Pre-and Post-Operational Effects of a Temperature Control Device on Physical, Chemical, and Biological Attributes of Shasta Lake, California: Phase1, Spring 1995 through Fall 1997

Open File Report 98-251

U.S. Department of the Interior
U.S. Geological Survey

USGS
science for a changing world

Prepared in Cooperation with
the U.S. Bureau of Reclamation

U.S. DEPARTMENT OF THE INTERIOR

U.S. GEOLOGICAL SURVEY

Pre- and Post-Operational Effects of a Temperature Control Device on Physical, Chemical, and Biological Attributes of Shasta Lake, California: Phase 1, Spring 1995 through Fall 1997

by
[1]Davine M. Lieberman, U.S. Geological Survey
[2]Michael J. Horn, U.S. Bureau of Reclamation

Open-File Report 98-251

Prepared in cooperation with the U.S. Bureau of Reclamation

---

[1]P.O. Box 25007 (D-8220), Denver, Colorado 80225

[2]P.O. Box 25007 (D-8220), Denver, Colorado 80225

Abstract:

A temperature control device (TCD) was installed on Shasta Dam, California which began to operate in spring 1997 for the purpose of cooling downstream river temperatures to aid recovery of chinook salmon in the Sacramento River. This study began in spring 1995 to investigate pre- and post-TCD effects on the limnology of Shasta reservoir. The maximum pool of cold water observed was a function of runoff, as bypass operations resulted in an almost complete yearly depletion of the cold water pool. Maximal surface temperatures and the strongest thermocline occurred during the month of August. Winter mixing was confined to the upper 50 m of the water column. Metalimnetic and hypolimnetic oxygen minima occur yearly in summer. Degree of oxygen depletion was related to distance from river inflow, while rate of movement of hypolimnetic minima downstream was dependent on deep level withdrawals from the dam. Preliminary observations show the TCD affected metalimnetic temperature gradients, movement of riverine inflows through the reservoir, and strength of DO minima.

Secchi transparency was highest in the main lake and the Sacramento Arm and, lowest in the McCloud and Pit Arm, and was indicative of differences in productivity between the arms. Patterns of nutrient cycling within the reservoir differed between epilimnetic and hypolimnetic waters. Hypolimnetic nutrient maxima were related to high flow events, while epilimnetic waters responded more to in reservoir events, such as stratification and changes in productivity. During spring and summer months, TCD releases of surface to mid-level water caused the hypolimnion to act as a nutrient sink. Composite chl *a* concentrations (0 to 5 m) ranged from BDL to 30 µg/L. Mean composite chl *a* concentrations for the period 1995 to 1997 were: main lake (2 µg/L), Sacramento River Arm (2 µg/L), McCloud River Arm (2.7 µg/L), Pit River Arm (5.2 µg/L), and the Pit-McCloud confluence (3 µg/L). Diatoms, *Melosira islandica* and *Melosira varians* dominated the algal population. Greatest algal biovolume occurred in the Pit River Arm. Biovolume increased significantly from 1995 to 1997. Seasonal algal peaks occured in spring, and again on early fall coinciding with development and breakdown of thermal stratification. Greatest mean zooplankton biomass occurred in the Sacramento River Arm and was lowest in the McCloud River Arm. Cladocerans were the dominant zooplankton at the main lake station and Sacramento River Arm. Cladoceran and copepod biomass was equally distributed at other stations.

In Shasta and Keswick tailwaters, <25 µm POM composed the greatest proportion of total POM contributing 90% to the total. Total POM averaged 0.8 g/m^3 and 0.9 g/m^3 in Shasta and Keswick tailwaters, respectively. In 1997, there was a significant increase in zooplankton biomass in the drift that was attributed to TCD operations. Copepods, cladocerans, and even rotifers increased. Cladocerans composed less and copepods made up more of the total zooplankton biomass. Releases of surface water during the most productive time of year resulted in entrainment of zooplankton and a significant downstream increase in biomass. As with zooplankton biomass, phytoplankton biovolume increased significantly (by a factor of 10) in 1997 over 1995 and 1996.

Table of Contents:

List of Figures:

Page

List of Tables

Page

I. Introduction:

Limnological research on Shasta Lake began in March 1995 as a joint effort between USGS/ Midcontinent Ecological Science Center (MESC), Bureau of Reclamation (BOR)/ Northern California Area Office (NCAO), and BOR/Technical Service Center (TSC) to investigate the potential changes in the biological, chemical, and physical characterisitics of the reservoir as a result of operating the temperature control device (TCD).

Shasta Lake is located 12 miles north of Redding in northern California, and is part of the Central Valley Project (CVP), a federal water project operated by the Bureau of Reclamation. Shasta Dam, a 602-foot high curved concrete gravity structure with a crest elevation of 1077 feet above mean sea level (MSL) was completed in 1945 forming the largest reservoir in California. The 35 mile long reservoir has 365 miles of shoreline, a surface area of 29,500 acres, a maximum depth of 157 m, and contains 4.5 million acre-feet of water at full pool. Limnologically the reservoir is characterized as monomictic, however, it does not turn over completely, and remains ice-free in the winter. The reservoir receives the majority of its water during the rainy season from the end of December through March each year (Fig. 1). The reservoir supports a two-level fishery with salmon and trout providing a cool water fishery, and a variety of Centrarchids, notably spotted bass, providing the warm water portion.

A temperature control device (TCD) was installed on Shasta Dam and began to operate on March 13, 1997 for the purposes of controlling downstream river temperatures to aid recovery of the endangered winter run chinook salmon, and to minimize loss of generating capacity as a result of releasing deeper, colder water through low level outlet works in order to meet downstream temperature criteria. Prior to operation of the TCD, late summer water temperatures in the Sacramento River were too warm for successful spawning of salmon. Proper operation of the TCD allows surface waters to be released in the spring and early summer, conserving the pool of cold water. As the season progresses withdrawals move deeper into the hypolimnion, and if need be, deeper than the old penstock intakes. Historically, Shasta was operated as a hypolimnetic deep release reservoir. In 1987, however, bypass releases were instituted as a conservation measure for chinook salmon. With the exception of surface withdrawl capabilities, TCD operations are meant to mimic bypass releases.

The current limnological study began in spring 1995, two years prior to operation of the TCD. The objectives of the study are to: 1) Compare pre- and post-operational changes on the physical, chemical, and biological attributes of Shasta Lake and tailwaters; 2) help develop better operational guidelines for the TCD in order to minimize negative impacts to Shasta Lake while providing optimum water temperatures downstream for chinook salmon spawning; and 3) apply limnological, fisheries, and modeling results to existing and planned TCD facilities.

We will test several hypotheses that address potential effects from operation of the TCD on the limnology of Shasta Lake. These include 1) increased winter mixing due to lower temperature gradient as a result of depleting the deep cold water pool in late fall; 2) delay of thermal stratification due to surface withdrawals during spring months; 3) increased degree of thermal

stratification due to shifts in withdrawal patterns; 4) lower summer primary and secondary productivity due to increased stratification; 5) increased hypolimnetic nutrient levels during spring and summer as a result of a shift to surface withdrawals; 6) potential decreased zooplankton biomass and increased algal production due to spring surface withdrawal; and 7) shifts in quantity and/or quality of downstream organic drift, potentially influencing riverine fish production.

II. Methods and Materials:

1. Reservoir Sampling:

Beginning April 1995, the reservoir was sampled monthly except during the months of October-December 1995, December 1996 and December 1997. Five stations were established at the beginning of the study: main lake (S6), Sacramento River Arm (S7), McCloud River Arm (S8), Pit River Arm (S9), and Pit-McCloud Confluence (S10). The sampling program was expanded in February 1996 to include 6 additional stations: Squaw Creek (S11), upper Pit (S12), upper upper Pit (S13), upper McCloud (S14), upper Sacramento (S15), and forebay (S16) (Fig. 2, Table 1). Stations S11 to S16 were only sampled for water temperature (^{0}C), conductivity (µS/cm), dissolved oxygen (mg/L), pH, turbidity (NTU), surface and composite chl *a* (µg/L), and secchi disk transparency (m). Stations were sampled at the deepest point on the thalweg of the historic river channel.

Water column profiles for water temperature (^{0}C), dissolved oxygen (mg/L), specific conductance (µS/cm), pH, and turbidity (NTU) were completed at each station using a Hydrolab Surveyor 3® connected to an H20® sonde unit. Data was collected at one meter intervals through the thermocline then every two to five meters thereafter until the bottom was reached. At each depth readings were allowed to stabilize before data was recorded. The sonde unit was calibrated according to Hydrolab specifications at the beginning of each sampling trip.

Nutrient samples (500 mL) were collected from surface, middle, and bottom depths through the summer of 1996 for total phosphorus (TP) (detection limit 0.002 mg/L), soluble reactive phosphorus (SRP) (detection limit 0.001 mg/L), dissolved nitrate and nitrite nitrogen (NO_3 + NO_2) (detection limit 0.002 mg/L), and ammonia (NH_4) (detection limit 0.003 mg/L). Middle depths were initially selected based on conductance (an identifier of riverine interflow). Surface and bottom samples were 0m and 85 m (if station depth allowed). Nutrient sampling protocols were changed in the summer of 1997 to reflect a better representation of the whole water column. From summer 1997 to present 0 m and 85 m collections remained constant, however, one mid-depth sample was added and the collection depths fixed at 10 m and 20 m. Total kjeldahl nitrogen (TKN) (detection limit 0.035 mg/L) and total suspended solids (TSS) (detection limit <1 mg/L) were collected only during the first few months of the study. Water samples were kept refrigerated until analyzed. Nutrient analyses were performed by the University of California, Davis.

Secchi transparency was measured using a standard 20 cm secchi disk and View Scope®. The

secchi disk was lowered in the water column until it disappeared, then brought up until visible, and lowered again until it disappeared. Secchi disk transparency was recorded at the point of disappearance.

Replicate zooplankton tows were collected from three depth intervals: 0-10 m, 10-20 m, and 20-30 m with a 64 μm birge-style closing net. Samples were stored in 125 mL amber bottles, and preserved with Acid Lugols solution. Samples were identified to species and enumerated by Dr. John Beaver, BSA, Beachwood Ohio. Biomass was calculated based on enumerated samples and individual species bio-volume estimates provided by Dr. Michael Brett (UC Davis, U. Washington).

Phytoplankton samples were collected from discrete depth intervals using a electric water pump and composited for analysis. Samples were collected from 0, 2, 5, 8, 10, 12, 15, 18, 20, 22, 25, 28, 30 m. At each depth, duplicate 1-gallon containers were filled and composited into 5 gallon buckets. Composited samples were 0-10 m, 10-20 m, and 20-30 m. 250ml of composited sample from each bucket was preserved in Lugols and later identified to species and enumerated and converted to biovolume. Dr. John Beaver, BSA, Beachwood, Ohio identified and enumerated samples and individual species bio-volume estimates were provided by Dr. Michael Brett (UC Davis, U. Washington).

Duplicate water samples for chlorophyll *a* (chl *a*) pumped from discrete depths were collected in a similar manner as phytoplankton except depth intervals used were 0, 5, 10, 15, 20, 25, and 30 m. Duplicate composite samples were additionally collected from 0-5 m using a pool hose lowered into the water column to a depth of 5 m. For analysis from 150 to 250 ml of sample was filtered onto Whatman GF/C filters (47 mm). Filters were kept frozen until analyzed. Samples were extracted with methanol and analyzed flourometrically by the UC Davis. Chl *a* concentrations were corrected for phaeophytin.

2. River Sampling:

Two river sites on the Sacramento River, one below Keswick Dam and the other below Shasta Dam, were sampled on a monthly basis (Fig. 2). The Keswick station was located approximately 14 km downstream from Shasta and about 0.8 km downstream of Keswick Dam. The Shasta site is about 0.8 km downstream of Shasta Dam.

Particulate organic matter (POM) samples were collected and fractionated into three sizes (>505 μm, >25 μm, and <25μm) using a series of plankton nets; all samples were collected in triplicate. >505 μm POM samples were collected using a 505 μm net 3 m in length with a 0.5 m diameter mouth. The 505 μm net was deployed in the current 3-4 minutes. A calibrated flow meter mounted across the mouth of the net was used to determine volume of water filtered. During extreme high flow events >505 μm samples were not collected. Three liters were collected for each <25 μm POM replicate by filtering water pumped from the river through a 25 μm plankton net into a 5 gallon bucket and collecting the filtrate. The >25 μm size fraction was obtained by

filtering 96 L through a 25 μm plankton net. All samples were stored chilled until processed. Samples were filtered through ashed, pre-weighed Gelman 47 mm glass fiber A/E filters. Filters were oven-dried at 75 °C for 24 hrs, weighed for dry weight, ashed at 500 °C for one hour, and weighed for ash weight. Dry weight (seston), ash weight, and ash-free dry weight (POM) were calculated as g/m^3 according to Strickland and Parsons (1968).

Four chlorophyll samples were collected from each river station. Two samples were collected as grab samples from the river surface, and two collected from the portion of water representing the <25 μm size fraction (as described above). Duplicate water samples for phytoplankton were collected and separated into >505 μm (500 mL), >25 μm (125 mL), and <25 μm (1 L) size fractions, according to the above netting protocols for POM. Duplicate zooplankton samples were collected from the >25 μm size fraction as described for POM above. Nutrients (NO_3 + NO_2, NH_4, TKN, TP, SRP) were collected at each station 0.5m below the surface. Chlorophyll, plankton and nutrient samples were processed following the same guidelines as those outlined for reservoir samples. Water temperature, pH, specific conductance, dissolved oxygen, and turbidity were taken 0.5 m below the water surface.

III. RESULTS AND DISCUSSION

1. Physical Limnology:

A. General Trends:

Physical data indicated Shasta Lake is monomictic, however, the reservoir did not turn over completely during this study and mixing was confined to the upper 50m of the water column. Thermal stratification extended from late April through the end of September (Fig. 3a-e). Duration of stratification was driven by ambient temperatures and amount of solar input, thus the length of time the reservoir stratified for was variable from year to year. Bypass operations probably had no significant impact on the timing of, or extent of, thermal stratification of the epilimnion. However, we do not have data to indicate whether bypass operations of the last 10 years differ in their effect from that of historical reservoir operations. Once stratification occurred typical reservoir and bypass operations generally withdrew water from well below the thermocline, and as such, should have had no real impact on epilimnetic heat budgets during summer months. Bypass operations did, however, as will be shown, affect thermal structure and water movement patterns, such as interflows, of hypolimnetic waters. Decrease in hypolimnetic cold water pools over the course of the summer was a direct consequence of bypass operations. Replenishment of the cold water pool in winter and spring was a function of runoff rate and temperature of inflow.

Figures (4a-e) indicate relative thermal resistance to mixing (RTRM) for each station by date, and provide a pictorial representation of stratification patterns that occurred within the reservoir. Graphs do not show true values for the large peaks due to the smoothing function of the graphical technique, but never the less indicated time periods and depths of greatest relative resistance to

mixing. RTRM was calculated as the difference in density of water due to temperature at two different depths standardized to the difference in density of water between 4 and 5^{0}C. Salinity differences were not incorporated into the resistance coefficients due to minimal contributions to density based on observed water column specific conductance values. The larger the RTRM coefficent (such as at the thermocline), the greater the density (temperature) gradient and thus a correspondingly larger amount of energy that is needed to mix water across the gradient.

Stratification developed in April, and the thermocline established at a depth range of 10-15 m (Figs. 3a-e, 4a-e). Maximal surface temperatures and the strongest thermocline occurred in August of the three years of this study (Figs. 5g,q, 6g,q). Maximal surface temperatures were 27-28 ^{0}C. Productivity decreases and nutrient declines during the summer were associated with the development of the thermocline, and increased RTRM. Fall disappearance of a strong thermocline was fairly rapid and by September lake surface temperatures had dropped 4-5 ^{0}C from the previous month (Fig. 4a-e). With the drop in temperatures there was gradual erosion of the thermocline as it was pushed deeper into the metalimnion. The time period of a weakening thermocline and greater potential mixing coincided with fall algal blooms each year of this study.

The metalimnetic zone in the reservoir represented a transition zone beginning at the thermocline (1 ^{0}C/m) and extending to the hypolimnion. This zone showed fairly consistent temperature change (<1 ^{0}C/m), though not to the degree of the thermocline. Early in the season the metalimnion was narrow, such as during April to May when the cold water pool in the reservoir was near maximum. As the reservoir warmed through surface heating, increases in inflow water temperature, and as hypolimnetic releases began removing the cool water pool, the metalimnetic zone thickened. RTRM indicated a fairly shallow metalimnion in April which deepened over the course of the summer to a maximal depth of about 50 m toward the end of September; this pattern was consistent for both 1995 and 1996. Below the metalimnion RTRM coefficients remain low and relatively constant, an indication of the hypolimnetic zone where temperatures changed little.

Metalimnnetic oxygen minima developed seasonally within the reservoir, tending to be most severe upstream with almost complete anoxia developing at some sites (Figs. 3a-e, 5g,q, 6g,q). Minima began developing in July and become strongest during peak stratification in August. Metalimnetic minima developed at the thermocline, and immediately above and extending into the high conductivity water of the interflow region at all reservoir stations. Minima develop from several causes. As inflow waters and their associated organic load move as interflow through the reservoir, respiration rates due to bacterial decomposition result in oxygen depletion. Upstream-downstream gradients occurred due to decreasing amounts of organic matter available for decomposition the further into the lake the interflow extends from the source. Secondly, as dead algal material settles out of the epilmnion settling rates slow as material encounter the denser water associated with the thermocline allowing more time for decompositional activity and consequently lowered oxygen levels. Plankton respiration in this zone may further be an important contributor to oxygen depletion. Sediment oxygen demand, at least early in the year is likely not an important contributor to the metalimnetic minima; it is, however, probably important in

development of hypolimnetic minima at upstream sites. All factors act together to some degree in the reservoir.

The upper portion of the metalimnion was important to reservoir productivity during summer months. Chl *a* peaked at the thermocline and measurable levels extended into the upper metalimnion. During summer, surface waters were depleted of nutrients quickly and productivity in near surface waters was low (Fig. 7). The 1% light level extended down to around 20-25 m during the summer, which represents the lower limit of active algal photosysnthesis. There was thus a 10-15 m band in the upper metalimnion suitable for algal growth. At least early in the development of the metalimnion each year, this zone contained higher nutrient levels than surface waters and there were correspondingly higher chl *a* concentrations (Fig. 7). As summer progressed nutrient levels decreased due to an ever thickening metalimnion with the temperature gradient acting to limit the degree of continued mixing.

Below the metalimnion, hypolimnetic temperatures remained relatively stable to the bottom of the reservoir. There was a slow decrease in temperature with depth (Fig. 3a-e), however, temperature changes were subtle enough that RTRM coefficients were relatively constant and small throughout this region (Fig. 4a-e). RTRM plots were used to determine the break between the metalimnion and hypolimnion, as even in the hypolimnion temperature change still occurred with depth. Hypolimnetic oxygen depletion developed over the course of the summer, significant declines began appearing in July with the severest depletion occurring during the months of September-November (Figs. 5h,i,r,s,t, 6h,i,r,s,t). Oxygen depletion was severest near the bottom sediments with levels gradually recovering towards the surface, and was a function of bacterial decomposition and associated sediment oxygen demand. Similar to metalimnetic responses, severest depletion occurred at upstream sites where organic load was highest.

Hypolimnion volume was near maximal in late winter to early spring, coincident with the end of the rainy season. Deep hypolimnetic temperatures varied slightly from year to year, though the range for the past four years is less than one degree, ranging from a low of 6.7 ^{0}C to a high of 7.5 ^{0}C. Minimal hypolimnetic temperatures were a function of inflow temperature (Fig. 8). During the three seasons of study to date, the hypolimnion has not mixed completely with the rest of the water column during winter mixing. Thorough mixing occurred to between 50 and 60 m consistently, though there was some exchange with deeper layers.

Mixing may have been influenced by operational patterns. Typical early winter releases were through the penstocks (el. 815), and given reservoir elevations for the same time period, what may be occurring is penstocks were not tapping the upper levels of the reservoir, while cold inflows in the range of 5-8 ^{0}C are replenishing the bottom waters up to the level of the penstocks. Cold inflows occurred from late December through March. As an example there was a fairly rapid break at about 45 m in February 1996 where water temperature drops from 10 ^{0}C down to about 8 ^{0}C. A depth of 45 m corresponded to a reservoir elevation of about 885 ft. This upper limit may represent the upper zone of influence of penstock releases (Tracy Vermeyen, USBR, Denver, Co. Pers. Comm.). During the three years of this study the level was fairly stable.

Without complete mixing the upper portion of the reservoir was unaffected by typical release patterns. During high flow events where upper bypass tubes were operated more of the surface layer was released. As such, the lowest temperatures observed for surface waters were influenced more by ambient weather conditions than inflow waters. Support for limited mixing was further provided by water column turbidity profiles conducted during winter and early spring 1997 (Fig. 9). A large flow event (Fig. 1) provided a turbid inflow into the reservoir that was monitored for several months. Initial movement was rapid, and within days inflow had reached the dam. One month later this plume had mixed fairly well. Surface turbidity indicated some mixing, but the majority of high turbidity water remained deep, with a well defined break at about 40 m (Fig. 9).

Bypass operations, with release target temperature near 9 ^{0}C in late summer, resulted in a gradual depletion of the hypolimnetic cold water pool over the course of late summer and fall. The maximum pool of cold water observed for the past three years was a function of runoff. Average inflow temperatures were below the 9 ^{0}C level for 3-4 months depending on the year. For example, in 1995-96 there was about a 110 day period of average 9 ^{0}C or lower, with inflow for the period being approx 2.8 million acre feet, during this time approximately 1 million acre feet was discharged, leaving an excess of approximately 1.8 million acre feet of 9 ^{0}C or cooler water in the reservoir.

B. Interflows:

The Sacramento and Pit Rivers provided the majority of inflow, with the Pit providing close to two-thirds (69% vs. 18%) of the total. Sacramento River water had lower specific conductance than water entering from the Pit river arm due to characteristics of the drainage basin. This provided the ability to distinguish the fate of the two rivers as they travel through the reservoir. Movement of riverine water through the reservoir was a function of both release depth, volume of releases at the dam, amount of inflow entering the reservoir and inflow density, primarily as a function of temperature. Pre-TCD upstream downstream patterns of water movement through the reservoir were similar from year to year. Variation existed in exact timing of events due to seasonal differences in flow rate and temperature of inflows. When warm temperatures demanded higher bypass releases, flow changes occurred at a faster rate in the reservoir because more water was being moved. As examples of water movement patterns, this report will focus on 1996 data as representative of typical bypass conditions for the Sacramento and Pit river arms.

Run-off temperatures were at or near their lowest in January -February (Fig. 8). For both rivers inflows were typically in the 7-9 ^{0}C range; similar to deep hypolimnetic temperatures. Surface temperature never dropped below 9-10 ^{0}C, thus river inflow always entered as interflow or underflow. A plunge point existed near the upstream end of each arm above which all water was river water and downstream of which was an isolated pool of lake water. During the spring as runoff increased, a shift in thermal structure could be seen in upstream sites, where cooler water was shallower than further downstream (Figs. 5a, j, 6a, j). This represented the wedge shaped mass of water sliding under the plunge zone and flowing down the narrow channel. As it moved downstream and the channel widened water dispersed to an equilibrium depth. Pit River inflow at

this time of the year moves along the bottom of the reservoir, and essentially functioned to replenish the cold water pool of the hypolimnion and pushed the warmer layers of water upward (Fig. 5b). The Sacramento river followed a similar pattern though tending to ride above the inflow from the Pit arm (Fig. 6b). Whereas the Pit river followed the bottom, the Sacramento river intrusion was most prominent in the 50-70 m depth range.

From March into April the interflows from both rivers elevated into the water column and by May stabilized at 10-20 m in depth, riding just under the thermocline, in the upper metalimnion. The Sacramento river plume was less visible in summer due to low inflows and conductivity more similar to average reservoir conditions. Effects of shallow interflows extended into the forebay of Shasta Lake, though not as prominently as at upstream sites (Figs. 5d,h, 6d,h). Inflows dispersed as a narrow band in the lake. This was attributed to the fact that releases were below this level and the plume is not pulled through the lake as when intrusions are near penstock depths (Figs. 5,6). As this intrusion band thickened through August it remained prominent but dispersal was evidenced by conductivity gradients of upper metalimnetic waters (Figs. 5,6). Beginning in September, as inflow temperatures declined at a rate more rapid than reservoir temperatures, interflows began settling in the water column (Figs. 5,6).

An upstream downstream gradient of oxygen depletion in the hypolimnion developed soon after the plume elevated off the bottom for both river arms. This bottom layer was isolated from further input until the plume descended in the fall. Since it was below the photic zone and there was no freshwater input, dissolved oxygen was not replenished and decompositional processes gradually utilized remaining dissolved oxygen. As typical with most reservoir systems, depression of dissolved oxygen levels was first apparent at upstream sites, gradually extending down reservoir. There was a one-month time lag between the Pit arm (S9) and the main lake (S6) in the appearance of severest DO depletion. Slow downstream movement in the summer, was a result of deep bypass withdrawals moving water through the reservoir. At upstream sites low DO water was associated with bottom sediments, however, further downstream the low DO center elevates into the water column and about mid-October can be observed in the main lake (S6) at about 100 m during 1995 and 1996, and at about 80 m during 1997, which approximate low-level bypass elevations, and indicate a possible cause-effect relationship.

2. Secchi Transparency:

Secchi transparency is a function of light absorption characteristics of water due to dissolved and particulate matter, either organic such as plankton, or inorganic such as suspended sediments. It reflects both the seasonal thermal structure and plankton productivity in the reservoir. Secchi transparency was highest in the main lake (Fig. 10a) and Sacramento Arm (Fig. 10b) and, lowest in the McCloud (Fig. 11a) and Pit Arm (Fig. 11b), and was indicative of differences in productivity between the arms. Secchi transparencies, reached a maximum in May, and again in October or November, and coincided with time of minimal algal productivity. Greater secchi depths in early summer were a result of decreased mixing, on-set of thermal stratification, subsequent depletion of nutrients and consequently a decrease in algal productivity. Secchi depth

transparency was lowest during the winter-spring rainy season, during spring algal blooms, and during late summer-early fall drawdown of the reservoir. During these times suspended sediment as a result of rains and/or reservoir drawdown and algal material caused a decrease in water clarity.

Secchi depth transparency trends were similar for 1995 and 1996, while 1997 data show some differences that reflected hydrology unique to 1997. Secchi transparencies for all stations were below 0.5 m from the beginning of January through March 1997 (Figs. 10-11). Torrential rains and severe flooding in northern California produced extremely turbid conditions and chocolate brown waters throughout the reservoir in 1997. The reservoir began clearing April and shortly after significant algal blooms occurred. Secchi transparency increased to about 2 m at all stations, lower readings were recorded in the upper reaches of river arms primarily due to greater concentration of suspended sediment. Following the April algal bloom, water clarity increased as a result of algal die-off from nutrient depletion and settling of sediments. In early summer 1997 secchi depths were greater than the two previous years during the same period (Figs. 10-11). Secchi transparency remained high until August, then decreased. Water were more turbid in late summer-early fall 1997 compared to 1995 and 1996 due to resuspension of particulate material in the water column as the reservoir was drawn down to its lowest level in three years. Draw down had a pronounced effect on water clarity, but at the same time may have brought in additional nutrients through resuspension which contributed to algal blooms throughout the reservoir in August 1997, particularly in the Pit river arm (S9).

3. Nutrient Dynamics:

Epilimnetic and hypolimnetic nutrient pools within the reservoir showed distinct spatial and seasonal patterns. There were, however, differences in cycling unique to each pool of water. Epilimnetic peaks were dependent on winter mixing whereas flow events and runoff played a more important role in the hypolimnion. Variation between years was likely driven by large scale climate responses that resulted in higher or lower inflows, shifts in outflow release patterns and variation in degree and duration of thermal stratification and resultant reservoir mixing. Spatial separation of nutrient levels within reservoir arms was related to distance from inflow and may be affected by contributions from exposed sediments due to changing reservoir levels, and local events such as algal blooms. All stations showed similar long-term patterns though there were substantial short term differences. Extremes for nutrient levels ranged from below detection limits (BDL) -160 μg/L for nitrite-nitrate, BDL to slightly over 30 μg/L for ammonia, BDL-60 μg/L for orthophosphate and BDL-100 μg/L for total phosphorus. Hypolimnetic values were typically higher than epilimnetic values for all nutrient parameters.

Nitrate and SRP (orthophosphate) levels reached epilimnetic seasonal peaks at all stations in January-February, during the time of maximal reservoir mixing (Figs. 12-13). Patterns of increase were similar between the two, however, SRP increased in surface waters prior to measurable increases in nitrate. For both nutrients increases began in October-November. Increases were a combination of both biological and physical dynamics. As the reservoir cooled and stratification

began to break down both nutrients were cycled into the surface waters, however, the reservoir tended to be more nitrogen limited, and nitrogen may be sequestered more rapidly than phosphorus during fall algal blooms. As such there was an apparent fall surplus of phosphorus. Some of the increase in phosphorus may be attributed to reservoir levels that were continually dropping during this time. Reservoir turbidity tended to increase in late summer due to wave action introducing fine sediments from a formerly inundated shoreline. As fall progressed, continued cooling, shortened day-length and greater mixing rates finally suppressed these blooms and both nutrient levels then increased.

Although runoff associated with winter storms began in mid-winter, winter nutrient peaks did not appear correlated with runoff events (Figs. 1,12,13). Peaks occurred at the time of greatest potential mixing, as indicate by reservoir temperatures. Chl *a* levels increased significantly beginning in February and were fairly uniform through at least the top 30 m of the water column, an indication of the mixing going on (Fig. 7). Although February reservoir temperatures were similar or even cooler then in January, lengthened photoperiod probably triggered the start of algal growth. As spring progressed nutrient levels decreased as algal production increased. Although nutrient levels appeared similar in different reaches of the reservoir, it may be somewhat misleading because higher algal abundance in the upper arms probably sequestered any new nutrients made available, or resulted in faster turnover times of available nutrients.

With few exceptions nutrient levels decreased and remained low through the summer due to biological uptake. Epilimnetic minima typically occur around April to May, though the minima for nitrogen occurs sooner in more productive areas such as at S9 and S10 (Figs. 12-13). Strong stratification within the reservoir prevented mixing, and once depleted, nutrients were not replenished until fall turnover. Algal production still occurred, but tended to be deeper and associated with the thermocline and upper metalimnion where their was increased availability of metalimnetic nutrients.

In surface waters nitrate-nitrogen was the dominant nitrogen form when nutrient levels were high, as during spring, and winter mixing. In late spring to early summer there were periods where epilimnetic ammonia spikes occurred. Spikes occurred at all stations, but were higher at upstream sites (Fig. 14). Occurrence times of spikes was not similar among stations. Chlorophyll levels were usually low when they occurred and ammonia spikes may represent an algal crash where there is a short peak due to decomposition. Other influences such as weather and periodic mixing may have also caused short-term peaks in ammonia

Hypolimnetic nutrient levels were on average greater than epilimnetic levels, and appeared to be driven more by hydrodynamic events. Though there was substantial variation within and between sites there were several emergent patterns. The beginning of winter runoff brings in nutrient laden water and a lot of detrital material, notably woody debris. Sudden changes in nutrient levels corresponded to changes in the hydrograph, and are apparent at all stations almost simultaneously (Figs. 12,13,15). Under normal operations (bypass) nutrient levels reached peak values at roughly the same time high flow events enter the reservoir though this may be an artifact of sampling

frequency. With only once per month sampling, we may miss time-lags associated with interflows from flood events. Using turbidity measurements as a marker for a high flow event in January, 1997 mass water movement was extremely rapid through the reservoir taking a week or less (Fig. 9). Inflows of greater than 100,000 acre/feet per day were recorded, reducing whole reservoir turnover time to less than 40 days. Ammonia and SRP levels showed large increases following high runoff events, due to high levels often associated with rain driven inflows. Change in TP at this time was due to contribution from SRP. Nitrate in contrast decreased during high flow events. The highest nutrient spikes occurred February-March 1996 and in early January for 1997, and corresponded to the two highest flow events. As reservoir waters warmed, effects were most noticeable in the hypolimnion due to underflow of riverine input. The epilimnion, except in extreme cases, remained relatively isolated, as runoff entered the reservoir as underflow. The flood turbidity graphs further support the idea that mixing is not deep (Fig. 9), and may explain why hypolimnetic nutrient levels are driven more by hydrologic events than were epilimnetic levels.

Following cessation of winter runoff ammonia levels decreased. Ammonia is utilized quickly or converted to nitrate in the well oxygenated waters, and essentially remained at low levels. Nitrate, and phosphorus decreased at a slower rate, mirroring changes in epilimnetic nutrient levels. The spring decrease reflected uptake by algae during the early spring months before the reservoir stratified. An increase in TP/SRP ratios suggested build up of biological organisms during the spring at least in surface waters. Total phosphorus decreased through the spring, as a result of lower nutrient level water in the deep interflow, and settling out of phosphorus due to sediment adsorption.

SRP levels were stable through the summer months and started increasing again in early fall (Fig. 13). Hypolimnetic nitrate increased early in summer then decreased as the season progressed. This may have been due to deep withdrawals. In a reservoir with epilimnetic withdrawal nutrient levels should increase over the summer in the hypolimnion. Deep withdrawals, however, constantly removed hypolimnetic water, and associated nutrients. Occasionally large ammonia spikes were observed, for example at S7 and S9 in October 1995 when deep layers were anoxic (Fig. 14).

Trends in N/P ratios in hypolimnetic waters were most notable in up-reservoir stations. The main forebay exhibited a more stable pattern (Fig. 17). Although co-limitation of nutrients occurred, DIN/SRP ratios indicated seasonal shifts in the availability of the two nutrients. Mid-summer spikes in N/P were primarily due to ammonia, however, neither ammonia or SRP were exceptionally high. There was a spring-fall shift in ratios with low ratios in early fall indicating quicker build up of SRP relative to TIN. Mid-winter ratios indicated a predominance of nitrate in surface waters. Spring decrease indicated more rapid sequestering of SRP relative to nitrate. Trends in hypolimnetic ratios were more stable and exhibited a distinct seasonal cycle versus the multiple cycles observed in surface waters. Hypolimnetic ratios peaked in late summer following a build up of DIN, while SRP levels remained stable. Ratios decreased in winter as high inflow events increased phosphorus concentrations at a greater rate than nitrogen. TP/SRP ratios

remained relatively low in the hypolimnion indicating much of the hypolimnetic nutrient pool was composed of orthophosphate. Epilimnetic TP/SRP ratios cycled to a much larger degree. Ratios were high in the late summer dropping during the rest of the year. This was indicative of a higher proportion of phosphorus tied up in plankton biomass. Increases may be a result of resuspension of reservoir sediments due to dropping reservoir levels in late summer, and/or possibly corresponds to increases in recreational usage and changes in wave action.

4. Primary and Secondary Productivity:

A. Chlorophyll:

Chl *a* is the predominant photosynthetic pigment in planktonic algae, and can be used to estimate total algal biomass. Chl *a* is often used to determine productivity and as a comparison with other aquatic systems to determine trophic status (Likens, 1975). Oligotrophic lakes range from 0.3 to 3.0 µg/L chl *a*, mesotrophic lakes from 2 to 15 µg/L chl *a*, and eutrophic lakes from 10 to 500 µg/L chl *a*. Composite chl *a* concentrations (0 to 5m) in Shasta Lake ranged from undetectable to 30 µg/L. Mean composite chl *a* concentrations for the period 1995 to 1997 were: main lake (2 µg/L), Sac River Arm (2 µg/L), McCloud River Arm (2.7 µg/L), Pit River Arm (5.2 µg/L), and the Pit-McCloud River Arm (3 µg/L) (Fig. 18). Based only on chl *a* Shasta Lake falls into the mesotrophic category.

Chl *a* tracked seasonal algal trends well. There were two well defined chl *a* peaks, one in spring and the other in late summer to early fall (Fig. 19 a-d). Chl *a* was highest at all stations as stratification began developing in the spring, and in the fall as stratification began breaking down. Spring peaks were usually highest, due to greater availability of nutrients in the reservoir from spring run-off and winter mixing. Development of spring chl *a* blooms was likely triggered by photoperiod, and a stabilizing water column. The spring chl *a* peak was not recorded in 1997 due to data loss because of a faulty spectrophotometer (i.e. used to analyze chlorophyll). If the chl *a* peak had been recorded it probably would have been greater than the March 1996 maxima, based on algal biovolume present at the time (Fig. 20 a-c). Once stratification fully developed, limiting mixing of nutrients, algal production utilized remaining nutrients and chl *a* levels decreased and remained low through the summer. Fall increases in chl *a* were generally associated with breakdown of thermal structure in the reservoir. Peak chlorophyll values in 1997, however, occurred in August during the time of greatest stratification. It is possible nutrient resuspension in surface waters due to reservoir drawdown and mixing of hypolimnetic waters contributed to this increase. Increased turbidity as a result of suspended sediments was an indication of potential nutrient influx. Chl *a* decreased from late fall into winter due to shortened photoperiod, cooling reservoir temperatures, and increase in mixing rates.

Spatially, chl *a* levels were higher uplake than downlake, particularly in the Pit River Arm (S9) and McCloud River Arm (S8) (Fig. 19c,d). The lowest composite chl *a* concentrations were reported in the main lake (S6), typical of a finger shaped reservoir. Kimmel et al. (1988) found that spatially, phytoplankton biomass and productivity generally decreased downstream within

reservoirs, reflecting uplake-to-downstream decreases in nutrient availability. Chl *a* values ranged from about 0 to 8.1 µg/L (Fig. 19a). Composite chl *a* productivity of the lower Sacramento River Arm (S7) was similar to the main lake (S6) (Fig. 19b). The upstream station on the Sacramento River Arm (S15) showed similar seasonal trends but had consistently higher chl *a* concentrations due to greater nutrient availability. The McCloud River Arm (S8) was yet more productive than the Sacramento Arm (S7) or the main lake (S6). Chl *a* in the upper McCloud River Arm (S14) was significantly greater than the lower arm only during August 1997 (Fig. 19c). The Pit River Arm (S9, S10, S12, S13) and Squaw Creek (S11) were the most productive sections of the reservoir in terms of chlorophyll production (Fig. 19d). The lowest secchi disk transparencies occurred at these stations due to greater algal productivity and suspended sediments in the water. Composite chl *a* concentrations ranged from about 0 µg/L to about 30 µg/L in the upper section (S13) of the Pit River Arm. Typical peaks were observed in spring, summer, and fall similar to other stations, although the amplitude of the peaks were significantly greater at these stations.

Vertical gradients of chl *a* were also apparent (Fig. 7). Highest chl *a* concentrations occurred below the top 5 m of water during summer months following development of the thermocline. Greatest algal biovolume was lower in the water column during the summer months as warmer surface temperatures, photoinhibition, and lack of nutrients may limit diatoms, the dominant species group in the reservoir. For example, in the Pit Arm (S9), the greatest percentage of bacillariophyta were collected from 10 to 30 m from May through October, while diatoms avoided the upper 10 m of water. The 1% light level for minimum photosynthetic activity was at 20 to 25 m in summer, while thermocline depth was 10-15 m. As such the majority of production was at or near the thermocline, extending into the upper metalimnion. This zone contained higher nutrients than surface waters yet still had adequate light for photosynthesis. This was reflected in the Pit River Arm (S9) and Pit-McCloud confluence (S10) during late summer when greater concentrations of chl *a* were found below 10 m (Fig. 7).

Flooding in January 1997 had a pronounced effect on chlorophyll production in the reservoir. Flood waters entering the headwaters of the reservoir made the arms extremely turbid. Chl *a* concentrations were very low at upper arm stations and probably light limited. During this time, however, there was production in the main lake (S6) which was clearer. By February, mixing of the water column had introduced enough turbidity that even in the main lake algal productivity stalled. The waters began to clear in March, and in April there was a tremendous bloom in diatoms resulting from the high nutrient load from the floods.

B. Phytoplankton:

A total of 134 phytoplankton species were collected from the main lake (S6) and Pit River Arm (S9) stations (Table 2). The algal community of Shasta Lake was dominated by diatoms, and occasionally by cryptomonads. The dominant diatom species were *Melosira islandica* and *Melosira varians*. *Melosira* sp. is one of the most ubiquitous of algal genera, and is widely distributed in all types of water bodies. *Rhodomonas minuta var. nannoplanctica* and *Rhodomonas minuta* were the primary cryptomonads. Cryptomonads (cryptophyta) were only

dominant from June through September 1995. Cryptophytes (Fig. 21b) were collected at all times but represented a significant proportion of the total phytoplankton only when diatom biovolume was low. Since 1995, diatoms (bacillariophyta) have remained the dominant group through 1997 (Fig. 21a). Blue-green and green algae were never common in the main lake (S6). There were occasional blue-green blooms *(Anabaena* spp.), as during January 1996 (Fig. 20b), however, low-levels of nutrients in Shasta Lake typically do not favor the dominance of blue-green assemblages.

Spring blooms peaked in March or April at the on-set of thermal stratification when nutrients were abundant and became available to phytoplankton (Figs. 20b-c). In May, as stratification intensified mixing ceased, and algal blooms crashed as nutrients were depleted from the epilimnion. Phytoplankton biovolume remained low from early to mid summer. In late summer biovolume increased and a fall algal bloom occurred. Biovolume increases during August or earlier may have been due to drawdown of the reservoir. As reservoir levels dropped wave action resuspended sediment and may have released nutrients. Fall blooms coincided with the breakdown of thermal stratification and circulation of nutrients from the metalimnion. Plankton populations began developing again in January or February. In the main lake (S6) the majority of diatom biovolume was concentrated in the top 10 m of the water column during the spring bloom. Phytoplankton moved downward in the water column into the 10-20 m, and 20-30 m depth interval (Fig. 21a) as the summer season progressed. Downward movement of phytoplankton may have been from settling and the preference of cooler water temperatures found deeper. Cryptophyta biovolume remained low throughout the year. Cryptomonads are known to adapt to low light levels (Wetzel 1975) and were collected in all 3 depth intervals (Fig. 21b).

Algal biovolume in the main lake (S6) increased significantly from 1995 to 1997 (Fig. 20a-c). In the main lake (S6), biovolume increased to 45,000 µg/L during April 1997. This was an increase of almost 10 fold from the maximum algal peak in March 1996 (4700 µg/L). The increase may have been caused by influx of nutrients from winter floods. In January and February following flooding, algal production developed in the main lake. By February flood waters had circulated to the point that the entire reservoir was turbid and suppressed production. Secchi disk transparency decreased to less than 0.5 m. When the main lake began to clear, the secchi disk transparency increased to about 2 m and algal blooms reached a maximum in April. The reservoir became thermally stratified in May, algal blooms collapsed, and secchi disk transparency reached a maximum.

There was greater algal biovolume in the Pit River Arm (S9) than in the main lake (S6). During September 1995, algal biovolume was dominated by *Melosira varians* in the Pit River Arm (S9) and biovolume reached 12,000 µg/L (Fig. 22a). In 1996, spring blooms peaked in April, one month later than in the main lake (Fig. 22b). Diatom biovolume reached 17,000 µg/L. The lowest biovolume occurred in June, then increased from July through October. In the spring of 1997 maximum biovolume was 38,000 µg/L, but did not exceed the bloom (45,000 µg/L biovolume) at S6 during the same period. Diatoms were concentrated in the upper 10 m of the water column only during early spring (Fig. 23a) when light and temperature were limiting at greater depths.

The rest of the year, diatoms were collected from deeper in the water column below 10 m, due to deeper light penetration, cooler water temperatures, and greater nutrient concentrations in the metalimnion. Cryptomonad biovolume was greater in the Pit River Arm than in the main lake station. Biovolume was evenly distributed throughout the water column (Fig. 23b) and occurred throughout the season, but cryptomonads were never dominant in the Pit River Arm (S9).

There was greater phytoplankton biovolume during 1997 compared to the two previous years (Fig. 22a-c). The bloom in April 1997 was significantly greater than blooms occurring in 1995 or 1996. Zooplankton biomass significantly decreased during the same period indicating less cropping of phytoplankton, and therefore larger algal blooms. Phytoplankton biovolume increased in late summer to 30 µg/L (0 to 5 m) from reintroduced nutrients from reservoir drawdown and again in the fall from reservoir turnover.

C. Zooplankton:

A total of 33 zooplankton species were collected from all stations: 8 cladocera, 3 copepoda, and 22 rotifera species (Table 3). The dominant cladocerans were *Daphnia pulex* and *Daphnia laevis* which are herbivorous zooplankters. The dominant copepod species were the cyclopoids, *Leptodiaptomus ashlandi* and *Limnocalanus macrurus.* Cyclopoids for the most part are carnivorous. Numerous species of rotifers were present throughout the year (Table 3) but were not a significant proportion of total zooplankton biomass. Greatest mean zooplankton biomass occurred in the Sacramento River Arm (S7) and the least zooplankton biomass in the McCloud River Arm (S8) for the sampling period, 1995 to 1997 (Fig. 24). Cladocerans were the dominant zooplankton at the main lake station (S6) and Sacramento River Arm (S7). Cladoceran and copepod biovolume was equally distributed at other stations (Fig. 25).

In 1995, total zooplankton biomass reached about 1200 µg/L in the Sacramento River Arm (S7) and Pit River Arm (S9). Zooplankton biomass was about 400 µg/L in the McCloud River Arm (S8). Total zooplankton biomass at all stations decreased in 1996 (Fig. 25b). Zooplankton biomass in the Pit River Arm (S9) and Pit-McCloud Confluence (S10) decreased more than 50 percent from the previous year. There was a further decrease in biomass in 1997 in the main lake (S6) and the Sacramento River Arm (S7). Total zooplankton biomass in the McCloud (S8), Pit (S9), Pit-McCloud Confluence (S10) did not significantly change in 1997. Overall, a significant reduction in zooplankton biomass has occurred at all stations since this study began in 1995. One possible explanation may be greater fish densities (i.e. threadfin shad) in the reservoir as compared to 1995 that are feeding on zooplankton. As a result of fewer zooplankton, phytoplankton biovolume has increased dramatically in the main lake (S6) and Pit (S9) since 1995, particularly during 1997 (Lieberman 1996).

Cladocerans were dominant in the main lake (S6) from spring through mid summer. The rest of the year copepods dominanted. Cladoceran and copepod springtime blooms sometimes occurred at the same time, although the copepod blooms were not as large as cladoceran blooms. Cladocerans composed as much as 85 percent of the total zooplankton biomass during the spring

bloom (Fig. 26 a). In 1995, we missed the spring bloom because we began sampling in June. Cladoceran biomass reached a maximum in June, followed by a copepod peak in August in the main lake (Fig. 26 b). In 1996, cladocerans and copepods bloomed simultaneously in April. In 1997, copepods bloomed in April followed by a cladoceran bloom in May. Blooms usually coincided with algal peaks. Following spring algal blooms, nutrients were depleted, phytoplankton died off and cladocerans populations decreased. This was followed by copepod dominance. Cladocerans and copepods were collected down to 30 m, although the majority of the zooplankton were in the upper 20 m of the water column (Figs. 27 a,b). Net tows from surface to bottom during a field sampling trip were used to determine if we were missing zooplankton by only collecting to 30 m. Significant numbers of zooplankton below 30 m were not collected in the tows. Zooplankton often exhibit diel migration throughout the water column to escape fish predation or for energetic reasons. At Lake Powell, Utah-Arizona, Bureau of Reclamation and USGS scientists have observed daily migration of zooplankton down to depths of 30 m in the water column (M.Horn, personal observation). At Shasta Lake, during a hydroacoustic fishery survey in April 1997, we observed some movement of zooplankton with hydroacoustic equipment. Zooplankton were concentrated at about 20 to 30 m early in the evening but at about 5 m after dark. The zooplankton may not be moving through the water column as freely as observed in some other reservoirs. In mid-summer zooplankton moved to below 10 m when the thermocline began to decline in the water column (Fig. 27 a,b). Early in the season during cladoceran-copepod blooms the majority of plankton were collected from the top 10 m down to 30 m.

Seasonal zooplankton trends in the Sacramento River Arm (S7) were similar to the main lake station (Fig. 28 a,b). In fact, the Sacramento River Arm (S7) was similar to the main lake in secchi depth transparencies, nutrient concentrations, and chl *a* concentrations. The main lake (S6) and the Sacramento River Arm (S7) were not as productive as the Pit River Arm (S9) in terms of chl *a* and algal production although there was greater zooplankton biomass. Specifically, cladoceran biomass was higher in the Sacramento River Arm (S7) and we speculate this may be due in part to lower fish abundance in this section of the reservoir. Hydroacoustic surveys of the lake were conducted and data will be helpful in answering these questions in regard to location and biomass of fish. The majority of copepods (Fig. 29a) were collected in the upper 20 m of the water column. For example, during blooms in August 1995 and April 1996 the greatest biomass of copepods were collected from 10 to 20 m. Cladoceran (Fig. 29b) blooms during the same time were concentrated in the upper 20 m of the water column.

Similar seasonal trends were observed in the McCloud River Arm (S8), although at this station the cladoceran and copepod peaks usually did not coincide (Figs. 30 a,b). Competition for food may be a probable reason. Copepods generally dominated except during major cladoceran blooms. This was the only station where the cladoceran bloom (125 µg/L biomass) in May 1997 was greater than in the two previous years. The zooplankton bloom was still lower than in the main lake (S6) and Sacramento River Arm (S7). It is significant to note that in 1997, cladocerans were dominant for a greater period of time (May to July) compared to previous years. A similar trend was observed in the Pit River Arm (S9) and Pit-McCloud confluence (S10). The distribution

of zooplankton throughout the water column is illustrated in Fig. 31 a,b.

Overfall, the Pit River Arm (S9) was the most productive. The Pit River Arm (S9) had the greatest concentration of chl *a* and algal biovolume, and lowest secchi depth transparencies. Total zooplankton biomass has decreased since 1995. Copepods dominated in both 1995 and 1996. In 1997, from April through September, cladocerans dominated (Fig. 32a) and composed at least 50 percent of the total zooplankton biomass. This general trend towards cladoceran dominance was observed at all stations. The increase in percent composition of cladocerans was due to increase in biomass (Fig. 32b) and the general decrease in copepod biomass during the 1997 spring and summer months. Zooplankton were collected deeper in the water column as the season progressed (Fig 33 a,b). The Pit-McCloud Confluence (S10) was most similar to the Pit River Arm (S9) in productivity. Cladocerans were dominant from March through September 1997 (Fig. 34 a,b). Most zooplankton were concentrated in the upper 30m of water (Fig. 35 a,b).

There were twenty-two species of rotifers collected. The maximum biomass of rotifers was collected from the Pit River Arm (S9) (Fig. 36). In Shasta, rotifer biomass composed less than 0.5 percent of the total zooplankton. The highest biomass (0.33 µg/L) occurred at the Pit in August 1995. Often times, rotifers are associated with more eutrophic conditions meaning the more productive a body of water, the greater the rotifer biomass (Lieberman, 1986). Most stations had less than 0.05 µg/L biomass of rotifers. Seasonal trends for rotifers showed spring and summer maxima followed by fall and winter minima. Similar trends were observed for cladocerans and copepods and may be why rotifers were not dominant in the reservoir. Rotifers cannot compete for food with larger zooplankton (Lieberman, 1986) but at the same time are not inhibited by larger algae and high bacterial densities that may exist in a reservoir during the summer months (Orcutt and Pace, 1984).

Significant decreases in zooplankton have occurred since 1995, particularly in the main lake (S6) and Sacramento River Arm (S7). Compounding factors that may have affected the zooplankton include the January 1997 floods, TCD operations that began in March, and grazing of zooplankters by possible increase in fish abundance. The floods of January and February 1997 flushed the reservoir and zooplankton may have been flushed downstream. When the reservoir inflows entered on January 1, 1997 at greater than 200,000 cfs flushing rate of the reservoir decreased to approximately 8-10 days. Hayward and Van Den Avyle (1986) observed residence times of at least 50 to 250 days were needed to allow establishment of plankton populations that reflected the productive potential as well as effects of species interactions in the reservoir. Brook and Woodward (1956) found that the water exchange rate had to be greater than 18 days for significant development of zooplankton, and Johnson (1964) demonstrated that when the mean flushing rate was less than 15 days, the effect on development was not linear. Cowell (1967) found that high flushing rates of 8-10 days caused little in situ zooplankton production.

5. River Studies: Downstream Drift In Shasta and Keswick Tailwaters Below Shasta Lake:

A. Temperatures:

Temperatures in the Sacramento river showed similar seasonal trends in both Shasta and Keswick tailwaters (Fig. 37). Water temperatures in Keswick tailwaters were slightly warmer. River temperature increased from April through October and decreased from October through March. The TCD began to operate on March 13, 1997 and upper and mid gates were opened in various combinations. In April, upper gates were open, and warmer water was released from the top layer of the reservoir, producing river temperatures higher than those recorded in April 1995 and 1996. In May, the mid gates were opened following reservoir stratification and temperatures dropped in the river because cooler metalimnetic water was being released. In June, upper and mid gates were used in combination and temperatures increased over what was recorded for the two previous years. Opening and closing the top and middle gates changed downstream water temperatures significantly. In general, there may be warmer temperatures in Shasta and Keswick tailwaters with the TCD operating during winter and spring than in previous years. Cooler temperatures will occur in the river during the late summer and early fall due to release of hypolimnetic waters. This shift may mean species changes in macroinvertebrate and plankton downstream.

B. Particulate Organic Matter (POM):

POM supports the base of the food web in many aquatic systems and is one of many factors determining the nature of biotic communities in a riverine ecosystem. Composition of POM is diverse and includes living and dead, whole and fragmented plant and animal material. POM provides the trophic connection between microbial assemblages and macroconsumers. It is the particulate fraction that is directly available to filter feeders whereas the dissolved fraction is not. The dissolved fraction usually is present in greater concentrations but not useable as food by freshwater filter-feeding zooplankton. If these compounds were made available by becoming particulate they could represent an important source of carbon in suspension. Detrital material is an important part of the food web and provides energy for bacterial and aquatic fauna and is commonly found to be the major proportion of stomach contents of many fish species, particularly salmonids. Microbial maceration and animal consumption can convert relatively large sized POM to finer sizes which dominate the organic matter pools. Usually the plankton only composes a small fraction of the POM except during the spring when plankton productivity is high in the reservoir. When limnoplankton is discharged from the reservoir it becomes an important part of the POM in the river.

The <25 μm POM composed the greatest proportion of total POM (Fig. 38 a,b) and contributed about 90% to the total POM in Shasta and Keswick tailwaters. <25 μm POM levels were higher in Keswick tailwater than at Shasta (Fig. 39a). The >25 μm and the >505 μm size fraction contributed the remainder. Detrital and small plankton fragments composed the majority of the <25 μm size fraction. The <25 μm size fraction has been broken down from the coarse POM and contributes greatly to the food base. Numerous investigators have reported that this smallest size fraction dominates in streams both natural and manmade (Maciolek, 1966; Fisher and Likens, 1973; Naiman and Sedell, 1979; Webster et al., 1979; Vannote 1980; Lieberman and Burke, 1993). <25 μm POM was greatest in Shasta tailwaters during the spring (March 1996 and April

1997) which coincided with algal blooms in the reservoir. The <25 μm size fraction also increased during the winter floods of February 1997. The <25 μm size fraction reached 1.5 g/m^3 in Shasta tailwaters and 2.0 g/m^3 in Keswick tailwaters.

Maximum POM concentrations are often associated with storm events such as flash floods (Fisher and Minckley, 1978) when stream bank erosion and sediments can contribute detrital material (Bilby and Likens, 1979). Detrital material associated with flooding at Shasta included large amounts of logs and woody debris washed from upstream into the reservoir. There was no correlation between discharge and <25μm POM. POM tended to fluctuate from month to month with greater concentrations occurring during spring time and flood events.

