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DISSERTATION

INTERDISCIPLINARY MODELING AT SHASTA LAKE

Submitted by

Laurel Saito

Department of Civil Engineering

In partial fulfillment of the requirements

For the Degree of Doctor of Philosophy

Colorado State University

Fort Collins, Colorado

Fall 1999

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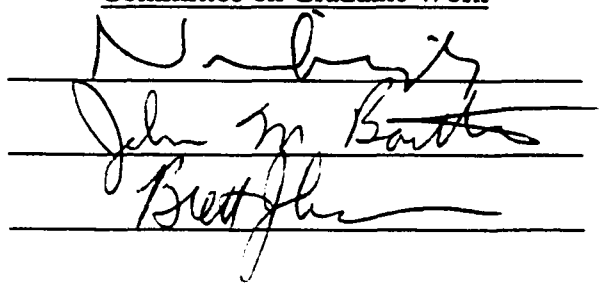
November 2, 1999

WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY LAUREL SAITO ENTITLED INTERDISCIPLINARY MODELING AT SHASTA LAKE BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

Committee on Graduate Work



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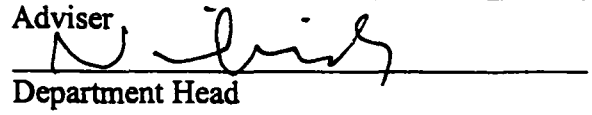

Department Head

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ABSTRACT OF DISSERTATION

INTERDISCIPLINARY MODELING AT SHASTA LAKE

As we approach the millennium, water management is focused on ecosystem management and sustainability. These perspectives recognize the interconnectedness of ecosystems that is inherently interdisciplinary. Modeling using available scientific knowledge and theories can bridge the cultural gap between the ability to scientifically predict effects of management alternatives with certainty, and the need to make management decisions.

A project is presented in which linked engineering and ecological models were employed to assess the effects of changes in reservoir operation on the Shasta Lake fishery. A recently installed temperature control device (TCD) on Shasta Dam enables reservoir managers to enhance the downstream thermal environment of endangered salmon while maximizing hydropower generation. Altered reservoir operations could affect in-reservoir fish growth through changes in thermal, nutrient and primary productivity patterns.

A two-dimensional hydrodynamic model (CE-QUAL-W2) simulated water temperatures, nutrients and phytoplankton with and without the TCD operating under different hydrologic conditions. CE-QUAL-W2 water temperature predictions were input into bioenergetics models to predict fish growth through space and time. A food

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web-energy transfer model developed using stable isotope analysis was applied with CE-QUAL-W2 net algal production predictions to assess changes in fish growth potential.

CE-QUAL-W2 predicted that, regardless of hydrologic condition, epilimnetic thermal changes were insignificant. Therefore, because fish generally reside in the epilimnion, the linked hydrodynamic and bioenergetics simulations concluded that fish would not experience substantial changes in scope for growth due to thermal effects of TCD operation. CE-QUAL-W2 predicted increased reservoir net algal production during stratification with TCD operation under hydrologically wet conditions, and the opposite during dry years. The estimated food web indicated that rainbow trout were not dependent on phytoplankton production, and therefore would not be affected by changes in net algal production. Bass were most connected to the phytoplankton food web pathways, and could experience more growth in wet years, and less growth in dry years with TCD operation.

Active involvement of engineers and biologists in developing and applying the linked modeling approach at Shasta Lake provided ecosystem-level insight into reservoir management impacts. Further interdisciplinary efforts are needed to effectively address ecosystem management and sustainability of water resources.

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seeking more from myself and gently guiding me along the paths that have been so meaningful to me is beyond the scope of any words.

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Chapter 1. INTRODUCTION/PROBLEM STATEMENT

The human experiment of the last ten thousand years has been to modify and simplify ecosystems

Smith (1996)

Since the beginning of civilization, humans have harnessed fresh water resources for food production, for transportation, and for energy

Gleick (1993)

For six millennia, the human race has been involved in a battle to control water resources

Strong et al. (1993)

If the creator were a corporate manager, he or she might well pose the question: 'has human management added value to the earth?' From a purely human perspective, the reply would likely be an emphatic yes. After all, there are now 5.5 billion of us. But any corporate manager knows that, when inventories are depleted and the physical plant is allowed to deteriorate, it is possible to make money in the short term while watching your

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net worth waste away...businesses routinely make decisions with short-term costs, but obvious benefits to their long-term sustainability.

Christensen et al. (1996)

Water is the source of all life

The Koran, as cited in Clarke (1993)

en'gi·**neer'**ing: *the putting of scientific knowledge in various branches to practical uses*

Webster's New World Dictionary of the American Language (1984)

When one thinks of water resources engineering, statements such as these might come to mind, for engineers have indeed historically been involved in altering the natural environment to try to mitigate the unpredictability of this valuable resource. The role of the engineer in water resources planning and development is described by the American Society of Civil Engineers (ASCE) as carrying 'out the wishes of the general public which prevailed at the time' (ASCE 1984). To illustrate, ASCE (1984) provides a history of engineering involvement in water resources development since the establishment of the United States until the early 1980's. Initially, water resources development was driven by national desires to facilitate the joining of newly settled regions of the vast country, and to provide national security. Thus, engineers were involved in developing hydropower, providing irrigation, and building infrastructure to protect and transport people and goods. Engineers became involved in developing water treatment plants and wastewater systems in the latter part of the 19th century as population growth and the

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development of large urban areas led to concerns about health hazards due to water pollution (ASCE 1984; Chapra 1997). In the first half of the 20th century, economic goals became the driving force behind national policy, and as a result, this was a period of intense development of water resources projects that were built predominantly for economic objectives. Beginning in the 1960's, concerns about environmental protection led to legislation such as the Endangered Species Act and the National Environmental Policy Act. The enactment of these laws caused water resources management to begin addressing multi-objective goals such as satisfying economic development and environmental protection. Through all of these changes, the role of the engineer remained the same, but the focus of issues shifted from merely being concerned with hydrologic systems, to addressing water quality, and social and environmental issues as well (ASCE 1984).

As we approach the millennium, the focus has shifted to issues such as ecosystem management and 'sustainability.' After all these years of manipulating the world's water resources and looking at relatively short-term needs, we have become more aware of the long-term effects of our actions, and how little we really know in the grand scheme of science and the universe. These revelations have led many to a greater concern about the future health of our planet. Thus, engineers are moving from an old paradigm of optimizing technological performance to being 'good stewards of the Earth and its resources' (Wiedenhoeft 1999).

Ecosystem management, a concept originating out of the biological sciences (Morrissey 1996), embodies the 'integration of ecological, economic, and social factors in order to maintain and enhance the quality of the environment to meet current and

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future needs' (Thomas 1996). Similarly, ASCE defines sustainability for water resources management as including the concepts of (ASCE 1998):

- 'Nature: the view that river-aquifer-estuarine systems and their environment and ecosystems have value in their own right and hence efforts to protect these resources, habitats and biodiversity are warranted.
- Current generation: the recognition that those who live today have demands they expect to be met from water resource systems...
- Future generations: the view that our descendants have the right to at least the same if not a better economy, environment, and quality of life that we enjoy today.'

Both of the concepts of ecosystem management and sustainability arise out of the current competing societal desires to maintain a healthy environment and enjoy strong economic growth (Morrissey 1996). These concepts also share an awareness of the interconnectedness of ecosystems. Willis (1997) attributes the first use of the word 'ecosystem' to A.G. Tansley, who attempted to describe the physical and biological components of an environment that are interrelated. Thus, it is natural that the concept of ecosystem management is applied to an increased awareness of the interconnectedness of ecological systems. This interconnectedness is not only interdisciplinary, but also shared among agencies and organizations, and ecological and social variables (Yaffee 1996).

Historically we have tended to look at specific concerns and management alternatives that were driven largely by particular disciplines. For example, engineers have traditionally applied technology to monitor and regulate flows, while fisheries biologists have studied fish populations, entomologists have studied aquatic insects, limnologists have analyzed lake chemistry and biology, geomorphologists have looked at how flows alter and shape landscapes, recreation managers have tried to provide

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environments that satisfy recreationalists such as boaters and anglers, economists have looked at the costs and benefits of the flow of water resources and corresponding services, and sociologists have looked at how decisions are made and affect people.

Today it seems obvious that addressing specific water resources issues in any one of these disciplines invariably involves effects that concern other disciplines. Attempts to address one issue often have consequences that exacerbate existing issues or concerns, or create new ones (Holliman 1974; Jørgensen et al. 1992; Lackey et al. 1975; Straskraba 1994). For example, in reservoir management, peak hydropower demands can be met by altering discharge quantities several times over the course of a day, yet these fluctuations can affect downstream channel morphology and create habitat limitations for downstream species that become a concern for biologists. In-reservoir water levels can also fluctuate greatly with this type of operation and have effects on recreational uses and in-reservoir habitat (Cantor 1985; Flug 1997; Stanford et al. 1996; Takeuchi et al. 1998).

While the need for interdisciplinary research and management seems clear, it is still painfully difficult to implement. Why is this? One reason is the different perceptions each discipline has of the other, which Cullen (1990) analogizes as 'cultural' differences, and parodies as:

> '*Engineers* don't care why it works as long as they think it does.
> *Scientists* don't care if it works or not as long as they understand why.
> *Economists* don't care either way if the internal rate of return is OK.
> *Managers* don't know unless someone bothers to tell them.
> *Planners* know how it should have turned out.'

Part of the cultural difference arises out of the different spatial and temporal scales at which management decisions must be made and the scales at which ecosystem processes occur (Christensen et al. 1996). For example, Carpenter (1989) concluded that

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investigations of ecosystem responses to manipulations of lake food webs needed to include about 10 replicate lakes, with manipulations that increased piscivore biomass at least 5-10 times for several years. Minns et al. (1996) illustrated the different spatial and temporal scales of events that affect fish populations (Figure 1), but the time and space scales for observing the effects are often even greater. Scientists require this large diversity of temporal and spatial scales because of the prevalent uncertainty in ecological science. The uncertainty arises out of the complexity and dynamic nature of ecosystems (i.e., they are always changing), lack of understanding of the processes that occur, and variability and quality of data used to understand the systems (Christensen et al. 1996; Hilborn and Ludwig 1993).

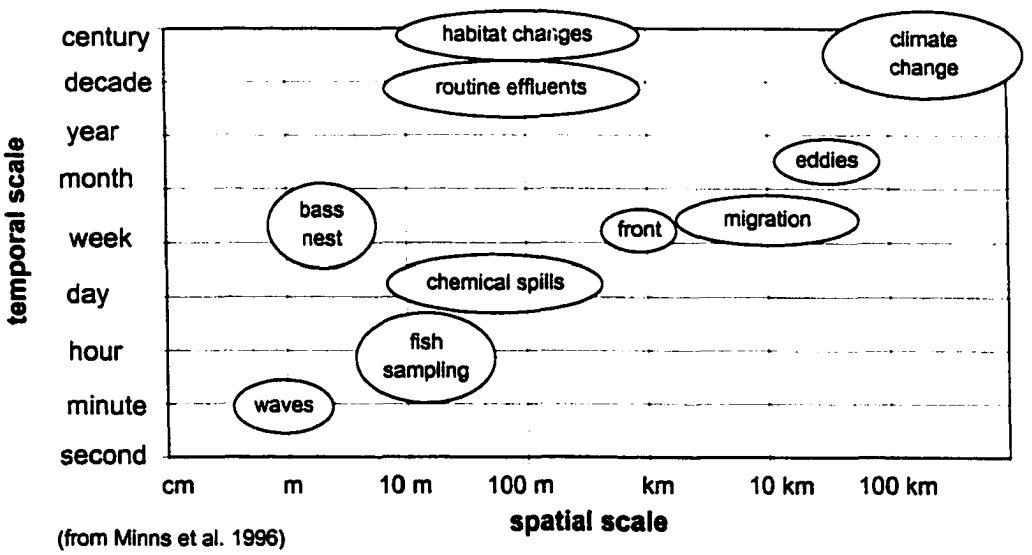

Figure 1. Illustration of spatial and temporal scale of events and activities that affect fish populations

However, management decisions are often made along political or jurisdictional boundaries that do not coincide with the spatial scale of ecosystems, and the time frames in which the decisions are made do not accommodate the temporal scale scientists need to

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make quantifiable predictions. Policy-makers want to make clear, defensible decisions, and they expect scientists to provide the knowledge needed to make such decisions (Costanza 1993). Managers often have a misperception that scientists will tell them 'the truth,' and will all agree about what the truth is. In reality, science is constantly *seeking* truth by challenging and testing hypotheses or theories (Müller 1997), and the different scientific disciplines rarely agree on the best management alternative because of their different perspectives and training (Cullen 1990).

Where do engineers fit into this picture? Webster's dictionary (1984) defines engineering as 'the putting of scientific knowledge in various branches to practical uses.' Thus, engineers are a link between scientists and managers. As such, engineers must be systems thinkers, looking at interdisciplinary approaches, not just multidisciplinary ones (Müller 1997).

One of the ways in which engineers have approached water resources management is through mathematical modeling. In this era of addressing ecosystem management and sustainability issues, Thomann (1998) notes that the application of models can be very useful. Models based on available scientific knowledge and theories can be used to bridge the cultural gap between the ability to scientifically predict with reasonable certainty, and the need to make decisions. If models are recognized as valid, they can increase understanding, facilitate communication, and assist in decision-making (Thomann 1998).

The challenge for modelers is to come up with models that are valid to all disciplines that the models are purported to represent. This is the issue tackled by this dissertation work. It is the premise of this dissertation that rather than attack modeling

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from a multidisciplinary perspective (i.e., use an engineering model to predict water temperatures and then ask a fisheries expert for an opinion on the resulting effects on the reservoir fishery), it is more productive to pursue interdisciplinary modeling, in which the expertise from each discipline is used concurrently in model development, application, and interpretation. In the case study project presented in this dissertation, linked engineering and ecological models were employed to assess the effects of changes in reservoir operation on the reservoir fishery. This interdisciplinary modeling is a promising step towards the needed interdisciplinary dialogue for ecosystem management and sustainability of our water resources.

The remainder of this dissertation will begin with a description of the case-study interdisciplinary modeling project at Shasta Lake in Chapter 2. This section ends with a summary of the modeling objectives. Chapter 3 presents a literature review of modeling approaches that have been used to address the modeling objectives summarized in Chapter 2. The literature review concludes with a synthesis of the methods selected for application in the Shasta Lake project.

The next four chapters present details of the modeling methods used at Shasta Lake. Chapter 4 describes the overall framework of the linked modeling approach that was chosen for the project. Chapter 5 describes the development, calibration and application of CE-QUAL-W2, the hydrodynamic and water quality model used to simulate water temperature, nutrients and phytoplankton. The development and application of the bioenergetics model to calculate scope for growth for rainbow trout and smallmouth bass is presented in Chapter 6. Chapter 7 details the development and

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application of the food web-energy transfer model, including the sampling and analysis of stable isotopes.

Chapter 8 presents and discusses the model results from all three models. A discussion of the overall modeling approach is provided in Chapter 9, and conclusions are made in Chapter 10.

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Chapter 2. PROJECT DESCRIPTION

The case-study project presented in this dissertation is an interdisciplinary modeling effort at Shasta Lake (Figure 2) in northern California. Shasta Lake and Dam are located about 19 kilometers (km) (12 miles) north of the town of Redding (Figure 3). The reservoir is the largest storage reservoir in California, with a maximum capacity of 5.615×10^9 m^3 (4,522,090 acre-feet), a surface area of 11,940 hectares (ha) (29,500 acres), a maximum depth of 157.6 m (517 feet), a length of 56 km (35 miles), and a shoreline of 587 km (365 miles). Shasta Dam, a curved concrete gravity dam with a height of 183 m (602 feet) and a crest length of 1054 m (3460 feet), is the second largest dam in mass in the United States. The dam is situated on the Sacramento River, and also impounds water from the Pit and McCloud Rivers (USBR 1994).

