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# Effects of a temperature control device on nutrients, POM and plankton in the tailwaters below Shasta Lake, California

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**Abstract**

A temperature control device (TCD) was installed by the U.S. Bureau of Reclamation on Shasta Dam, California, in March 1997 for controlling downstream river temperatures. Temperature modification was required to aid recovery of the endangered winter run chinook salmon (*Oncorhynchus tshawytscha*) in the Sacramento River, and to minimize loss of generating capacity as a result of releasing deeper, colder water through low level outlet works to meet downstream temperature criteria. This study began two years prior to operation of the TCD, to compare pre- and post-operational changes on downstream tailwaters, including nutrients, particulate organic matter (POM) and plankton. During epilimnetic withdrawals from January to mid-June, and mid-level withdrawals through August, operation of the TCD was associated with decreases in dissolved nitrate–nitrate concentrations, localized increases in small particulate organic matter (SPOM) at Shasta tailwaters, increases of bacillariophyta ($<25$ $\mu$m size fraction), and increases in copepod biomass. These changes can potentially influence the food base of the river and therefore fish production in the Upper Sacramento River, including the chinook salmon.

## Introduction

Selective withdrawal is becoming a popular alternative for western U.S. reservoirs where operations have impacted native fishes. A selective withdrawal system has the capability to withdraw water from a stratified reservoir equipped with multilevel outlet structures to meet temperature or other downstream water quality objectives (Ford, 1990). With the exception of Cassidy's (1989) extensive review of selective withdrawal impacts on physicochemical conditions in the tailwater and some reference to biological impacts, there are few reports of effects on downstream nutrients, particulate organic matter (POM), phytoplankton and zooplankton that may show significant impacts downstream but are not well documented (Miller, 1984).

Particulate organic matter is important to the riverine community because it provides a trophic connection between microbial assemblages and macroconsumers (Kondratieff & Simmons, 1985). Particulate organic matter constitutes the heterotrophic base of the food web in many aquatic systems and is an important factor structuring biotic communities in riverine ecosystems (Vannote, 1980). Impoundments created by dams may trap sediment and the POM transported into them (Petts, 1984), although Lieberman & Burke (1993) found that this was not the case in the series of impoundments located on the lower Colorado River. Depending on depth of reservoir discharge (Martin & Arneson, 1978), tailwaters usually contain a high density of lentic phytoplankton and zooplankton that decreases rapidly with distance from the outfall (Hynes, 1970; Novotny & Hoyt, 1982). Wright (1967) concluded that lakes with surface-water outflow tend to dissipate heat and trap nutrients, whereas reservoirs with subsurface outflow dissipate nutrients and store heat.

A temperature control device (TCD) was installed on Shasta Dam on the Sacramento River, California, and began to operate in late winter of 1997 for the purposes of controlling downstream river temperatures to aid recovery of the endangered winter run chinook salmon (*Oncorhynchus tshawytscha)* and to minimize loss of generating capacity while releasing deeper, colder water through low level outlet works. The goal is to maintain temperatures at or below 13 °C for approximately 80 km downstream. The Sacramento and its tributaries support spring, fall, late fall and winter races. Chinook salmon can migrate only as far as Keswick Dam which is located about 15 km below Shasta Dam. The winter run start their upstream migration in mid-December and continue into August. They spawn from mid-April through mid-August and young fish begin to emerge from the gravel in mid-July. Rearing and migration extend through the following spring. Water temperature is a significant factor affecting chinook salmon spawning success, egg survival and juvenile fish growth in the Sacramento River. Chinook salmon have limited temperature tolerances and occur in the Sacramento River at all times of the year and at various life stages. Therefore, maintaining favourable water temperatures along with optimum environmental conditions may enhance survival (Fisher, 1993).

Historically, Shasta Dam was operated as a hypolimnetic release reservoir. In 1987, bypass releases were instituted as a conservation measure for chinook salmon. At present, operation of the TCD allows top strata of water to be released from January through June via upper gates, conserving the pool of cold water for use from late summer through fall. As the summer season progresses withdrawals move deeper into the hypolimnion, and if needed, deeper than the old penstock intakes. With the exception of surface withdrawal capabilities, TCD operations mimic bypass releases.

The current limnological study began in spring 1995, 2 years prior to operation of the TCD and continued for 2 years after TCD initiation. Our objective was to compare pre- and post-TCD operational changes on the quantity and/or quality of nutrients, POM and plankton potentially influencing the riverine food base and, therefore, fish production.

## Methods

Shasta Lake is located 19 km north of Redding in northern California, and is part of the Central Valley Project, a federal water project operated by the Bureau of Reclamation. Shasta Dam, a 183 m high curved concrete gravity structure with a crest elevation of 328 m above mean sea level was completed in 1945 forming the largest reservoir in California. The 56 km long reservoir has 588 km of shoreline, a surface area of 11 940 hectares, a maximum depth of 145 m, and contains $5.55 \times 10^9$ $m^3$ of water at full pool. The reservoir is characterized as monomictic, however, it does not turn over completely, and remains ice-free in the winter. High runoff from rain and snowmelt occurs from the end of December through March each year. Three tributary arms of Shasta are formed by the Pit, McCloud and Sacramento Rivers inflows. The outflow waters for the reservoir form the Sacramento River. Salmon and other migratory fish are trapped by Keswick Dam which creates an afterbay for Shasta Lake. Water travel time from Shasta Dam to Keswick Dam ranges from 17 h in the spring to 81 h in the fall depending on discharge from Shasta Lake.

