less than in 1974 (580 $mg/m^2$-day), estimated 7 years after wastewater diversion was complete. This decrease was not evaluated statistically due to the limited number of historical values.

There are two important advantages in using AHOD as an indicator of lake quality:

1. AHOD determines the oxygen demand rate in the hypolimnion due to bottom sediments and settled particulate matter, both largely the result of algal production of organic matter in the epilimnion and littoral regions.

2. AHOD normalizes for hypolimnetic depth by expressing the rate in areal units to enable lake-to-lake comparison, regardless of DO concentration.

The second advantage of using AHOD is especially pertinent to Lake Washington and explains why anoxia did not develop in the lake prior to wastewater diversion. Prior to diversion, the volumetric rate of DO depletion was 0.043 mg/L-day (AHOD of 810 $mg/m^2$-day/19 m). At that rate, zero DO throughout the hypolimnion would be reached in 233 days, although anoxia would have occurred sooner near the sediment surface. Anoxia did not result because stratification did not persist for that long. Lake Washington remains stratified for 150 to 180 days. However, if hypolimnetic depth were only half of 19 m, anoxia would have been reached in 116 days, sooner near the bottom, and easily within the stratified period.

For comparison, AHOD in Lake Sammamish declined from a mean of 423 $mg/m^2$-day before and shortly after wastewater diversion (1968) to a mean of 312 $mg/m^2$-day from 1974 to 1984, but it continues to experience anoxia each year (Welch et al., 1996). The reason Lake Sammamish reaches anoxia, but Lake Washington does not, is its shallower mean depth, which is half that of Lake Washington. So there is half the hypolimnetic volume and half the DO to oxidize organic matter settling through the water column and accumulated in the surficial sediment. As a result, Lake Sammamish goes anoxic every summer in spite of the lake having a lower AHOD than Lake Washington.

The Lake Washington hypolimnion remains oxic during the stratified period as shown by isopleths of DO for all data from 1993 to 2001 (Figure 13). Minimum DO observed near the bottom did not drop below 2.5 mg/L. This indicates that the sediment-water interface did not go anoxic. At a level of DO >2.5 mg/L above the sediment-water interface, the potential for P release from sediment is minimized by maintaining P in a bound form with iron. Also of interest in Figure 13 is the high epilimnetic DO during March and April. The high DO signifies that mild supersaturation occurred in the range of 110-120%, a result of the photosynthetic activity of the spring algal bloom. The figure illustrates that high DO concentrations occurred at a depth of 30 m or more in the spring. Water column mixing at that time probably carried the supersaturated water, which was produced in the photic zone, to depths well below the photic zone.

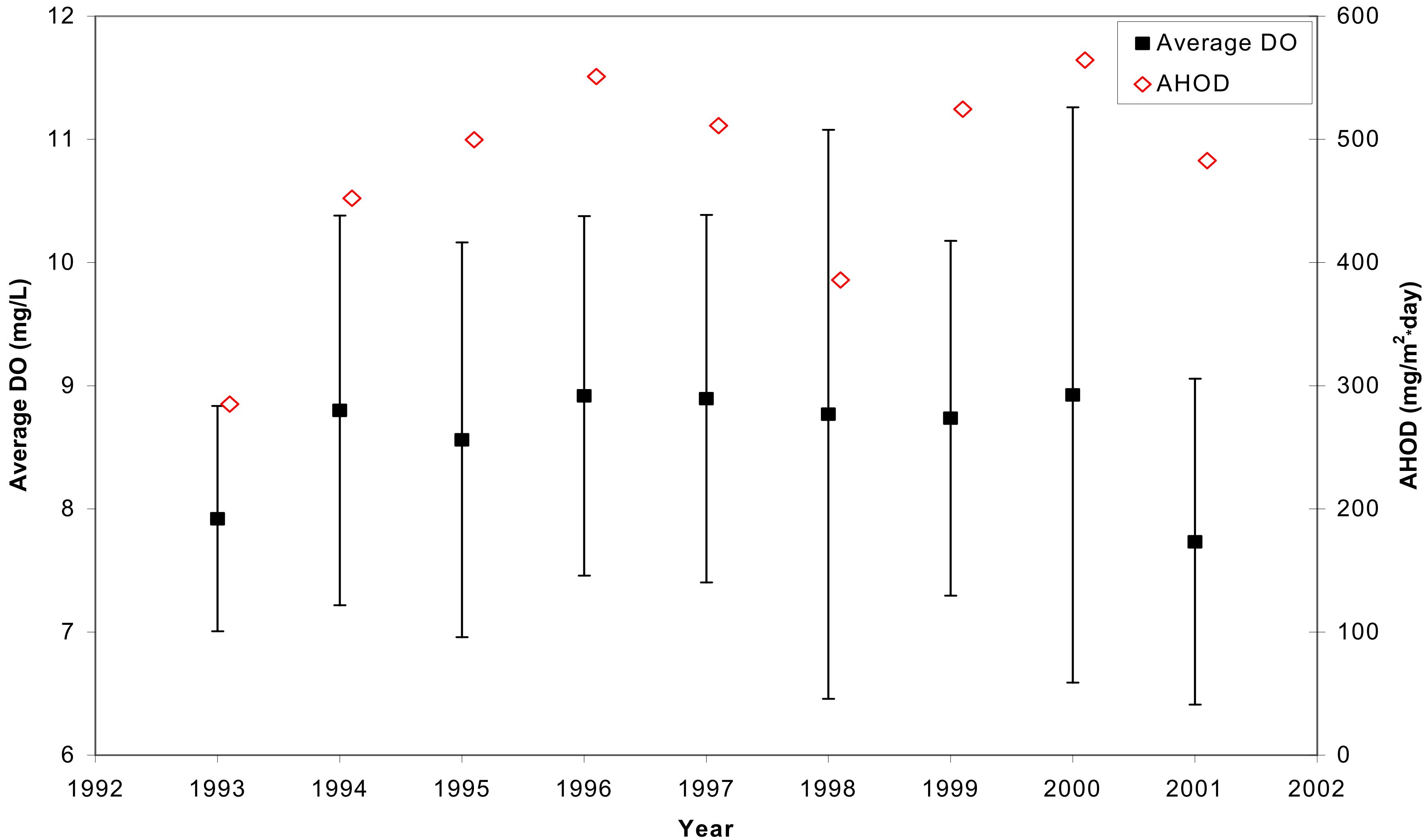


**Figure 12. Mean AHOD and Dissolved Oxygen Concentration (+/- SD) in the Hypolimnion of Lake Washington for Periods of Stratification (May Through October) From 1993 to 2001**

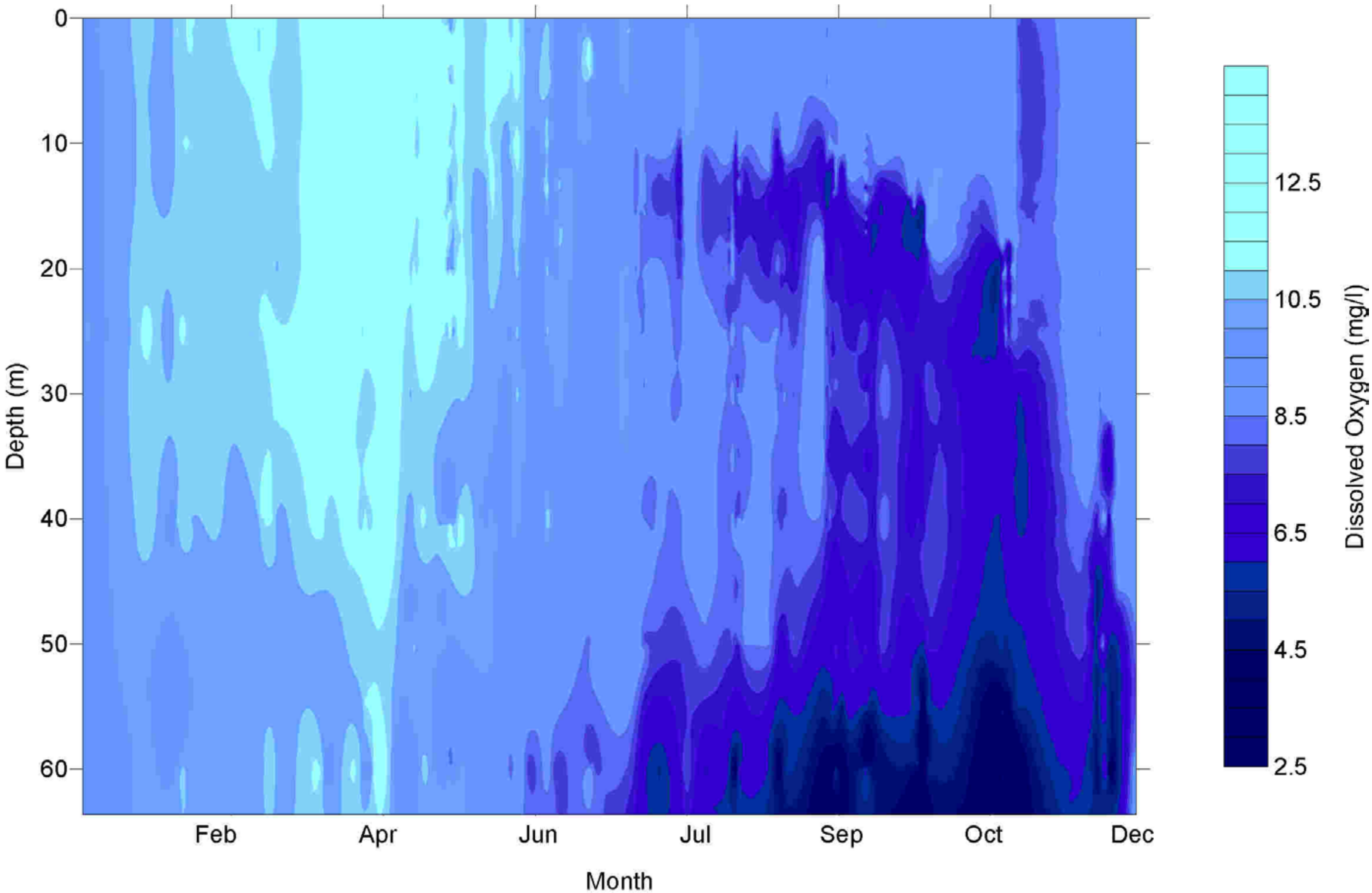


**Figure 13. Dissolved Oxygen Profile at the Deep Water Station (0852) in Lake Washington for all Data From 1993 to 2001**

### 4.2.2. Conductivity

Lake Washington conductivity was measured during the study period from 1992 through 2001. There was no pattern in the conductivity data by station, depth, or time. The conductivity ranged from a low of 60 to a high of 173 μmhos/cm, averaged between 80 and 108 μmhos/cm, and was around 90 μmhos/ml throughout the year and at all depths. This range is typical of soft water, lowland Puget Sound lakes and compares with conductivity observed during a nearshore study of Lake Washington conducted in 1981 through 1984 (METRO, 1985).

### 4.2.3. Alkalinity

During the study period between 1990 and 2001, the mean alkalinity for each station ranged from 34 to 40 mg/L. Annual alkalinity means are presented for the deep water station (0852) in Figure 14. The alkalinity at this station was similar to that observed throughout the lake (see Appendix A for station means and standard deviation). The annual variability probably reflects the inflow alkalinity, which is a measure of bicarbonate leaching from the watershed and can vary depending upon the intensity and timing of precipitation. Alkalinity in Lake Washington has increased over the past few decades, possibly in response to increased soil disturbance within the watershed (Edmondson, 1994).

### 4.2.4. pH

The pH of Lake Washington ranged from a low of 6.4 to a high of 9.2 between 1992 and 2001. The pH profile for the deep lake station (0852) for data collected between 1994 and 2001 is presented in Figure 15 (note: only partial monitoring at this station was conducted until 1994 and only complete data years were included in Figure 15). The pH profile illustrates several basic characteristics about the lake. The pH was only observed to be less than 7.0 in the top 10 m during late fall and early winter when the lake is completely mixed and photosynthetic activity is limited by light. In addition, the low pH throughout the water column is in part due to the mixing of low pH hypolimnetic water with the epilimnetic water at overturn. The pH in the hypolimnion tends to decrease to the mid to high 6 range as the stratified period progresses from spring to late summer due to respiration and the degradation of organic materials.

The pH profile in Figure 15 illustrates that pH is more dynamic than temperature (see Figure 4) during periods of mixing, because unlike temperature, the pH varies vertically in late winter through spring. The increase in pH from 6.4 to 6.8 observed in winter to a pH approaching neutrality (7.0) is due to mixing (allowing carbon dioxide to escape to the atmosphere) and the influence of alkalinity (buffering the pH). In the spring (April through June), as thermal stratification is established, pH in the surface waters reaches maximum levels in response to maximum photosynthetic production. The pH remains relatively elevated in the epilimnion through the summer period, coinciding with photosynthesis.

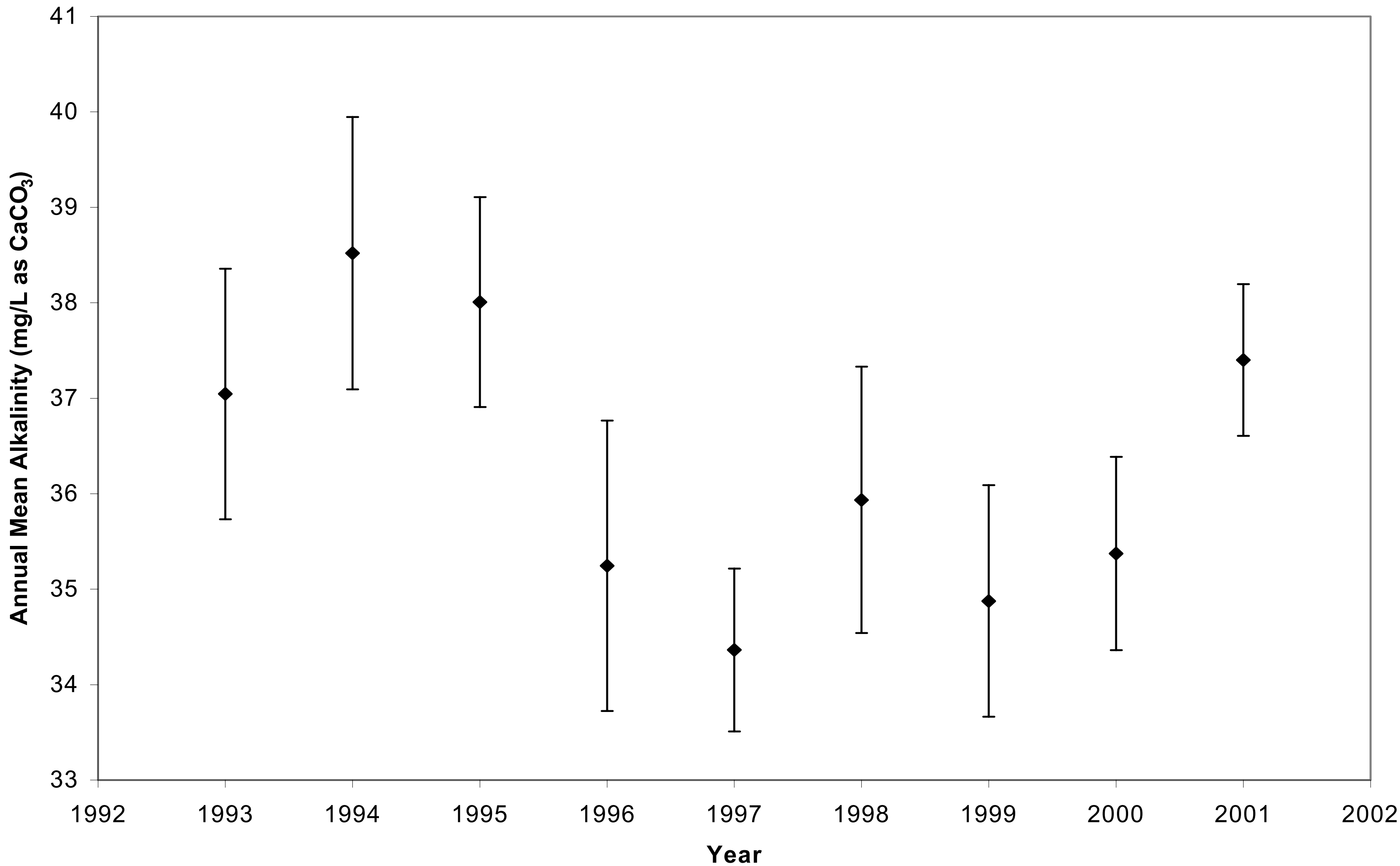


**Figure 14. Annual Mean Water Column Alkalinity ($CaCO_3$ mg/L) for Station 0852 in Lake Washington for 1993 to 2001**

Note: Means +/- SD are arithmetic.

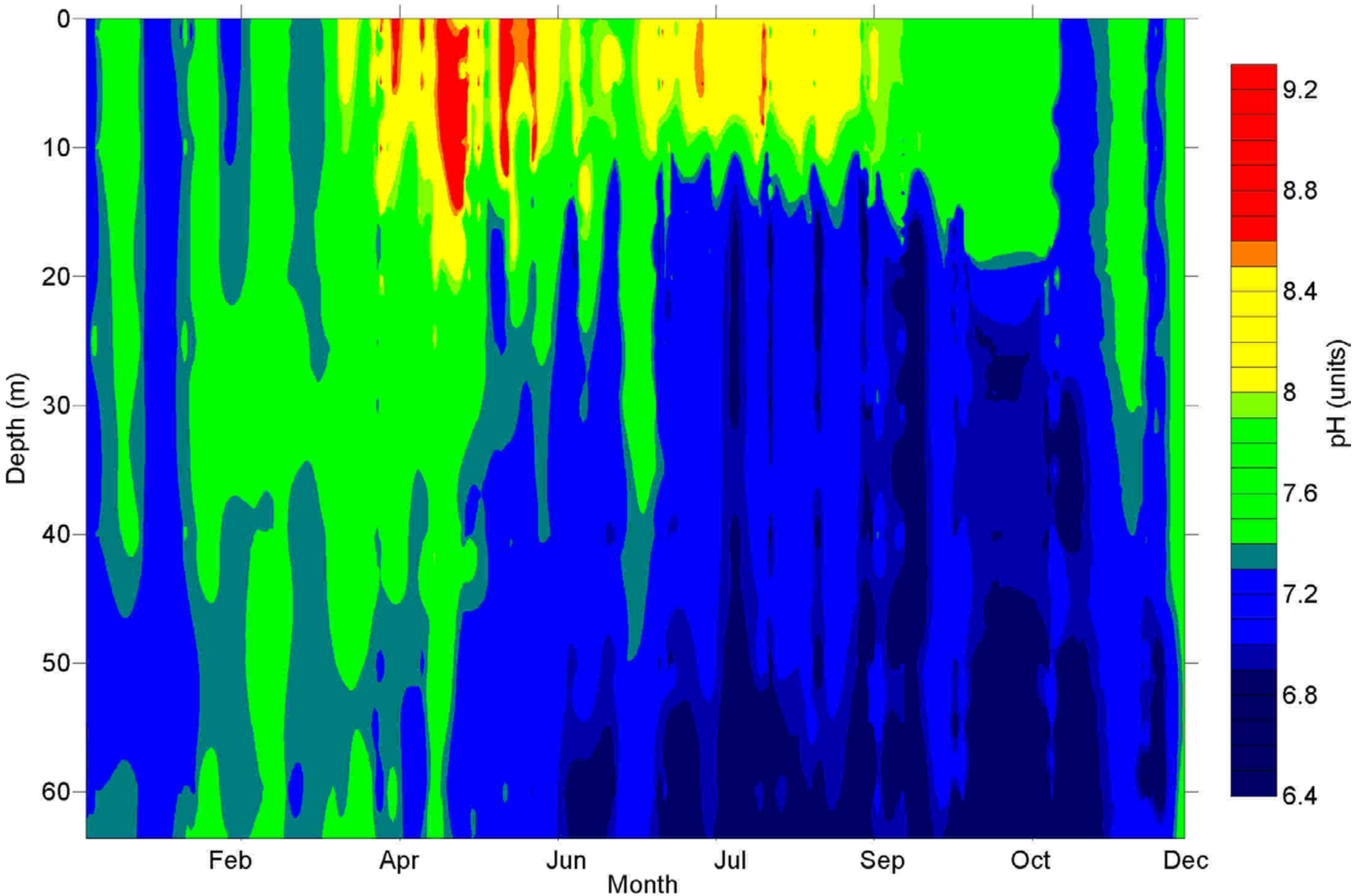


**Figure 15. pH Profile for the Deep Lake Washington Station (0852) for Data Collected From 1994 to 2001**

It is interesting to note that pH was not elevated directly above, at, or below the thermocline. An elevated pH around the thermocline would indicate that there was a layer of algae photosynthesizing at a greater rate than the algae in other depths of the euphotic (photosynthetic) zone or that algae were more abundant at this layer. The fact that this is not occurring in the lake confirms two things: (1) the lake is mixing vertically within the epilimnion, and (2) there is not an increase in photosynthesis at or near the thermocline, as seen in many large oligotrophic lakes (Hutchinson, 1957).