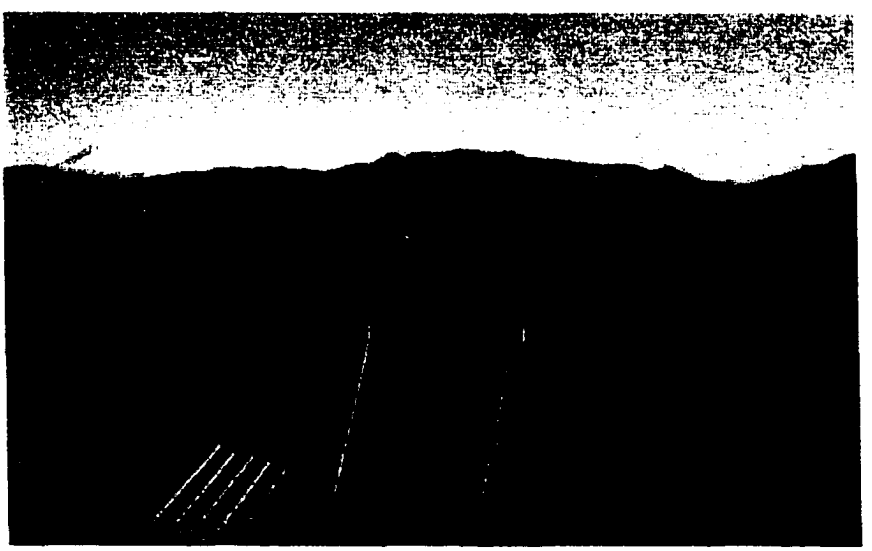

Figure 2. Shasta Dam and Lake, with Mount Shasta in the background

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Figure 3. Location of Shasta Dam and Lake

Shasta Dam was built and is operated by the U.S. Bureau of Reclamation (USBR). Its multipurpose operations include controlling floodwater, storing winter runoff for irrigation use, providing maintenance of navigation flows, providing water supply for municipal and industrial use, improving fisheries and wildlife, protecting the Sacramento-San Joaquin Delta from the intrusion of saline ocean water, and generating hydroelectric power. Located just below Shasta Dam, the Shasta power plant has an installed capacity of 583,000 kilowatts. Five 4.6-m (15-foot) penstocks carry water from the dam to the five main generating units in the power plant (USBR 1983; USBR 1994).

About 14.5 km (9 miles) downstream of Shasta Dam is another 48 m (157 feet) high concrete gravity dam called Keswick Dam. The dam creates Keswick Reservoir, which serves as an afterbay for hydropower releases from Shasta Dam and the Spring Creek Powerplant, located on Spring Creek just upstream of Keswick Dam. Keswick

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Dam also has a power plant that has three generating units with a total capacity of 75,000 kilowatts (USBR 1983; USBR 1994).

Recreational activities are very popular at Shasta Lake. The U.S. Forest Service (USFS) manages general recreational activities in and around the reservoir, while the California Department of Fish and Game (CDFG) is responsible for fishery management, including fish stocking, fishing regulations and enforcement, and related activities (CDFG 1991).

Shasta Lake also supports both cold water and warm water fisheries. According to CDFG creel census data, the most abundant warm water game fish are Alabama spotted bass, smallmouth bass, largemouth bass, channel catfish, white catfish, brown bullhead, black crappie, green sunfish, and bluegill. Stocked rainbow trout dominate the cold water fishery (CDFG no date; CDFG 1991).

Trout are stocked annually in the spring (usually April and/or May) and fall (usually October and/or November). Approximately 250,000 hatchery-reared yearling trout consisting of several species are stocked each year. Approximately 50,000 yearling chinook salmon are also stocked each November. CDFG does not do any annual stocking of warm water fish, but CDFG released about 5,000 largemouth bass in 1990, and 2,500 white sturgeon in 1989 (CDFG no date).

Many of the fish present at Shasta Lake have been introduced, including kokanee salmon, Alabama spotted bass, largemouth bass, smallmouth bass, threadfin shad, black crappie, and channel catfish (CDFG 1991). Table 1 summarizes the relative abundance of fish species present in Shasta Lake as of March 1991.

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Table 1. Relative abundance of fish species present in Shasta Lake

Common name	Scientific name	Relative abundance
Pacific brook lamprey	*Lampetra pacifica*	Unknown
white sturgeon	*Acipenser transmontanus*	Rare
threadfin shad	*Dorosoma petenense*	High
chinook salmon	*Oncorhynchus tshawytscha*	Rare
kokanee salmon	*Oncorhynchus nerka*	Rare
rainbow trout	*Oncorhynchus mykiss*	High
brown trout	*Salmo trutta*	Low
tui chub	*Gila bicolor*	Low
Sacramento blackfish	*Orthodon microlepidotus*	Low
golden shiner	*Notemigonus crysoleucas*	Rare
hardhead	*Mylopharodon conocephalus*	Low
Sacramento squawfish	*Ptychocheilus grandis*	Medium
Carp	*Cyprinus carpio*	Medium
black bullhead	*Ictalurus melas*	Unknown
brown bullhead	*Ictalurus nebulosus*	Rare
white catfish	*Ictalurus catus*	Low
channel catfish	*Ictalurus punctatus*	Medium
green sunfish	*Lepomis cyanellus*	Medium
bluegill	*Lepomis macrochirus*	Low
black crappie	*Pomoxis nigromaculatus*	Low
white crappie	*Pomoxis annularis*	Unknown
largemouth bass	*Micropterus salmoides*	Low
smallmouth bass	*Micropterus dolomieui*	High
Alabama spotted bass	*Micropterus punctulatus henshalli*	High
tule perch	*Hysterocarpus traski*	Rare
sculpin spp.	*Cottus spp.*	Rare

Source: CDFG (1991)

The central issue behind the modeling effort at Shasta Lake is the propagation of the endangered winter-run chinook salmon fishery downstream of Shasta Dam. Also known as King salmon, the chinook salmon has been the most abundant salmon species in California (McGinnis 1984). Historically, salmon propagation on the Sacramento River and its tributaries was known to be good from the mid-1800s. The first salmon hatchery in the United States was constructed in 1872 on the McCloud River several km upstream of its confluence with the Pit River in a one-room cabin with 24 hatching troughs (Stone 1874, as cited in Heizer 1973; Stone 1896 as cited in Cone and Ridlington

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1996). Stone (1874, as cited in Heizer 1973) gave the following description of the salmon populations in the McCloud River in 1872:

> 'It will be seen by the previous notes that there are salmon in the Lower Sacramento every month in the year. It is not so in the upper tributaries of the river...The salmon have stated times for arriving in the upper tributaries and for remaining in them, and at other periods of the year there are no salmon in these streams.
>
> The salmon arrive in the mouth of the McCloud in March, but are scarce in that month. In April and May they become plentiful but are not large, the average weight not exceeding ten or twelve pounds. They remain plentiful through June and July, during the latter part of which months they receive an accession from Pit river...In August, there is a large run of salmon up the McCloud, composed of larger fish. The salmon are now, in August, the largest and most abundant of any time of the year in the McCloud. They begin to spawn in the lower portions of the McCloud during the last half of August. By the middle of September the salmon begin here to die, and from this till the end of the month they die very rapidly...About this time—the latter half of September—a new run of salmon makes its appearance in the McCloud, called the 'fall run'...During October there are no salmon in the McCloud, except the few newcomers of the 'fall run,' and by the first of November all the salmon are gone form the river except one or two individual stragglers here and there...From November till March there are no salmon in the McCloud River.'

In the early 1900's, the salmon fishery on the Sacramento River was very important to the California commercial salmon fishery. However, by 1927, the commercial salmon catch had declined considerably (Clark 1929). Although Clark (1929) warned that the 35 dams already constructed on the Sacramento River as of 1928 were directly or indirectly affecting salmon migration, Shasta Dam was still completed in 1945. The dam prevented the chinook salmon from migrating to their native spawning habitat upstream of the dam, and exacerbated the decline in chinook salmon populations. In 1989, the winter-run chinook salmon were listed as a protected species under provisions of the Federal and California State Endangered Species Acts. The impending listing led the USBR to change its reservoir operations in 1987 to release colder water in

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the late summer and fall through low level outlets. Correspondingly, the USBR released warmer water in the spring and early summer through high level outlets in order to preserve the cool water pool (Vermeyen 1998a). The control of release temperatures was done to enable the winter-run chinook salmon to survive in the anthropogenically created habitat downstream of the dam (Healey 1994).

In 1987, the dam consisted of three outlet options: over the spillway, through the penstocks that release water through the power generating facilities, and through release outlets at three levels. The penstocks were only capable of withdrawing water from one elevation. Thus, any releases through the low- or high-level outlets bypassed the penstocks (Figure 4(a)). In order to maintain adequate downstream temperatures for the cold-water chinook salmon fishery, releases were often required through these outlets, reducing the amount of hydropower that could be generated. It was estimated that from 1987 through 1996 inclusive, approximately $63 million in power generation was lost through the bypass outlets (Vermeyen 1998a).

In order to improve the USBR's ability to control downstream water temperatures while maximizing hydropower generation, an $80 million temperature control device (TCD) was installed at Shasta Dam. The TCD began operating in February 1997 and now allows the passage of water from various elevations through the penstocks (Figure 4(b)). Operation of the TCD was expected to permit increased hydropower generation while maintaining the downstream temperature requirements (Vermeyen 1998a).

The TCD includes a 76.2 m (250 feet) wide by 91.4 m (300 feet) high shutter structure that has a separate unit for each of the five penstocks to the power plant, as shown in Figure 5. Each unit has adjustable shutters that allow shallow, deep and/or

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intermediate withdrawals from three inlets through the penstocks. There is also a low level intake structure that allows withdrawals from depths deeper than the previous outlet structure.

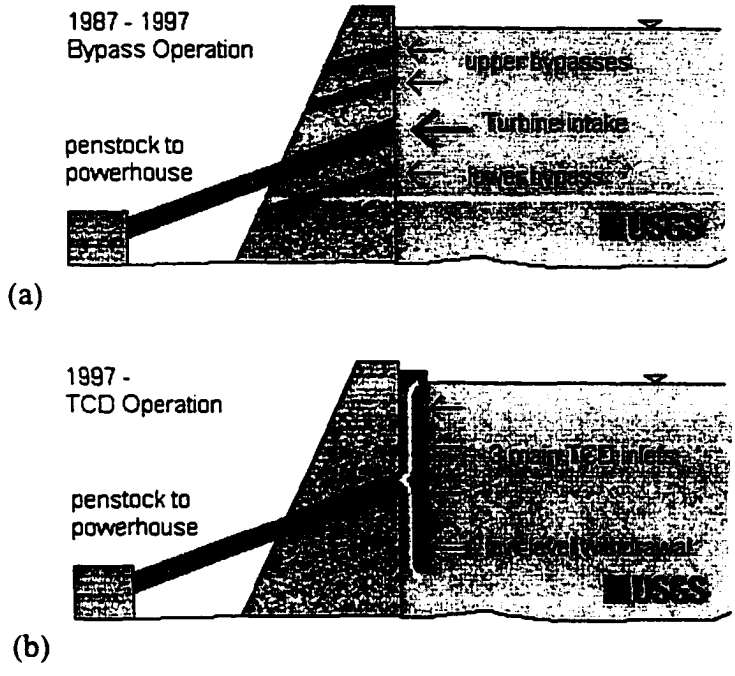

Figure 4. Schematics of releases from Shasta Dam (a) prior to installation of the temperature control device (TCD), and (b) after installation of the TCD

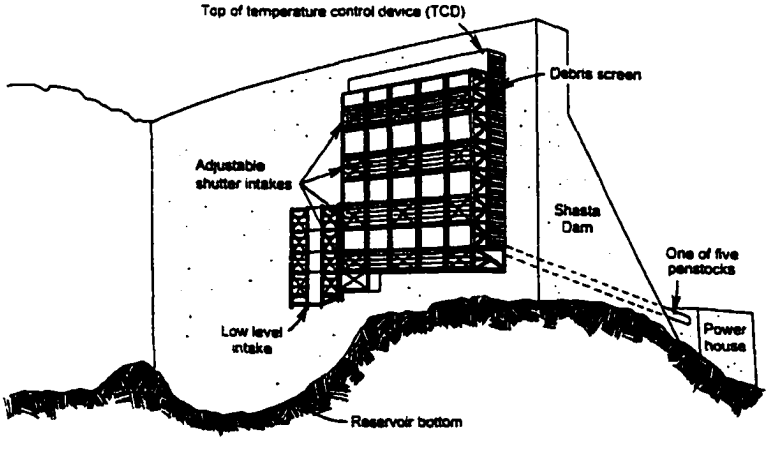

Figure 5. Shasta Dam TCD, with one adjustable shutter structure for each of the five penstocks to the power plant, and a low level intake structure

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The general operational strategy for the TCD is to release cold water in the summer and fall in order to reduce salmon mortality, and provide warm winter and spring releases to enhance salmon growth. The critical period for salmonid spawning and egg survival is September and October. Temperature is so important during this period because exceeding the temperature target by 2°C for a single day can be more detrimental to the chinook salmon population than being over by 1°C for two days (Kilgour et al. 1985). Thus, one of the intents of the TCD operation is to reduce both the magnitude and duration of high release temperatures.

Another concern of reservoir managers is the effect of the TCD operations on in-reservoir water quality and fishery dynamics. It is expected that the operational changes due to TCD could alter the patterns of thermal stratification, which in turn could affect nutrient and productivity dynamics in the reservoir. Changes in food availability or thermal habitat could then affect fish production in the reservoir.

A number of projects were undertaken in 1995 by the USBR and the U.S. Geological Survey (USGS, formerly National Biological Service (NBS)) to investigate in-reservoir and downstream effects of new reservoir operations following installation of the TCD. Projects (described below) included an entrainment study by the USBR; acoustic Doppler profiling by the USBR; a collaborative limnological study of the impact of reservoir releases on downstream productivity by the USGS and USBR; a collaborative in-reservoir limnological study by the USBR and USGS; and a reservoir modeling study by the USGS. Each of these projects gathered or used both pre- and post-TCD data (Hovecamp et al. 1996; NBS 1995). The entrainment study and acoustic Doppler profiling are ongoing studies; all other studies have been completed.

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2.1 Fish entrainment study

The fish entrainment study involves monthly monitoring of fish that are being entrained by the power plant releases (Figure 6). Because the releases in the spring and summer are likely to be from upper elevations of the reservoir, it is possible that all life stages (i.e., larvae, juvenile, and adult) may experience increased entrainment (Hovecamp et al. 1996).

Figure 6. Ray Bark collecting fish from entrainment nets at Shasta Dam

2.2 Acoustic Doppler profiling

An acoustic Doppler current profiler (ADCP) has been used at Shasta Lake to determine near-field and far-field hydraulic characteristics of reservoir withdrawals (Figure 7). A boat-mounted ADCP was used in 1995 and 1996 to measure pre-TCD velocity profiles around the penstock intakes and spillway outlet structure. After the TCD began operating, a self-contained acoustic Doppler profiler (ADP) was installed that has internal storage capacity and a processor so that it can remotely collect velocity

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profiles (Vermeyen 1997). The ADP is located at the reservoir surface approximately 400 feet upstream from the TCD and takes velocity measurements every ten minutes (Vermeyen 1998b).

Figure 7. Tracy Vermeyen and Steve Hiebert with acoustic Doppler current profiler

2.3 Downstream limnology study

Limnological data was collected in the tailwaters of Shasta Dam and Keswick Dam to assess the effects of the reservoir releases on downstream productivity (Figure 8). Measurements were made on a monthly basis at sites located approximately 0.8 and 14 km downstream of Shasta Dam. Another sampling site was located approximately 0.8 km downstream of Keswick Dam. Chlorophyll, nutrients, water temperature, pH, specific conductance, dissolved oxygen, and turbidity measurements were collected. In

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particular, the quantity and quality of particulate organic matter (POM) was investigated (Lieberman and Horn 1998; NBS 1995).