Beginning in April 1995, two river sites on the Sacramento River were sampled monthly (Fig. 1) to determine if the TCD had any effect on downstream tailwaters. The uppermost site was in the tailwater of Shasta Dam (0.8 km downstream from Shasta Dam) and most likely affected by changes in reservoir operation due to the TCD simply because of proximity to the reservoir. The tailwater of Keswick Dam (15 km downstream from Shasta Dam and about 0.8 km downstream of Keswick Dam) was sampled to detect any differences in data between Shasta and Keswick tailwaters. There are a few minor tributaries that enter into the mainstem river between Shasta and Keswick Dam. Samples were not collected during high flows from October to December 1995, December 1996 and December 1997. Eleven sampling stations on Shasta Lake (Lieberman & Horn, 1998) were also established but only the main part of the lake will be discussed in brief in this paper to help explain downstream variability in the tailwaters. Detailed effects of the TCD on the reservoir are presently being prepared for a manuscript by the authors.

Grab water samples were collected at each river station and in the reservoir at surface, 10 m, 20 m and 85 m for total phosphorus (1 $\mu$g/l detection limit), soluble reactive phosphorus (1 $\mu$g/l detection limit), dissolved nitrate and nitrite nitrogen (1 $\mu$g/l detec-

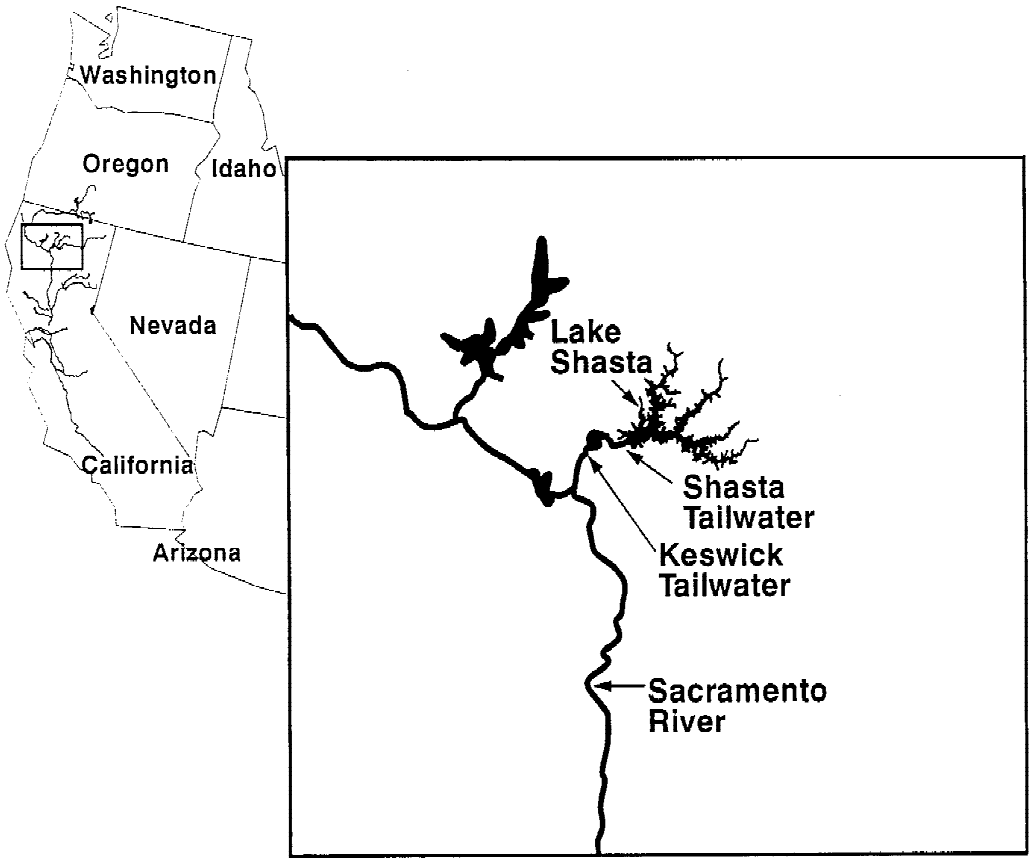


*Figure 1.* Location of Shasta tailwater and Keswick tailwater sampling sites on the Upper Sacramento River below Shasta Lake, California.

tion limit), and ammonium nitrogen (2 $\mu$g/l detection limit) and later analyzed at UC-Davis labs. Values less than the detection limit were reported by the lab and used in data analysis. Water samples were collected at each station 0.5 m below the surface, stored at 4 °C, and shipped overnight express to UC-Davis labs for analysis. Water samples for the dissolved fractions were filtered immediately upon arrival. Methods for nutrient analysis were followed according to Brzezinsik (1987), Kamphake (1967), Murphy & Riley (1962) and Strickland & Parsons (1972).

Particulate organic matter (POM) samples were collected and fractionated into three sizes: larger than 505 $\mu$m (LPOM for large particles), 25–505 $\mu$m (MPOM for medium particles) and 25–1.2 $\mu$m (SPOM for small particles) using a series of plankton nets (Fisher & Likens, 1973; Gurtz et al., 1980; Elser & Kimmel, 1985). Three replicate samples to increase sensitivity of collections (Lieberman & Burke, 1993) were collected for large POM, using a 505 $\mu$m net (three meters in length with a 0.5 m diameter mouth). The 505 $\mu$m net was deployed in the current for 3–4 min to collect each replicate. A calibrated flow meter mounted across the mouth of the net was used to determine the volume of water filtered. During extreme high flow events LPOM samples were not collected due to unsafe boating conditions in the river. Triplicate water samples for MPOM were obtained by pumping 96 l of water through a 25 $\mu$m plankton net and bucket. The sample was collected in the bucket and transferred to a 500 ml amber sample bottle. This procedure was repeated three times and collected material greater than 25 $\mu$m. Three liters were collected for each SPOM replicate by filtering water pumped from the river with an electric sump pump through a 25 $\mu$m plankton net into a 9 l bucket and collecting the filtrate. This was repeated three times. All samples were processed within 2 h of collection. Samples were

filtered through ashed, pre-weighed Gelman 47 mm glass fiber A/E filters (particle retention 1.2 $\mu$m). Filters were oven-dried at 75 ° C for 24 h, weighed for dry weight, ashed at 500 ° C for 1 h and weighed for ash weight. Dry weight (seston), ash weight (inorganic) and ash-free dry weight (POM) were calculated as g/m$^3$ according to Strickland & Parsons (1972). The greater than 505 $\mu$m concentration was subtracted from the greater than 25 $\mu$m concentration to obtain the 25–505 $\mu$m concentration.

