow was 108% of the 7-year average, TP load was 114% of the average. Other years (with the exception of 1996) tended to have percent of average TP loads lower than their respective percent of average water inputs. It may be that during several low flow years (e.g., 1991 and 1992), watershed sources of TP accumulate, and are then flushed in a following high flow year. Moreover, the volume weighted TP concentration of the Sprague R. in 1993 is higher than any other year (Table 3), indicating additional watershed contributions of TP. Because the Sprague R. watershed is impacted by wetland and riparian loss, flood plain grazing, agriculture, and channel degradation, it would be prone to TP export, especially during major runoff events.

Biweekly time series for major TP balance terms are presented in Fig. 8 and Appendix 6. As with flow, significant seasonal inflow patterns are evident for all major sources, with the Williamson R. (due to the influence of snowmelt driven runoff from the Sprague River), showing

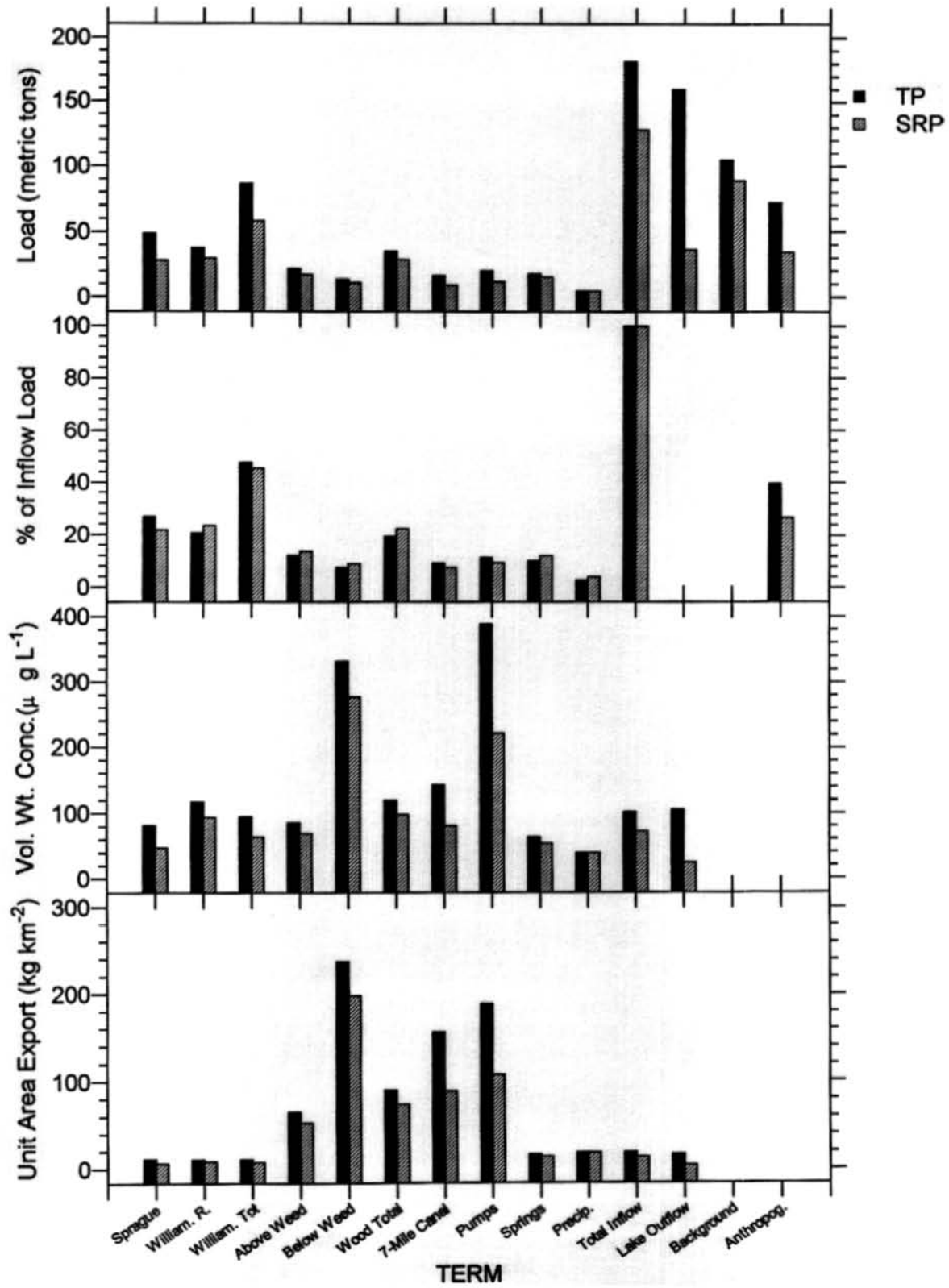


Fig. 6. Average annual phosphorus balance for water years 1992-1998

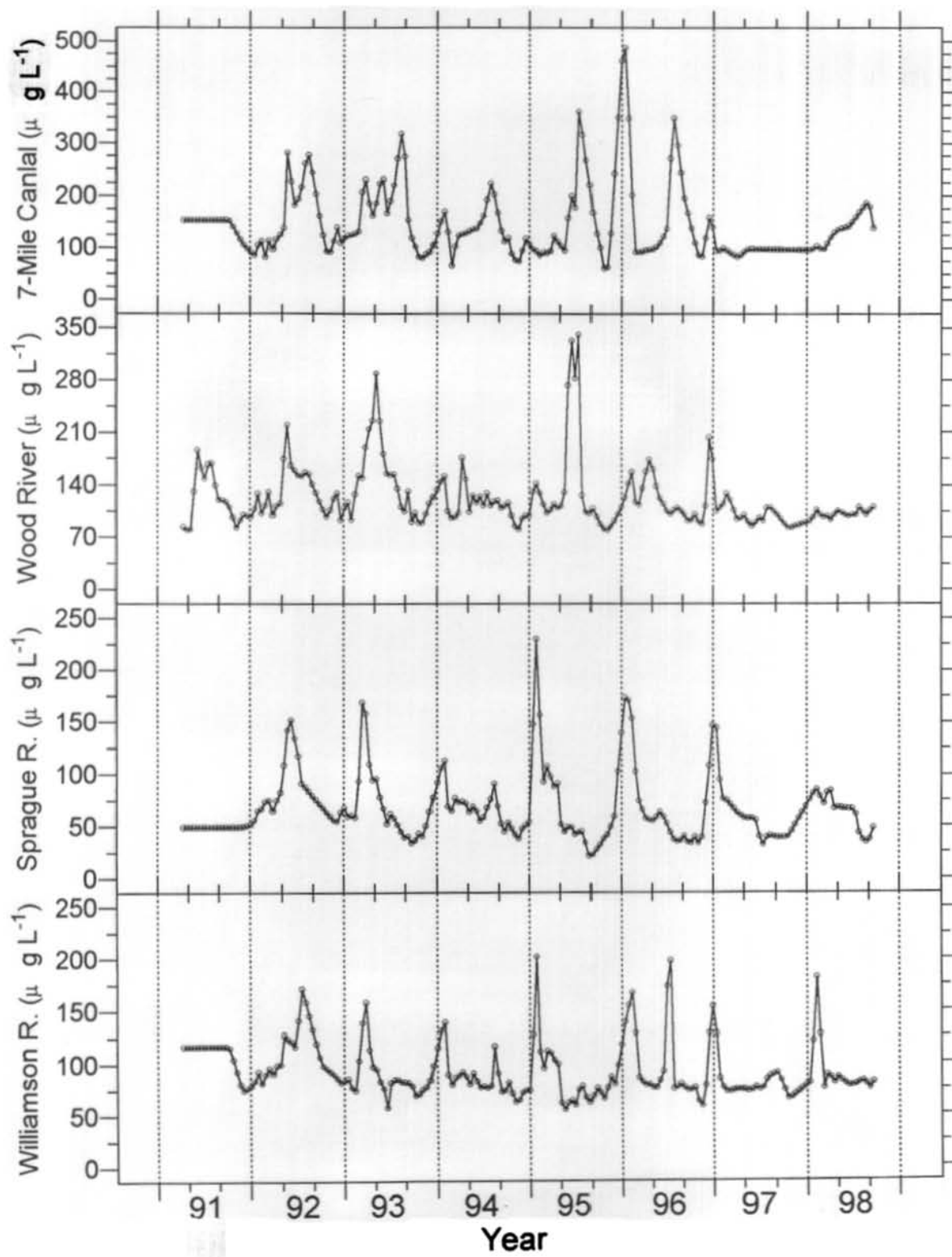


Fig. 7. Biweekly time series of volume weighted total phosphorus concentration (1991-1998).