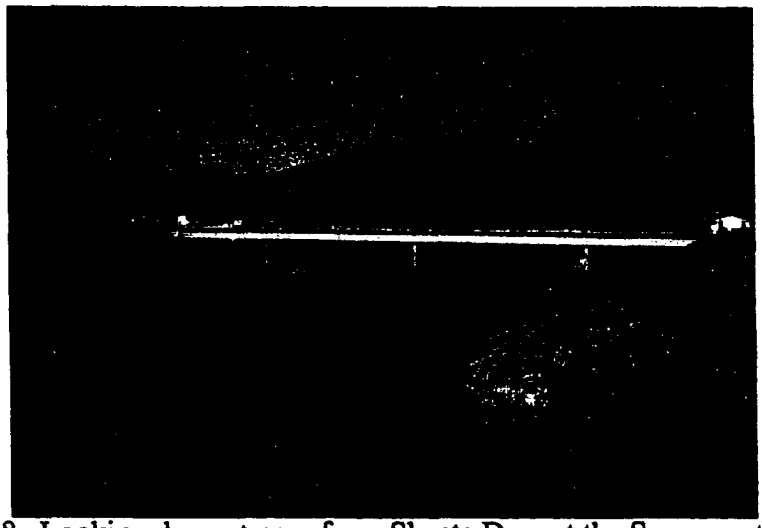

Figure 8. Looking downstream from Shasta Dam at the Sacramento River

2.4 In-reservoir limnology studies

The USGS conducted an in-reservoir limnology study that involved the monthly collection of biological, chemical and physical data from sampling stations throughout the reservoir (Figure 9). Data was collected in the Pit River, Sacramento River, and McCloud River arms and the main reservoir area from 1995 through 1999. Monitored parameters included water transparency, water temperature, dissolved oxygen, pH, conductivity, turbidity, dissolved inorganic nitrogen, ammonia, nitrite, soluble reactive phosphorus, total phosphorus, chlorophyll, phytoplankton, and zooplankton (Lieberman and Horn 1998; NBS 1995).

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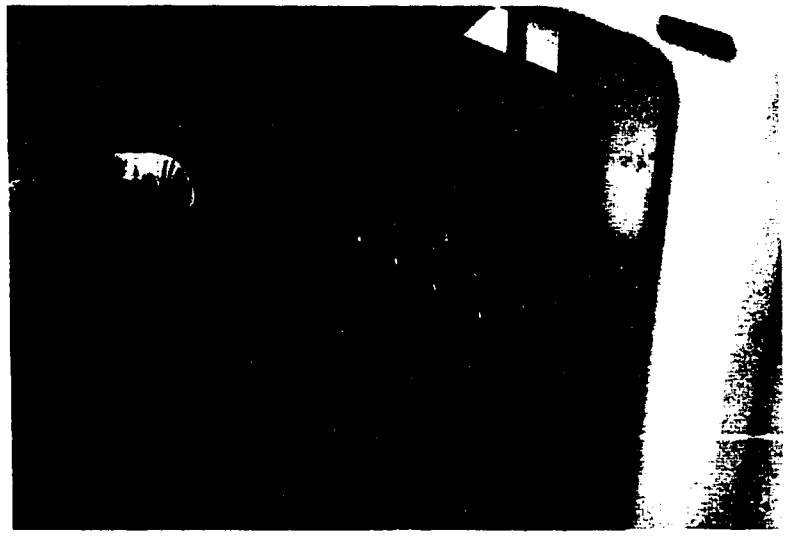

Figure 9. Davine Lieberman preparing phytoplankton samples on sampling boat

Another in-reservoir study was conducted by the USBR that evaluated water temperature, inflow and outflow data from a long-term data set and three years of limnological sampling at three sites in the main body of the reservoir. The objectives of the study were to understand the historical limnology of Shasta Lake, and predict the impacts of TCD operation on biological processes in the reservoir (Brett et al. 1998).

2.5 Reservoir modeling study

A reservoir modeling study was proposed in 1995 to develop fish growth, survival, reproduction, production, standing crop or habitat models for Shasta Lake that could simulate reservoir fishery responses to changes in reservoir operation. The research study plan proposed to develop a reservoir water quality model to evaluate the limnological effects of alternative reservoir operations. The biological, chemical, and physical data collected in the in-reservoir limnological study was to be used as input data and for calibration and validation purposes. A reservoir fish population and production

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model was also proposed to be developed, possibly as an advancement of the reservoir water quality model. Hydroacoustic surveys and fish sampling with vertical gill nets were anticipated to assist in the evaluation of the accuracy of the combined limnological and fish models (NBS 1995).

The NBS proposal also intended to interface the detailed reservoir models with reservoir operations by using data from the downstream limnological study with the reservoir models to determine recommended TCD operations. An optimal TCD outlet choice algorithm model was proposed for development as well (NBS 1995).

This reservoir modeling study was the case study project for this dissertation. Modeling objectives were to (Hanna et al. 1999):

1) compare and contrast outfall and reservoir water quality and conditions with and without the TCD operating
2) look at TCD operations and effects under a range of different hydrologic year conditions (i.e., wet, dry, normal, etc.)
3) determine the longitudinal and vertical extent of the TCD's effects in the reservoir
4) determine if there are any carryover effects of multi-year TCD operations
5) investigate the ability of using TCD operations to meet the downstream temperature targets under a variety of conditions
6) look at the potential effects of TCD operations on the opportunity for growth of sport fish species

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In particular, this dissertation addresses the interdisciplinary application of models to assess the effects of TCD operations on the in-reservoir sport fishery. The modeling project involved the investigation of changes in thermal and water quality patterns in the reservoir, and subsequent changes in patterns of nutrient availability, primary productivity, and fish growth in the reservoir.

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Chapter 3. LITERATURE REVIEW

Mathematical modeling is an appropriate technique for addressing the interdisciplinary study objectives listed in Chapter 2. Mathematical models employ one or more mathematical equations to quantify a system or process (Jørgensen 1988; Puccia and Levins 1985; Reckhow and Chapra 1983). The ideal model of Shasta Dam and Lake would be able to address:

- physical processes like circulation, mixing, thermal stratification, etc.
- physical components such as solar radiation, temperature, pressure, and density
- chemical processes such as oxidation-reduction; gas-solution; precipitation-dissolution
- chemical components such as dissolved organic and inorganic substances, particulate organic material
- biological processes such as primary production, photosynthesis, grazing, egestion, excretion, respiration, mortality, predation, and competition
- biological components such as primary producers, zooplankton, zoobenthos, fishes
- the interrelationships between all of these processes and components; for example, physical processes can move chemical and biological components, aquatic organisms can change the concentrations of constituents by metabolic uptake, transformation,

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storage, and release, and chemical and biological processes are affected by physical processes such as water temperature, pressure, available energy

It is evident that a diversity of disciplines would be needed to develop this ideal model. Hydrology and hydromechanics are needed to describe the movement and mixing of water. The description of interactions at the air-water interface requires the disciplines of meteorology and atmospheric physics. Chemistry, chemical kinetics, and biochemistry play parts in describing the fate of substances in water. Aquatic ecology is needed to model the biology, habitat and interactions between the different aquatic life forms present in the reservoir ecosystem (Orlob 1983c).

Historically, the development of mathematical models to address issues in reservoir ecosystems has been distinguished primarily by disciplinary efforts. This is partly because the issues themselves were assigned to particular disciplines, and the models were therefore structured according to their specific goal(s), which were influenced by the discipline that was developing them.

The two primary disciplines that have been involved in addressing the interdisciplinary issue of assessing water quality and fishery impacts of reservoir operations are engineering and ecology. Engineers, especially civil engineers, have been involved in developing water quality models to assist them in their responsibilities of planning and policy development, systems design and analysis, process and functional design, construction of physical works, and operation and maintenance of systems and facilities (Chapra 1997; Krenkel and Novotny 1980; McGauhey 1968). Engineers have generally focused on the physical and chemical processes and components of reservoir

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water quality models, which has led to the development of a class of models called 'hydrodynamic models.' These models include equations to describe reservoir hydrodynamic characteristics such as water velocities, velocity gradients, shears, and turbulence in order to model circulation and mixing phenomena (Mauersberger 1983).

Ecologists and biologists, on the other hand, have tended to focus on the ecological aspects of the system rather than on the hydrodynamics of circulation, mixing, and stratification. While ecological models can be used to simulate response to changes in chemical and biological conditions, interactions between physical processes and biological and chemical processes are limited (Hamilton and Schladow 1997; Orlob 1983a, 1983b). Popular biological models include index models such as morphoedaphic indices or habitat suitability indices, bioenergetics models, compartment models and dynamic individual-based and population models.

A literature review was made of models that have been applied to investigate the effects of physical and/or chemical processes on biological components of aquatic ecosystems, primarily in lake or reservoir systems. To describe these models, there are several different kinds of classifications that could be made:

- *Distributed-parameter versus lumped-parameter models*: In distributed-parameter models, parameter quantities are considered to be continuous functions of time and space. Lumped-parameter models assume that the parameters are constant within a defined volume (Beck 1983; Jørgensen 1988; Reckhow and Chapra 1983). Lumped-parameter models often consist of ordinary differential equations, while distributed-parameter models usually use partial differential equations (Beck 1983).

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- *Stochastic models versus deterministic models*: Stochastic models use probability density functions for certain parameters and variables, and the model predictions are therefore also probability density functions (Reckhow and Chapra 1983). If it is assumed that the variability and random error of the input data and parameters are zero, then the model is considered deterministic, and its predicted values do not allow quantification of variability or random error (Beck 1983; Jørgensen 1988).

- *Dynamic models versus steady-state models*: Dynamic models describe systems that vary with time, while steady-state models describe behavior that is assumed constant over time (Beck 1983; Jørgensen 1988; Reckhow and Chapra 1983).

- *Mechanistic models versus empirical or black-box models*: Empirically-based models are based more on analyzing data than on theoretical principles, while mechanistic models are based more on theoretical principles than on fitting data sets. An example of a typical empirical model is a multiple regression equation that predicts a certain parameter based on values of one or more other variables. In contrast, a mechanistic model will employ equations developed from theories such as the laws of thermodynamics to predict a certain parameter (Reckhow and Chapra 1983). Empirical models are sometimes referred to as black-box models because output from these models only reflect the effect of changes in input data, but the models do not represent the causes of these effects (Beck 1983; Jørgensen 1988).

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- *Reductionistic models versus holistic models*: Reductionistic models include as many details of the system as possible, while holistic models use general system principles to represent the system (Jørgensen 1988).

- *Qualitative models versus quantitative models*: Quantitative models produce numerical predictions for model output. Qualitative models do not require numerically precise input or parameters, and their output is consequently also not numerically precise. However, qualitative models can determine how to direct a system in a desired direction (Puccia and Levins 1985).

- *Simulation versus optimization models*: Simulation models describe the system, but interpretation of the 'best' outcome is left to the model user. Optimization models include algorithms that find a solution that is best, usually subject to constraints (Reckhow and Chapra 1983).

To describe models in this literature review, it was decided to classify the models as hydrodynamic, compartmental, ecologically mechanistic, or empirical. The first three categories are generally mechanistic approaches, although a certain degree of empiricism is always present when an equation is developed to fit observed data points. For example, bioenergetics models are arguably mechanistic because they include equations for several physiological processes. However, many of those equations were developed from curves fit to observed data.

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3.1 Hydrodynamic models

The classical mechanistic reservoir water temperature modeling approach involves the modeling of the hydrodynamics of the water body. This approach is based on the following general principles (Mauersberger 1983; Lynch 1986):

- conservation of mass
- conservation of momentum
- conservation of energy
- boundary conditions and initial conditions
- second law of thermodynamics

The hydrodynamics approach has been popular because modelers have recognized that the dynamics of lakes and impounded waters are driven by the movement and mixing of water due to variations in water density (Johnson 1981; Stefan et al. 1989). The reservoir's hydrodynamics affects water quality by transporting constituents via advection, diffusion, and dispersion. Advection refers to the portion of the constituent transport that is due to the directed movement of water. Therefore, the fluid velocity describes the constituent's advected velocity. Diffusion involves the movement of the constituent due to random water motion, and tends to cause the constituent to spread out from regions of high to low concentration. Transport due to random water motion on the molecular level is called molecular diffusion, while random water motion on a larger scale due to eddies causes turbulent diffusion. Dispersion also causes the constituent to spread out, but is due to water velocity gradients in space. In lakes, diffusion tends to dominate over dispersion in distributing constituents, while the opposite is true for

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advective systems such as streams or river-run reservoirs. Models may combine molecular diffusion, turbulent diffusion, and dispersion into one or more terms (Ferrara 1986; Chapra 1997).

Hydrodynamic models can be zero-, one-, two-, or three-dimensional in space. Zero-dimensional models assume that constituent concentrations or water temperatures do not vary in space. These models are also referred to as fully mixed models, completely mixed models, and completely stirred tank reactors. These models calculate a global balance over the entire volume of the reservoir (Ferrara 1986; Stefan et al. 1989).

One-dimensional (1D) reservoir or lake models consist of series of horizontal layers and allow for the spatial variation of constituents in the vertical direction. This enables thermal stratification to be simulated. Calculations are carried out of vertical lines in space (Ferrara 1986; Stefan et al. 1989).

Two-dimensional (2D) models are usually applied to long, deep reservoirs in which spatial uniformity is assumed along the width of the reservoir (i.e., lateral averaging) (Ferrara 1986; Stefan et al. 1989). Vertically averaged (i.e., single-layer) 2D models are sometimes used in shallow water bodies with unsteady flows (Watanabe et al. 1983).

The most complex models are the three-dimensional (3D) models, in which calculations are carried out over points in space. Constituent concentrations vary in all directions. Because of their computational intensiveness, 3D models are only recommended for large-scale lake situations such as the Great Lakes where an extensive data base exists (Ferrara 1986).

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One- and two-dimensional models have been applied frequently to reservoir and aquatic ecosystem issues (Table 2). The 1D models applied by Hamilton and Schladow (1997), Riley and Stefan (1988a, 1988b), and Summer et al. (1990) are all related to the model developed by Imberger et al. (1978). This model is distinguished by its integral energy approach to represent mixing processes. The 1D representation of the impoundment is composed of a thin surface layer that produces turbulent kinetic energy that gets exported to a central layer below it. These two layers represent the mixed, or epilimnetic layer. Below this layer, a thin layer separates the turbulent epilimnion from the relatively quiescent hypolimnion. The hypolimnion has a weak temperature gradient with sporadic mixing (Krenkel and French 1982). According to Hamilton and Schladow (1997), the hydrodynamic component of the model does not need to be calibrated because 'coefficients' in the model are purported to be constants, applicable for any lake (Krenkel and French 1982). Riley and Stefan (1988a, 1988b), however, included a calibration module with their version of the model that provides visual output to the user as well as goodness-of-fit estimators. They also stipulated that their model was designed to apply to small to moderate sized lakes (i.e., on the order of tens of square km or less).

In contrast to the integral energy models, 2D laterally averaged models such as CE-QUAL-W2 use longitudinal dispersion and vertical mixing of both momentum and heat. The turbulent incompressible flow and heat transport equations are averaged in the lateral direction, then integrated over horizontal slabs of uniform density (Krenkel and French 1982). A key assumption of this approach is that hydrostatic pressure is a function of depth only (Cole and Buchak 1995), which assumes that water density differences in the water column are negligible.