Two grab chlorophyll samples were collected about 0.5 m below the river surface. In addition, duplicate water samples for chlorophyll *a* were pumped from 0, 5, 10, 15, 20, 25, 30 m from the main part of the reservoir and composited into a 0–30 m water sample for analysis. From 150 ml to 250 ml of water was filtered through Whatman GF/C filters (47 mm, particle retention 1.0 $\mu$m) immediately after collection. Filters were frozen until processed. Samples were extracted with methanol, analyzed fluorometrically for chlorophyll *a* ($\mu$g/l) by University of California-Davis lab (UC-Davis), and corrected for pheophytin according to Strickland & Parsons (1972).

Duplicate water samples for phytoplankton in the tailwaters were collected, separated into size fractions of greater than 505 $\mu$m (500 ml), 25–505 $\mu$m (125 ml) and 1.2–25 $\mu$m (1 l), according to the above netting protocols for POM, and preserved in Lugols solution (Eaton et al., 1995). Duplicate water samples for phytoplankton were pumped from 0, 5, 10, 15, 20, 25, 30 m from the main part of the reservoir and composited into a 0 to 30 m water sample (250 ml). Phytoplankton were identified to species and reported as biovolume ($\mu$g/l) by BSA Environmental Services, Inc. Cell biovolumes of all taxa were quantified on a per milliliter basis using the Utermohl method (Lund et al., 1958). Biovolumes were estimated using formulae for solid geometric shapes that most closely matched the cell shape (Burkholder & Wetzel, 1989). Subaliquots were examined after cells settled for at least 20 h at 400× under phase contrast with an inverted microscope. Homogeneity in settling was checked to determine minimum number of fields necessary to obtain consistency in enumerating abundant taxa (greater than 5% of the total biovolume) per unit chamber area (Burkholder & Wetzel, 1989).

Duplicate zooplankton samples in the tailwaters were collected from the 25 to 505 $\mu$m size fraction as described for POM above and preserved with Lugols solution (Eaton et al., 1995). Duplicate zooplankton tows were collected from 0 to 30 m in the main part of the reservoir with a 64 $\mu$m birge-style closing net. Zooplankton were identified to species and reported as biomass ($\mu$g dry weight per liter) by BSA Environmental Services, Inc and based on established length/width relationships (Dumont et al., 1975; McCauley, 1984; Lawrence et al., 1987). The lengths or the lengths and widths of each species were measured from a composite sample formed by pooling 5 ml from each sample for that date. Number of specimens examined was equal to 25 for common species and lesser for more rare taxa. In accordance with McCauley (1984), biomass was computed for the appropriate number of individuals for each sample location and the arithmetic mean biomass was multiplied times the species abundance to produce a species biomass for each sample.

The study objective was to examine changes in tailwater variables at Shasta and Keswick before and after TCD operation. Change in river systems is difficult to prove given the constraints of hypothesis testing in lotic systems (e.g. Hurlbert, 1984; also see Cherry, 1998 and Johnson, 1999). Classical inferential statistics cannot be used to demonstrate changes because of inherent problems with pseudoreplication, inability to randomly select samples from sites, and lack of independence between sites in the same river. Estimation and presentation of information on the variability of data are most useful in showing differences in such studies (Hurlbert, 1984; Cherry, 1998). Box and whisker plots were therefore used (Sigma Plot, 1999, version 5.0, SPSS Inc.) to graphically compare data to evaluate parameter changes at Shasta (upstream) and Keswick (downstream) tailwaters between pre- and post-TCD phases, and between stations. Data used to represent the pre-TCD phase included months from April through September 1995 and January through August 1996 ($n$=14), excluding data collected from late fall to early winter. During the pre-TCD phase, either hypolimnetic withdrawals or bypass releases were discharged from the reservoir, and as a result the top 20–30 m of water in the reservoir were not disturbed (Hanna, 1999). Data from the post-TCD phase only included the months from February through July 1997 and January 1998 through July 1998 ($n$=12 or 13) when surface to mid-level withdrawals, via top to mid gates, were discharged from the reservoir. This scenario was not possible in the pre-TCD phase. The interquartile range of the box represented data in the 25–75th percentile range, and whiskers in the 10–90th percentile range. Dots represented data outliers in the 10–90th percentile range. We defined differences (in-

creases or decreases) in data only when there were lack of overlap in the 25–75th percentile range (interquartile) between boxes. The software program described above did not include the option of using confidence intervals in the box plots.

## Results

Nutrients in Shasta Lake influenced downstream tailwaters to some extent. Nutrients in Shasta Lake ranged from minimum detection limits (MDL) – 160 $\mu$g/l for nitrite-nitrate, MDL – 30 $\mu$g/l for ammonia, MDL – 60 $\mu$g/l for soluble reactive phosphorus (SRP) and MDL – 100 $\mu$g/l for total phosphorus (TP). Hypolimnetic values were typically higher than epilimnetic values. For example, in the main part of Shasta Lake, nitrite-nitrate concentrations increased in the hypolimnion in late spring to 80 $\mu$g/l during May 1996, then decreased to about 50 $\mu$g/l by September 1996, and below 40 $\mu$g/l by January 1997. By contrast, in the epilimnion nitrite-nitrate maximum values reached 60 $\mu$g/l in January 1996 and decreased to the MDL in April and May 1996. This pattern was reflected in nutrient concentrations downstream (Lieberman & Horn, 1998). Much of the hypolimnetic nutrient pool was composed of soluble reactive phosphorus. Total phosphorus (20–40 $\mu$g/l) to SRP (30–90 $\mu$g/l) ratio was found to be generally less than two in the hypolimnion of the reservoir. Epilimnetic TP/SRP ratios were highest in late summer (maximum TP/SRP ratio was equal to 27; concentrations were 27 and 1 $\mu$g/l, respectively) during deep level withdrawals.