The pH in the nearshore waters was similar to that near the surface (0 to 9 m depth) in the pelagic region, indicating no difference in the horizontal distribution of pH (ANOVA; $p < 0.05$, $n = 10$, annual means).

### 4.2.5. Phosphorus

The annual, volume-weighted, whole-lake mean TP concentration is often used as a long-term indicator of a lake's enrichment status. Whole-lake mean TP represents the TP in the lake that is often predicted with mass-balance models and integrates the effects of unequal distribution of inflow due to effects of wind, stratification, surface inflow, and contributions from bottom sediment.

The annual whole-lake (log-transformed) mean TP concentration ranged from 10 to 18 µg/L from 1992 to 2001. Data were insufficient during 1990 and 1991 to calculate volume-weighted whole-lake means. Annual mean concentrations for 1998 through 2001 (10 to 12 µg/L) were substantially lower than the means observed in the previous 6 years (14 to 18 µg/L) (Figure 16). The difference between whole-lake mean TP from 1998 through 2001 and earlier years (1993 through 1997) was statistically significant (ANOVA; $p < 0.05$, $n = 12$, monthly means). However, closer examination showed that the difference was only between 2000 and 2001 means and the 1994 mean. Data for 1992 were omitted from statistical analysis because only one pelagic station was sampled.

Trend analysis of the annual TP means (arithmetic) using Kendall rank correlation indicate that there is a statistically significant trend toward decreasing TP concentration from 1993 to 2001 for whole-lake and pelagic area ($p < 0.05$), but not for nearshore area. No trend was identified when testing annual mean TP for 1993 through 1997 or 1998 through 2001. This reinforces the observation from the data presented in Figure 16 that there are two separate groups of means, 1990 through 1997 and 1998 through 2001.

Comparison with historical data (see Figure 3) shows that the range in annual whole-lake TP concentration has remained rather stable, not withstanding the recent downward trend. Although there has been a slight decline in whole-lake TP concentration over the last 10 years, TP is near the quickly established equilibrium for 1976 through 1979, following wastewater diversion. The January whole-lake concentration (index used by Edmondson and Lehman, 1981) averaged 15 µg/L from 1992 to 2001, well within the range for the annual means and only slightly less than the 4-year mean of 17 µg/L (1976 through 1979) reported by Edmondson and Lehman. The winter (January through March) whole-lake means for 1992 through 2001 averaged 16 µg/L, similar to that in January.

Nearshore volume-weighted mean TP concentrations were significantly greater than the pelagic means, in spite of high variability (ANOVA; $p < 0.05$, $n = 9$, annual means). While the annual volume-weighted mean TP in the nearshore areas was greater than in the whole-lake and pelagic areas, the year-to-year pattern was similar to that observed for the whole-lake means (Figure 16). The lower whole-lake concentrations indicate that the open water (pelagic) portion of lake dominated the effect of the higher nearshore concentrations. That would be expected given the relative volume of the two areas.

The pattern of lower whole-lake and nearshore means observed from 1998 to 2001, compared to previous years, was consistent for each season (Figures 17 and 18). That is, volume-weighted mean concentrations were usually lower in 1998 through 2001 in each season whether they were from the lake as a whole or nearshore only. Whole-lake winter means were generally less variable than for other seasons (Figure 17). The lower variability of winter concentrations makes them a good indicator of year-to-year trends. The completely mixed condition of the lake and higher water exchange in winter tends to evenly distribute constituents, accounting for much of the lower variability. In contrast to the whole-lake mean TP concentrations, nearshore means usually showed greater variability and were higher during winter than in other seasons (Figure 18). The minimal effect of the higher, more variable nearshore concentrations on the lower, more stable winter whole-lake concentrations was also due to the high flushing and complete mixing.

An assessment of causes for lower TP concentrations from 1998 to 2001 is beyond the scope of this report. However, there are two factors responsible for such annual variation in other large lakes that might apply here: (1) reduced external loading, and (2) longer water residence time. Reduced external loading would simply mean less TP income to replace the losses through sedimentation and outflow, while increased residence time would increase the loss to bottom sediments. These explanations would require that inflows were less during those 4 years than previously. While flows have not been examined directly, precipitation was not consistently low during those years. Another explanation could be reduced external loading due to improved stormwater treatment. Even if there were a trend of improved treatment, a step-change to account for the 4-year, lower concentrations is highly unlikely.

Examination of the annual volume-weighted mean TP concentrations in the hypolimnion and epilimnion (Figure 19) shows that the lower whole-lake TP in recent years is due in part to the marked decline in hypolimnetic TP. There is a significant trend toward reduced hypolimnetic TP concentrations over the last 10 years from an annual mean of 33 µg/L in 1992 to 13 µg/L in 2001. Trend analysis using Kendall rank correlation showed a statistically significant ($p < 0.05$, $n = 9$, annual arithmetic means) decrease in hypolimnetic TP concentration from 1993 through 2001. No trend was indicated for epilimnetic TP over the same period. This trend toward reduced TP concentration may in part account for the similar trend observed for whole-lake TP means discussed earlier. For this analysis, the hypolimnion was defined as all water from 25 m to 60 m and the epilimnion layer between the surface and a depth of 20 m for the whole year. This trend is similarly apparent for volume-weighted TP concentrations in the hypolimnion during the spring-fall stratified periods only.

The TP concentrations in Lake Washington are in large part a reflection of the P loading from the Cedar River, which has good water quality and relatively low P concentration. From 1995 to 2000, the Cedar River contribution averaged 50% of the total annual flow into Lake Washington, and that source contained an annual volume-weighted mean TP concentration of 17.2 µg/L, while the other 50% of inflow from the Sammamish River, other tributaries, and nearshore non-point sources had a combined mean of 56 µg/L (Table 1, Arhonditis et al. unpublished manuscript). Hence, the Cedar River is essentially diluting other TP sources to the lake. The expected lake TP concentration resulting from the Cedar River load (25% of the total) is only about 7 µg/L [TP inflow (1 – R)], using the TP retention coefficient (R) from Edmondson and Lehman (1981). Thus, if Lake Washington received only Cedar River water, its concentration would be about half the 1993 through 2001 whole-lake average (15 µg/L). There is little internal loading of P during summer from bottom sediments in Lake Washington, so there is little hypolimnetic entrainment of bioavailable P in the surface waters during the growing season. Respective inflow TP loads and expected lake concentrations from the remaining inputs averaged 71 and 28 µg/L. Without the high-quality Cedar River inflow, the quality of Lake Washington would be many times poorer, given that 63% of the lake's immediate watershed is urbanized.

The importance of the low Cedar River TP concentration relates directly to the size of the spring algal bloom. That is, soluble P, which is strongly influenced by the Cedar River input as indicated, remains relatively high during winter when mixing and low incident light combine to prevent algal utilization and growth. When incident light increases and warms the surface water, temporary thermal stability of the water column occurs. The temporary stability reduces the mixing depth of the algae, allowing the algae to produce more biomass than is respired, utilize the available P, and develop a bloom. While this explains the timing and magnitude of spring algal blooms in deep lakes generally (Welch, 1992), diurnal variations in solar heating and wind mixing may cause year-to-year variations in bloom magnitude. Nevertheless, the large historical magnitude of algal blooms in Lake Washington, beginning in spring and continuing into summer, was caused by the change in winter soluble P concentration determined by external loading (Edmondson, 1969; Edmondson and Litt, 1982).

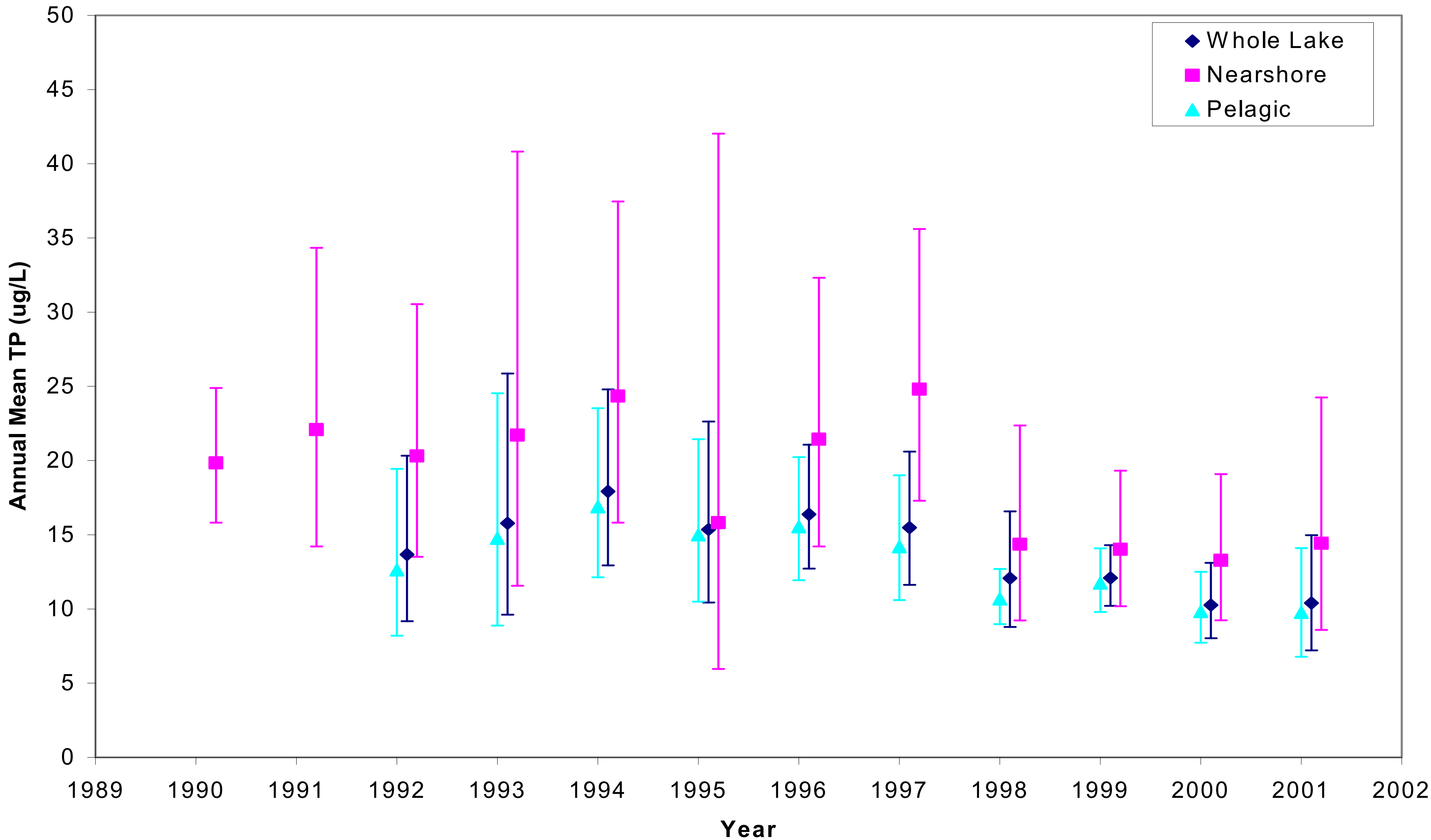


**Figure 16. Annual Volume-Weighted Whole-Lake (1992 to 2001), Pelagic (1992 to 2001), and Nearshore (1990 to 2001) Mean Total Phosphorus Concentrations in Lake Washington**

Note: Means +/- SD are based on log-transformed data.

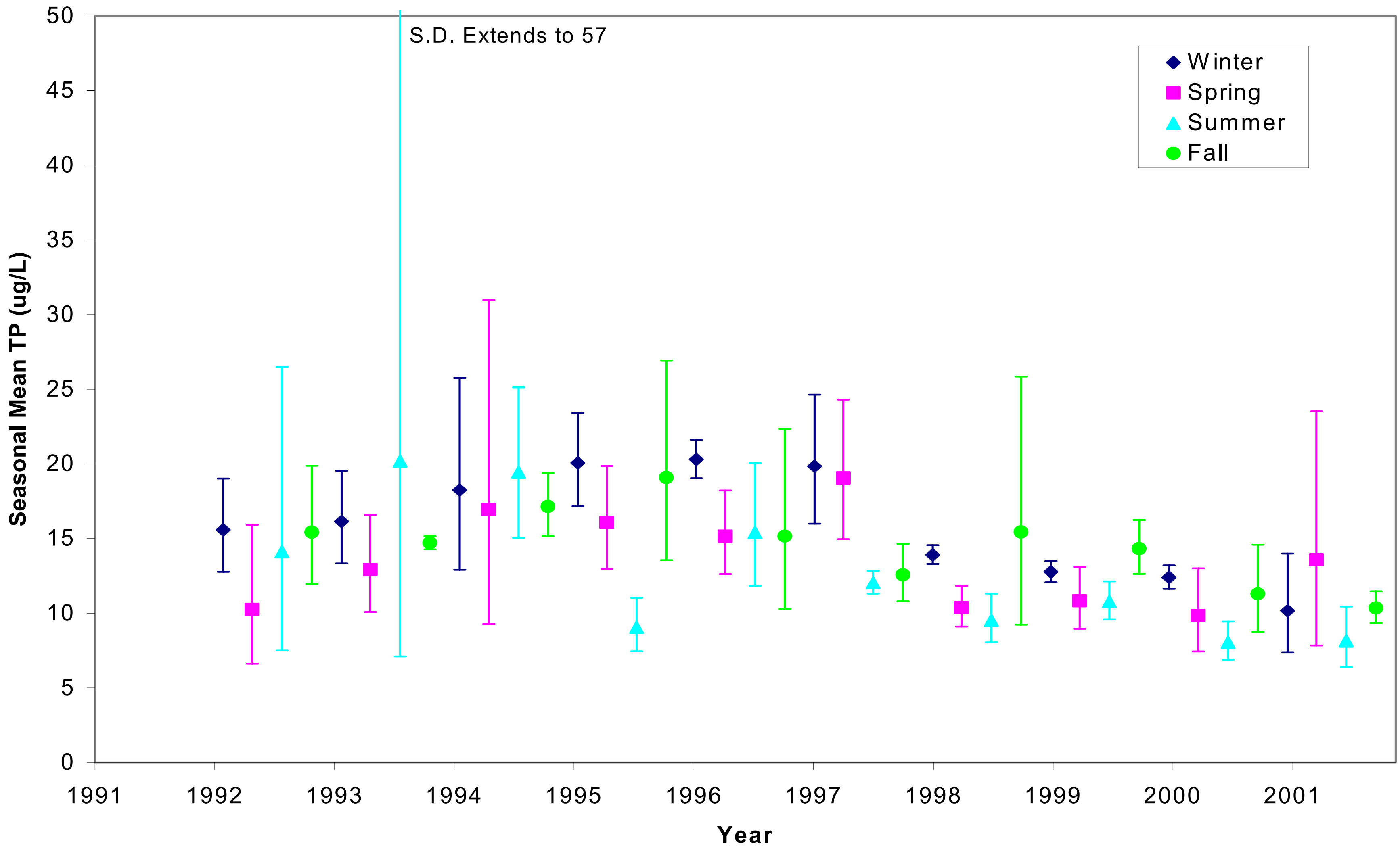


**Figure 17. Seasonal Volume-Weighted Whole-Lake Mean Total Phosphorus Concentrations in Lake Washington From 1992 to 2001**

.Note: Means +/- SD are based on log-transformed data.

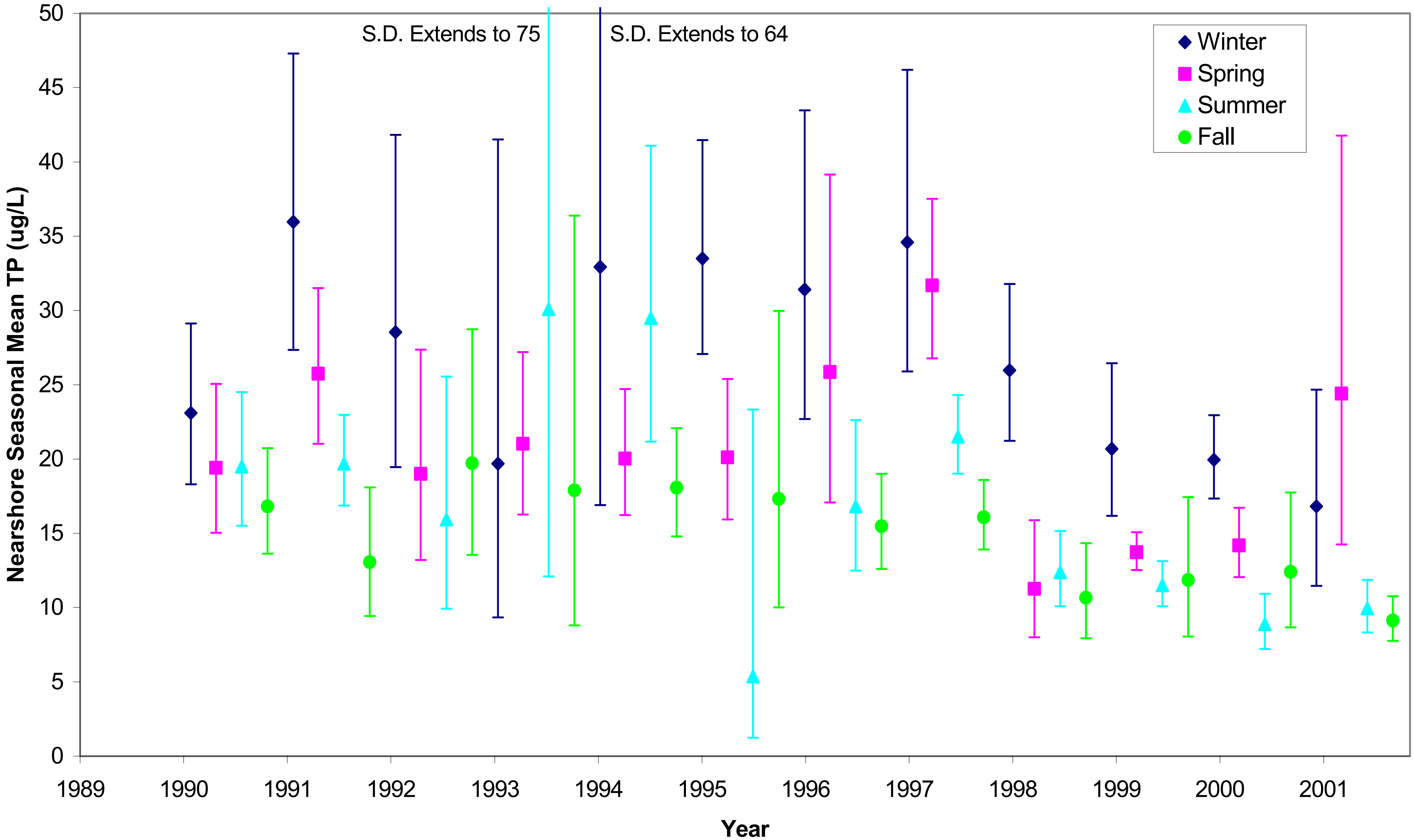


**Figure 18. Seasonal Volume-Weighted Nearshore Mean Total Phosphorus Concentrations in Lake Washington From 1990 to 2001**

.Note: Means +/- SD are based on log-transformed data.