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Table 2. Hydrodynamic models for aquatic ecosystem issues

Citation	Location	Constituents[a]	Description	Calib/validation?
Chang et al. (1992)	Douglas Reservoir, TN	WT, DO, striped bass habitat	Applied BETTER (2D) with global climate models and striped bass habitat criteria to assess effects of global warming on striped bass habitat	Yes for WT, DO; no for global climate models and striped bass habitat/no
Chen and Wells (1976)	Boise River, ID	WT, DO, BOD, SS, NUTS, DET, COLI, ALGAE (2), ZOO (1), FISH (3), BENTH (1), AINS (1)	Used 1D river model with general mass balances for water quality and biological constituents to assess management alternatives for waste discharges	Yes for WT, DO, NUTS, COLI, ALGAE, BOD/no
Garvey et al. (1998)	Quabbin Reservoir, MA	WT, COLI	Applied CE-QUAL-W2 (2D) with coliform model developed from experiment	Yes for WT; no for COLI/no
Giorgino and Bales (1997)	Rhodhiss Lake, NC	WT, DO, NUTS, ALGAE (1), DOM, DET, FE, BOD	Used CE-QUAL-W2 (2D) to investigate effects of changes in wind, air temperature, and nutrient loadings	Yes/no
Hamilton and Schladow (1997)	General	WT, ALGAE (3), BOD, DO, NUTS, ZOO (1)	Describe DYRESM (1D)	Claim not necessary for WT
Imberger et al. (1978)	Wellington Reservoir, Australia	WT, SAL	1D precursor to DYRESM	Yes/no
Johnson et al. (1997)	Blue Mesa Reservoir, CO	WT, FISH (1)	Used CE-THERM (1D) with foraging model to provide inputs to bioenergetics model to predict changes in fish behavior and growth due to alternative reservoir management and hydrologic scenarios	Yes/no
Marotz et al. (1996)	Hungry Horse and Libby Reservoirs, MT	WT, ALGAE, ZOO, AINS, TINS, FISH (2)	Used 1D thermal model with a hydrologic model, primary productivity model, and fish growth efficiency models; other species were modeled with regression relationships	Yes/no
Martin (1988)	DeGray Lake, AR	WT, DO, NUTS, DOM, DS, ALGAE (1), FE	Applied CE-QUAL-W2 (2D) to look at reservoir dynamics	Yes/no
Martin and Wlosinski (1986)	DeGray Lake, AR	WT, DO, ALGAE (1), NUTS, DET, DOM	Compared CE-QUAL-R1 (1D) and CE-THERM-W2 (2D) predictions of DO	Yes/no
Ziransky and Shteinman (1993)	Lake Kinneret, Israel	WT, NUTS, DO, SS, others?	Applied BETTER (2D) to investigate sediment transport	No/no
Riley and Stefan (1988a, 1988b)	Several MN lakes	WT, CHLA (3), NUTS, DO, SS, DS, SI, ZOO	Applied MINLAKE (1D) to evaluate lake restoration alternatives	Yes (included in model)
Rounds and Wood (1998); Rounds et al. (1998)	Tualatin River, OR	WT, CL, SS, DS, DOM, ALGAE (1), DET, NUTS, DO, CAR, ALK, pH, ZOO (1), BOD	Used CE-QUAL-W2 (2D) to evaluate assimilation capacity of low-gradient stream reaches	Yes/no
Summer et al. (1990)	Eagle Lake watershed, MN	WT, NUTS, CHLA	Linked 1D model with AGNPS (watershed nonpoint source model) to evaluate effects of land use on watershed	Not reported/no
USACE (1995a)	General	WT, SS, DS, ALK, pH, COLI, FE, MN, SU, DET, ZOO FISH, DOM, NUTS, SI, ALGAE (3), MAC, C	Describes CE-QUAL-R1 (1D)	Not applicable

[a]
AINS = aquatic insects (# of species in parentheses)
ALGAE = algae or phytoplankton (# of species in parentheses)
ALK = alkalinity
BOD = biochemical oxygen demand (also carbonaceous BOD)
BENTH = benthos (# of species in parentheses)
CAR = carbon species (inorganic, CO_2, bicarbonate, or carbonate)
CHLA = chlorophyll *a* (# of species in parentheses)
CL = chloride
COLI = coliforms or fecal coliform
DET = detritus
DO = dissolved oxygen
DOM = dissolved organic matter
DS = dissolved solids
FE = iron
FISH = fish (# of species in parentheses)
MAC = macrophytes
MN = manganese
NUTS = nutrients (any nitrogen and phosphorus)
SAL = salinity
SI = silica
SS = suspended solids
SU = sulfur species
TINS = terrestrial insects
WT = water temperature
ZOO = zooplankton (# of species in parentheses)

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Water quality and biological processes in the hydrodynamic models in Table 2 tend to be modeled with an overall mass balance approach of keeping track of sources and sinks of constituents. Nitrification (i.e., $NH_3 \rightarrow NO_2 \rightarrow NO_3$) and oxygen processes are usually included, and models that include biological components such as algae and zooplankton account for nutrient uptake and release. Most of the models represent algae growth as being limited by either light or nutrients (Bender et al. 1990; Chen and Wells 1976; Cole and Buchak 1995; Hamilton and Schladow 1997; Riley and Stefan 1988b; USACE 1995a). CE-QUAL-W2 and BETTER only simulate one algal compartment (Bender et al. 1990; Cole and Buchak 1995), while other models have included more than one algal compartment, zooplankton, and/or fish (Chen and Wells 1976; Hamilton and Schladow 1997; Riley and Stefan 1988a, 1988b; USACE 1995a).

3.2 Compartment models

While the water quality and biological components of most of the hydrodynamic models resemble compartment model approaches, in this literature review, the compartment model classification is reserved for ecological models that do not include hydrodynamic processes. Generally, the compartment models included in Table 3 include more detail on the dynamics of each compartment than their counterpart applications in the hydrodynamics models.

Rather than model water temperature and dissolved oxygen as in the hydrodynamic models, compartment models include them as input variables if they are used in the model (Canale et al. 1976; Cochrane et al. 1987; Jørgensen 1983). Some

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compartment models include life history characteristics such as spawning for fish (Cochrane et al. 1987) or foraging for zooplankton (Canale et al. 1976).

Table 3. Compartment models for aquatic ecosystem issues

Citation	Location	Constituents[a]	Description	Calib/validation?
Burns et al. (1991)	A continental shelf	ALGAE (2), ZOO (2), FISH (2), BACT (3), BENTH (2), DET, POM, DOM, AINS	Energy-flow network using trophic unfolding concept	No/no (compared to predictions from another model)
Canale et al. (1976)	Lake Michigan	NUTS, SI, ZOO (9), ALGAE (4)	Used to estimate effects of eutrophication and higher forage fish levels on plankton and nutrient distributions	Yes, visually/no
Christensen and Pauly (1993b)	General; reference includes many applications of model	Any trophic compartment	ECOPATH, a steady-state network model	No/no (developed models from field data)
Cochrane et al. (1987)	Hartbeespoort Dam, South Africa	NUTS, CHLA, ZOO, FISH (3)	Separated reservoir into epilimnion and hypolimnion with different processes in each; used to evaluate management alternatives to control eutrophication	Yes for NUTS, CHLA, ZOO/no
DeAngelis et al. (1989)	Conceptual	NUTS, DET, MAC, ALGAE, ZOO, FISH, BENTHOS, BACT	Investigated food web resilience in response to changes in nutrient input and trophic structure	No/no
Flint (1986)	Lake Ontario	ALGAE (1); ZOO (2); BENTH (1); FISH (5); MYSIDS	Modeled carbon flow by estimating food web and transfer efficiencies; used to estimate food web requirements for top predators	No/no (but developed from field data)
Jørgensen (1983)	General	NUTS, DET, BENTH, ZOO, FISH, DO, BACT	Description of dynamic model	No/no
Ulanowicz (1986)	Conceptual	Any trophic compartment	Flow networks	No/no
Walters et al. (1997)	General	Any trophic compartment	ECOSIM, a dynamic version of ECOPATH	No/no

[a] Abbreviations are same as for Table 2

Several compartment model approaches emphasize energy or material flows through food webs or trophic networks. These approaches are based on concepts of energy flow introduced by Lindeman (1942) and applied by Odum (1957) in which available production decreases with higher trophic levels. These applications generally do not include temperature-dependent rates, but use inputs of biomass or production for each compartment instead, and then estimate flows between compartments. Network flow analysis, which has been incorporated into the generalized ECOPATH model of

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Christensen and Pauly (1992), has been applied in numerous ecosystems using 'known' inputs of biomass for various compartments of fish, zooplankton, detritus, benthos, etc. (Burns et al. 1991; Christensen and Pauly 1993b; Halfon et al. 1996; Heymans and MacLachlan 1996; Monaco and Ulanowicz 1997; Niquil et al. 1999; Pace et al. 1984; Whipple 1998). Because the approaches used in all of these applications are so similar, not all are listed in Table 3. Calibration of these models is usually not done because they are constructed from 'known' biomass or production values, although the biomasses and productions are usually estimated from available data. In addition, flows between compartments are difficult to measure.

Most of the network analysis models are steady-state models (Burns et al. 1991; Christensen and Pauly 1993b; Halfon et al. 1996; Heymans and McLachlan 1996; Monaco and Ulanowicz 1997; Niquil et al. 1999; Pace et al. 1984; Ulanowicz 1986; Whipple 1998), but Walters et al. (1997) modified ECOPATH so that it could model dynamic systems. Their model, called ECOSIM, enables the modeling of multispecies dynamics, but, as with ECOPATH, it has not been calibrated (Walters et al. 1997).

3.3 Ecologically mechanistic models

While the compartment models are arguably ecologically mechanistic, this category of models includes those that specifically model the physiology, life history or movements of biological organisms (Table 4). Individual-based models are reductionist approaches that track the attributes of individual fish through time and space. Typically, such models include phenotypic-specific feeding and predation, as well as growth and mortality. They have most often been applied to young-of-year fish, although they can be

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used to simulate all life history stages (Van Winkle et al. 1993a). Although designed for the responses of individuals to their local environment, Huston et al. (1988) argue that individuals from these models can be aggregated to investigate population, community or ecosystem dynamics. However, the detail and data required at the individual level make the feasibility of such applications difficult at best. At a much lower level of resolution, population dynamics models can incorporate many of the same characteristics of individual-based models, but with generalizations for populations. For example, Cole et al. (1995) incorporated a population dynamics model in their comprehensive management model for New Mexico sport fisheries that includes egg, larval, fishing, and natural mortality as well as reproduction and fish stocking (Cole et al. 1995). At an intermediate level of resolution, other fish dynamic models follow cohorts through life history stages in space and time (Bartholow et al. 1993). In general, most of these models are not calibrated because of the large amount of field data that would be needed. Bartholow et al. (1993) performed a 'qualitative' calibration using measured, but uncertain, data for outmigrating fish, while Van Winkle et al. (1997) applied a 'reference' calibration in which they selected desired model growth, physiological condition, and reproductive success based on literature and experience that model parameters were adjusted to meet.

Bioenergetics models are another popular mechanistic approach that incorporate the energetic components of fish physiology by applying an energy mass balance in which consumed energy is balanced by energy losses due to metabolism, waste losses, and reproduction and growth (Hanson et al. 1997; Hewett 1989):

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Table 4. Ecologically mechanistic models for aquatic ecosystem issues

Citation	Location	Constituents[a]	Description	Calib/validation?
Bartholow et al. (1993)	Trinity River, CA	Chinook salmon	Salmonid population model that tracks spatially distinct cohorts; can be linked with stream temperature model and physical habitat simulation system model	Yes/no
Brandt et al. (1992); Goyke and Brandt (1993); Mason et al. (1995)	Chesapeake Bay, Lake Ontario, Lake Michigan	Striped bass, chinook salmon, lake trout	Used hydroacoustics to gather data on prey density and distribution that was input into foraging model; measured WT and DO were input with foraging data into bioenergetics model to spatially evaluate fish growth	No/no
Cole et al. (1995)	State of NM	SS, NUTS, WT, NPP, FISH, BENTH, AINS, CRA	Fish population dynamic model is linked with hydrologic and economic models to evaluate state management options	None given/no
Hayes et al. (1996)	Conceptual	Lake whitefish	Linked habitat conditions and empirical stock-recruitment relationships to fish population dynamics	No/no
Johnson et al. (1997); Stockwell and Johnson (1997)	Blue Mesa Reservoir, CO	Kokanee salmon	Used hydrodynamic model (CE-THERM) with foraging model to provide inputs to bioenergetics model to predict changes in fish behavior and growth due to alternative reservoir management and hydrologic scenarios	Yes/no
Post and McQueen (1994)	Experiment	Yellow perch	Used type II functional response for consumption and foraging costs with bioenergetics model	No (developed from experimental data)/no
Van Winkle et al. (1993a)	Synthesis	Smallmouth bass, bluegill, Atlantic cod, capelin, winter flounder, striped bass, lake trout, spot	Individual based models to predict fish responses or examine populations trends	No (often developed from field data)/no
Van Winkle et al. (1993b)	Not applicable	Striped bass, smallmouth bass	Compared responses of fish with different life history characteristics using individual-based model	No/no
Van Winkle et al. (1997)	Not applicable	Rainbow trout	Simulated effects of climatic temperature change with individual-based model	'reference' calibration/no
Wilson et al. (1991)	Conceptual	Multiple FISH	Dynamic model includes community predation	No/no

[a] Abbreviations are same as for Table 2, plus
CRA = crayfish
NPP = net primary productivity

$$C = R + F + U + \Delta B \qquad \text{Equation 1}$$

where

C = energy of food consumed

R = energy of metabolism

F = energy egested

U = energy excreted

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ΔB = energy invested in growth

Metabolism includes standard metabolism (respiration), specific dynamic action, and activity. The energy invested in growth includes somatic growth and gonad production. Available energy is expended according to a hierarchy in which metabolism requirements are satisfied first, followed by egestion and excretion requirements. Remaining energy is then considered available for growth (Hewett 1989). Although there have been numerous bioenergetics applications using the generalized version of this model (see Hanson et al. 1997), all of the bioenergetics applications in Table 4 were linked with other models such as foraging models for food consumption or water temperature models (Brandt et al. 1992; Goyke and Brandt 1993; Johnson et al. 1997; Mason et al. 1995; Post and McQueen 1994; Stockwell and Johnson 1997).

3.4 Empirical models

As noted previously, empirical models are 'curve-fitting' approaches, in which equations are derived to fit observed data. The strength of these models is their generality and ease of application, and their ability to generate testable hypotheses (Peters 1986). The uncertainty of predictions with these methods is also usually quantifiable. As seen in Table 5, a wide variety of these approaches are used to address aquatic ecosystem issues. Many of these models were developed to address eutrophication issues, so most include some means of estimating the effects of nutrient loads on algae, chlorophyll *a*, or fish.

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Table 5. Empirical models for aquatic ecosystem issues

Citation	Location	Constituents[1]	Description	Calib/validation?
Canfield and Bachmann (1981)	704 natural and artificial north temperate lakes	NUTS, CHLA, SD	Determined regression relationships between constituents and physical lake variables	Yes /no (randomly divided data set in two)
Christie and Regier (1988)	21 large north temperate lakes	THA, THV, FISH (4)	Used fish temperature preferendums, measured WT, and lake hypsography to determine THA, THV; correlated THA and THV with sustained yield	Yes/no
Fukushima and Muraoka (1988)	90 Japanese lakes	CHLA, COD, NUTS	Correlated NUTS with COD and CHLA	Yes/no
Hanson and Leggett (1982)	4 data sets of north-temperate lakes from literature	DS, NUTS, FISH, BENTH	Investigated regression equations and MEI between constituents and lake characteristics to predict fish yield	Yes/no
Heidtke et al. (1986)	Green Bay, WI	NUTS, CHLA, SD, TRST	Obtained phosphorus loading from nonpoint model; used steady-state model to determine spatial phosphorus profiles, then applied empirical formulas to determine CHLA, SECCHI, and TRST to evaluate water quality response to changes in land use	Yes/no
Jenkins (1982)	294 reservoirs > 202 ha in US	FISH, DS	Applied MEI to estimate reservoir sport fish crop and yield	Yes/yes
Jones et al. (1998)	119 Missouri reservoirs	CHLA, NUTS	Looked at different levels of aggregation of data to see if that affected correlations	No
Magnuson et al. (1990)	Lake Michigan and Lake Erie	WT, TH,FISH (3)	Global climate models provided input into limnological models that determined WT; estimated TH from WT and hypsography; evaluated sustained yield using THV	Yes/no
Marotz et al. (1996)	Hungry Horse and Libby Reservoirs, MT	WT, ALGAE, ZOO, AINS, TINS, FISH (2)	Used 1D thermal model with a hydrologic model, primary productivity model, and fish growth efficiency models; other species were modeled with regression relationships	Yes/no
Ploskey et al. (1986)	607 reservoirs in US	NUTS, ALGAE, ZOO, SD, CHLA, DS, FISH	Comprehensive investigation of correlations between physical characteristics of reservoirs and constituents to predict fish yield	Yes/no
Reckhow (1988)	80 lakes and reservoirs in southeastern US	NUTS, CHLA, SD	Developed empirical relationships to predict constituents	Yes/no
Regier and Meisner (1990)	General	WT, NUTS, CHLA, FISH	Used empirical relationships to show linkages between climate change, water quantity, fish standing crop, water quality	Yes/no
Seip and Ibrekk (1988)	8 Norwegian lakes	ALGAE, NUTS, ZOO, FISH, MAC, SD, CHLA	Used an expert system to choose a series of regressions to improve predictions to assess effects of lake eutrophication	Yes/no
Walker (1986)	USACE reservoirs	NUTS, CHLA, DO, SD	Empirical model with nutrient balance and eutrophication models in a spatially segmented network; incorporates some uncertainty estimation	Yes/no

[1] Abbreviations are same as for Table 2, plus
COD = chemical oxygen demand
MEI = morphoedaphic index
SD = Secchi disk measurements
TH = thermal habitat
THA = thermal habitat area
THV = thermal habitat volume
TRST = trophic state

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A commonly-used empirical approach is the use of regression analysis, which is also known as least squares analysis. The typical application of this approach involves predicting the value of a variable y by using one or more measured variables x_j using a linear relationship (Reckhow and Chapra 1983):

$$y = \beta_0 + \sum_{j=1}^{k} \beta_j x_j \qquad \text{Equation 2}$$

where

β's = regression coefficients

k = total number of measured variables

For example, Seip and Ibrekk (1988) included a relationship between chlorophyll a ($CHLA$) in mg m^{-3} and total phosphorus (p) in mg m^{-3} in their model:

$$CHLA = 0.55p - 1.03 \qquad \text{Equation 3}$$

The most common nonlinear regression approach involves the logarithmic transformation of the measured and/or predicted variables. For example, if both measured and predicted variables are transformed by taking their base 10 logarithms, the resulting relationship would be of the following form (Reckhow and Chapra 1983):

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$$\log y = \beta_0 + \sum_{j=1}^{k} \beta_j \log x_j$$ Equation 4

where the parameters are as defined in Equation 2. A commonly used relationship between *CHLA* (mg m^{-3}) and *p* (mg m^{-3}) of this form is (Chapra 1997):

$$\log(CHLA) = 0.76 \log(p) - 0.259$$ Equation 5

In applying these regression approaches, the modeler uses some sort of squared deviation criterion to determine which regression equation and associated coefficients best fits the data. Thus, most of these models are not validated to an independent data set although they are generally considered calibrated to the original data set.