During the post-TCD phase, only nitrite-nitrate concentrations (Fig. 2a–d) decreased between pre- and post-TCD phases in Shasta tailwater (range 55–186 $\mu$g/l (pre-TCD), MDL–68 $\mu$g/l (post-TCD)) and Keswick tailwater (range 38–113 $\mu$g/l (pre-TCD), MDL–60 $\mu$g/l (post-TCD)), exhibited by a lack of overlap in the 25–75th interquartile range. The interquartile ranges overlapped for ammonium nitrogen, SRP and TP and for all nutrient concentrations upstream in Shasta tailwaters to downstream in Keswick tailwaters.

There was overlap in the interquartile range of LPOM and MPOM (Fig. 3a,b) between pre- and post-TCD phases. There was an increase in SPOM at Shasta tailwaters between the pre- and post-TCD phase, whereas downstream at Keswick, SPOM did not change dramatically as illustrated by overlap of the interquartile range (Fig. 3c). The range of SPOM

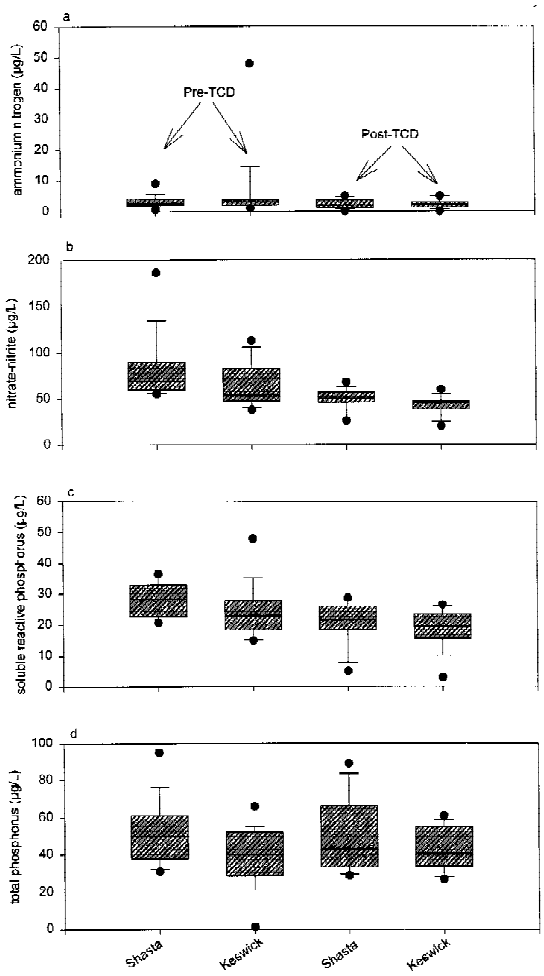


*Figure 2.* Comparison of nutrient concentrations in Shasta and Keswick tailwaters between pre-TCD and post-TCD phases. (a) ammonium nitrogen; (b) nitrate-nitrite; (c) soluble reactive phosphorus; and (d) total phosphorus. Data used to represent pre-TCD phase included the months from April through September 1995 and January through August 1996 ($n$=14), excluding data collected from late fall to early winter. Data from the post-TCD phase only included the months from February through July 1997 and January 1998 through July 1998 ($n$=12) when surface to mid-level withdrawals (i.e. top-to- mid-gates opened) were discharged from the reservoir. The interquartile range of the box represents from 25th to 75th percentile data, and whiskers from 10 to 90th percentile data. Dots represent outliers, that datum outside of the 10 to 90th percentile. We assumed differences in data only if there were lack of overlap in 25 to 75th percentile data between boxes.

at Shasta and Keswick tailwaters was 0.37–1.56 $\mu$g/l and 0.41–1.58 $\mu$g/l during the pre-TCD phase, and 0.56–2.47 $\mu$g/l and 0.52–2.77 $\mu$g/l during post-TCD phase, respectively. SPOM contributed about 90%