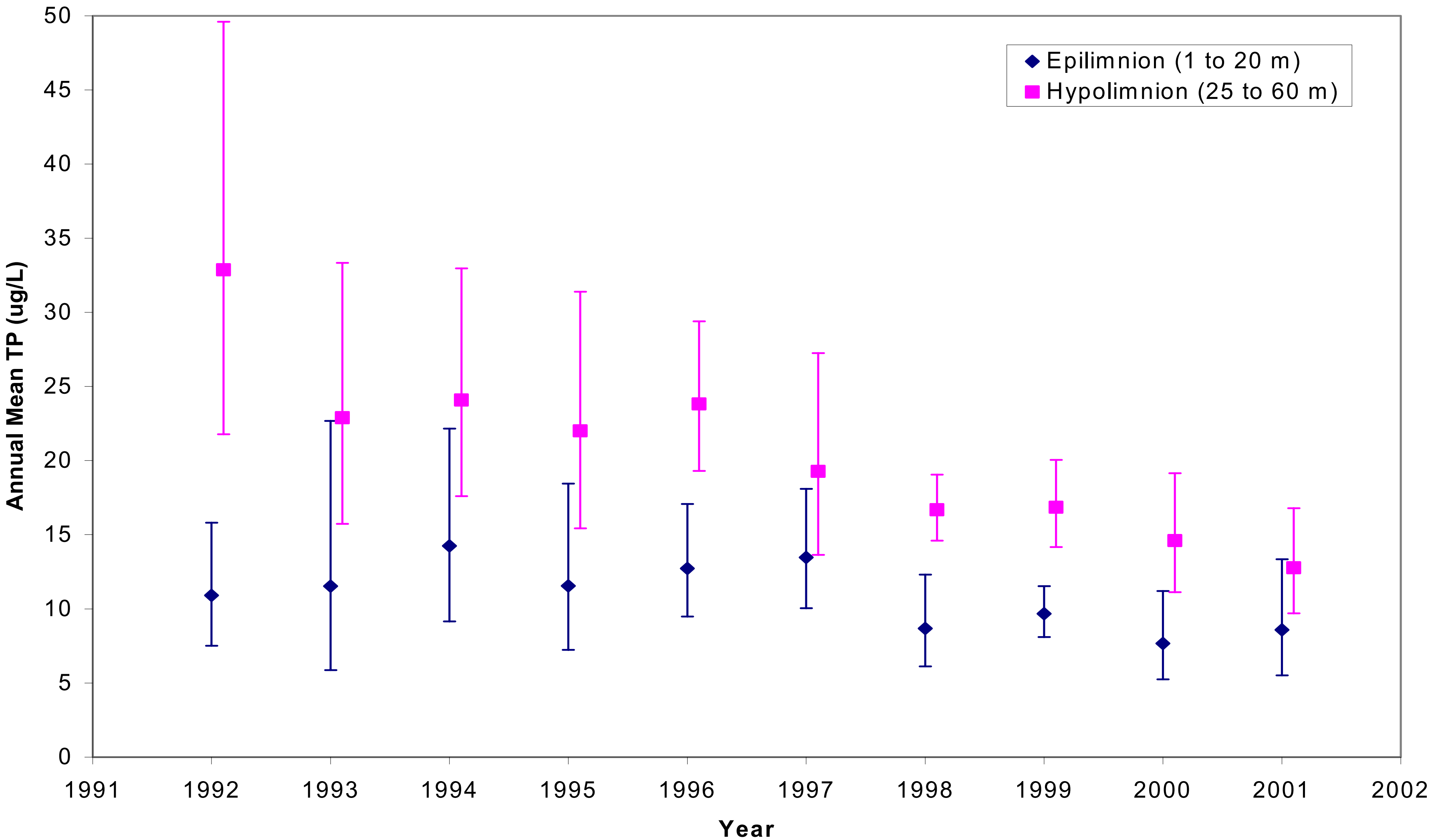


**Figure 19. Volume-Weighted Annual Mean Total Phosphorus Concentrations in Lake Washington From 1992 to 2001 in the Epilimnetic and Hypolimnetic Layers**

.Note: Means +/- SD are based on log-transformed data.

### 4.2.6. Soluble Reactive Phosphorus

Mean annual volume-weighted soluble reactive phosphorus (SRP) concentrations ranged from 2 to 11µg/L in the nearshore areas and 3 to 8 µg/L in both the pelagic area and the whole lake (Figure 20). SRP concentrations in the nearshore were lower from 1998 to 2001 than the previous 8 years, especially in 2001 (Figure 20). Pelagic and whole-lake mean SRP concentrations were lower only for the last 2 years. ANOVA analysis of monthly mean concentrations from 1993 through 2001 showed that SRP concentrations in the nearshore, pelagic, and whole-lake areas from 2001 were significantly different from those from 1993 through 1997 ($p < 0.05$, $n = 12$, monthly means). Data from 1990 through 1992 were omitted due to limited sampling.

As would be expected in P-limited systems (see Section 4.2.7), the mean SRP concentrations were usually higher during winter than spring and summer due to greater loading and reduced algal uptake (Figure 21). Spring SRP concentrations were usually lowest due to the maximum utilization by algae, as indicated above. The generally low summer concentrations are a reflection of low external loading, due to low summer inflow and settling of particulate P with sinking algae. Fall concentrations were typically higher than those in spring and summer, reflecting the increase in external loading, entrainment from higher hypolimnion concentrations, and also limited plankton algal uptake due to increasing growth restrictions as light and temperature decreased. In general, the SRP concentrations in the lake were largely less than 10 µg/L, the level below which biomass increase is strongly dependent on P (Sas, 1989).

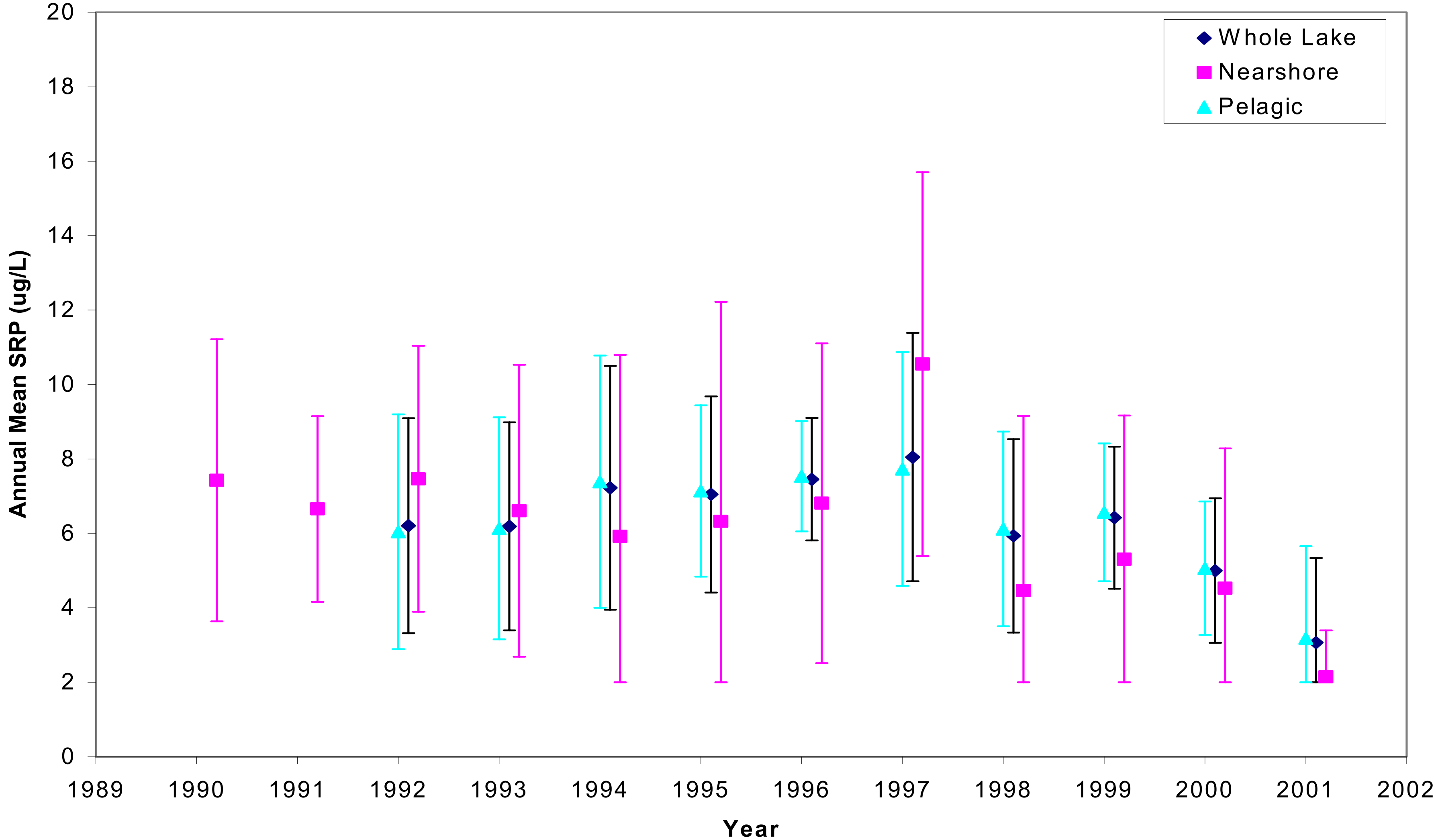


**Figure 20. Annual Volume-Weighted Nearshore (1990 to 2001), Pelagic (1992 to 2001), and Whole-Lake (1992 to 2001) Mean SRP Concentrations**

Note: Means +/- SD are arithmetic.

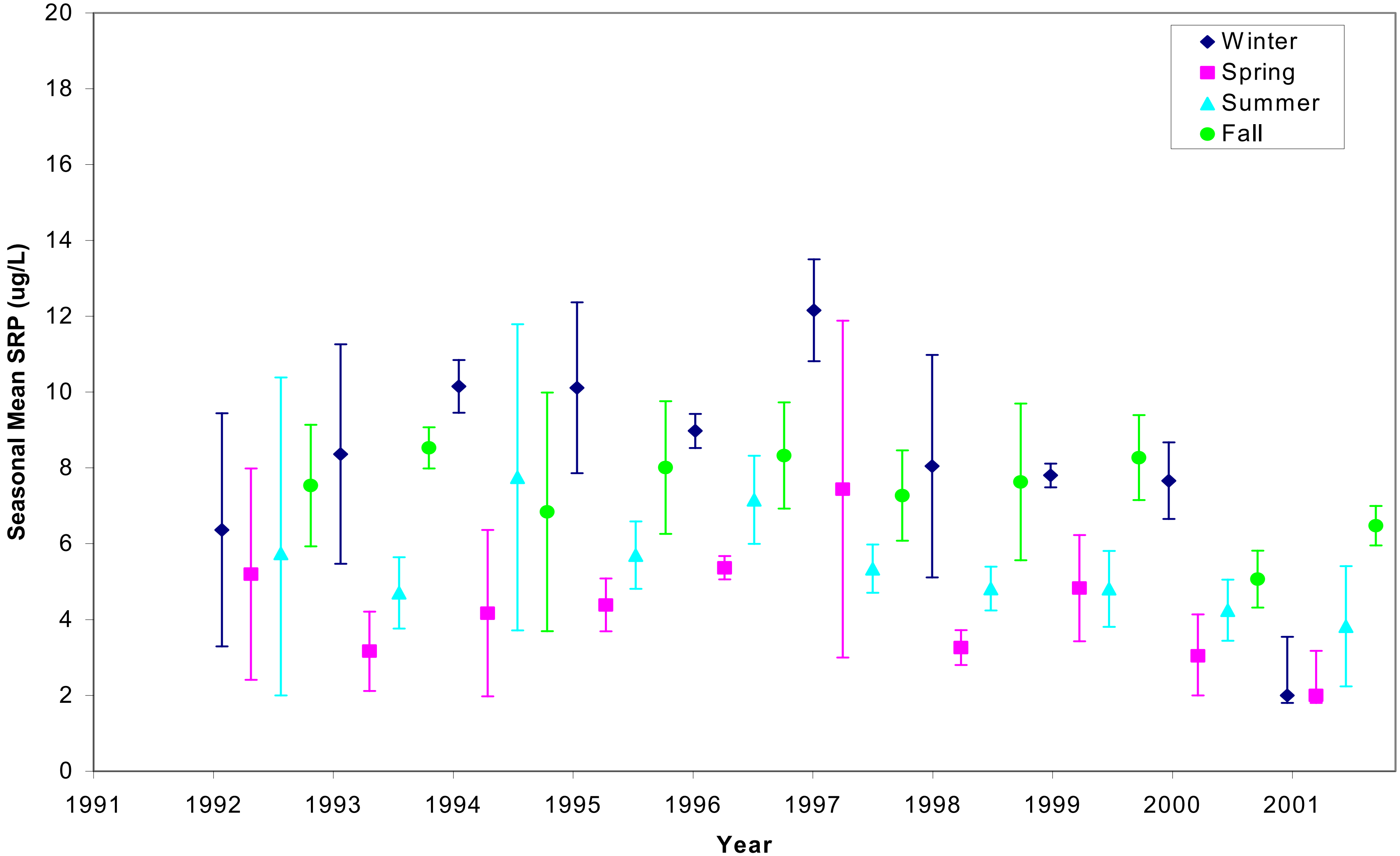


**Figure 21. Seasonal Volume-Weighted Whole-Lake Mean SRP Concentrations in Lake Washington From 1992 to 2001**

Note: Means +/- SD are arithmetic.

## 4.2.7. Nitrogen

The epilimnetic and hypolimnetic annual mean volume-weighted TN concentrations in Lake Washington followed a similar year-to-year pattern from 1993 through 2001(Figure 22). The epilimnion contained significantly less nitrogen than did the hypolimnion (ANOVA; $p < 0.05$, $n = 9$, annual averages), which is to be expected due to the settling out of particulate matter during stratified periods and algal uptake.. The hypolimnetic annual mean volume-weighted TN concentrations ranged from 320 to 440 µg/L, and the epilimnetic annual mean volume-weighted TN concentrations ranged from 100 to 275 µg/L. Although it would appear from Figure 22 that there was a trend toward increasing nitrogen concentration over time in both the epilimnion and hypolimnion, the relatively low annual mean TN concentration in 2001 counters any statistical significance. Analysis of monitoring data from future years will determine if there is a trend.

Annual mean TN at the nearshore areas was consistently higher than in pelagic areas, ranging from 275 to 390 µg/L and 160 to 330 µg/L, respectively. Whole-lake means were nearly the same as pelagic concentrations, as would be expected since the nearshore volume represents a small portion of the lake.

From 1993 to 1999, whole-lake spring TN concentrations were, in general, higher than for other seasons (Figure 24). In contrast, whole-lake nitrate-nitrogen concentrations were usually highest in the winter (Figure 25). This seasonal influence on nitrate-nitrogen was also noticeable in the nearshore areas (Figure 26). The high winter nitrate-nitrogen means were probably due to the influence of reduced uptake by wintering terrestial plants and increased storm water runoff . Although nitrate concentrations decreased in the summer, concentrations remained an order of magnitude higher than SRP.

The annual mean nitrate-nitrogen whole-lake concentration ranged from 122 to 212 µg/L, nearshore concentrations ranged from 99 to 196 µg/L, and pelagic concentrations ranged from 116 to 215 µg/L (Figure 27). Nitrate-nitrogen makes up a relatively large fraction of the lake TN, as can be seen by comparing Figure 23 with Figure 27. No long-term trend was found for nitrate-nitrogen, nor was there a difference between the nearshore and pelagic areas (ANOVA test, $p < 0.05$, $n = 9$, annual means). Note that the dataset for nitrate-nitrogen is more complete than the TN dataset; therefore, Figures 25 through 27 start in 1990 and 1992 instead of 1993.

As would be expected due to the oxic water column, the ammonium-nitrogen in the lake was relatively low and only made up a fraction of the TN in the lake. The whole-lake volume-weighted ammonium-nitrogen average was highly variable over this period, and no long-term significant trend was observed. The 1998 through 2001 annual means were lower than 1994 through 1997 annual means ($p < 0.05$, $n = 12$, monthly means) (Figure 28). Because of the variability in the data, a longer period of record is required to determine if there has, in fact, been a decreasing trend.

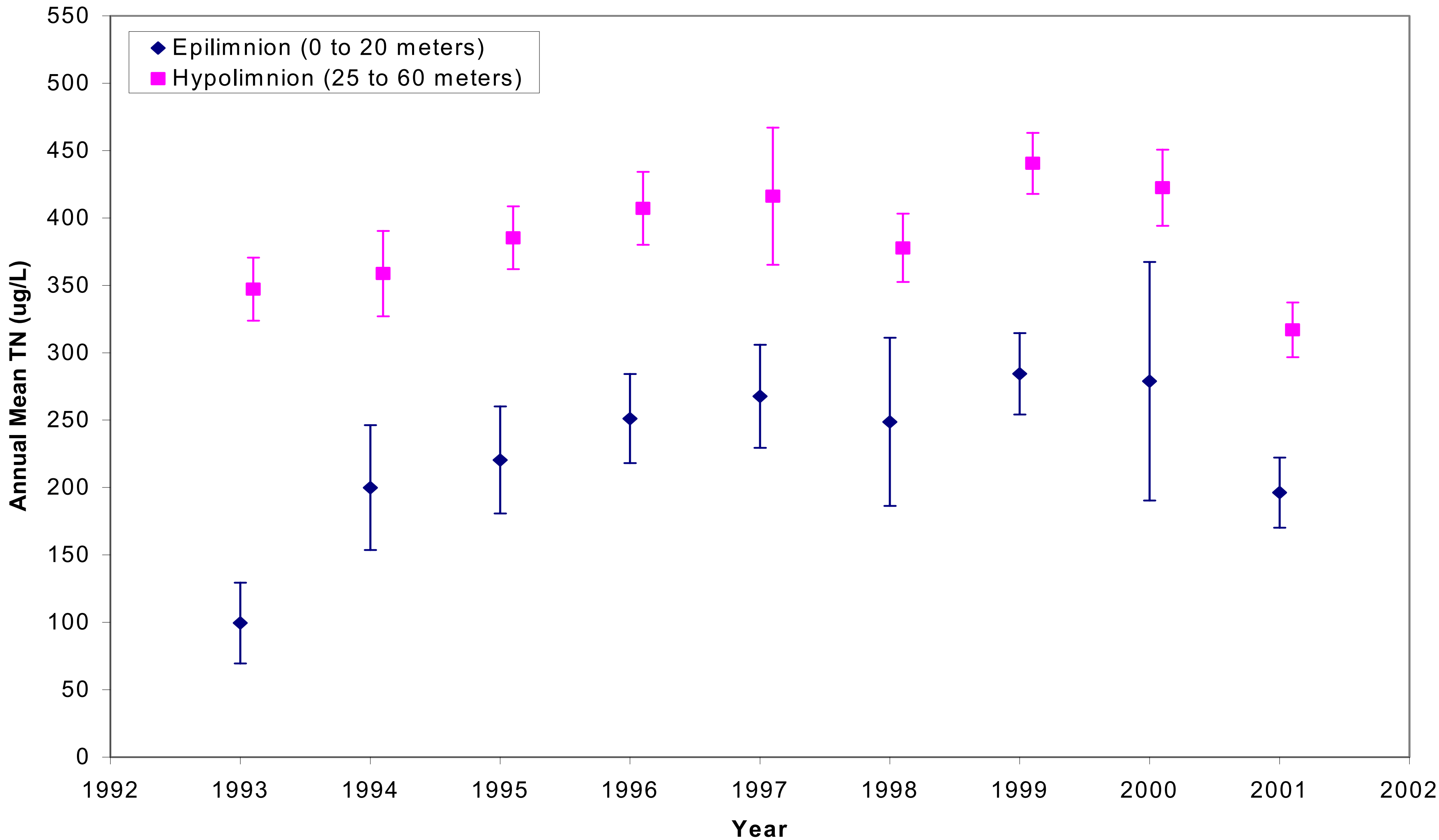


**Figure 22. Annual Volume-Weighted Mean Total Nitrogen (TN) Concentration for the Epilimnion and Hypolimnion of Lake Washington**

Note: Means +/- SD are arithmetic.

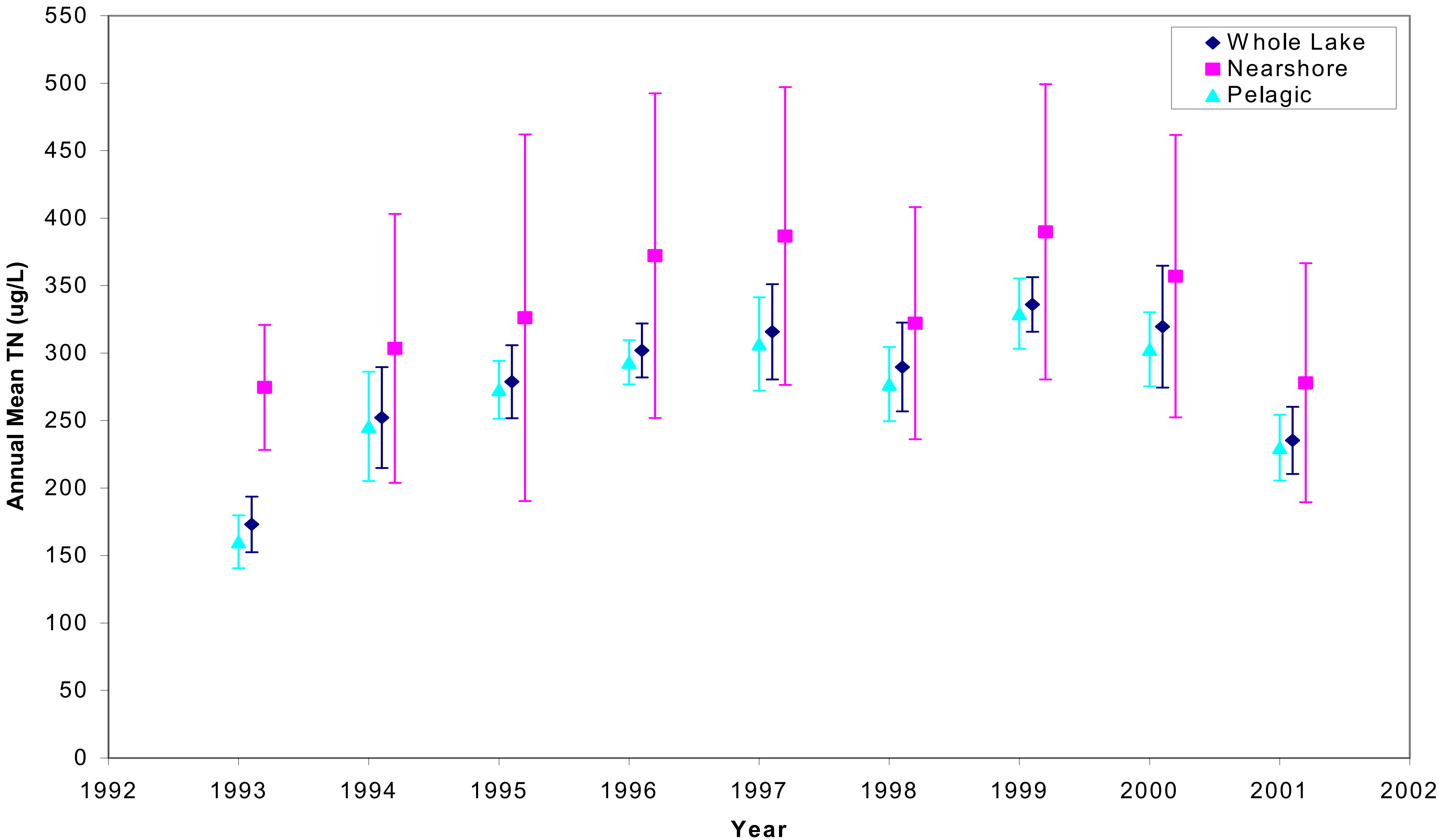


**Figure 23. Annual Volume-Weighted Mean Whole-Lake, Nearshore, and Pelagic Total Nitrogen (TN) Concentration for Lake Washington, 1993 to 2001**

Note: Means +/- SD are arithmetic.