The <25 μm PIM (particulate inorganic matter) (Fig. 40a) also fluctuated significantly during the start of the rainy season, flood events, and rising water levels which increased channel sediment resuspension and decreased the proportion of particulate organic matter in suspended material. The majority of inorganic drift was in the <25 μm size fraction and the majority of the <25 μm size fraction was inorganic material (Fig. 40a). In February 1997, PIM increased to 12 g/m^3 in Shasta tailwaters and to 16 g/m^3 in Keswick tailwaters (Fig. 41a). During this period turbidity levels in the reservoir increased dramatically and much of the resuspended sediment was flushed downstream contributing to muddy river waters. PIM concentrations in Keswick tailwaters were higher than in Shasta tailwaters.

The >25 μm POM was composed of larger plankton and detrital material. This size fraction contributed the smallest proportion of POM to the total. The Keswick station was a good example of how downstream POM in the >25 μm size fraction fluctuated independently of Shasta tailwaters (Fig. 39b). During February 1996 >25 μm POM increased to a maximum of 0.08 g/m^3 at Keswick whereas POM upstream did not increase. Tributaries entering the Sacramento River above Keswick Dam added detrital material, producing spikes in Keswick tailwaters that did not appear at Shasta. The >25 μm PIM in Shasta and Keswick tailwaters peaked in March 1997 (Fig. 40b. 41b), one month later than the <25 μm PIM (Fig. 40a). By March, the reservoir began to clear and turbidity levels decreased considerably. The >25 μm size fraction contributed very little to the total PIM in Shasta tailwaters because the reservoir tended to act as a sediment trap except during flood events (Fig. 40b). Downstream the inorganic matter increased in Keswick tailwaters (Fig. 41b) from suspended sediment brought in from the river bank.

The >505 μm size fraction made up a greater proportion of the total POM than the 25 μm to 505 μm size fraction. It was composed of green algae (Cladophora strands) and larger zooplankton. The <25 μm and the >25 μm size fractions were almost always greater downstream in Keswick than in Shasta tailwaters whereas the >505 μm size fraction (Fig. 39b) showed greater concentrations upstream in Shasta tailwaters. Since March 1996, the >505 μm POM has been greater in Shasta tailwaters. Increases in 1996 were possibly due to more cladophora collected in the 505 μm nets from surrounding cladophora beds and not due to increased zooplankton biomass. Increases in >505 μm POM during 1997 were due to increases in zooplankton biomass caused by operation of the TCD. The peak in April 1997 was due to

increased copepod biomass and the June, 1997 peak to increased cladoceran biomass. Upper gates were opened April 1997 during the copepod bloom and again in June when cladocerans were dominant.

Many investigators have found that both living and non-living plankton released from reservoirs provide support to large populations of filter feeding macroinvertebrates downstream. POM significantly decreased 14 km downstream in Keswick tailwaters due to settling out of plankton. Travel time between Shasta and Keswick was a matter of hours depending on flows. Limnoplankton decreased over this stretch and smaller size fraction detrital material increased. The total >505 μm size fraction (Figs. 40c, 41c) in Shasta and Keswick tailwaters was dominated by the organic portion. PIM did not increase during the flood event of 1997 as observed with the two smaller size fractions. The majority of silt and sediment coming out of Shasta was in the <25 μm size fraction. Larger size fraction sediments settle out quickly because of hydrdynamic changes at the head of the reservoir and are not transported through the reservoir. Increases in PIM in Keswick tailwaters during February 1996 was due to increased sediment flow from above Keswick dam and not an influence from Shasta lake since there was not an increase observed at Shasta tailwaters during this time. Depending on the timing and location of reservoir releases, particulate content of outflows may vary considerably. However, if released waters are rich in particulate matter, macroinvertebrate populations of Trichoptera, Ephemeroptera, and Diptera may be substantially enhanced.

Overall, total POM in Keswick tailwaters was higher than in Shasta tailwaters (Fig. 42) for the sampling period from 1995 to 1997. Ward (1975) found POM concentrations increased with distance downstream from Cheesman Lake on the South Platte River, Colorado. Total POM averaged 0.8 g/m^3 and 0.9 g/m^3 in Shasta and Keswick tailwaters, respectively. This was comparable to results found by other investigators in other 6^{th} and 7^{th} order streams in the western USA (Webster et al. 1979). Lieberman and Burke (1993) reported that the mean total POM concentration was 0.80 g/m^3 below Davis Dam on the lower Colorado River. Birge and Juday (1934) reported mean POM concentrations for 57 northern Wisconsin natural lakes as 0.83 g/m^3.

C. Zooplankton:

Zooplankton composed a portion of the total particulate organic matter. Zooplankton identified were in the >25 μm and >505 μm size fractions. Zooplankton breaks down into detrital material in the river and may contribute to the <25 μm POM. There was a significant increase in zooplankton biomass in 1997 compared to the previous years in Shasta and Keswick tailwaters (Figs. 43a, 44a). Copepods, cladocerans, and rotifers showed increases. Rotifers were found for the first time as significant biomass in Shasta tailwaters (Fig. 43a). There was a change in the percent composition of major zooplankton groups from 1995 and1996, to 1997 (Figs. 43b and 44b). Cladocerans composed less and copepods made up more of the total zooplankton biomass.

Changes in zooplankton drift were attributed to operation of the TCD in early spring. Releases of epilimnetic water during the most productive time of year resulted in entrainment of zooplankton

and a downstream increase in biomass. Hypolimnetic withdrawals in previous years introduced little zooplankton biomass to Shasta tailwaters due to deeper level releases. Upper bypass intakes did discharge water from an upper depth of about 20 m, but upper most surface waters were not withdrawn. In 1995 and 1996 there were greater spring zooplankton blooms in the main lake (S6) compared to 1997 but this was not reflected in downstream drift (Fig. 45a) because withdrawals occurred below the depth of greatest zooplankton production. Zooplankton biomass blooms in the main lake (S6) significantly decreased in spring 1997 whereas the zooplankton drift in the river significantly increased. This was a direct result of TCD operation. The upper most strata of water was discharged downstream through the TCD when the reservoir was at full pool.

There was a increase in zooplankton biomass in Shasta tailwaters during April 1997 as the top gates of the TCD were open. Copepods increased to 18 µg/L, dominated by the adult cyclopoid *Leptodiaptomus ashlandi*, immature cyclopoids, and copepodids. Cladocerans increased and were dominated by *Daphnia pulex*. Rotifers significantly increased in April. In May, a combination of mid and top gates were used and the zooplankton biomass for all three groups decreased. During June, top gates were opened once again and cladocerans increased in the tailwater drift. Copepod biomass in Shasta tailwaters was greater for July and August than in the two previous years.

Increase drift of zooplankton biomass affects POM composition. Release of limnoplankton from the reservoir may shift the composition of POM from detrital to limnoplankton based during the most productive times of the year. During the summer of 1997 the top and middle gates of the TCD were open until August. With operation of the TCD, organic matter was being expelled from the reservoir as drift. Epilimnetic releases favored living and dead limnoplankton as observed in 1997 whereas hypolimnetic releases contributed nutrients to downstream communities. Dance (1981) listed the presence of lakes/impoundments as an important factor controlling the concentration of POM being transported through a river system. A number of investigators have reported reservoirs trap POM coming into them, thereby modifying and disrupting the natural downstream drift (Maciolek, 1966; Armitage 1977; Goldman and Kimmel, 1978), although Lieberman and Burke (1993) did not find this to be true of the Lower Colorado River reservoirs.

There was a decrease in zooplankton biomass from upstream in Shasta tailwaters to downstream in Keswick tailwaters (Fig. 45b) and has been observed in other studies (Lieberman and Burke, 1993; Soballe and Bachmann, 1984). The change from a lentic environment in the reservoir to a lotic environment in the river does not favor reservoir produced plankton. Travel time from Shasta to Keswick was short, a matter of hours, as compared to the residence time in the reservoir of over one year. Even though limnoplankton decreased from upstream to downstream the detrital content increased as material was broken down. Total POM concentration in Keswick tailwaters was greater than in Shasta tailwaters, but zooplankton biomass was lower. Grazing, sedimentation, and mechanical destruction, as well as filtering, may have effected the decrease in numbers.

Lentic zooplankton were affected most severely by the change to lotic conditions, and rapidly decreased in numbers downstream. Ward (1975) found that copepods dominated the zooplankton

in the South Platte River below Cheesman Lake, Colorado but within 5 km of the dam the rotifers were the most abundant organisms. Importantly, Ward (1975) demonstrated that the persistence of zooplankton, downstream from a lentic source, was primarily related to their size and form (in terms of their exoskeletal components). The larger organisms, having a smaller surface: volume ratio, sink more rapidly, become entangled more easily, and be more vulnerable to mechanical destruction and fragmentation, than the smaller ones.

D. Phytoplankton and Chlorophyll:

Three size fractions of phytoplankton corresponding to POM size fractions were identified and enumerated. The <25 μm size fraction composed the greatest phytoplankton biovolume in Shasta and Keswick tailwaters (Figs. 46 a,b). As with zooplankton biomass, phytoplankton biomass increased significantly in 1997 following operation of the TCD. In April 1997 there was a large diatom bloom in the main lake (S6) and this was reflected in a 10-fold biovolume increase in the tailwaters of both stations. The top gates of the TCD were open in April allowing for surface releases from the reservoir. Most of the phytoplankton in the reservoir were concentrated in the upper 20 m of the water column. Chl *a* in the river was generally greater downstream in Keswick than in Shasta tailwaters (Fig. 47). Chl *a* ranged from about 1.1 μg/L to 2.7 μg/L but was often below 1.0 μg/L. Chl *a* at Shasta tailwaters exceeded chlorophyll concentration at Keswick during February 1997.

Phytoplankton biovolume was higher in Shasta tailwaters than in Keswick tailwaters. Phytoplankton released from Shasta Lake during algal blooms (i.e. April 1997) appeared to fall out of the drift quickly and did not always reach Keswick tailwaters in the form of algal material. Reservoirs provided an important source of phytoplankton for downstream reaches, although the lentic species may be selectively eliminated by filtering, sedimentation, and destruction during transport by the river. Hynes (1970) found that diatoms and blue-green algae survived downstream transport better than green algae and desmids , although in the Sacramento River drift that did not always appear to be true.

Composition of the <25 μm biovolume in Shasta tailwaters was 90 percent bacillariophyta (diatoms) (Fig. 48a). The dominant diatom collected in the tailwaters was *Melosira varians*. At Keswick tailwaters composition was 90 percent diatoms, 5 percent green algae, 2 percent blue-greens, and 3 percent other algae. At Keswick other algal species were introduced into the river upstream from the dam from allochthonous sources other than Shasta Lake. The >25 μm biovolume was 87 percent diatoms in Shasta and 96 percent diatoms in Keswick tailwaters (Fig. 48b). The >505 μm size fraction was composed of about 70 percent diatoms and 27 percent green algae in Shasta and 95 percent diatoms and 3 percent green algae in Keswick (Fig. 48c). The >25 μm and >505 μm size fractions were represented by fewer diatoms in Shasta tailwaters while the opposite was true of Keswick. This was a good example of how the drift composition in Shasta and Keswick tailwaters were somewhat different.

Mean phytoplankton biovolume increased significantly in the drift during 1997 after TCD

operations began. As observed with zooplankton drift, the two factors that contributed to greatest biovolume increases downstream were surface withdrawal and withdrawal during peak production from the reservoir. In support of this statement are past data that show little downstream drift when withdrawals were made through bypasses, below 20 m in the water column.

IV. Post TCD Results:

As of this report the TCD has been in operation for one season. The following represent some preliminary observations of potential changes that have occurred. Physical limnology of the reservoir shows several significant changes to patterns of water movement through the reservoir. Approximately 20 km upstream in the river arms were affected by TCD operations (Hanna et al. 1998). It is difficult to determine if onset of stratification was delayed by TCD operations simply due to annual variability in the onset of stratification. Once stratified, however, thermal gradients were much sharper. This was a direct result of how the TCD withdrew mid-level water. Dissolved oxygen minima were more intense upstream, and there was a delay in the timing of the minima moving downstream until low level releases were used in late summer, moving this body of water downstream.

In spring and summer operations of the TCD caused the reservoir hypolimnion to act as a nutrient sink. Previously, hypolimnetic waters had been discharged throughout the season and nutrient levels did not build to the level observed this last year. With surface releases there may be additional accumulation of nutrients available during fall turn-over from hypolimnetic waters for algal production. Fewer nutrients were transported downstream during the spring and early summer when surface waters are released with TCD operations. More nutrients left the reservoir later in the season, late summer and fall, when bottom gates were opened and hypolimnetic (coolest waters) released downstream. Some questions that remain to be answered are; 1. Will there be residual (carryover) effects from the previous year and an accumulation of nutrients in the reservoir because of epilimnetic releases? 2. Will shifts in timing of nutrient releases affect downstream biota? All of these questions come to mind when thinking about how the TCD may affect and change the nutrient budget of the reservoir and at the same time may cause increased algal production in the reservoir, if in fact, additional nutrients become available in the upper strata of the water column. Paulson (1981) showed that epilimnetic rather than hypolimnetic releases of water from Hoover Dam would enhance productivity of the reservoir by reducing NO_3-N losses from the reservoir. At Shasta, selective withdrawal allowed for a combination of releases to occur during the spring-summer months.

During this first season of TCD operation there were larger algal blooms in the main lake during the period of surface withdrawals than previously observed. Downstream, diatoms increased dramatically in tailwaters due to surface water releases during the most productive time of year, and from the depth of greatest algal production. Generation time of algae is extremely short (a few days), and even after a flood as observed in the data, algae recovered rapidly. If flushing rate does not exceed mean doubling time of phytoplankton, increased inflow may enhance

phytoplankton productivity by increasing nutrient availability (Carmack et al. 1979).

We observed significant decreases in the zooplankton biomass in the reservoir and at the same time zooplankton drift has significantly increased in downstream tailwaters. During March 1997 the upper gates of the TCD were open before the zooplankton bloomed. In April, at the main lake station (S6) copepods were at maximum biomass when the top gates were open, obviously effecting the upper strata of the water column where zooplankton were most abundant (as discussed above). We hypothesized that POM would increase in the Sacramento river with operation of the TCD and discharge of upper strata of water during spring months. In the downstream drift there were significant increases in copepods. In May, both the top and middle gates were open during the same time when cladocerans were at a maximum and zooplankton decreased in the river, but then in June only the top gates were open once again and zooplankton increased in the river. We have observed an increase in the zooplankton from our plankton collections in the river immediately after the TCD began operation this past spring. The increase in zooplankton in the drift was directly related to depth of withdrawal and time or season of withdrawal. Zooplankton were most concentrated in the upper 30 m of water and most productive during the spring months.

The decrease of zooplankton in the reservoir at all stations may be linked to upper strata of water being released during the most productive time of year. Lower zooplankton biomass means a loss of food for fish. Zooplankton entrainment and cropping of zooplankton may be occurring simultaneously. It is difficult, however, to show whether reservoir fish have increased since 1995 because of the lack of data from Shasta Lake. After the reservoir stratified, the algal production decreased because of nutrient depletion and the zooplankton biomass naturally decreased during mid summer due to food limitation. At Hungry Horse reservoir, Montana (Brian Marotz, pers. comm, 1998) downstream loss of zooplankton through the dam was predicted to increase as warm epilimnetic waters were released. 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 the reservoir fishery.

V. Management Implications:

Studying the TCD on Shasta will provide other investigators with valuable information on selective withdrawal and how operations may effect the physical, chemical, and biological aspects of a reservoir. Currently, selective withdrawal systems are functioning on Hungry Horse Dam, Montana; Libby Dam, Montana; Cachuma Dam, California; Casitas Dam, California; Folsom Dam, California; and Flaming Gorge Dam, Utah. There has been considerable research on the physical aspects of these systems. In the future, a TCD is being planned for one of the largest reservoirs in the country, Lake Powell in Utah (Glen Canyon Dam). Additional research is needed on the biological effects from TCD operations in these reservoirs and downstream. At Hungry Horse Dam researchers are examining zooplankton entrainment from the selective withdrawal structure on Hungry Horse Dam. On Hungry Horse there were shutter gates that were added to the original design to reduce zooplankton entrainment while still meeting the target discharge

temperature (Brian Marotz, pers. comm.). On Lake Powell one of the primary concerns is whether TCD operations would alter the thermal structure of the reservoir. Thread-fin shad are the primary forage fish in Lake Powell. Currently winter minimum temperatures in the reservoir approach the thermal minimum for shad. If the TCD results in lower water temperatures, shad will likely die-off and the recreational fishery will be substantially reduced.

Our results on Shasta Lake and tailwaters have shown some effects from operating the TCD on Shasta Lake that include: 1) Cooler water tailwater temperatures during the summer months; 2) Significant algal blooms in April 1997 over the two previous years; 3) Significant decrease in spring zooplankton blooms; 4) Increase in algal biovolume in Shasta and Keswick tailwaters; 5) Significant increase in zooplankton biomass in Shasta and Keswick tailwaters; 6) Change in composition and increase in particulate organic matter in Shasta and Keswick tailwaters; 7) Shift in water movement patterns through the reservoir.

The goal of this project and in our continued investigation of the limnological aspects of Shasta was to identify potential water quality or biological problems that might arise within Shasta Lake due to TCD operations; identify changes in downstream drift and nutrient release that may potentially influence fish production and; allow for optimal use of the TCD to achieve specified downstream temperature requirements for chinook salmon. Though the primary use of the TCD was to provide benefit for one particular endangered species, its effects could negatively impact other species such as recreational fishes or endangered species that might use the reservoir. A good limnological understanding of the reservoir will allow proper operations that minimize impacts to all species of interest.

VII. Acknowledgments:

This study was funded through U.S. Geological Survey and U.S. Bureau of Reclamation. We are grateful to Clair Stalnaker, USGS, Midcontinent Environmental Science Center, for all of his meticulous assistance in keeping this project on track. We thank Craig Sarsfield, BOR, Northern California Area Office for technical support and sound advice on the temperature control device. Thanks to Shawn Duffy for phytoplankton data entry and Janet Martin for providing help collecting data on Shasta reservoir.

VI. Literature Cited:

Armitage, P.D. 1977. Invertebrate drift in the regulated river Tees, and an unregulated tributary Maize Beck, below Cow Green dam. Freshwater Biol. 7:167-183.

Bilby, R.E. and G.E. Likens. 1979. Effects of hydrologic fluctuations on the transport of fine particulate organic carbon in a small stream. Limnol. Oceanogr. 24:69-75.

Birge, E.A. and C. Juday. 1934. Particulate and dissolved organic matter in inland lakes. Ecol. Monogr. 4:440-474.

Brook, A.J. and W.B. Woodward. 1956. Some observations on the effects of water inflow and outflow on the plankton of small lakes. Journal of Animal Ecology. 25: 22-35.

Carmack, E.C., C.B. Gray, C.H. Pharo, and R.J. Daley. 1979. Importance of lake-river interaction on seasonal patterns in the general circulation of Kamloops Lake, British Columbia. Limnol. Oceanogr. 24:634-644.

Cowell, B.C. 1967. The copepoda and cladocera of a Missouri river reservoir: a comparison of sampling in the reservoir and the discharge. Limnology and Oceanography. 12:125-136.

Dance, K.W. 1981. 'Seasonal aspects of transport of organic and inorganic matter in streams' in Lock, M. and D. Williams (Ed.). Perspectives in Running Water Ecology. Plenum Press, New York. pp.69-96.

Fisher, S.G. and Likens, G.E. 1973. Energy flow in Bear Brook, New Hampshire, an integrative approach to stream ecosystem metabolism. Ecol. Monogr. 43:421-439.

Fisher, S.G. and W.L. Minckley. 1978. Chemical characteristics of a desert stream flash flood. J. Arid Environ. 1:25-33.

Goldman, C.R. and Kimmel, B.L. 1978. 'Biological processes associated with suspended sediment and detritus in lakes and reservoirs'in Cairns, J., Jr., E.F. Benfield, and R. Webster (Ed.). Current Perspectives on River-Reservoir Ecosystems. North American Benthological Society, Springfield, Illinois. pp. 19-44.

Hanna, B., J. Bartholow, L. Saito, and J. Sandelin. 1998. Results of mock Shasta TCD operations on in-reservoir and discharge water temperatures using CE-QUAL-W2. U.S. Geological Survey. USGS/MESC/3302.

Hayward , R.S. and M.J. Van Den Avyle. 1986. The nature of zooplankton spatial heterogeneity in a nonriverine impoundment. Hydrobiologia. 131: 261-271.

Hynes, H.B.N. 1970. The ecology of running waters. University of Toronto Press, Toronto.

Johnson, W.E. 1964. Quantitative aspects of the pelagic, entomostracan zooplankton of a multibasin lake system over a 6-year period. Verhandlungen der Internationalen Vereingung fuer Theoretische und Angewandte Limnologie. 15:1119-1125.

Kimmel, B.L., D.M. Soballe, S.M. Adams, A.V. Palumbo, C.J. Ford, and M.C. Bevelhimer. 1988. Inter-reservoir interactions: Effects of a new reservoir on organic matter production and processing in a multiple-impoundment series. Verh. Internat. Verein. Limnol. 23: 985-994.

Lieberman, D.M. 1996. Use of silver carp (*Hypophthalmichthys molotrix*) and bighead carp

(*Aristichthys nobilis*) for algae control in a small pond: changes in water quality. Journal of Freshwater Ecology. 11(4):391-397.

Lieberman, D.M. 1986. Variation in the crustacean and rotifer plankton of riverine and nonriverine lower Lake Havasu, Arizona. M.S. thesis.

Lieberman, D.M. and T.A. Burke. 1993. Particulate organic matter transport in the lower Colorado River, South-Western USA. Regulated Rivers: Research and Management. 8:323-334.

Likens, G.E. 1975. Primary productivity of inland aquatic ecosystems. In: Lieth, H. and R.H. Whittaker (ed.). Primary Productivity of the Biosphere. Springer-Verlag, New York. pp. 185-202.

Maciolek, J.A. 1966. Abundance and character of microseston in a California mountain stream. Verh. Int. Ver. Limnol. 16:639-645.

Naiman, R.J. and Sedell, J.R. 1979. Characterization of particulate organic matter transported by some cascade mountain streams. J. Fish. Res. Board Can. 36:17-31.

Orcutt, J.D. Jr. and M.L. Pace. 1984. Seasonal dynamics of rotifer and crustacean zooplankton populations in a eutrophic, monomictic lake with a note on rotifer sampling techniques. Hydrobiologia. 119:73-80.

Paulson, L.J. 1981. Nutrient management with hydroelectric dams on the Colorado River system. Lake Mead Limnol. Res. Cen. Tech. Rept. 8. Univ. Nevada, Las Vegas, NV. 39pp.

Soballe , D.M. and R.W. Bachmann. 1984. Influence of reservoir transit on riverine algae transport and abundance. Can. J. Fish. Aquat. Sci. 37:130-137.

Strickland , J.D.H. and T.R. Parsons. 1968. A Practical Handbook of Seawater Analysis. Fish.Res. Board of Canada, Ottawa, Canada.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. and Cushing, C.E. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37L130-137.

Ward, J.V. 1975. Downstream fate of zooplankton from a hypolimnial release mountain reservoir. Verh. Int. Ver. Limnil. 19:1798-1804.

Webster, J.R., Benfield, E.F., and Cairns, J., Jr. 1979. 'Model predictions of effects of impoundments on particulate organic matter transport in a river system' in Ward, J.V. and Stanford, J.A. (Eds). The Ecology of Regulated Streams. Plenum Press, New York. pp. 339-364.

Wetzel, R.G. 1983. Limnology. W.B.Saunders Co. New York.

Table 1. UTM coordinates of sampling stations in Shasta Lake.

Station	Description	UTM Coordinates	
S6	Main Lake	0550316E	4510493N
S7	Sacramento	0552009E	4518078N
S8	McCloud	0559229E	4515403N
S9	Pit	0562344E	4512933N
S10	Pit-McCloud	0557367E	4512563N
S11	Squaw Creek	0568062E	4515311N
S12	Upper Pit	0568825E	4510733N
S13	Upper Upper Pit	0574026E	4513335N
S14	Upper McCloud	0564616E	4525048N
S15	Upper Sacramento	0551101E	4523261N
S16	Forebay	0549686E	4508330N

Table 2. Phytoplankton species collected from stations S6, S7, S8, S9, and S10.

Phytoplankton Species	Division
Achnanthes spp	Bacillariophyta
Amphora ovalis	Bacillariophyta
Amphora spp	Bacillariophyta
Asterionella formosa	Bacillariophyta
Asterionella spp	Bacillariophyta
Caloneis limosa	Bacillariophyta
Cocconeis placenta	Bacillariophyta
Cocconeis placentula	Bacillariophyta
Cocconeis spp	Bacillariophyta
Cyclotella bodanica	Bacillariophyta
Cyclotella meneghiniona	Bacillariophyta
Cyclotella operculata	Bacillariophyta
Cyclotella spp	Bacillariophyta
Cyclotella stelligera	Bacillariophyta
Cymatopleura elliptica	Bacillariophyta
Cymatopleura solea	Bacillariophyta
Cymatopleura spp.	Bacillariophyta
Cymbella lanceolata	Bacillariophyta
Cymbella minuta	Bacillariophyta
Cymbella spp	Bacillariophyta
Cymbella tumida	Bacillariophyta
Diatoma spp.	Bacillariophyta
Diatomella spp	Bacillariophyta
Diploneis smithii	Bacillariophyta
Epithemia spp.	Bacillariophyta
Eunotia pectinalis	Bacillariophyta
Eunotia spp	Bacillariophyta
Fragilaria construens	Bacillariophyta
Fragilaria crotonensis	Bacillariophyta
Fragilaria spp	Bacillariophyta
Gomphonema constrictum	Bacillariophyta
Gomphonema olivaceum	Bacillariophyta
Gomphonema spp	Bacillariophyta
Gyrosigma spp.	Bacillariophyta
Melosira granulata	Bacillariophyta
Melosira islandica	Bacillariophyta

Melosira italica	Bacillariophyta
Melosira spp	Bacillariophyta
Melosira varians	Bacillariophyta
Meridion circulare	Bacillariophyta
Navicula cryptocephala	Bacillariophyta
Navicula radiosa	Bacillariophyta
Navicula spp	Bacillariophyta
Nitzschia acicularis	Bacillariophyta
Nitzschia denticula	Bacillariophyta
Nitzschia linearis	Bacillariophyta
Nitzschia sigmoidea	Bacillariophyta
Nitzschia spp	Bacillariophyta
Opephora spp	Bacillariophyta
Pinnularia obscura	Bacillariophyta
Pinnularia spp.	Bacillariophyta
Pleurosigma delicatulum	Bacillariophyta
Rhoicosphenia spp	Bacillariophyta
Rhopalodia gibba	Bacillariophyta
Rhopalodia gibbera	Bacillariophyta
Rhopalodia gibberula	Bacillariophyta
Stephanodiscus hantzschii	Bacillariophyta
Stephanodiscus spp	Bacillariophyta
Surirella angustata	Bacillariophyta
Surirella ovalis	Bacillariophyta
Surirella ovata	Bacillariophyta
Surirella robusta	Bacillariophyta
Surirella spp	Bacillariophyta
Synedra delicatissima	Bacillariophyta
Synedra delicatissima var. angustissima	Bacillariophyta
Synedra pulchella	Bacillariophyta
Synedra spp	Bacillariophyta
Synedra ulna	Bacillariophyta
Tabellaria fenestrata	Bacillariophyta
Tabellaria flocculosa	Bacillariophyta
Tabellaria spp	Bacillariophyta
Unidentified centric diatom	Bacillariophyta
Unidentified pennate diatom	Bacillariophyta
Ankistrodesmus spp.	Chlorophyta
Chlamydomonas spp.	Chlorophyta
Cladophora spp.	Chlorophyta
Closterium acerosum	Chlorophyta
Closterium spp	Chlorophyta
Coelastrum microporum	Chlorophyta

Dictyosphaerium pulchellum	Chlorophyta
Geminella mutabilis	Chlorophyta
Kirchneriella spp.	Chlorophyta
Mougeotia parvula	Chlorophyta
Mougeotia spp.	Chlorophyta
Oocystis pusilla	Chlorophyta
Oocystis spp	Chlorophyta
Palmodictyon spp.	Chlorophyta
Pandorina spp	Chlorophyta
Pediastrum duplex	Chlorophyta
Quadrigula lacustris	Chlorophyta
Rhizoclonium hieroglyphicum	Chlorophyta
Rhizoclonium spp	Chlorophyta
Scenedesmus bernardii	Chlorophyta
Scenedesmus bijuga	Chlorophyta
Scenedesmus quadricauda var. longispina	Chlorophyta
Scenedesmus spp	Chlorophyta
Schizomeris spp.	Chlorophyta
Sorastrum spinulosum	Chlorophyta
Sphaerocystis schroeteri	Chlorophyta
Spirogyra parvula	Chlorophyta
Spirogyra spp.	Chlorophyta
Staurastrum longiradiatum	Chlorophyta
Staurastrum paradoxum	Chlorophyta
Stigeoclonium lubricum	Chlorophyta
Stigeoclonium polymorphum	Chlorophyta
Stigeoclonium spp	Chlorophyta
Strombidium oculatum	Chlorophyta
Ulothrix lubricum	Chlorophyta
Ulothrix spp	Chlorophyta
Unidentified coccoid green	Chlorophyta
Unidentified filament	Chlorophyta
Dinobryon spp	Chrysophyta
Mallomonas pseudocoronata	Chrysophyta
Cryptomonas spp	Cryptophyta
Rhodomonas minuta	Cryptophyta
Rhodomonas minuta var nannoplanctica	Cryptophyta
Anabaena spp.	Cyanophyta
Chroococcus spp	Cyanophyta
Microcystis aeruginosa	Cyanophyta
Oscillatoria spp.	Cyanophyta
Stigonema spp	Cyanophyta

Unidentified blue-green filament	Cyanophyta
Euglena gracilis	Euglenophyta
Euglena spp.	Euglenophyta
Phacus spp.	Euglenophyta
Trachelomonas spp	Euglenophyta
Ceratium hirundella	Pyrrophyta
Ceratium hirundinella	Pyrrophyta
Glenodinium spp.	Pyrrophyta
Gymnodinium spp.	Pyrrophyta
Peridinium inconspicuum	Pyrrophyta
Peridinium spp.	Pyrrophyta
Peridinium willei	Pyrrophyta
Rhodochorton spp.	Rhodophyta

Table 3. Zooplankton species collected from stations S6, S7, S8, S9, and S10.

Zooplankton Species	Group
Biapertura affinis	Cladocera
Bosmina lonirostris	Cladocera
Ceriodaphnia spp.	Cladocera
Chydorus sphaericus	Cladocera
Daphnia laevis	Cladocera
Daphnia pulex	Cladocera
Diaphanasoma brachyurum	Cladocera
Leptodora kindtii	Cladocera
Diacyclops thomasi	Copepoda
Leptodiaptomus ashlandi	Copepoda
Limnocalanus macrurus	Copepoda
Asplanchna girodi	Rotifera
Brachionus havanaensis	Rotifera
Brachionus angularis	Rotifera
Colurella spp.	Rotifera
Conochilodies dossarius	Rotifera
Euchlanis dialata	Rotifera
Filinia longiseta	Rotifera
Gastropus hypotus	Rotifera
Gastropus stylifer	Rotifera
Hexarthra mira	Rotifera
Kellicottia longispina	Rotifera
Keratella cochlearis	Rotifera
Keratella quadrata	Rotifera
Lecane spp.	Rotifera
Platyias quadricornia	Rotifera
Polyarthra vulgaris	Rotifera
Pompholyx sulcata	Rotifera
Trichocerca spp.	Rotifera
Synchaeta pectinata	Rotifera

Figure 1. Seasonal patterns of inflow (cfs), discharge (cfs) and reservoir elevation (Feet above msl) for Shasta Lake, CA, 1995-1997

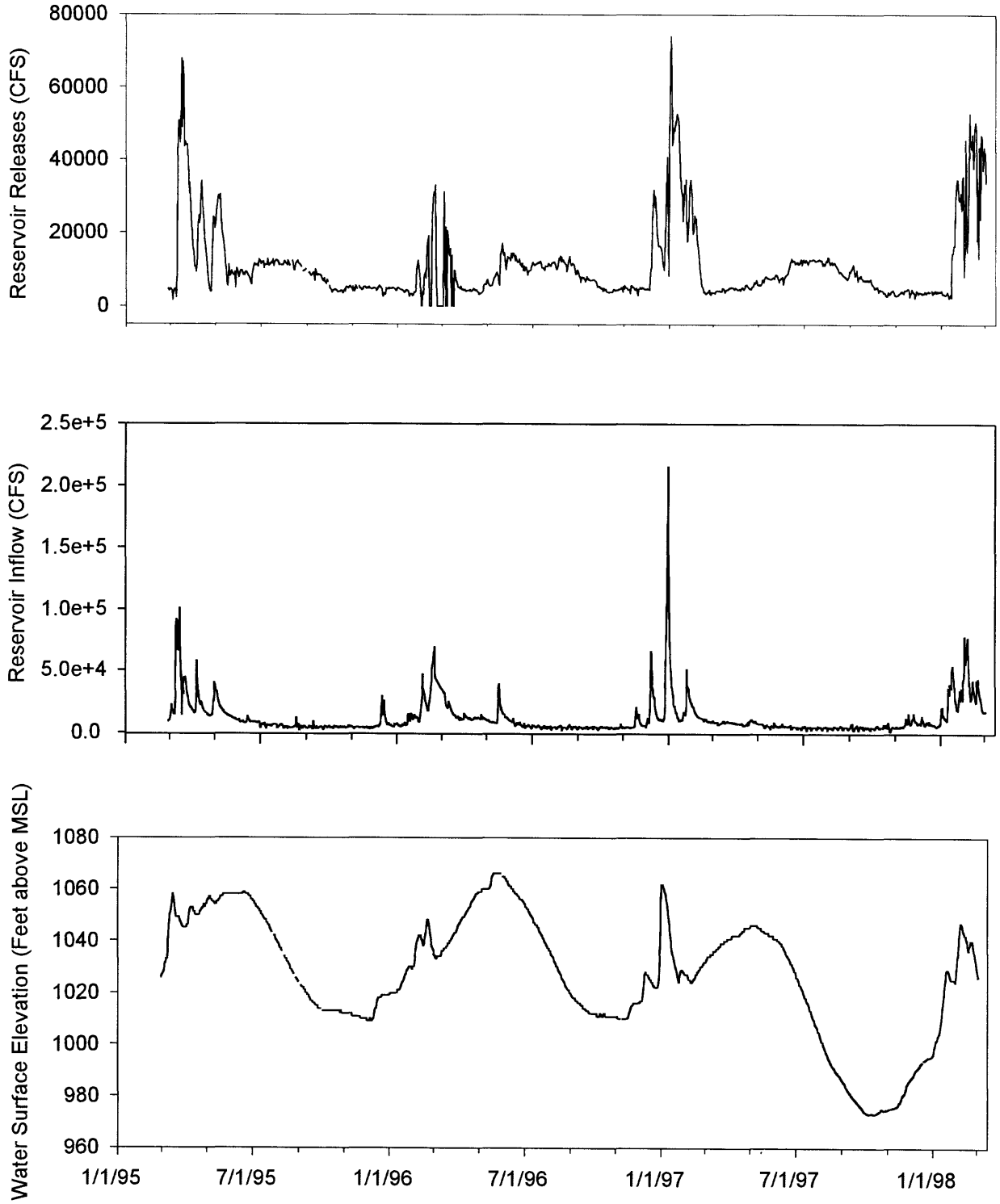


Figure 2. Map of limnological sampling stations in Shasta Lake. Main lake (S6), Sacramento River Arm (S7, S15), McCloud River Arm (S8 and S14), Pit River Arm (S9, S12, S13), Squaw Creek Arm (S11), and Pit-McCloud Confluence (S10). River stations are located in Shasta tailwaters (S0) and Keswick tailwaters (K1).

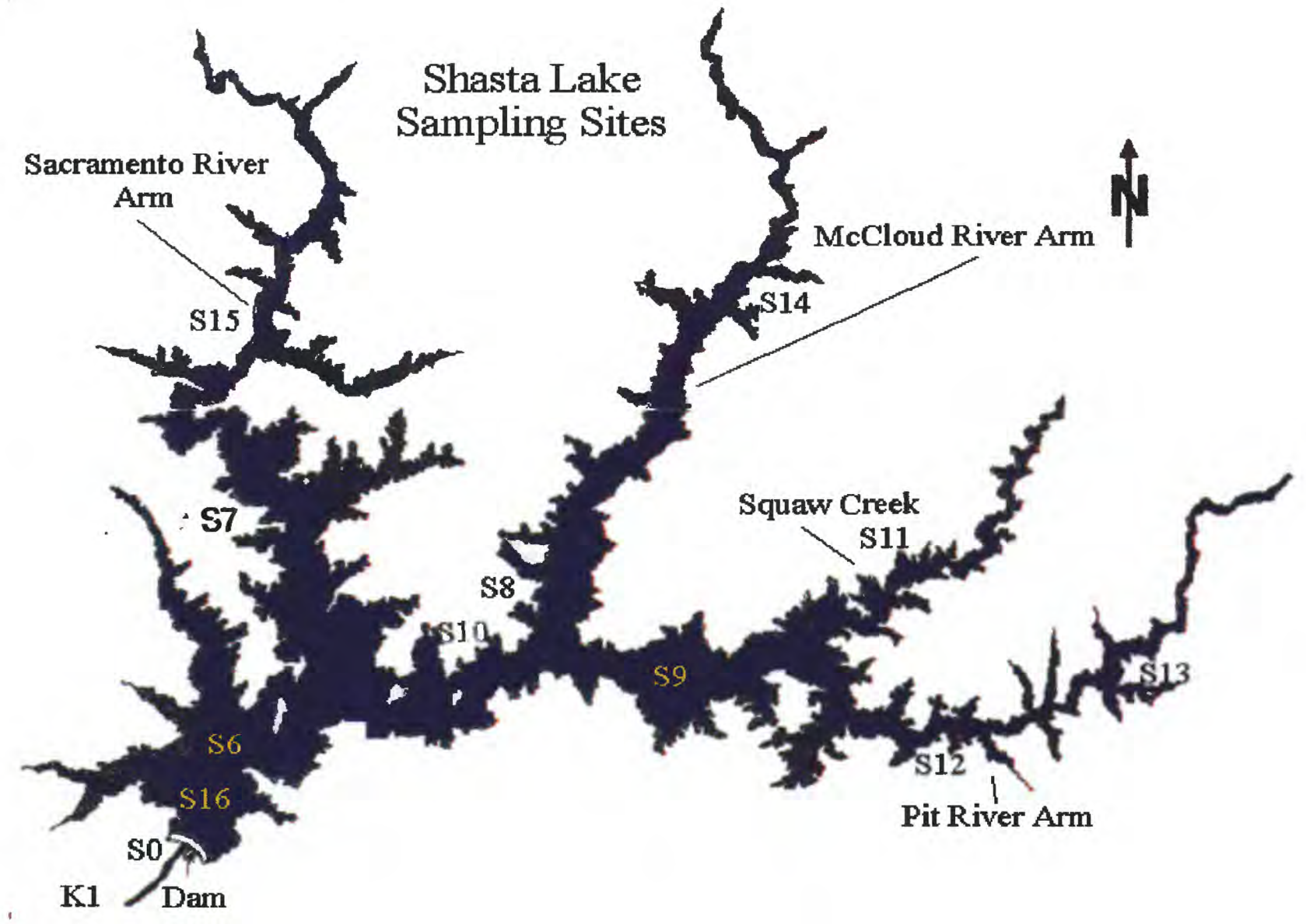


Figure 3 a-e. Time series isopleths of dissolved oxygen (mg/L), temperature (^{0}C) and specific conductance (μS/cm) for Shasta Lake sampling stations. a. S6; b. S7; c. S8; d. S9; e. S10

a.

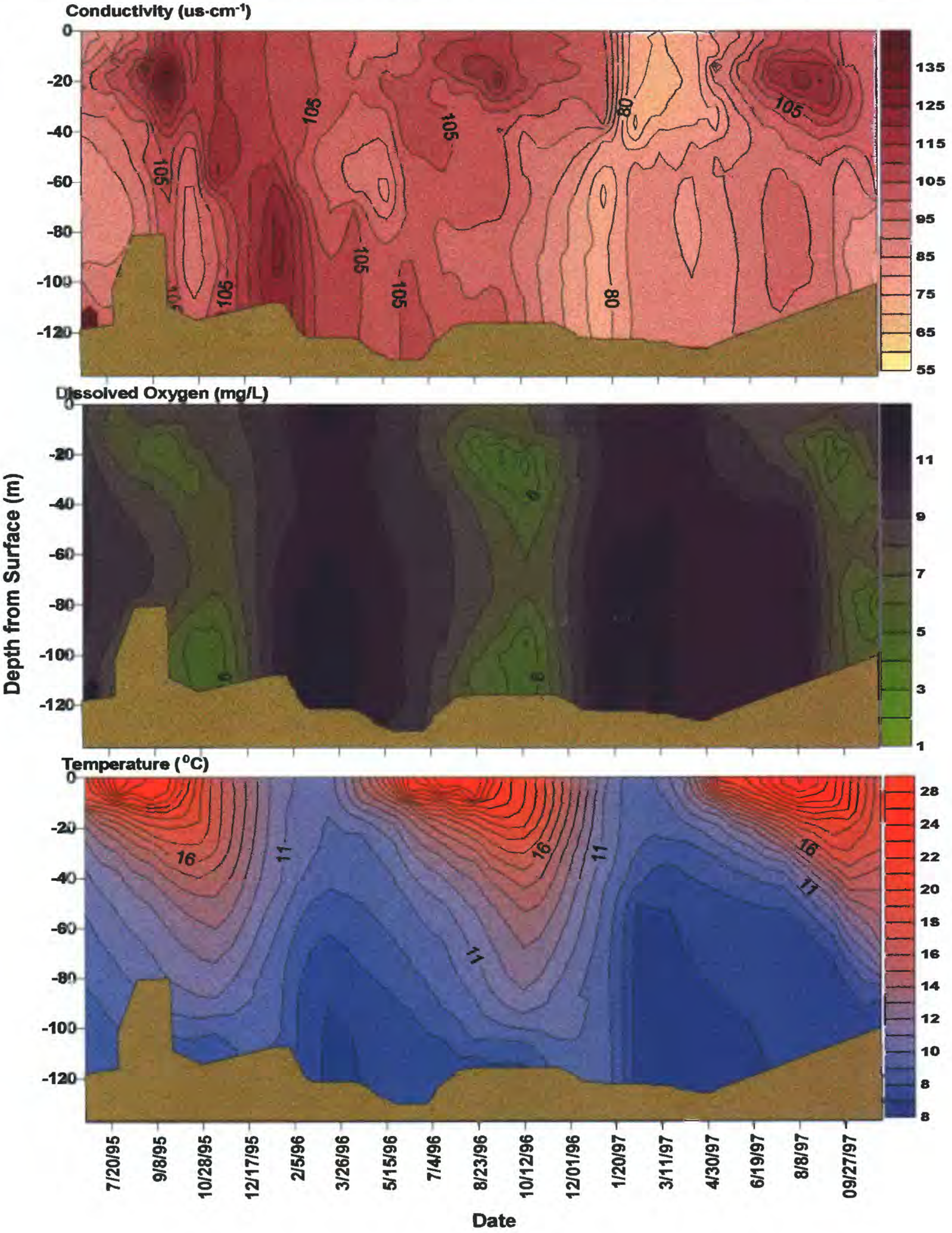


Figure 3 cont.
b.

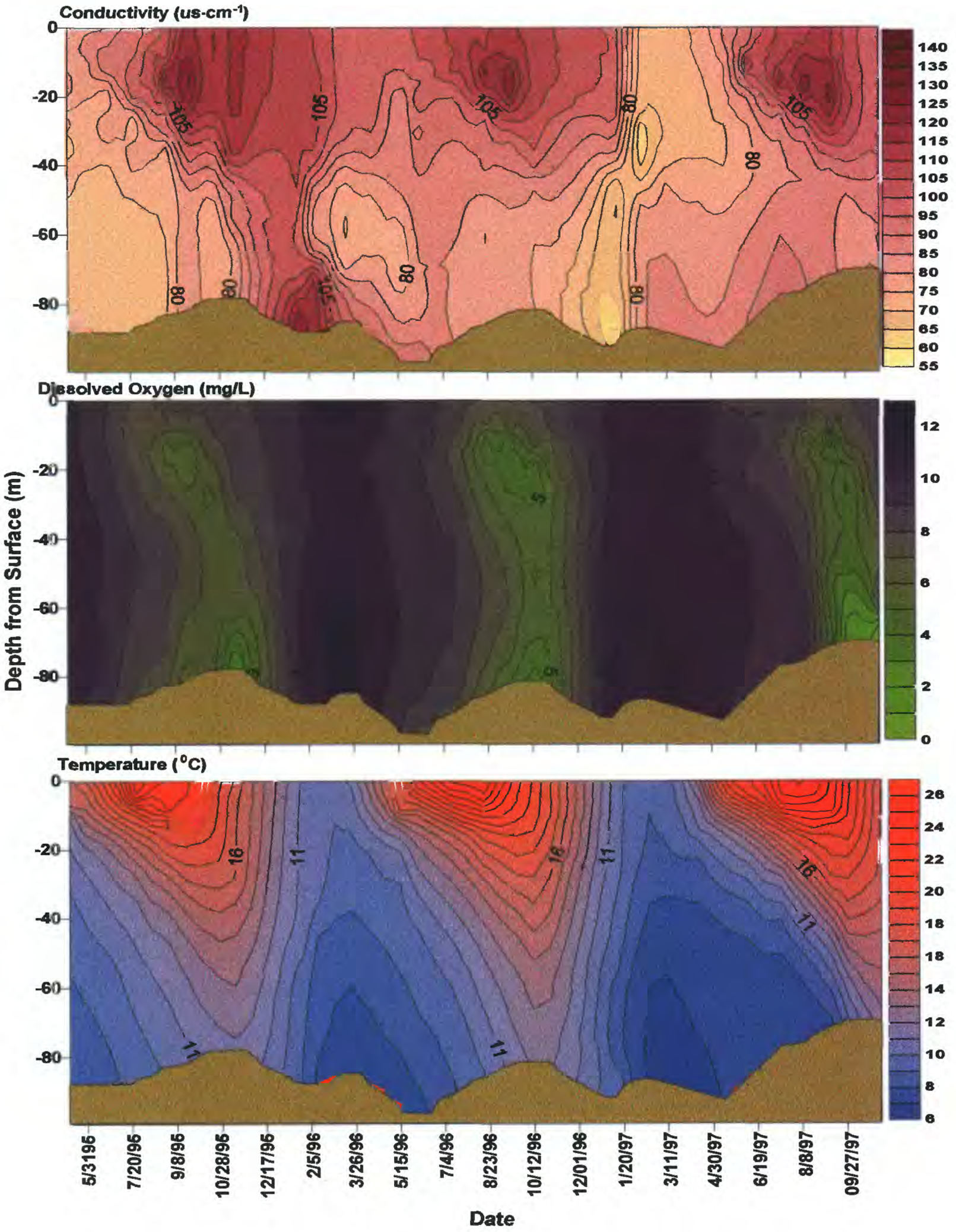


Figure 3 cont.
c.

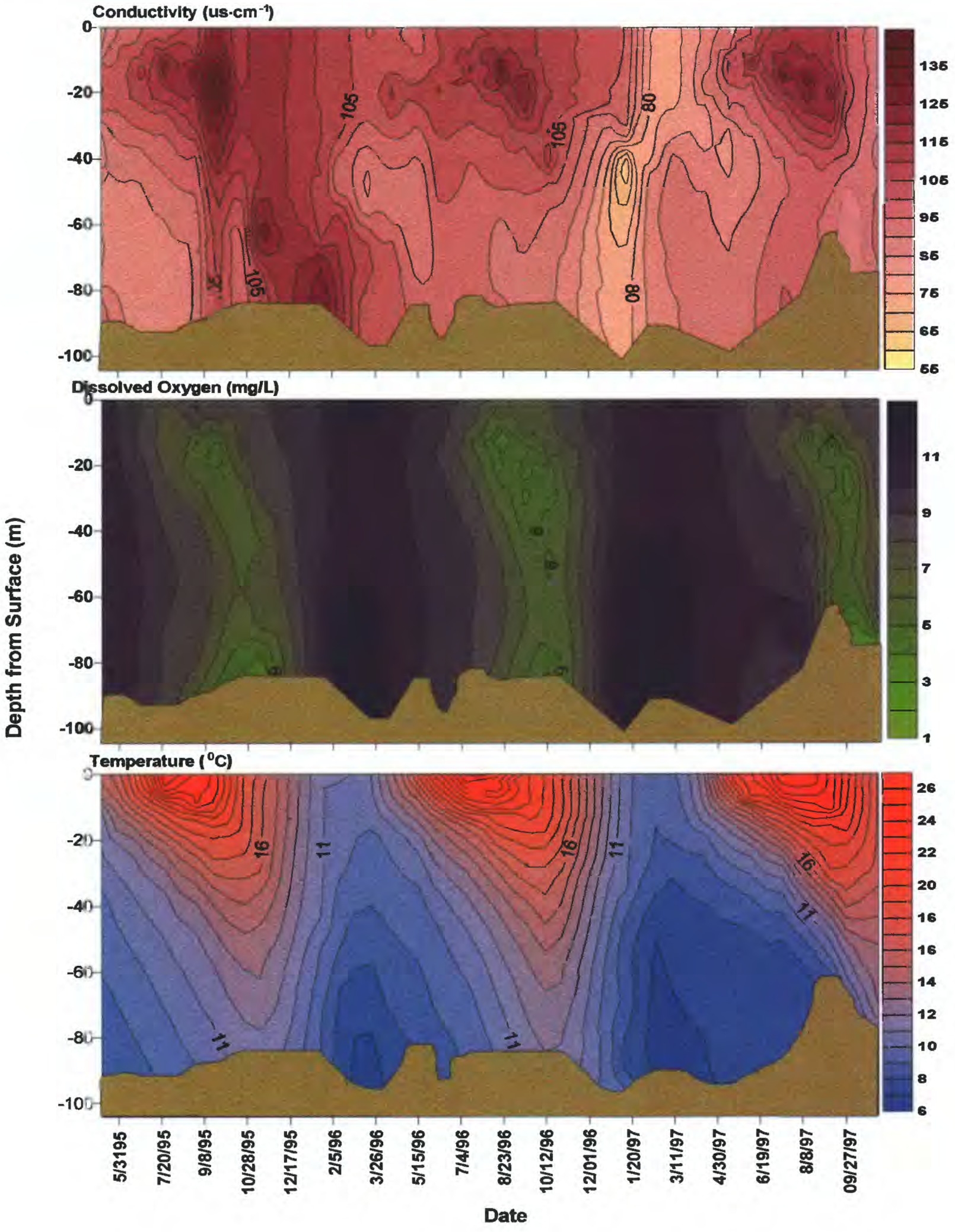


Figure 3 cont.
d.

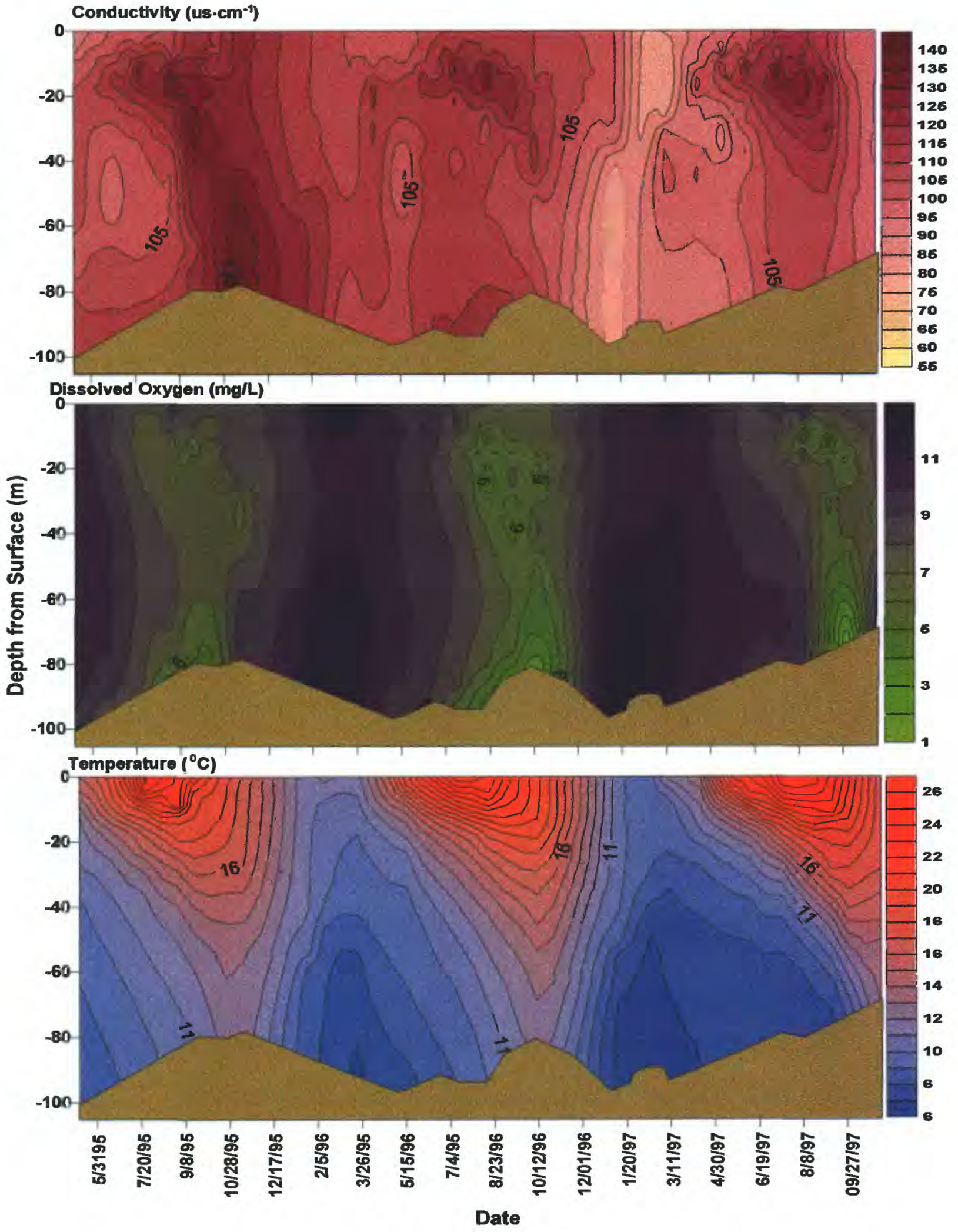


Figure 3 cont.
e.