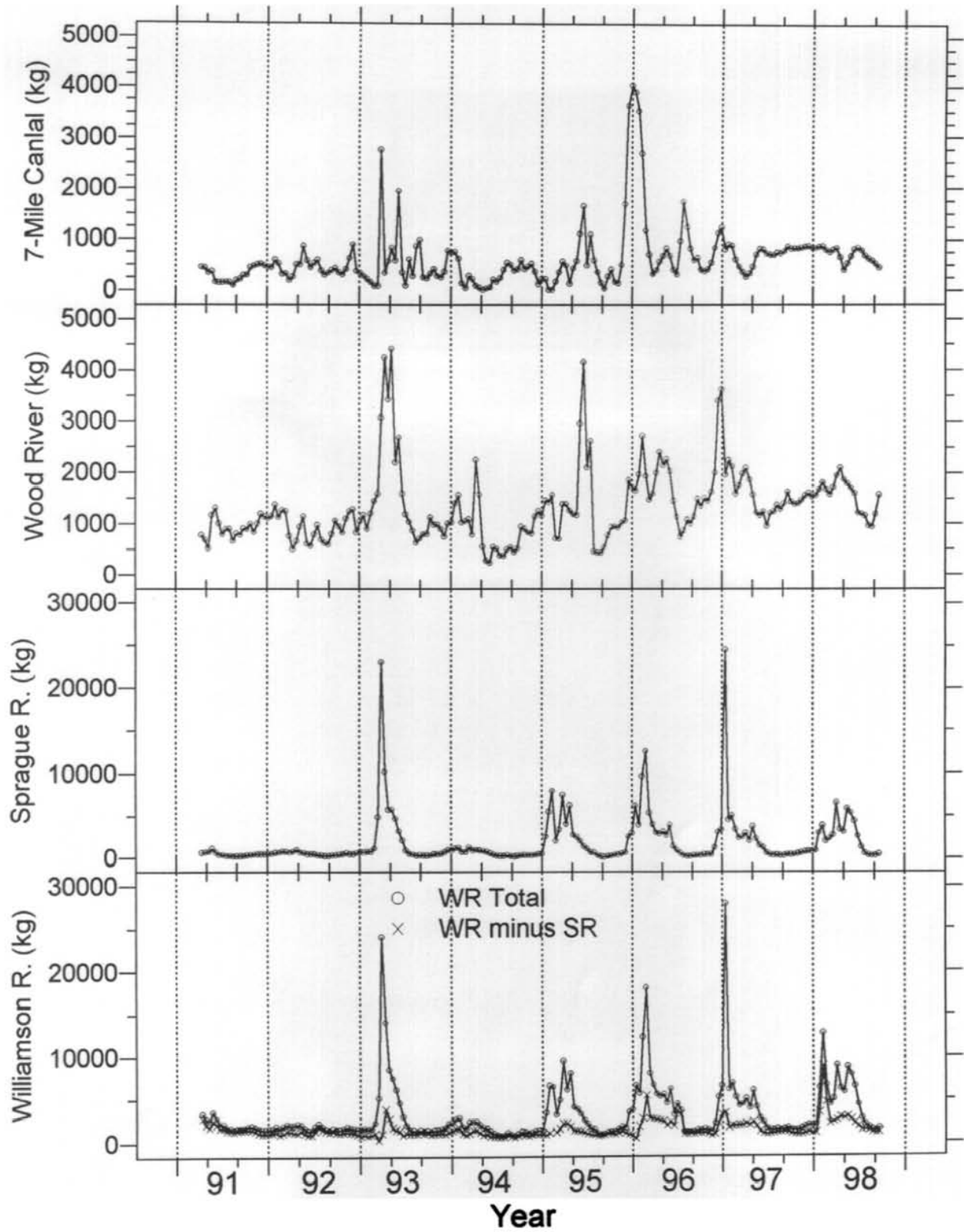


Fig. 8. Biweekly time series of phosphorus balance terms (1991-1998).

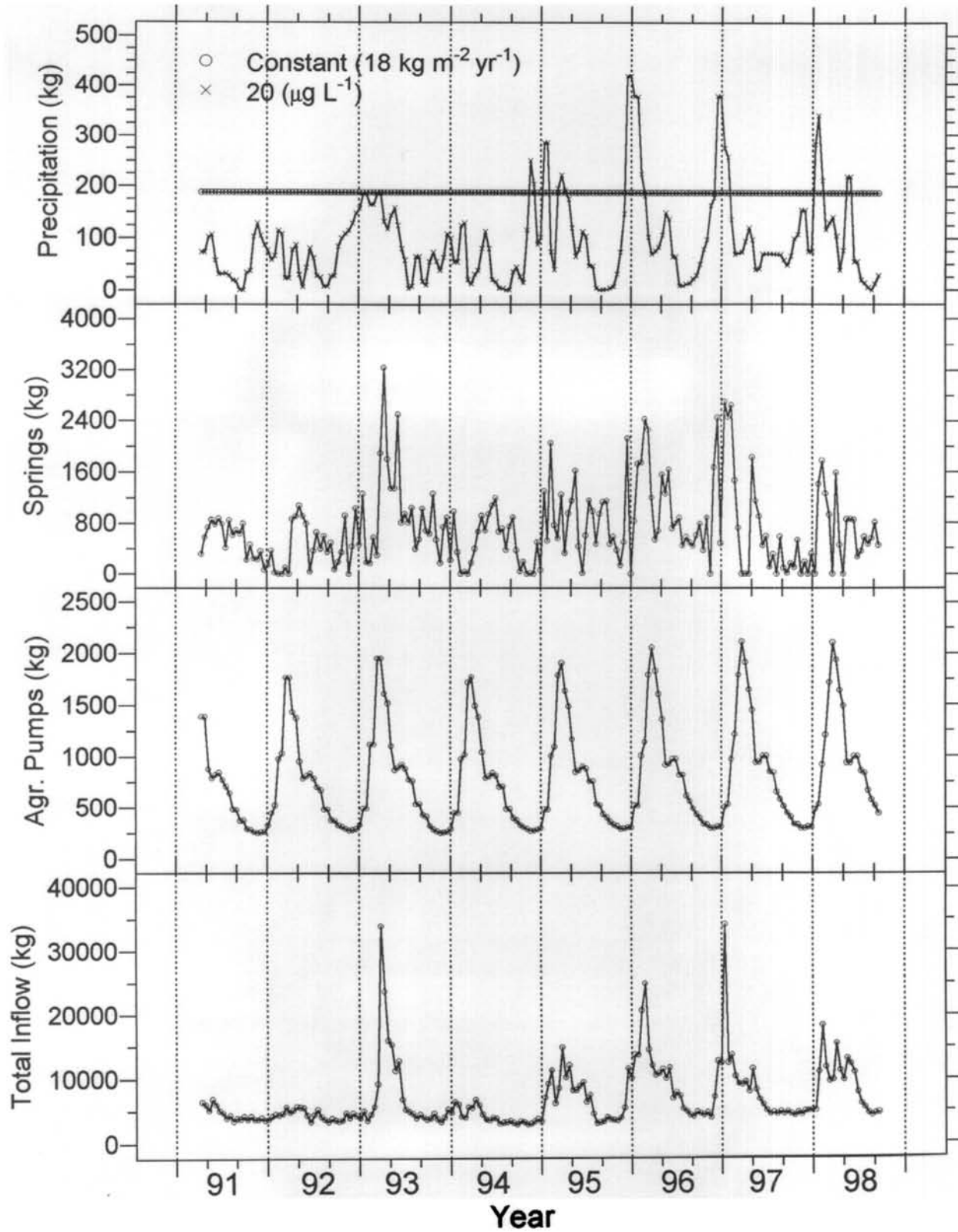


Fig. 8. (continued).