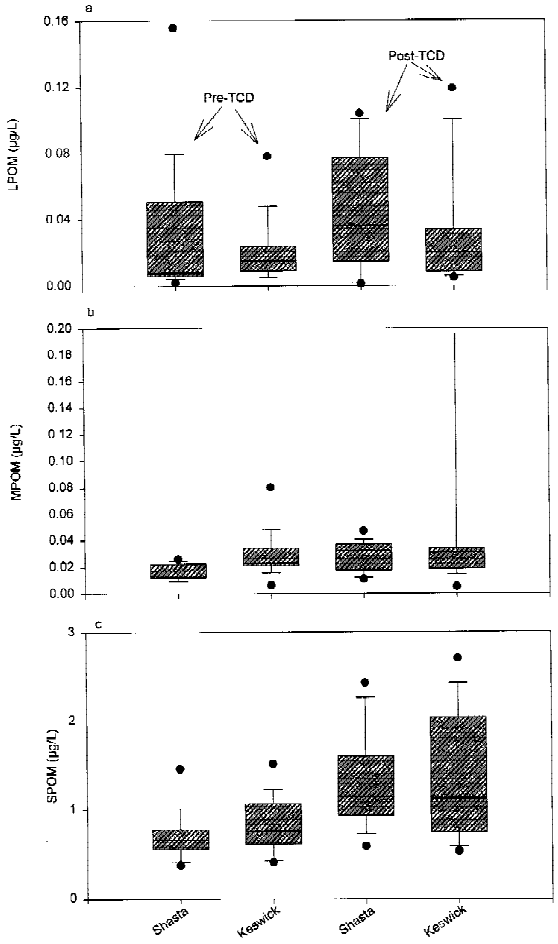


*Figure 3.* Comparison of particulate organic matter concentrations in Shasta and Keswick tailwaters between pre-TCD and post-TCD phases. (a) large particulate organic matter (LPOM); (b)medium particulate organic matter (MPOM); and (c) small particulate organic matter (SPOM). Data used to represent pre-TCD phase included the months from April through September 1995 and January through August 1996 ($n$=13), excluding data collected from late fall to early winter. Data from the post-TCD phase only included the months from February through July 1997 and January 1998 through July 1998 ($n$=13) when surface to mid-level withdrawals (i.e. top-to-mid-gates opened) were discharged from the reservoir. See Figure 2 for explanation of box plots.

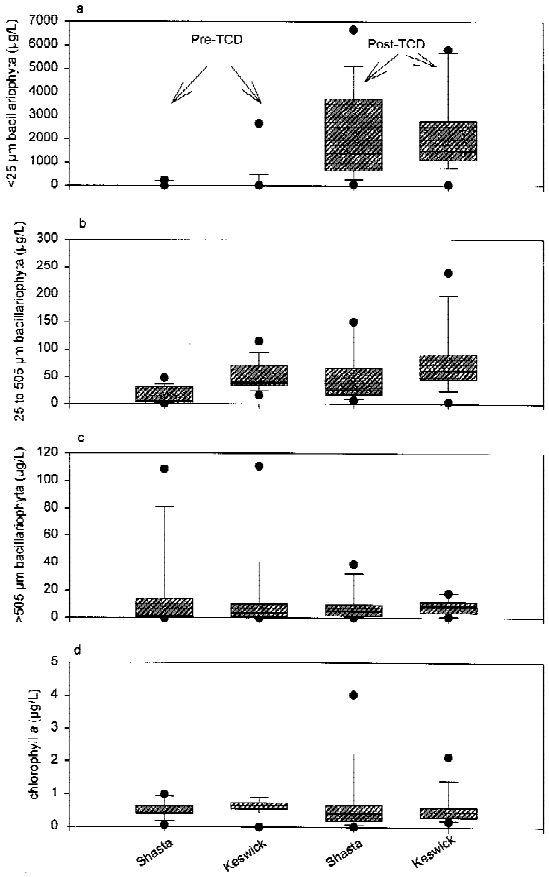


*Figure 4.* Comparison of bacillariophyta biovolume and chlorophyll *a* concentration in Shasta and Keswick tailwaters between pre-TCD and post-TCD phases. (a) <25 $\mu$m bacillariophyta; (b) 25–505 $\mu$m bacillariophyta (c) >505 $\mu$m bacillariophyta; and (d) chlorophyll *a* concentration. Data used to represent pre-TCD phase included the months from April through September 1995 and January through August 1996 ($n$=14), excluding data collected from late fall to early winter. Data from the post-TCD phase only included the months from February through July 1997 and January 1998 through July 1998 ($n$=13 (bacillariophyta), $n$=12 (chlorophyll *a*)) when surface to mid-level withdrawals (i.e. top-to- mid-gates opened) were discharged from the reservoir. See Figure 2 for explanation of box plots.

to the total POM and was composed mainly of detrital and small plankton fragments. Composition of the MPOM included phytoplankton, zooplankton and plant fragments, as well as detrital material. Cladophora strands, large zooplankton (i.e. *Daphnia laevis*, *Leptodiaptomus ashlandi*) and detrital material were the major components of the LPOM. Cladophora beds that occurred in Shasta tailwaters were more common than downstream in Keswick tailwaters, although there were no differences in LPOM concentrations between the two sites.

Most phytoplankton in the tailwaters were diatoms (<25 $\mu$m) that form blooms during spring and fall months in Shasta Lake. Phytoplankton is often sparse during summer due to depletion of nutrients and thermal stratification that occurs in the reservoir

(Lieberman & Horn, 1998). Composition of the algal drift (<25 $\mu$m) included more than 98% diatoms in Shasta tailwater, and 90% diatoms, 5% green algae, 2% blue-greens and 3% other algae in Keswick tailwater. Dominant diatoms collected from the tailwaters were *Melosira varians* and *Melosira rubens*. These are the same species that dominate the phytoplankton of Shasta Lake (Lieberman & Horn, 1998). Bacillariophyta biovolume (<25 $\mu$m) in Shasta tailwaters ranged from 0.06 to 243 $\mu$g/l (pre-TCD) and 40 to 6656 $\mu$g/l (post-TCD), and in Keswick tailwaters from 0.08 to 2657 $\mu$g/l (pre-TCD) and 26 to 5810 $\mu$g/l (post-TCD). An increase in diatoms was observed between the pre- and post-TCD phase, exhibited by lack of overlap in the interquartile range (Fig. 4a). Interquartile range of the biovolume data (<25 $\mu$m) increased in the post-TCD phase. Biovolume in the 25–505 $\mu$m (Fig. 4b) and >505 $\mu$m size fractions (Fig. 4c) fluctuated less predictably. There were no differences in biovolume data between stations. There was extensive overlap in the interquartile range of chlorophyll *a* concentrations (Fig. 4d) between Shasta and Keswick tailwaters as well as between pre- and post-TCD phases. Chlorophyll *a* concentrations were generally below 1.0 $\mu$g/l in the tailwaters, except during periods of flooding (January 1997), and when top gates of the TCD were open coinciding with peak algal blooms in the reservoir. Range increased in chlorophyll *a* concentrations (Fig. 4c) during the post-TCD phase, similar to what was observed in bacillariophyta data (Fig. 4a,b).