Another empirical approach is based on the phosphorus mass balance model for a well-mixed lake (Chapra 1997):

$$V \frac{dp}{dt} = W - Qp - k_s Vp$$ Equation 6

where

V = volume (m^3)

p = total phosphorus concentration (mg m^{-3})

t = time (yr)

W = total phosphorus loading rate (mg yr^{-1})

Q = outflow (m^3 yr^{-1})

k_s = first-order settling loss rate (yr^{-1})

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At steady state, this equation becomes:

$$p = \frac{W}{Q + k_s V}$$ Equation 7

If everything is divided through by Q, the general model becomes:

$$p = \frac{p_{in}}{1 + k_s T_w}$$ Equation 8

where

p_{in} = inflow concentration of total phosphorus (mg m^{-3})

T_W = hydraulic detention time (yr)

This model has been widely used, and modifications of this approach were included in Canfield and Bachmann (1981), Fukushima and Muraoka (1985), and Reckhow (1988).

Another popular use of empirical models is as indices. For example, one of the most widely used indices is the morphoedaphic index (MEI), which predicts annual fish yield from empirical relationships with abiotic factors. The underlying concept is that the morphology of a system determines how energy and matter introduced into that system are channeled, cascaded, dissipated, or retained (Ryder 1982). The basic morphoedaphic index is:

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$$MEI = \frac{N}{\bar{z}}$$ Equation 9

where

N = represents a nutrient value

$\bar{z}$ = mean depth of lake (m)

The predicted fish yield can be determined from its correlation with the MEI, which is determined from regional lake data sets (Wootton 1990). In the original index, the nutrient value used was the concentration of total dissolved solids. Ryder (1982) stated that the nutrient value could be the concentration of parameters such as alkalinity, phosphorus, nitrogen, chlorophyll *a*, algae, zooplankton, or benthos. Because MEIs are derived from lake data sets for certain regions, modifications may be necessary if they are to be applied in areas with different climatic characteristics. Leach and Herron (1992) state that the MEI was found to be the best overall estimator of fish yield in the Great Lakes when compared with seven other empirically-derived fish yield models. Jenkins (1982) investigated the use of the MEI with different predictor variables for reservoirs, but concluded that total dissolved solids and mean depth were still the best predictor variables.

Another set of popular empirical models is the stock-recruitment models. The idea behind these models is that the stock of parental cohorts can be used to predict the recruitment of their offspring into the fishery. In 1957, Beverton and Holt proposed the following stock-recruitment curve (Wootton 1990):

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$$R = \frac{1}{\left(a + \frac{b}{P}\right)}$$ Equation 10

where

R = number of recruits

P = number of parental cohorts or stock

a = curve parameter

b = curve parameter

The Beverton-Holt curve is characterized by an asymptote, which might represent a maximum species abundance due to food availability or space. For species in which high densities lead to decreases in recruitment, Ricker proposed an alternate curve in 1975 (Wootton 1990):

$$R = aPe^{-bP}$$ Equation 11

where the parameters are defined as in Equation 10.

3.5 Other models

There are other models that are not easily classified into the breakdowns chosen for this literature review (Table 6). The only one that has had considerable use in applied aquatic ecology is the habitat suitability index (HSI) approach. HSI models describe habitat and can be used to evaluate the suitability of fish habitat and predict fish standing crop or production. These models produce an HSI that is a unitless number between 0

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and 1. An HSI of zero indicates completely unsuitable habitat, while optimal habitat has an HSI of one. There are three degrees of HSI models: 1) regression HSI models; 2) descriptive HSI models; and 3) 'mechanistic' models based on suitability indices (Terrell et al. 1982). The regression models were developed by the National Reservoir Research Program and are similar to the regression approaches of Ploskey et al. (1986). Descriptive HSI models rate habitat on the presence or absence of optimal or selected values of certain habitat variables. These models are black-box models that do not attempt to model cause and effect relationships between variables (Terrell et al. 1982).

Table 6. Other models for aquatic ecosystems

Citation	Location	Constituents[a]	Description	Calib/validation?
Bosserman and Ragade (1982)	Conceptual	Primary producers, carnivores, herbivores, detritivores	Fuzzy model approach	No/no
Oksanen et al. (1981)	Conceptual	Primary producers, herbivores, carnivores	Isoclinic approach to study interactions	No/no
Scheffer (1991)	Conceptual	NUTS, FISH, ALGAE, ZOO	Isoclinic approach	No/no
Terrell et al. (1982)	Conceptual	Many different species of fish	Habitat suitability indices approach	No/no

[a] Abbreviations are same as for Table 2

The 'mechanistic' HSI models are developed in a series of steps designed to use variables that represent key habitat features in mathematical equations that yield a single numerical description (i.e., HSI) of habitat suitability. The relationship between each variable and species carrying capacity is represented graphically. These graphs are used to determine the appropriate values of the habitat variables, which are then aggregated mathematically into life requisite equations for food, cover, water quality, reproduction, life-stage, or other components. These model components are then aggregated into a species HSI equation with another mathematical equation (Terrell et al. 1982).

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These models are described as 'mechanistic' because their structure allows modelers to trace the effects of habitat changes on overall habitat suitability. The hierarchical nature of the models also allows easy modification to incorporate different or site-specific habitat requirements. However, these models are difficult to calibrate because HSI cannot be directly measured. According to Terrell et al. (1982), the ability of mechanistic HSI models to predict species-specific standing crop was very low. However, Terrell et al. (1982) advocate the applicability of these models for planning purposes where quantitative accuracy is not necessary.

The fuzzy modeling approach of Bosserman and Ragade (1982) is similar to the HSI approach because it involves rule curves for different model compartments. The rule curves reflect the variability or continuous (i.e., nondiscrete) nature of the input variables.

Finally, the isocline approaches of Oksanen et al. (1981) and Scheffer (1991) are purely conceptual, and therefore not very useful for aquatic ecosystem management. The approaches generate predictions but are not based on any actual measured data, and the concepts are very abstract and difficult to follow.

3.6 Issues concerning model application

Each of the modeling approaches described in the literature review has its advantages and disadvantages. Because models are simplifications of reality, they are incomplete and have inherent biases because the modeler has chosen to ignore certain aspects of reality in order to group phenomena or simplify mathematical relations. Puccia and Levins (1985) note that the ideal model would maximize precision, realism and generality, but there seems to be a fundamental principle of modeling that all three

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cannot be maximized at the same time. Precision has to do with how closely the model mimics certain system variables (i.e., how many significant figures), while accuracy reflects how realistically the model accounts for all relevant variables and relations. The generality of a model allows it to be applied across a broad range of similar systems (Hall and Day 1977).

The models presented in the literature review tend to have tradeoffs of complexity and accuracy. While real ecosystems are acknowledged to be highly complex, no model can be accurate over the complete range of ecosystem structure and behavior, so sacrifices have to be made. For example, dynamic models are complex in time, spatial models are articulated in space, and compartment models are articulated in system components (state variables). Models that are highly accurate tend to have narrow perspectives and do not say enough about system interactions (i.e., they say much about little), while comprehensive models are more articulated (complex), but their accuracy suffers (i.e., they say little about much) (Beck 1983; Costanza and Sklar 1984; Reckhow and Chapra 1983). Empirical models generally fall into the former category, while mechanistic models usually fall into the latter.

Models tend to have three types of error in their results, and the seriousness of each of these depends on the complexity of the model (Crockett 1994):

1. *Simplifications of the system made by the model*: Ideally, a model helps to make some phenomena more understandable, but at the same time, it creates sources of error because of the assumptions used to simplify reality (Puccia and Levins 1985).

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2. *The numerical solution scheme*: Models that employ numerical techniques to solve differential and partial differential equations can have problems due to numerical dispersion and stability.

3. *Inaccuracies in the input data*: Errors can be inherent in the data used, or the model can interpret the data incorrectly. One must be aware of the uncertainties in the input data, from measurements to assumptions to model objectives (Loucks 1983).

The first error is a problem for all models, but is especially an issue with empirical models. Because they are generally simple and use data that is easy to gather, it is easy for them to be misapplied. The empirical models do not include the causal mechanisms for the relationships that they use, so applying the models outside of the range in which they were developed can be inappropriate. For example, Jones et al. (1998) looked at a data set of 119 Missouri reservoirs under different temporal aggregations and found that the interpretation of the data was misleading if empirical inferences were made at the wrong scale. Caution should also be used when applying empirical relationships across boundaries such as spatial scales (e.g., to different regions, or to individual lakes when developed for regional interpretation), species (e.g., to bass species when developed for trout), or any other variable that might be outside of the range for which the relationships were originally developed.

The second issue is a concern for models that incorporate numerical methods. Most empirical models do not have this problem, especially those that use simple linear regressions.

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The third issue is a problem for all models, but the relative differences in data uncertainty are probably a source of misunderstanding between engineers and ecologists in applying models. Physical and chemical data are generally much easier to collect, interpret and model than biological data (Crockett 1994). Because there are usually many more data points for physical and chemical data, the uncertainty of the data is easier to quantify, and should actually become smaller with more observations if the errors are truly random (Winter 1981). In contrast, ecological data is more difficult to gather and replicate, and be statistically significant (Carpenter 1989). This is one of the reasons why most of the models in Table 2 have been calibrated, while most of the models in the other tables have not been. Data needed to calibrate hydrodynamics models are relatively easy to collect with small but quantifiable data uncertainty compared to data for a network flow model or an individual-based fish model.

3.7 Linked models

Because models that attempt to include lots of detail quickly become too complex, some level of detail must be sacrificed to maintain the utility and understandability of the model (Fagerström 1987; Heidtke et al. 1986; Hildén and Kettunen 1985). An alternative approach that was employed by a number of the studies in this literature review was linked modeling. Several researchers have advocated linked modeling approaches in aquatic ecosystem modeling (Crockett 1994; DeAngelis and Cushman 1990).

A major advantage of this approach is that well-accepted models for specific processes can be used. For example, hydrodynamics models are good at modeling water

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temperature and water movement, and bioenergetics models are good at modeling fish growth at different water temps. Linking the water temperatures from the hydrodynamics model with the bioenergetics model allows both models to be used without compromising their proven methodology.

The other advantage of linked modeling approach is that it is simpler and easier to follow. Models that are too huge and complicated will not be used because they cannot be understood (Fagerström 1987; Hildén and Kettunen 1985).

The approach does not allow the simultaneous modeling of all processes and variables because data from one model must be taken and passed to another. There is also a danger that uncertainties from one model are passed on to the next model and so on (DeAngelis and Cushman 1990), but comprehensive models will also have errors that are propagated. It may actually be easier to trace such error with linked models because it is not hidden in the internal interactions in the model.

Four of the hydrodynamics model applications in Table 2 involved the linkage of the hydrodynamics model with another model. Chang et al. (1992) used global climate models to provide meteorological input into the BETTER model, and then use the water temperature and dissolved oxygen predictions from BETTER to assess changes in striped bass habitat due to global warming. To evaluate the effects of reservoir operations and hydrologic conditions on kokanee salmon behavior and growth, Johnson et al. (1997) linked CE-THERM-predicted water temperatures with foraging and bioenergetics models. Marotz et al. (1996) used the water temperature predictions from a 1D thermal model in a linear regression relationship for primary productivity and in growth models for trout and kokanee salmon. They used the model to assess effects of reservoir

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operations on the riverine and reservoir fisheries. The other application involved linking water quality output from a watershed-level nonpoint source model with a 1D lake model to evaluate the effects of land use on the watershed (Summer et al. 1990). In addition, Hildén and Kettunen (1985) suggested that a hydrodynamic model could be linked with a model of environmental change and a fish population model to evaluate environmental impacts on fish stocks and fisheries.

In addition to the linkages mentioned to hydrodynamics models, global climate models have been linked to empirical models to predict effects of climate change on fisheries (Magnuson et al. 1990; Regier and Meisner 1990), and bioenergetics models have been applied with foraging models (Brandt et al. 1992; Goyke and Brandt 1993; Mason et al. 1995; Post and McQueen 1994). The dynamic fish population model of Bartholow et al. (1993) could be linked with a stream temperature model and a physical habitat simulation model. Other researchers have applied series of empirical models to address lake eutrophication (Fukushima and Muraoka 1988; Heidtke et al. 1986; Reckhow 1988; Seip and Ibrekk 1988; Walker 1986).

A linked modeling approach using the two-dimensional hydrodynamic and water quality model CE-QUAL-W2 and two ecological models was selected to address the assessment of the effects of the TCD on the reservoir fishery at Shasta Lake. Water temperature output from CE-QUAL-W2 was input into a bioenergetics model to predict fish scope for growth through space and time in the reservoir. This linkage was most similar to the model approach of Johnson et al. (1997), but provided better spatial resolution on the thermal impacts of the TCD because of the two-dimensional representation of the reservoir. It was also an improvement over the linkages of

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bioenergetics applications in Chesapeake Bay, Lake Michigan and Lake Ontario (Brandt et al. 1992; Goyke and Brandt 1993; Mason et al. 1995) because the model was able to simulate water temperatures through time and space.

The other ecological model linked with CE-QUAL-W2 was a food web-energy transfer model. The food web model was developed using stable isotope analysis. The food web-energy transfer model was similar to the compartment models of energy flow, but not as sophisticated and complex because biomass or quantifiable fish data was not available at Shasta Lake. Although stable isotope analysis has been used to investigate food web interactions in previous studies (see description in Chapter 7), it has only been applied in a predictive manner with a multiple regression relationship with lake characteristics such as phosphorus and color (France et al. 1997). The combination of stable isotope analysis with a food web-energy transfer model that relates net algal production estimates from CE-QUAL-W2 with fish growth is an innovative, but simple to follow approach, as detailed in the remainder of this dissertation.

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Chapter 4. METHODS - LINKED MODELING

The linked modeling approach at Shasta Lake involved the application of three models: a two-dimensional hydrodynamic and water quality model (CE-QUAL-W2), a bioenergetics model, and a food web-energy model. This chapter describes the manner in which these models were linked. Subsequent chapters describe the development and application of each of the models separately.