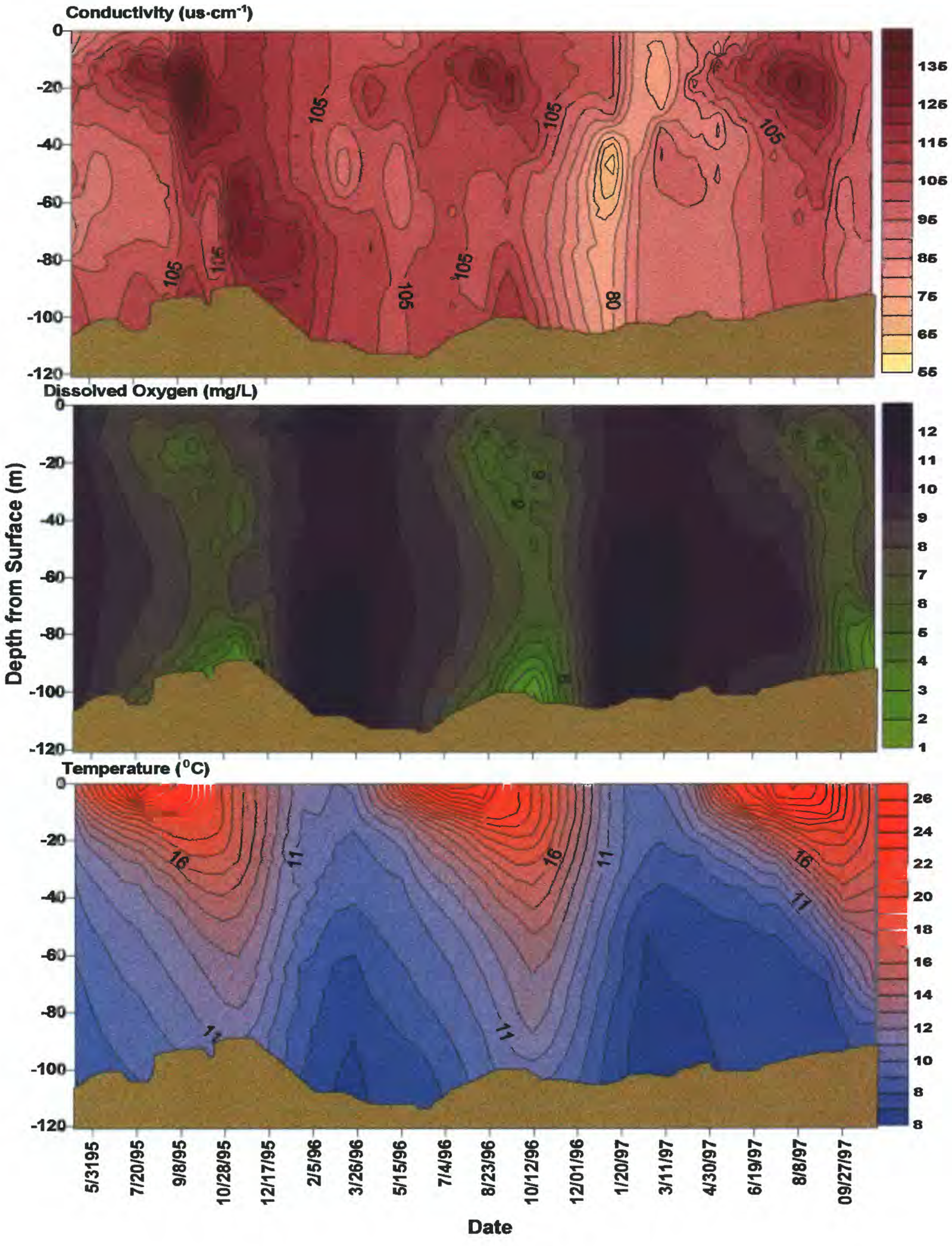


Figure 4 a-e. Time series isopleths of Relative Thermal Resistance to Mixing (RTRM) coefficients for Shasta Lake sampling stations a. S6; b. S10; c. S9; d. S8; e. S7.

a,b.

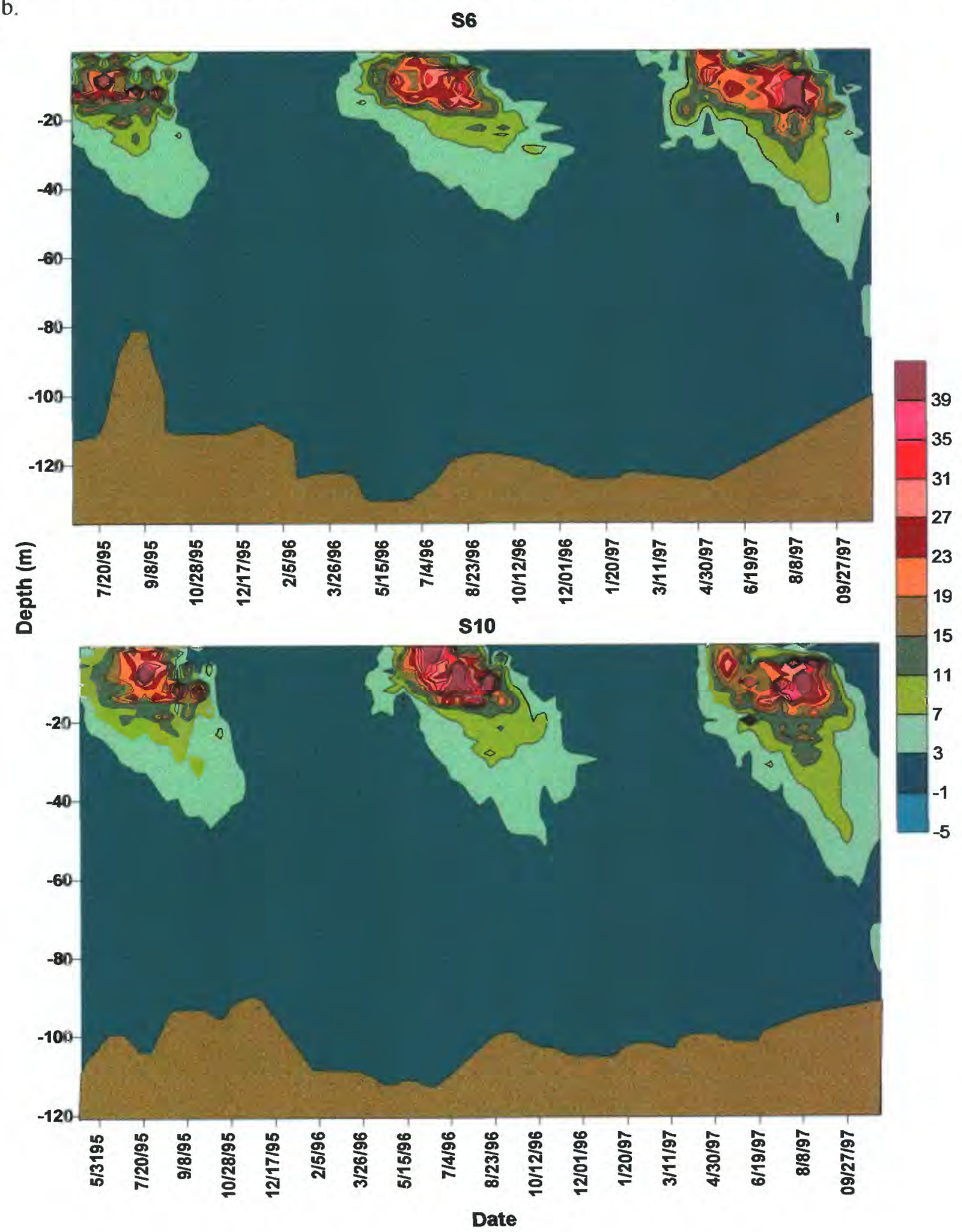


Figure 4 cont.
c,d.

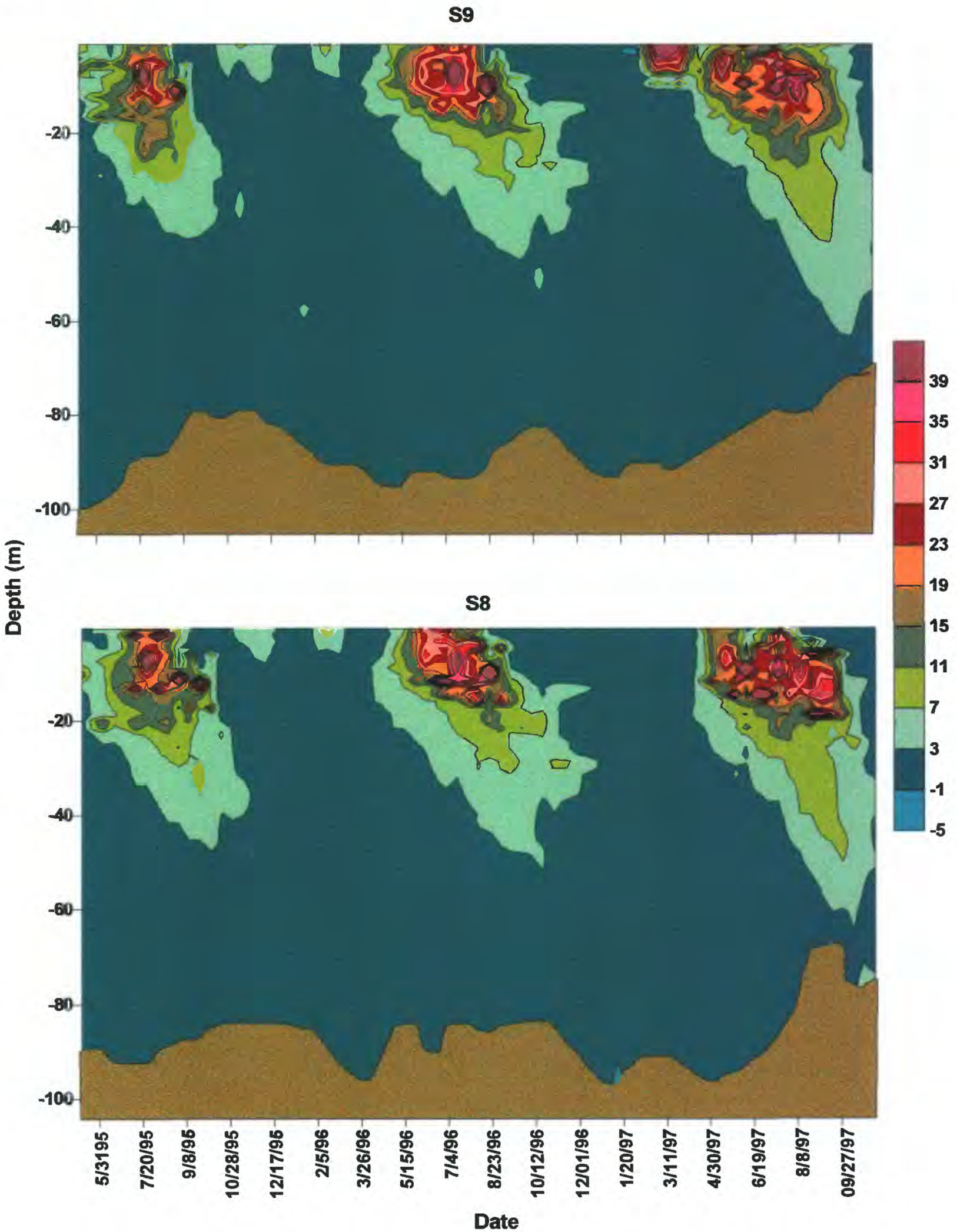


Figure 4 cont.
e.

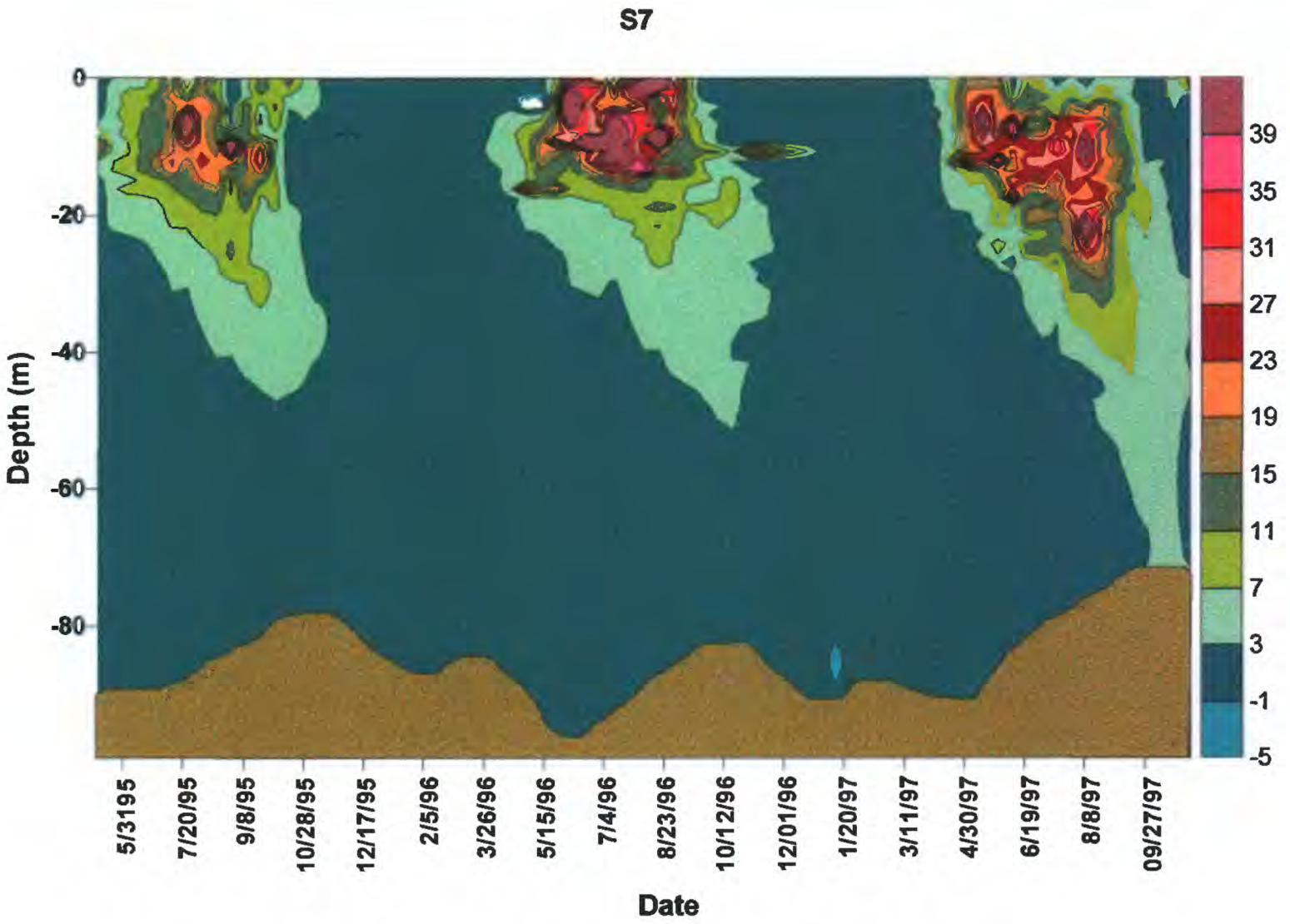


Figure 5 a-t. Monthly isopleth description of temperature (^{0}C), dissolved oxygen (percent saturation), pH, and specific conductance (μS/cm) for the Pit River arm for 1996 and 1997. a. February 1996; b. March 1996; c. April 1996; d. May 1996; e. June 1996; f. July 1996; g. August 1996; h. September 1996; i. October 1996; j. January 1997; k. February 1997; l. March 1997; m. April 1997; n. May 1997; o. June 1997; p. July 1997; q. August 1997; r. September 1997; s. October 1997; t. November 1997.

a.

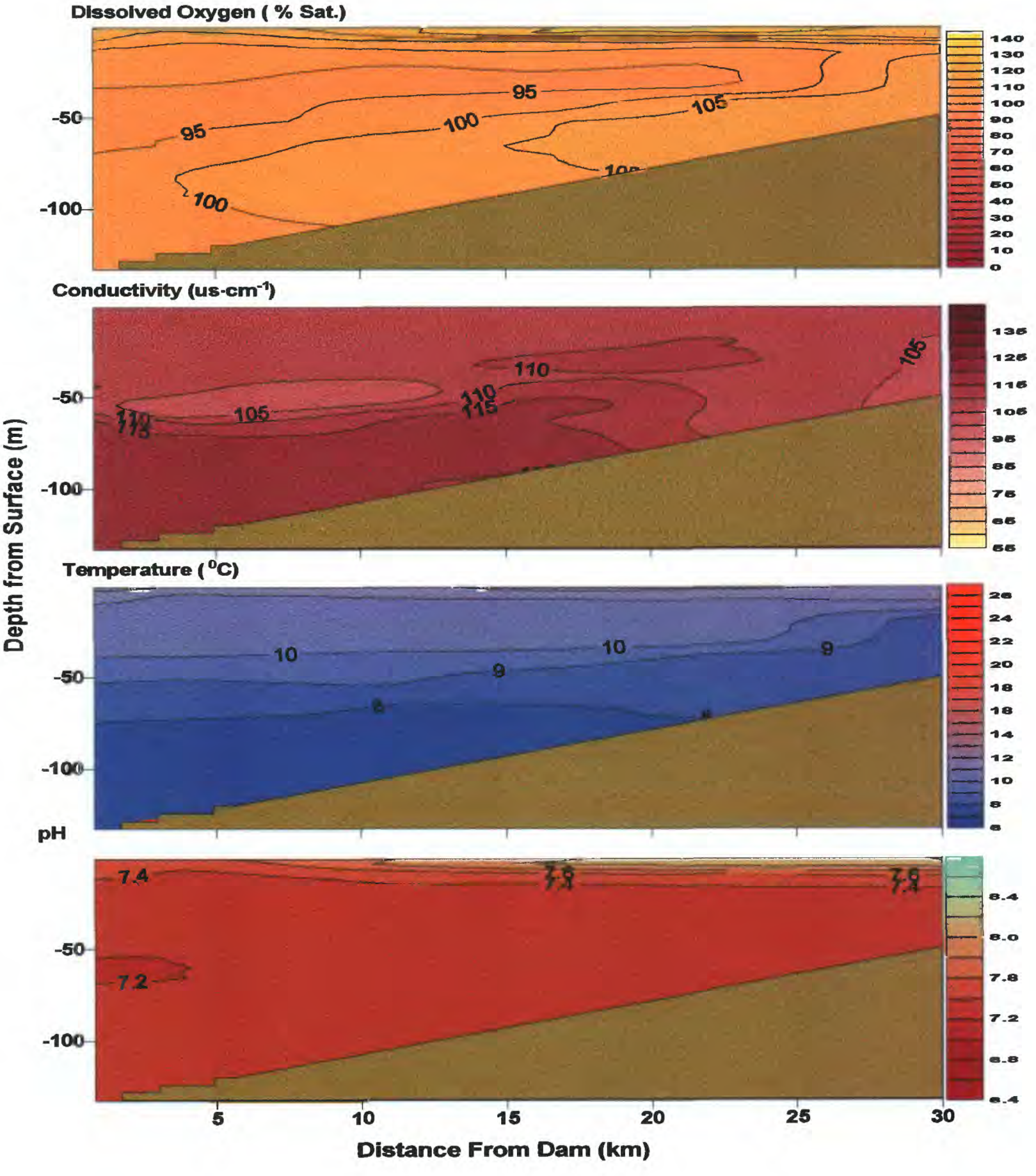


Figure 5 cont.
b.

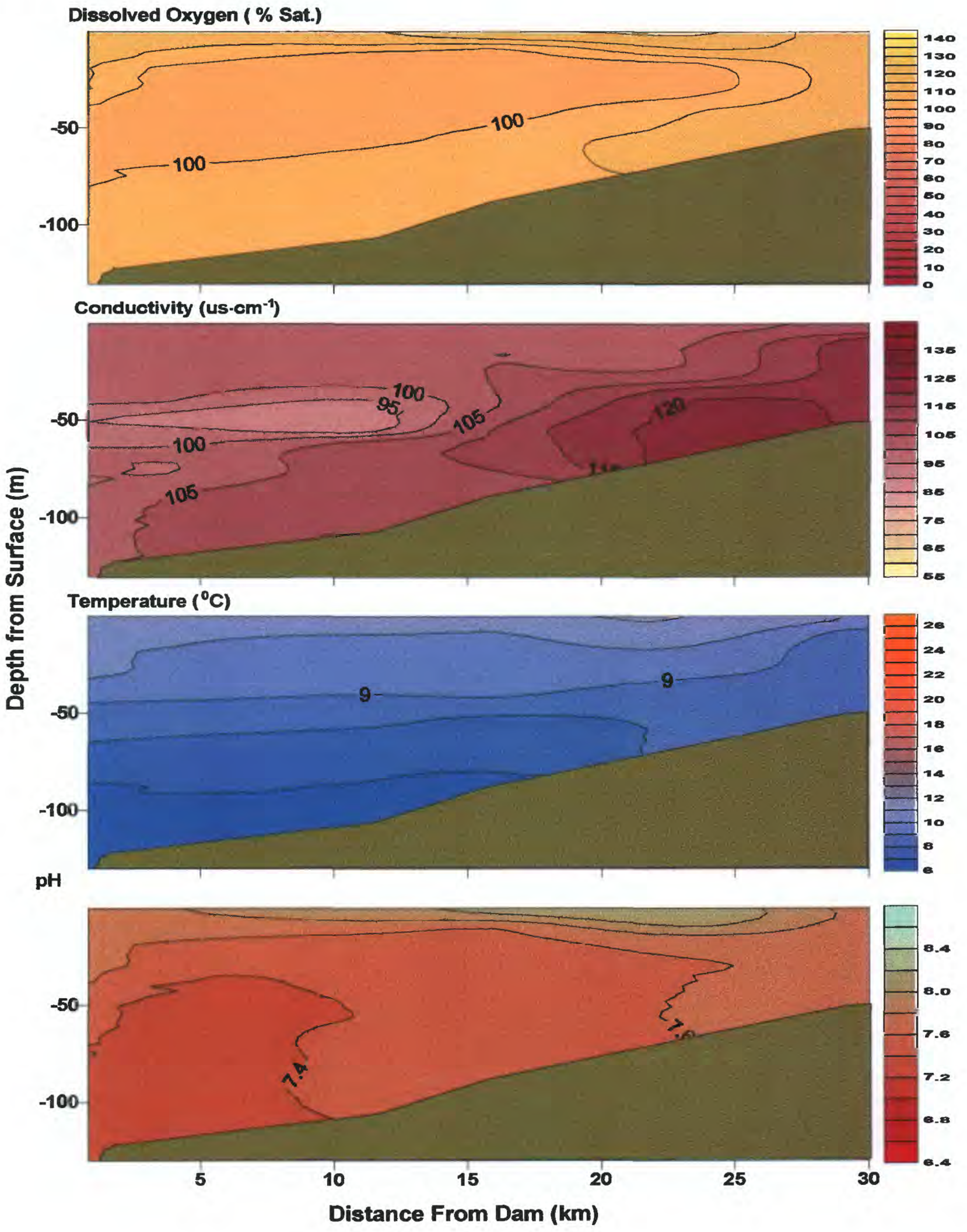


Figure 5 cont.
c.

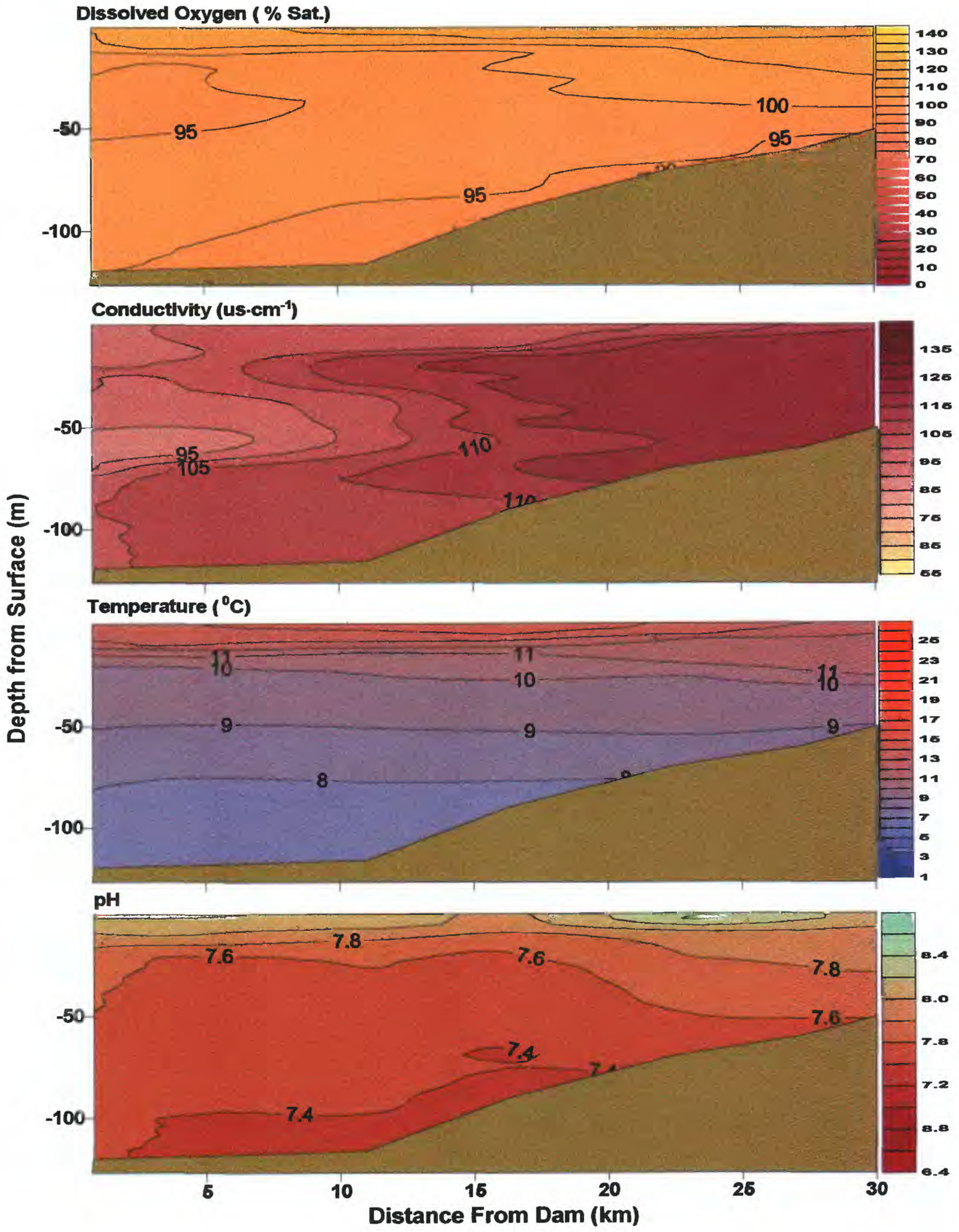


Figure 5 cont.
d.

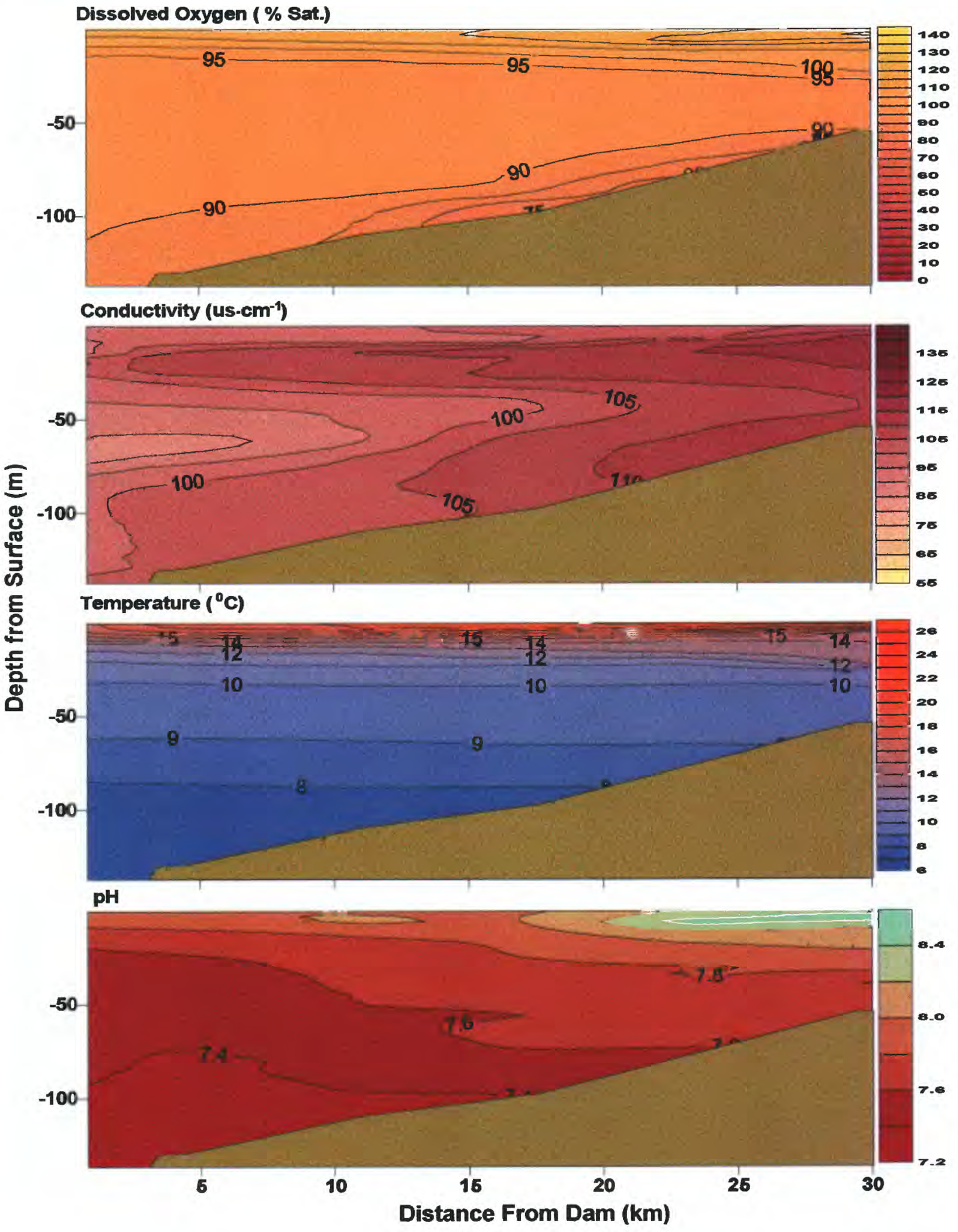


Figure 5 cont.
e.

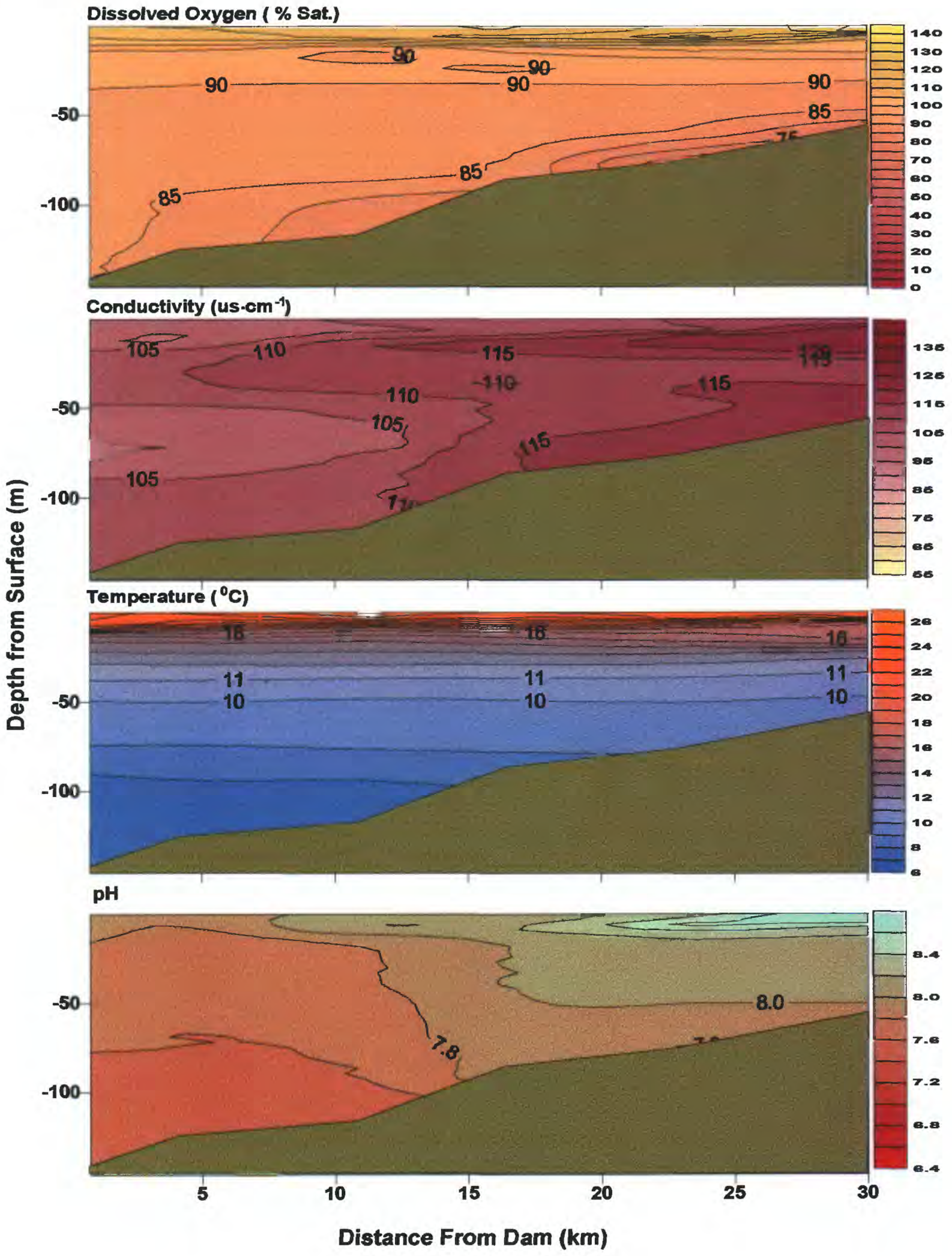


Figure 5 cont.
f.

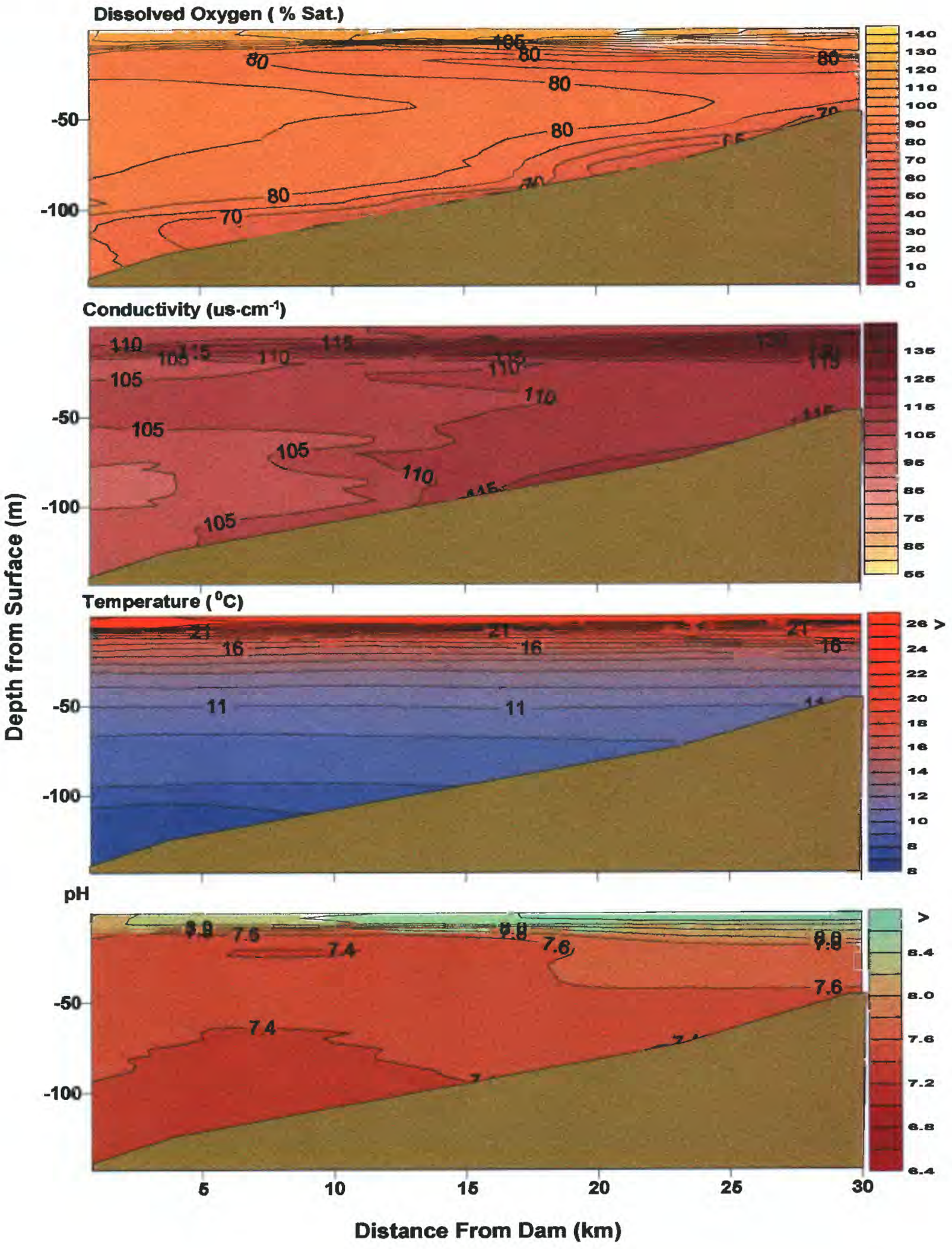


Figure 5 cont.
g.

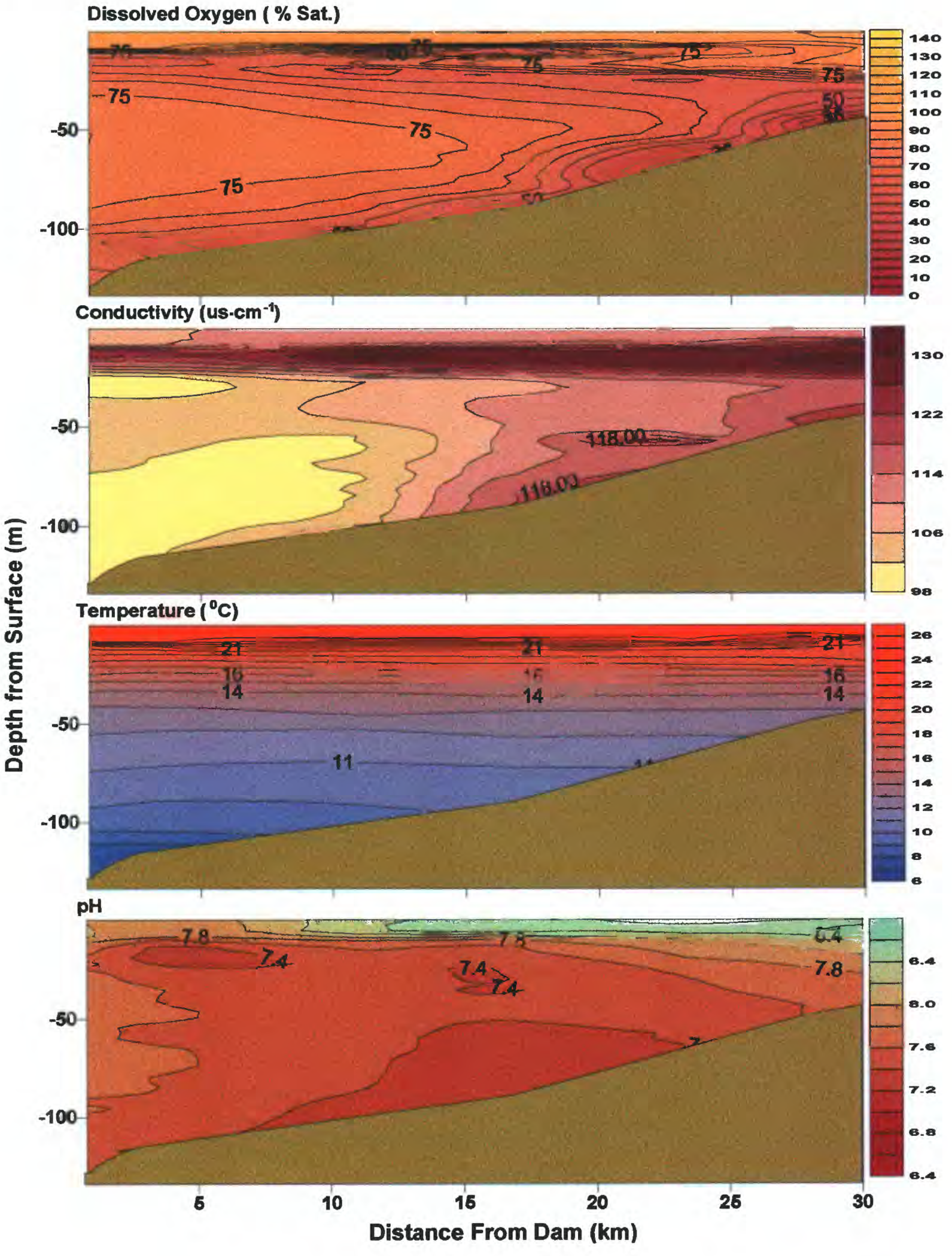


Figure 5 cont.
h.

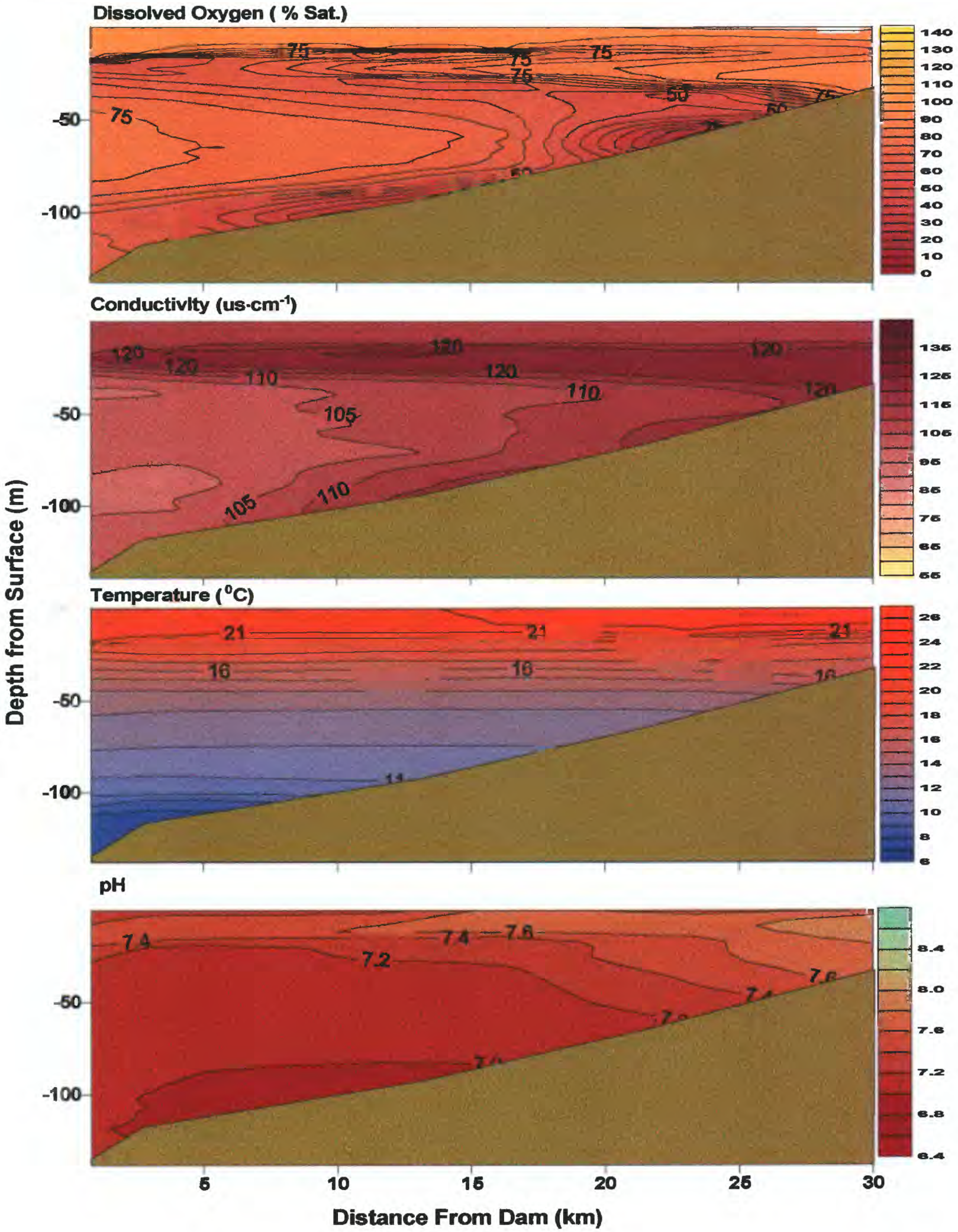


Figure 5 cont.
i.

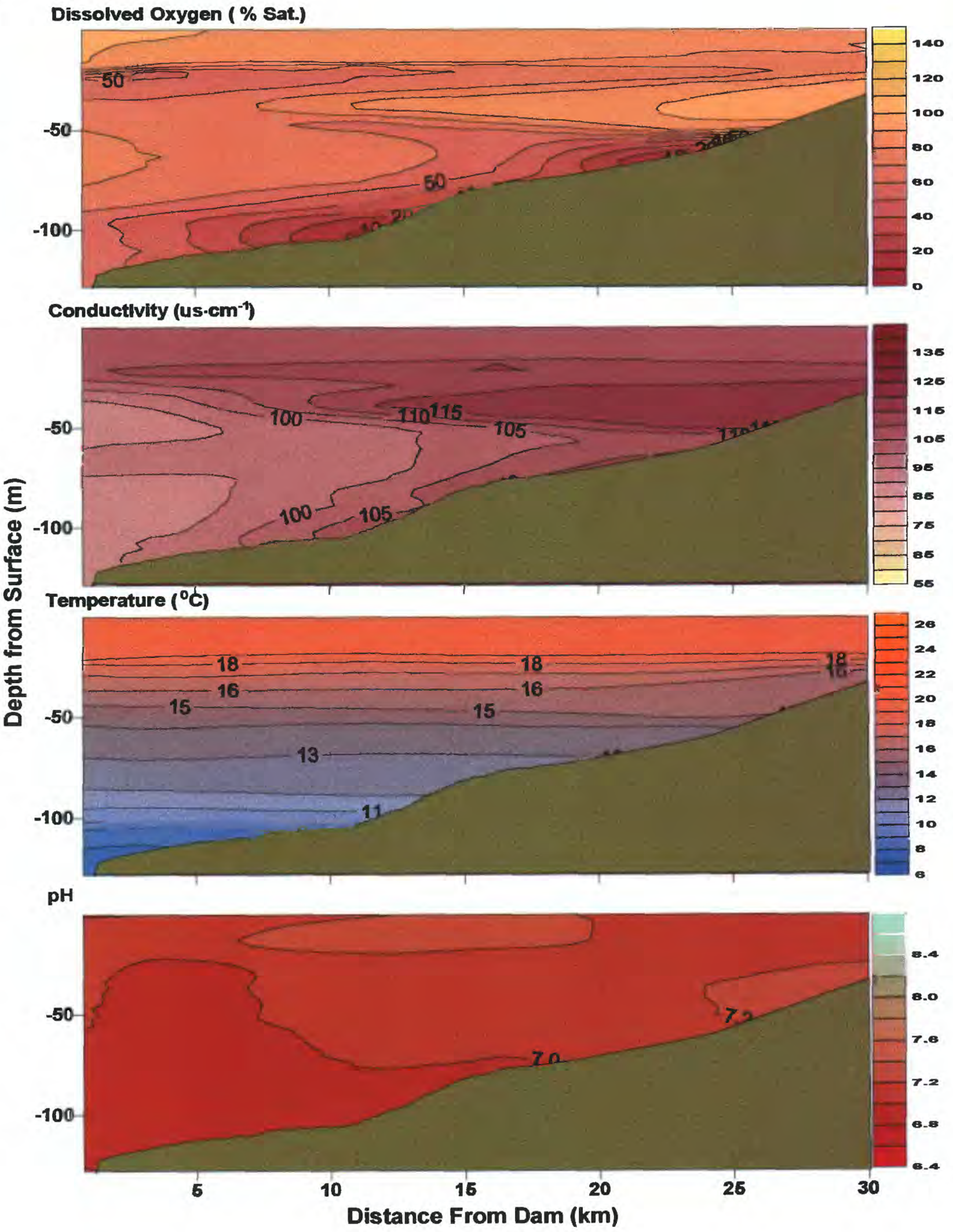


Figure 5 cont.
j.

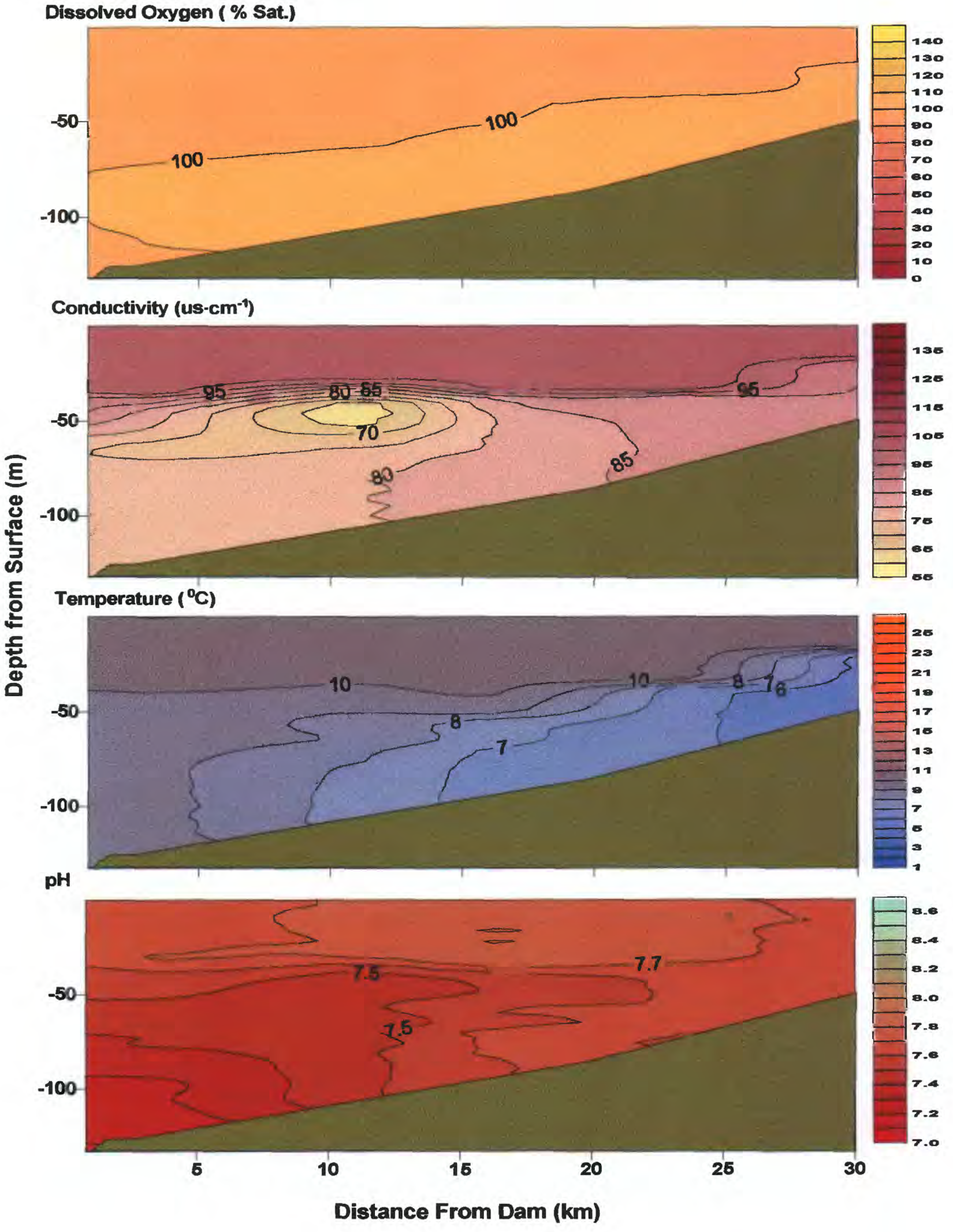


Figure 5 cont.
k.

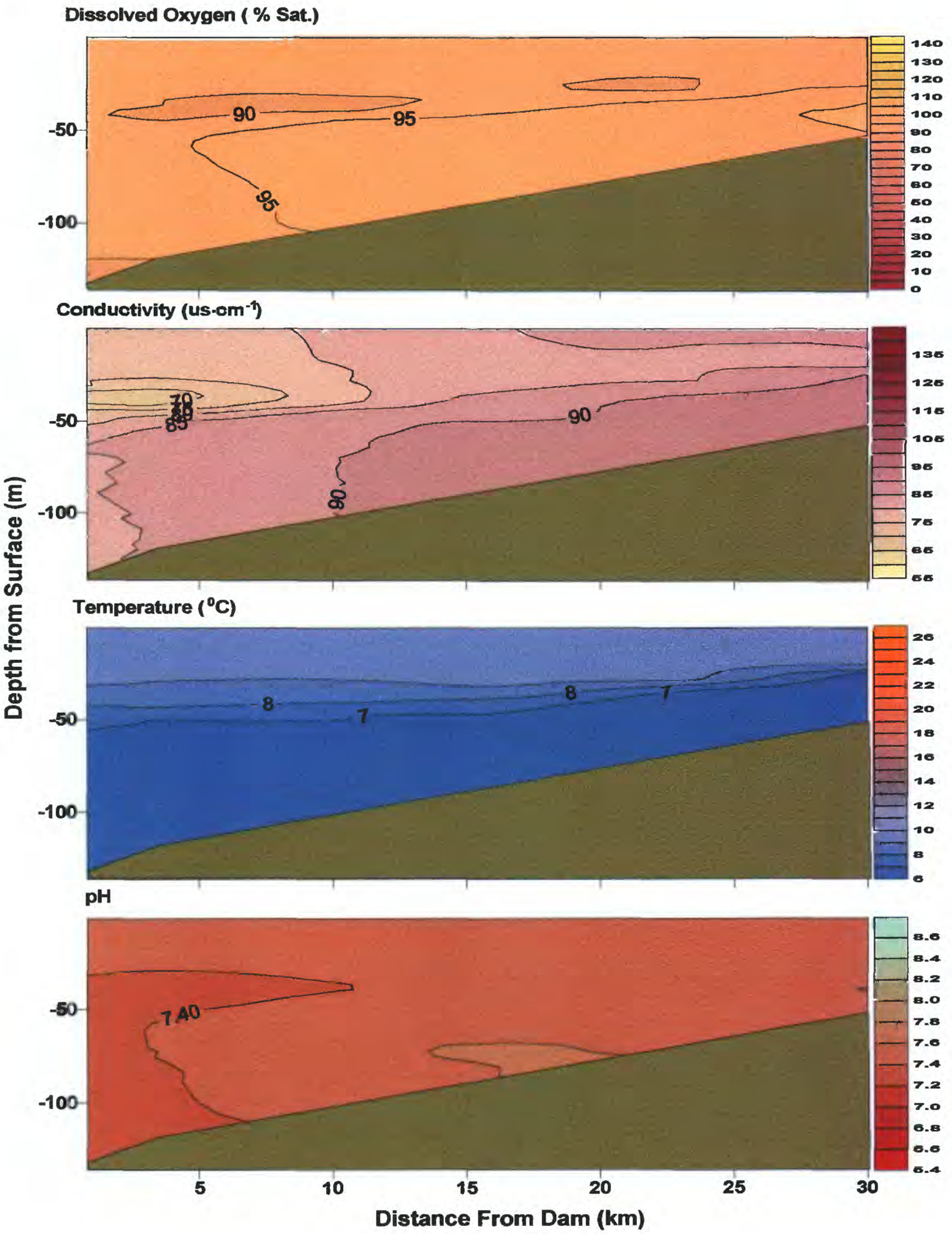


Figure 5 cont.
l.