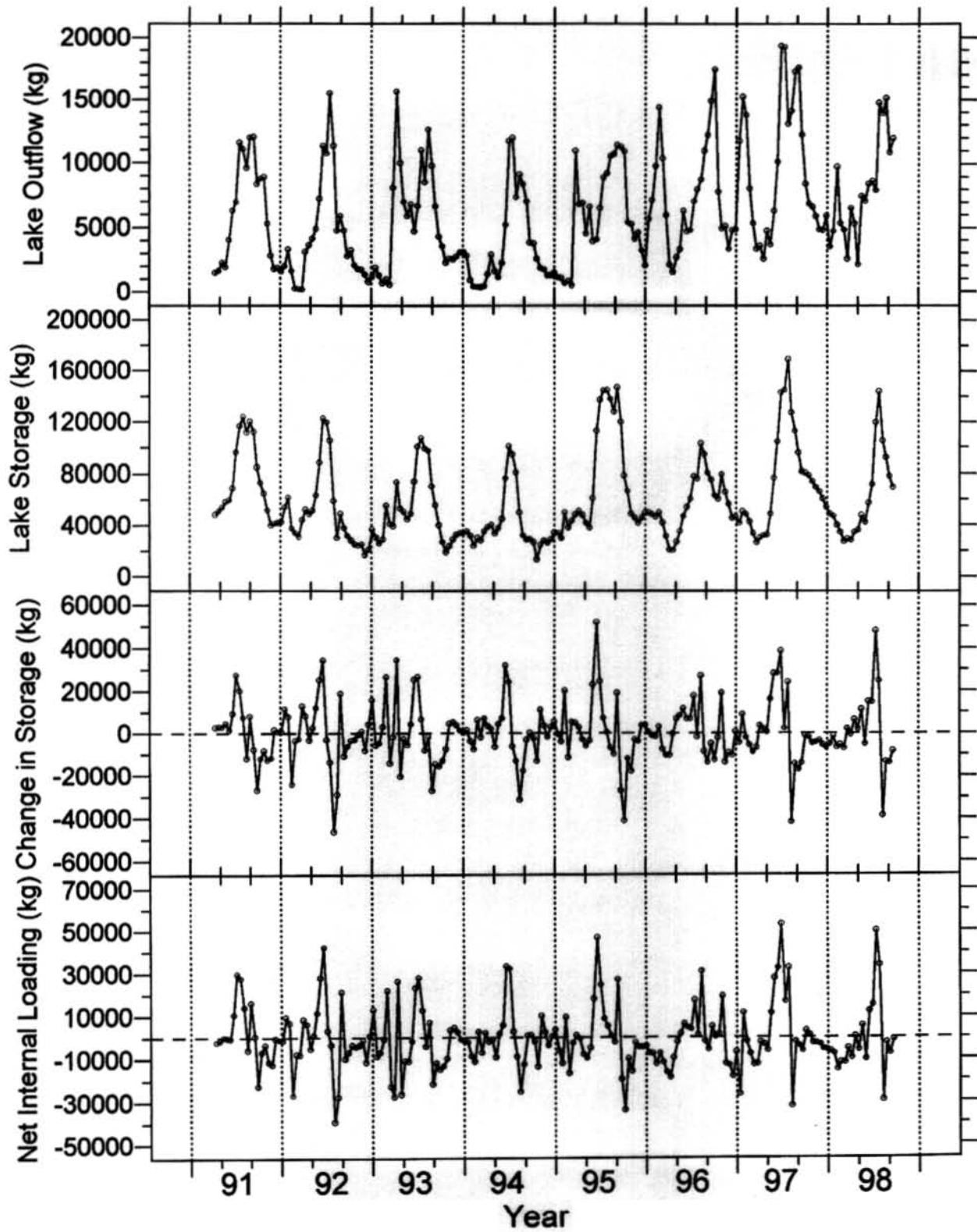


Fig. 8. (continued).

the most pronounced springtime peaks in 1993 and 1995-1998 (Fig. 8). The Wood R. and 7-Mile Canal also show some seasonal peaks coinciding with snowmelt, but they also show other midseason spikes likely due to irrigation return flow. It is evident that the Sprague River contributes the majority of the Williamson R. load during peak flows, but that the Williamson proper (minus Sprague) contributes the majority of the load during low flow summer months.

An estimate of the particulate phosphorus load (PP) was taken as the TP load minus the SRP load (Fig. 9). These data clearly show an increase in the loading of PP during high runoff events for the Williamson and Sprague Rivers. During these high flow events, which typically occur January-May, PP can increase to 60% of the TP load, compared to less than 5% during summer low flow periods (Fig. 9). There are also noticeable spikes of PP load occurring in the Wood R. and 7-Mile Canal systems, but they are not limited to high runoff periods. This pattern could be consistent with flood irrigation practices that would tend to be pulsed in nature, and where overland runoff could increase the proportion of particulates. The increase in PP loading as well as the increase in volume weighted concentration during elevated flows (Figs. 7 and 9) are both indicative of degraded watershed conditions. In a healthier watershed (e.g., intact riparian areas and flood plains) the concentration should tend to decrease at high flows through dilution, and particulate loading should only increase slightly.

Lake outflow TP load tends to increase during high runoff events in the winter and spring, as well as during the summer period when inflow load is low (Fig. 8). It is clear from this trend, and the increase in lake TP storage (at a time when lake water storage is decreasing), that TP is being internally loaded from the sediments. This is confirmed by the large net internal loading occurring during late spring and early summer of each year (Fig. 8). These large net internal loading events are generally followed by a substantial decline, indicating a large sedimentation event. Such events coincide with algal bloom crashes (Kann 1998). An estimate of the magnitude of the internal loading among years can be gained by summing all positive growing season internal loading values calculated using the 0.14/week (0.01/day) sedimentation rate. The values for each of the 1992 through 1998 water years are as follows:

WY	Int. Load	Ext. Load	Tot. Load	% Ext.	% Int.
1992	294	113	407	28%	72%
1993	265	208	473	44%	56%
1994	195	112	307	36%	64%
1995	394	169	563	30%	70%
1996	212	241	453	53%	47%
1997	376	220	596	37%	63%
1998	257	208	465	45%	55%
Average	285	182	466	39%	61%