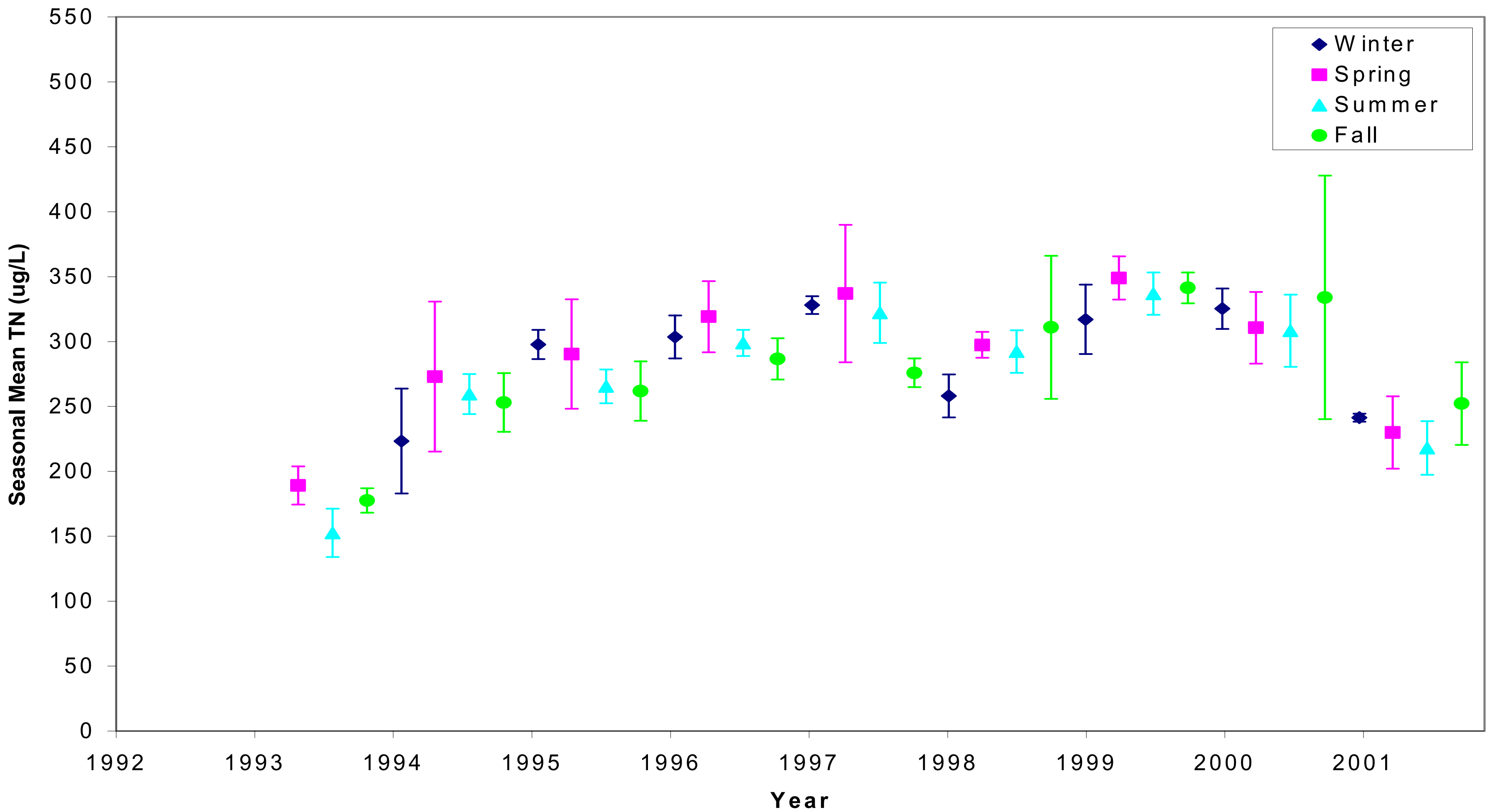


**Figure 24. Seasonal Volume-Weighted Mean Whole-Lake Total Nitrogen (TN) Concentrations for Lake Washington, 1993 to 2001**

Note: Means +/- SD are arithmetic.

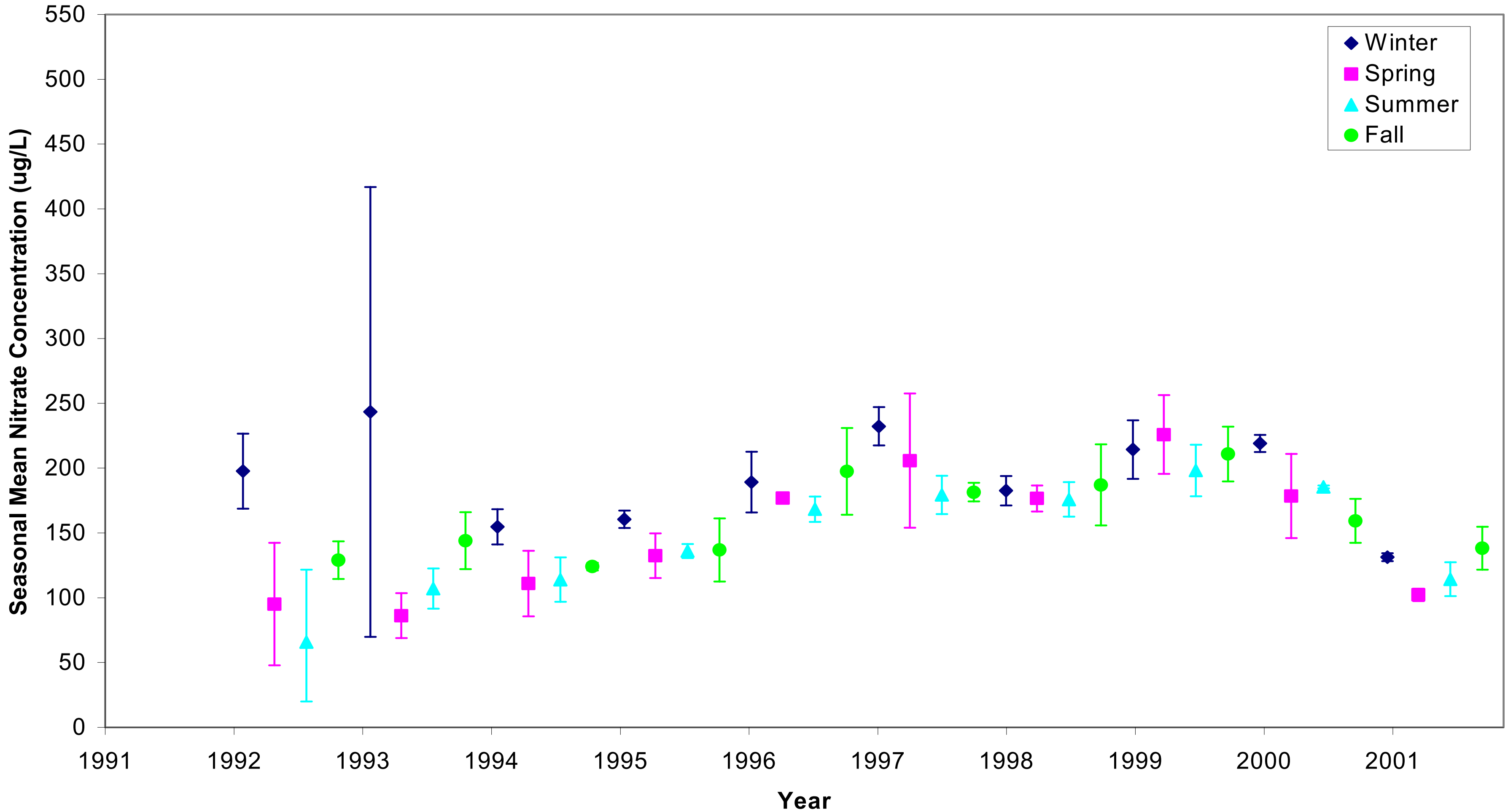


**Figure 25. Seasonal Volume-Weighted Mean Whole-Lake Nitrate-Nitrogen for Lake Washington, 1992 to 2001**

Note: Means +/- SD are arithmetic.

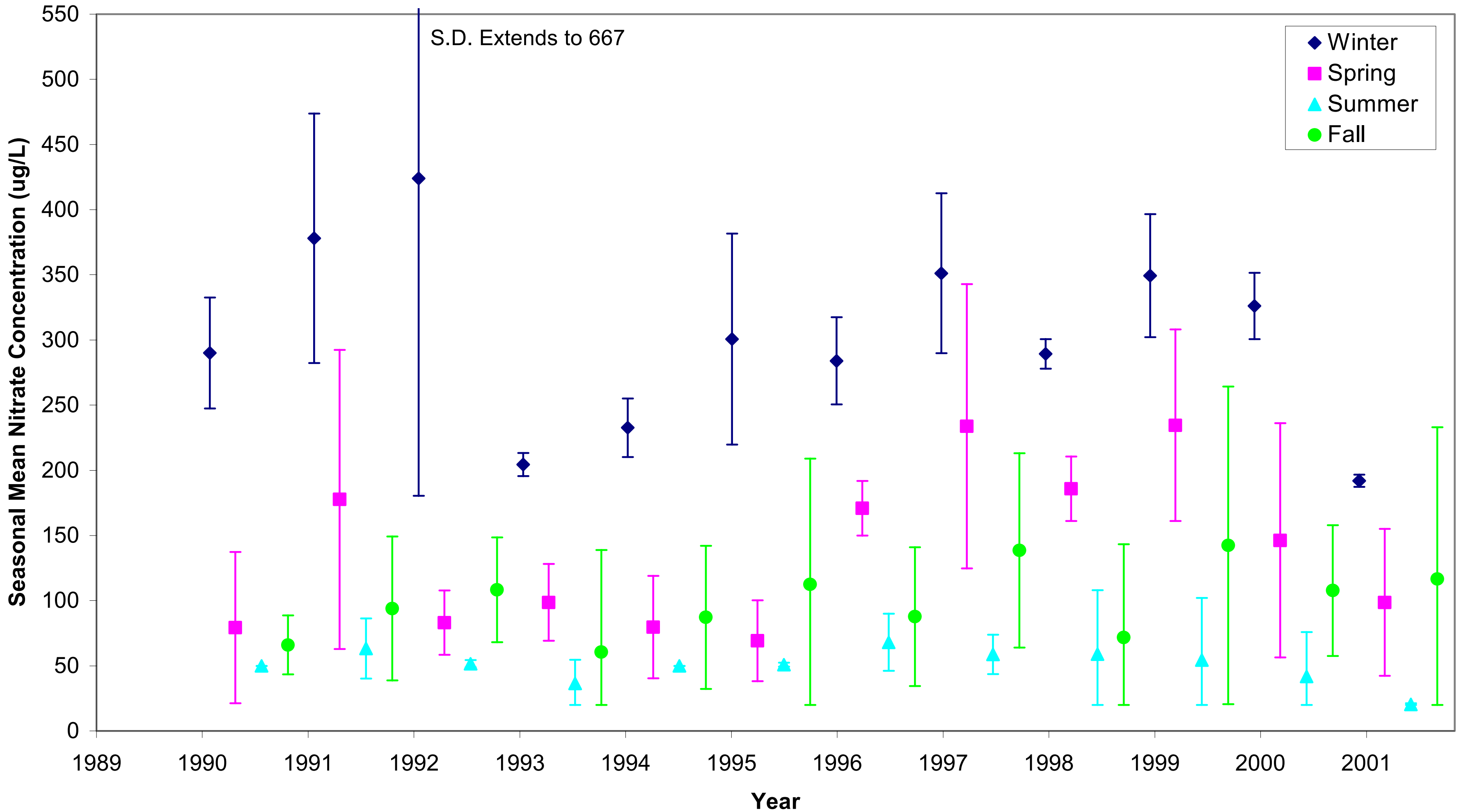


**Figure 26. Seasonal Volume-Weighted Mean Nearshore Nitrate-Nitrogen Concentrations for Lake Washington, 1990 to 2001**

Note: Means +/- SD are arithmetic.

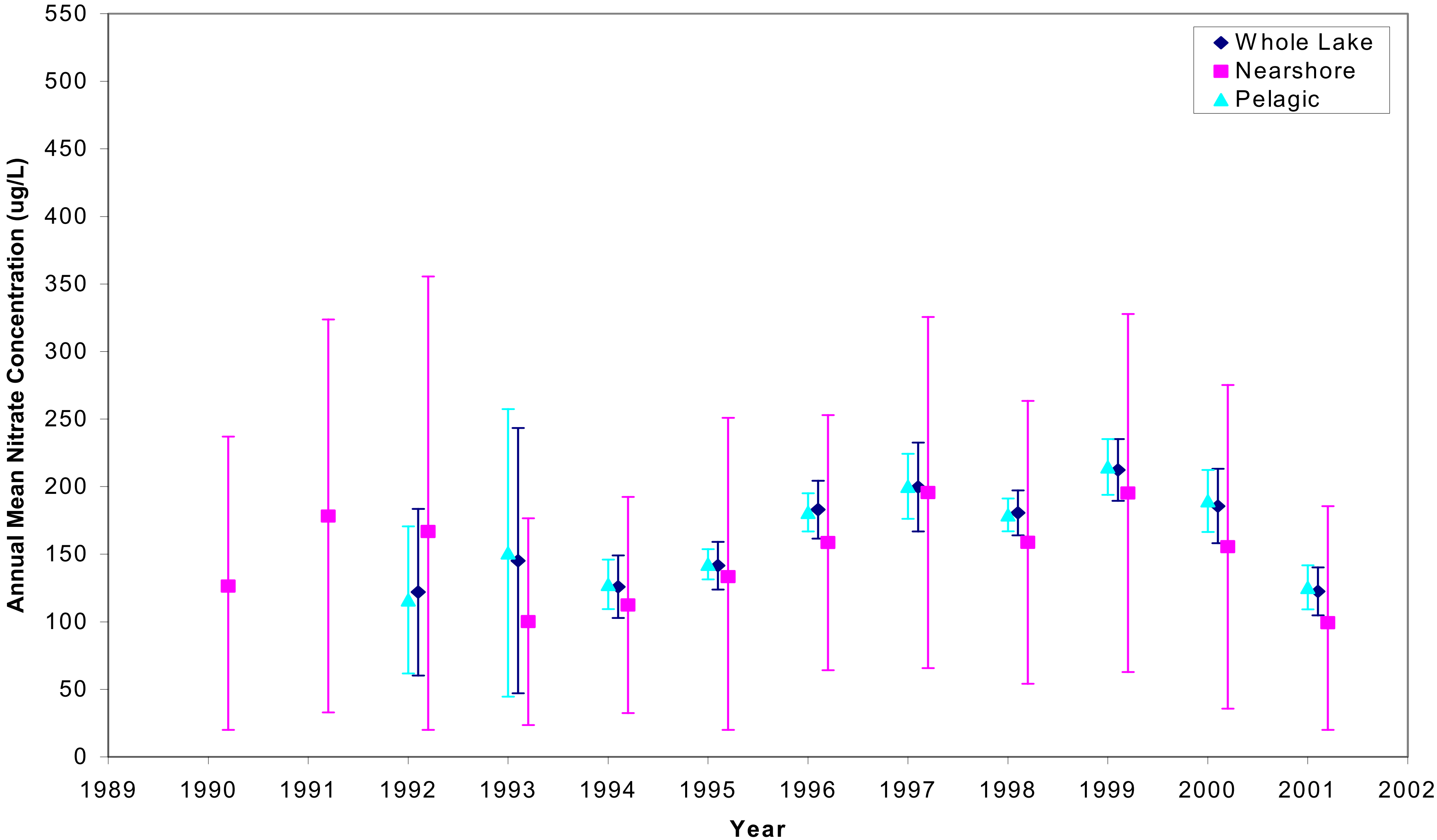


**Figure 27. Annual Volume-Weighted Mean Whole-Lake, Nearshore, and Pelagic Nitrate-Nitrogen Concentrations for Lake Washington, 1990 to 2001**

Note: Means +/- SD are arithmetic.

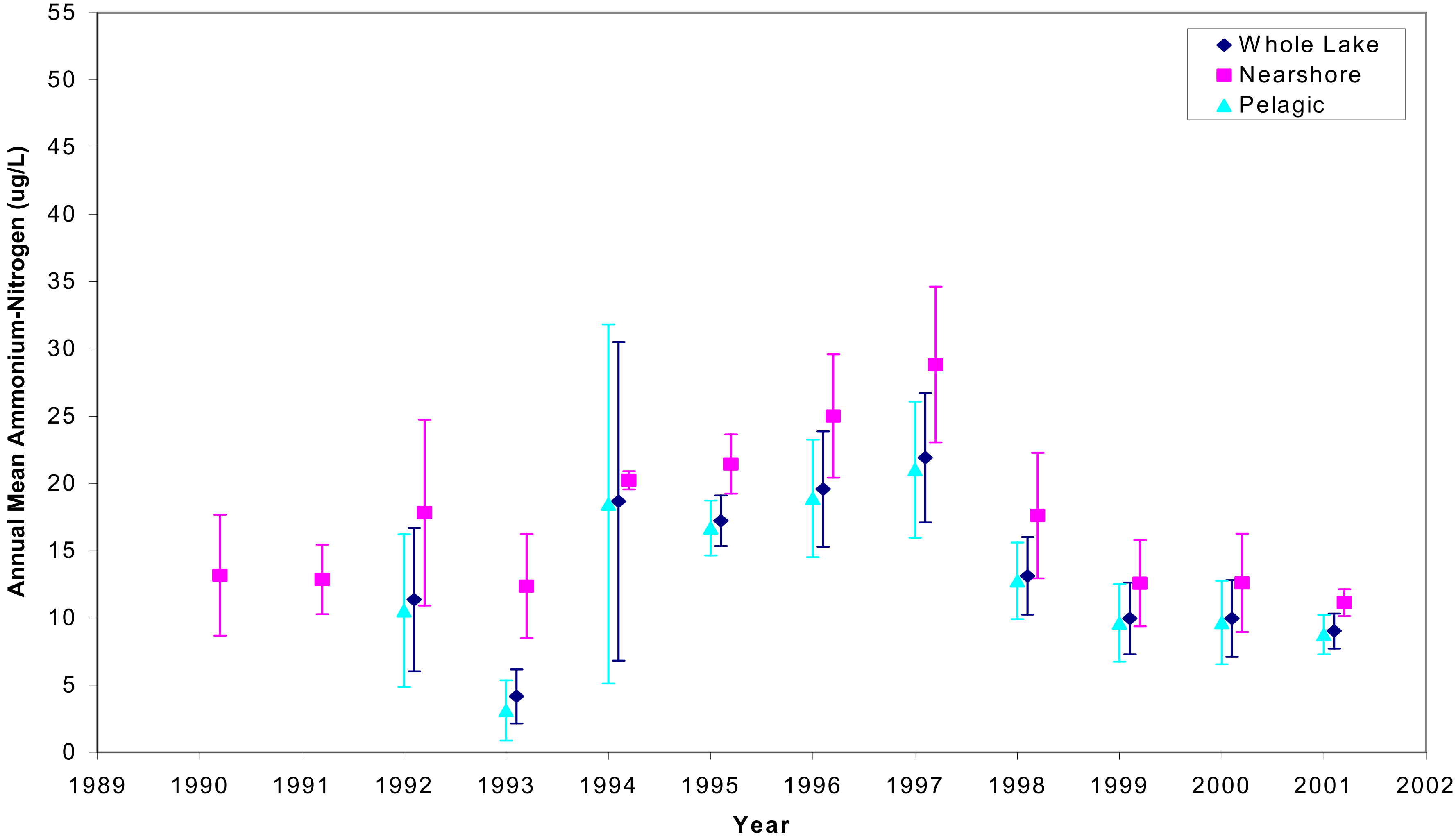


**Figure 28. Annual Volume-Weighted Mean Whole-Lake, Nearshore, and Pelagic Ammonium Concentrations for Lake Washington, 1990 to 2001**

Note: Means +/- SD are arithmetic. Scale is smaller than for Figures 18 through 23.