Zooplankton in the tailwaters were composed mainly of copepods and cladocerans. Copepods in the river were dominated by the adult cyclopoid *Leptodiaptomus ashlandi*, immature cyclopoids and copepodids. Cladocerans were dominated by *Daphnia pulex*. Copepod and cladoceran blooms occurred during spring and fall in the reservoir, simultaneous with blooms in phytoplankton. Copepod biomass was similar at both tailwater stations during pre- and post-TCD phases, but did increase during the post-TCD phase (Fig. 5a) as indicated by lack of overlap of the interquartile range between pre- and post-TCD phases. Copepod biomass in Shasta tailwater ranged from 0.08 to 1.36 $\mu$g/l (pre-TCD) and 0.18 to 16.92 $\mu$g/l (post-TCD), and in Keswick tailwater from 0.16 to 1.29 $\mu$g/l (pre-TCD) and 0.00 to 13.34 $\mu$g/l (post-TCD).

Data show an increase in the 75–90th percentile range and an increase in magnitude of outliers for copepod, cladoceran, rotifer and total zooplankton biomass during post-TCD years (Fig. 5a–d), although

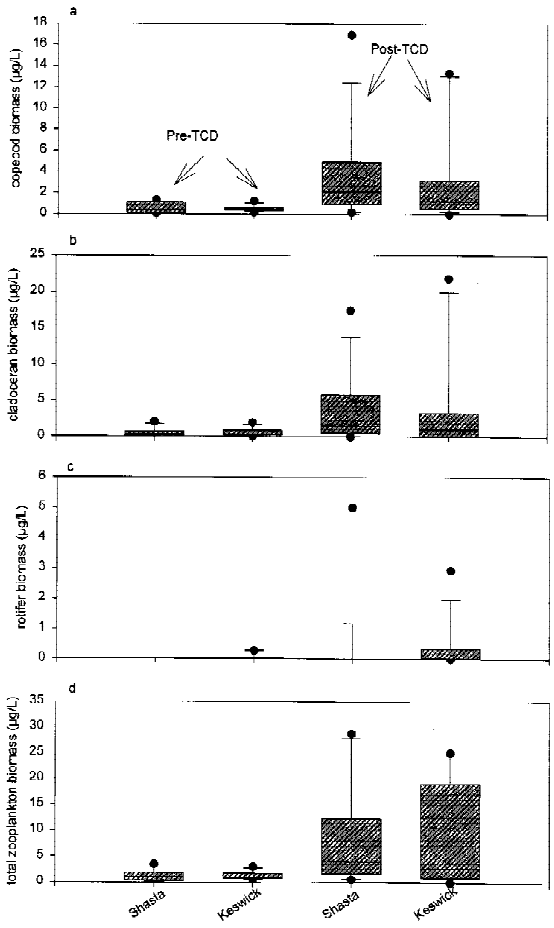


*Figure 5.* Comparison of zooplankton biomass in Shasta and Keswick tailwaters between pre-TCD and post-TCD phases. (a) copepod biomass; (b) cladoceran biomass; (c) rotifer biomass; and (d) total zooplankton biomass. Data used to represent pre-TCD phase included the months from April through September 1995 and January through August 1996 ($n$=14), excluding data collected from late fall to early winter. Data from the post-TCD phase only included the months from February through July 1997 and January 1998 through July 1998 ($n$=13), when surface to mid-level withdrawals (i.e. top-to-mid-gates opened) were discharged from the reservoir. See Figure 2 for explanation of box plots.

differences in biomass of cladocerans and rotifers were not observed (Fig. 5b,c). Rotifers were extremely scarce in Shasta tailwaters during the entire study. An increase in total zooplankton biomass did occur at Shasta but not at Keswick tailwater (Fig. 5d) after the TCD began to operate as illustrated by data in the box and whisker plots.

Seasonal fluctuations of cladocerans, copepods, rotifers, and <25 $\mu$m biovolume of bacillariophyta