On average, external loading was 39% of the total loading to the lake, while internal loading was 61%. On an annual basis there tends to be a net retention of TP in the lake due to the significant sedimentation events from algal crashes and the likely settling of PP during high runoff (e.g., annual average retention is 25 metric tons; Table 3). However, it is evident from the negative retention (positive internal loading) during the May through September period (Fig. 8; Table 4)

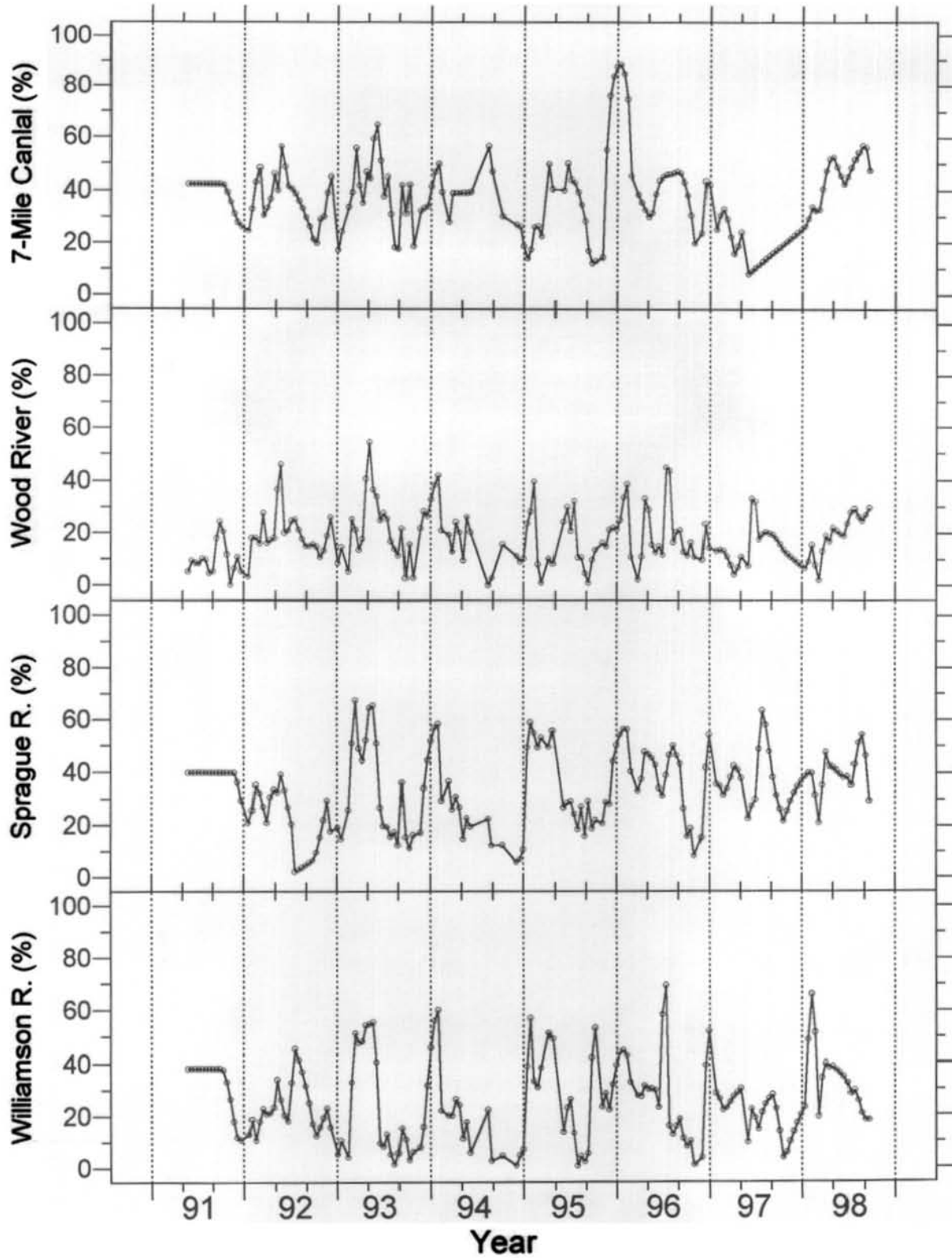


Fig. 9. Biweekly time series of particulate phosphorus load (PP) as a percent of the total phosphorus load (1991-1998).

that internal loading is a significant source of phosphorus to the lake. Although there is a high contribution of internal TP loading to lake TP concentration during the algal growing season, it has been noted that the mobilization of phosphorus from iron has the potential to respond rapidly when primary productivity and pH maxima are reduced (Marsden 1989). The rapid response may be due to a reversal of the positive feedback mechanism described earlier (see introduction).

Using a constant loading rate for the precipitation input (Fig. 8) causes error in the seasonal mass balance dynamics, and would most affect estimates of net internal loading during the summer months. It is likely that using a constant loading rate overestimates precipitation input during this period of low precipitation (Fig. 5). To determine the relative effect of this error a concentration of  $20 \mu\text{g L}^{-1}$  TP was applied to the precipitation volume to determine seasonal load distribution (Fig. 8). Given the magnitude of the net internal loading during the summer period, the error induced by an overestimation of precipitation TP load would likely be only 2-3% on the average.

The total nitrogen balance (Fig. 10 and 11; Tables 3 and 4) indicates that Upper Klamath Lake is a seasonally significant source of nitrogen. On an annual basis there is a net negative retention of TN (average annual = -143% of external inputs; Table 3). On a seasonal basis the 1992-1998 WY range is between -259% and -627% of external inputs (Table 4). This shows that internal sources of nitrogen far exceed external sources. The WY's 1995, 1996, and 1997 had internal nitrogen loading higher than other years, with 1997 nearly 2X higher than the average for this period. The main source for this increase in internal nitrogen loading is nitrogen fixation by the blue-green alga *Aphanizomenon flos-aquae* (Kann 1998). Another potential source is the mobilization of inorganic nitrogen from lake sediments during anaerobic bacterial decomposition.