### 4.2.8. Nutrient Limitation

Algal growth, or more properly termed productivity, is limited by nutrients or environmental conditions (temperature, solar energy). In lakes, the primary productivity is usually limited or controlled by nutrient availability. Primary productivity in lowland Puget Sound area lakes, as for most lakes in the world, is typically limited by the macronutrient, phosphorus. Nitrogen can also limit algal growth at times. To estimate which nutrient is limiting the productivity, the TN:TP ratio can be used. Generally, if the molecular TN:TP ratio is greater than 16:1, then the algal productivity is considered limited by P availability (Carroll and Pelletier, 1991). Nutrient ratios are usually expressed on a weight (mass) basis, e.g., μg TN:μg TP. The Redfield TN:TP ratio of 16:1, calculated using the number of atoms, is approximately equivalent to 7:1 by weight.

The annual mean TN:TP ratios were similar for the epilimnion and the whole lake (Figure 29). There was a trend toward increasing TN:TP ratio in the lake from 1994 through 2001 that was statistically significant ($p < 0.05$, $n = 8$). (TN:TP ratios were not calculated prior to 1994 due to incomplete data records.) The TN:TP ratios exceeded the Redfield ratio of 7:1 (Figure 29), which is often considered the threshold for P limitation. This would indicate that the lake is becoming increasingly limited by P. The dramatic decrease in the TN:TP ratio that occurred in 2001 was due in part to a decrease in epilimnetic and whole-lake TN concentration observed in 2001. At the same time, the TP concentration in both the epilimnion and the whole lake increased in 2001 from 2000 concentrations. Nevertheless, the TN:TP ratio for 2001 was still well above the P-limit threshold.

The overall trend of increasing TN:TP ratio from 1994 through 2001 is directly related to the trends in annual TN and TP concentrations (see Figures 23 and 16). Although TN increased from 1994 through 2000, except for a reduction in 2001, the change in TP in the opposite direction was even greater, especially from 1998 to 2000 (see Sections 4.2.5 and 4.2.6 for TP and TN trend analysis). This then is the cause for the increase in TN:TP ratio over the same period. As discussed in previous sections, the TP concentration in the lake is a function of external loading, sedimentation, and flushing. Future investigations may help determine the causes for these trends when loading from land uses relative to storm water runoff and flows from major water supplies, i.e. Cedar River, are better defined.

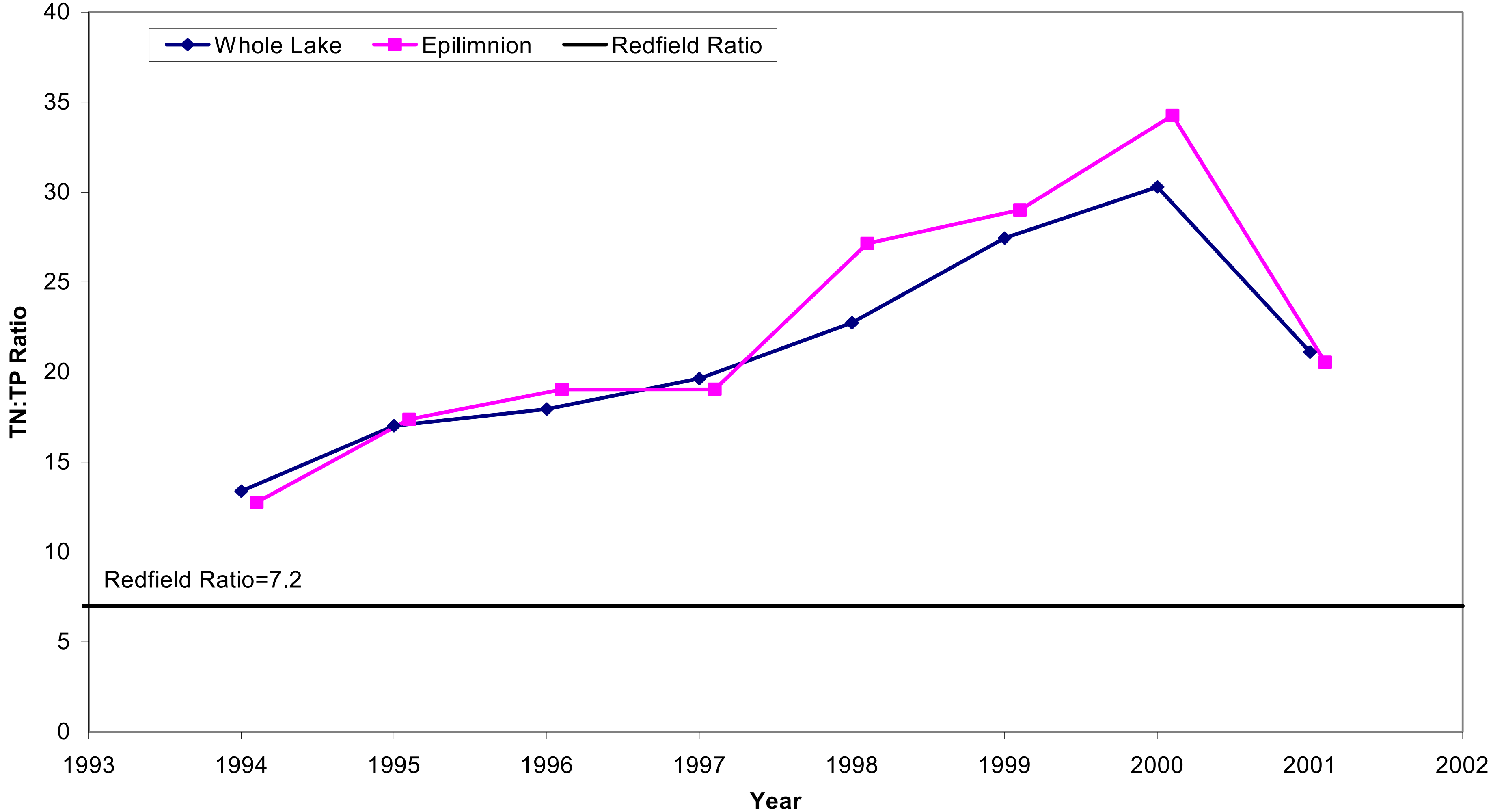


**Figure 29. Annual Whole-Lake and Epilimnetic Ratios of Total Nitrogen (TN) to Total Phosphorus (TP) for Lake Washington From 1994 to 2001, Compared to the Redfield Ratio**

Note: Ratios are based on molecular weights.

# 4.3. Biological Conditions

## 4.3.1. Chlorophyll *a* (chl *a*)

The photosynthetic pigment chl *a* represents a reliable estimate of algal biomass (total wet or dry weight) and is used universally as an index of biological response to nutrient enrichment. Mean growing season concentrations are commonly used to indicate trophic state and the acceptability of water quality for recreation and water supply use. While chl *a* is a good indicator of total algal biomass, biovolume estimates from microscopic examination of individual taxa are necessary to determine their relative contribution to total biomass.

From a historical perspective, chl *a* content declined sharply from 1969 to 1970, proportional to P decrease, following wastewater diversion (Edmondson, 1970). Chl *a* declined yet again after 1976 to a summer mean of 3 μg/L when zooplankton grazing increased (Edmondson and Litt, 1982). Summer mean chl *a* from 1990 through 2001 ranged from 0.5 to 3.6 μg/L with a 12-year mean of 2.4 μg/L (Figure 30).

The highest chl *a* concentrations in Lake Washington occurred during early to mid spring (Figures 30 and 31), when available light in the water column reached an intensity that allowed the growth of wind-mixed algal cells, mostly diatoms, to exceed losses. Temporary density stability in the water column, which results from increased surface warming, is important for maintaining algal cells in the lighted zone. Mixing during cooler nights distributes chl *a* to greater depths, as indicated by high DO concentrations at 20 m (see Section 4.2.1). The spring mean chl *a* concentrations have ranged from 3.2 to 13.5 μg/L from 1990 to 2001, with a 12-year mean of 6.7 μg/L. The spring means were significantly higher than mean concentrations from other seasons (ANOVA; $p < 0.05$, $n = 12$, annual means).

Summer chl *a* concentrations in the lake were usually lower because settling of the spring bloom removed much of the P from surface water, transferring it to the bottom via sedimentation. Thus, summer algal biomass in the photic zone is constrained by P availability. With higher external loading, more P would be available in the epilimnion during summer, permitting higher summer chl *a* as was the case during the period of high loading from wastewater (Edmondson, 1969).

Annual mean chl *a* concentrations in the pelagic areas were consistently less than those in nearshore areas (Figure 32), but the means were not significantly different (ANOVA; $p < 0.05$, $n = 9$, annual means). Neither was pelagic chl *a* during the more productive spring period significantly different from nearshore chl *a* (Figure 33) (ANOVA; $p < 0.05$, $n = 9$, annual means). Thus algal biomass was apparently rather evenly distributed across the lake. Moreover, the significantly higher nearshore TP concentrations apparently did not result in more algae. The rate of exchange between nearshore and pelagic water was probably too great to allow a growth response to the higher nearshore TP. In addition, nearshore TP may be elevated due to inputs of particulate P from the tributaries.

Particulate P does not provide nutrients for algae, it probably settles in the vicinity of the tributary input.

Year-to-year trends in annual mean chl *a* concentrations were not evident from 1990 through 2001 for any season separately (see Figure 30). Chl *a* may have been expected to decline from 1998 to 2001 in response to the significantly lower TP concentrations. However, even the higher concentrations during spring were not significantly different over the 12-year period (ANOVA; $p < 0.05$, $n = 3$, spring monthly means). Moreover, the spring chl *a* was only weakly related to TP concentrations, and the correlation was not statistically significant ($p < 0.05$ and $n = 10$, annual means; Figure 34). In general, year-to-year variation in chl *a* was more likely due to differences in water column stability and light availability than the relatively small differences in TP (see discussion in Section 4.2.5).

The high spring chl *a* concentrations tended to dominate annual means (Figure 35). The high annual concentrations in 1994, 1995, 2000, and 2001 corresponded with the high spring concentrations during those years (see Figure 30).

While year-to-year variations in mean chl *a* concentrations did not relate strongly with TP, even in the spring, the long-term control of TP on chl *a* and transparency becomes more convincing when the overall 10-year means are compared with model predictions (Carlson, 1977) and historical data. Figure 36 shows that the overall 10-year means fit closely to the predicted lines in spite of considerable variation among the individual yearly summer concentrations. The figure further indicates that small differences in TP are not apt to explain small year-to-year variations in chl *a*. Not withstanding year-to-year variations due largely to climatic conditions, algal biomass and transparency are strongly dependent on TP concentrations in Lake Washington over a wide range of external loading.

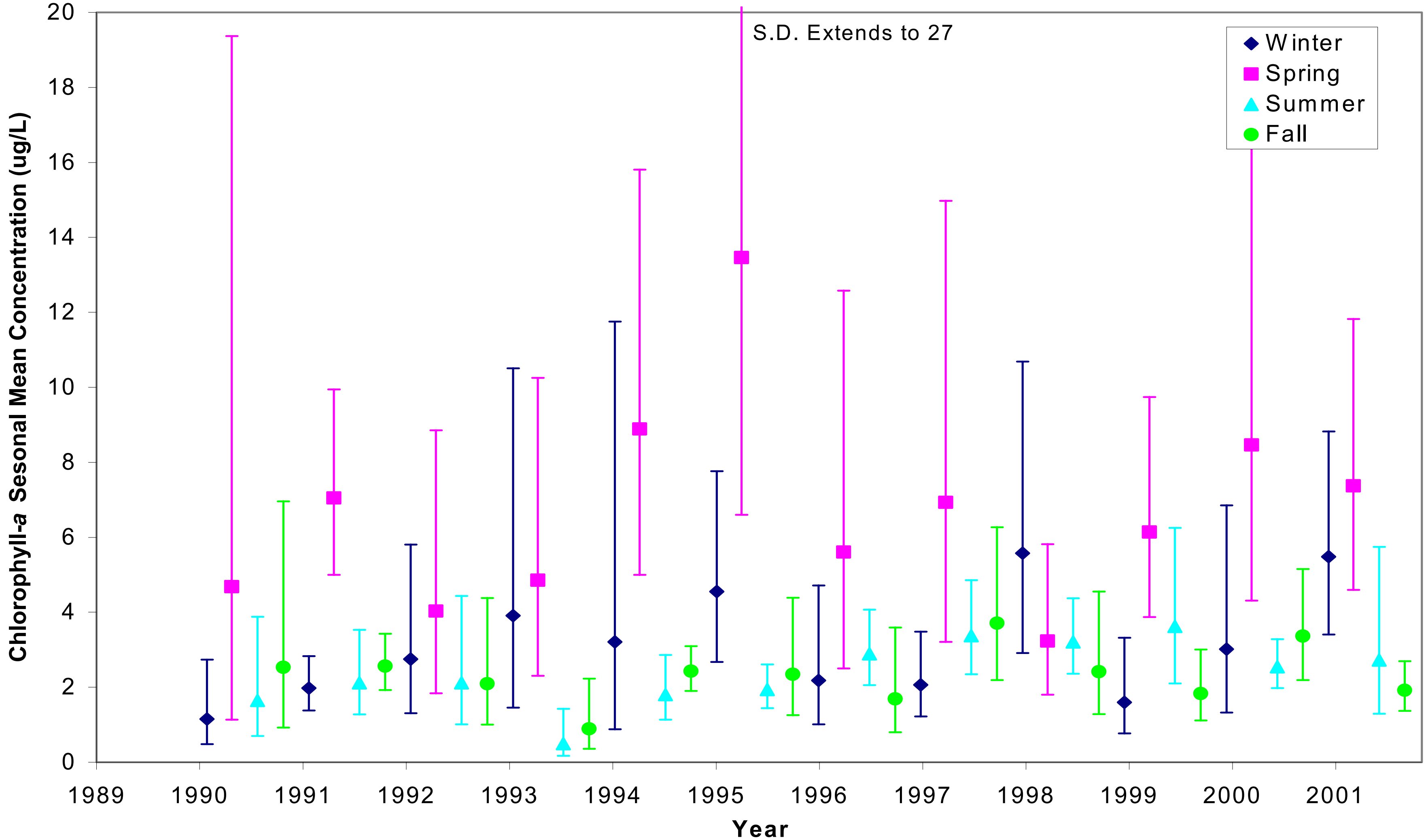


**Figure 30. Seasonal Mean Chlorophyll *a* Concentrations for the Combination of Pelagic and Nearshore Stations Unweighted for Area**

Note: Means +/- SD are based on log-transformed data.

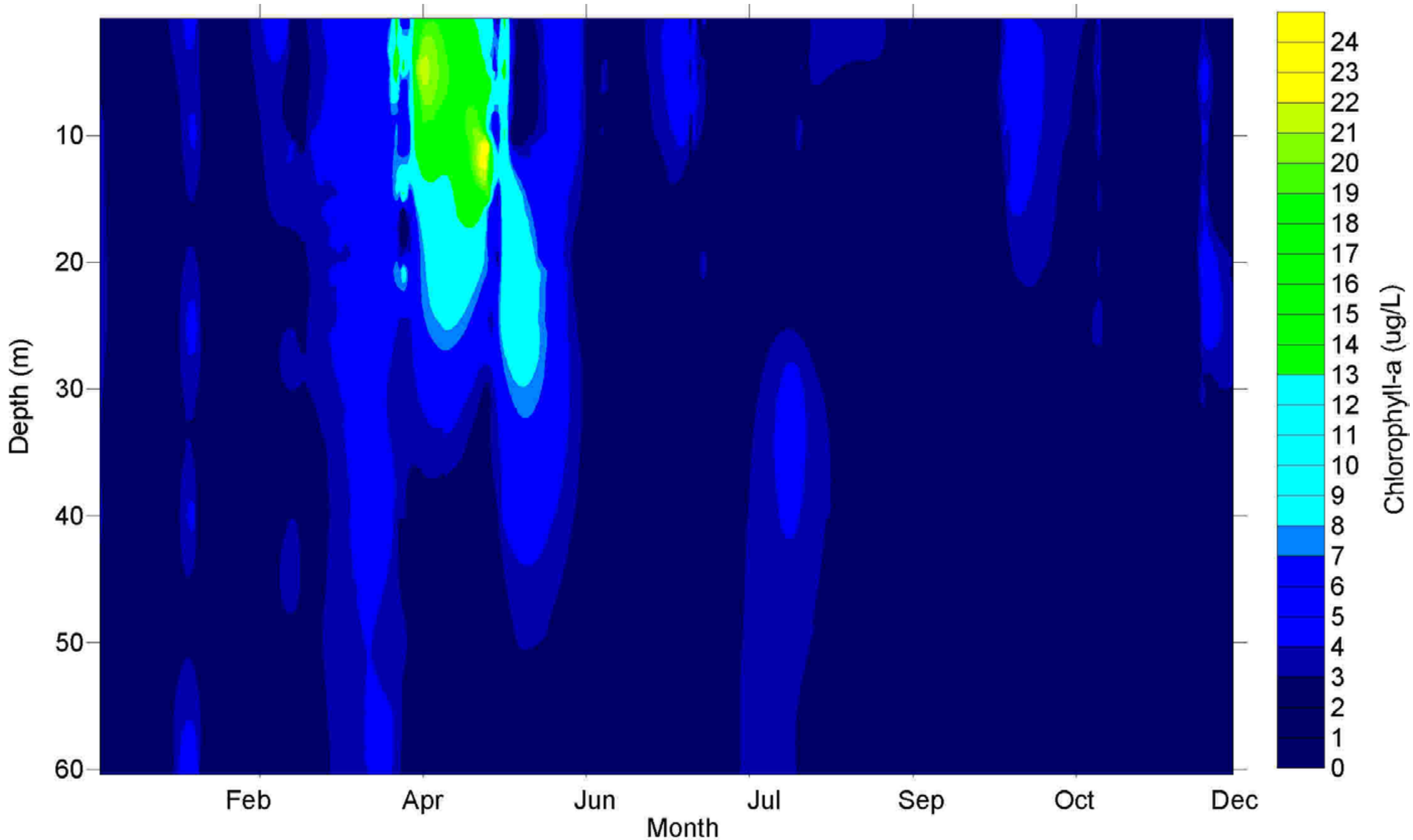


**Figure 31. Isopleths of Chlorophyll *a* Concentrations at the Deep Station (0852) From 1993 to 2001**

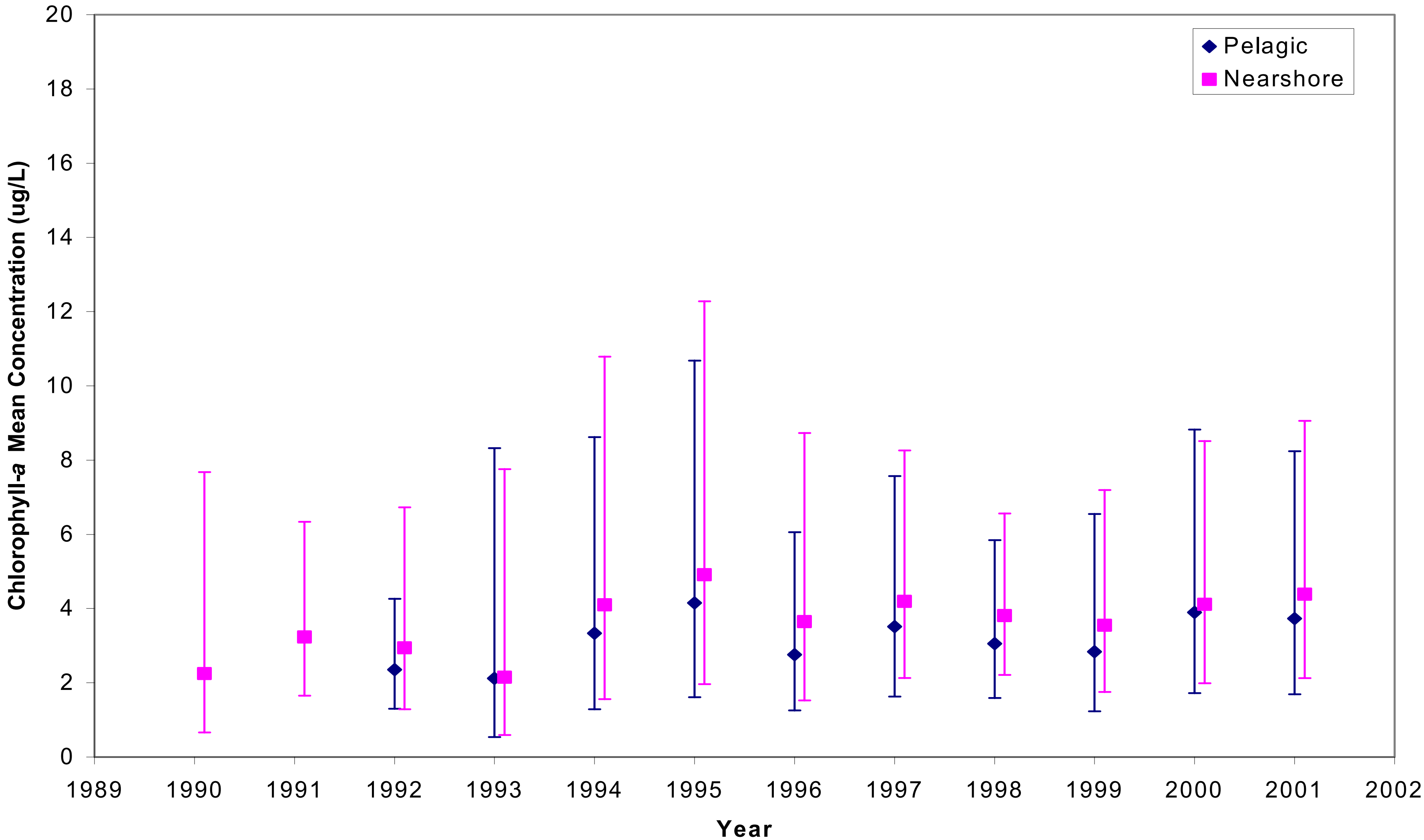


**Figure 32. Annual Mean Chlorophyll *a* Concentrations for Pelagic and Nearshore Stations Unweighted for Area**

Note: Means +/- SD are based on log-transformed data.