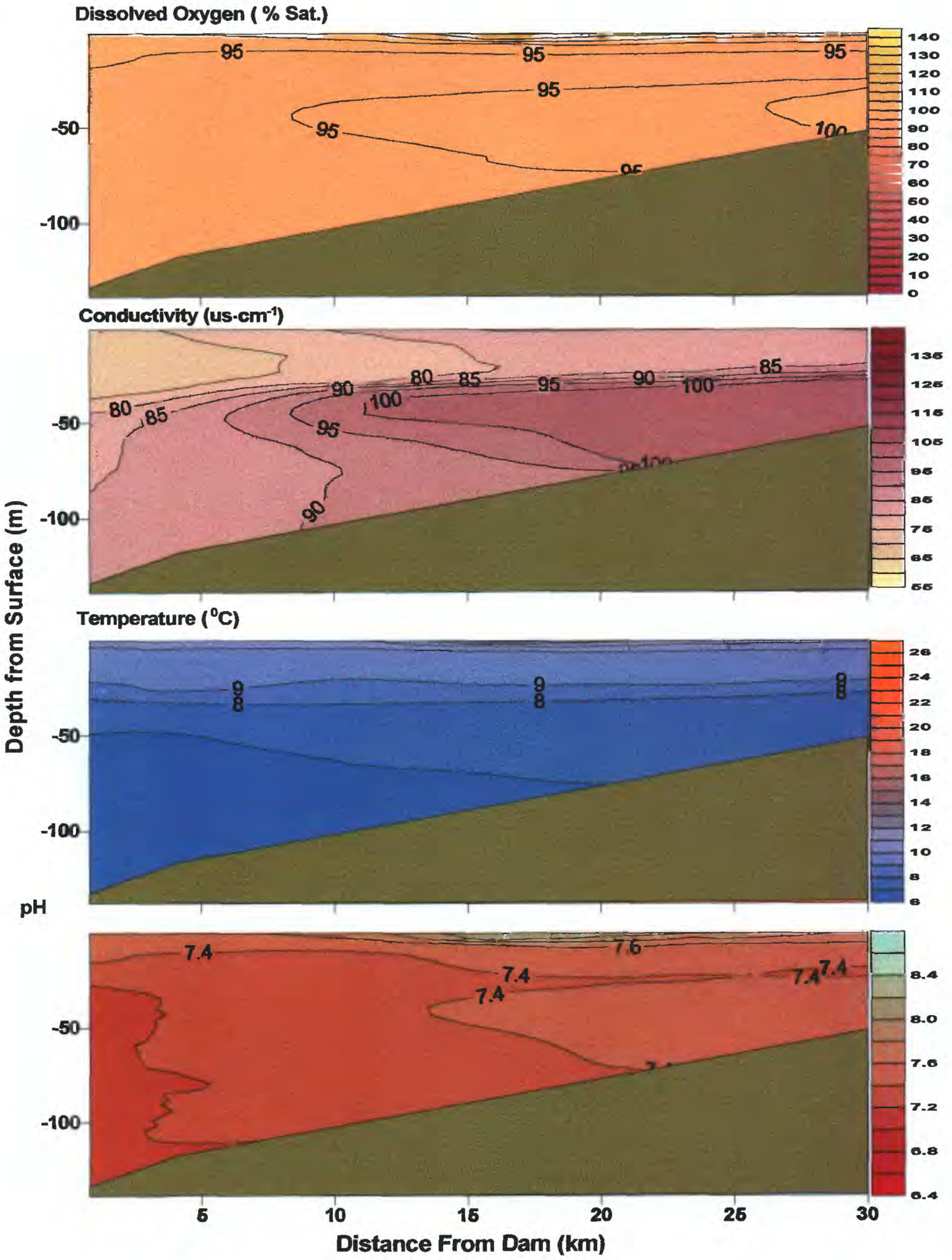


Figure 5 cont.
m.

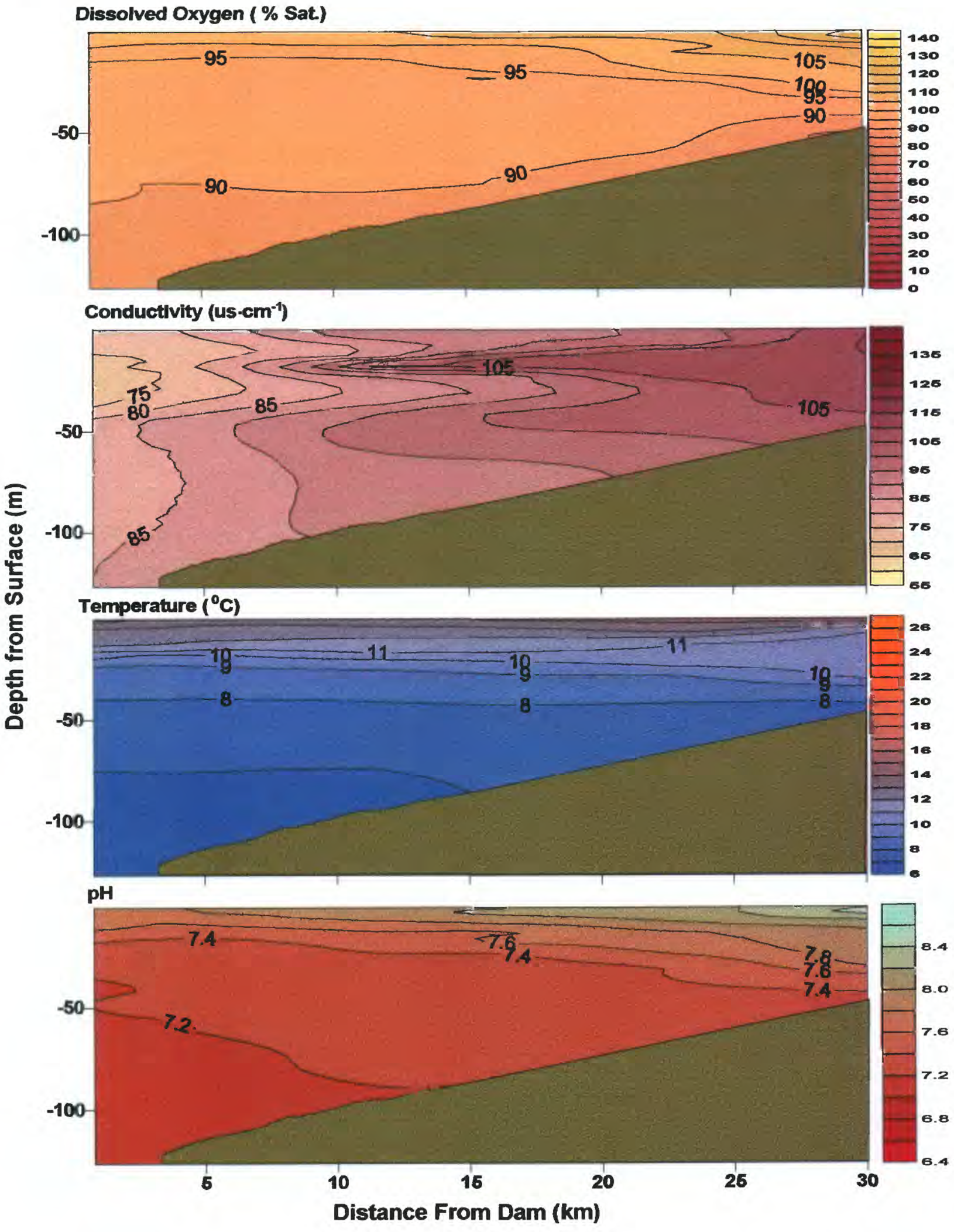


Figure 5 cont.
n.

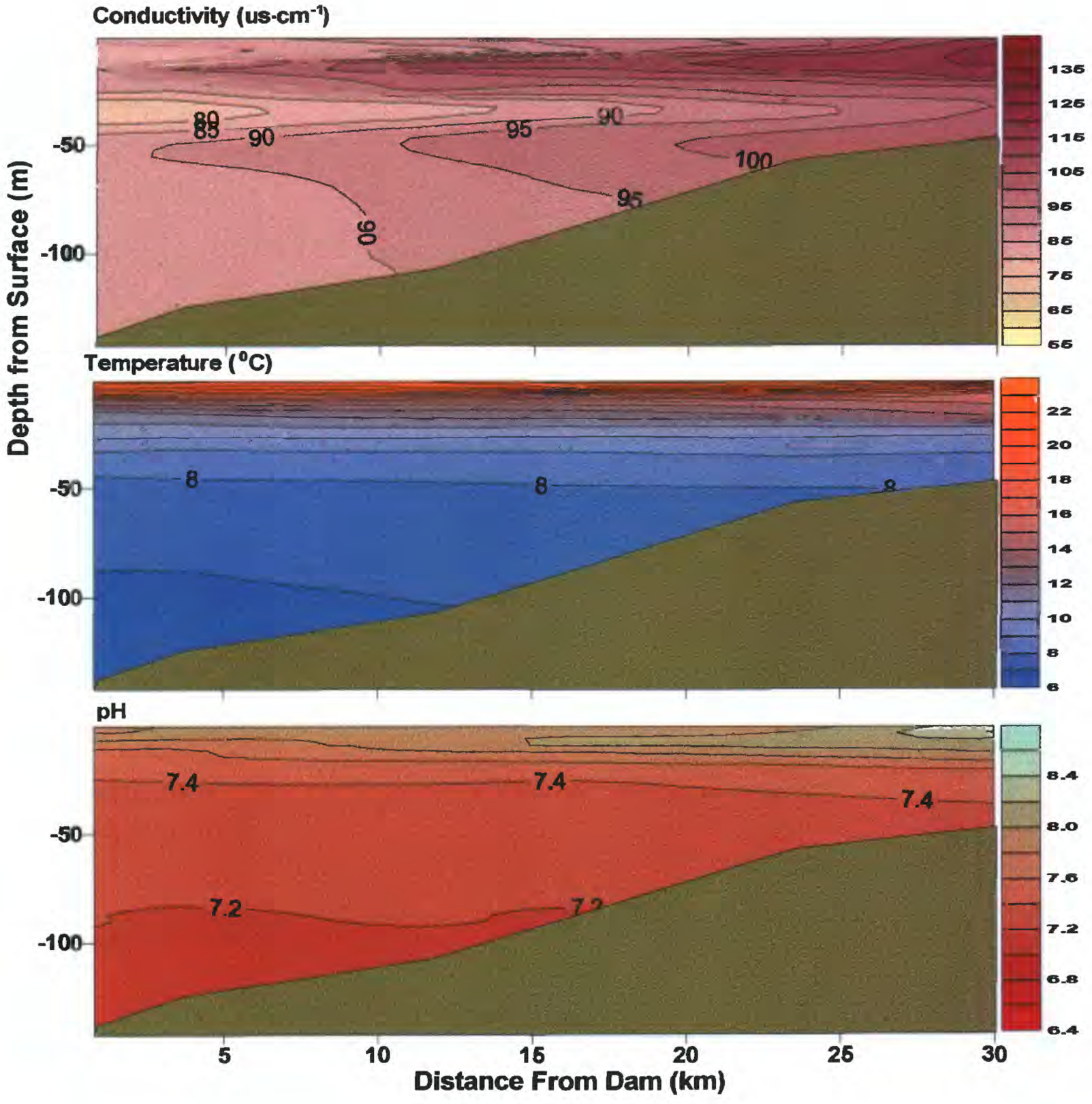


Figure 5 cont.
o.

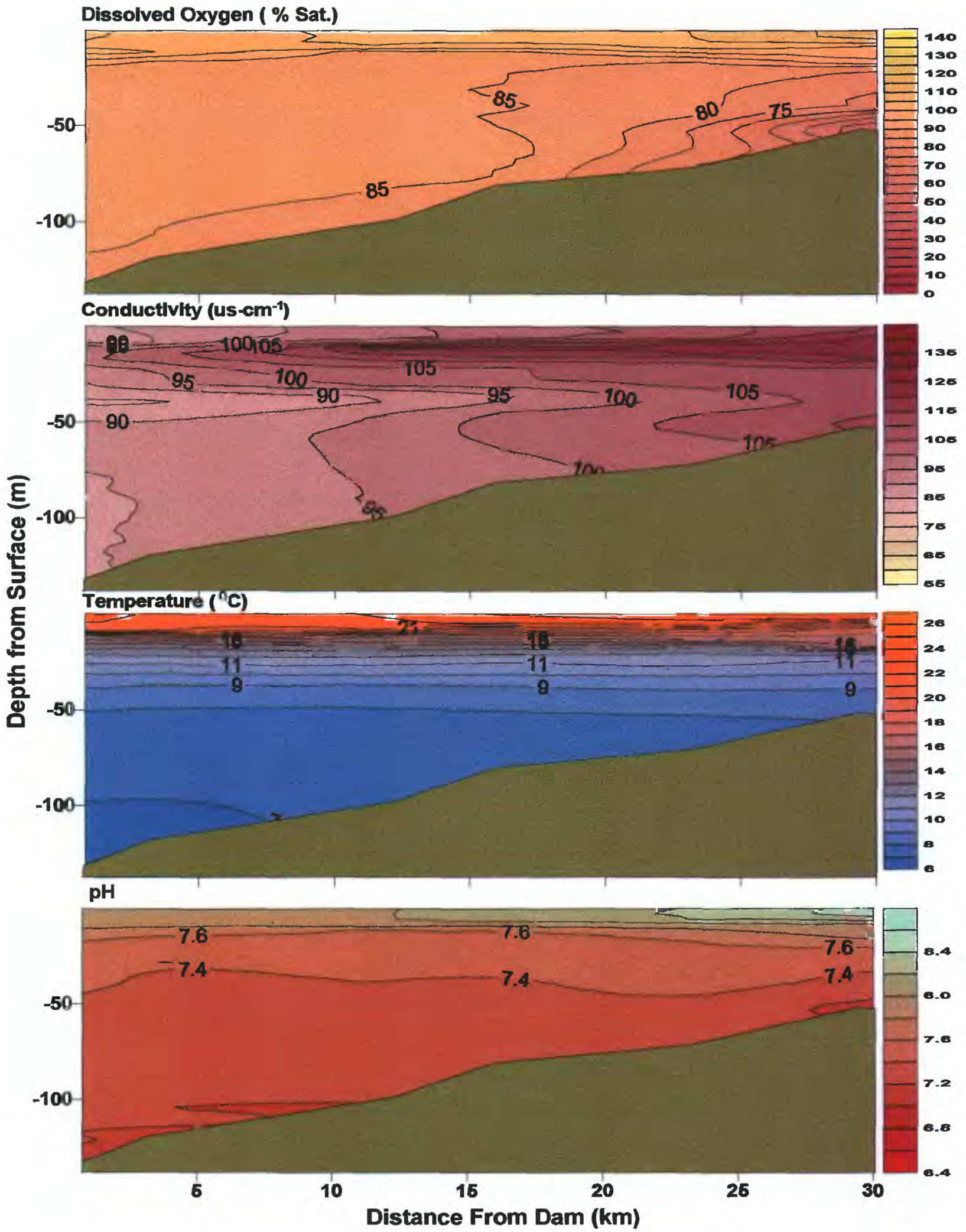


Figure 5 cont.
p.

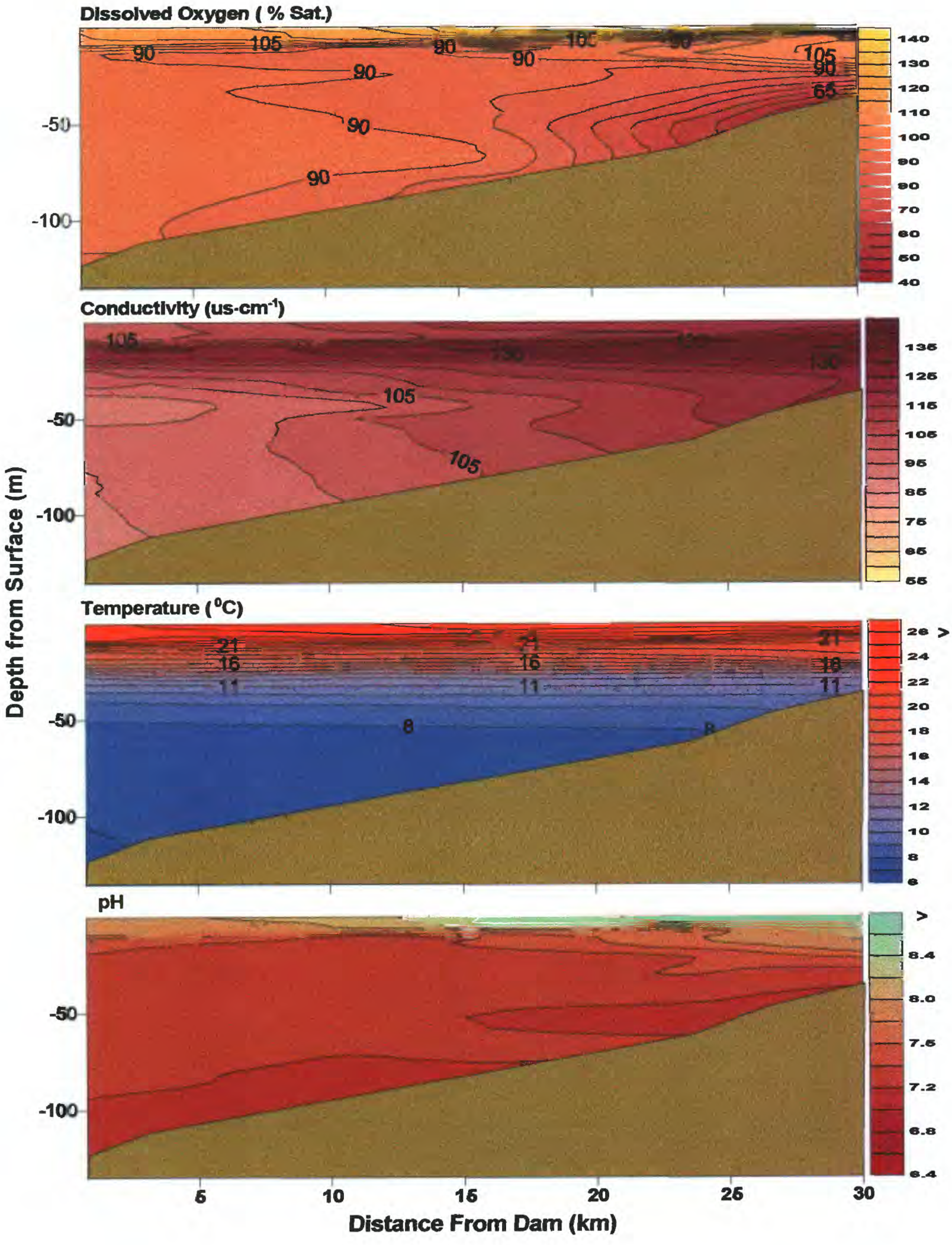


Figure 5 cont.
q.

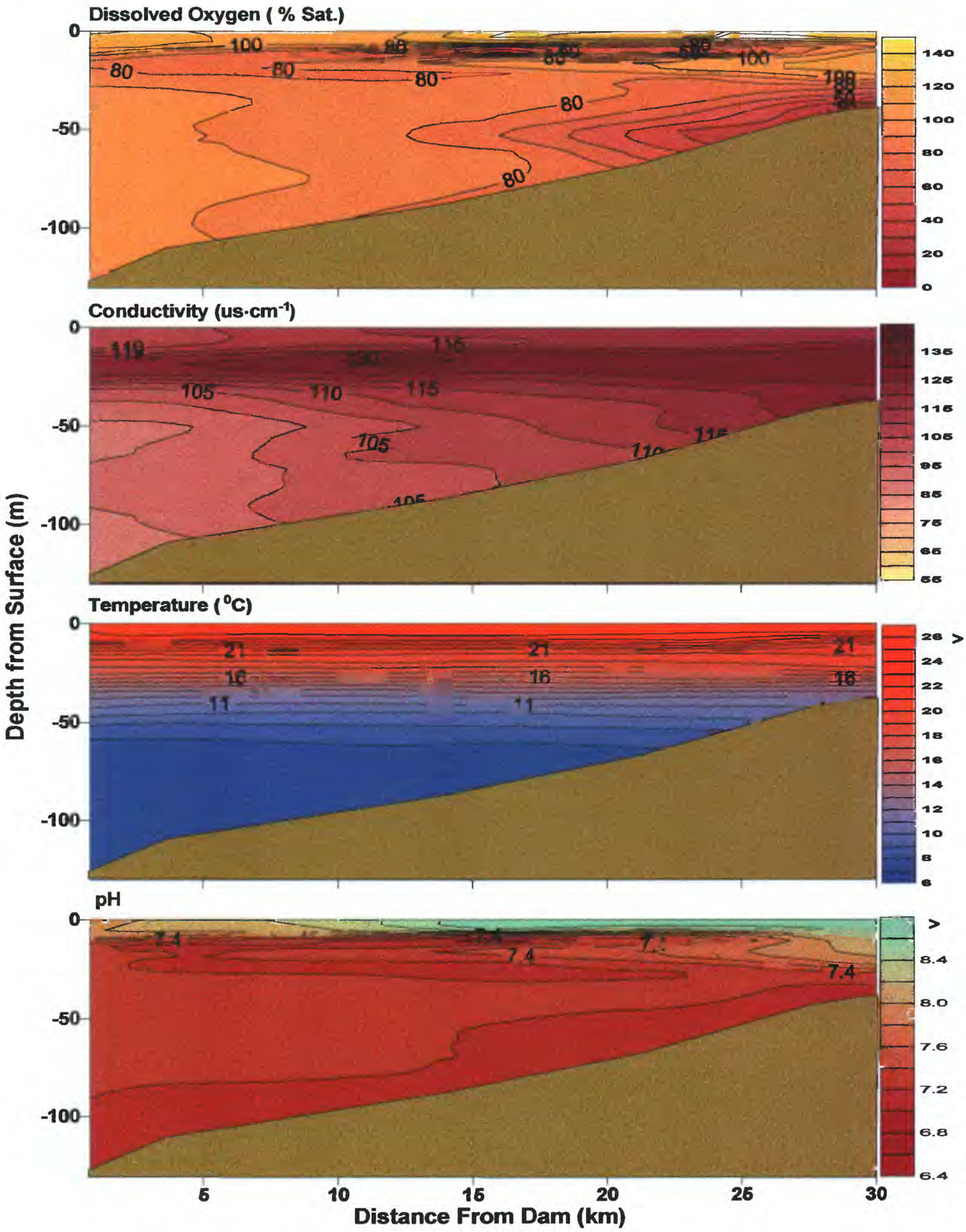


Figure 5 cont.
r.

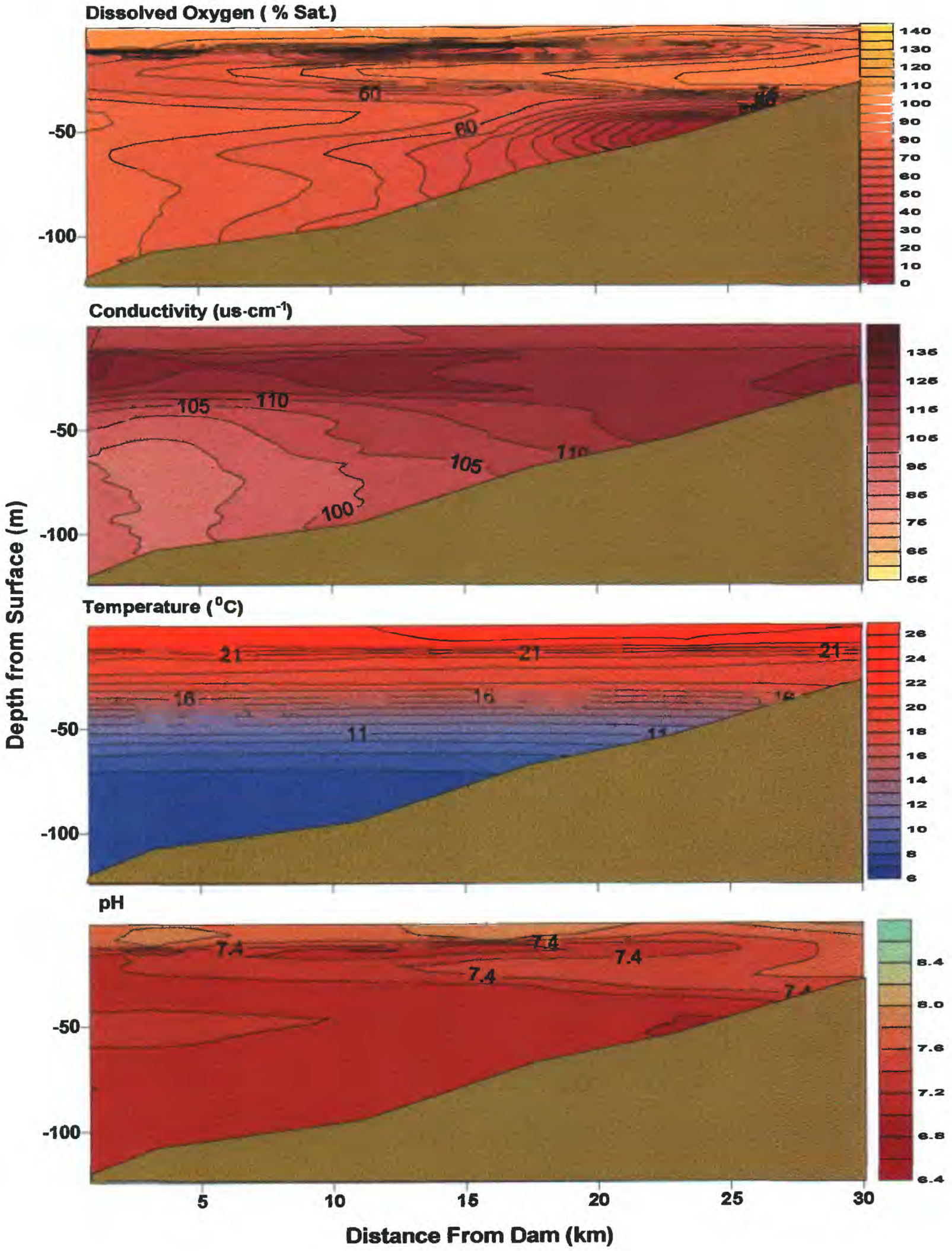


Figure 5 cont.
s.

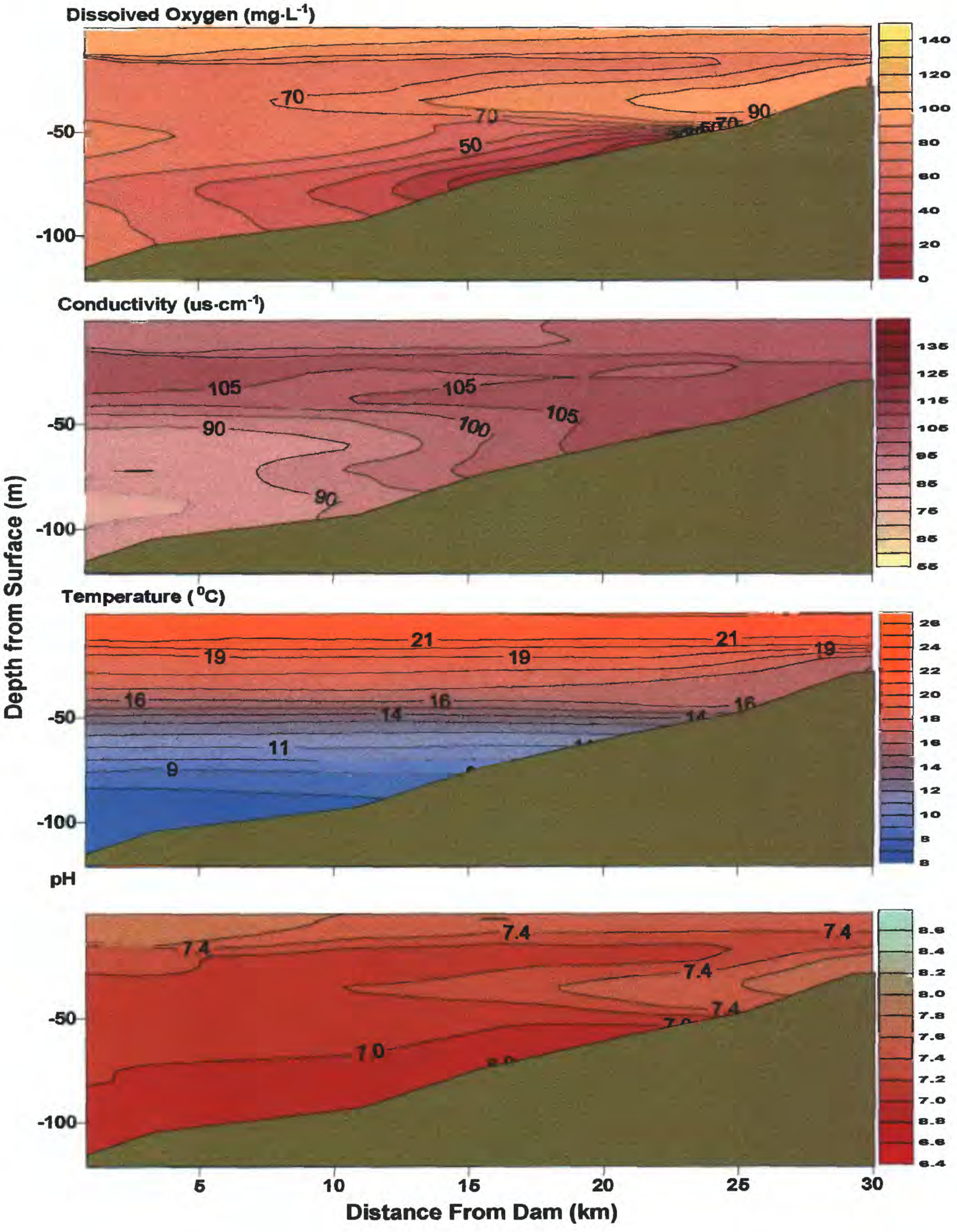


Figure 5 cont.
t.

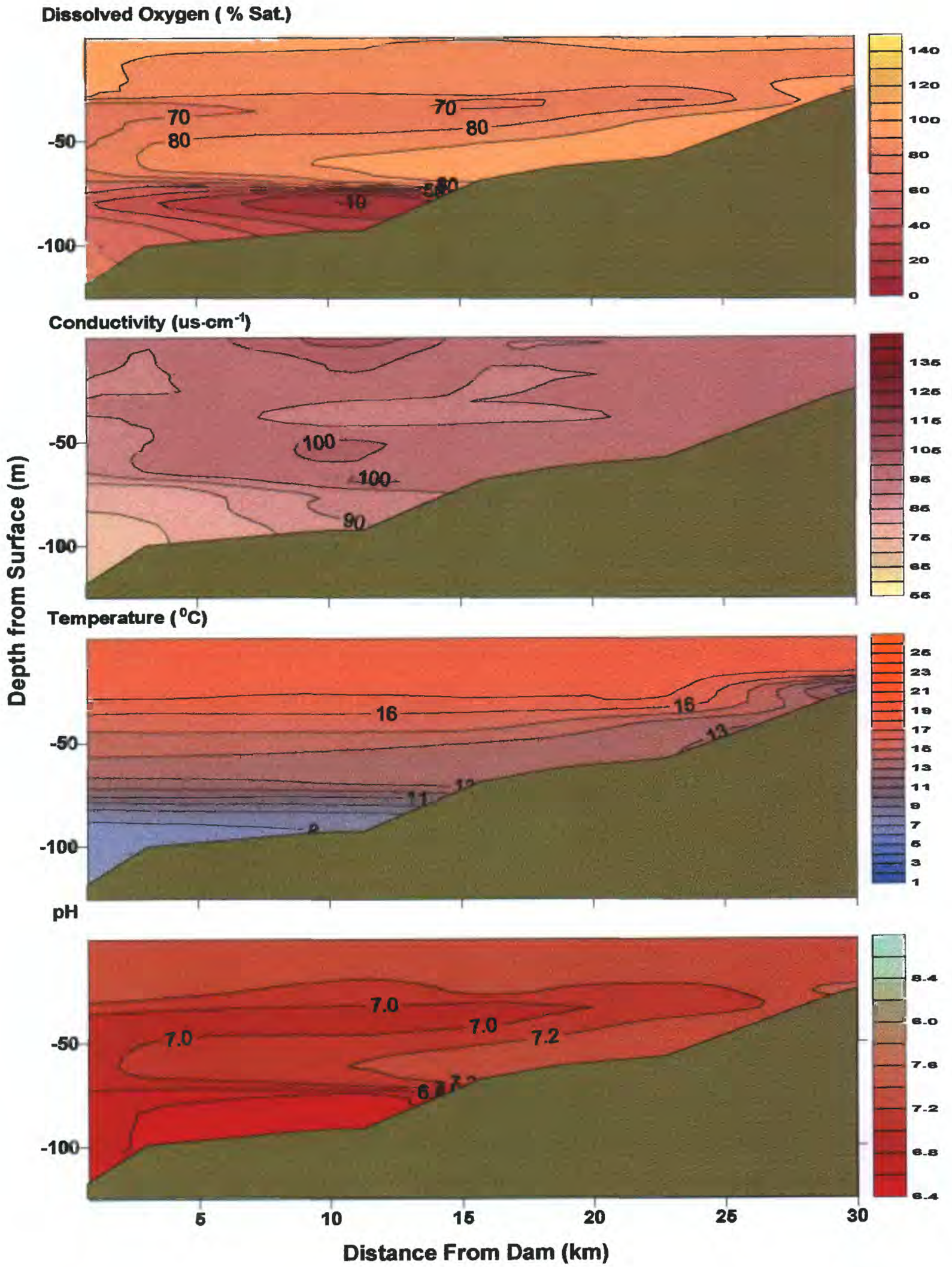


Figure 6 a-t. Monthly isopleth description of temperature (^{0}C), dissolved oxygen (percent saturation), pH, and specific conductance (μS/cm) for the Sacramento River arm for 1996 and 1997. a. February 1996; b. March 1996; c. April 1996; d. May 1996; e. June 1996; f. July 1996; g. August 1996; h. September 1996; i. October 1996; j. January 1997; k. February 1997; l. March 1997; m. April 1997; n. May 1997; o. June 1997; p. July 1997; q. August 1997; r. September 1997; s. October 1997; t. November 1997.

a.

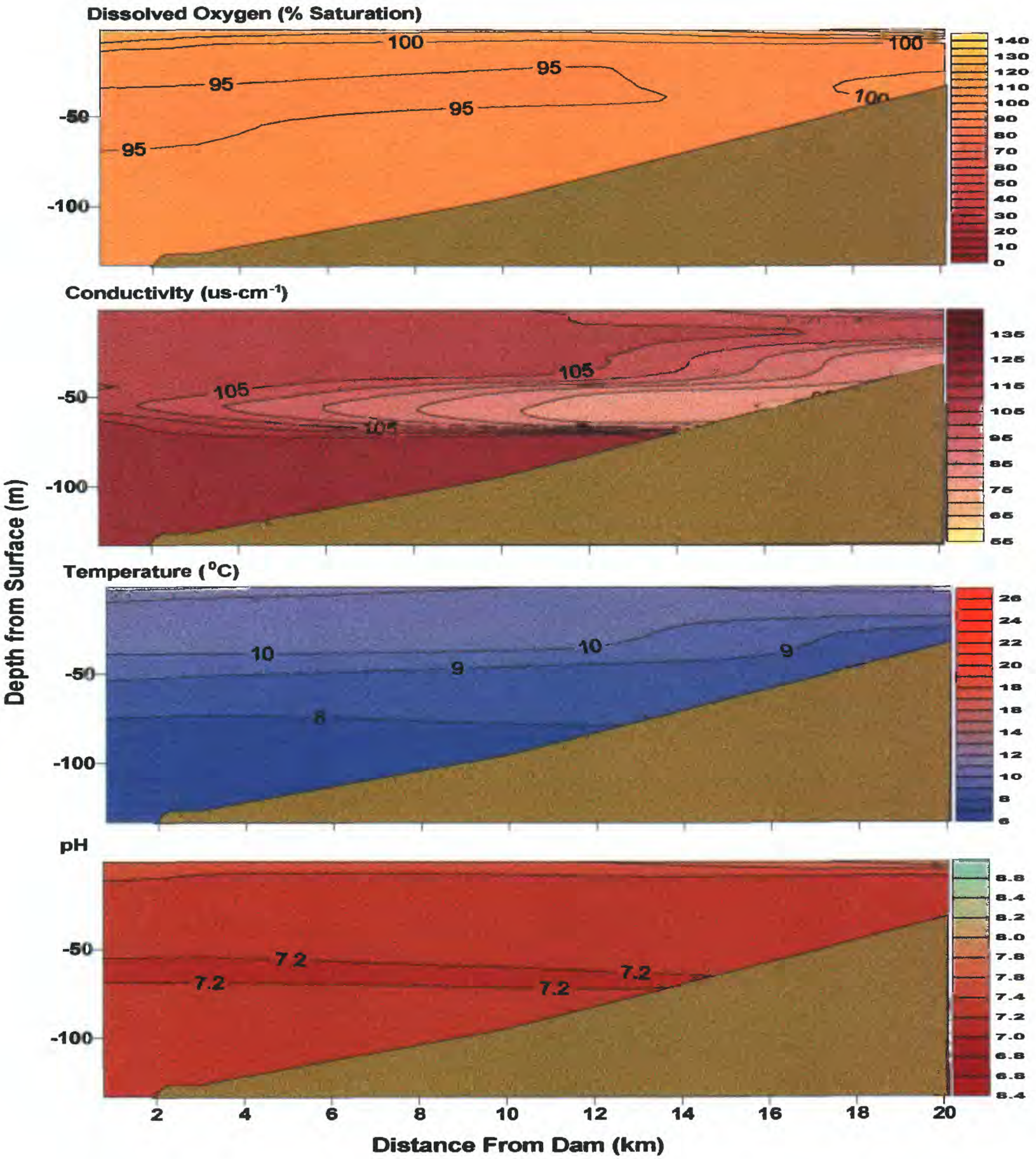


Figure 6 cont.
b.

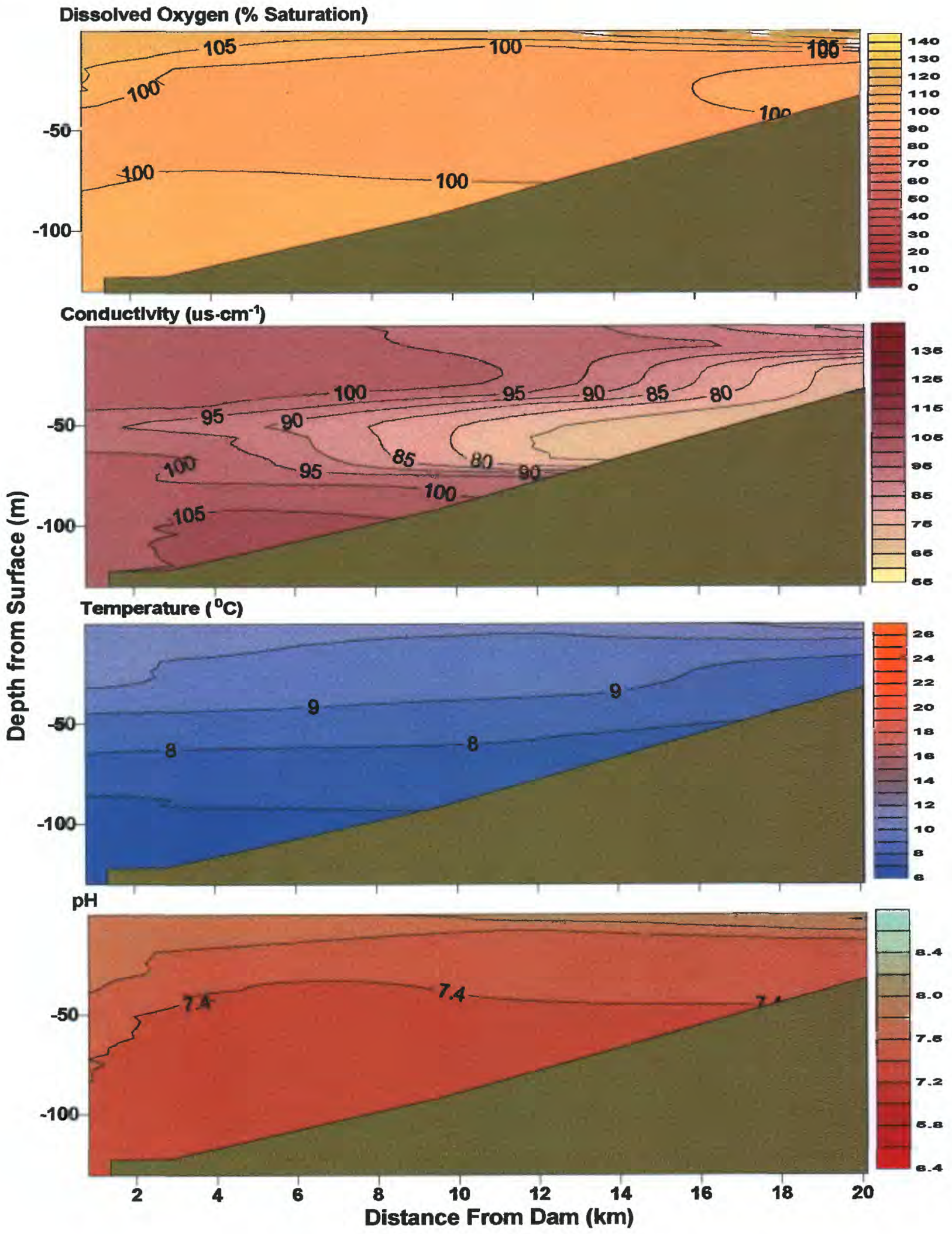


Figure 6 cont.
c.

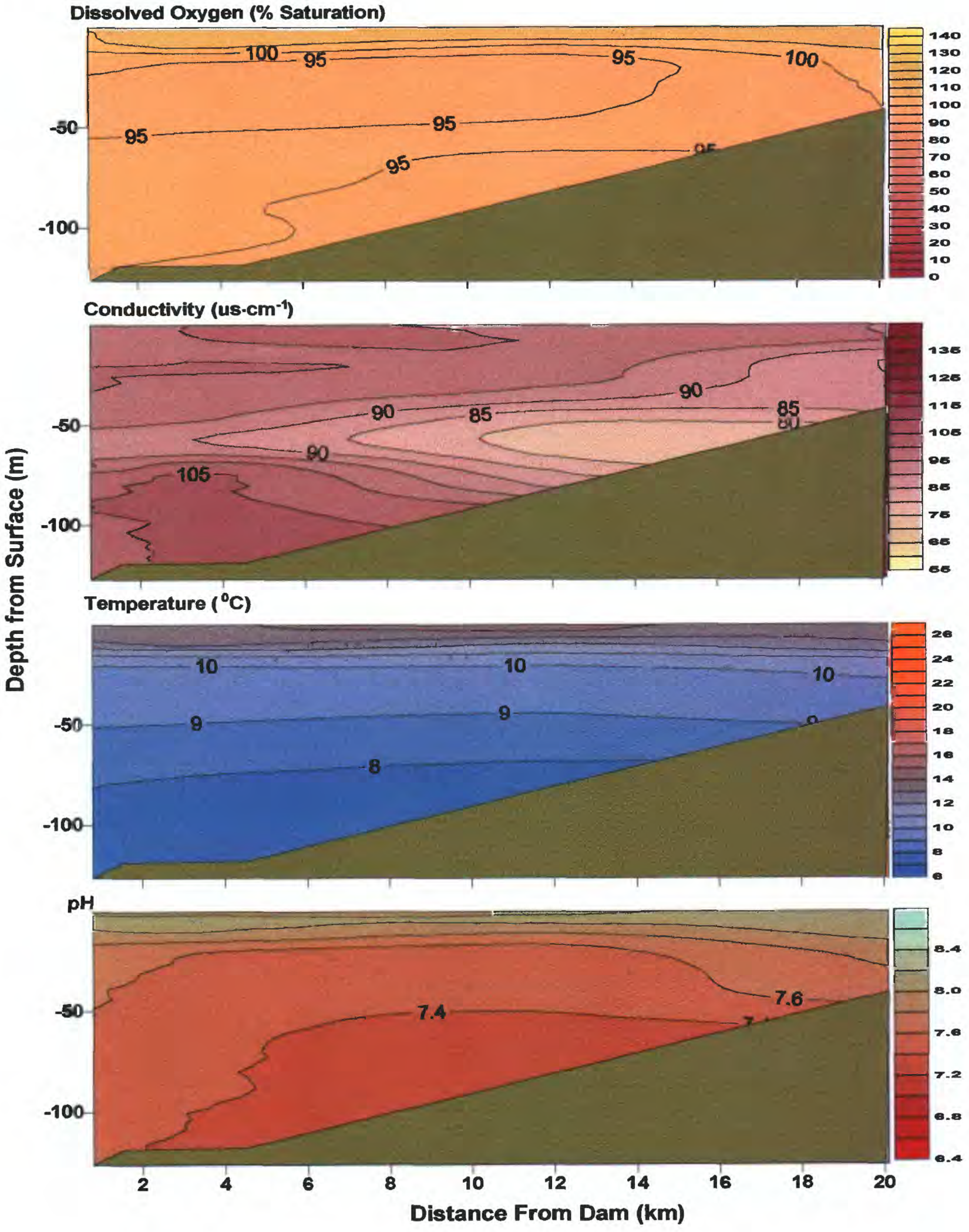


Figure 6 cont.
d.

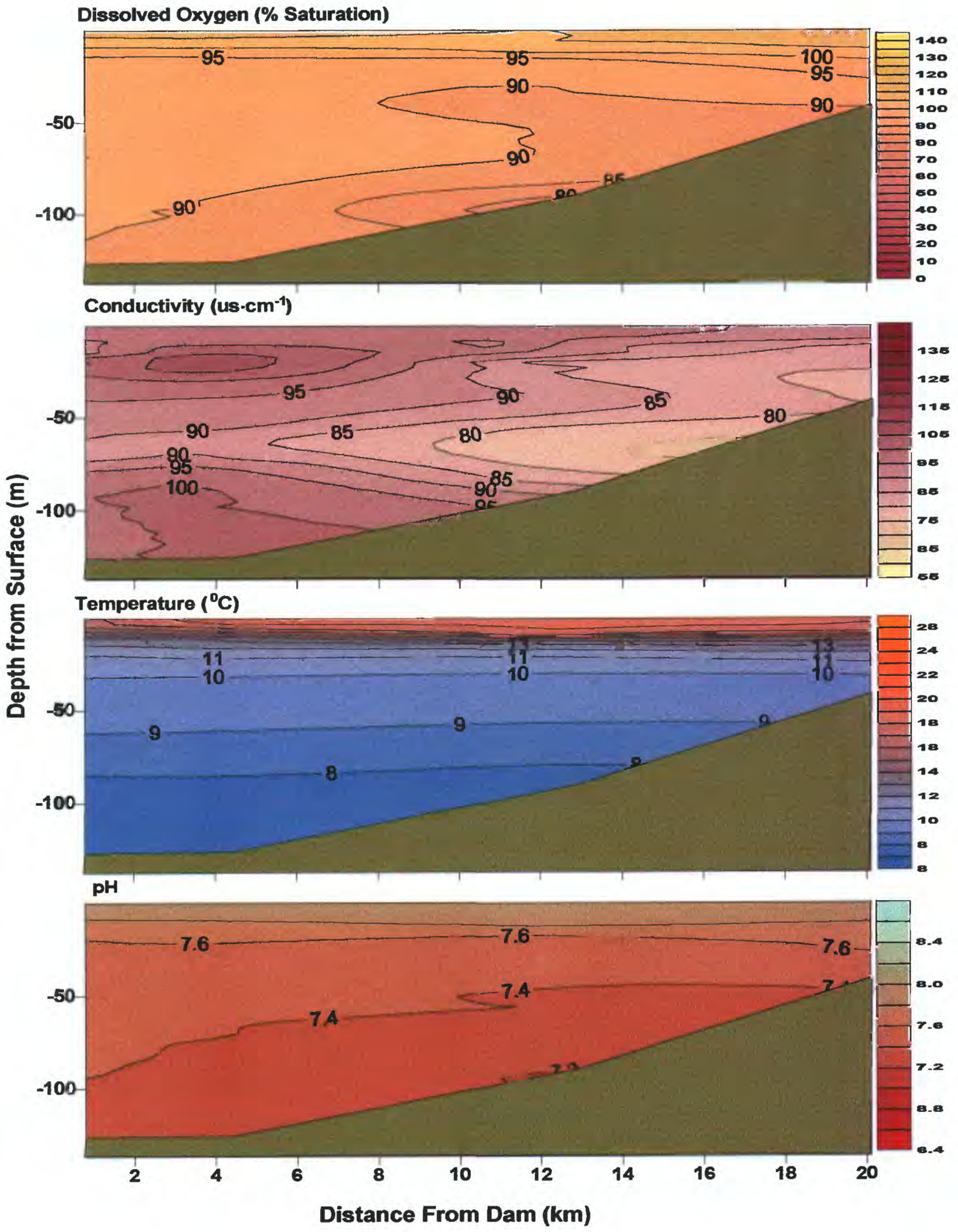


Figure 6 cont.
e.

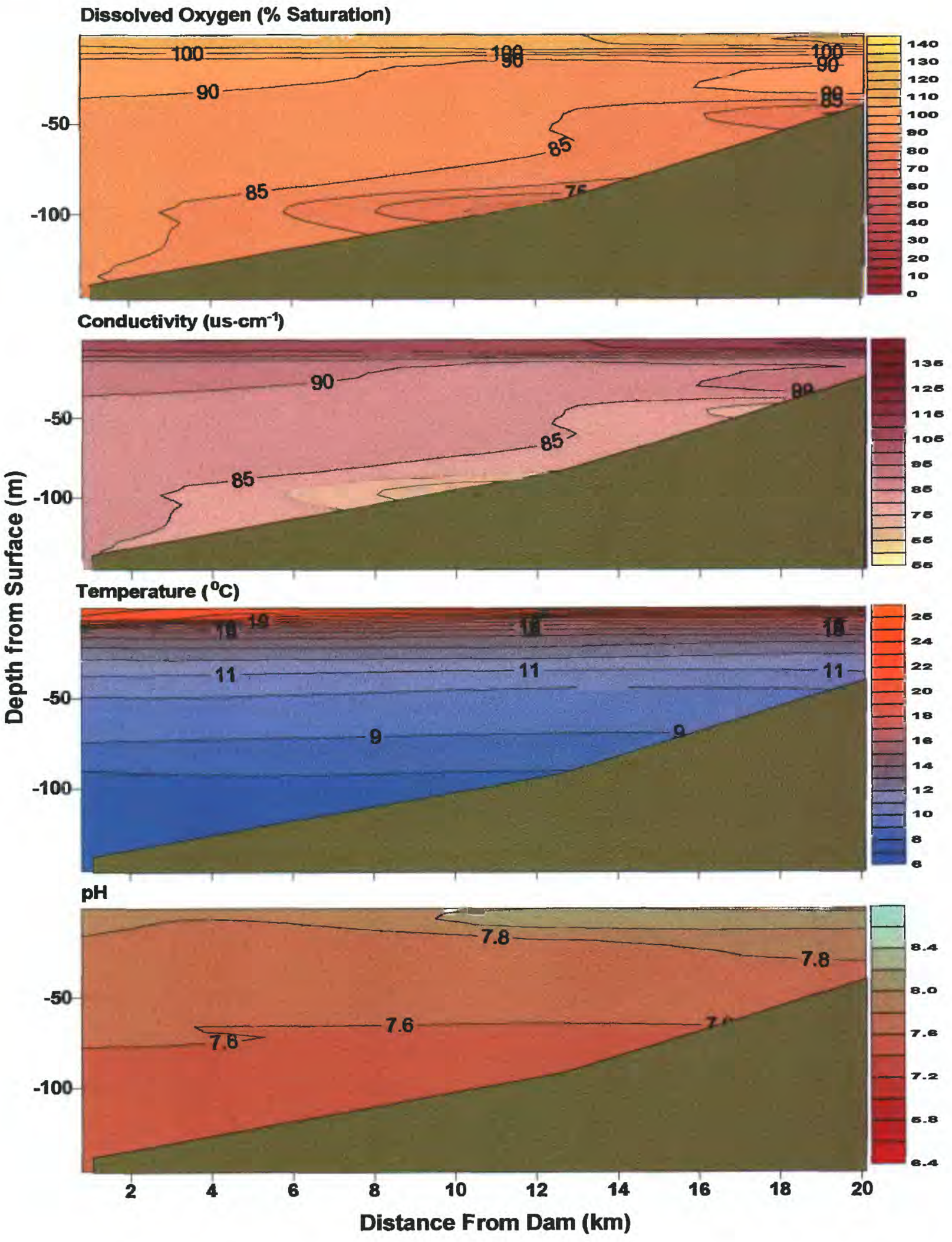


Figure 6 cont.
f.

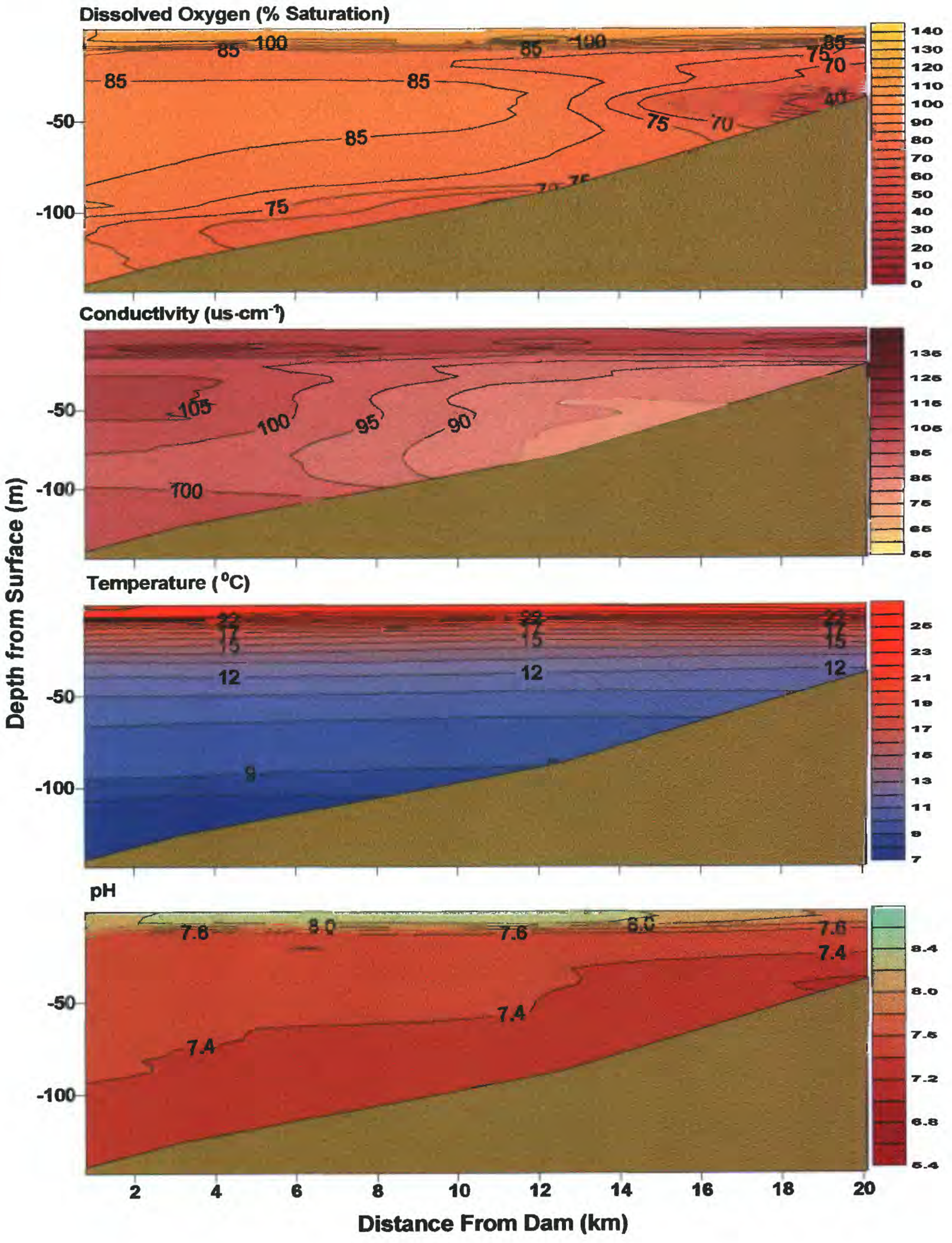


Figure 6 cont.
g.