The three models were linked in the following manner (Figure 10):

- CE-QUAL-W2 was run to generate water temperature and phytoplankton predictions through space and time under different operating and hydrologic scenarios

- The water temperature predictions were input into the bioenergetics model to develop predictions of scope for growth for fish species through space and time

- The phytoplankton predictions were input into the food web-energy transfer model to develop predictions of changes in fish growth over the stratified period under different hydrologic conditions due to the different operating scenarios

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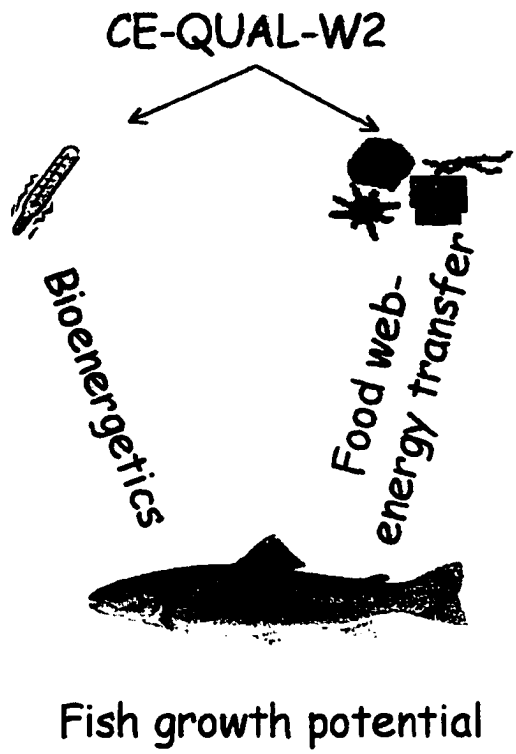

Figure 10. Schematic of model linkages for Shasta Lake project

The linkage of bioenergetics with the water temperatures from the hydrodynamics model was a logical extension of the merits of both of these models. Because hydrodynamics models incorporate the movement and mixing of water due to variations in water density (Johnson 1981; Stefan et al. 1989), they have been very successful at predicting water temperature patterns in lakes and impounded waters (Garvey et al. 1988; Giorgino and Bales 1997; Imberger et al. 1978; Martin 1988; USACE 1995b). Many thermal models can predict temperatures throughout the water column to within 1°C, with the main limitation in predictive ability being the quality of the input data (Imberger 1985). It is also well known that the thermal environment affects fish from incubation in the egg stage to movement, digestive and metabolic processes, growth and reproduction,

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location and survival of adults (Coutant 1976). Bioenergetics models that incorporate temperature-dependent energy balances to model fish energetics have been used for many years (Hanson et al. 1997; Hewett 1989). However, Brandt and Hartman (1993) pointed out that one of the limitations of applying bioenergetics was an assumption of water temperatures. Thermal changes in time periods of less than one day were often not available even though fish physiological rates and processes change over the course of the day. By linking the bioenergetics model with the hydrodynamics model, this limitation can be reduced because hydrodynamics modeling allows good water temperature predictions to be made over time and space on a less-than-daily basis.

To facilitate the linkage between CE-QUAL-W2 and the bioenergetics model, Jeff Sandelin, a contractor with the USGS, created a graphical user interface using Visual C++ that read output files from CE-QUAL-W2 and applied the bioenergetics equations for rainbow trout and smallmouth bass. Although a Windows-based bioenergetics model was available for over 20 species of fish (Hanson et al. 1997), the interface and corresponding programs were needed in order to apply the bioenergetics equations through space and time. This interface also enabled the output of water temperature, water quality, and fish growth predictions in spreadsheet or Tecplot[1] format.

The linkage of the food web-energy transfer model to CE-QUAL-W2 was not as straightforward as the linkage of the bioenergetics model. Because physiological ecology is traditionally closely connected with the physical sciences, it is easy to show the connection to causal mechanisms such as water temperature. However, when broadening

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[1] Tecplot is a graphical plotting program that was used to generate figures displaying data over time and space for visualization of model predictions

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predictive modeling to ecological communities and ecosystems and their dynamics, it is harder to make such connections (DeAngelis 1988). Because of the limited food web data available, the food web-energy transfer model was not dynamic in space or time. Instead, it was developed as a representation of the general food web during the period in which the reservoir was stratified when most biological growth was expected to occur. The phytoplankton output from CE-QUAL-W2 was therefore synthesized into overall predictions of the total net algal production in the whole reservoir and in the epilimnion over the stratified period (see Chapter 7).

To facilitate this synthesis, Jeff Sandelin altered the CE-QUAL-W2 program to output model predictions through space and time into a binary file. Blair Hanna, another contractor with the USGS, wrote a C++ program that pulled the desired data out of the binary file and computed metrics that were output into a text file. This text file was imported into a spreadsheet where algal production and fish growth calculations were made.

The next three chapters describe the methods used to develop and apply CE-QUAL-W2, the bioenergetics model and the food web-energy transfer model.

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Chaper 5. METHODS - CE-QUAL-W2

5.1 General model description

CE-QUAL-W2 is a two-dimensional hydrodynamic and water quality model that is developed and distributed by the U.S. Army Corps of Engineers (USACE). Its precursor was the laterally averaged reservoir model (LARM) developed by Buchak and Edinger in 1982. LARM evolved into the generalized longitudinal-vertical hydrodynamic and transport model that allowed modeling of multiple branches and estuarine boundary conditions. The USACE added water quality algorithms to the hydrodynamic model to create the current CE-QUAL-W2 model (Cole and Buchak 1995).

CE-QUAL-W2 models many processes internally, which increases model complexity because the model incorporates many interconnected equations. Computation time is also increased because of necessary complex solution schemes. However, because more of the physics of the system is modeled internally, the number of calibration parameters is greatly decreased.

The CE-QUAL-W2 model includes a number of special features (see Cole and Buchak (1995) for more detailed discussion of these features):

- An algorithm that adjusts the timestep to ensure hydrodynamic stability
- A selective withdrawal algorithm
- A higher-order transport scheme to reduce numerical diffusion

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- Time-weighted vertical advection and fully implicit vertical diffusion
- Ability to fill in gaps of input data using a step function or linear interpolation
- An ice-cover algorithm
- Internal calculation of terms for surface heat exchange modeling
- Allows variable layer heights and segment lengths
- Surface layer effects can extend through multiple layers
- Input data does not need to be input at constant intervals (i.e., can be accepted at any time frequency)
- Includes sediment/water heat exchange modeling

The model uses lateral averaging, which means that the reservoir is simplified into two dimensions (i.e., length and depth), as shown in Figure 11. It is assumed that conditions are uniform across the width of each segment. Vertical profiles of modeled parameters are predicted for each segment (Cole and Buchak 1995).

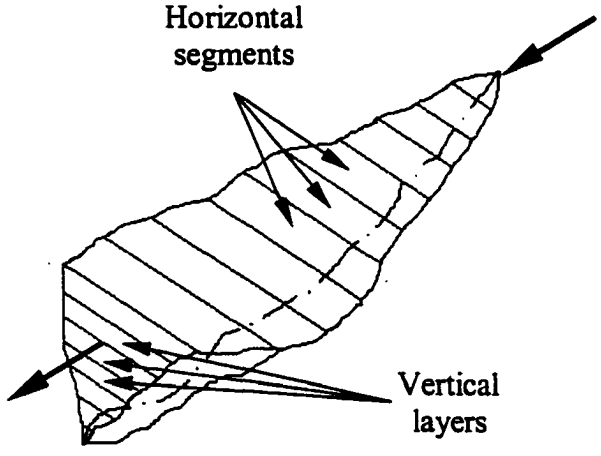

Figure 11. Two-dimensional segment and layer configuration of CE-QUAL-W2

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The hydrodynamics modeling approach is used to determine water surface elevations, horizontal and vertical velocities, water temperature, water density, vertical eddy viscosities and diffusivities, and constituent transport. The model employs the principles of conservation of momentum, conservation of mass, and conservation of energy in six fundamental equations with six unknown variables (Cole and Buchak 1995):

- two conservation of momentum equations
  - horizontal momentum
  - vertical momentum due to hydrostatic pressure

- three conservation of mass equations
  - vertically integrated continuity to determine the free water surface elevation
  - internal continuity of flow
  - constituent transport

- one conservation of energy equation
  - equation of state to relate water density to temperature, total dissolved solids (TDS), and total suspended solids (TSS)

The six unknown parameters in these equations are horizontal velocity, vertical velocity, hydrostatic pressure, free water surface elevation, constituent concentration, and water density. In solving the six equations for these six unknowns, the solution scheme

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uses two key assumptions: 1) that each cell in the model is homogeneous (i.e., two-dimensional); and 2) hydrostatic pressure is a function of depth only (i.e., water density differences in the water column are negligible).

The numerical solution scheme used by CE-QUAL-W2 is the finite difference method with a space-staggered grid. Model variables are defined either at cell boundaries or at the cell center. Mixed implicit-explicit time differencing is used. At each time step, the following solution strategy is used to solve the hydrodynamic equations (Cole and Buchak 1995; USACE 1995b):

- At the beginning of the new time step, the water densities are computed from the equation of state using the water temperature, TDS, or TSS values computed at the previous time step

- Using these new water densities and the last update for water velocities, water surface locations are computed implicitly using a centered difference scheme from the water surface equation

- Using these new water surface elevations, new horizontal velocities are computed explicitly using backward and central difference schemes from the horizontal momentum equation

- Using these new horizontal velocities, new vertical velocities are computed from the internal continuity equation

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- Using the new horizontal and vertical velocities, constituent concentrations are calculated from the transport equation

The model then proceeds to the next time step and repeats the solution process.

The surface water temperatures are calculated in the surface layer cells using one of two options for modeling the heat budget in CE-QUAL-W2: 1) term-by-term heat balance, or 2) equilibrium temperature approach. These calculations are made using previous timestep data. The term-by-term heat balance is calculated according to the following equation (Cole and Buchak 1995; USACE 1995b):

$$H_n = H_s + H_a + H_e + H_c - (H_{sr} + H_{ar} + H_{br}) \qquad \text{Equation 12}$$

where

H_n = net rate of heat exchange across the water surface (W m^{-2})

H_s = incident short wave solar radiation (W m^{-2})

H_a = incident long wave radiation (W m^{-2})

H_{sr} = reflected short wave solar radiation (W m^{-2})

H_{ar} = reflected long wave radiation (W m^{-2})

H_{br} = back radiation from the water surface (see Equation 13) (W m^{-2})

H_e = evaporative heat loss (see Equation 14) (W m^{-2})

H_c = heat conduction (see Equation 16) (W m^{-2})

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If the short wave solar radiation (H_s) is not measured, it is calculated from sun angle relationships (i.e., latitude, time of day, and time of year), air conditions (i.e., altitude, humidity, and dust), and cloud cover. The long wave atmospheric radiation (H_a) is calculated from air temperature and cloud cover or air vapor pressure using Brunts formula. The reflected short wave radiation (H_{sr}) is calculated from the sun altitude angle and cloud cover, while the reflected long wave radiation (H_{ar}) is 3% of H_a (Cole and Buchak 1995; USACE 1995b).

The following equation is used to calculate the water surface back radiation (H_{br}) (Cole and Buchak 1995):

$$H_{br} = \varepsilon \sigma^{*} (T_s + 273.15)^4 \qquad \text{Equation 13}$$

where

ε = emissivity of water = 0.97

σ^{*} = Stephan-Boltzman constant = 5.67×10^{-8} W m^{-2} $°K^{-4}$

T_s = water surface temperature (°C)

The evaporative heat loss (H_e) is dependent on air temperature, dewpoint temperature, and wind speed according to the following equation (Cole and Buchak 1995; USACE 1995b):

$$H_e = f(W)(e_s - e_a) \qquad \text{Equation 14}$$

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where

$f(W)$ = evaporative wind speed function (see Equation 15) (W m^{-2} mm Hg^{-1})

e_s = saturation vapor pressure at the water surface due to water surface temperature (mm Hg)

e_a = atmospheric vapor pressure due to dewpoint temperature (mm Hg)

The evaporative wind speed function is an empirical relationship determined from investigating waterbodies of different sizes, shapes and locations (Cole and Buchak 1995; USACE 1995b):

$$f(W) = 9.2 + 0.46W^2 \qquad \text{Equation 15}$$

where

W = wind speed (m s^{-1})

Surface heat conduction is a function of the evaporative wind speed function, water surface temperature, and air temperature (Cole and Buchak 1995):

$$H_c = C_c f(W)(T_s - T_a) \qquad \text{Equation 16}$$

where

C_c = Bowen's coefficient = 0.47 mm Hg $°C^{-1}$

T_a = air temperature (°C)

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The alternative equilibrium temperature approach for computing the surface heat exchange involves finding an equilibrium temperature (T_e) such that the net rate of surface heat exchange (H_n) is zero (Cole and Buchak 1995):

$$H_n = -K_{aw}(T_s - T_e) \qquad \text{Equation 17}$$

where

K_{aw} = coefficient of surface heat exchange (see Equation 18) (W m^{-2} $°C^{-1}$)

The coefficient of surface heat exchange (K_{aw}) is obtained by taking the derivative of H_n (from Equation 12) with respect to surface water temperature (Cole and Buchak 1995):

$$K_{aw} = \frac{dH_n}{dT_s} \qquad \text{Equation 18}$$

where the terms in Equation 12 are estimated using linear approximations with the equilibrium temperature (T_e) in place of surface water temperature (T_s).

All of the heat exchange terms in both approaches only enter and exit surface layer cells, with the exception of short wave solar radiation, which penetrates the surface and decays exponentially with depth according to Beers Law (Cole and Buchak 1995):

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$$H_s(z) = (1-\beta)H_s \exp^{-\eta z}$$ Equation 19

where

$H_s(z)$ = short wave radiation at depth z (W m^{-2})

β = fraction absorbed at the water surface

η = extinction coefficient (m^{-1})

CE-QUAL-W2 includes water quality algorithms for 21 constituents in addition to temperature (Cole and Buchak 1995). While the algorithms are separate from the hydrodynamics equations, the water quality kinetics are tied internally to the hydrodynamics because at each time step, the hydrodynamics move constituents into and out of model cells, where the kinetic equations are employed (Cole and Buchak 1995; USACE 1995b).

The water quality modeling in CE-QUAL-W2 generally uses a mass balance approach (Chapra 1997; Cole and Buchak 1995):

$$\frac{\partial c}{\partial t} = SS_B + SS_K$$ Equation 20

where

c = constituent concentration (g m^{-3})

SS_B = boundary sources and sinks for constituent

SS_K = internal sources and sinks due to kinetic reactions

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For example, the mass balance for algae in CE-QUAL-W2 is as follows (Cole and Buchak 1995):

$$\frac{\partial \Phi_a}{\partial t} = K_{ag}\Phi_a - K_{ar}\Phi_a - K_{ae}\Phi_a - K_{am}\Phi_a - \frac{\omega_a}{\Delta z}\Phi_a \qquad \text{Equation 21}$$

where

Φ_a = algal concentration (g m^{-3})

K_{ag} = algal growth rate (s^{-1})

K_{ar} = algal dark respiration rate (s^{-1})

K_{ae} = algal excretion rate (s^{-1})

K_{am} = algal mortality rate (s^{-1})

ω_a = algal settling rate (m s^{-1})

Δz = cell thickness (m)

In order to solve the equations, boundary conditions must be specified (Table 7). Some are input by the model user, while others are calculated by the model (Cole and Buchak 1995).