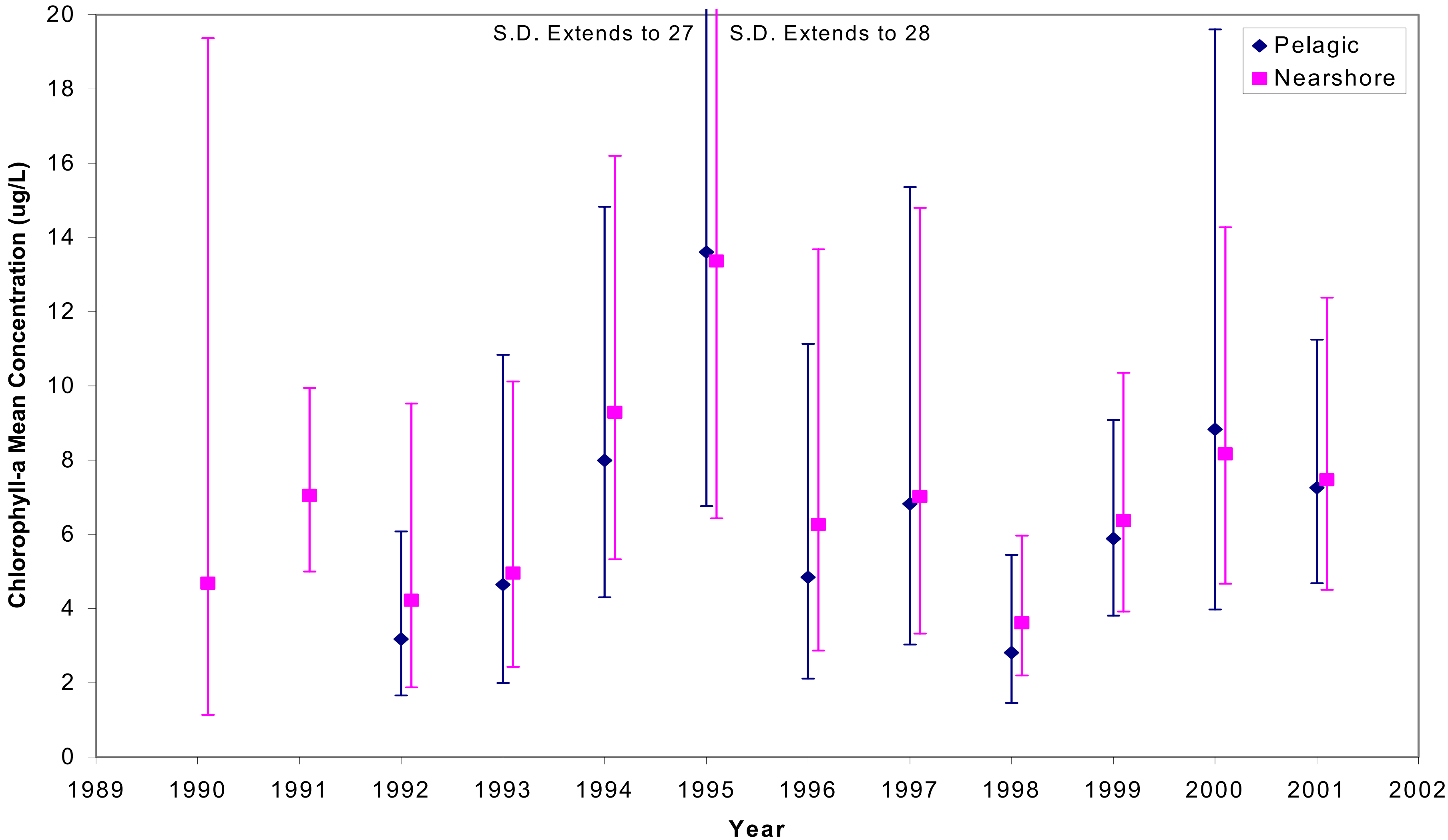


**Figure 33. Spring Mean Chlorophyll *a* Concentrations for Pelagic and Nearshore Stations Unweighted for Area**

Note: Means +/- SD are based on log-transformed data.

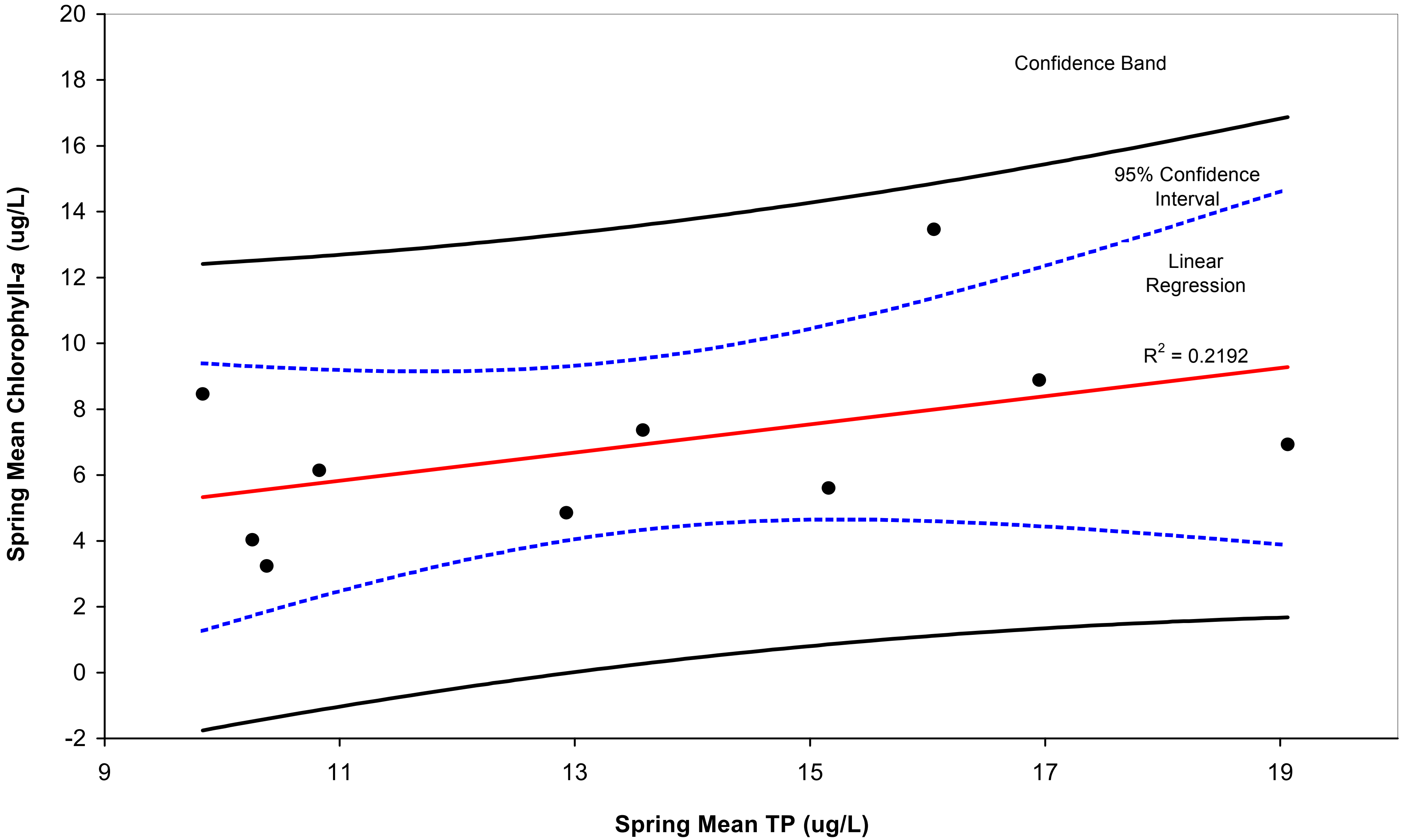


**Figure 34. Relationship Between Spring Whole-Lake TP and Spring Chlorophyll *a* at Combined Stations From 1992 to 2001**

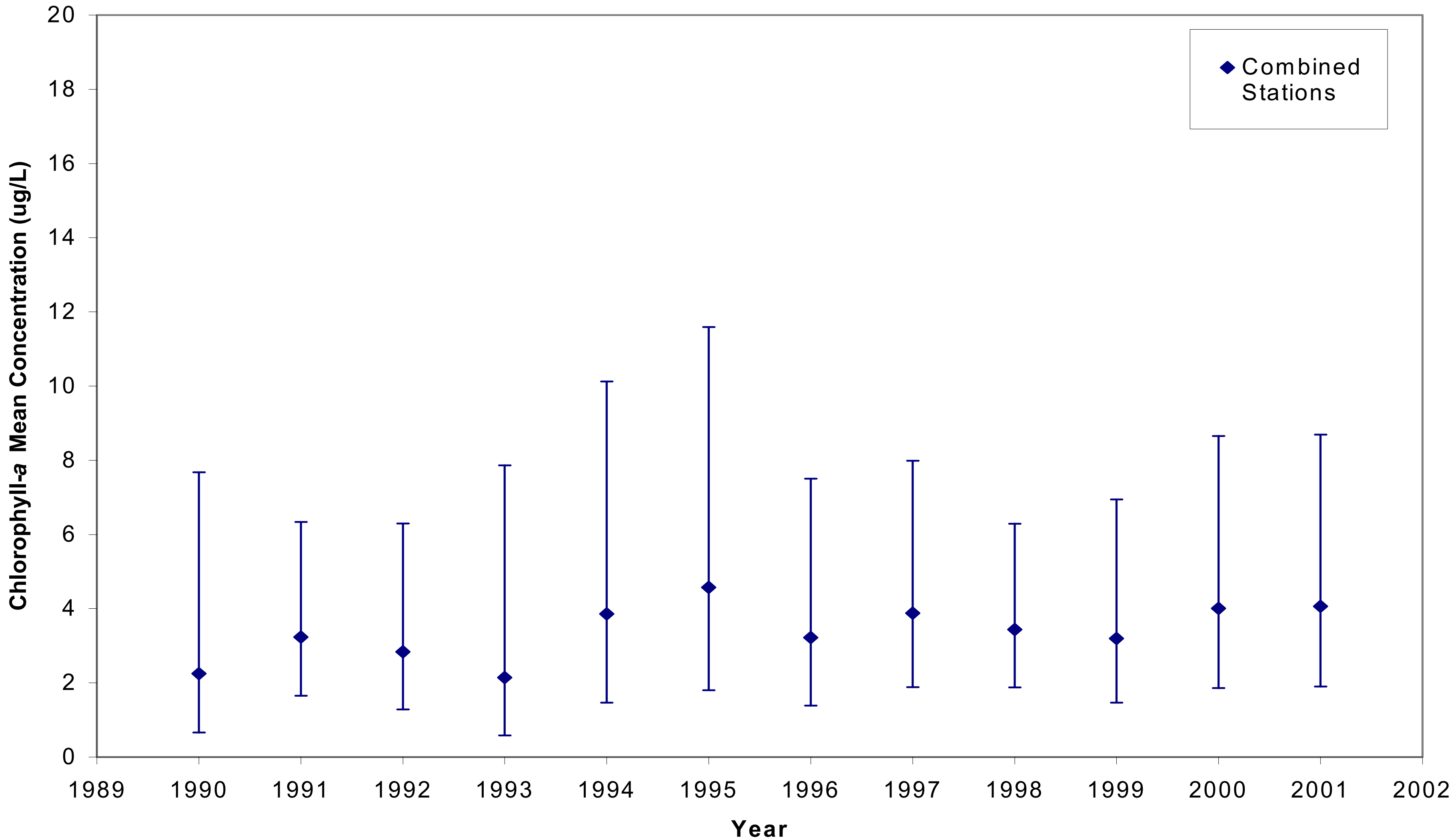


**Figure 35. Annual Mean Chlorophyll *a* for Combined Pelagic and Nearshore Stations From 1990 to 2001**

Note: Means +/- SD are based on log-transformed data.

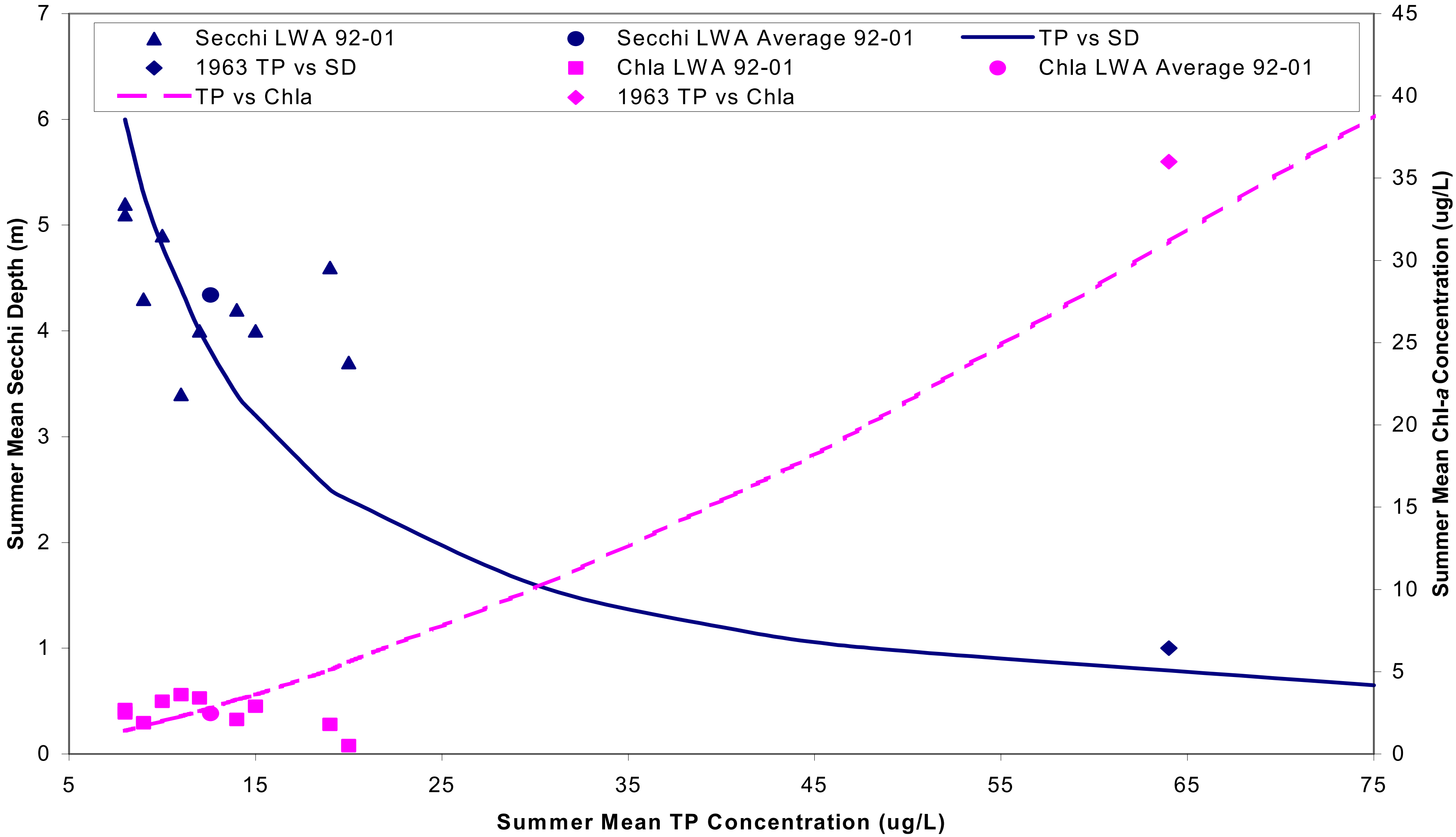


**Figure 36. Relation of Chlorophyll *a*, Total Phosphorus, and Secchi Depth From 1992 to 2001 and Prior to Wastewater Diversion (1963) to the Model Predictions for Carlson's (1977) Trophic State Index**

## 4.3.2. Fisheries

The following section contains excerpts and information from the WRIA 8 Salmon and Steelhead Habitat Limiting Factors Report (Kerwin, 2001) summarizing the current fisheries community in the Cedar-Sammamish Watershed and the current salmonids that utilize Lake Washington. For further information on the salmon population status and habitat conditions in the Cedar-Sammamish Watershed and Lake Washington, refer to the*Salmon and Steelhead Habitat Limiting Factors Report for the Cedar-Sammamish Basin (Water Resource Inventory Area 8)* (Kerwin, 2001), available from the Washington State Conservation Commission.

### *4.3.2.1. Current Fish Status*

Many stocks of the wild salmonid population in the Cedar-Sammamish Watershed, as well as in the Puget Sound ecoregion, have declined significantly. In March 1999, the National Marine Fisheries Service (NMFS) listed Puget Sound chinook salmon as a threatened species under the Endangered Species Act (ESA). In November 1999, the U.S. Fish and Wildlife Service (USFWS) listed bull trout as a threatened species under the ESA (Kerwin, 2001).

The fisheries community in the Cedar-Sammamish Watershed comprises both native and non-native species (Tables 7 and 8). The historically important and current fishery is dominated by chinook salmon, sockeye salmon, coho salmon, kokanee, steelhead, and rainbow and costal cutthroat trout, as well as native char or bull trout (Table 7). Additionally, 24 non-native fish species have been introduced into the Cedar-Sammamish Watershed, creating numerous new trophic interactions with native species (see Table 8 for a complete list). This includes one non-native salmonid (Atlantic salmon).

*Chinook Salmon*

The Cedar-Sammamish Watershed supported an average yearly total run of approximately 9,600 adult chinook salmon from 1968 to 1997. This number represents the fish returning to the river and those that were harvested. However, total returns for naturally produced fish during the past 9 years have averaged less than 550 adult fish. Returns of naturally produced chinook salmon to the Cedar-Sammamish Watershed have experienced the same decline that has occurred in many of the other Puget Sound drainage basins (Kerwin, 2001).

*Coho Salmon*

Coho salmon escapement estimates (the number of coho salmon that survived fish predation and angler pressures) for the tributaries of Lakes Washington and Sammamish from 1980 to 1999 averaged 8,058 fish and ranged from 399 to 20,002 fish. However, escapement estimates are not always indicative of overall habitat productivity because they do not necessarily reflect the harvest of Cedar-Sammamish Watershed Basin origin subadult and adult coho salmon. The Cedar River coho salmon stock was identified as unique based on its spawn timing and its geographic isolation. However, the status of this stock appears to be on a downward trend in escapement. Between 1980 and 1999,

the average escapement was 3,710 fish. While there has been insufficient or no escapement data collected in 4 of the ensuing 10 years, the most recent 2 years indicate extremely poor returns. Since 1991, where data are available, the average coho salmon escapement has been 697 fish (Kerwin, 2001). Coho salmon population decline in the Cedar-Sammamish Watershed can be attributed to spawning and rearing habitat degradation and changes in oceanic conditions.