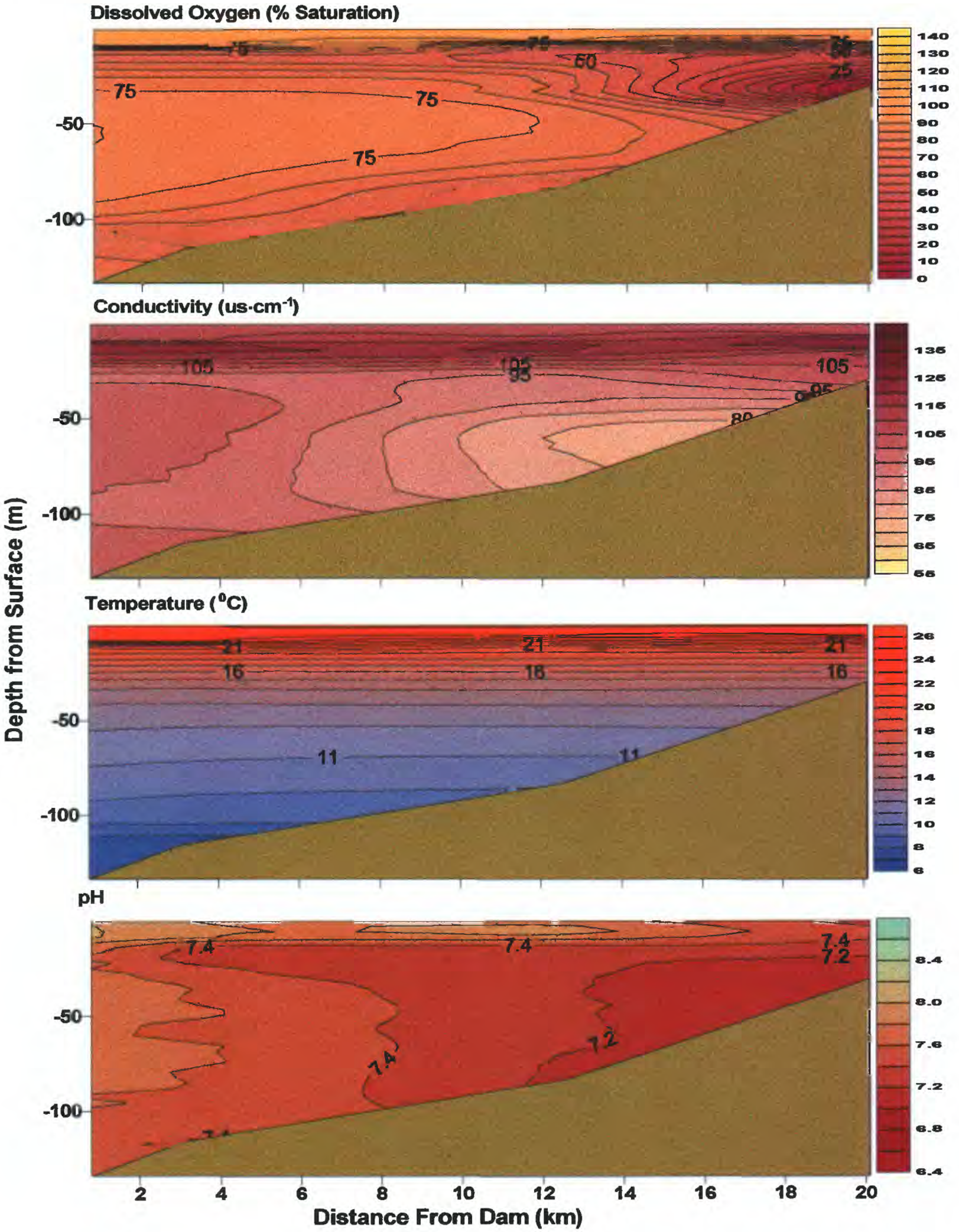


Figure 6 cont.
h.

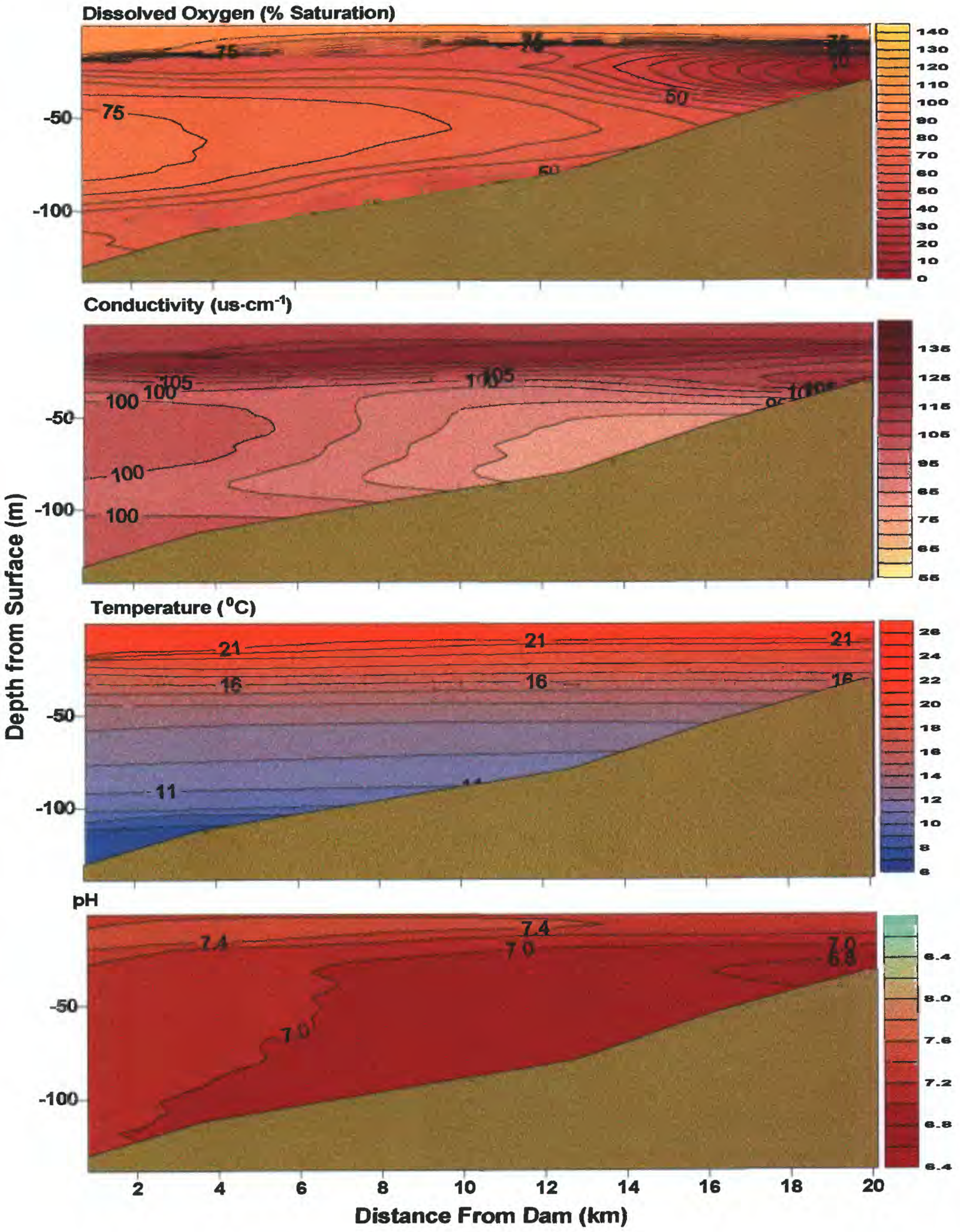


Figure 6 cont.
i.

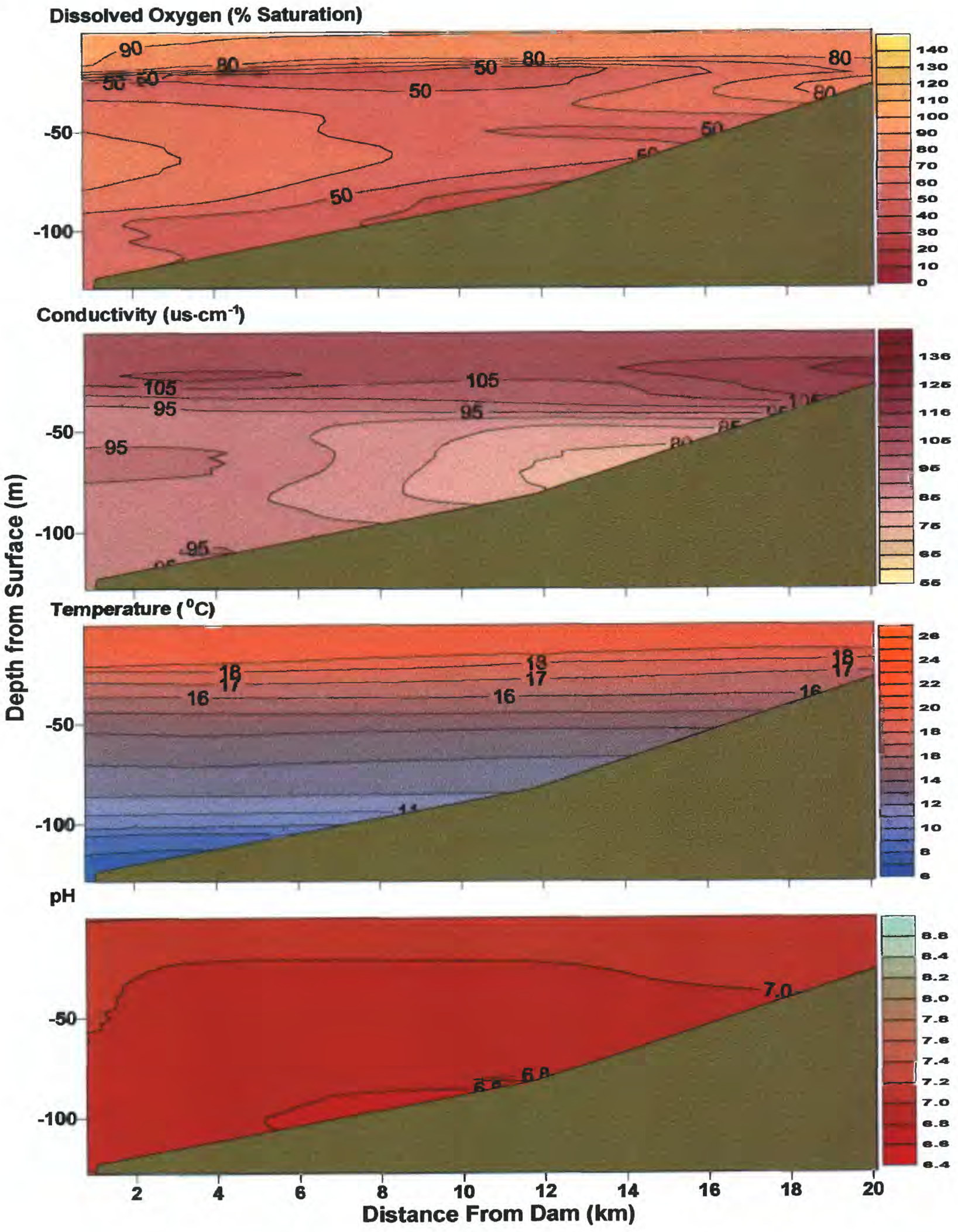


Figure 6 cont.
j.

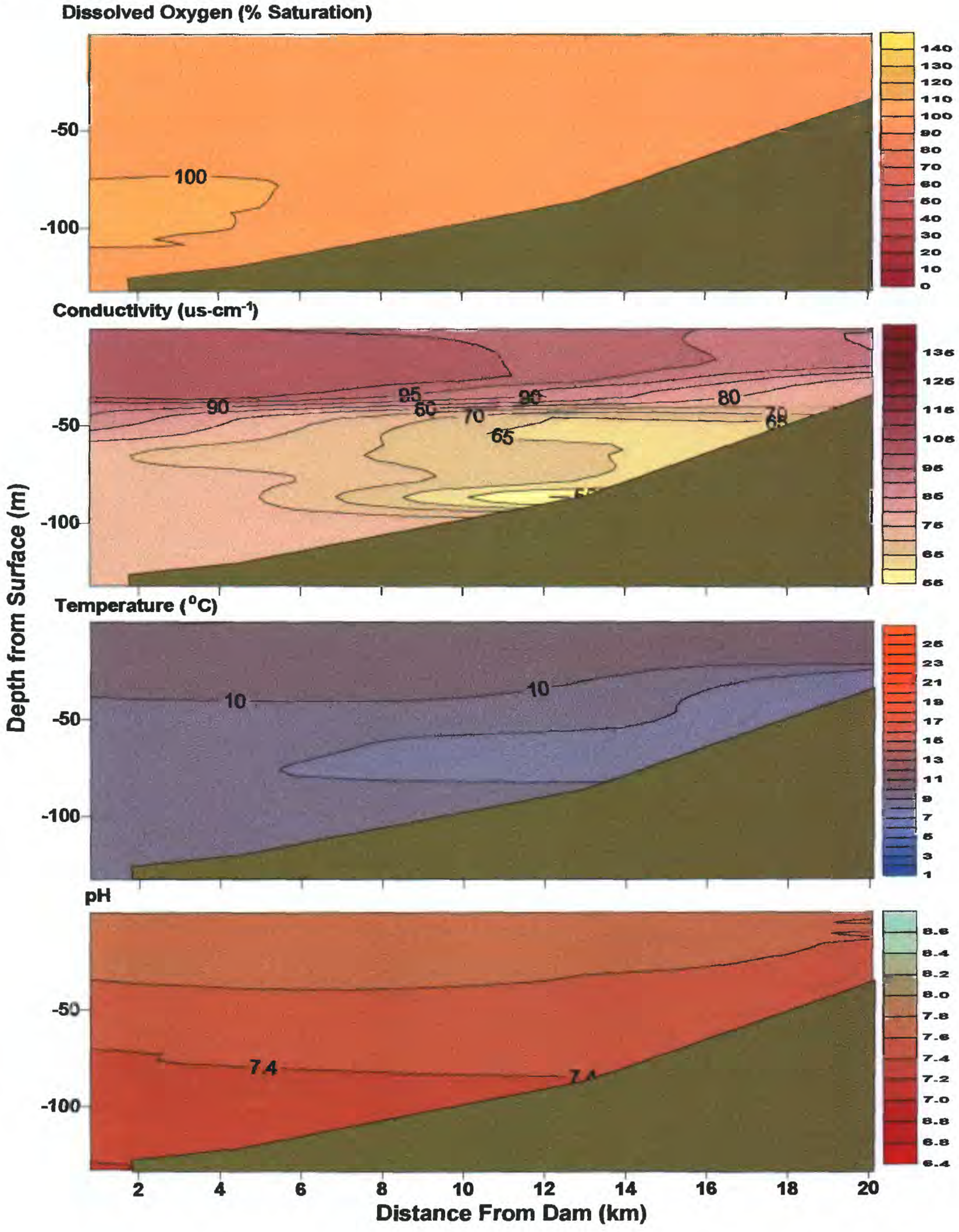


Figure 6 cont.
k.

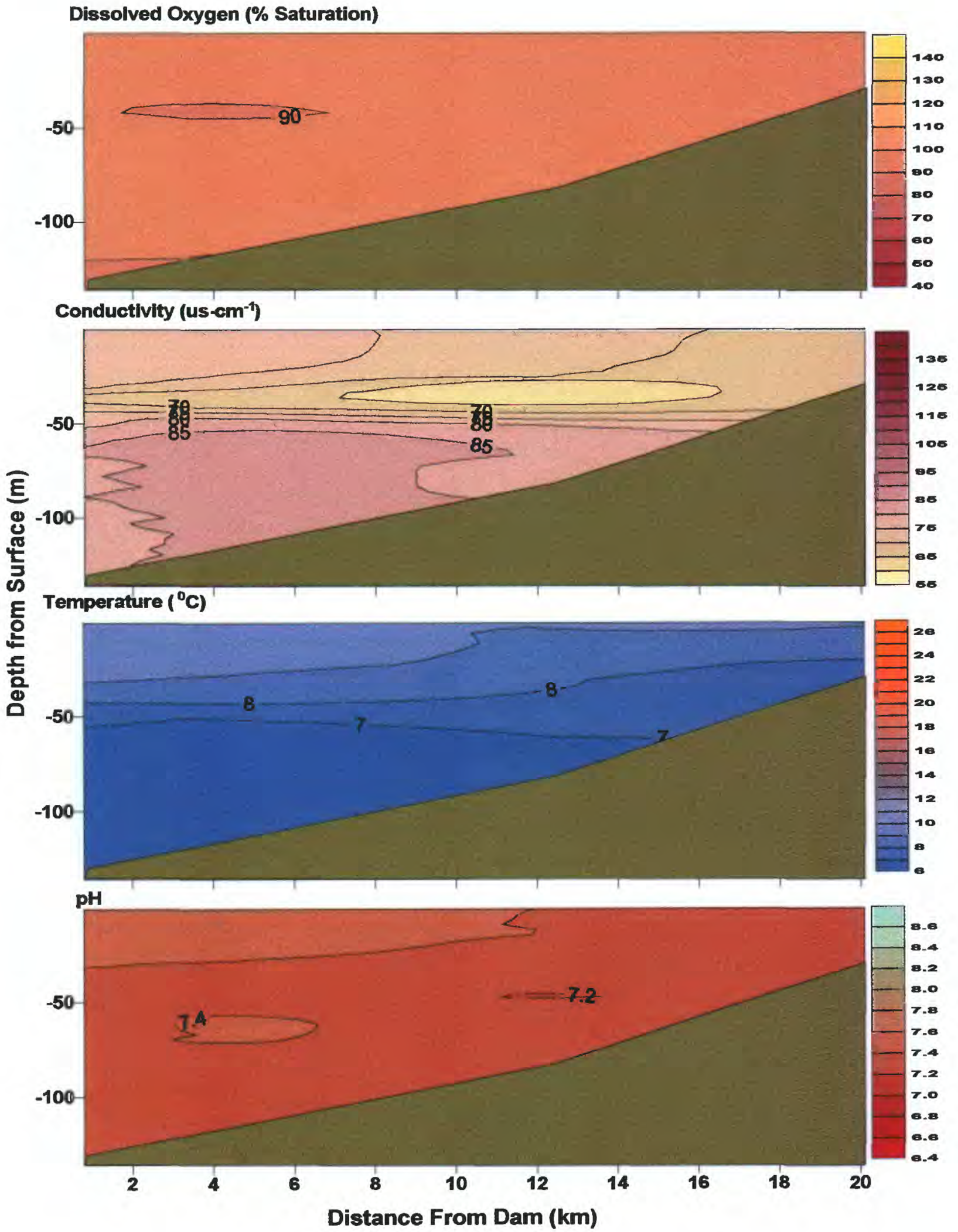


Figure 6 cont.
l.

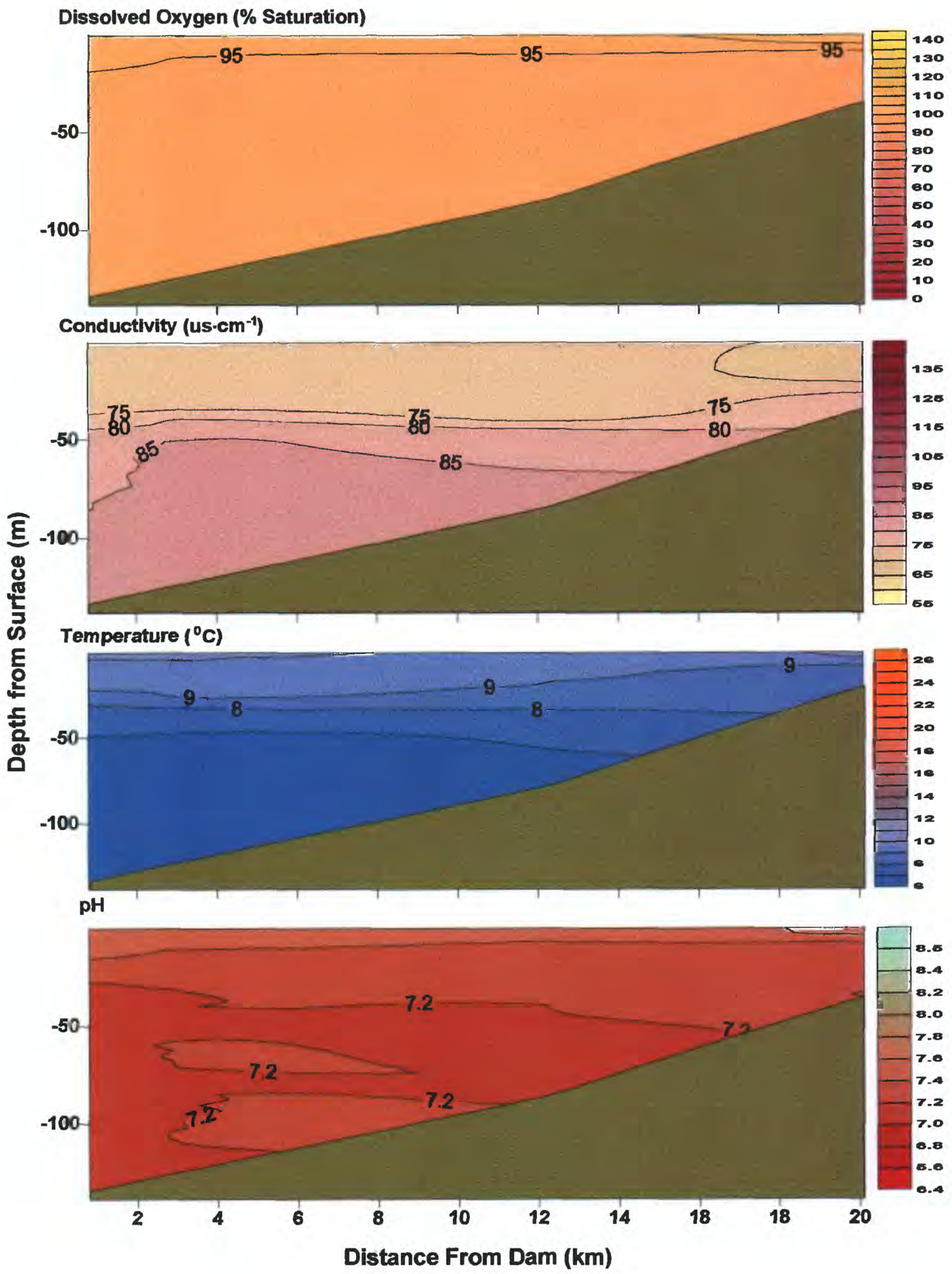


Figure 6 cont.
m.

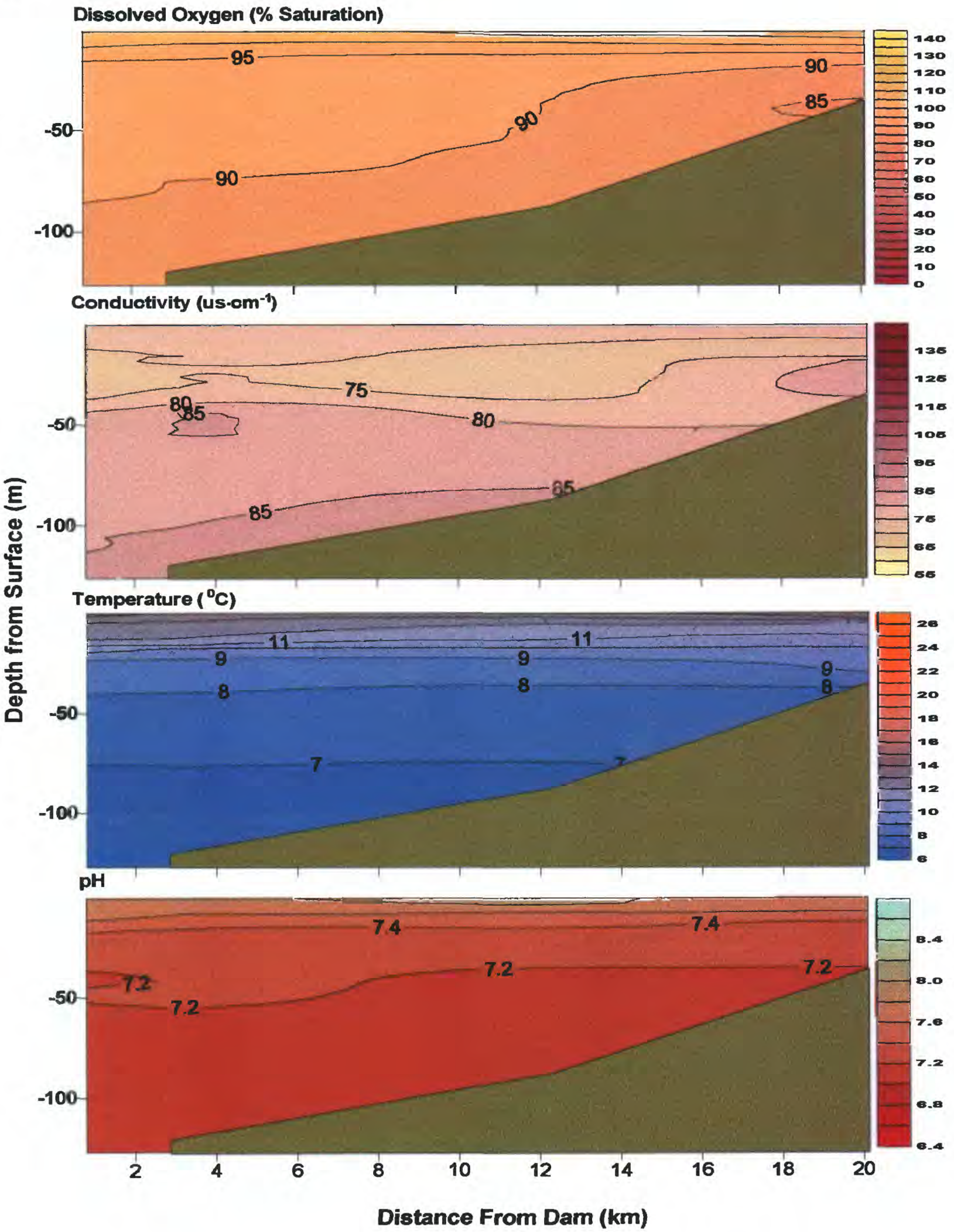


Figure 6 cont.

n.

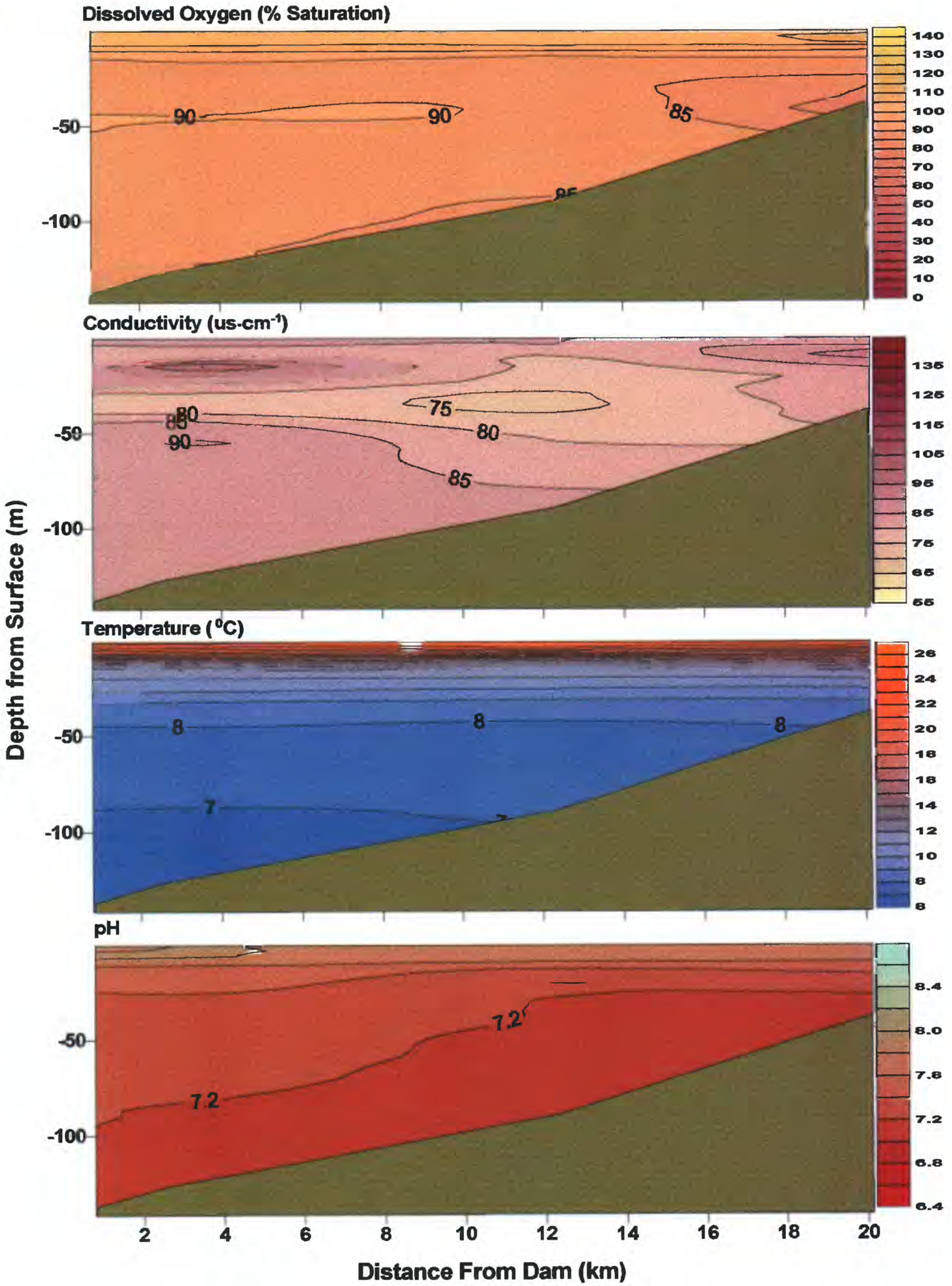


Figure 6 cont.
o.

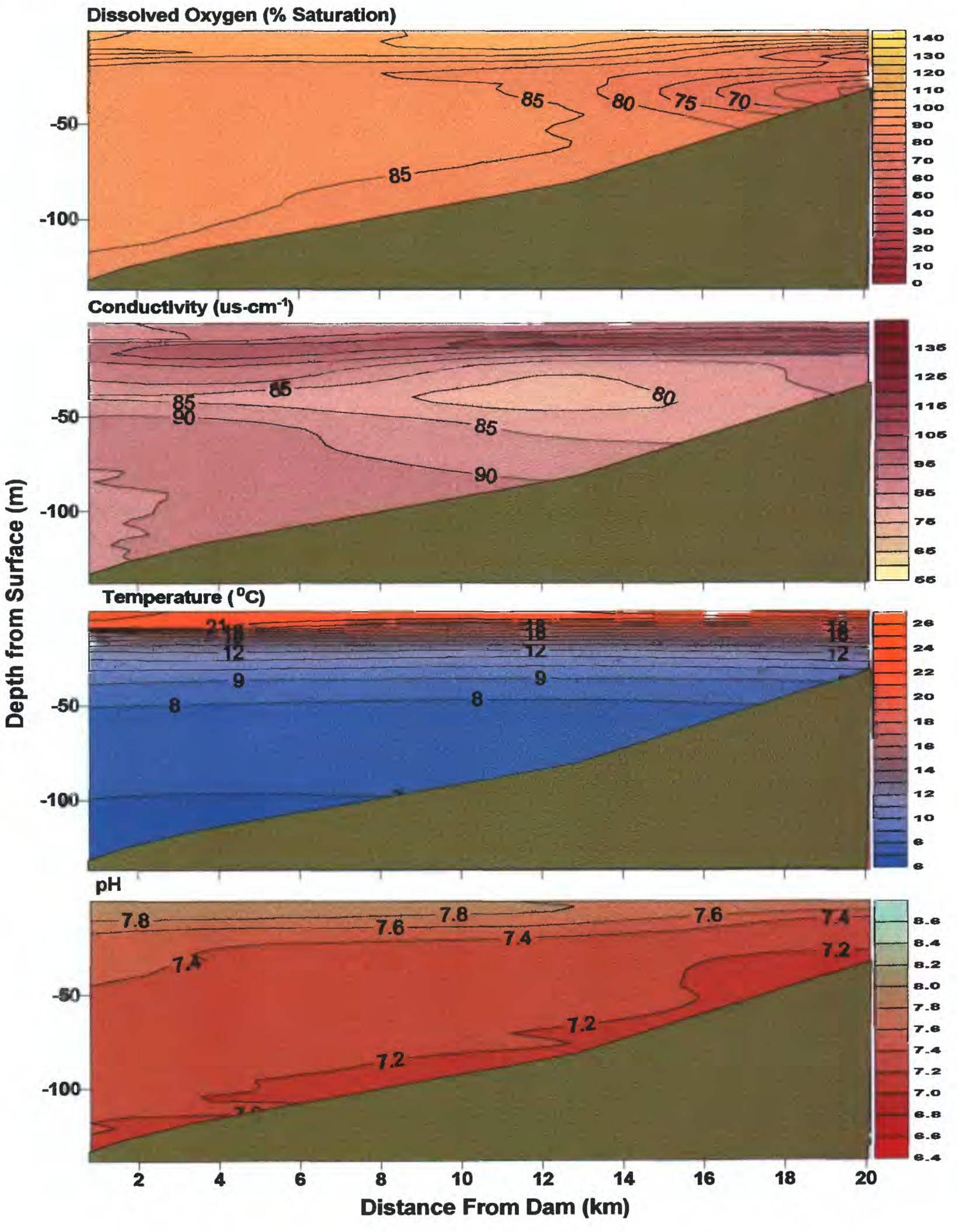


Figure 6 cont.
p.

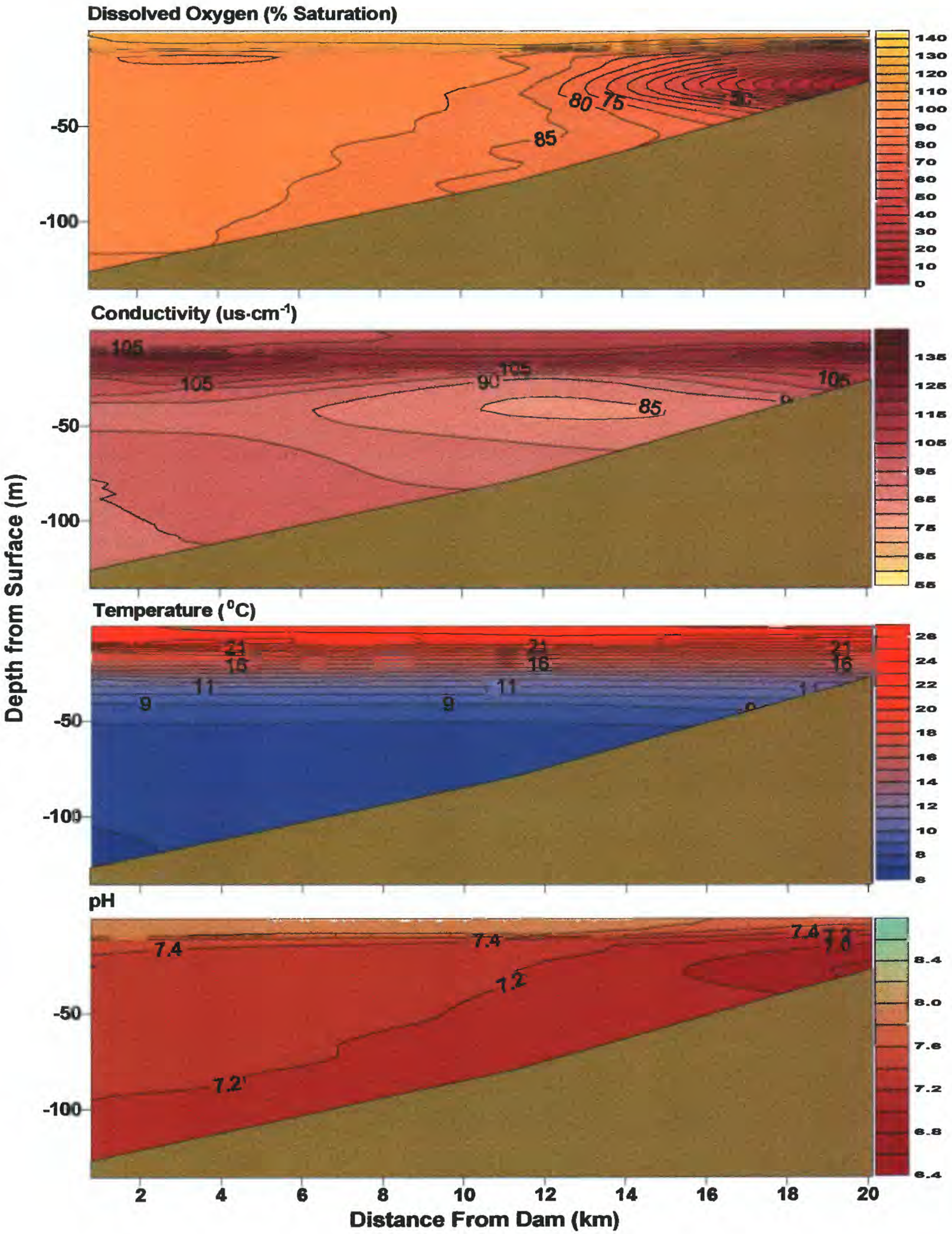


Figure 6 cont.
q.

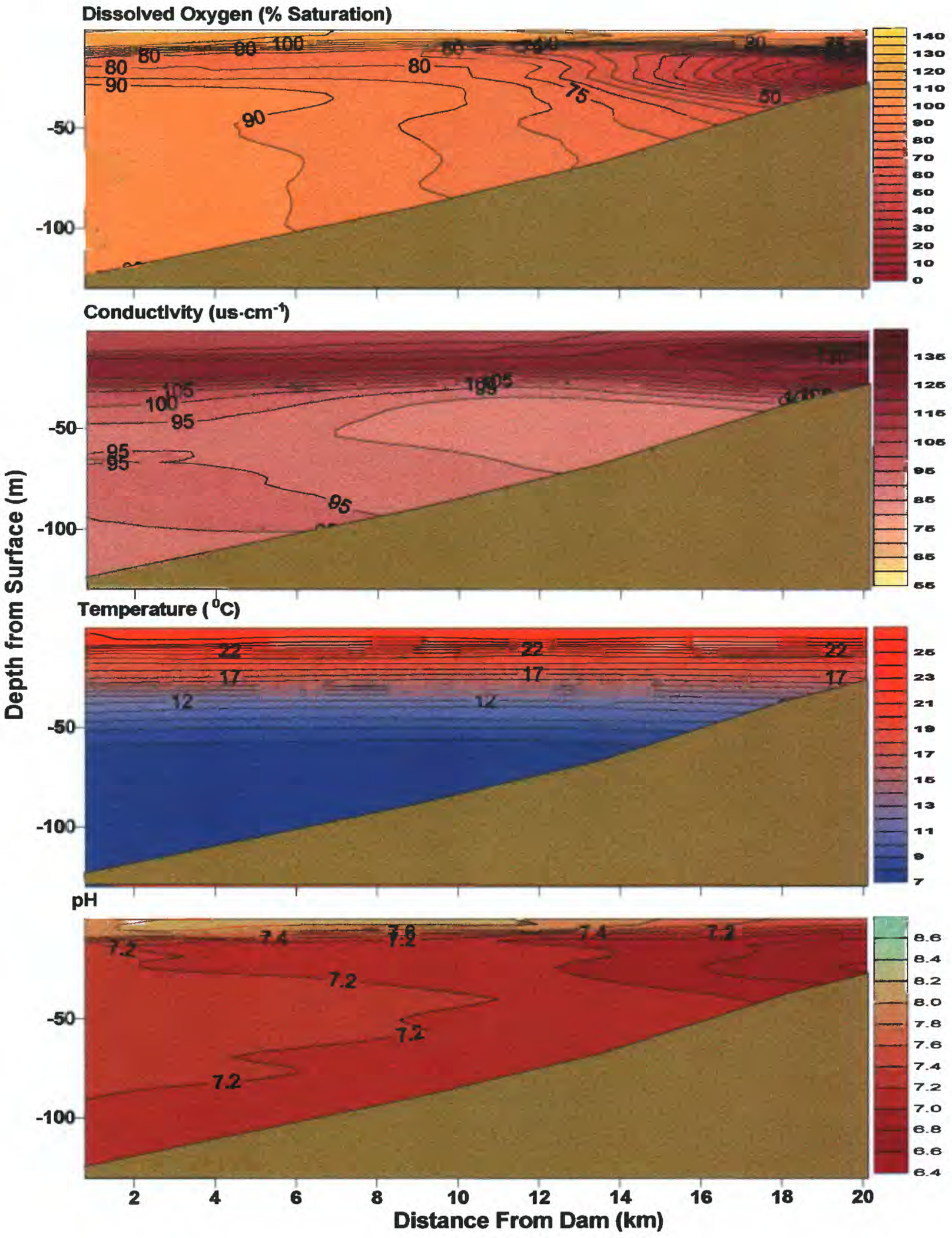


Figure 6 cont.
r.

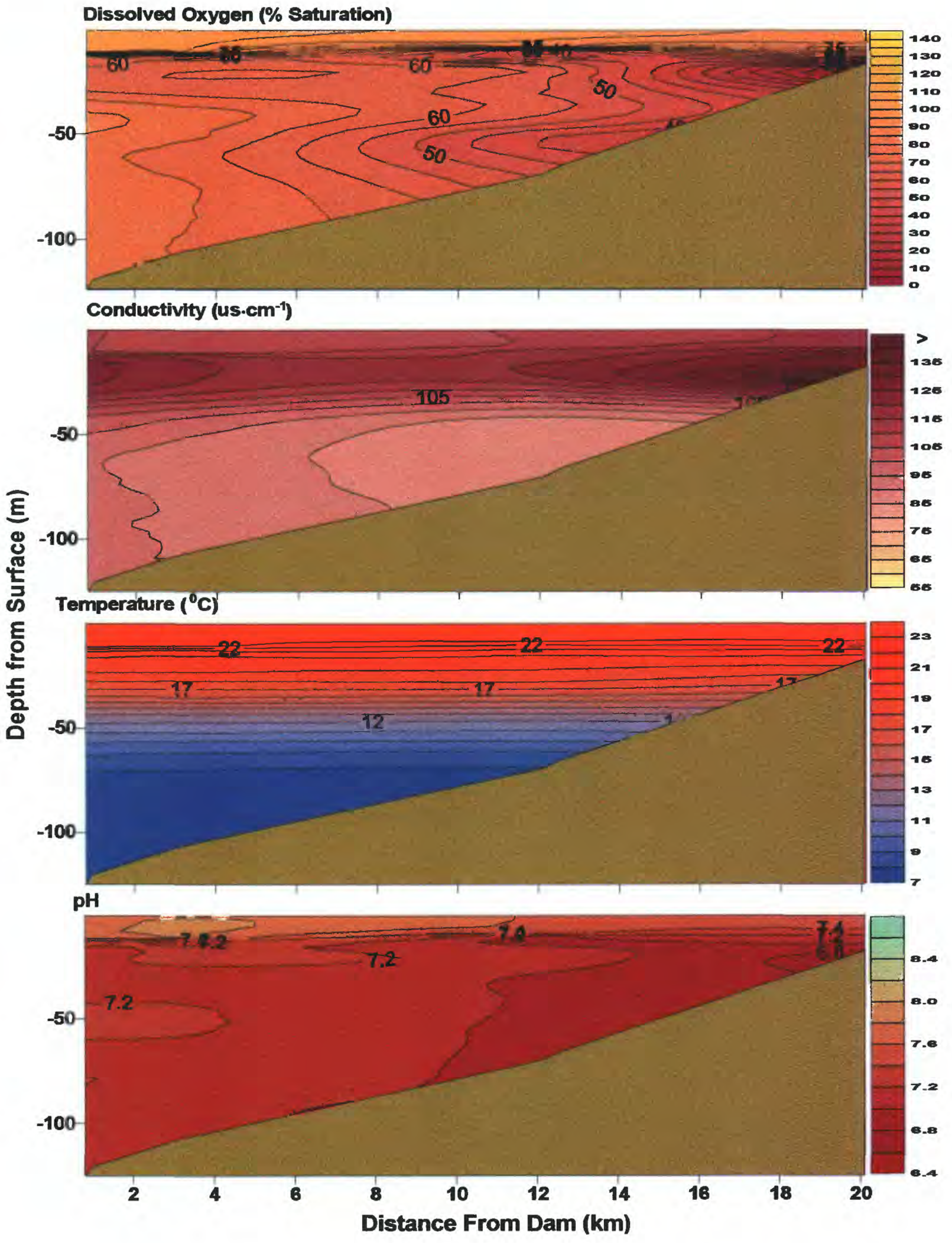


Figure 6 cont.
s.

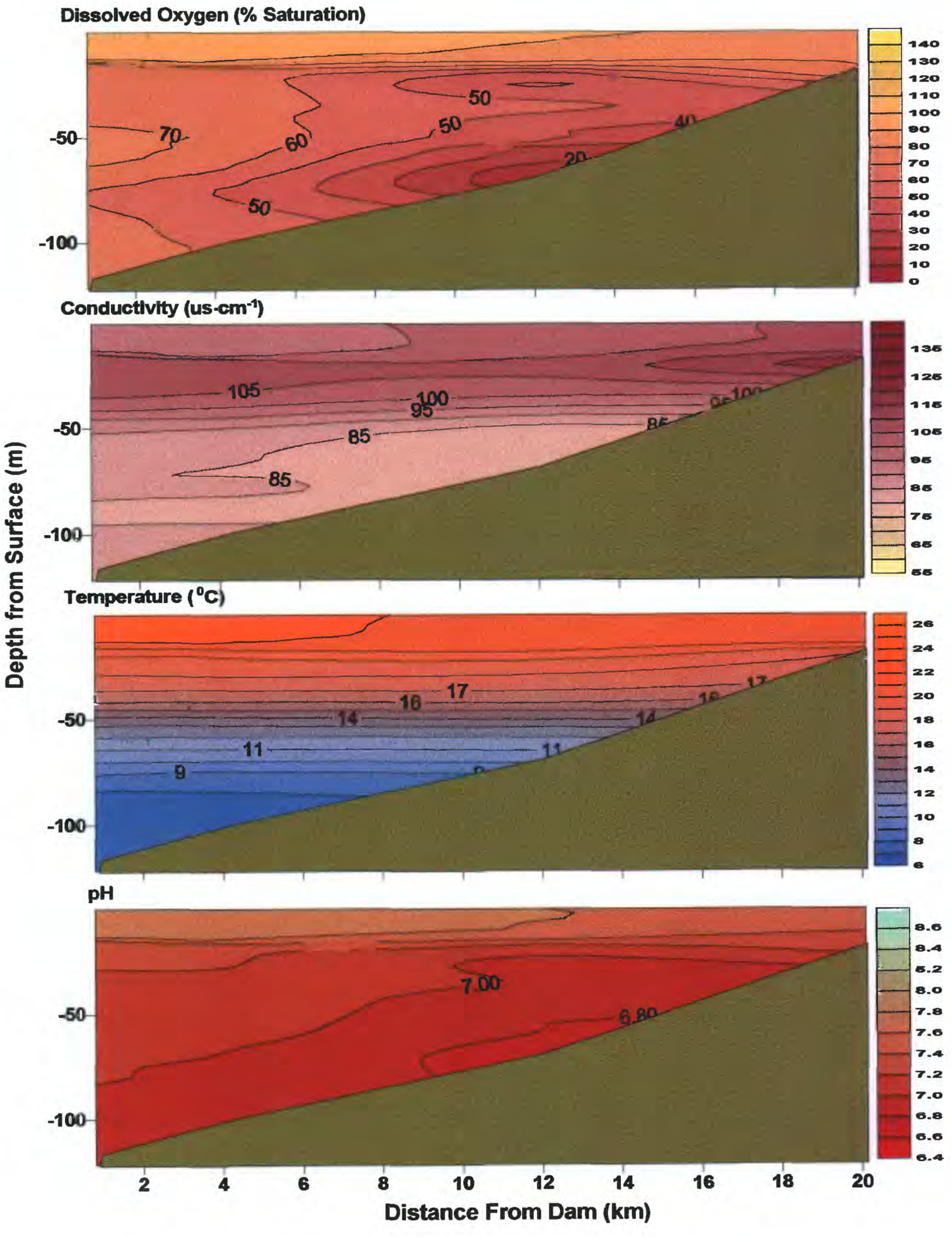


Figure 6 cont.
t.

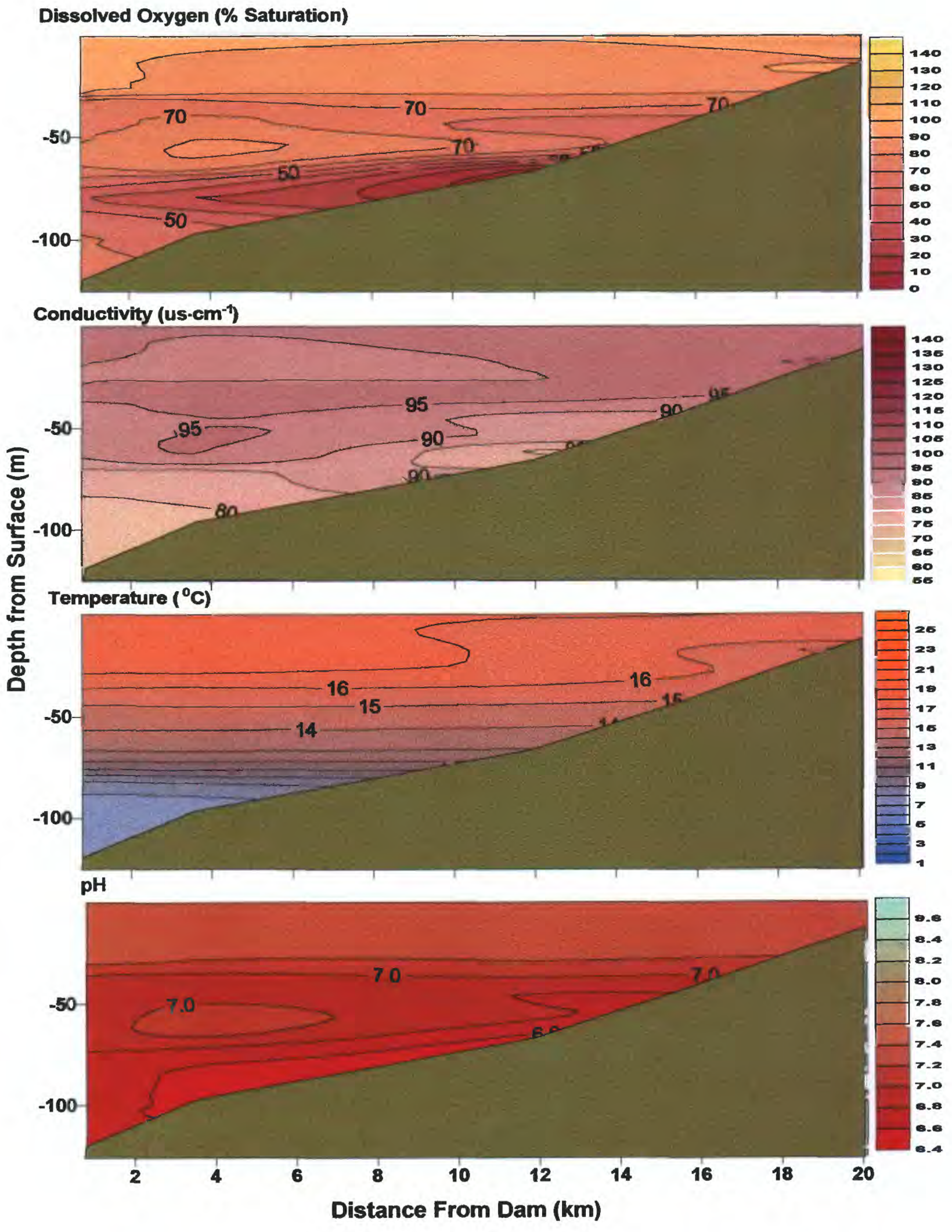


Figure 7. Comparison of chl *a* concentrations (µg/L) from surface to 30m between the main lake (S6), Sacramento River Arm (S7), McCloud River Arm (S8), Pit River Arm (S9), and the Pit-McCloud Confluence (S10) from August 1995 thru October 1997.