Biweekly time series for SRP and TIN balance terms are presented in Appendix 6, but will not be discussed here. While these data will be useful for integrating with lake water quality and algal dynamics, the main purpose here was to construct a TP mass balance budget for integration with other TP loading models.

Additional exploration and statistical analysis will enhance our ability to understand dynamics in the Klamath Lake system, but the main purpose of this phase of the study was to formulate and present a mass balance budget for water and phosphorus. The mass balance models presented here are based on rigorous data collection and provide good representation of seasonal and interannual dynamics. They thus will provide a solid base for future nutrient, algal biomass, and water quality modeling efforts.

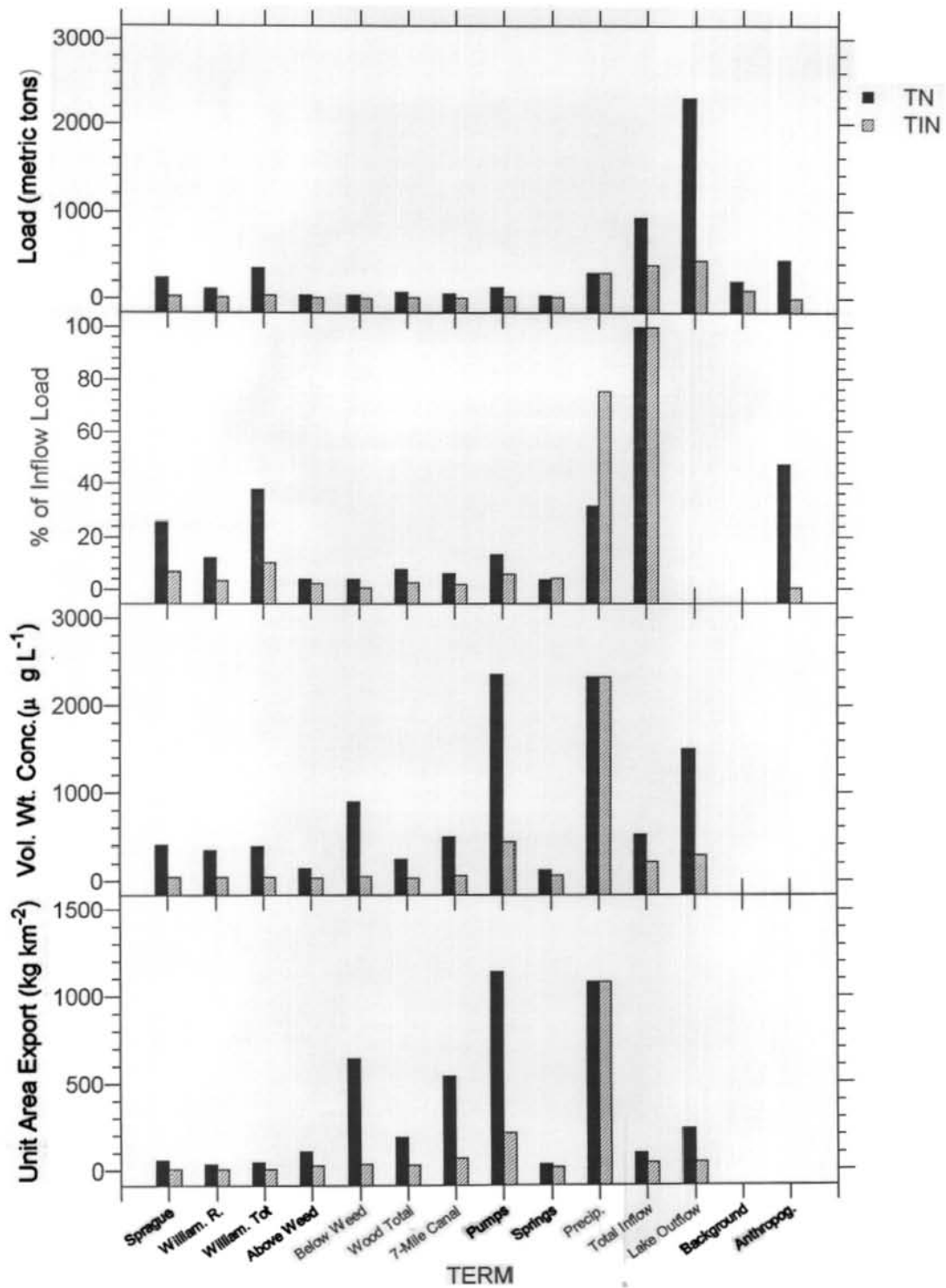


Fig 10. Average annual nitrogen balance for water years 1992-1998.