**Table 7.**
**Salmon Species and Stocks Found in the Cedar-Sammamish Watershed, with NMFS and USFWS Listed or Proposed ESA Listing Status as of June 2000[a]**

| Stock | Stock Origin | Production Type* | Stock Status (SASSI/SASI) | ESA Status (NMFS/USFWS) |
|---|---|---|---|---|
| Issaquah Creek Summer/Fall Chinook | Non-native | Composite** | Healthy | Listed as Threatened |
| North Lake Washington Tributary Summer/Fall Chinook | Native | Wild | Unknown | Listed as Threatened |
| Cedar River Summer/Fall Chinook | Native | Wild | Depressed | Listed as Threatened |
| Cedar River Coho | Mixed | Wild | Depressed | Not Currently Listed |
| Lake Washington and Lake Sammamish Tributary Coho | Mixed*** | Composite | Depressed | Not Currently Listed |
| Winter Steelhead | Native | Wild | Depressed | Not Currently Listed |
| Lakes Washington and Sammamish Tributary Sockeye | Unknown | Wild | Depressed | Not Currently Listed |
| Lake Washington Beach Spawning Sockeye | Unknown | Wild | Depressed | Not Currently Listed |
| Lake Washington-Cedar River Sockeye | Non-native | Composite | Depressed | Not Currently Listed |
| Issaquah Creek Summer-Run Kokanee | Native | Wild | Critical | Petitioned as Endangered |
| Big Bear, Little Bear, and North Creeks Residualized Sockeye | Naturally reproducing | Wild | Unknown | NA |
| Late-Run Lake Sammamish Kokanee | Native | Wild | Unknown | Petitioned as Endangered |
| Lake Washington Rainbow Trout | Non-native | Composite | Unknown | Not Currently Listed |
| Chester Morse Bull Trout | Native | Wild | Unknown, but stable | Listed as Threatened |
| Coastal Cutthroat Trout | Native | Wild | Unknown | Not Currently Listed |

[a] Excerpt from WRIA 8 Salmon and Steelhead Habitat Limiting Factors Report (Kerwin, 2001).

* Production type is the method of spawning and rearing that produces the fish, which constitutes the stock.

** A stock sustained by both wild and artificial production.

*** A stock whose individuals originated from commingled native and non-native parents, and/or by mating between native and non-native fish, or a previously native stock that has undergone substantial genetic alteration.

**Table 8.**
**Introduced or Non-Native Fish Species Found in the Cedar-Sammamish Watershed[a]**

| Common Name | Scientific Name | Population Status | Origin |
|---|---|---|---|
| American shad | *Alsoa spaidissima* | Uncommon strays | E. N. America |
| Atlantic salmon | *Salmo salar* | Stray, can exceed 1,000/yr | N.A. & Europe |
| Black bullhead | *Ictalurus melas* | Extinct | E. N. America |
| Black crappie | *Pomoxis migromaculatus* | Common | E. N. America |
| Bluegill | *Lepomis macrochirus* | Common | E. N. America |
| Brook trout | *Salvelinus fontinalis* | Rarely caught | E. N. America |
| Brown bullhead | *Ictalurus nebulosus* | Rare, may be extinct | E. N. America |
| Brown trout | *Salmo trutta* | No observed reprod. | N. Europe |
| Channel catfish | *Ictalurus punctatus* | Rarely caught | E. N. America |
| Cherry salmon | *Oncorhynchus masou* | Extinct | Japan |
| Common carp | *Cyprinus carpio* | Abundant | Asia |
| Fathead minnow | *Pimephales notatus* | Unknown | E. N. America |
| Goldfish | *Carassius auratus* | Intermittent | Asia |
| Grass carp | *Ctenopharengodon idella* | Triploids only | Asia |
| Lake trout | *Salvelinus namaycush* | Extinct | NE NA+AL |
| Lake whitefish | *Coregonus clupeaformis* | Extinct | NE NA+AL |
| Largemouth bass | *Micropeterus salmoides* | Common | E. N. America |
| Pumpkinseed sunfish | *Lepomis gibbosus* | Abundant | E. N. America |
| Smallmouth bass | *Micropterus dolomieui* | Common | E. N. America |
| Tench | *Tinca tinca* | Abundant | Europe |
| Warmouth | *Lepomis gulosus* | No observed reprod. | E. N. America |
| Weather loach | *Misgurnus angillicaudatus* | No observed reprod. | NE Asia |
| White crappie | *Pomoxis annularis* | Uncommon | E. N. America |
| Yellow perch | *Perca flavescens* | Abundant | NE N. America |

[a] Excerpt from WRIA 8 Salmon and Steelhead Habitat Limiting Factors Report (Kerwin, 2001).

### *Winter Steelhead*

The Cedar-Sammamish Watershed winter steelhead stock has been characterized as depressed. Population declines began in the mid-1980s, similar to other Puget Sound winter steelhead stocks. These declines have been attributed to a multitude of factors, including degraded habitat, harvest, and largely to a change in ocean conditions. However, escapement estimates from recent years indicate an upward trend with the exception of poor returns in 2000 and 2001 (Kerwin, 2001).

*Sockeye Salmon*

In Lake Washington, there are three known production units of sockeye salmon (i.e., groups classified by spawning location). The first and largest production unit is the Cedar River population. The Cedar River produces the greatest proportion of sockeye salmon returning to the Lake Washington Basin. However, this particular stock is depressed and has a declining long-term population trend. The second production unit consists of sockeye salmon spawning in tributaries of Lake Washington other than the Cedar River. This production unit is also depressed with a declining long-term population trend. The third and smallest production unit is the Lake Washington beach spawning stock. This stock has seen the greatest declines of the three production units, for reasons that are unclear. It has been hypothesized that the construction of docks and/or the introduction and explosive distribution of Eurasian watermilfoil may be partially responsible (Kerwin, 2001).

*Kokanee*

Cedar-Sammamish Watershed kokanee, the resident form of sockeye salmon, have been separated into two distinct stocks based on a number of key characteristics, the most important being run timing and unique genetic traits (Young et al., 2001). The early-run stock of kokanee that return to Issaquah Creek are considered native to the Lake Sammamish drainage.

Another stock of kokanee salmon enters east and south Lake Sammamish tributaries (e.g., Laughing Jacobs, Ebright, and Lewis Creeks) from October through early January. These adult kokanee are morphologically distinct from the kokanee mentioned above, with a heavy spotting pattern along their entire dorsal surface and both caudal lobes along with varying degrees of red coloration laterally.

Finally, what has been thought to be a separate kokanee stock present in Bear Creek (sometimes referred to as Big Bear Creek) and Swamp Creek is now believed to be genetically closer to sockeye salmon and has been called a residualized sockeye stock (Young et al., 2001).

*Rainbow Trout*

Rainbow trout in Lake Washington have two life history strategies, the anadromous steelhead and the resident rainbow trout. The life history of the steelhead is similar to salmon species; spawning and rearing occur in freshwater, then the fish migrate to marine waters as juveniles and return to their native streams to spawn. The resident rainbow trout complete their entire life in fresh water. In WRIA 8, resident rainbow trout are hatchery-produced fish that are released into the system for “put-grow and take” or “put and take” recreational fisheries. The hatcher-produced fishery is not believed to be self-sustaining as there is no evidence of natural reproduction and recreational harvest is high (Kerwin, 2001).

### *Coastal Cutthroat Trout*

Assessing populations of coastal cutthroat trout in the Cedar-Sammamish Watershed Basin is particularly difficult. Ludwa et al. (1997) estimated the abundance of coastal cutthroat trout in McAleer Creek at 8 fish per 50 m of stream (Kerwin, 2001). In that same study, the number of coastal cutthroat trout in Lyons Creek was estimated at 30 fish per 50 m of stream. Scott et al. (1986) examined Kelsey Creek in 1979 and found 4 to 5 fish per 50 m, but that was increased to 23 fish per 50 m in 1996 (Ludwa et al., 1997).

### *Native Char (Bull Trout)*

There are known reproducing populations of both adfluvial and stream-resident bull trout in the upper Cedar River, in and above Lake Chester Morse (Berge and Mavros, 2001). Adfluvial populations spend much of their lives in lakes but spawn and rear in streams. The stream-resident populations complete their entire life history in streams. Bull trout have been observed in the lower Cedar River below Landsberg (Berge and Mavros, 2001). Surveys were conducted in 2001 and 2002 in tributaries to the lower Cedar River to determine if a self-sustaining population exists in the lower Cedar River Basin. A native char population may also occur in Issaquah Creek, as indicated by a single observation of a char in Carey Creek, a tributary located in the upper Issaquah Creek Basin (Berge and Mavros, 2001). Further surveys for char populations were proposed for 2001 and 2002 to determine the presence and distribution of native char in the Issaquah Creek Basin. Redd counts conducted from 1992 to 2000 ranged from 2 to 236 redds, but turbidity decreased the viewing conditions in some years, and likely caused underestimation of the number of redds (Kerwin, 2001).

### *Lake Washington Salmonids*

The five salmonid species that use Lake Washington are sockeye salmon, coho salmon, chinook salmon, coastal cutthroat trout, and rainbow/steelhead trout (Table 9). Anadromous forms of each of these species are present, so individuals are present in the lake both as adults during migrations to spawning grounds and as juveniles. Sockeye salmon are known to spawn along some beaches of the lake, and there are unconfirmed reports of chinook salmon spawning in littoral or shallow shoreline areas of the lake (Kerwin, 2001).

**Table 9.**
**Salmonid Species that Utilize Lake Washington**

| Common Name | Scientific Name |
|---|---|
| Chinook salmon | *Oncorhynchus tshawytscha* |
| Sockeye salmon | *Oncorhynchus nerka* |
| Coastal cutthroat trout | *Oncorhynchus clarki clarki* |
| Rainbow/steelhead trout | *Oncorhynchus mykiss* |
| Coho salmon | *Oncorhynchus kisutch* |

Populations known to reside specifically in Lake Washington include non-anadromous forms of winter steelhead (rainbow trout), sockeye salmon (kokanee), and cutthroat. Resident rainbow trout spend their entire life in Lake Washington. The resident rainbow trout population was sustained with hatchery plants because they rarely successfully

reproduce in WRIA 8 (Beauchamp, 1987). Recently, however, releases of hatchery rainbow trout have been all but eliminated. Non-anadromous coastal cutthroat trout also occur in Lake Washington and are much more abundant than the anadromous form (Nowak, 2000). Kokanee is the freshwater, resident form of *O. nerka*, sockeye salmon (Foote et al., 1989; Wood, 1995). Some progeny from the parents of anadromous sockeye salmon may also remain in Lake Washington for all or a portion of their lives (resident/anadromous sockeye) (Kerwin, 2001).

Salmonids primarily use two major habitat zones of the lake: the littoral and limnetic regions. The littoral zone is defined as the shallow water portion of the lake associated with the shoreline; the limnetic zone is the water column portion of the lake extending down to, but not including, the lake's bottom. These two habitat zones have been impacted and altered by human activities throughout the watershed. The alteration of habitat zones in Lake Washington is believed to have an impact on the salmonid species that utilize the lake. Below is a summary of the limiting habitat factors and impacts on Lake Washington and its salmonid population (Kerwin, 2001):

- The riparian shoreline of Lake Washington is highly altered from its historic state. Current and future land-use practices all but eliminate the possibility of the shoreline functioning as a natural shoreline to benefit salmonids because of the lack of structure and shoreline vegetation, as well as hydraulic changes due to bulkheads and docks.
- Introduced non-native plant and animal species have altered trophic interactions between native animal species.
- Riparian habitats are generally non-functional and are disconnected.

## 4.4. Trophic State Indices

Trophic state indices (TSIs) were developed by Carlson (1977) to provide a tool to apply TP, Secchi transparency, and chl *a* values to a uniform scale that can be used to compare lake condition. The mathematical relationship allows for the conversion of TP, Secchi transparency, and chl *a* values to an index number for each parameter between 0 and 100. The greater the index number, the more eutrophic the lake. The agreement between summer average Secchi transparency, chl *a*, and TP for Lake Washington with Carlson's relationships was discussed in Section 2 and shown in Figure 36 (complete data set can be found in Table A-26, Appendix A). Carlson's computational equations are:

$$\mathrm{TSI}_{(SDD)} = 10\left(6 - \left(\frac{\ln \mathrm{SDD}}{\ln 2}\right)\right)$$

SDD = Secchi disk depth, m

$$\mathrm{TSI}_{(Chla)} = 10\left(6 - \left(\frac{2.04 - 0.68 \ln \mathrm{Chl}\, a}{\ln 2}\right)\right)$$

Chl *a* = Chlorophyll *a*, mg/m$^3$

$$\mathrm{TSI}_{(TP)} = 10\left(6 - \left(\frac{\ln(48/\mathrm{TP})}{\ln 2}\right)\right)$$

TP = Total phosphorus, µg/L

The TP-TSIs in Lake Washington show a statistically significant decline (t-test, 95% confidence interval) comparing 1998 through 2001 to earlier years, suggesting a shift from mesotrophic to lower mesotrophic or upper oligotrophic (Figure 37). This is consistent with decreased TP concentrations measured during those years (see Figure 16). The cause or causes of the TP decline have not been identified at this time. Secchi-TSIs showed a similar decline in the last four years of this study (Figure 38). Chl *a*-TSI did not show a corresponding decline (Figure 39), and in fact, increased significantly at the nearshore sites. This range of TSI values for all three constituents (32 to 49) indicates that Lake Washington is mesotrophic.

TSIs are often used to determine if something is limiting chl *a* other than TP, such as turbidity and light (Carlson, 1977). Carlson provided a method of charting the TSI 'residuals' to visually demonstrate areas of inconsistency between indicators (Figure 40a and 40b). Points that fall within the center area (+/- 5 on both axis) represent years in which the three indicators are in general agreement. For most years in this study period, residual TSI values for Lake Washington fell within the center axis range, suggesting agreement between the indicators. If chl *a* was limited by something other than TP, the TSI for chl *a* would be much lower than TSIs for TP and Secchi and the TSI residual point would end up in the lower left quadrant. This occurred at both nearshore and pelagic stations in 1993.

The TSI residuals for the pelagic stations in 1998, 2000, and 2001 fall within the upper right quadrant due low TP and high water clarity – this can occur when algal communities are dominated by larger species/colonies between which the Secchi disk remains visible at high concentrations of chlorophyll *a*. As discussed previously, the magnitude of the spring blooms (March/April) are likely determined more by other factors such as the availability of light and zooplankton-phytoplankton dynamics, than by TP. Arhonditsis et.al. (2003) found that while zooplankton played a dominant role in determining the phytoplankton maximum, it was not clear whether grazing rates or nutrient limitation is the primary cause for the decline in phytoplankton biomass following the spring bloom. They also suggest that since nutrient recycling by zooplankton (*Daphnia pulicaria* and *Daphnia thorata*) provides 60 to 90 percent of the phosphorus input to the mixed layer (Richey 1979), that phytoplankton-*Daphnia* dynamics are a significant regulatory factor for the phytoplankton community properties (abundance and composition) from late spring until the end of September (Schindler unpublished data). Changes in the phytoplankton-*Daphnia* dynamics in recent years may have resulted in less zooplankton nutrient recycling in the mixed layer and perhaps changes in dominant chlorophytes and/or cyanobacteria from late spring to fall. Zooplankton and phytoplankton species and composition were not investigated for this report.

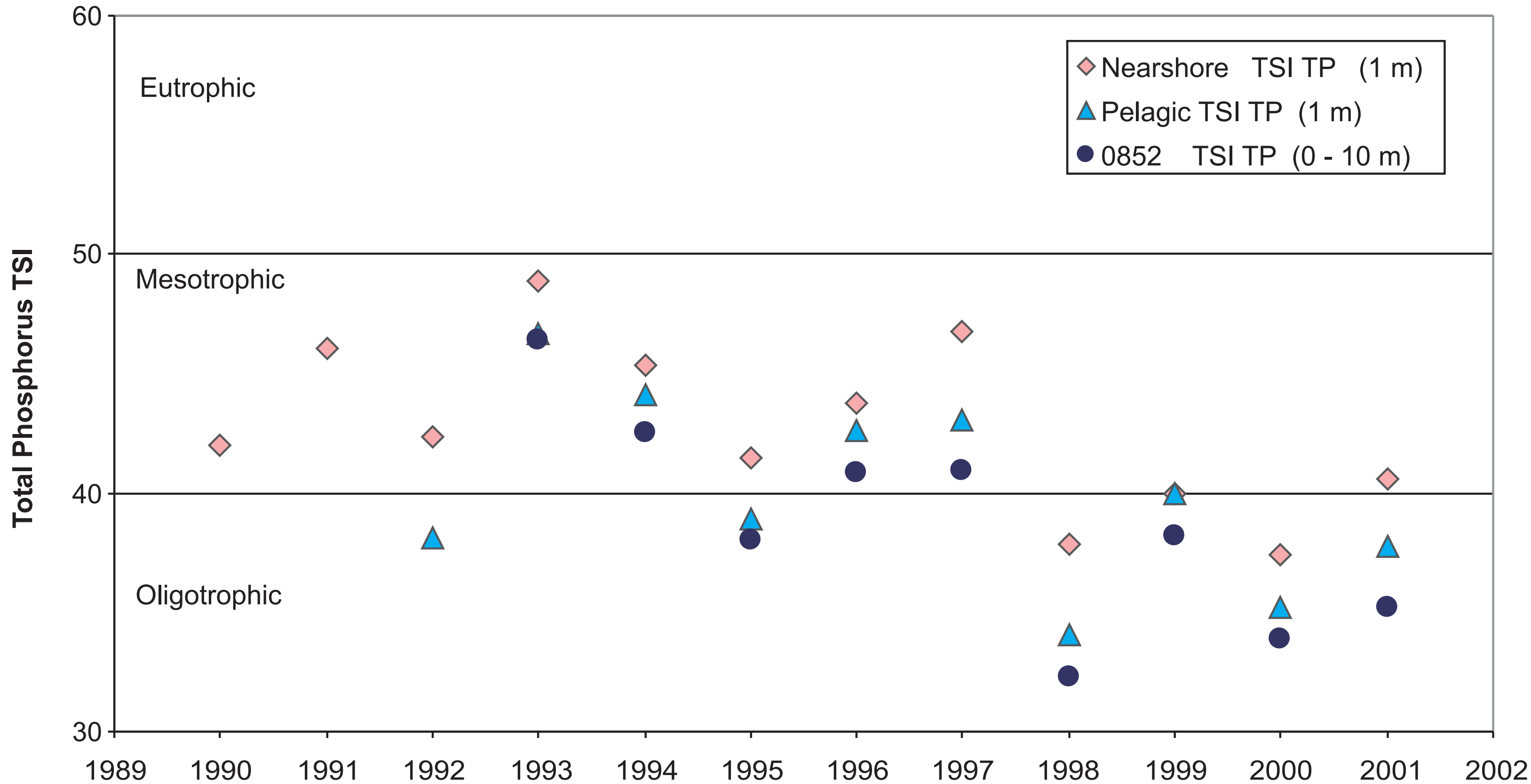


**Figure 37. Summer Mean (May - Sept) Total Phosphorus Trophic State Index for Nearshore Stations, Pelagic Stations, and Station 0852.**

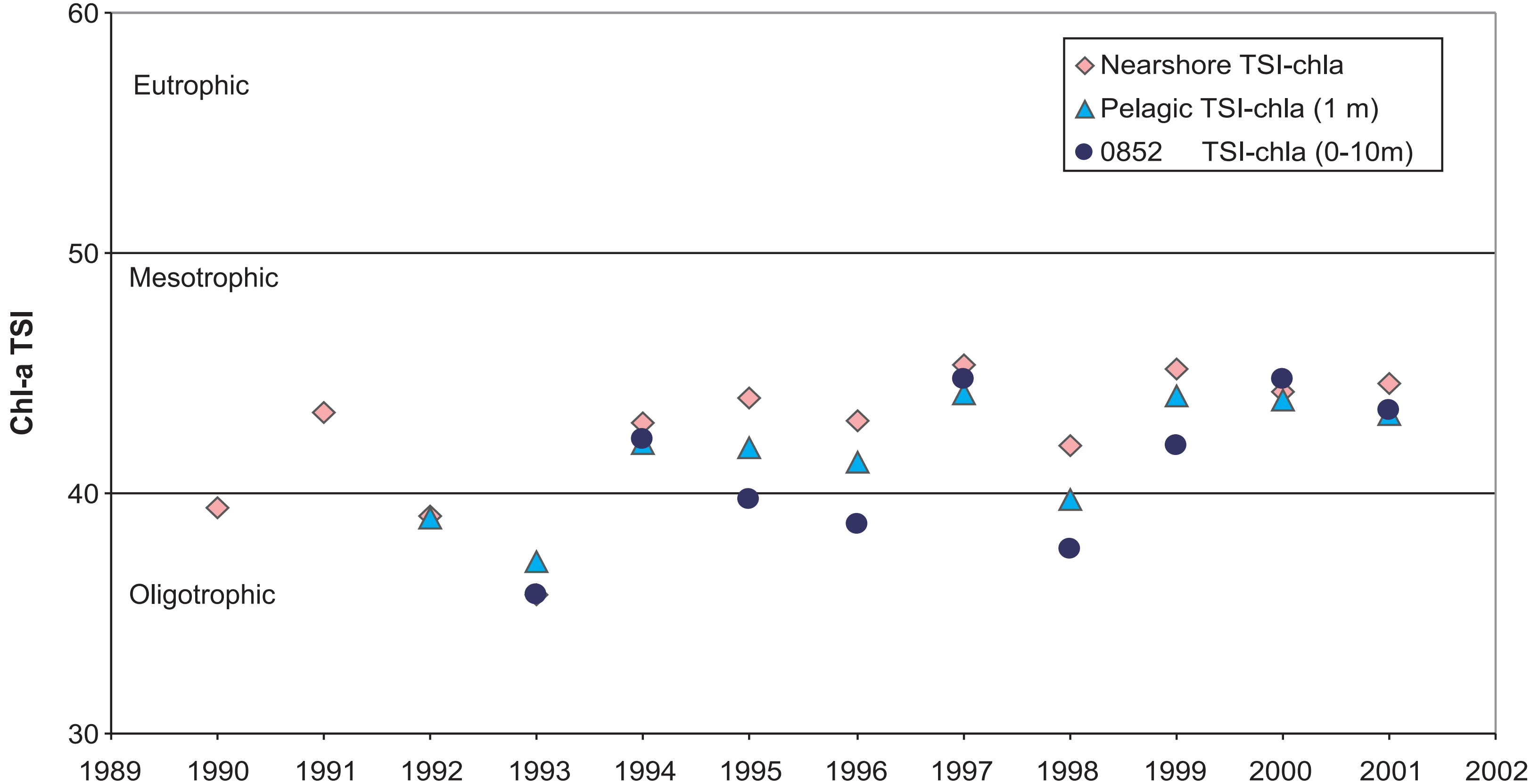


**Figure 38. Summer Mean (May - Sept) Chl-a Trophic State Index for Nearshore Stations, Pelagic Stations, and Station 0852.**