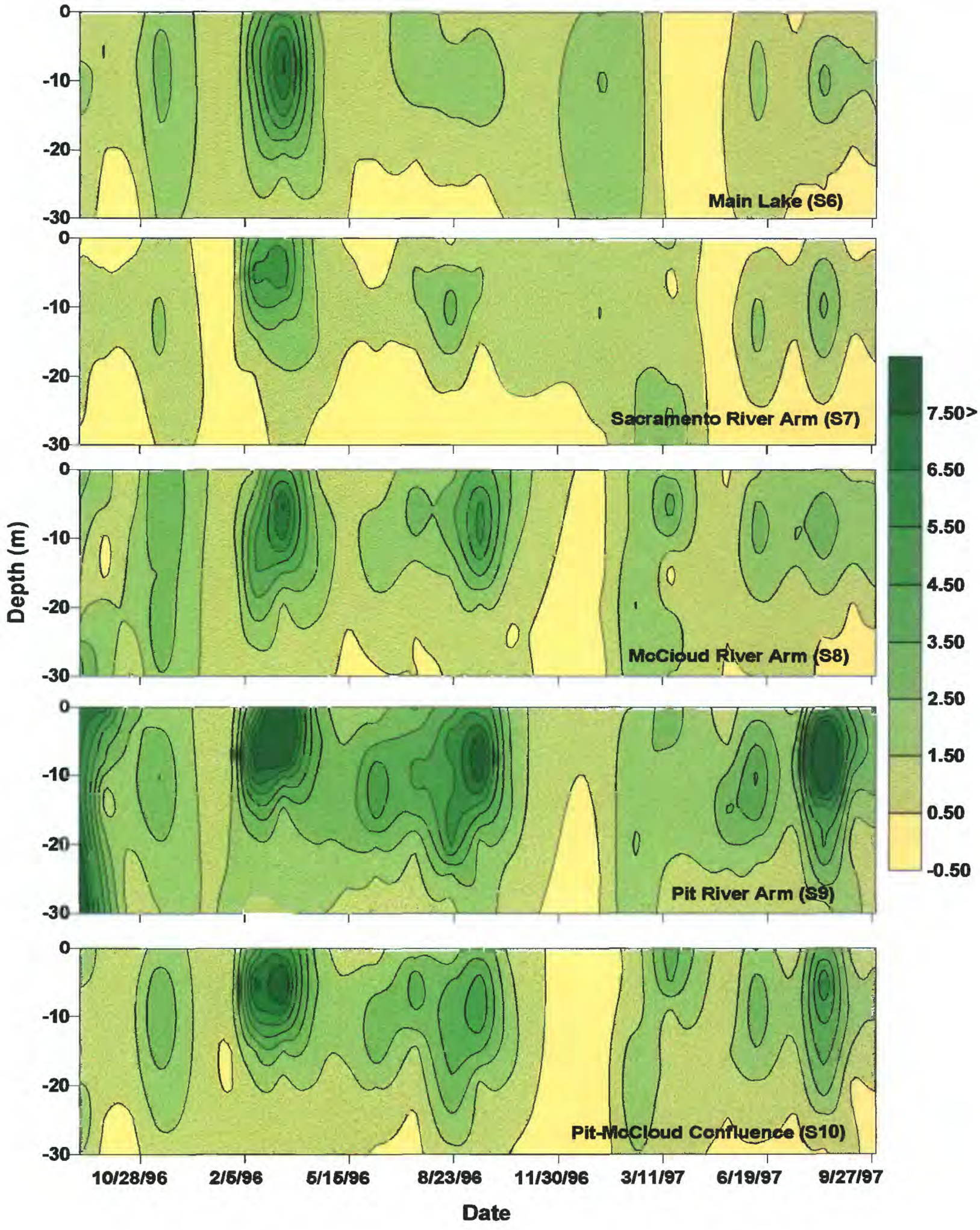


Figure 8. Seasonal cycle of inflow temperatures for the Sacramento river at Delta, station (DLT), and for the Pit river at station (PMN).

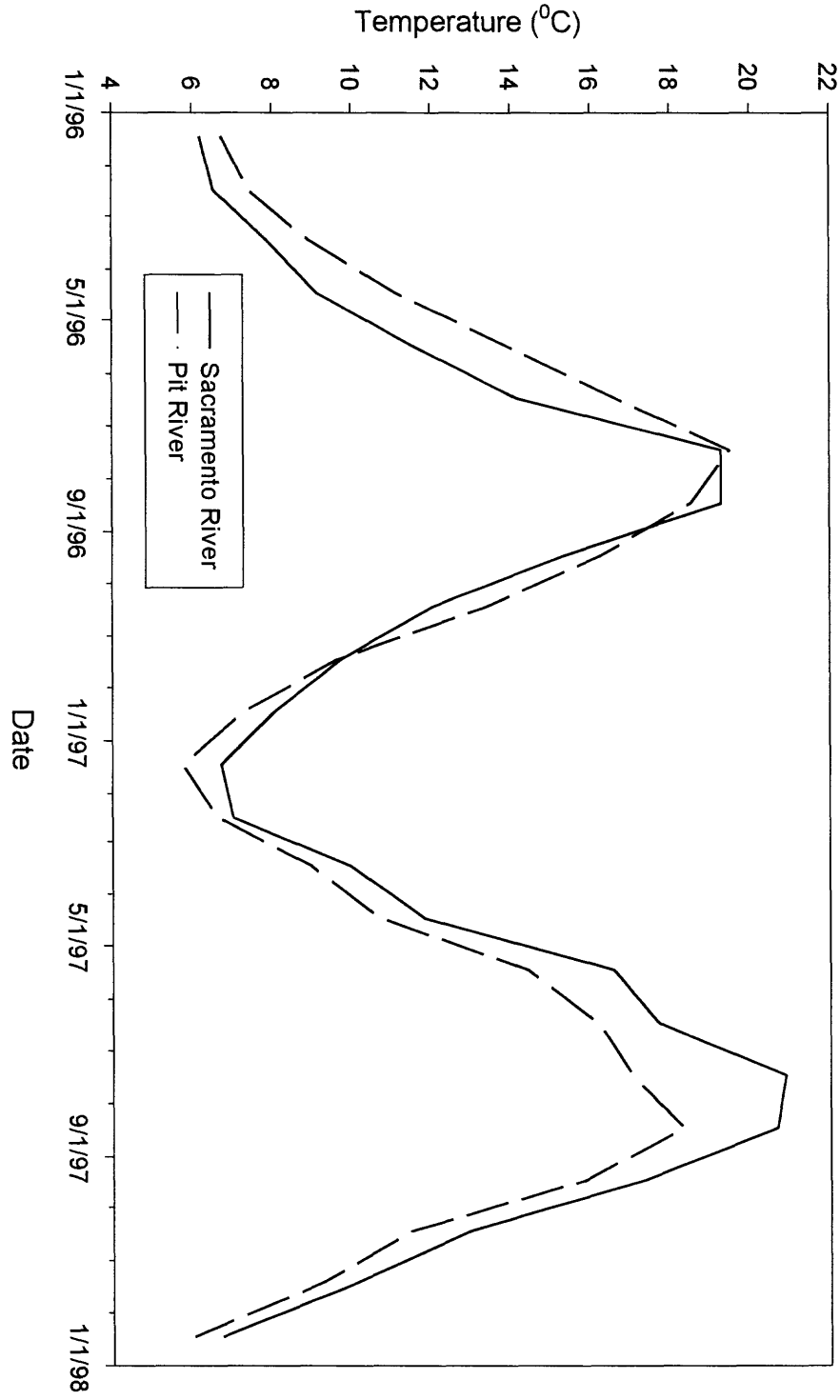


Figure 9. Isopleth description of 1997 flood inflow patterns using turbidity as an indicator for October 1996, January 1997, February 1997, March 1997.

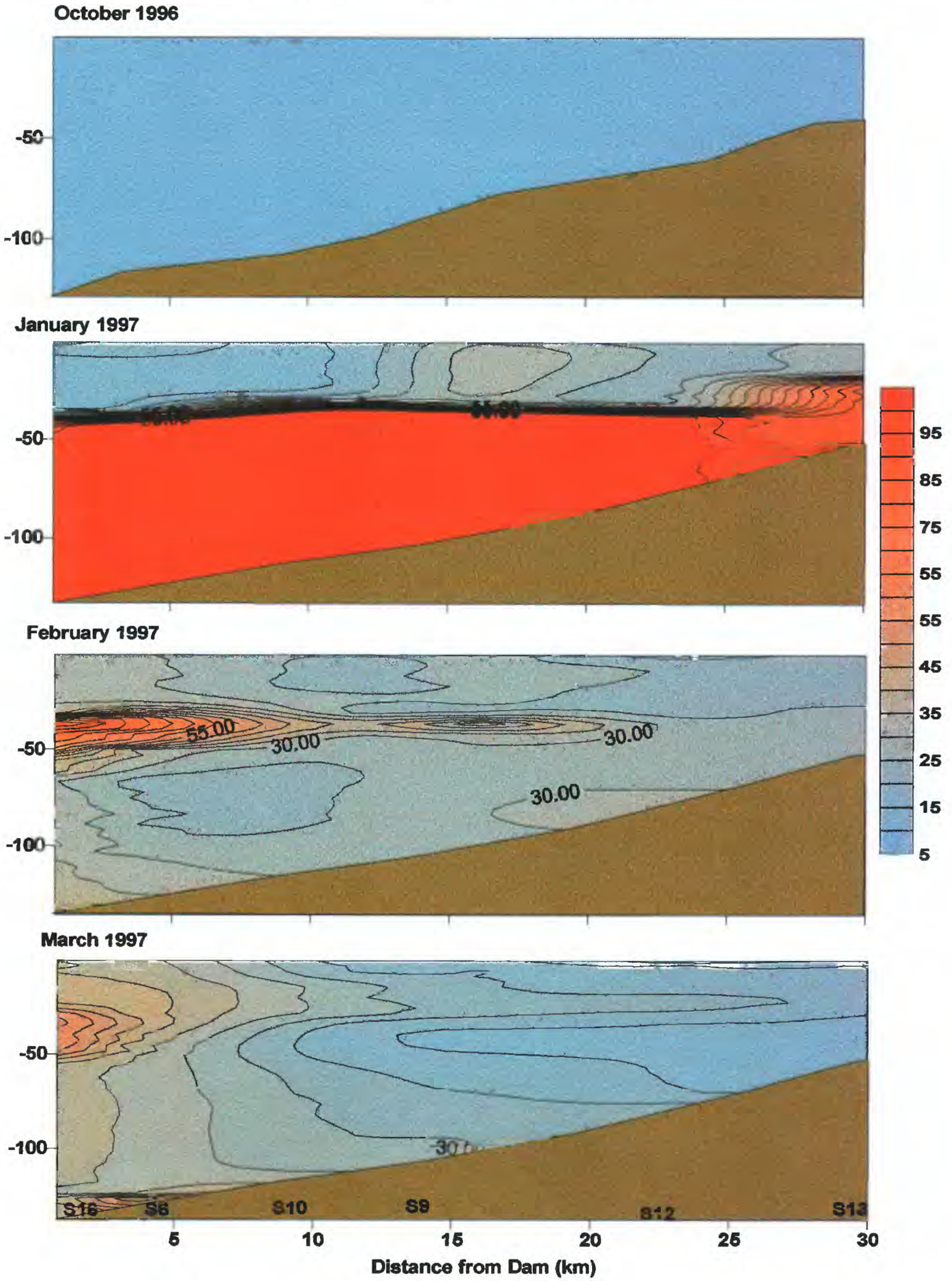
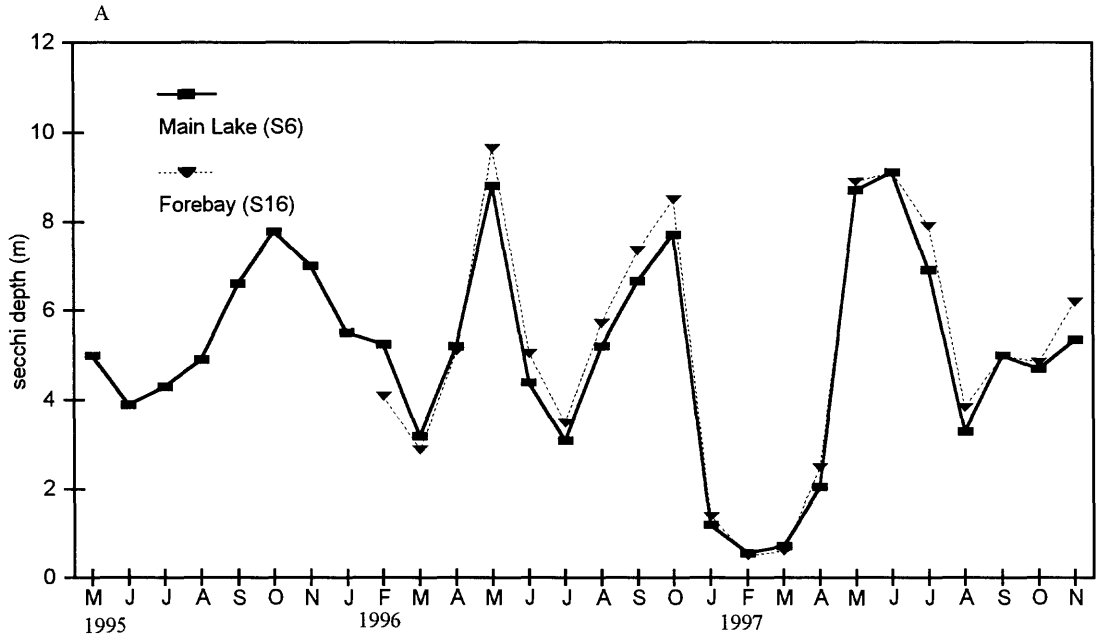


Figure 10 a-b. Secchi depth transparencies (m) for Shasta Lake from May 1995 thru November 1997. a. main lake (S6) and forebay (S16); b. Sacramento River Arm (S7) and upper Sacramento River Arm (S15).



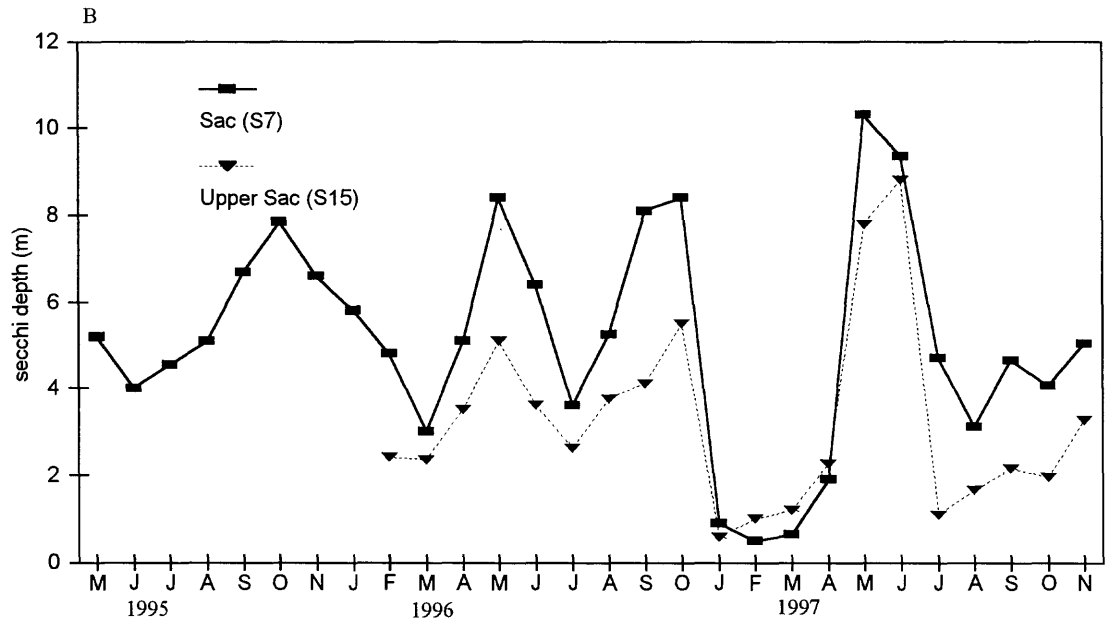
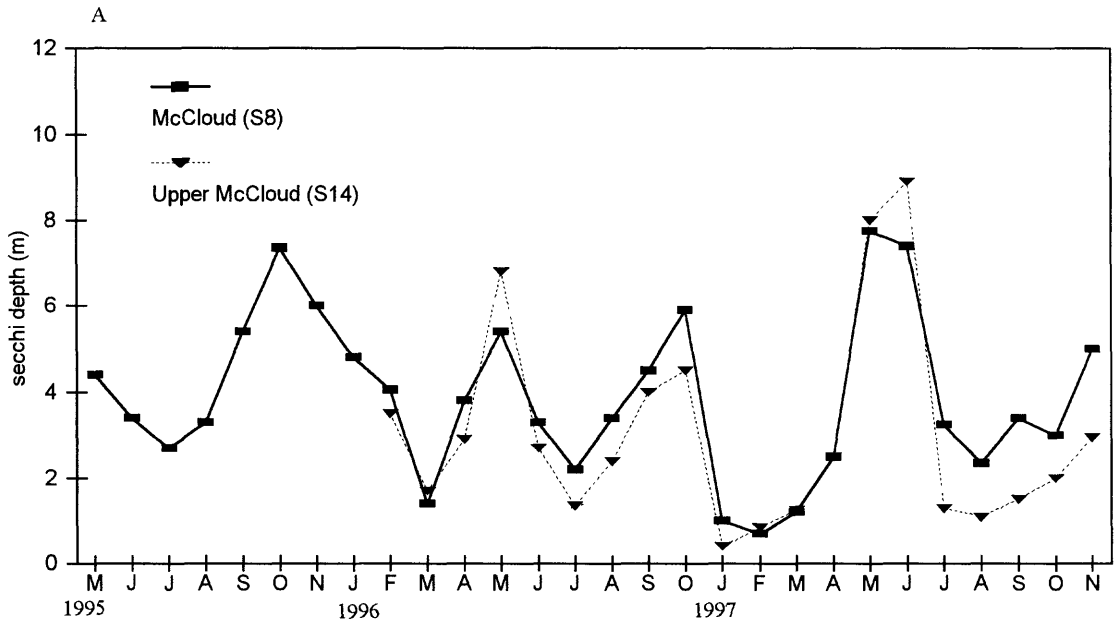


Figure 11 a-b. Secchi depth transparencies (m) for Shasta Lake from May 1995 thru November 1997. a. McCloud River Arm (S8) and upper McCloud River Arm (S14); b. Pit River Arm (S9), upper Pit (S12), upper upper Pit (S13), and Squaw Creek (S11).



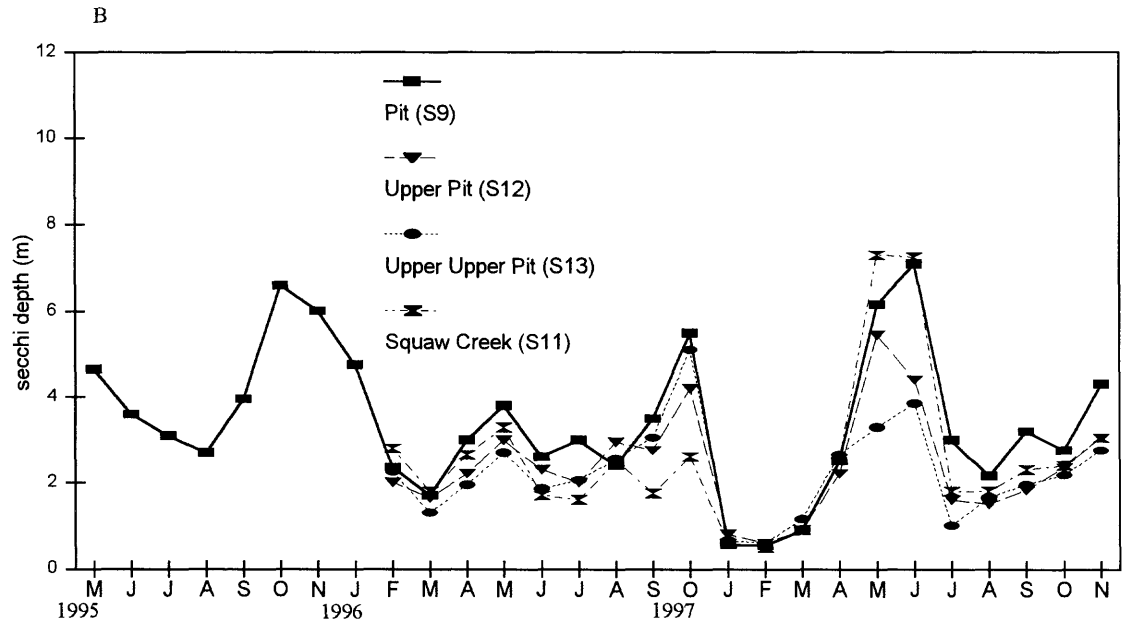


Figure 12. Seasonal patterns of epilimnetic and hypolimnetic nitrate-nitrogen for Shasta Lake sampling stations from 1995-1997.

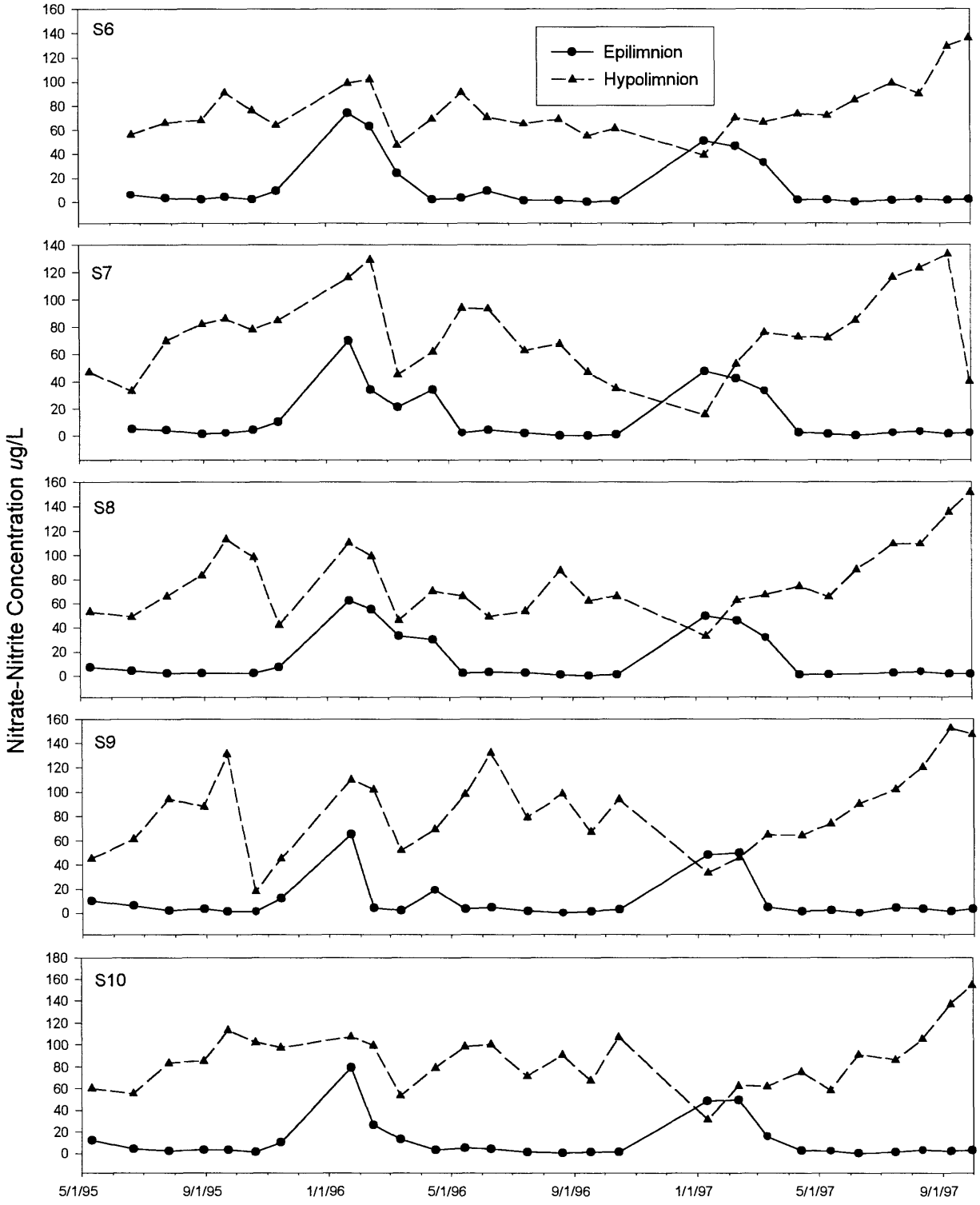


Figure 13. Seasonal patterns of epilimnetic and hypolimnetic orthophospahte (SRP) for Shasta Lake sampling stations from 1995-1997.

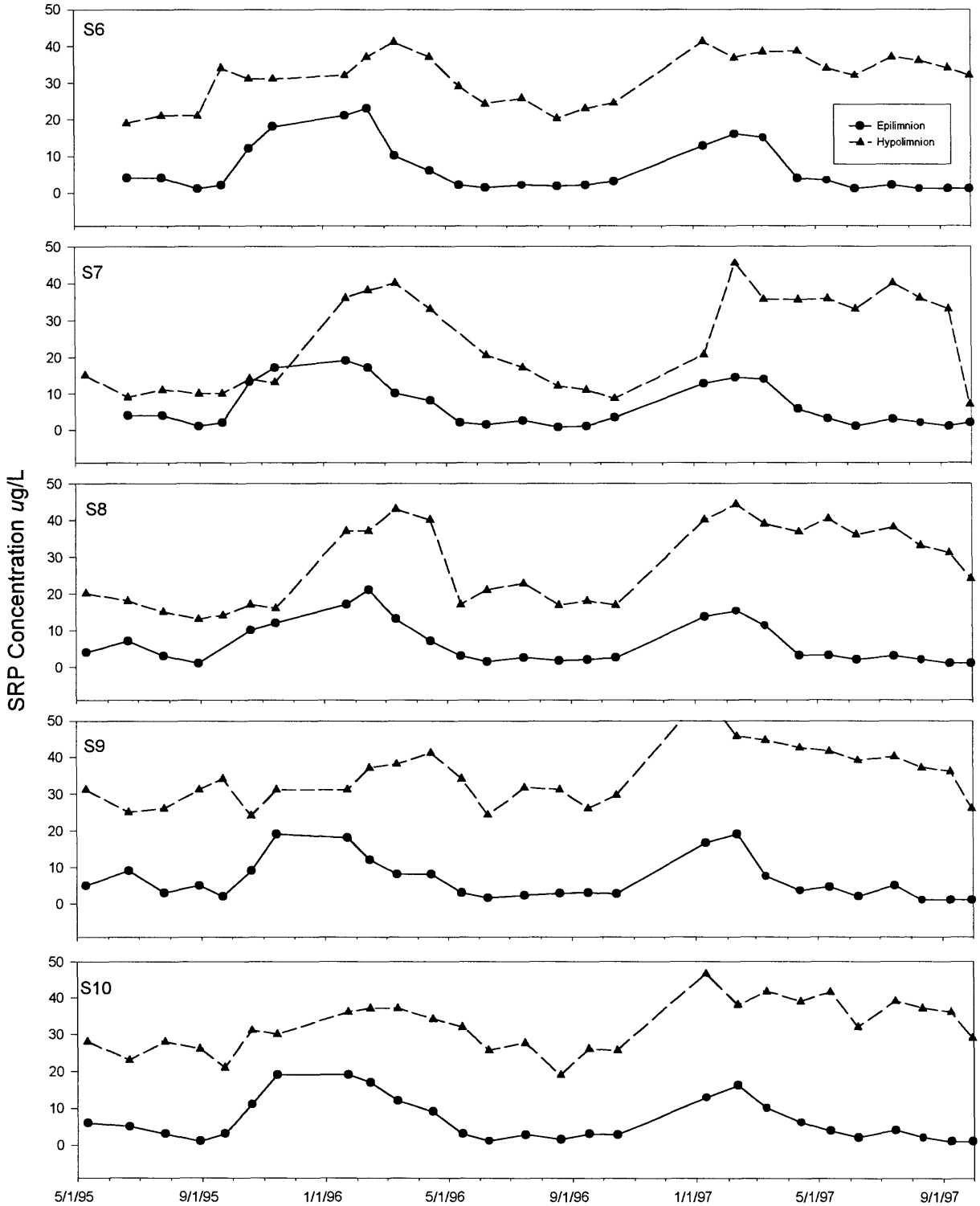


Figure 14. Seasonal patterns of epilimnetic and hypolimnetic ammonia (NH_4) for Shasta Lake sampling stations from 1995-1997.

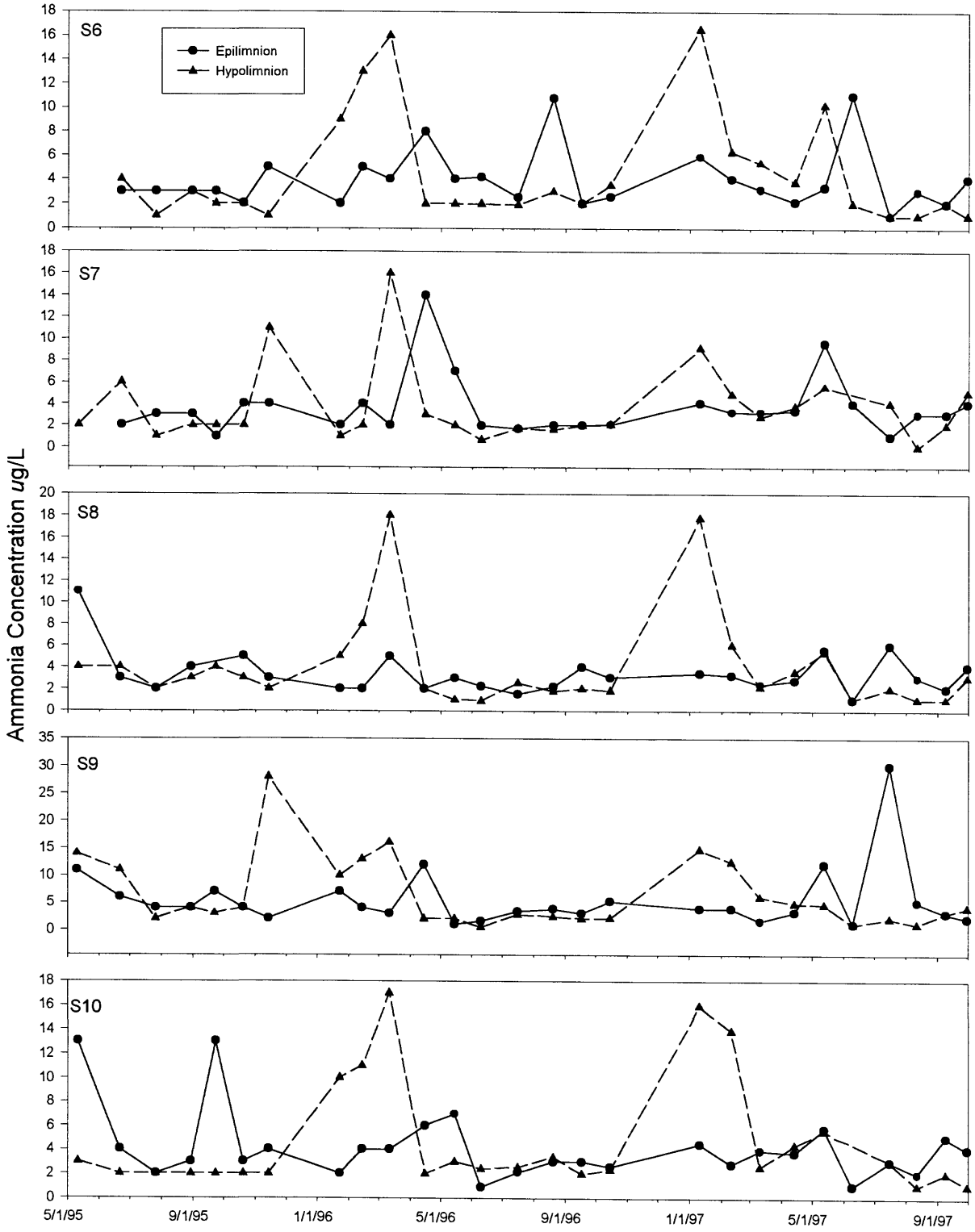


Figure 15. Seasonal patterns of epilimnetic and hypolimnetic total phosphorus (TP) for Shasta Lake sampling stations from 1995-1997.

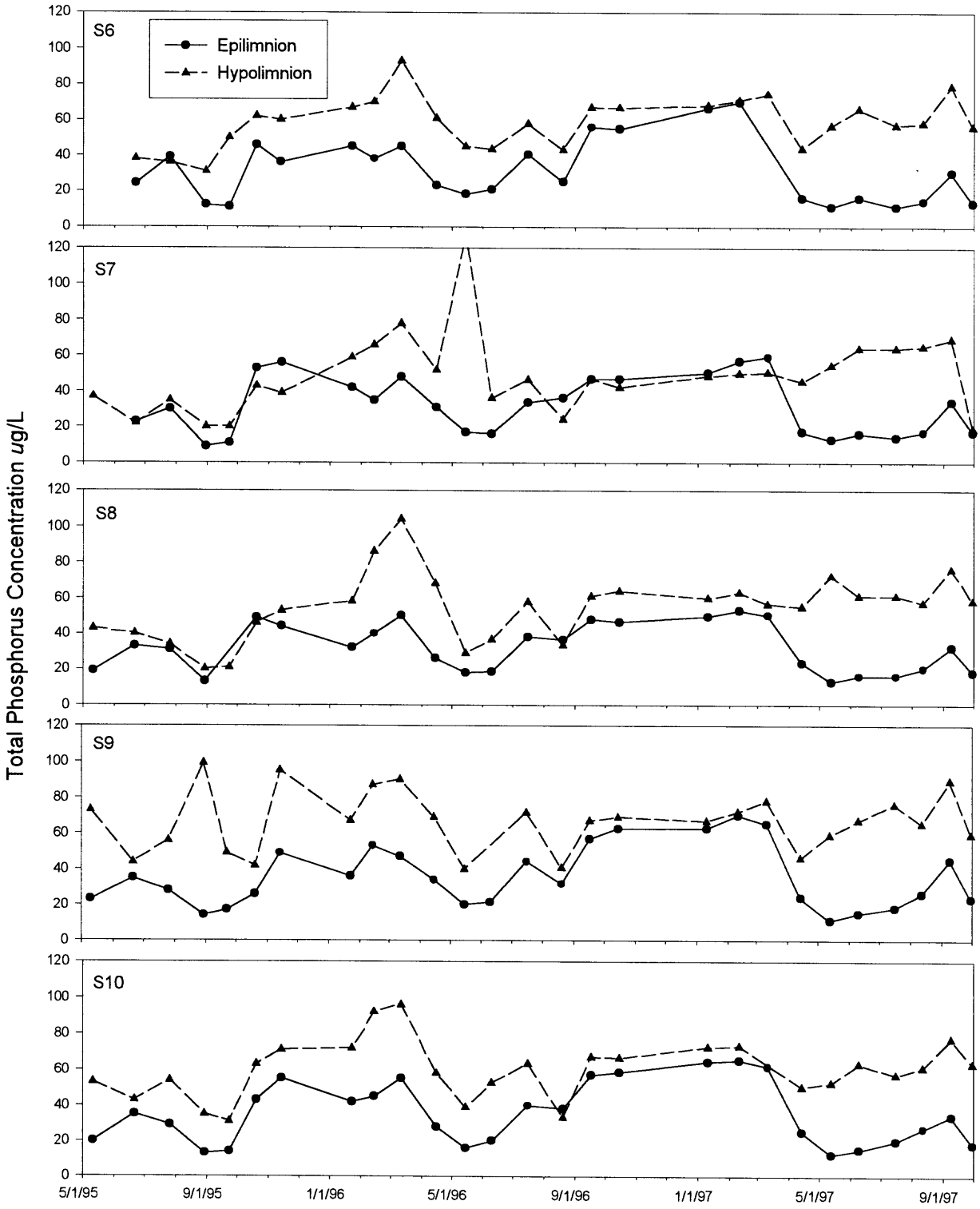


Figure 16. Seasonal patterns of epilimnetic and hypolimnetic TP/SRP ratios for Shasta Lake sampling stations from 1995-1997.

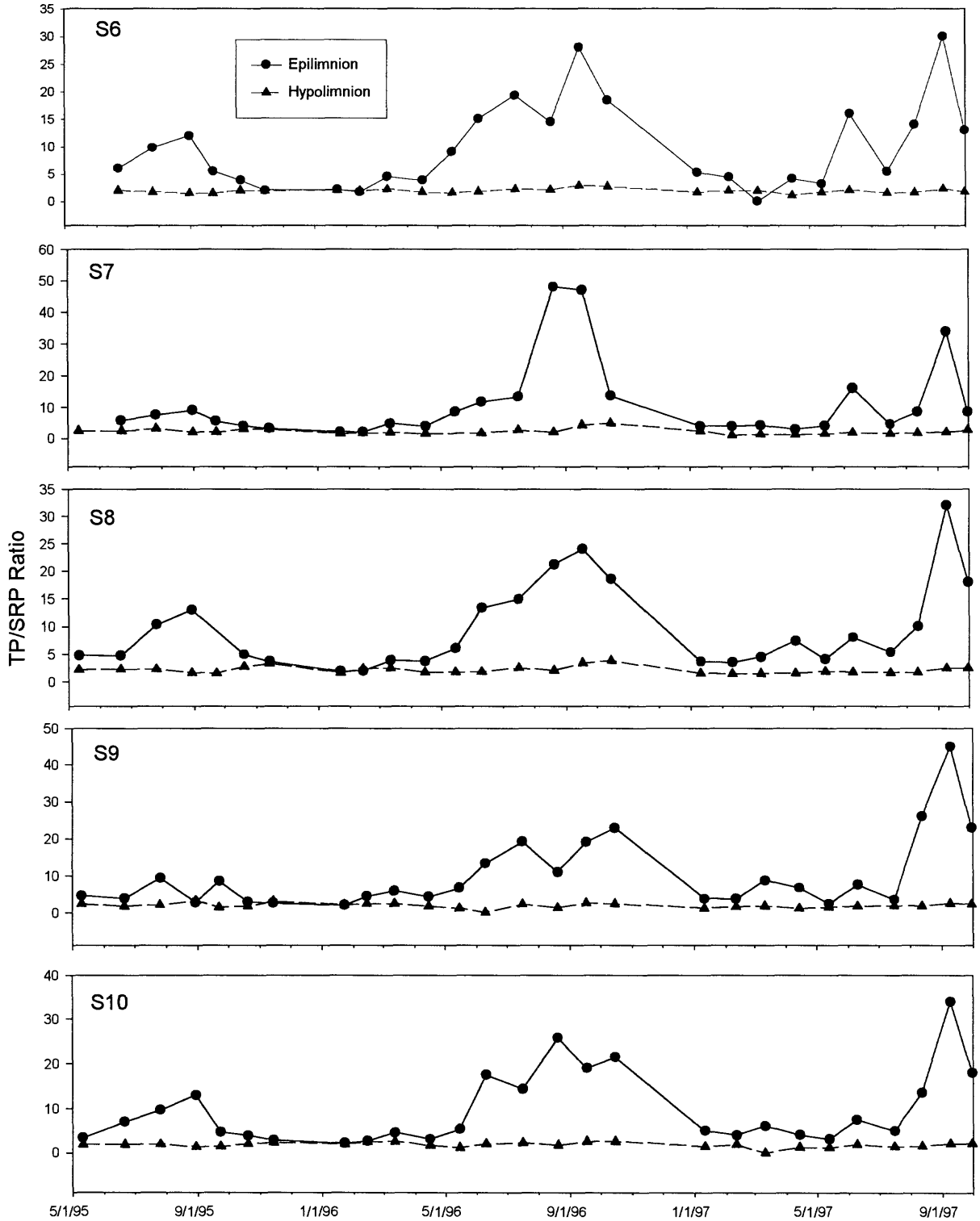


Figure 17. Seasonal patterns of epilimnetic and hypolimnetic DIN/SRP ratios for Shasta Lake sampling stations from 1995-1997.

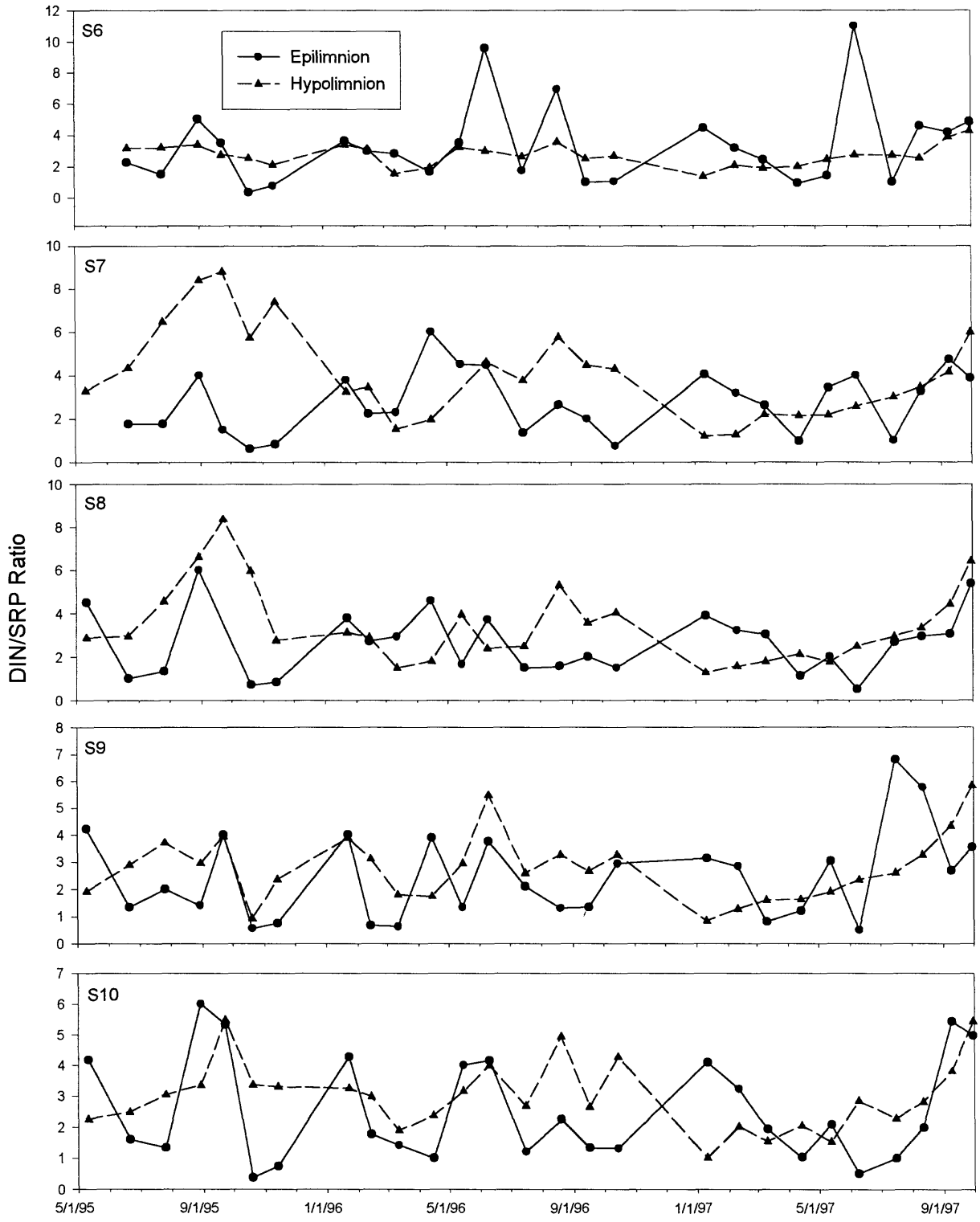


Figure 18. Comparison of total composite chl *a* concentrations (µg/L) (0-5 m) between stations from 1995 thru 1997.

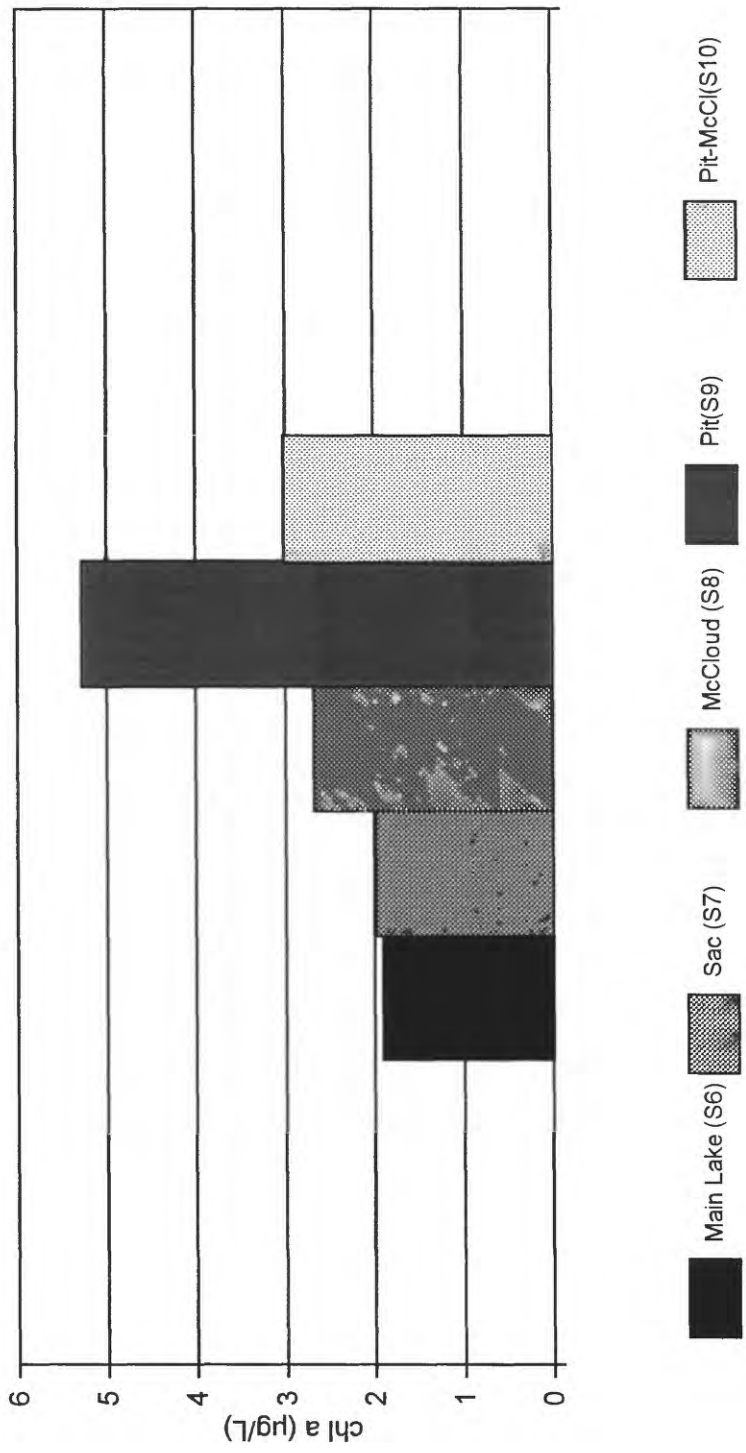
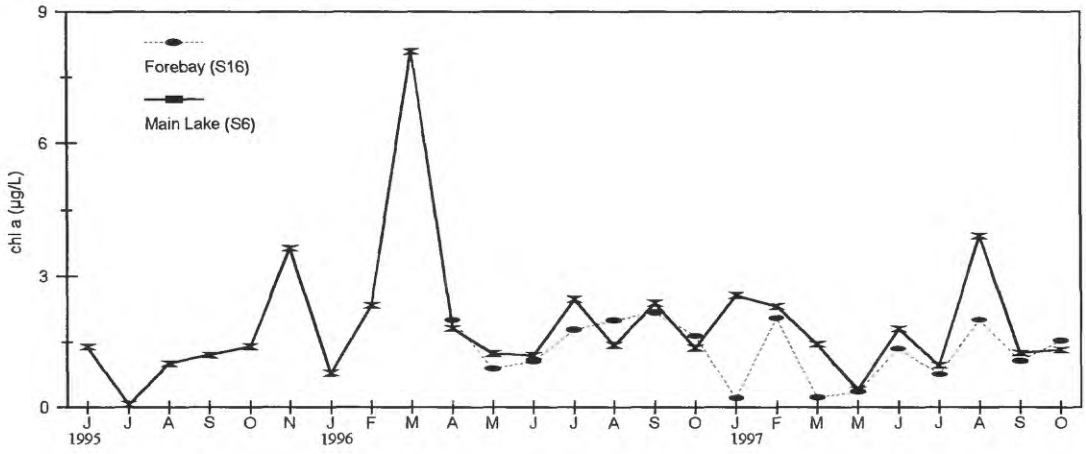
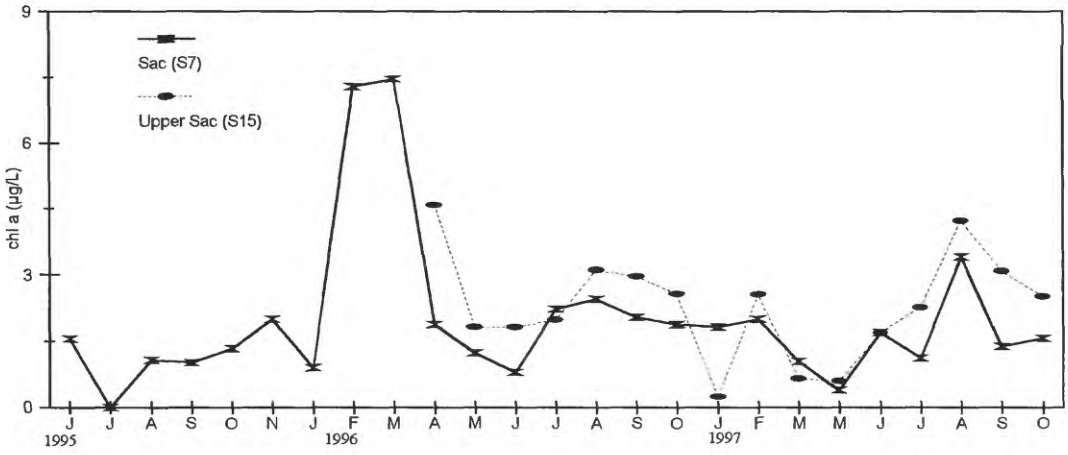


Figure 19 a-d. Seasonal trend of composite chl *a* concentrations (µg/L) (0 to 5 m) from May 1995 thru October 1997. a. main lake (S6) and forebay (S16); b. Sacramento River Arm (S7) and upper Sacramento River Arm (S16); c. McCloud River Arm (S8) and upper McCloud River Arm (S14); d. Pit River Arm (S9), upper Pit (S12), upper upper Pit (S13), Squaw Creek (S11), and Pit-McCloud Confluence (S10).

a.



b.



c.

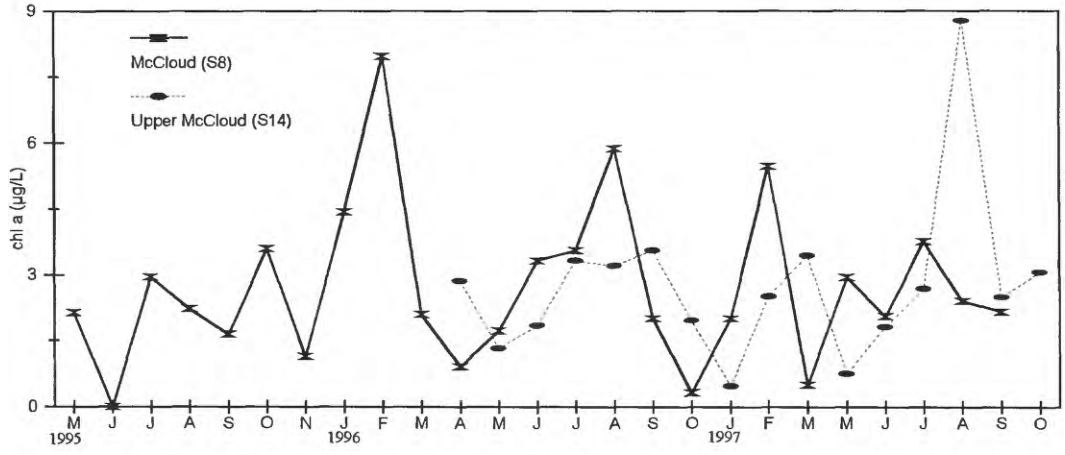


Figure 19 cont.
d.

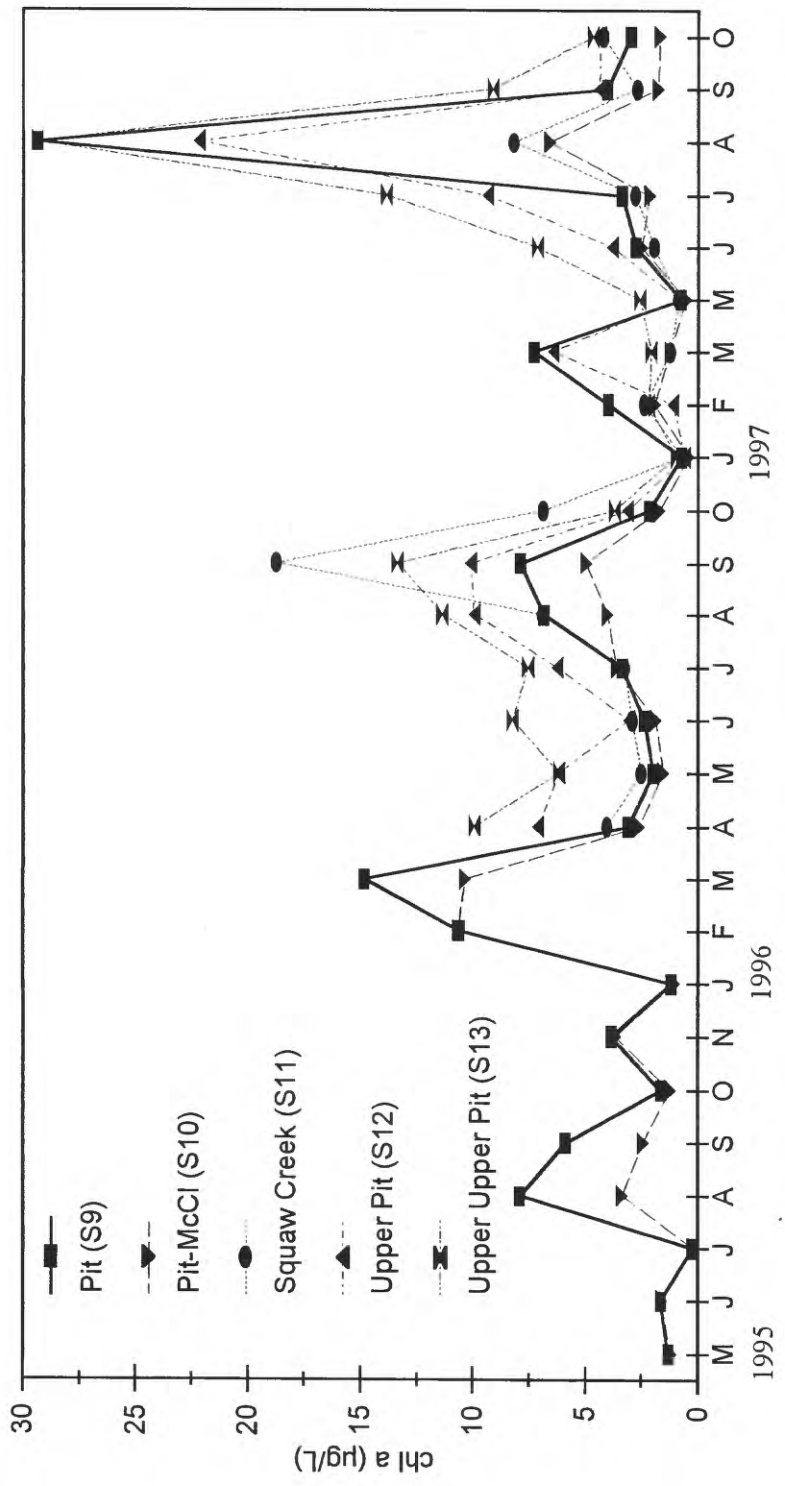


Figure 20 a-c. Seasonal trend of dominant phytoplankton groups collected from 0-30 m from the main lake (S6). a. June 1995 thru November 1996; b. January 1996 thru October 1996; c. January 1997 thru October 1997.