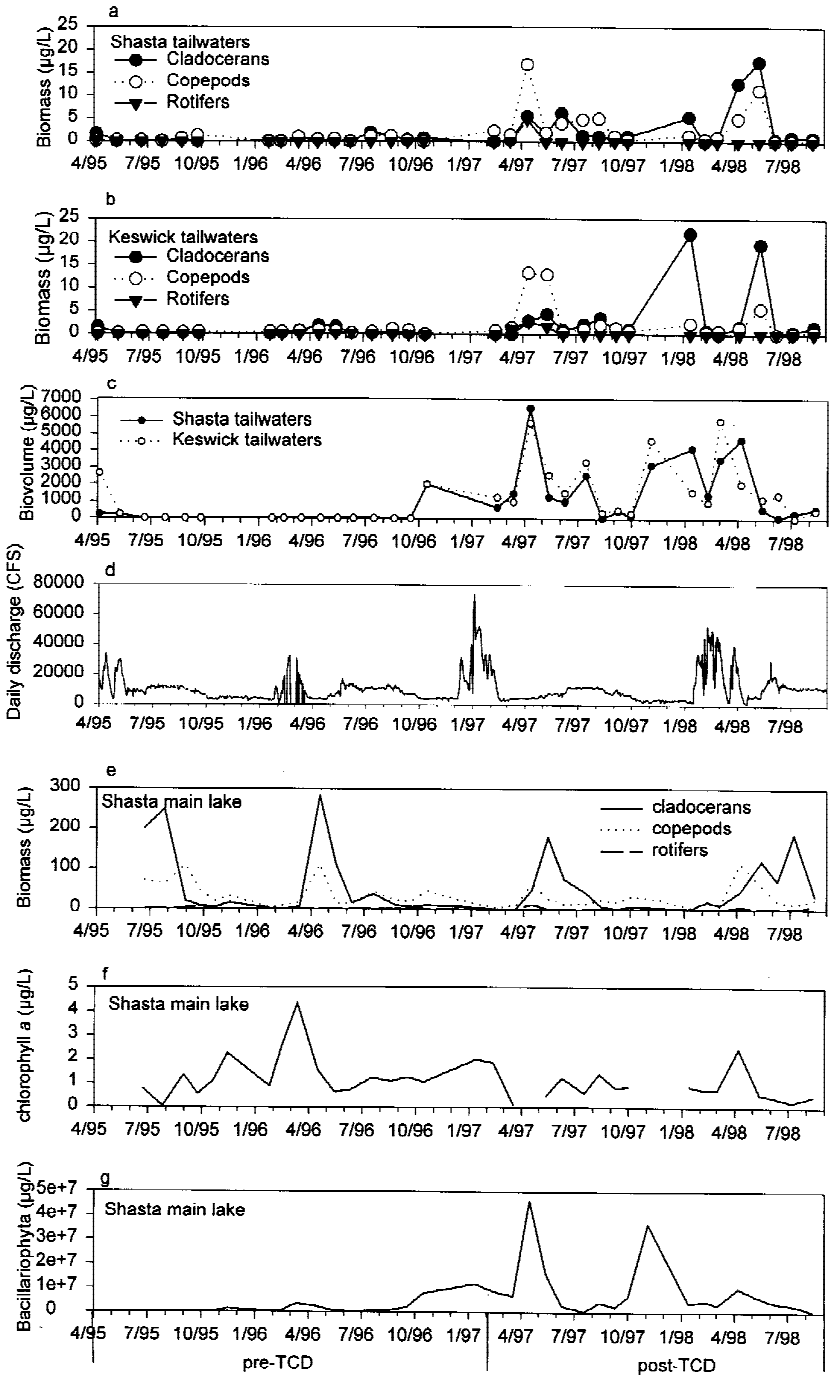


*Figure 6.* Comparison of pre-TCD (April 1995 to February 1997) to post-TCD (April 1995 to September 1998) phases. Seasonal trends of (a) cladocerans, copepods, and rotifers in Shasta tailwaters (b) cladocerans, copepods and rotifers in Keswick tailwaters (c) bacillariophyta (<25 $\mu$m) biovolume in Shasta and Keswick tailwaters (d) daily discharge from Shasta Dam (e) cladocerans, copepods and rotifers (0–30 m depth) in the main part of Shasta Lake (f) chlorophyll *a* concentration (0–30 m depth) in the main part of Shasta Lake (g) bacillariophyta biovolume (0–30 m depth) in the main part of Shasta Lake.

in the tailwaters (Fig. 6a–c) during 1997 and 1998 were more pronounced than during the 2 previous years. Biomass/biovolume of plankton during 1995 and 1996 (pre-TCD phase) was sparse in the tailwaters. Fluctuating discharge from the reservoir (Fig. 6d) was minimal during a typical diel cycle and its effect on plankton in the tailwaters appeared to be negligible except for extreme flood events. Daily discharge from Shasta Dam was regulated on a seasonal basis and only spiked during the rainy period from winter through early spring. Zooplankton and chlorophyll *a* typically peaked in the spring after reservoir turnover, whereas bacillariophyta peaks were sporadic (Fig. 6e–g); increases were only observed during the post-TCD phase in downstream tailwaters.

## Discussion

There are numerous articles in the literature discussing selective withdrawal that include Cassidy's (1989) review of selective withdrawal impacts of tailwaters on physicochemical conditions with some reference to biological communities, and Petts (1984) thorough discussion on epilimnial and hypolimnial releases of phytoplankton and zooplankton from reservoirs. By comparison, there have been few articles that discuss changing reservoir operations to a cyclical yearly pattern of hypolimnetic-epilimnetic-metalimnetic-hypol imnetic discharge. Operating a reservoir for top-to-mid level releases from winter through summer, and then shifting to bottom releases during fall months may cause downstream communities to experience increases or decreases in nutrients, plankton and POM depending on the depth and time of year of withdrawal from the reservoir (Petts, 1984; Kimmel et al., 1990).

Only concentrations of nitrite–nitrate decreased in Shasta and Keswick tailwaters during top-to-mid-level releases with TCD operations. Typically, nutrient concentrations in the hypolimnion were higher than epilimnetic values for all nutrient parameters in the reservoir (Lieberman & Horn, 1998) which meant during the pre-TCD phase deep water withdrawal would tend to deplete the system of nutrients (Soltero et al., 1973) and promote stream production (Neel, 1963). Nitrite–nitrate and SRP concentrations reached seasonal peaks on the surface in January and February in Shasta Lake, during the time of maximal reservoir mixing, and discharge of surface waters with operation of TCD. After this period, surface waters continued to be discharged while the epilimnetic minima developed in April or May due to biological uptake and lack of mixing due to thermal stratification. Once depleted, epilimnetic nutrients were not replenished until fall turnover. The described nutrient cycling of Shasta Lake indicates that levels of nutrients would decrease in tailwaters by opening top-to-mid gates of the TCD, but we found that only nitrite–nitrate concentrations decreased in both tailwater stations and levels of ammonium nitrogen, soluble reactive phosphorus and total phosphorus concentrations were unchanged. By contrast, Martin & Stroud (1973) found that total organic carbon and total phosphorus concentrations were proportionately higher in the tailwater of Nolin Lake, a hypolimnetic discharge reservoir, than in the Barren River Reservoir, which discharged epilimnetic water.