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Table 7. Summary of boundary conditions for CE-QUAL-W2

Boundary condition	Calculated by model (CALC) or input by user (INPUT)	User input data required for calculation
Inflow quantity data	INPUT	None
Inflow quality data	INPUT	None
Outflow quantity data	INPUT	None
Upstream or downstream heads at ending model segments	INPUT	None
Upstream or downstream heads where branches meet in model	CALC	None
Surface heat exchange	CALC	Meteorological input data, site location, time
Solar radiation	CALC	Meteorological input data, site location, time
Wind stress	CALC	Meteorological input data, wind sheltering coefficient
Gas exchange	CALC	Meteorological input data, wind sheltering coefficient
Evaporation	CALC	Meteorological input data, wind sheltering coefficient
Precipitation quantity data	INPUT	None
Precipitation quality data	INPUT	None
Shear stress	CALC	Chezy coefficient
Heat exchange	CALC	Bottom exchange coefficient, sediment temperature
Sediment water quality fluxes	INPUT or CALC	None

Cole and Buchak (1995)

5.2 CE-QUAL-W2 model setup and water temperature calibration

The CE-QUAL-W2 model of Shasta Lake was first developed and calibrated for simulating water temperatures, and then later calibrated for other modeled constituents. This procedure was used because CE-QUAL-W2 internally models the hydrodynamic processes, so very few coefficients must be calibrated for water temperature. The process of water temperature calibration involved the investigation and adjustment of these coefficients, and the methods for estimating input data. The best available data was used

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to input into the model and represent Shasta Lake, but available data was not always as complete or detailed as desired.

The Shasta Lake CE-QUAL-W2 model was initially configured by Michael Sheetz, a contractor with the USGS. The work performed for this dissertation involved modifying the model from its initial configuration, calibrating it, and applying it. The model was refined during the water temperature calibration process by investigating five general categories of data: bathymetry, inflow distribution, inflow temperatures, outflow distribution, and meteorology. Water temperatures measured in 1995 by the USGS at several sampling stations throughout the reservoir (Table 8) were compared with model-predicted water temperatures for various runs to evaluate the goodness-of-fit and sensitivity of the model to different parameters. The water temperatures were collected using a Hydrolab Surveyor 3® connected to a H2O® sonde unit. Measurements were made at depth intervals of 1 m through the epilimnion, and then every 2 to 5 m to the bottom of the reservoir (Lieberman and Horn 1998).

5.2.1 Bathymetry

Measurements of bathymetric characteristics were made using USGS topography maps that were completed in 1934 before the reservoir was filled. The calibration of reservoir bathymetric representation resulted in the application of varying layer depths and revised bottom layers. Thus, the CE-QUAL-W2 model representation of Shasta Lake (see Figure 12), which includes the main reservoir area and five branches (the Backbone Creek Inlet, Sacramento River arm, McCloud River arm, Squaw Creek, and the Pit River arm), was as follows:

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Table 8. Locations and dates of USGS water temperature sampling in 1995

USGS station[a]	Location	Model segment[a]	Sampling day of month						
			May	Jun	Jul	Aug	Sep	Oct	Nov
S6	Backbone Ck. confl.	19	nm[b]	20	27	31	22	19	14
S7	Sacramento River	54	11	20	27	31	22	19	14
S8	McCloud River	41	11	22	26	30	22	19	14
S9	Pit River	13	11	20	26	30	22	19	14
S10	McCloud R. confl.	16	11	21	26	30	22	19	14

[a] Refer to Figure 12 for locations
[b] nm = no measurements

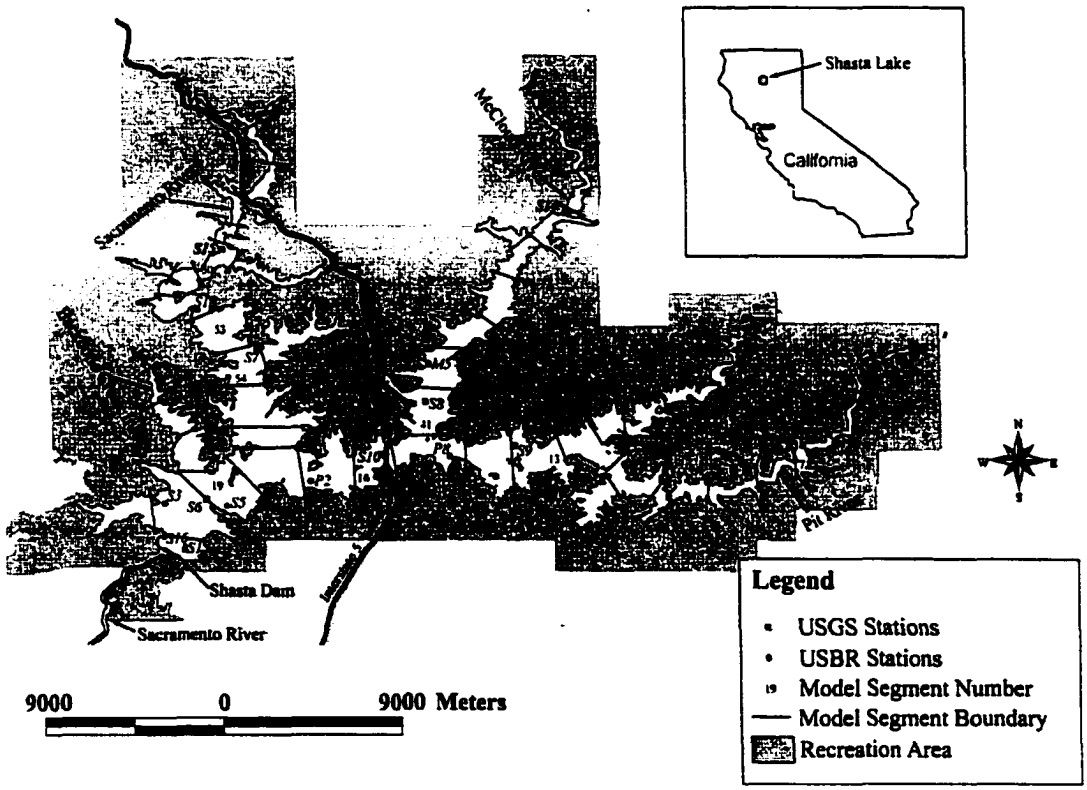

Figure 12. CE-QUAL-W2 representation of Shasta Lake

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- 63 segments approximately 2 km in length
- up to 51 vertical layers with layer depths that vary as follows:

  Elevation 295.2 – 325.2 m: 1.5 m layers (21 layers)

  Elevation 238.2 – 292.2 m: 3.0 m layers (19 layers)

  Elevation 172.2 – 232.2 m: 6.0 m layers (11 layers)

Adjustment of the Place QIN Control (PQC) switch, which determines how the inflows are distributed by CE-QUAL-W2 amongst the layers in the upstream reach of each branch, was also made during the calibration of bathymetric characteristics. If the PQC switch is OFF, inflows are distributed evenly amongst all the layers. If the PQC switch is ON, the inflows are distributed to the layer(s) whose density most closely matches the inflow density. During calibration, the PQC switch was set to OFF because some of the runs with the varying layer depths would not run with the PQC switch set to ON due to model instability. Sensitivity runs were done in which water temperatures and distribution of inflows were compared with the switch ON and OFF. Water temperature comparisons were made with constant layer depth runs because CE-QUAL-W2 would not complete its runs under certain scenarios with varying layer depths. Differences in model-predicted water temperatures were usually less than 0.5°C. The depths over which the inflows were distributed with the switch ON were very similar to the depths of distribution with varying layer depths, but with the switch OFF. Because model results were so much better with varying layer depths (see Table 11), it was concluded that further model runs would incorporate varying layer depths, but with the PQC switch off.

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5.2.2 Inflow distribution

Although the model has five branches, historical inflow data was only available for the four main branches. 1995 inflow data provided by the USGS was used for the Pit, McCloud and Sacramento Rivers, but Squaw Creek inflows were not available for 1995. A comparison of various regression relationships between historical Squaw Creek inflows and corresponding Pit, McCloud and Sacramento River inflows indicated that the McCloud River had the best correlation with Squaw Creek inflows. Thus, 1995 Squaw Creek inflows were calculated using the following regression relationship with the McCloud River that was developed from historical data ($r^2 = 0.83$):

$$Q'_{Sq} = -346.474 + 0.338274(Q_M) \qquad \text{Equation 22}$$

where

Q'_{Sq} = estimated Squaw Creek inflow (cfs)

Q_M = measured McCloud River inflow (cfs)

Because of the large negative intercept, a minimum flow value of 0.42 m^3 per second (cms) (15 cubic feet per second (cfs)) was assumed based on the average of historical minimum annual flow values in Squaw Creek for calendar years 1946 through 1963.

The USBR also provided daily computed inflow data at Shasta Dam, which should be comparable to the sum of the USGS daily flows of the four branches. Differences between the two sources of data were primarily due to precipitation and

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evaporation since the four branches were by far the largest rivers in the watershed above Shasta Dam, and the USGS gages were located several km upstream of the dam. In order to balance the inflows from the rivers with the USBR computed inflows, the flow differences were distributed to the four branches according to the average fraction of the total flow of the four branches that each had carried over the historical period of record (i.e., January 1946 to December 1963).

5.2.3 Inflow temperatures

Water temperature data used in the model was obtained for the Pit, McCloud and Sacramento Rivers from the California Data Exchange Center's website. The hourly data was converted to degrees Celsius and averaged for each day. Gaps in the available data for these rivers were filled by using a regression relationship between flow and water temperature (Bartholow 1989):

$$T' = A_1 + A_2 \ln(Q) + A_3 \sin(JD) + A_4 \cos(JD) \qquad \text{Equation 23}$$

where

T'	=	estimated temperature (°C)
Q	=	flow (cfs)
JD	=	Julian date converted to radians
A_1, A_2, A_3, A_4	=	coefficients to fit the data

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Some manual adjustments were made to fit the regressed data to the gaps in the actual data. Since water temperatures were not available for Squaw Creek, its water temperatures were assumed to be the same as those for the McCloud River. Flow in Squaw Creek had been correlated with the McCloud River, and the drainage basins have similar land use and cover.

5.2.4 Outflow distribution

For 1995 operations, the USBR provided daily operation records for three types of releases from Shasta Dam: power, spill, and outlet. Power releases were those that went through the penstocks for power generation. Spill releases were those that went over the crest of the dam, and there were no such releases in 1995. Outlet releases in 1995 went through the bypass pipes that are located at three levels. The actual distribution of the outlet releases by elevation was unknown for 1995, but USBR personnel provided seasonal operating priorities that were used to estimate the distributions (Table 9) (Fujitani 1997).

Jeff Sandelin wrote a program using Delphi/C++ to facilitate the generation of the CE-QUAL-W2 outflow file with the appropriate distributions of the bypass releases. The program reads daily release data from a text file, performs any unit conversions necessary, and then distributes bypass releases according to outlet capacities and seasonal operating rules and priorities. The locations and physical capacities of the outlets are defined by the user, as are the seasons and their corresponding operating rules. The user can also specify a file that contains daily water surface elevations to ensure that an outlet is submerged by at least 9.1 m (30 feet) before releases are made from that outlet.

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Table 9. 1995 bypass release operations at Shasta Dam

Season	Description of releases
January 1-31	All bypass releases through the 257 m (842 feet) outlets
February 1-28	Bypass releases first through the 257 m (842 feet) outlets, then the 226 m (742 feet) outlets, and lastly through the 275 m (942 feet) outlets
March 1-31	230.3 cms (8132 cfs) of the bypass releases through the 226 m (742 feet) outlets, 69.1 cms (2440 cfs) through the 287 m (942 feet) outlets, and the remainder through the 257 m (842 feet) outlets
April 1-May 12	All bypass releases through the 287 m (942 feet) outlets
May 13-31	233.7 cms (8252 cfs) of the bypass releases through the 226 m (742 feet) outlets, and the remainder through the 257 m (842 feet) outlets
June 1-December 31	All bypass releases through the 226 m (742 feet) outlets

Source: Fujitani (1997)

5.2.5 Meteorology

The meteorological input file required air temperature, dewpoint temperature, wind speed, wind direction, and cloud cover data. There was no meteorological data available at Shasta Lake until March 1996, when a meteorological station located at Shasta Dam began collecting data. For 1995 and the first few months of 1996, the closest available meteorological data was from a station located at the Redding airport

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approximately 15 km to the south of Shasta Dam, and at about 200 m less in elevation. Therefore, all data for the 1995 meteorological input file was developed in some manner from 1995 meteorological data at the Redding airport. Relationships were explored between the available 1996-97 Shasta data and Redding data so that the 1995 Redding data could be used to generate representative 1995 hourly-averaged Shasta data. These relationships included linear and higher-order regressions, multiple regressions, fuzzy rule-based modeling, and relationships published in literature. The fuzzy analysis methods performed about the same as the regression analyses. Because the regression techniques are built into most spreadsheet software, the level of effort needed to implement fuzzy modeling was much greater, so it was not pursued. Thus, the methods employed to transfer the Redding meteorological data to Shasta Lake were either relationships published in literature or regression relationships.

Air temperatures at Shasta Lake were computed by adjusting air temperatures at Redding for elevation using the dry adiabatic lapse rate:

$$T_S' = T_R - 1.71°C \qquad \text{Equation 24}$$

where

T'_S = estimated air temperature at Shasta (°C)

T_R = measured air temperature at Redding (°C)

1.71°C = adjustment based on assumption that the dry adiabatic lapse rate is about 9.76°C per 1000 m (Ahrens 1991; Morel-Seytoux 1990)

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Dewpoint temperatures were calculated by first determining relative humidity at Redding from measured dewpoint and air temperatures (Equation 25) (Bartholow 1989). The relative humidity at Redding was then adjusted to relative humidity at Shasta using a correction for the change in elevation between the two locations that was based on the ideal gas law (Equation 26) (Theurer et al. 1984). The relative humidity at Shasta was then converted to dewpoint temperature at Shasta (Equation 27) (Bartholow 1989):

$$RH'_R = \left[\frac{(112 - 0.1T_R + T_{dpR})}{(112 + 0.9T_R)}\right]^8 (100\%) \qquad \text{Equation 25}$$

$$RH'_S = RH'_R\left[(1.064)^{(T_R - T'_S)}\frac{(T'_S + 273.16)}{(T_R + 273.16)}\right] \qquad \text{Equation 26}$$

$$T'_{dpS} = \left[(112 + 0.9T'_S)\left(\frac{RH'_S}{100}\right)^{1.8}\right] - 112 + 0.1T'_S \qquad \text{Equation 27}$$

where

RH'_R = estimated relative humidity at Redding (%)

RH'_S = estimated relative humidity at Shasta (%)

T_R = air temperature at Redding (°C)

T'_S = estimated air temperature at Shasta (°C)

T_{dpR} = dewpoint temperature at Redding (°C)

T'_{dpS} = estimated dewpoint temperature at Shasta (°C)

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Wind speeds were calculated using a multiple regression equation with the Redding wind speeds:

$$u'_{wS} = 1.286497 + 0.253806(u_{wR}) - 0.00241(u_{wR})^2 - 0.0716\sin(JD) - 0.04722\cos(JD) \quad \text{Equation 28}$$

where

u'_{wS} = estimated wind speed at Shasta (m s^{-1})

u_{wR} = wind speed at Redding (m s^{-1})

JD = Julian date converted to radians

Wind direction at Redding was used at Shasta Lake without adjustment because the r^2 values for the wind direction data were extremely low regardless of the technique used. Cloud cover data at Redding was also used at Shasta Lake without adjustment because cloud cover data was not available at Shasta Lake to set up any relationships.

The other meteorological parameter investigated was the wind sheltering coefficient (WSC). This coefficient is used in CE-QUAL-W2 to account for the effect of surrounding terrain on sheltering a water body from winds observed at a meteorological station that may be located at a distance from the water body. A WSC of 1.0 indicates that the measured wind speeds were not sheltered by the terrain adjacent to the water body (Cole and Buchak 1995). Since the multiple regression of the Redding wind speeds effectively located the winds at Shasta Lake, the use of a WSC of 1.0 was reasonable.

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5.2.6 Results of water temperature calibration

The following changes were made during the calibration of the water temperature predictions from CE-QUAL-W2:

- New bathymetry = varying layer depths; PQC switch off; new method of calculating bottom layers
- New inflow distribution = new USGS flows for Pit and McCloud Rivers; new regression for Squaw Creek flows with minimum flow of 0.42 cms; new fractions to distribute flow discrepancies
- New inflow temperatures = new calculated inflow temperatures due to new inflow distribution where there were gaps in available inflow temperature data
- New meteorology = new air temperatures due to change in reduction factor for dry adiabatic lapse rate; new dewpoint temperatures because of new air temperatures; new wind speeds calculated using multiple regression techniques; WSC changed to 1.0

Table 10 summarizes the input data for runs that were made to represent the revised model input files after each category of data was calibrated.