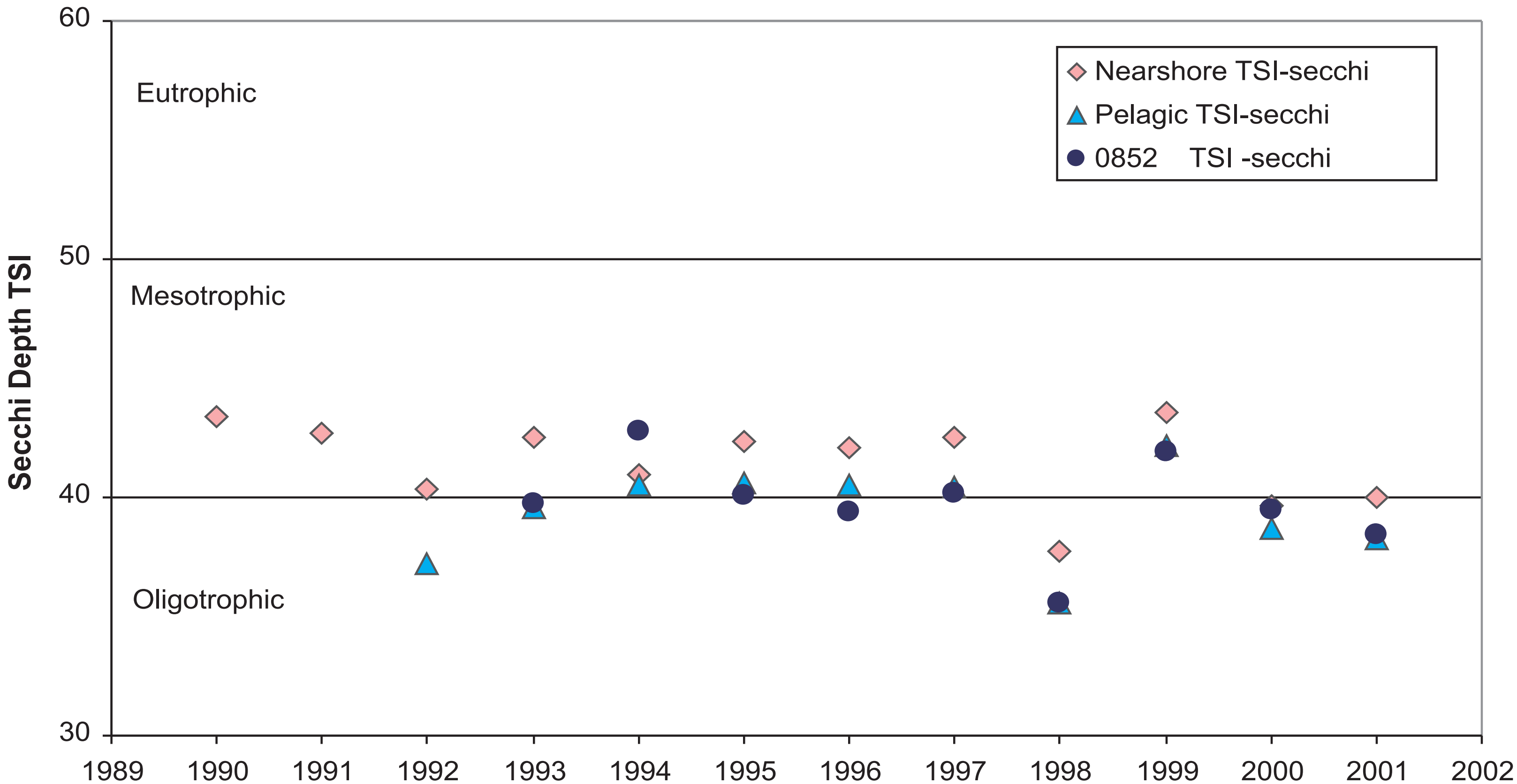


**Figure 39. Summer Mean (May - Sept) Secchi Depth Trophic State Index for Nearshore Stations, Pelagic Stations, and Station 0852.**

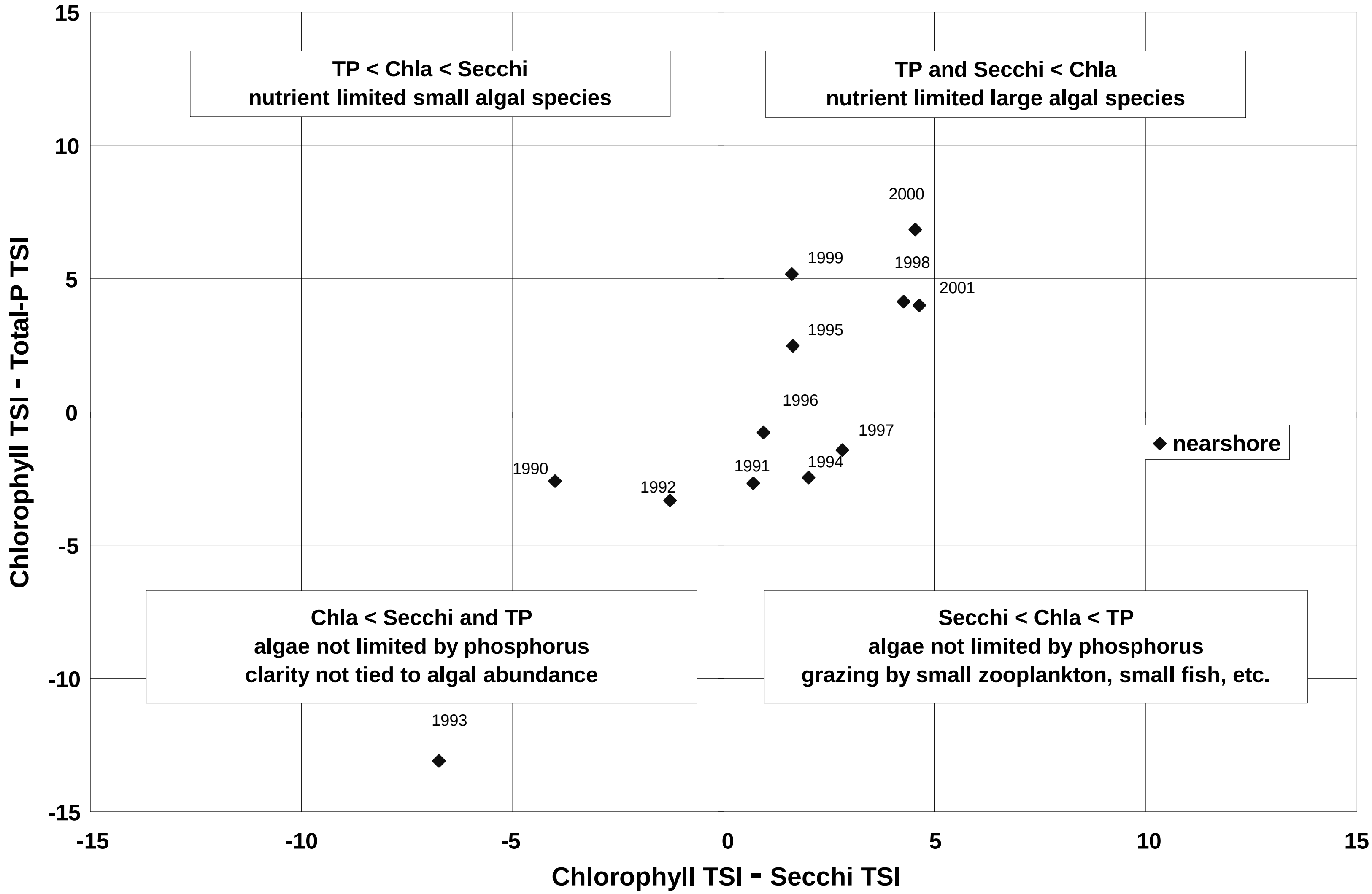


**Figure 40a. Summer Mean (May - Sept) Trophic State Indices Residuals for Lake Washington Nearshore Stations 1990 - 2001.**

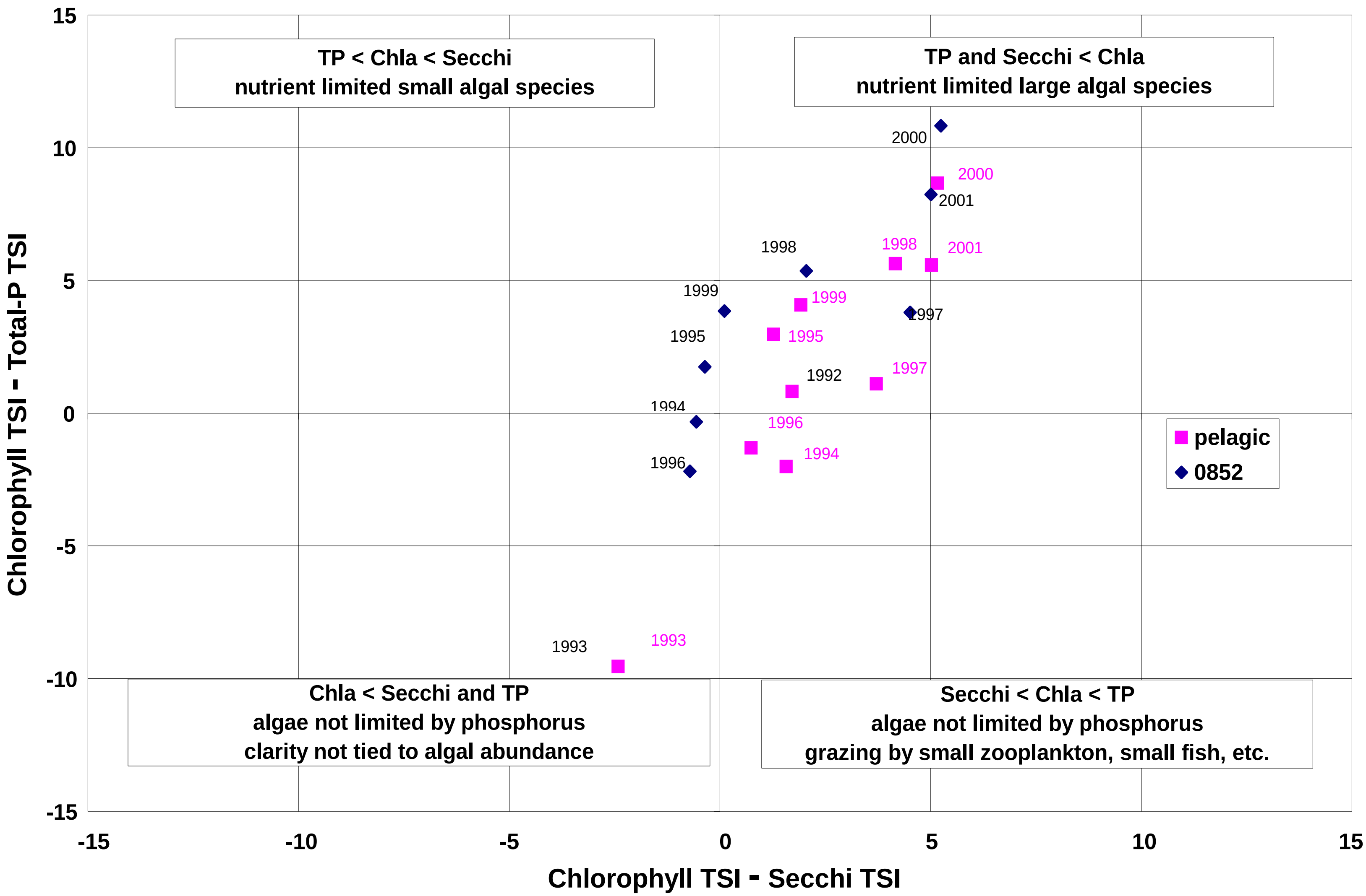


**Figure 40b. Summer Mean (May - Sept) Trophic State Indices Residuals for Lake Washington Pelagic Stations and Station 0852 from 1992 to 2001.**

# 4.5. Metals and Organics

## 4.5.1. Metals Analysis

Lake water was analyzed for 21 metals (dissolved + total) (see Table B-1 in Appendix B for the list of analytes), 18 of which were detected at concentrations greater than the method detection limit (MDL) (Table 10). Table A-25 in Appendix A contains summary statistics for all metals analyzed. Sampling for metals and organic constituents in Lake Washington was recently initiated as part of the SWAMP project; very few data are available prior to 2000.

King County's Major Lakes Monitoring Program and the King County Environmental Laboratory (KCEL) have used inductively coupled plasma mass spectrometry (ICP-MS; USEPA methods 200.8 and 6020) to measure metal concentrations since 1998. ICP-MS is capable of achieving the detection limits required to detect metals at typical ambient water concentrations. Methods used prior to 1998, such as inductively coupled plasma optical emission spectroscopy (ICP-OES; USEPA methods 200.7 and 6010), had detection limits 5 to 20 times higher than ICP-MS, and did not detect metals at typical ambient concentrations.

Mercury is analyzed using cold vapor atomic absorption (CVAA; USEPA method 245.2) or cold vapor atomic fluorescence (CVAF; USEPA 1631 modified).

In samples where metals were not detected, one-half the MDL was assumed as the maximum metal concentration.

**Table 10.**
**Metals Detected in Lake Washington Water Samples at Concentrations Greater Than Method Detection Limit for Method Indicated**

| | | |
|---|---|---|
| Aluminum, dissolved, ICP-MS | Cobalt, dissolved, ICP-MS | Molybdenum, dissolved, ICP-MS |
| Aluminum, total, ICP | Cobalt, total, ICP-MS | Molybdenum, total, ICP-MS |
| Antimony, dissolved, ICP-MS | Copper, dissolved, ICP-MS | Nickel, dissolved, ICP-MS |
| Antimony, total, ICP-MS | Copper, total, ICP-MS | Nickel, total, ICP-MS |
| Arsenic, dissolved, ICP-MS | Iron, dissolved, ICP | Silver, total, ICP-MS |
| Arsenic, total, ICP-MS | Iron, total, ICP | Thallium, dissolved, ICP-MS |
| Barium, dissolved, ICP-MS | Lead, dissolved, ICP-MS | Thallium, total, ICP-MS |
| Barium, total, ICP-MS | Lead, total, ICP-MS | Mercury, total, CVAF |
| Calcium, dissolved, ICP | Magnesium, dissolved, ICP | Vanadium, dissolved, ICP-MS |
| Cadmium, total, ICP-MS | Magnesium, total, ICP | Vanadium, total, ICP-MS |
| Chromium, dissolved, ICP-MS | Manganese, total, ICP-MS | Zinc, dissolved, ICP-MS |
| Chromium, total, ICP-MS | Mercury, total, CVAA | Zinc, total, ICP-MS |
| | Mercury, dissolved, CVAF | |

### 4.5.2. Organics Analysis

Lake water was analyzed for 163 organic compounds (see Table B-2 in Appendix B for the list of analytes). Twenty organic compounds were detected at concentrations greater than their MDLs (Table 11). See Table A-25 in Appendix A for summary statistics for all organics analyzed.

**Table 11.**
**Organic Compounds Detected in Lake Washington Water Samples at Concentrations Greater Than Method Detection Limit**

| | | |
|---|---|---|
| 2,4-D | Benzo(k)fluoranthene | Fluoranthene |
| 2-Nitrophenol | Bis(2-Ethylhexyl)Phthalate | Indeno(1,2,3-Cd)Pyrene |
| Acenaphthylene | Caffeine | Isophorone |
| Benzo(a)pyrene | Chrysene | Naphthalene |
| Benzo(b)fluoranthene | Dimethyl Phthalate | Phenanthrene |
| Benzo(g,h,i)perylene | Di-N-Octyl Phthalate | Phenol |
| Pyrene | Total PAH Immunoassay | |

### 4.5.3. Metals and Organic Compounds Compared to Water Quality Standards

Concentrations of metals and organic compounds in Lake Washington water samples were compared to *Water Quality Standards for Surface Waters of the State of Washington* (WAC 173-201A) to assess how well Lake Washington meets concentrations established to protect beneficial uses (e.g., public health, fish and wildlife use, and recreation). These standards include numeric criteria[1] established for protection of aquatic life, including both acute and chronic exposure concentrations.

WAC standards for the compounds analyzed are summarized in Table 12. Many of the metal standards are hardness-dependent. The hardness-dependent standards were calculated using the mean sample hardness of 37.2 mg/L (see Table C-1 in Appendix C for the hardness-dependent water quality standard equations). The standard deviation of hardness was only 2.0 mg/L (n = 455 samples). Therefore, use of the mean hardness was considered appropriate for calculating hardness-dependent metal water quality standards.

Within the study period, a single metals sample exceeded WAC numerical standards. Dissolved lead, reported as 1.39 µg/L on January 7, 2002, at Station 0826 (mid-lake off Sand Point), exceeded the chronic standard of 0.715 µg/L. Dissolved lead (ICP-MS) concentrations are plotted by date in Figure 41. The highest dissolved lead concentration was compared to the 14 other samples containing dissolved lead above the MDL; using Grubb's test for detecting outliers, this concentration was found to be an outlier. Lead concentrations in samples collected on January 7, 2002, from Station 0831 (mid-lake south), Station 0852 (Madison Park), and Station 0890 (south of I-90, south-central basin), while below water quality standards, were also elevated compared to samples

[1] Criteria refers to the maximum concentration of a chemical allowed under national or state regulations.

previously collected. With the most elevated lead levels occurring on January 7, 2002, at three different stations, sample contamination could have occurred in the lab, or lead levels may have been elevated in Lake Washington that day. The implication of the lead criteria exceedance will be examined in the screening level risk assessment.

**Table 12.**
**Washington State Freshwater Acute and Chronic Numerical Water Quality Standards for Protection of Aquatic Life for Compounds Analyzed**

| Parameter Analyzed | Acute Standard (µg/L) | Chronic Standard (µg/L) |
|---|---|---|
| Aldrin | 2.5 | 0.0019 |
| Arsenic, dissolved | 360 | 190 |
| Cadmium, dissolved | 1.21[a] | 0.47[a] |
| Chlordane | 2.4 | 0.0043 |
| Chloride | 860,000 | 230,000 |
| Chlorpyrifos | 0.083 | 0.041 |
| Chromium (III), dissolved | 244.1[a] | 79.2[a] |
| Copper, dissolved | 6.70[a] | 4.88[a] |
| 4,4'-DDD | 1.1 | 0.001 |
| 4,4'-DDE | 1.1 | 0.001 |
| 4,4'-DDT | 1.1 | 0.001 |
| Endosulfan | 0.22 | 0.056 |
| Endrin | 0.18 | 0.0023 |
| Gamma-BHC (Lindane) | 2 | 0.08 |
| Heptachlor | 0.52 | 0.0038 |
| Lead, dissolved | 18.34[a] | 0.715[a] |
| Mercury, dissolved | 2.10 | - |
| Mercury, total | - | 0.012 |
| Nickel, dissolved | 613[a] | 68.1[a] |
| Parathion | 0.065 | 0.013 |
| Pentachlorophenol (PCP) | 9.07[b] | 5.73[b] |
| Polychlorinated Biphenyls (PCBs) or Aroclors | 2.0 | 0.014 |
| Selenium, total | 20 | 5 |
| Silver, dissolved | 0.63[a] | - |
| Toxaphene | 0.73 | 0.0002 |
| Zinc, dissolved | 49.5[a] | 45.2[a] |

[a] Indicates hardness-dependent standard. Average hardness of 37.2 mg/L was used to calculate these criteria.
[b] Indicates pH-dependent standard. A pH of 7 was used to calculate these criteria.

No organic compounds exceeded WAC numerical standards.

With regard to concentrations of metals and organic compounds, Lake Washington demonstrates generally very good water quality. Based on analysis of samples collected in 2000 and 2001, with the exception of lead in one sample, concentrations of metals and organic compounds in Lake Washington are below Washington State chronic and acute water quality standards.