Dance (1981) listed the presence of lakes and impoundments as an important factor controlling the concentration of POM being transported through a river system. The SPOM in Shasta and Keswick tailwaters comprised the greatest proportion of total POM, similar to what is found in many river systems (Maciolek, 1966; Fisher & Likens, 1973; Naiman & Sedell, 1979; Webster et al., 1979; Vannote, 1980; Lieberman & Burke, 1993). We found that the increase in SPOM was localized and occurred only in Shasta tailwaters, probably due to the increased release of bacillariophyta in the less than 25 $\mu$m size fraction, as well as an increase in detrital material. Travel time between upstream (Shasta) to downstream (Keswick) was relatively short depending on flows. During the most productive times of the year, the release of limnoplankton from the reservoir may shift the composition of POM in Shasta tailwaters from detrital to limnoplankton based. Zooplankton was discharged from the reservoir into the tailwaters during April and May when the top gates of the TCD were opened. Concentrations of SPOM in the Upper Sacramento River, were similar from upstream (Shasta) to downstream (Keswick), even though there was greater biovolume of small diatoms ($<$25 $\mu$m) collected during the post-TCD phase. Ward (1975) reported POM concentrations increased with distance downstream from Cheesman Lake on the South Platte River, Colorado.

Reservoirs provide an important source of phytoplankton for downstream reaches, although the smaller lentic species may be selectively eliminated due to size by filtering, sedimentation, and destruction during transport by the river (Hynes, 1970). Few phytoplankters in the pre-TCD phase were exported from the hypolimnetic zone of Shasta Lake and, therefore,

biovolume remained sparse in the river throughout the year. Peak phytoplankton productivity in the tailwaters coincided with springtime diatom blooms in the reservoir (Lieberman & Horn, 1998), only after the TCD began operation. Most of the phytoplankton in the reservoir tended to be distributed in the upper 20 m of the water column and were not disturbed until upper gates of the TCD began to discharge bacillariophyta (<25 $\mu$m) downstream in significant quantities. The less than 25 $\mu$m size fraction composed the greatest proportion of total chlorophyll *a* but did not mirror the increase in the less than 25 $\mu$m diatom biovolume. Possibly the primary pigment of the less than 25 $\mu$m diatom may have been chlorophyll *b* or *c*, and not chlorophyll *a*. Analyses for these other pigments were not performed. Only a small proportion of the total phytoplankton in the reservoir was composed of larger phytoplankton and therefore an increase was not observed in the tailwaters. The majority of phytoplankton production in the upper sections of the Sacramento River originated from reservoir contributions only after the TCD began operation and epilimnetic waters were discharged downstream. With present TCD operations, the river may be significantly influenced by biological drift from Shasta Lake. SPOM composition may begin to shift from detrital to plankton based due to increased biovolume.

Releases of epilimnetic surface waters during spring blooms in Shasta Lake resulted in discharging of greater copepod biomass and overall total zooplankton biomass, as well as greater variability in all zooplankton data, including cladoceran and rotifer biomass. In addition, outliers in the data (post-TCD phase) of copepods, cladocerans and rotifers represented the biomass outfall coinciding with operation of the TCD, and the tremendous concentration of zooplankton in the upper 30 m of water. Ward (1975) found greater numbers of survival for rotifers and small-bodied cladocerans in a lotic system. Collections further downstream from Keswick tailwaters are deemed necessary to get a better indication of the fate of organic drift in the Upper Sacramento River. Range in data during the pre-TCD phase was small as a result of withdrawals that typically occurred below the euphotic layer (upper 30 m) of the reservoir. The shallow bypass intakes and spillway could possibly discharge water from an upper depth of about 20 m or less during high flows which would allow some zooplankton released downstream, though this was not a common occurrence and, there was sparse zooplankton in the river before the TCD began to operate. Ward (1975) observed that dams which release water from the hypolimnion do not provide zooplankton to downstream reaches in sufficient quantities to supply a reliable food source for riverine biota. With TCD operation at Shasta Lake, it appears that epilimnetic releases may increase some of the zooplankton biomass (i.e. copepods) in the river during spring months without detrimentally affecting zooplankton biomass in the reservoir. Investigators reported at Hungry Horse Dam, Montana (Kubitschek, 1994; Marotz et al., 1996, Christenson et al., 1996) that reservoir loss of zooplankton was predicted as warm epilimnetic waters were released through a multi-level outlet. As a result, an engineering modification to the structure has allowed simultaneous release of water from two elevations to reduce zooplankton entrainment because it was deemed detrimental to reservoir fishery. Fall-out of zooplankton appeared to be minimal from upstream to downstream (15 km between stations), whereas other investigators have reported rapid decrease in zooplankton biomass in reservoir-river systems caused by the inability to withstand turbulence in a riverine environment and greater selectivity of predators toward larger zooplankton (Ward, 1975; Novotny & Hoyt, 1982; Soballe & Bachmann, 1984).

A TCD on Shasta Lake was built to provide cooler water temperatures to the Upper Sacramento River for protection of chinook salmon. We observed decreased nitrite-nitrate concentrations, localized increases in SPOM at Shasta tailwaters, increased bacillariophyta (<25 $\mu$m), and increased copepod biomass, with no apparent differences in data from upstream (Shasta tailwaters) to downstream (Keswick tailwaters) stations. Most of the changes were due to TCD operation and releases of productive epilimnetic waters from Shasta Lake. The operation of the TCD may change food web dynamics at downstream sites. These changes may benefit declining salmon populations by increasing food availability for fry. It should be realized, however, that the impacts of food web changes are difficult to predict. It is possible that increased zooplankton in the river could increase numbers of some organisms that could negatively impact salmon populations. We have presently extended our studies downstream below Keswick tailwaters, to determine how productivity changes from upstream to downstream in the Upper Sacramento River.

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