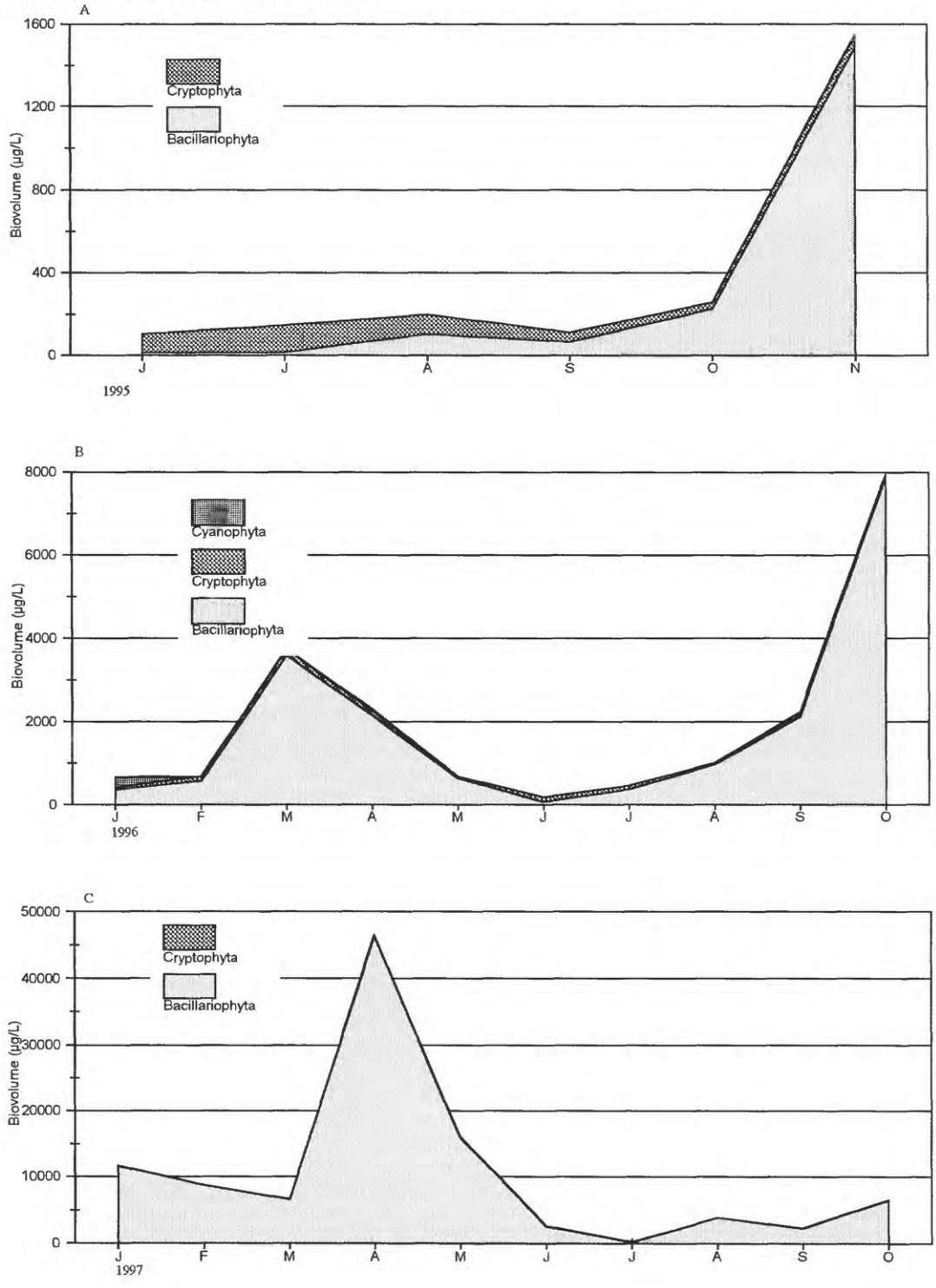
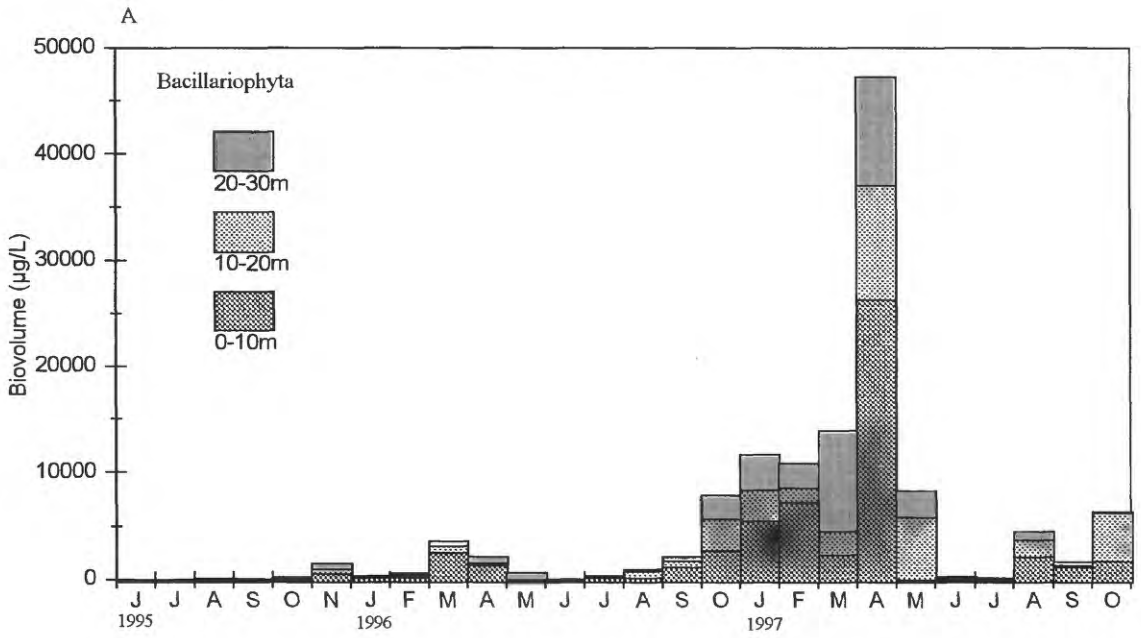


Figure 21 a-b. Seasonal depth distribution of phytoplankton collected from 0-10 m, 10-20 m, and 20-30 m in the main lake (S6) from June 1995 thru October 1997. a. bacillariophyta; b. cryptophyta.



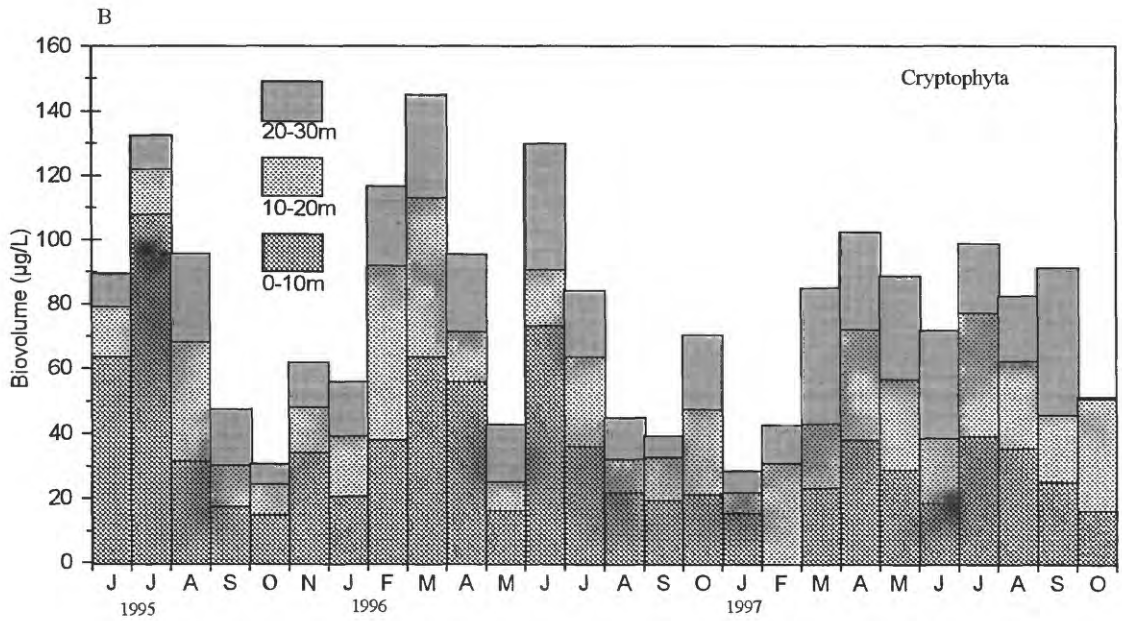
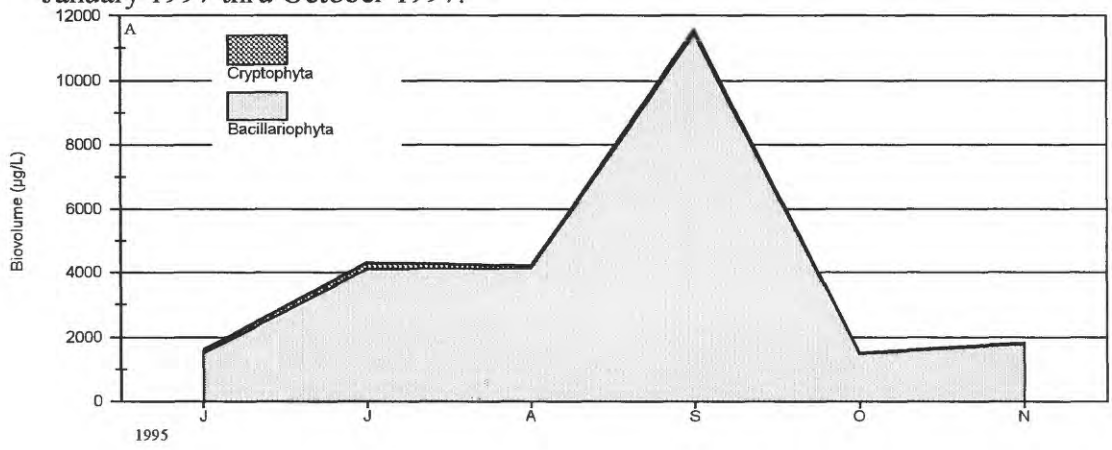
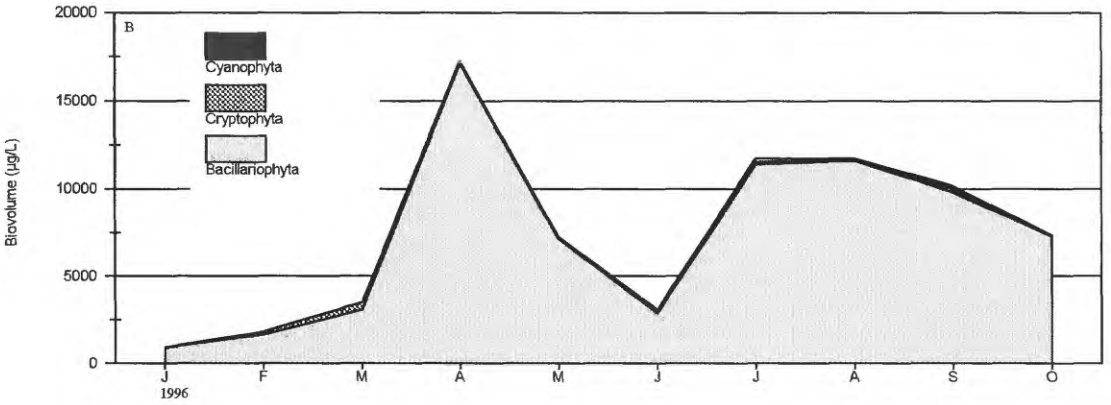


Figure 22 a-c. Seasonal trend of dominant phytoplankton groups collected from 0-30 m in the Pit River Arm (S9). a. June 1995 thru November 1995; b. January 1996 thru October 1996; c. January 1997 thru October 1997.





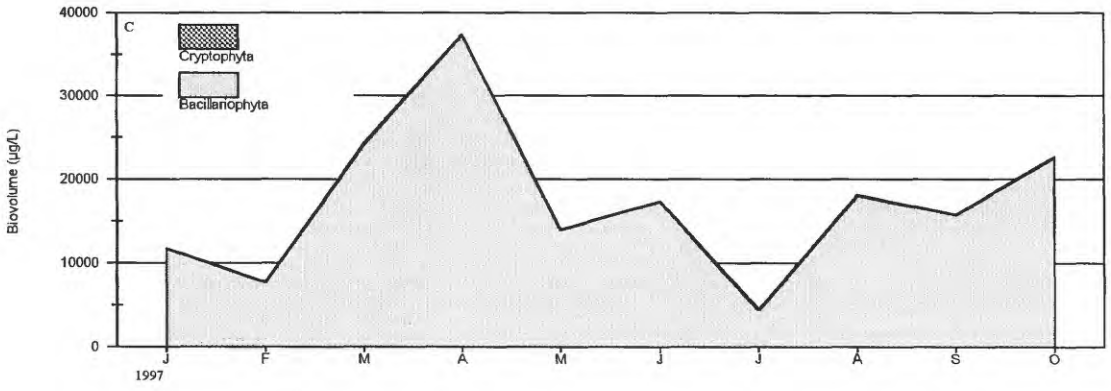


Figure 23 a-b. Seasonal depth distribution of phytoplankton collected from 0-10 m, 10-20 m, and 20-30 m in the Pit River Arm (S9) from June 1995 thru October 1997. a. bacillariophyta; b. cryptophyta.

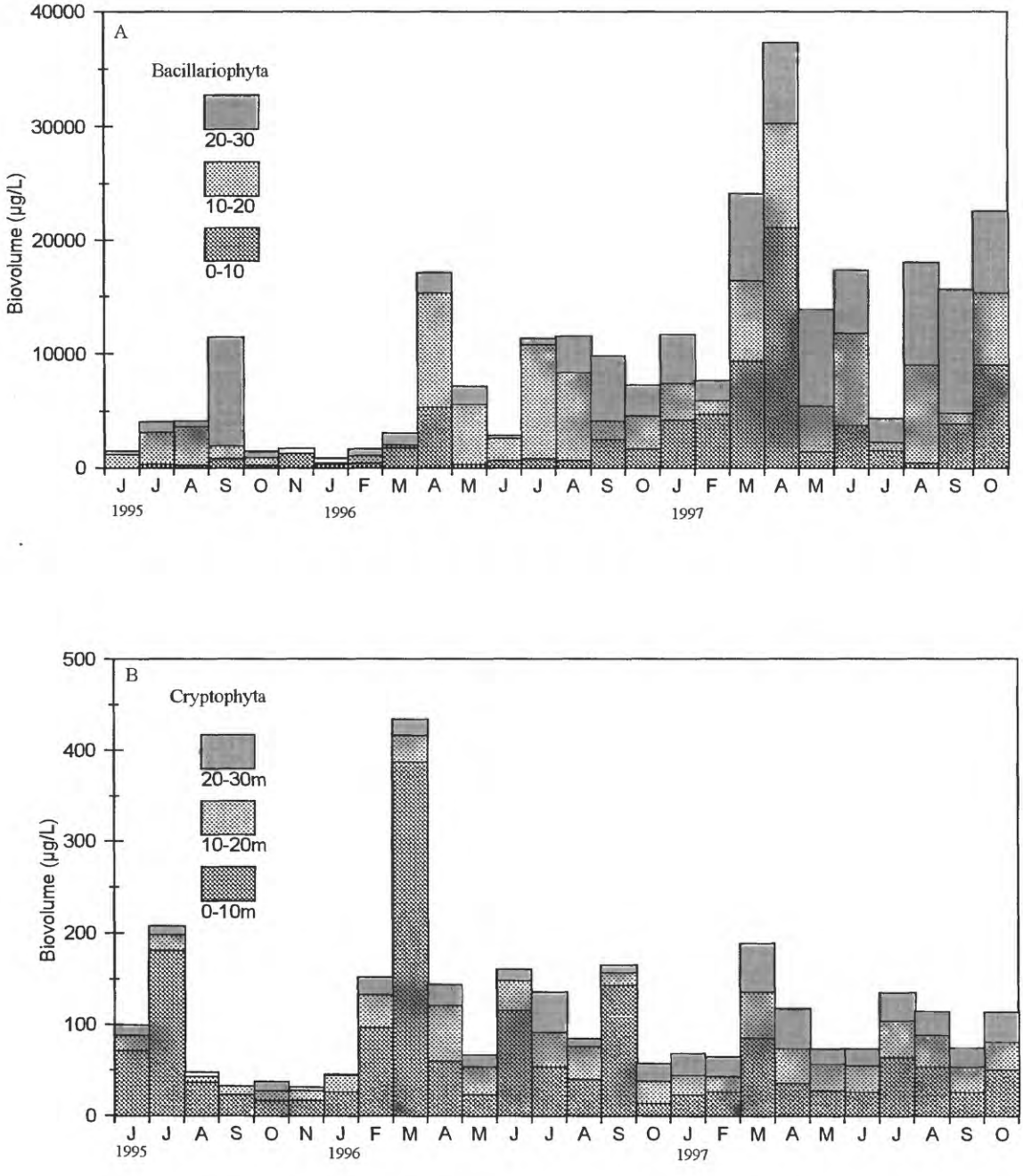


Figure 24. Mean total copepod and cladoceran biomass (µg/L) for for all sampling dates for the main lake (S6), Sacramento River Arm (S7), McCloud River Arm (S8), Pit River Arm (S9), and Pit-McCloud Confluence (S10) from 1995 thru 1997.

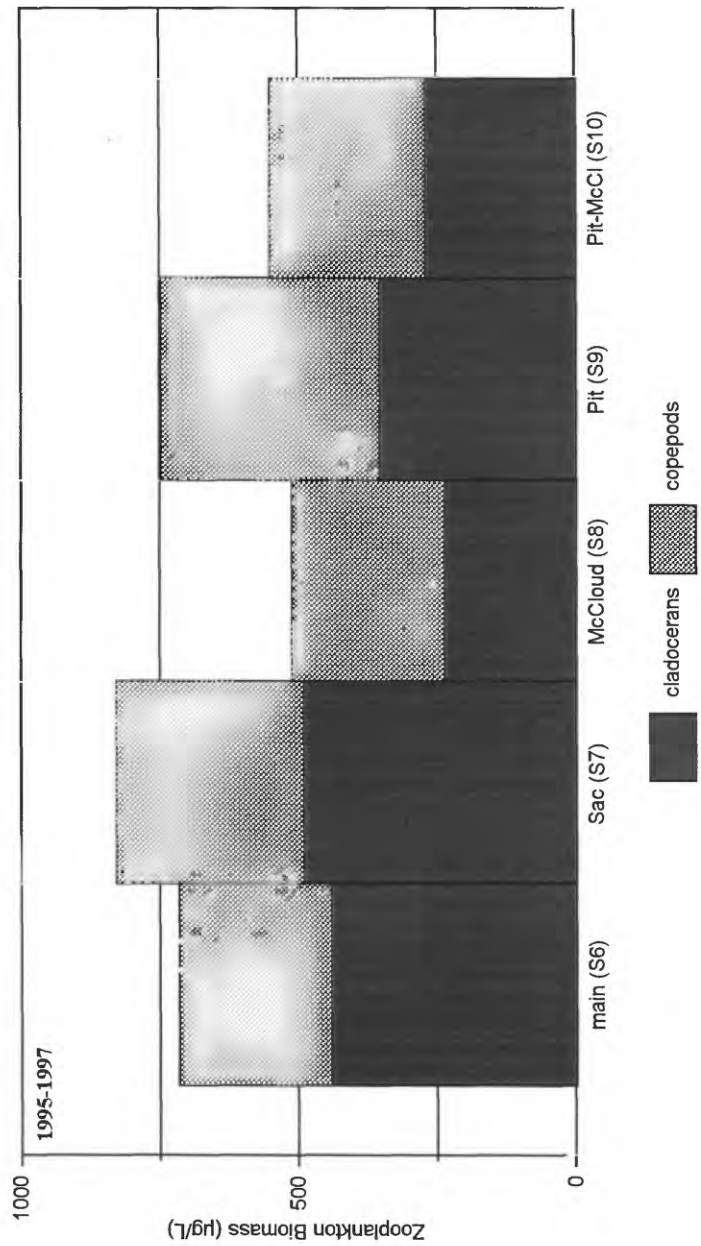


Figure 25. Mean copepod and cladoceran biomass (µg/L) for the main lake (S6), Sacramento River Arm (S7), McCloud River Arm (S8), Pit River Arm (S9), and Pit-McCloud Confluence (S10) for 1995, 1996 and 1997.

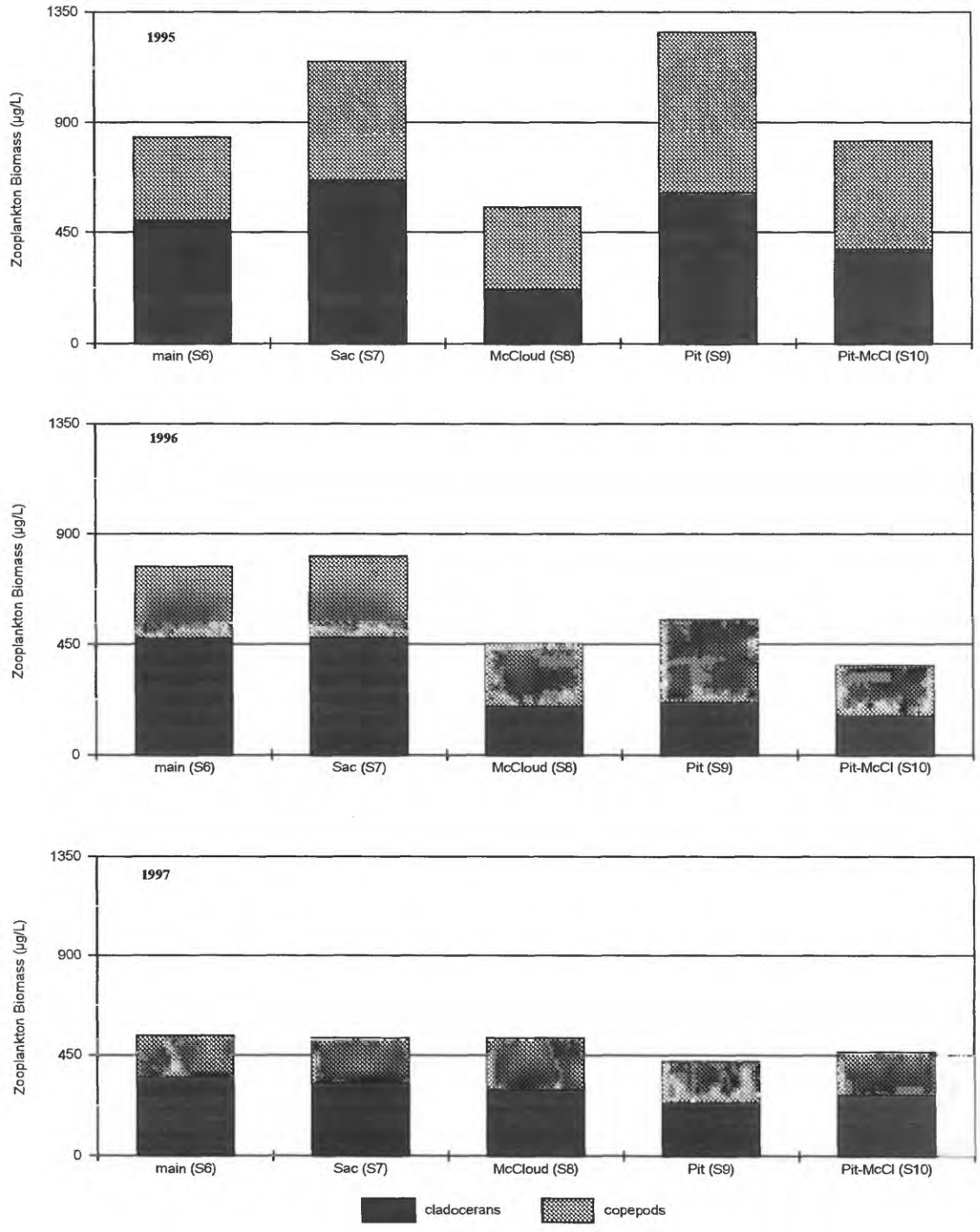
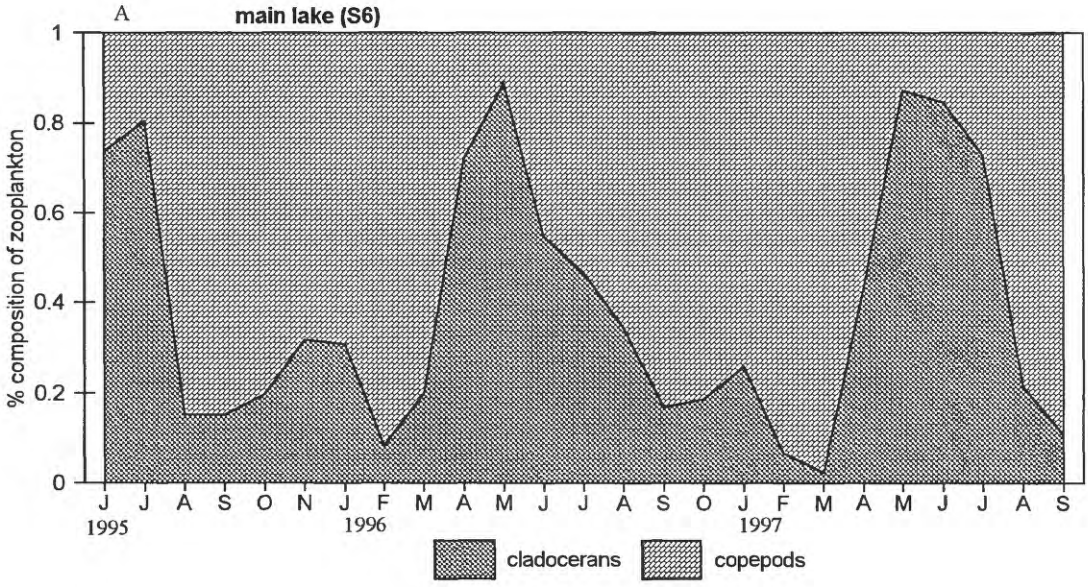


Figure 26 a-b. Trends in zooplankton abundance (0-30 m) in the main lake (S6) from June 1995 thru September 1997. a. Percent composition of copepods and cladocerans; b. Cladoceran, copepod, and rotifer total biomass (µg/L) trends.



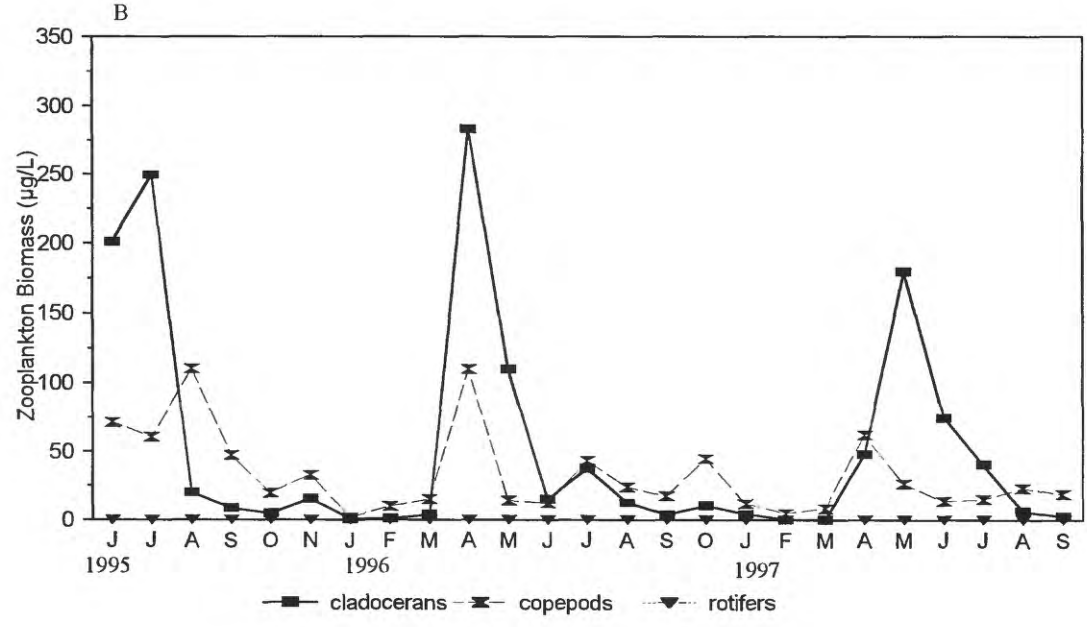


Figure 27 a-b. Seasonal distribution of copepod and cladoceran biomass in 0-10 m, 10-20 m, and 20-30 m depth intervals in the main lake (S6) from June 1995 thru September 1997. a. copepods; b. cladocerans.

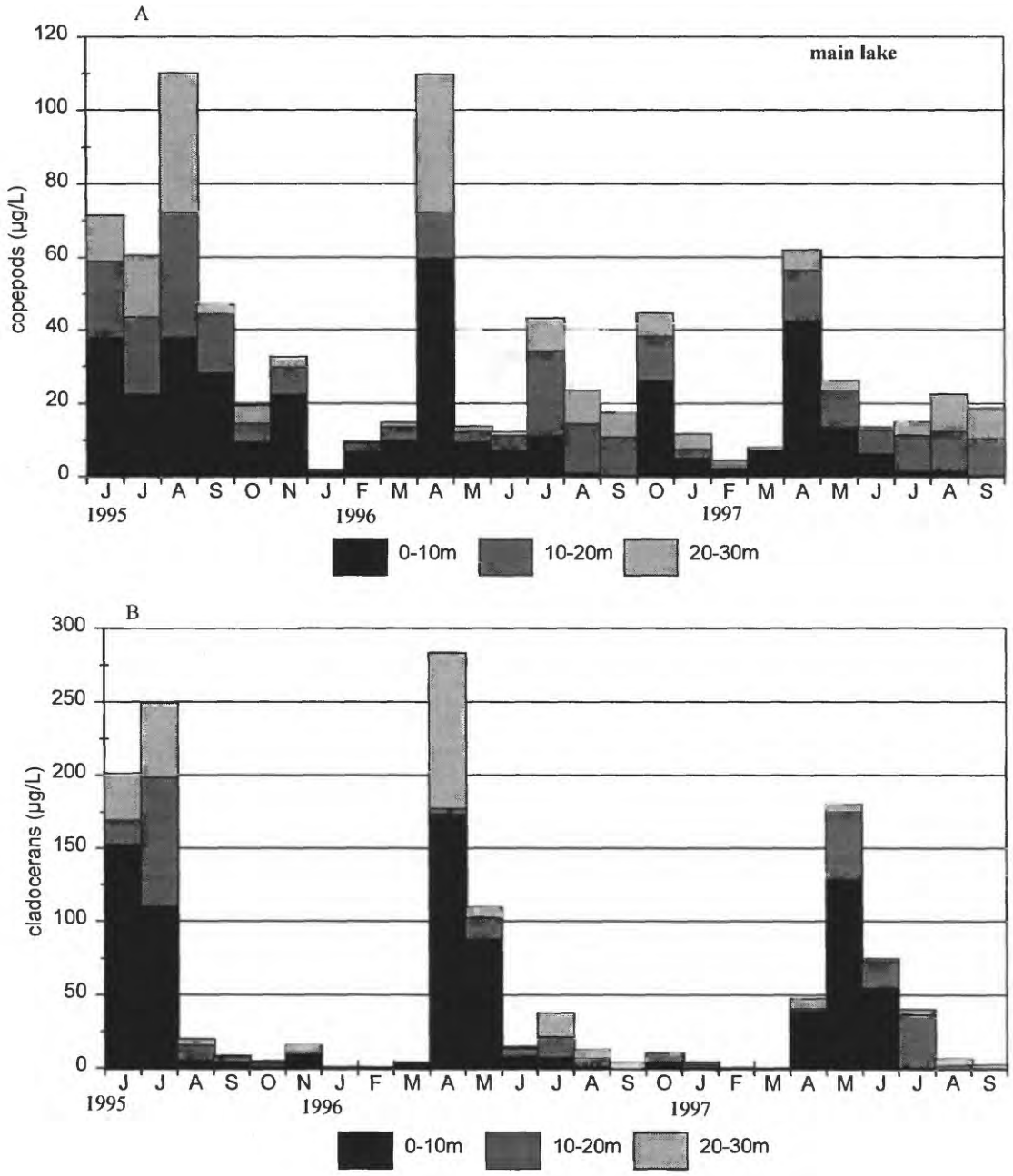
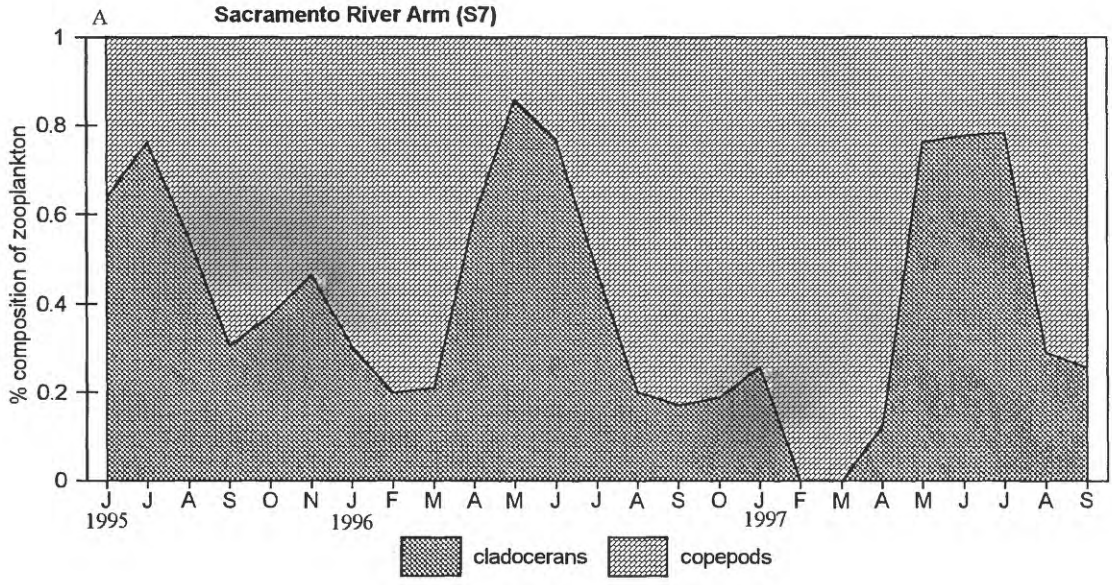


Figure 28 a-b. Trends in zooplankton abundance (0-30 m) for the Sacramento River Arm (S7) from June 1995 thru September 1997. a. Percent composition of copepods and cladocerans; b. Cladoceran, copepod, and rotifer total biomass (µg/L) trends.



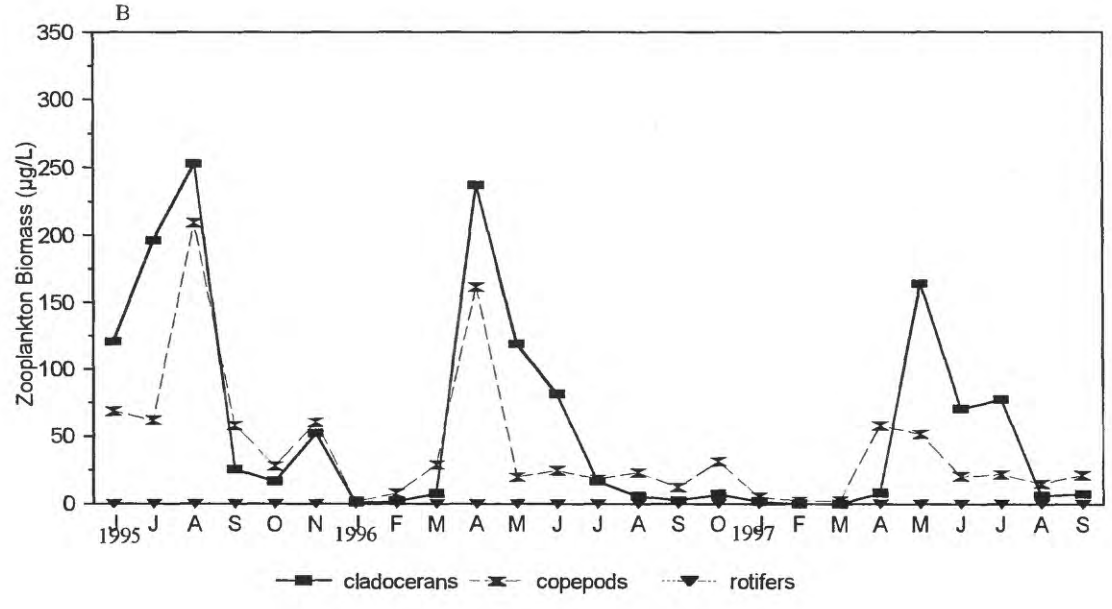


Figure 29 a-b. Seasonal distribution of copepod and cladoceran biomass in 0-10 m, 10-20 m, and 20-30 m depth intervals in the Sacramento River Arm (S7) from June 1995 thru September 1997. a. copepods; b. cladocerans.

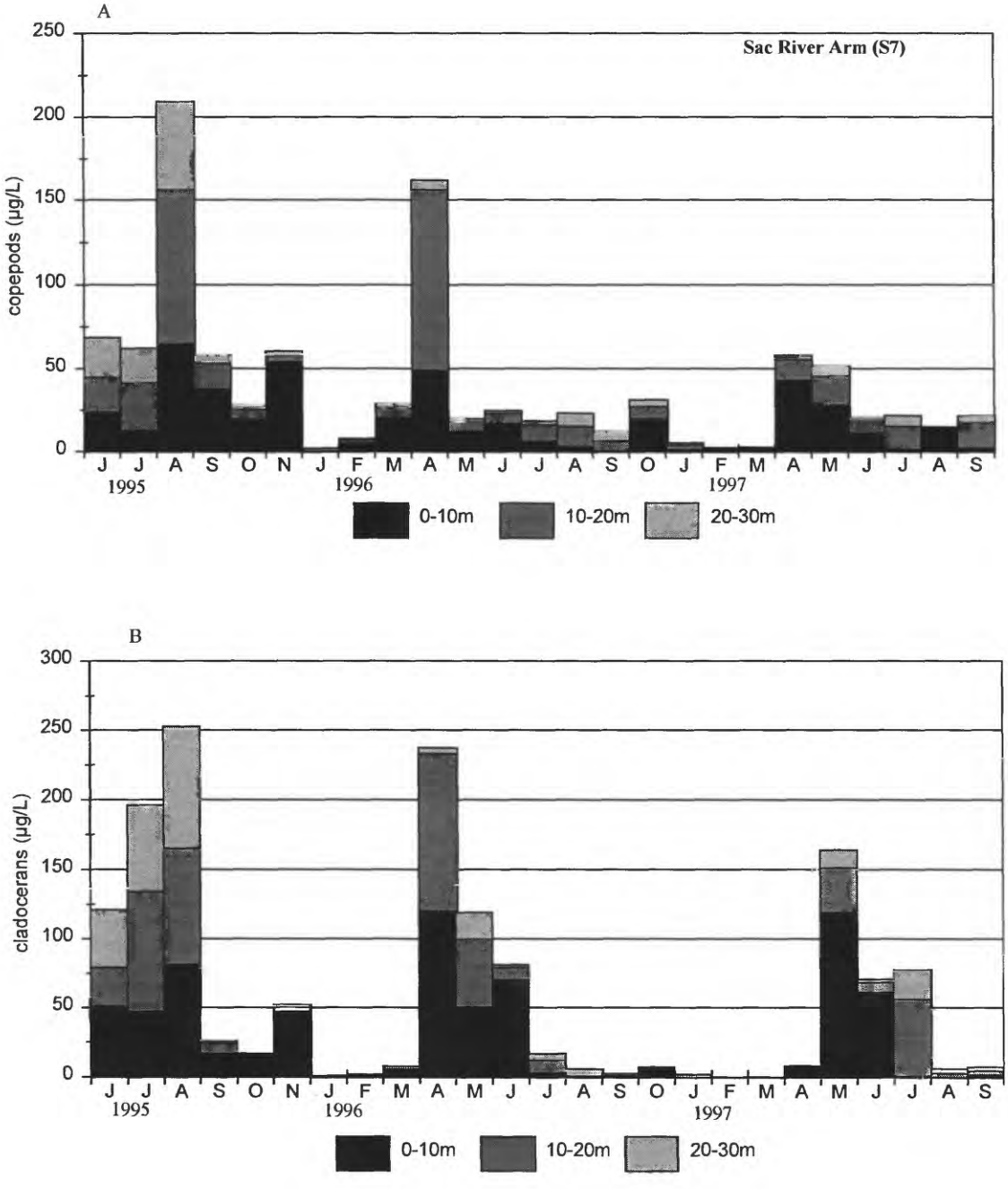
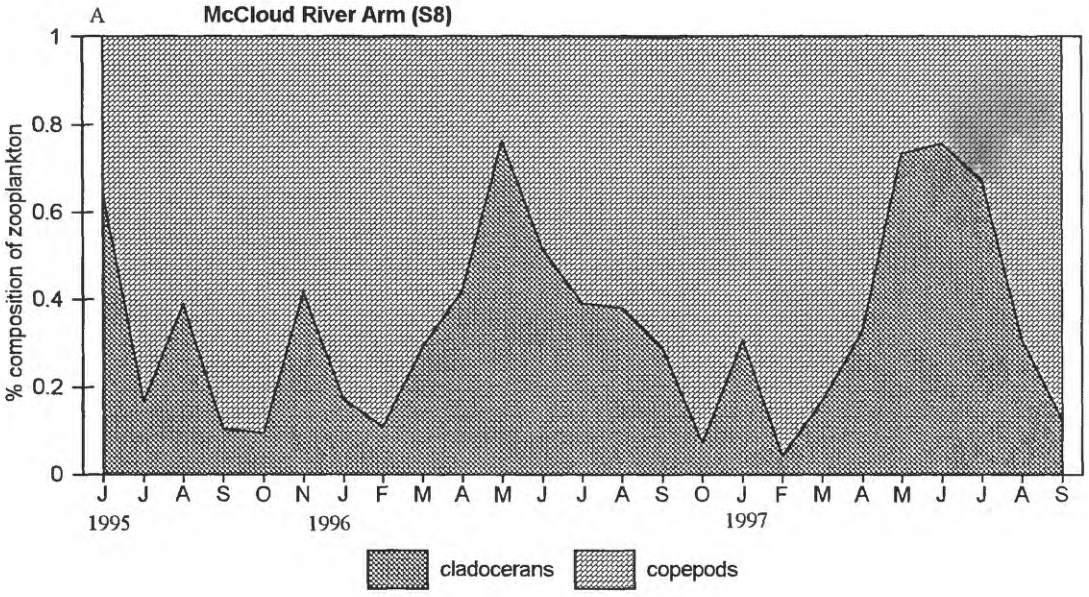


Figure 30 a-b. Trends in zooplankton abundance (0-30 m) for the McCloud River Arm (S8) from June 1995 thru September 1997. a. Percent composition of copepods and cladocerans; b. Cladoceran, copepod, and rotifer total biomass (µg/L) trends.



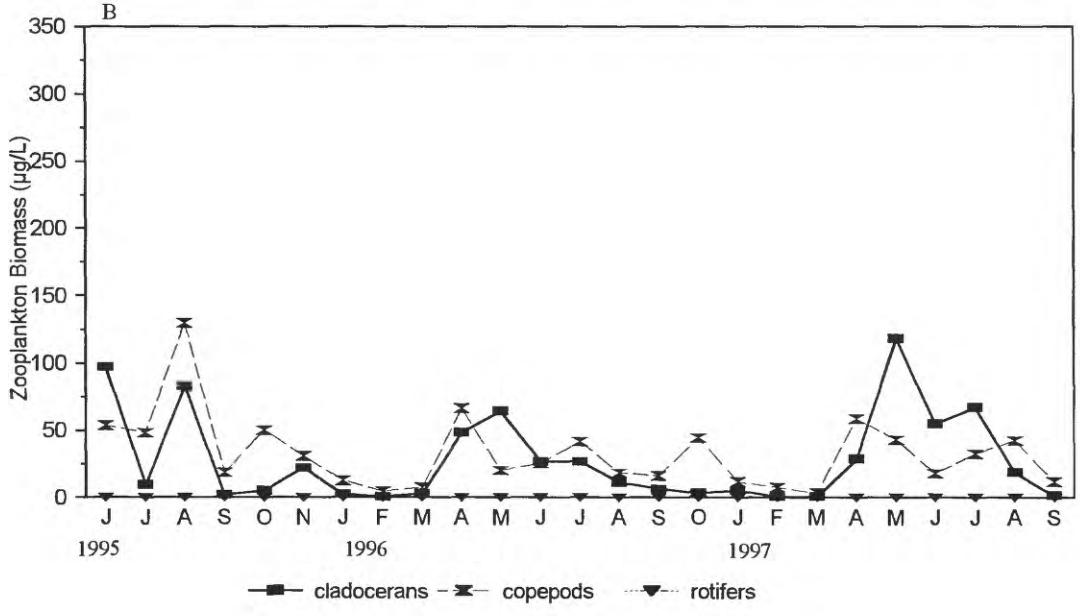


Figure 31 a-b. Seasonal distribution of copepod and cladoceran biomass in 0-10 m, 10-20 m, and 20-30 m depth intervals in the McCloud River Arm (S8) from June 1995 thru September 1997. a. copepods; b. cladocerans

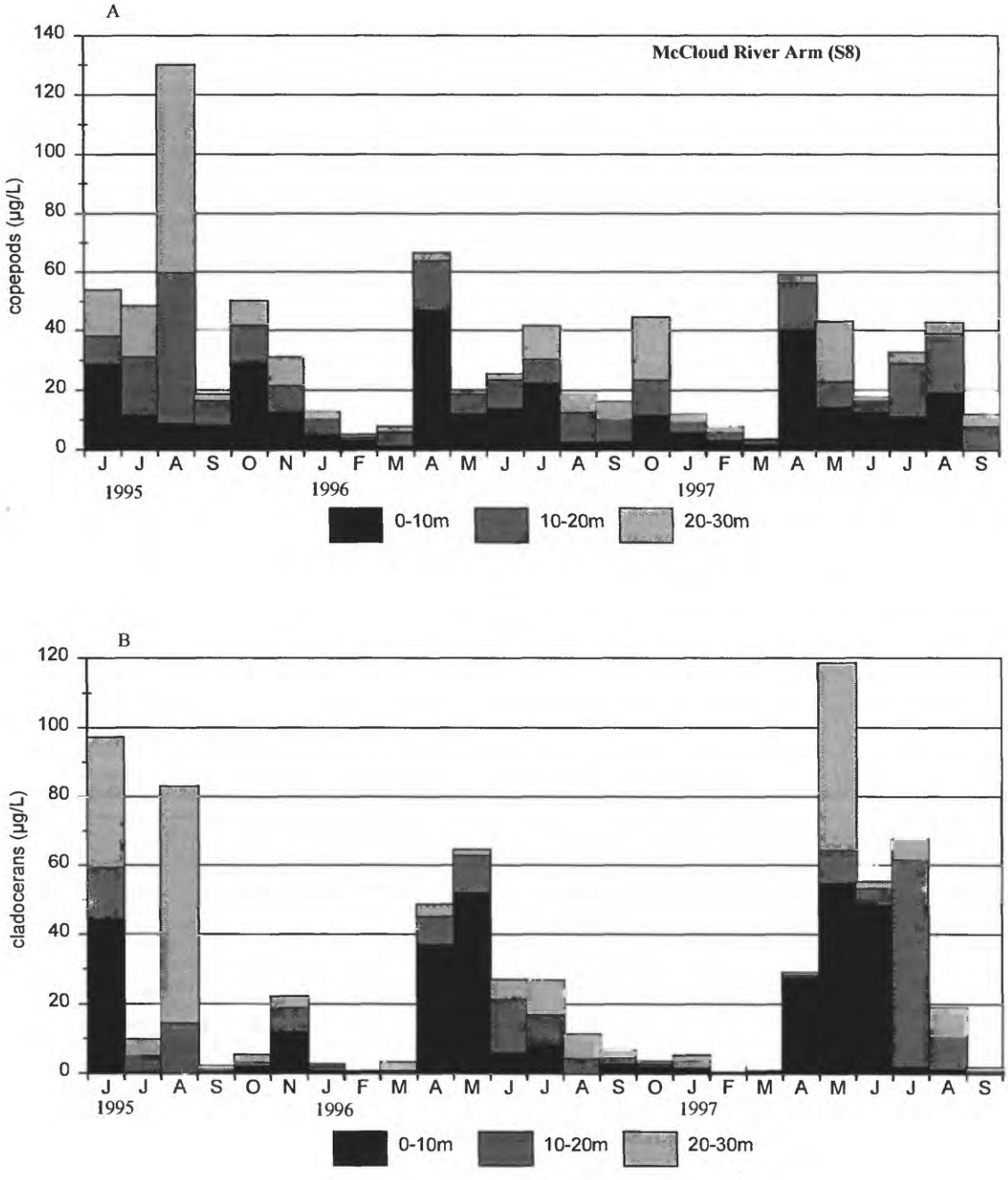
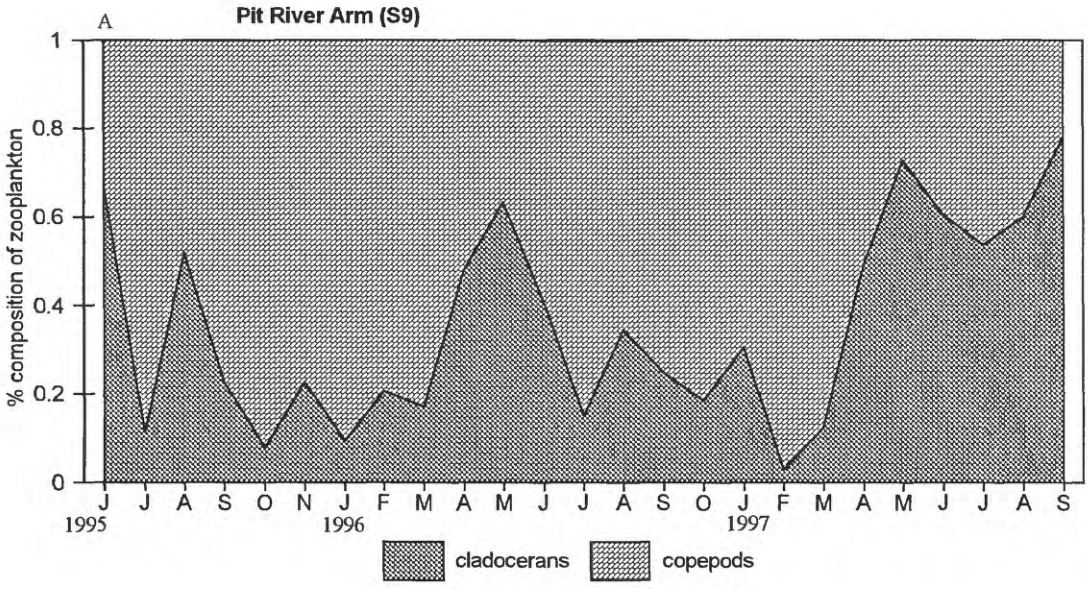


Figure 32 a-b. Trends in zooplankton abundance (0-30m) for the Pit River Arm (S9) from June 1995 thru September 1997. a. Percent composition of copepods and cladocerans; b. Cladoceran, copepod, and rotifer total biomass(µg/L) trends.



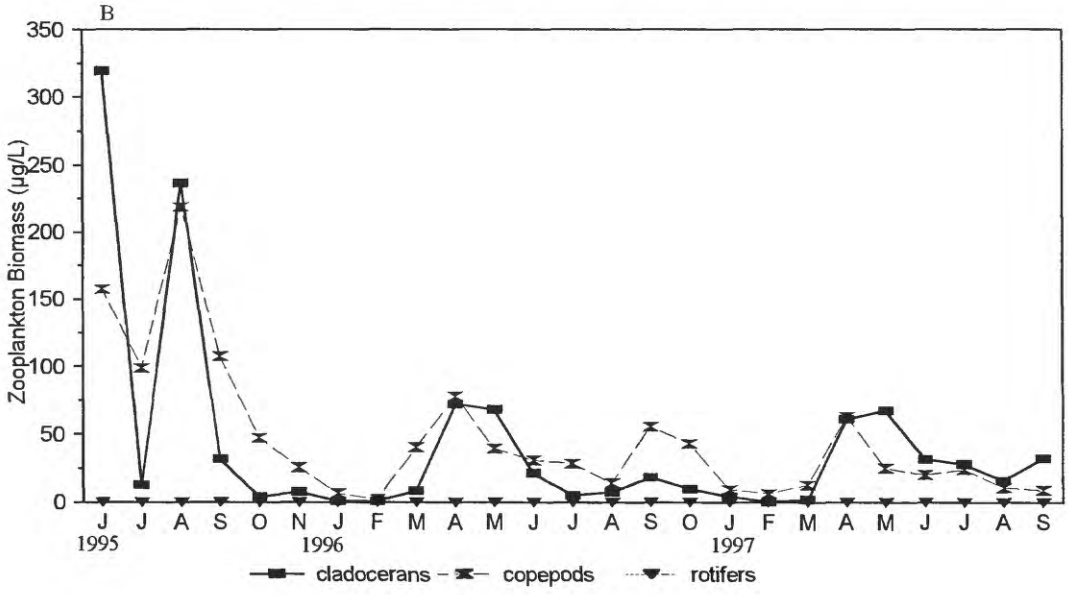
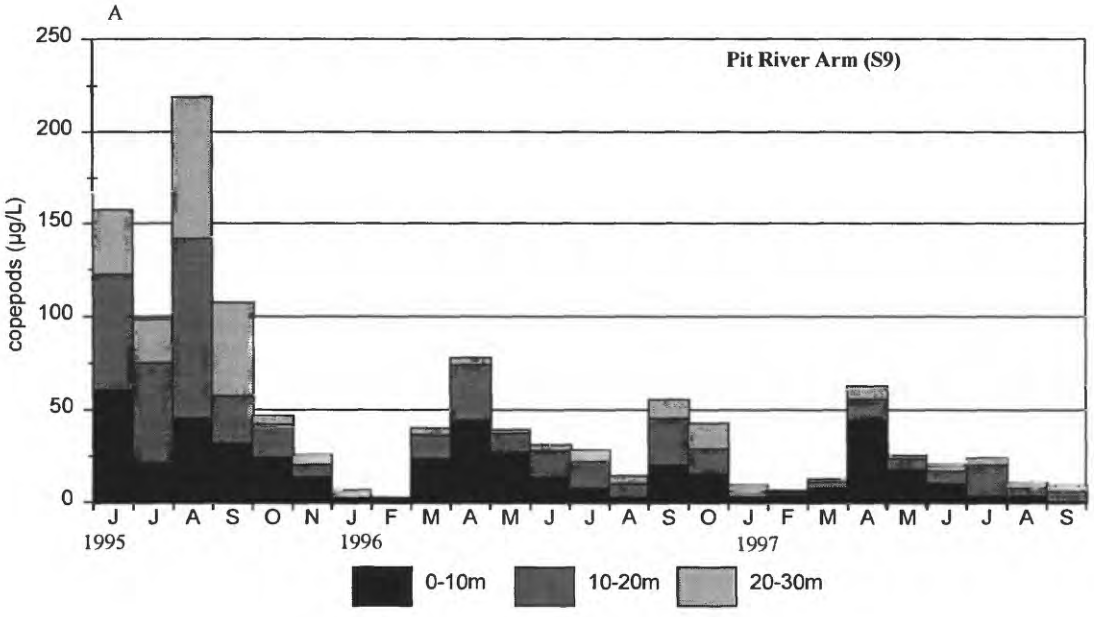


Figure 33 a-b. Seasonal distribution of copepod and cladoceran biomass in 0-10 m, 10-20 m, and 20-30 m depth intervals in the Pit River Arm (S9) from June 1995 thru September 1997. a. copepods; b. cladocerans.