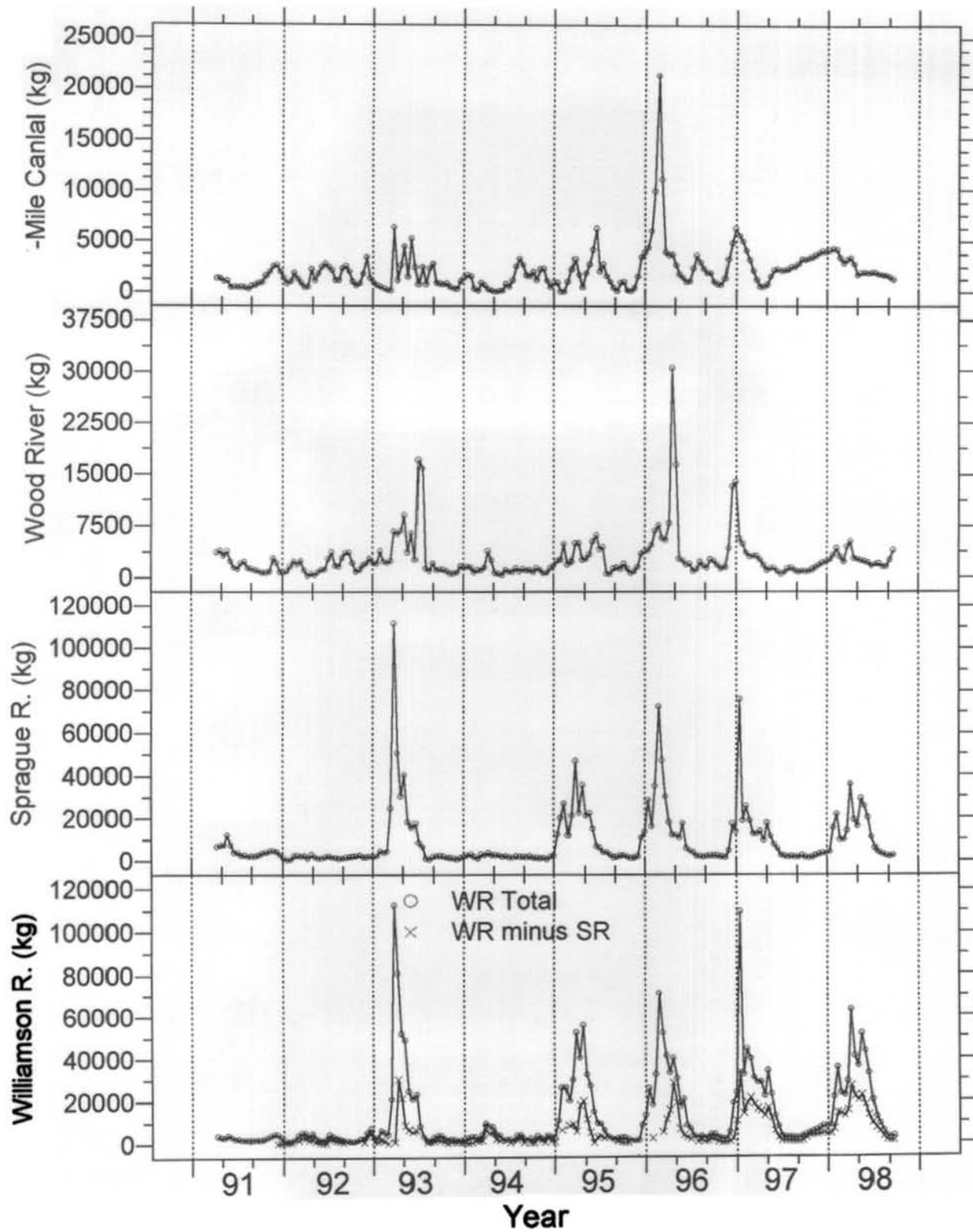


Fig. 11. Biweekly time series of nitrogen balance terms (1991-1998).