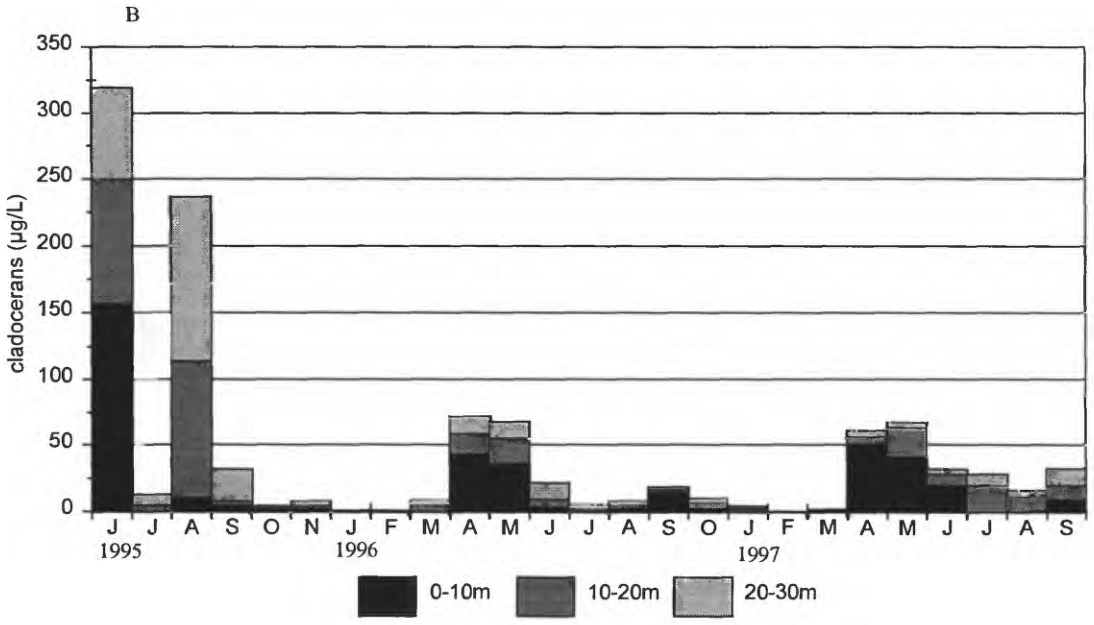
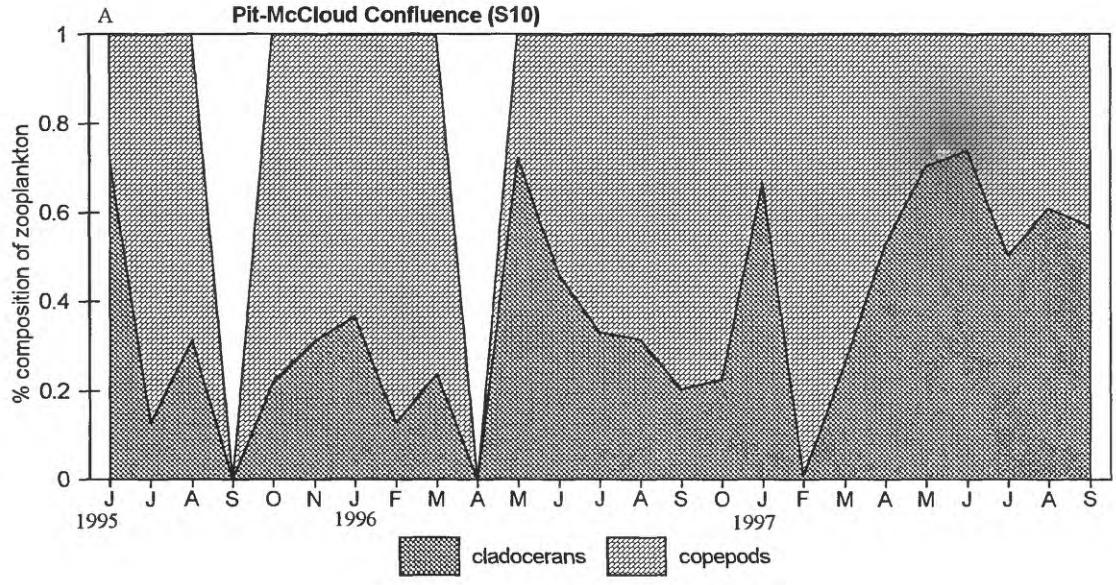


Figure 34 a-b. Trends in zooplankton abundance (0-30 m) for the Pit-McCloud confluence (S10) from June 1995 thru September 1997. a. Percent composition of copepods and cladocerans; b. Cladoceran, copepod, and rotifer total biomass (µg/L) trends.



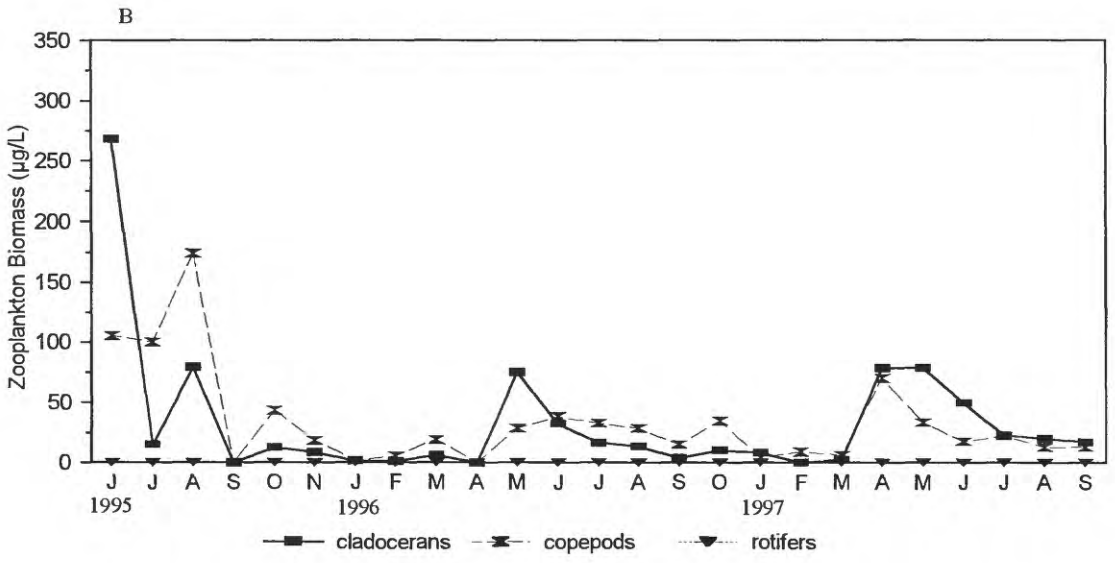


Figure 35 a-b. Seasonal distribution of copepod and cladoceran biomass in 0-10 m, 10-20 m, and 20-30 m depth intervals at the Pit-McCloud confluence (S10) from June 1995 thru September 1997. a. copepods; b. cladocerans.

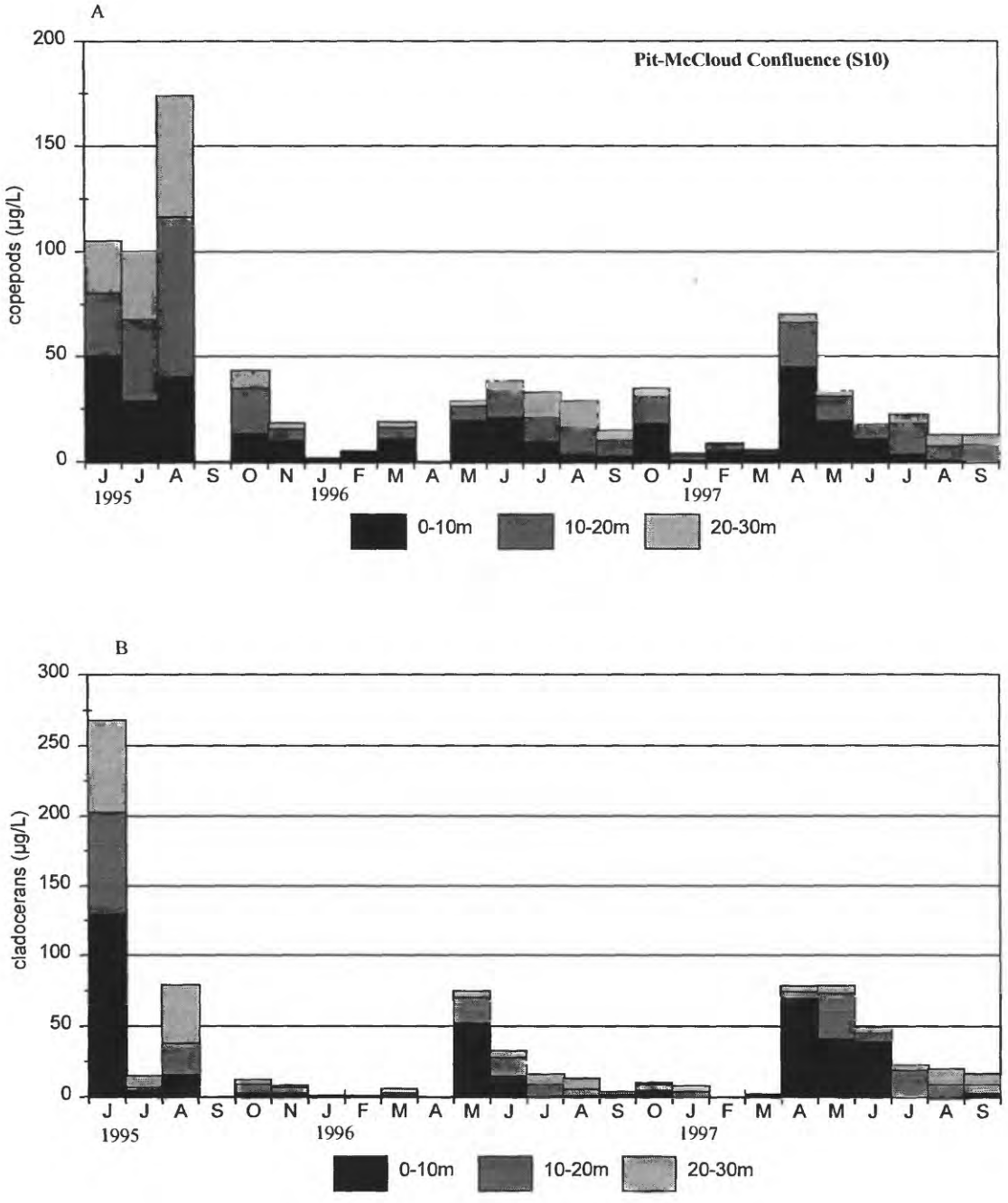


Figure 36. Seasonal trend of rotifer biomass (µg/L) (0-30 m) in the main lake (S6), Sacramento River Arm (S7), McCloud River Arm (S8), Pit River Arm (S9), and the Pit-McCloud Confluence (S10) from June 1995 thru September 1997.

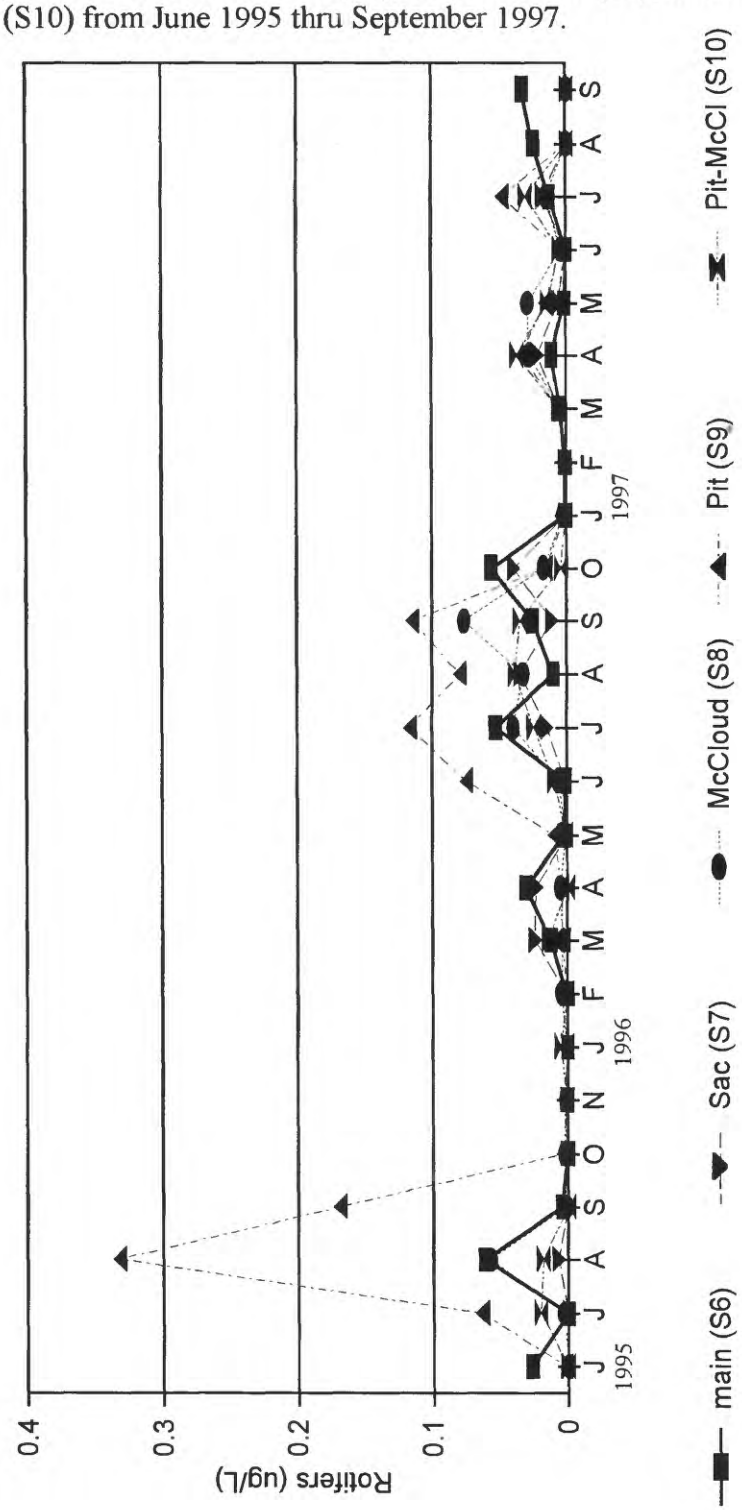


Figure 37. Water temperatures (^{0}C) in Shasta tailwaters and Keswick tailwaters from April 1995 thru November 1997.

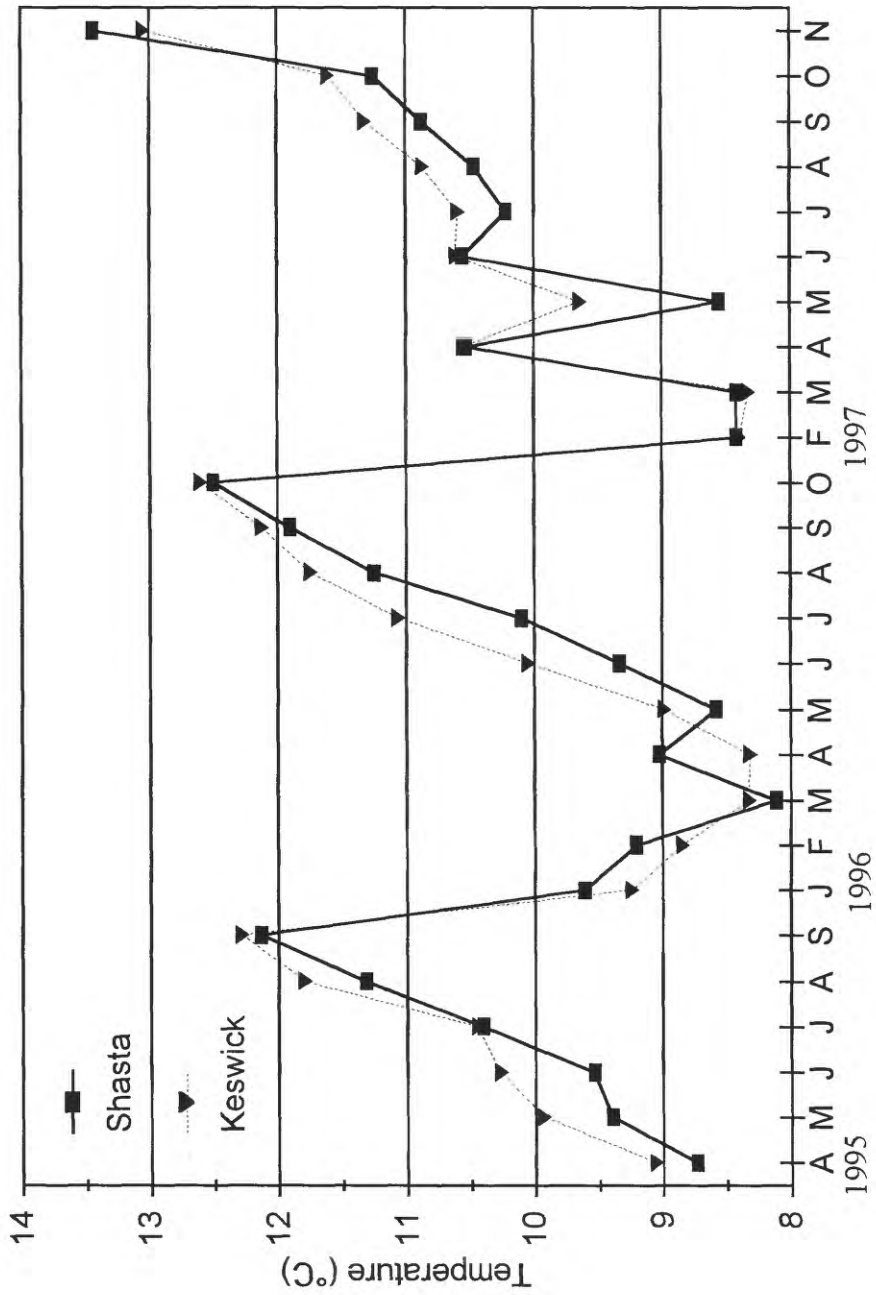


Figure 38 a-b. The relationship between size fractions of <25µm, >25, and >505µm POM concentrations (g/m^3) from April 1995 thru October 1997. a. Shasta tailwaters; b. Keswick tailwaters.

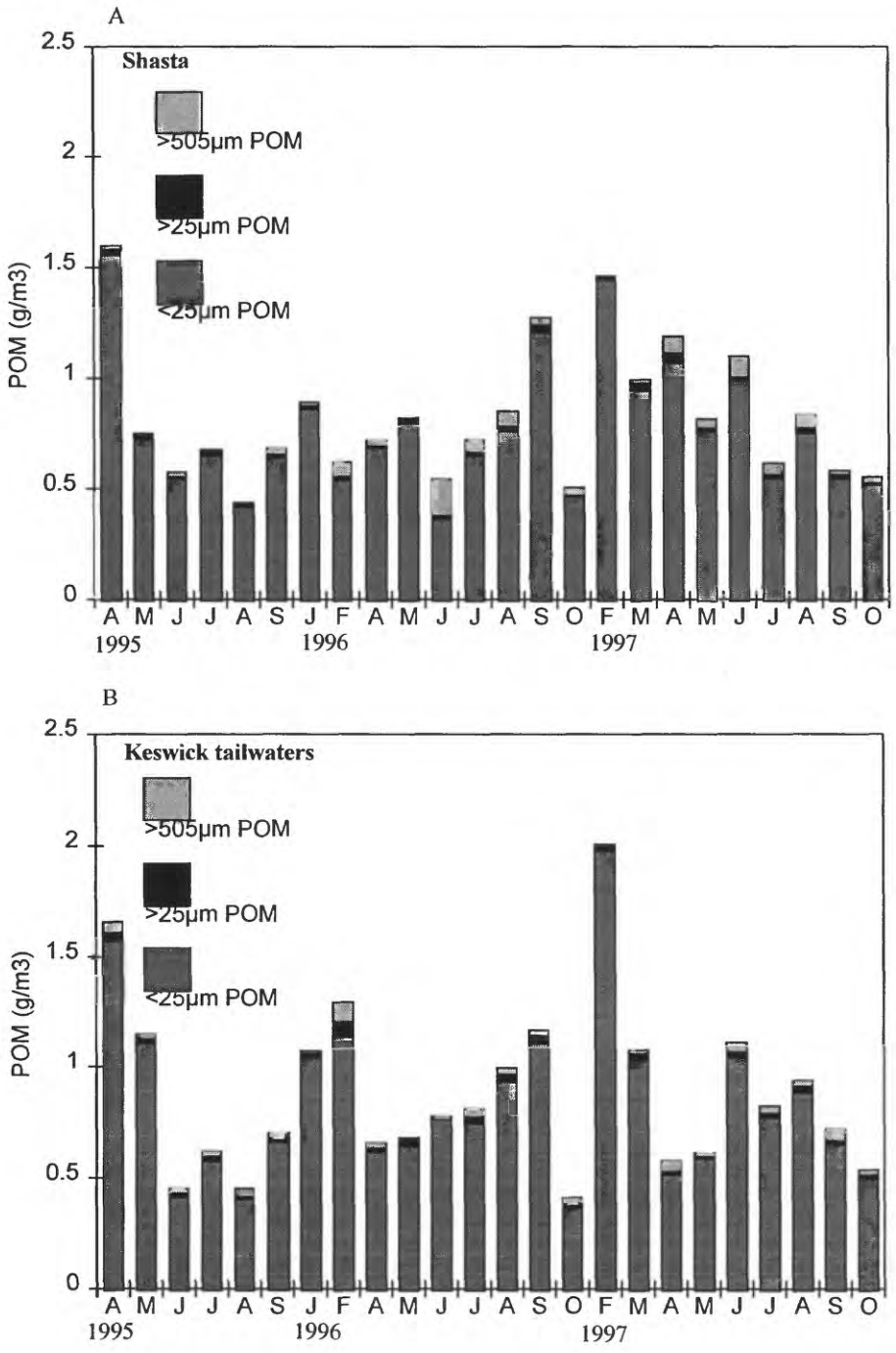
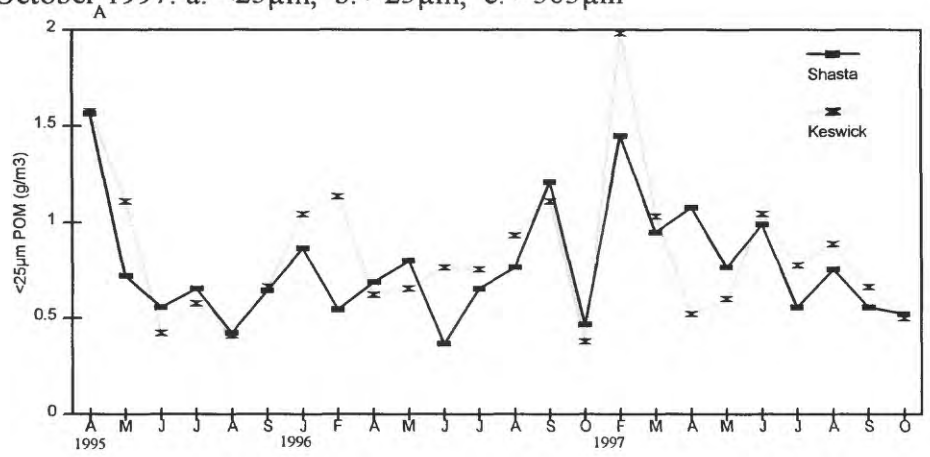
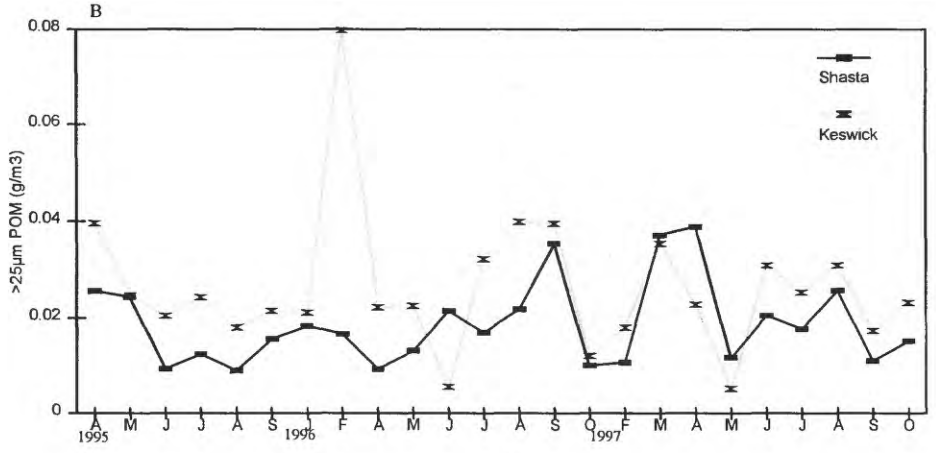


Figure 39 a-c. Seasonal POM (g/m^3) trnds in Shasta and Keswick tailwaters from April 1995 thru October 1997. a. <25µm; b. >25µm; c. >505µm





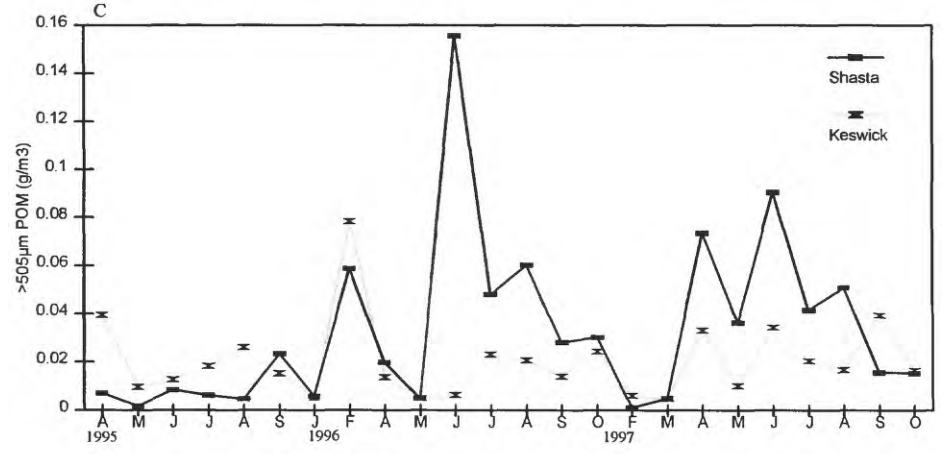


Figure 40 a-c. Composition of total particulate matter (g/m^3) in Shasta tailwaters from April 1995 thru October 1997. a. <25µm; b. >25µm; c. >505µm

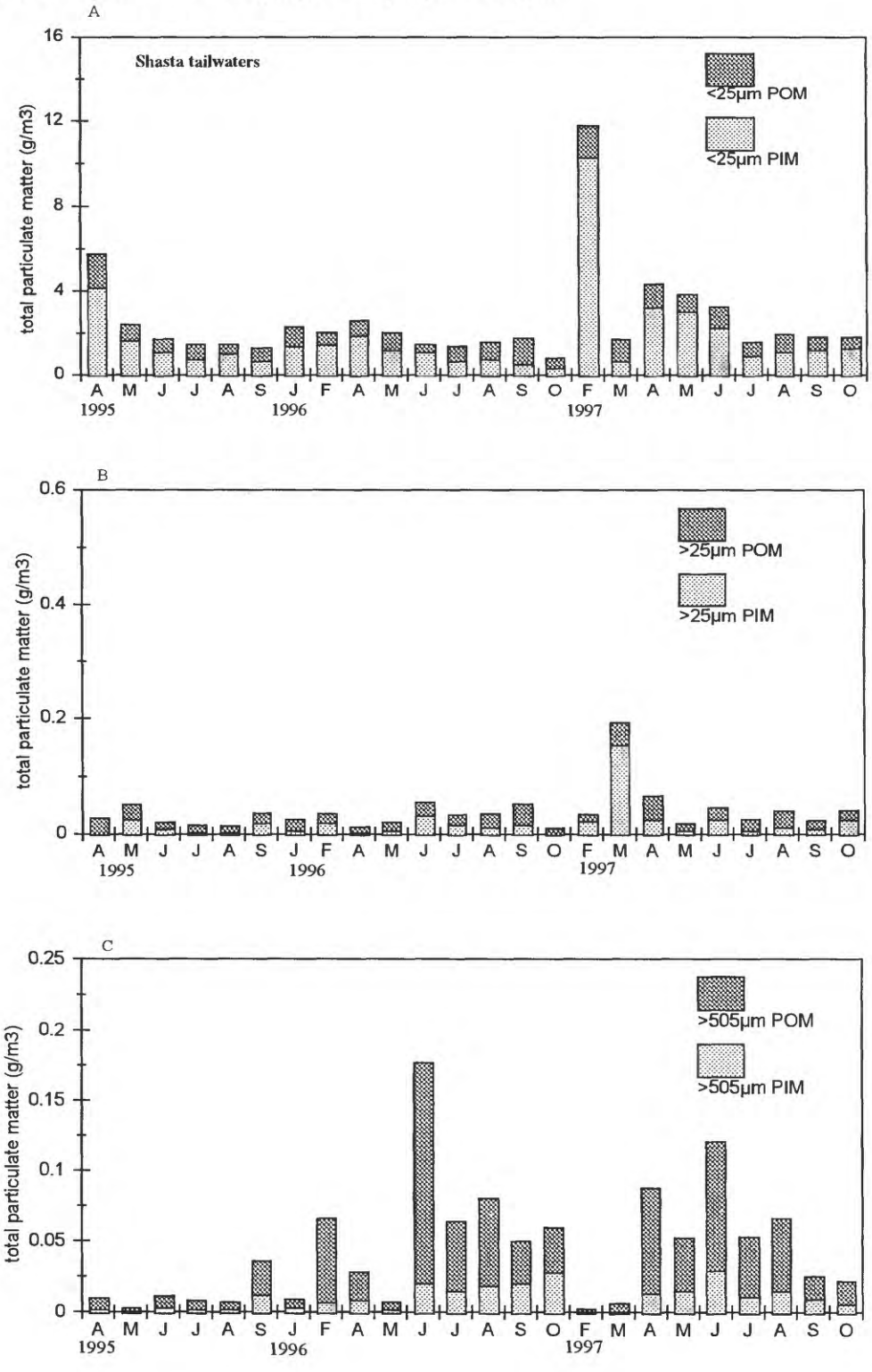


Figure 41 a-c. Composition of total particulate matter (g/m^3) in Keswick tailwaters from April 1995 thru October 1997. a. <25µm; b. >25µm; c. >505µm

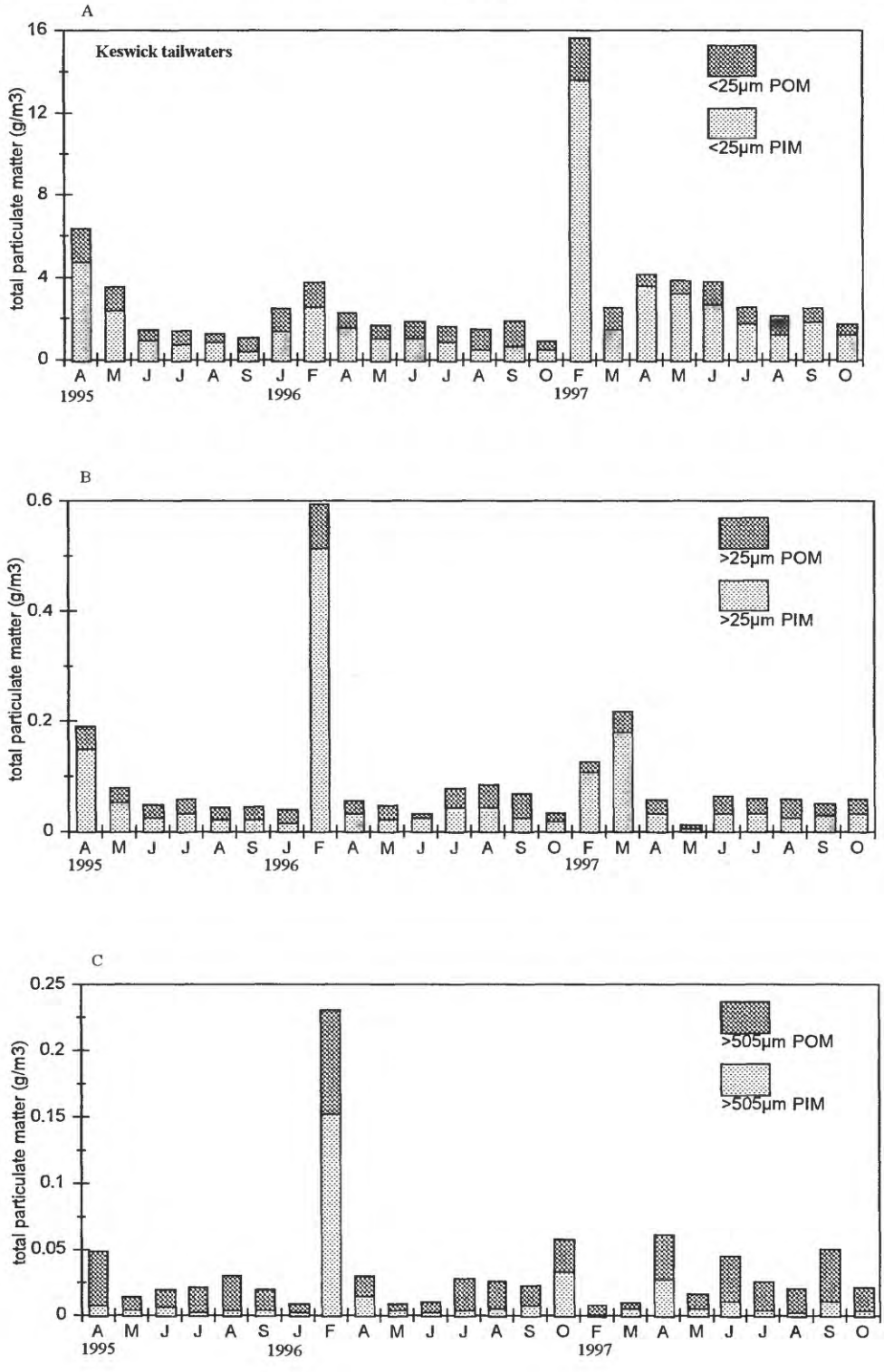


Figure 42. Mean total POM (g/m^3) at Shasta and Keswick tailwaters for the period 1995 thru 1997.

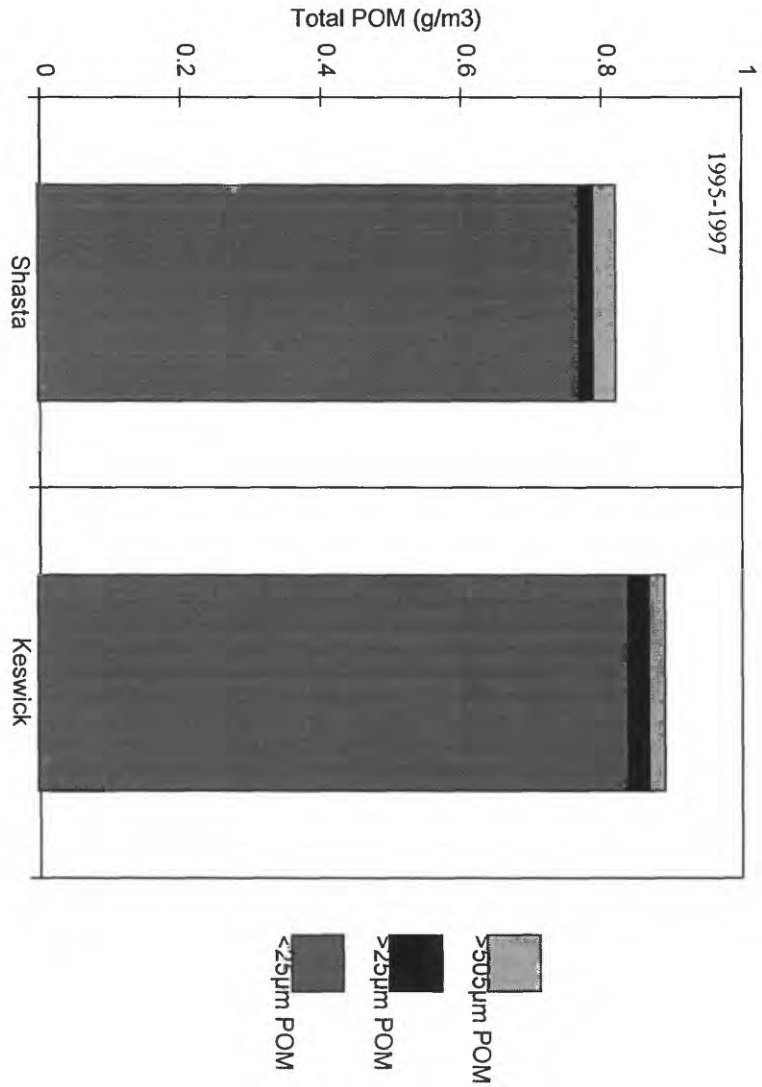


Figure 43 a-b. Seasonal composition of zooplankton for 1995, 1996, and 1997 in Shasta tailwaters. Major groups are cladocerans, copepods, and rotifers. a. biomass (µg/L); b. percent composition

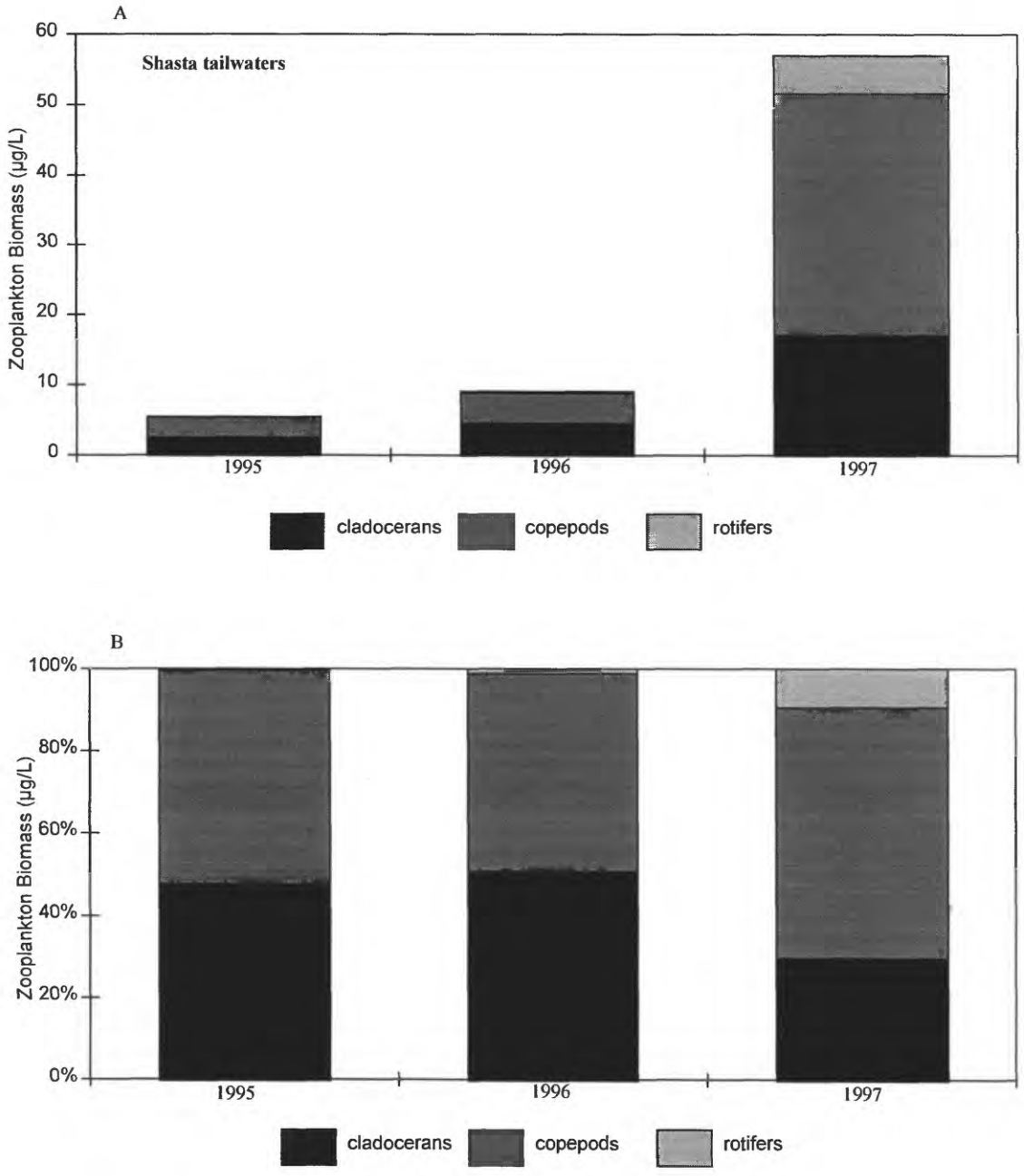


Figure 44 a-b. Seasonal composition of zooplankton for 1995, 1996, and 1997 in Keswick tailwaters. Major groups are cladocerans, copepods, and rotifers. a. biomass (µg/L); b. percent composition

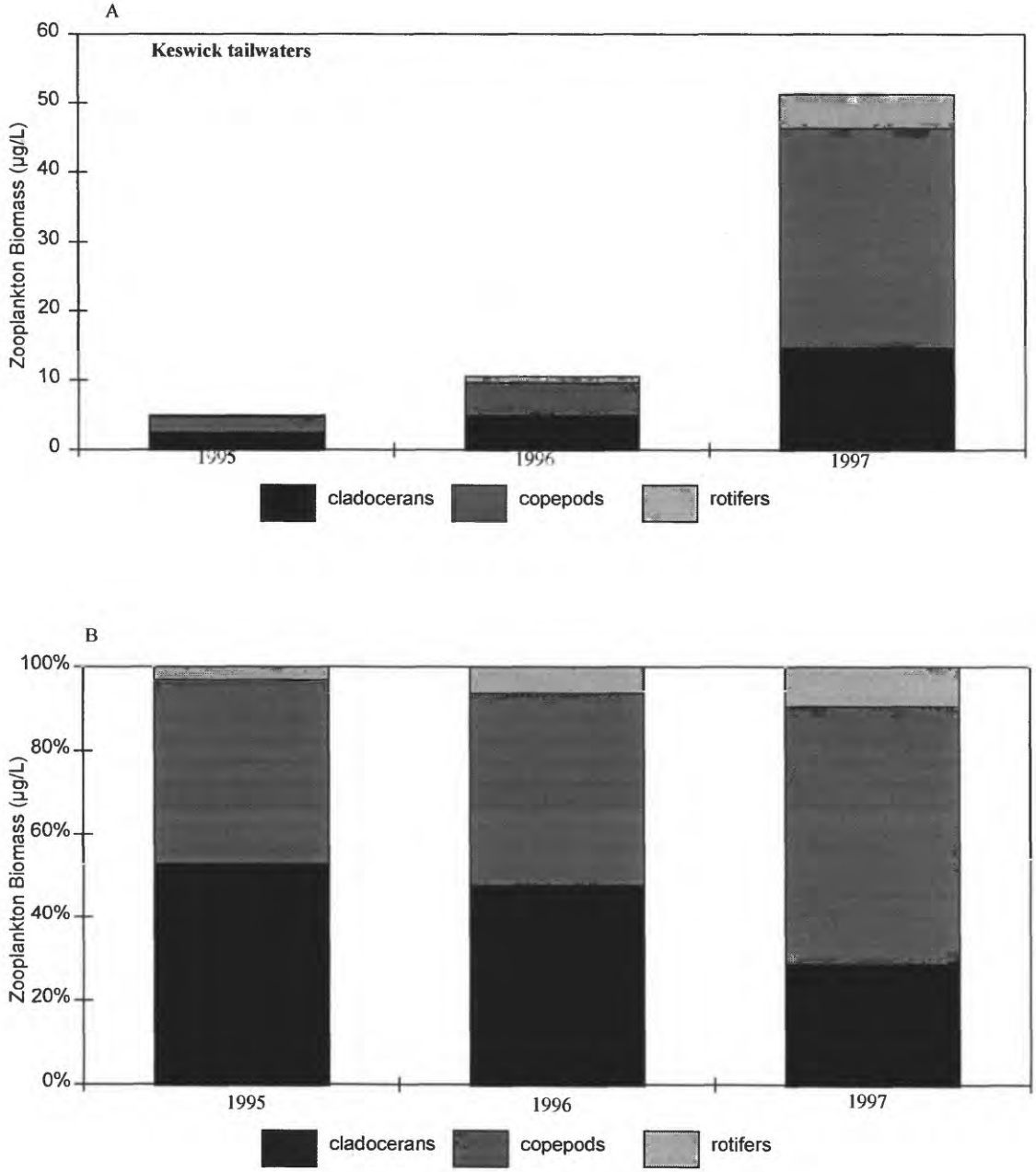


Figure 45 a-b. Cladoceran, copepod, and rotifer biomass (µg/L) drift from April 1995 thru October 1997. a. Shasta tailwaters; b. Keswick tailwaters.

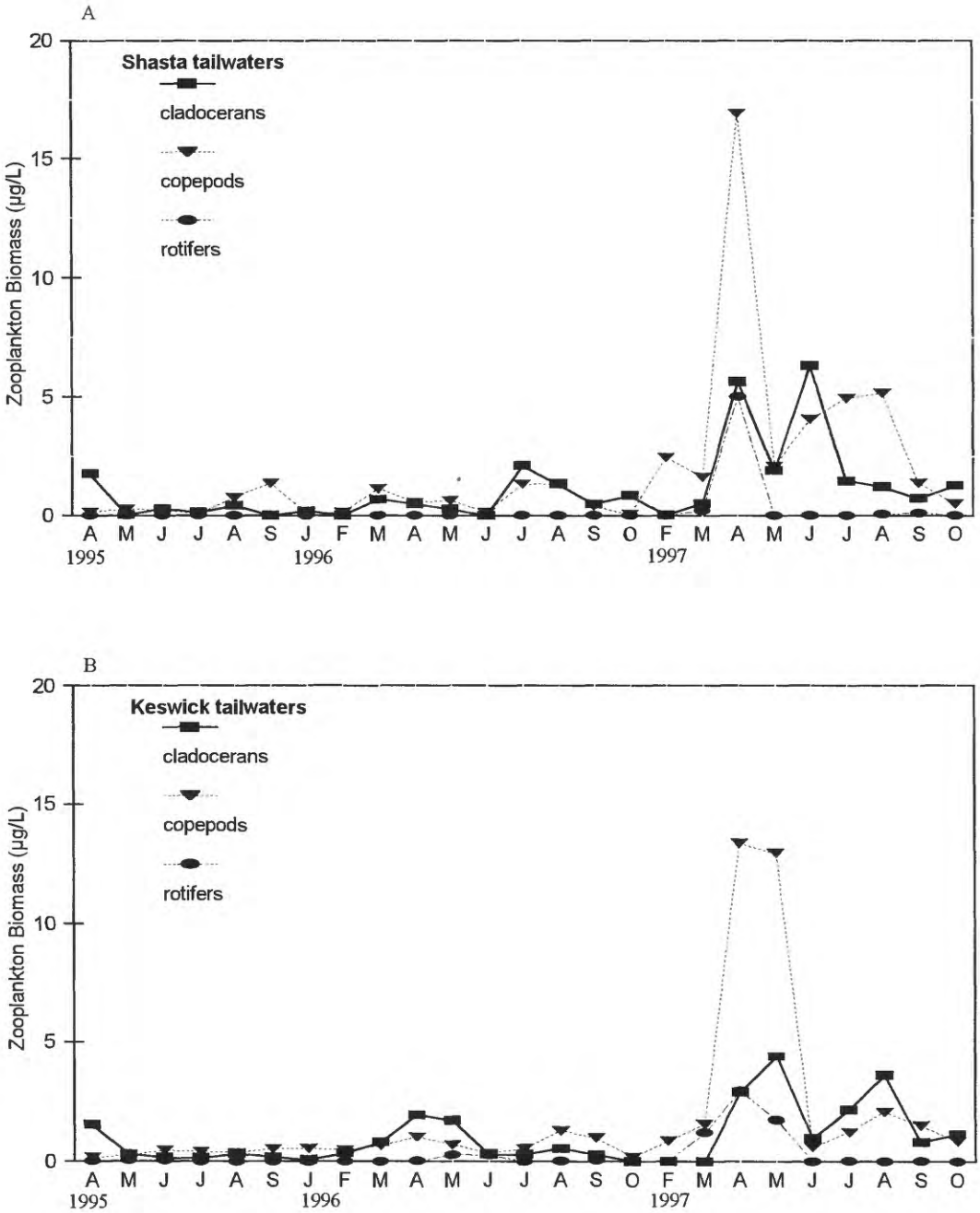


Figure 46 a-b. Total biovolume (µg/L) of phytoplankton size fractions (<25µm, >25µm, >505µm) from April 1995 thru September 1997. a. Shasta tailwaters; b. Keswick tailwaters.

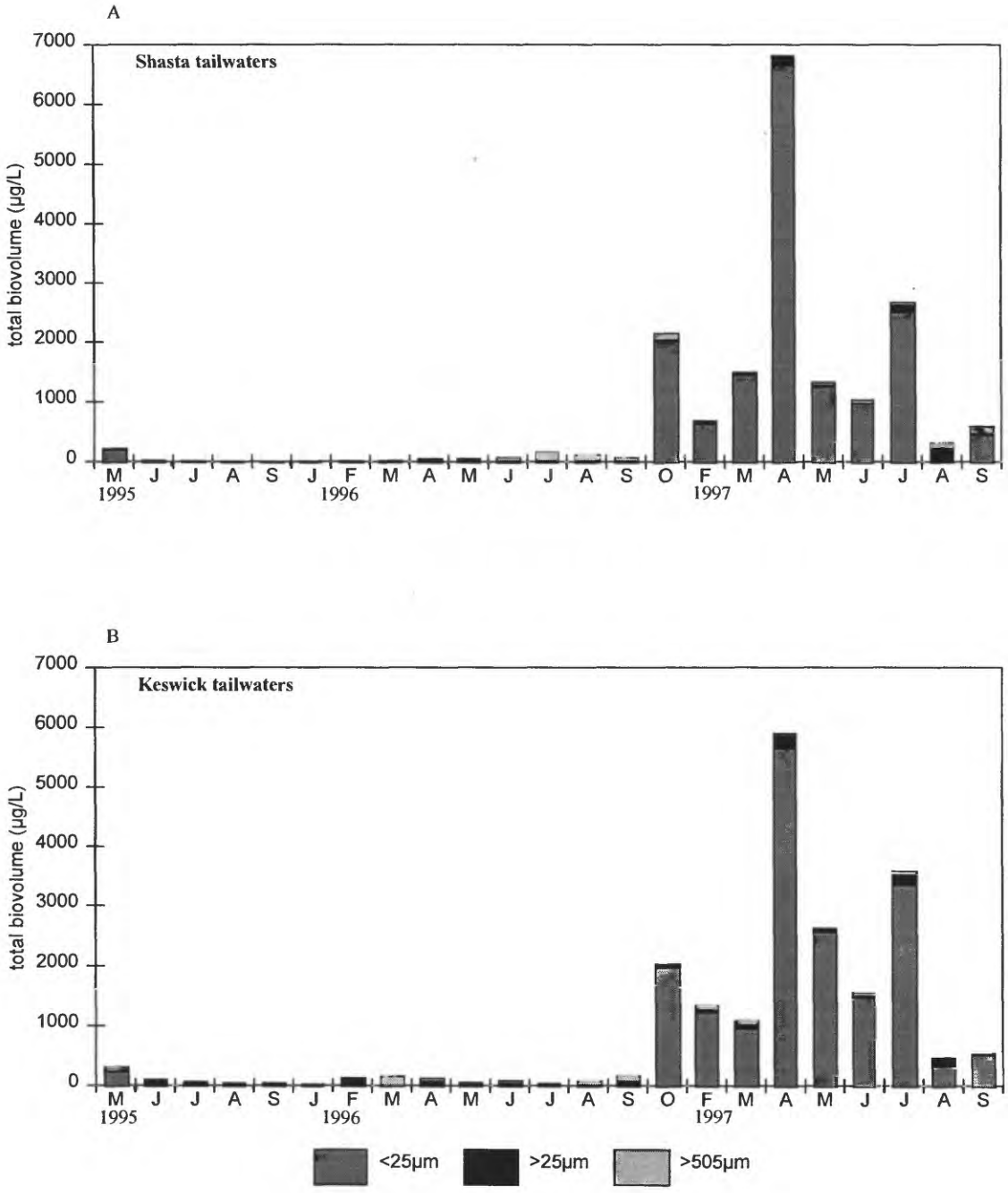


Figure 47. <25µm Chlorophyll *a* concentration (µg/L) in Shasta and Keswick tailwaters from April 1995 thru October 1997.

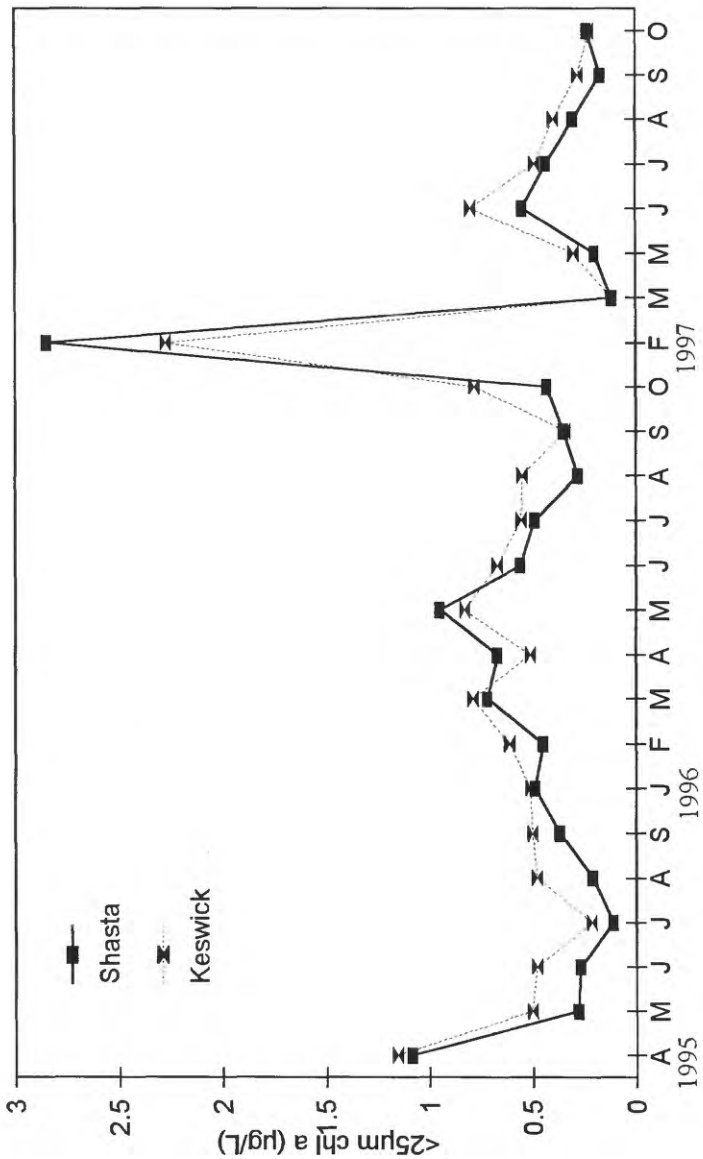


Figure 48 a-c. Percent composition of the <25µm phytoplankton biovolume in Shasta and Keswick tailwaters for the period 1995 thru 1997. a. <25µm; b. >25µm; c. >505µm