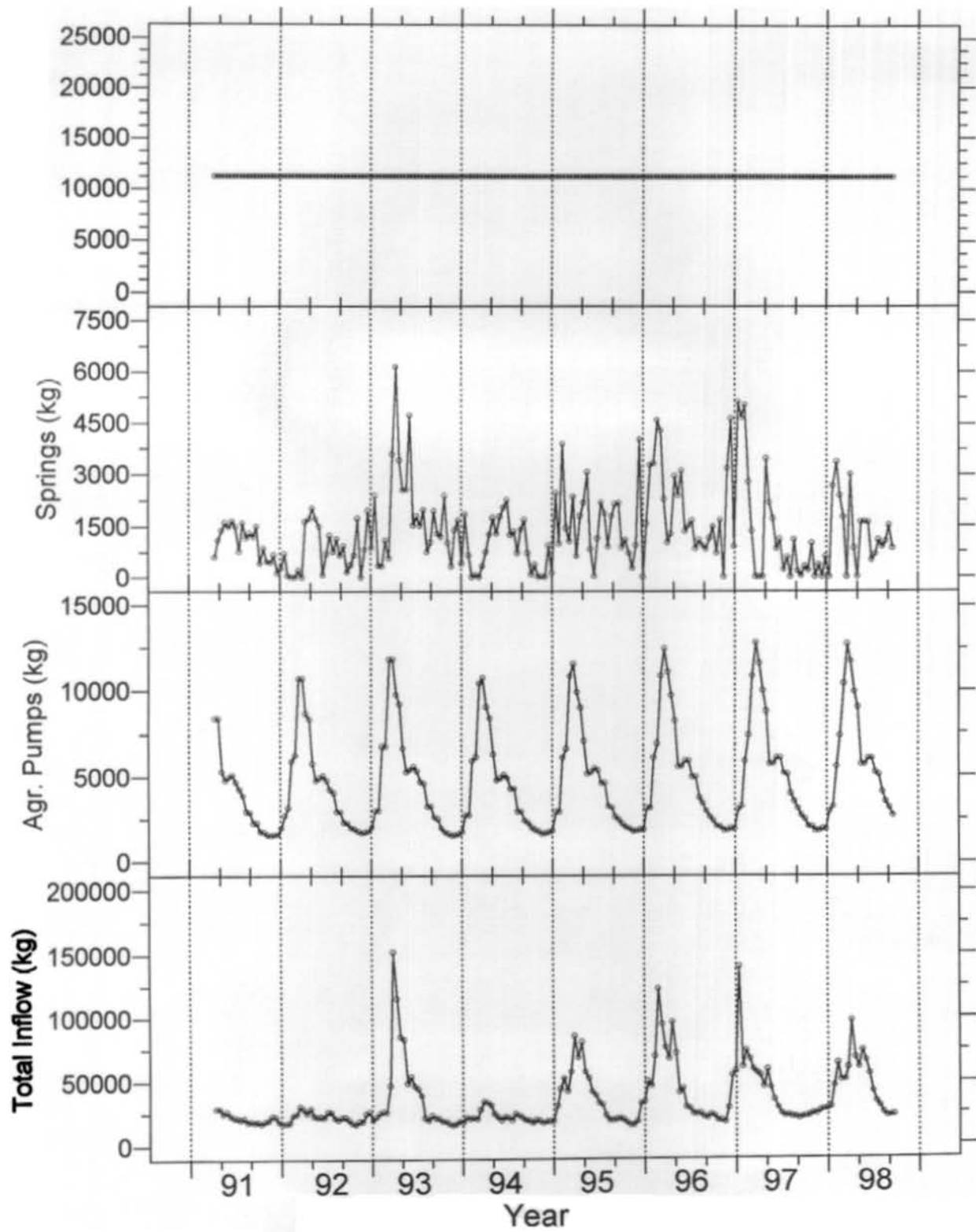


Fig. 11. (continued)

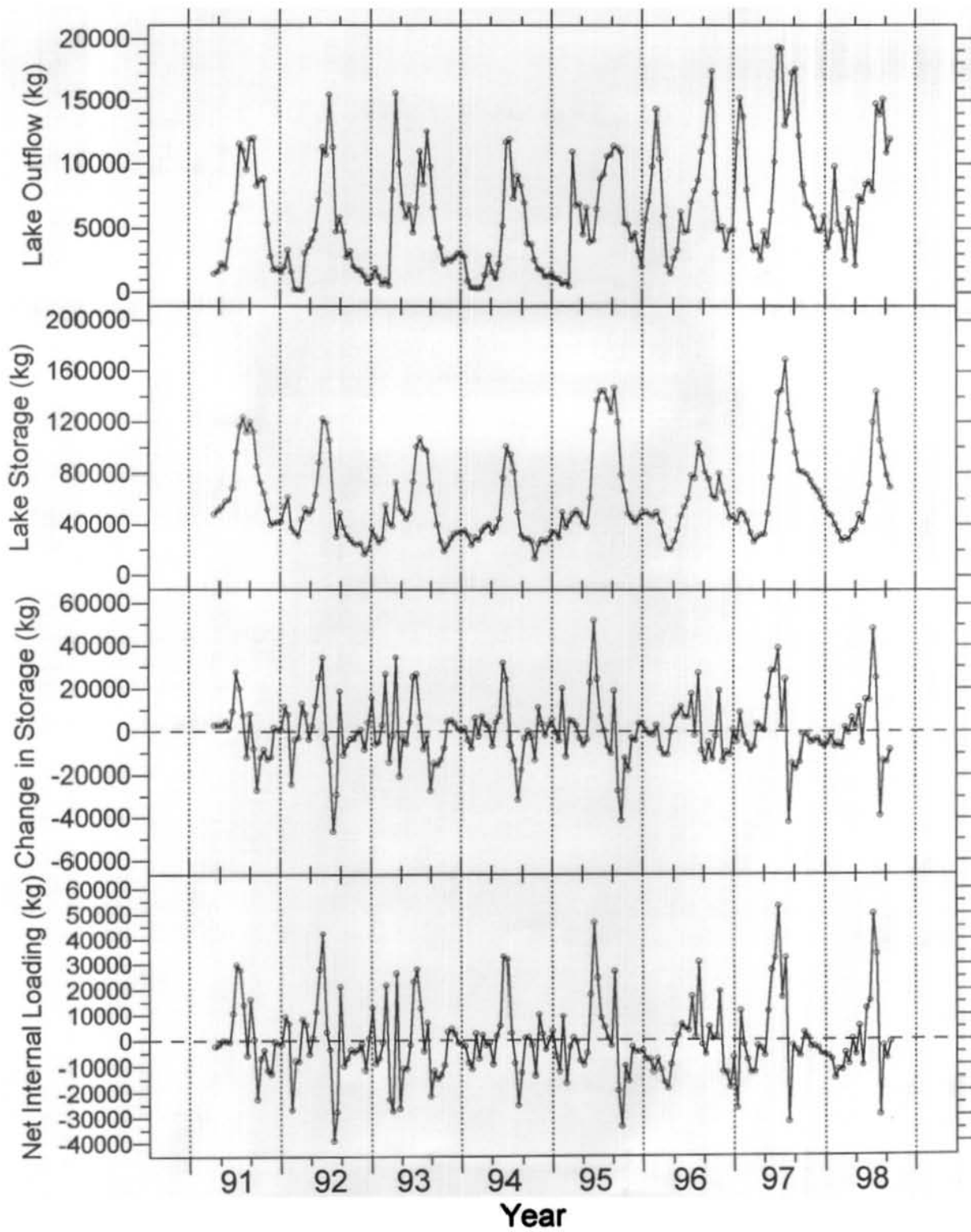


Fig. 11. (continued)

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**APPENDICES**  
(attached electronically)

- I. Lake-wide mean nutrient concentrations, 1991-1998.**
- II. Tributary nutrient loading and water discharge data, 1991-1998.**
- III. Estimation of daily flow for 7-Mile Canal, Wood R. at Dike Rd., and Wood R. at Weed Rd.**
- IV. Procedure for estimating flow and nutrient loads from agricultural pumps.**
- V. Biweekly time series for water balance terms.**
- VI. Biweekly time series for nutrient balance terms.**