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Table 10. Description of calibration changes of key runs

Run name	Bathymetry	Inflow distribution	Inflow temperatures	Outflow distribution	Meteorology
Oldbath	Old	Old	Old	Old	Old
Newbath	New	Old	Old	Old	Old
Newflows	New	New	Old	Old	Old
Newtemps	New	New	New	Old	Old
Newout	New	New	New	New	Old
Newcalib	New	New	New	New	New

To evaluate the fit of the model-predicted water temperatures with corresponding measured temperatures, the following statistics were calculated:

$$r^2 = \left[\frac{n\left(\sum_{i=1}^{n} T_i T_i'\right) - \left(\sum_{i=1}^{n} T_i\right)\left(\sum_{i=1}^{n} T_i'\right)}{\sqrt{\left[n\sum_{i=1}^{n} T_i^2 - \left(\sum_{i=1}^{n} T_i\right)^2\right]\left[n\sum_{i=1}^{n} T_i'^2 - \left(\sum_{i=1}^{n} T_i'\right)^2\right]}} \right]^2 \qquad \text{Equation 29}$$

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n} (T_i - T_i')^2}{n}} \qquad \text{Equation 30}$$

$$ABSE = \frac{\sum_{i=1}^{n} |T_i - T_i'|}{n} \qquad \text{Equation 31}$$

where

r = correlation coefficient

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$RMSE$ = root mean squared error (°C)

$ABSE$ = absolute mean error (°C)

T_i = measured water temperature for observation i (°C)

T'_i = predicted water temperature for observation i (°C)

n = number of observations

An r^2 value of 1.0 indicates a perfect fit, while an r^2 of 0.0 indicates no fit at all. Better fits result in smaller values of *RMSE* and *ABSE*. The statistics for the calibration runs in Table 10 indicated that run *Newcalib* provided the best fit of the model-predicted water temperatures with measured temperatures (Table 11 and Appendix A). Changes to bathymetry and meteorological input data resulted in the greatest improvement to the model-predicted water temperature profiles.

Table 11. Goodness-of-fit statistics for water temperature calibration runs

Parameter	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
r^2	0.948	0.969	0.969	0.969	0.968	0.981
RMSE (°C)	1.25	0.84	0.83	0.83	0.81	0.68
ABSE (°C)	0.96	0.64	0.64	0.64	0.62	0.54

5.3 Water temperature validation and sensitivity analyses

Validation runs for CE-QUAL-W2 water temperature predictions were made using input files for calendar years 1968, 1974, and 1976 (Table 12). For these runs, the

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only available measured water temperatures were at Shasta Dam, so the model predictions at Segment 21 (the first model segment upstream of the dam) were compared with measured water temperatures for 1968, 1974, 1976 and 1995. Generally, the measurements were made on a biweekly basis at uniform 7.6-m (25-foot) intervals, ranging from an elevation of 190.5 m (625 feet) to the water surface.

Table 12. Total volume of Shasta Lake inflow for historical years

	1968	1974	1976	1995
Total inflow (10^6 m^3)	174.6	302.3	118.5	348.6
% of 1995 inflow	50%	87%	34%	100%
Hydrologic classification[a]	Below normal	Wet	Critically dry	Wet

[a] Source: USBR(1990)

Initial runs were made for each of the years with model parameters set as they had been for the calibrated 1995 model run. Output from these runs was compared with the measured stage elevations and water temperatures at the dam.

All of the comparisons showed an increasing difference between the measured and modeled water surface elevations through the year (Table 13), with modeled water surface elevations over a meter higher than the measured elevations by the end of the year. The maximum discrepancy for all years was less than 0.5% of the reservoir volume. Because the difference was generally cumulative over time, it could have been due to evaporation and/or precipitation, both of which were assumed to be included in the

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inflows input into the model. Another cause of the discrepancy was that the measured elevations were only collected at the dam, while the model accounted for differences in elevation throughout the reservoir due to inflows, momentum, and seiche (Winter 1981). In addition, the translation of the elevations to reservoir storage was made using an empirically-derived storage-elevation curve. However, change in reservoir storage with elevation would not realistically correspond exactly with an empirical equation, and the deviation of reservoir bathymetry from the equation would vary with elevation.

Table 13. Summary of differences between measured and modeled water surface elevations

Year	Maximum difference (measured – predicted)	
	(m)	% of total volume
1968	-1.233	0.412
1974	-1.215	0.389
1976	-1.262	0.453
1995	-1.044	0.336

Comparisons of water temperatures from the initial runs were not as impressive for 1968, 1974 and 1976 as they were for 1995 (Table 14 and Appendix B). 1995 statistics shown in Table 11 are different than those shown in Table 14 because the validation statistics are comparing the measured values at Shasta Dam only. In 1968, 1974 and 1995, modeled temperatures in the hypolimnion were too high, especially later in the year. This phenomenon was not seen in 1976. Also, the very bottom layer in

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almost all of the 1976 profiles, the first profile of 1974, and the May and June profiles of 1995 was much colder than the layer above it, so that the profile appeared to have a 'tail' at the very bottom. In addition, modeled surface temperatures were too cold throughout the year in 1968.

Table 14. Results of initial water temperature validation runs

Run year	Maximum difference (measured – predicted) (°C)	RMSE (°C)	ABSE (°C)	r^2
1968	5.043	1.50	1.19	0.891
1974	-4.013	1.36	1.10	0.920
1976	5.854	1.35	0.96	0.938
1995	2.791	0.86	0.63	0.941

Additional runs were made to test the sensitivity of the model to changes in the following parameters:

- *Coefficient of bottom heat exchange (CBHE) and ground temperature (TSED)*: These parameters are used to compute the heat exchange at the ground-water interface. The recommended value for the CBHE is 7.0E-8 W m^{-2} s^{-1}, while the ground temperature is estimated from the average annual air temperature (Cole and Buchak 1995). All of the initial runs used a CBHE value of 7.0E-8 W m^{-2} s^{-1} and a TSED value of 15.9°C.

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- *Light extinction for water exclusive of suspended solids (EXH2O) and the fraction of incident solar radiation absorbed at the water surface (BETA)*: These parameters affect the amount of heat absorbed in the surface layer. Cole and Buchak (1995) describe the relationship between EXH2O and the net extinction coefficient (γ), and the corresponding relationship between γ and BETA. The values used in the initial runs were EXH2O = 0.40 m^{-1} and BETA = 0.45. However, because inorganic suspended solids and organic suspended solids were not simulated in any of these runs, these values did not satisfy the relationships presented in Cole and Buchak (1995).

- *Inflow temperatures*: Measured temperatures were only available for the Sacramento River in 1968, 1974 and 1976, so water temperatures for the Pit River and McCloud River were generated using relationships with the Sacramento River water temperatures. This procedure was different than that used for the 1995 input data.

- *Meteorological data*: The closest available meteorological data for 1968, 1974 and 1976 was from Red Bluff, which is 50 km south of Redding and at an elevation of 106.4 m, while the Redding meteorological station from which the 1995 data was extrapolated is about 15 km south of Shasta Dam at an elevation of 153.0 m. The same procedures were used to transfer the Red Bluff air and dewpoint temperatures to Shasta Dam as were used to transfer the Redding data, but Red Bluff wind speeds were used directly at Shasta, while a multiple regression was used to transfer Redding wind speeds to Shasta.

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A total of 44 runs were made (Appendix B) to investigate the sensitivity of the model to these parameters, with the best summary statistics for each year found under the conditions shown in Table 15. The 'best' summary statistics for each year occurred under different conditions. Also, none of the runs in 1968, 1974 or 1976 approximated the measured water temperatures as closely as the 1995 calibrated run according to the summary statistics.

Table 15. Summary of best overall summary statistics for 1968, 1974, 1976 and 1995 runs

Year	Run	RMSE (°C)	ABSE (°C)	r^2	Condition(s)[a]
1968	1968e	1.339	1.055	0.9137	Wind speed – 0.5 m s^{-1}
1974	1974b	1.105	0.896	0.9501	Wind speed – 0.5 m s^{-1}
1976	1976k	1.140	0.821	0.9545	EXH2O = 1.5 m^{-1}; BETA = 0.72
1995	1995c	0.810	0.589	0.9466	TSED = 10°C; CBHE = 8.0E-8 W m^{-2} s^{-1}

[a] All values were as set for initial conditions (i.e., EXH2O = 0.4 m^{-1}; BETA = 0.45; TSED = 15.9°C; CBHE = 7.0E-8 W m^{-2} s^{-1}) unless otherwise indicated in 'Condition(s)' column

In addition to adjusting the parameters, Cole (1998) suggested that the 'tail effect' observed in many of the water temperature profiles could be due to Segment 21 being the most downstream segment, and the adjacent upstream segment (Segment 20) being one layer shallower. Therefore, there were no horizontal velocities through the bottom layer in Segment 21, and the only way in which water could move in and out of the bottom layer was through diffusion with the layer above it. If there were little mixing occurring

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in the reservoir, the water and corresponding water temperature in the bottom layer of Segment 21 would tend to remain the same until mixing occurred. Cole (1998) further suggested that the effect could be eliminated by adding another layer to the bottom of Segment 20, thereby allowing horizontal velocities in the bottom layer of Segment 21. This layer was added and the model was rerun for each of the years. The tail effect was eliminated and the additional layer had virtually no effect on the modeled water surface elevations. Summary statistics of water temperature comparisons were slightly worse with the extra layer.

5.4 Conclusions of water temperature calibration, validation and sensitivity analyses

The results of the water temperature calibration and validation indicated that the performance of the model was related to the quality of the input data. More actual measured data was available in 1995 than in the other years, and the summary statistics were correspondingly better.

Predictions of water temperatures were most sensitive to changes in the bathymetry, WSC, and to a lesser degree, air temperature, cloud cover and wind speed. The use of variable layer depths greatly improved the water temperature predictions throughout the year, and the addition of an extra layer to the bottom of Segment 20 removed the 'tail effect' observed during the water temperature validation runs. The greatest effect of changing the wind sheltering coefficient occurred at or near the water surface, with lower coefficients resulting in higher estimated water temperatures at the surface. The highest sensitivity to air temperatures also occurred at or near the water surface, although the effect was not consistent with change in air temperature, which

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indicates that the interaction of air temperature with other model components such as wind speed and wind direction may have resulted in greater effects on the water temperature profile. Modeled water temperatures decreased as cloud cover increased, and uniformly reducing the wind speed by 0.5 m s^{-1} slightly improved model results for all years.

A run made with inflow temperatures reduced by 1°C for all of 1995 indicated that water temperatures in the upper layers were virtually unchanged, while predicted temperatures were reduced by more than 0.5°C at about 8-11 m below the water surface. Temperatures predicted by this run were generally less than 1°C lower than those predicted by the run without the temperature reduction.

Although the cause of the discrepancy in modeled water surface elevations with measured water surface elevations was not corrected, it was concluded that the discrepancy could be due to model representation of the reservoir that was more realistic than the stage measurements at the dam and the empirically-derived storage-elevation curve. In addition, the primary application of the model was to evaluate changes in reservoir operations, so that the same discrepancy would be present in comparison runs. Because the discrepancy was present in the calibration year (1995) as well as the other years, it undoubtedly was part of the overall error demonstrated by the model, but this error was relatively small because model-predicted temperatures compared very well with measured water temperatures over time and space. Other applications of CE-QUAL-W2 have observed comparable performance of the model in predicting water temperatures (Garvey et al. 1998; Giorgino and Bales 1997; Martin 1998; Rounds et al. 1998). The final calibrated parameters are shown in Table 16.

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Table 16. Calibrated values of the wind sheltering coefficient (WSC), coefficient of bottom heat exchange (CBHE), temperature of the sediments (TSED), and light extinction coefficients (EXH2O and BETA) for the Shasta Lake CE-QUAL-W2 model

WSC	CBHE (m^{-2} s^{-1})	TSED (°C)	EXH2O (m^{-1})	BETA
1.0	7.0E-8	15.9	0.40	0.45

5.5 CE-QUAL-W2 water quality and phytoplankton calibration

The Shasta Lake CE-QUAL-W2 model included the following constituents: total dissolved solids, labile dissolved organic matter (DOM), refractory DOM, one compartment of algae, detritus (or labile particulate organic matter (POM)), phosphate, ammonium, nitrate/nitrite, dissolved oxygen, sediment, and total inorganic carbon. Blair Hanna performed the calibration of the model for these parameters by comparing model predictions to measured data for calendar years 1995-96 (Bartholow et al. in review). Water temperature and dissolved oxygen data were collected at depth intervals of 1 m through the thermocline, and then every 2 to 5 m to the bottom of the reservoir. Chlorophyll *a* data were collected at the water surface and at depth intervals of 5 m to a depth of 30 m. Nutrient samples were collected at the surface, and in the metalimnion and the hypolimnion (Lieberman and Horn 1998).

Details for the construction of the water quality input files and the calibration of the CE-QUAL-W2 water quality model are provided in Bartholow et al. (in review). The set of model parameters that simultaneously predicted the best fit of the trends in the measured data for all of the constituents was chosen as the calibrated model (Table 17).

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Table 17. Nutrient and phytoplankton calibration parameters for the Shasta Lake CE-QUAL-W2 model

CE-QUAL-W2 parameter	Parameter description	Units	Calibrated value
Temperature			
WSC	Wind sheltering coefficient	--	1.0
CBHE	Coefficient of bottom heat exchange	$m^{-2}\ s^{-1}$	7.0E-8
TSED	Temperature of sediments	°C	15.9
Light extinction			
EXH2O[a]	Light extinction in water exclusive of suspended solids	m^{-1}	0.42
EXSS	Extinction due to inorganic suspended solids	$m^3\ m^{-1}g^{-1}$	0.00
EXOM	Extinction due to organic suspended solids	$m^3\ m^{-1}g^{-1}$	0.10
BETA[a]	Fraction of incident solar radiation absorbed at water surface	--	0.38
Algae			
AG	Growth rate	d^{-1}	3.0
AM	Mortality rate	d^{-1}	0.03
AE	Excretion rate	d^{-1}	0.04
AR	Dark respiration rate	d^{-1}	0.08
AS	Settling rate	$m\ d^{-1}$	0.20
ASAT	Saturation intensity for maximum growth	$W\ m^{-2}$	86
APOM	Fraction of algal biomass lost by mortality to POM	--	0.95
AT1	Lower temperature for algal growth	°C	0
AT2	Lower temperature for maximum algal growth	°C	15
AT3	Upper temperature for maximum algal growth	°C	25
AT4	Upper temperature for algal growth	°C	35
AK1,AK2, AK3, AK4	Fractions of algal growth at above 4 temperatures	--	0.1, 0.9, 0.9, 0.1
Dissolved organic matter (DOM)			
LDOMDK	Labile DOM decay rate	d^{-1}	0.30
LRDDK	Labile to refractory decay rate	d^{-1}	0.010
RDOMDK	Maximum refractory decay rate	d^{-1}	0.001
Particulate organic matter (POM)			
LPOMDK	POM decay rate	d^{-1}	0.10
POMS	Settling rate	$m\ d^{-1}$	0.30
Organic matter rate multipliers			
OMT1, OMT2	Decay rate adjustment temperature values	°C	5.0, 30.0
OMK1, OMK2	Decay rate fractions	--	0.1, 0.99
Sediment			
SDK	Sediment decay rate	d^{-1}	0.08

[a] EXH2O and BETA are different from Table 16 because suspended solids is being modeled

Source: Bartholow et al. (in review)

To evaluate goodness-of-fit, the reliability index (RI) was calculated in addition to ABSE and r^2. The RI increases as predictions diverge from observations, with an RI of 1.0 for a perfect fit (Leggett and Williams 1981):

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$$RI = \frac{1 + \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left[\frac{1-(y_i / x_i)}{1+(y_i / x_i)}\right]^2}}{1 - \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left[\frac{1-(y_i / x_i)}{1+(y_i / x_i)}\right]^2}} \qquad \text{Equation 32}$$

where

x_i = model prediction for observation i

y_i = measured value for observation i

n = number of observations

Table 18. Goodness-of-fit statistics for thermal and water quality constituents in Shasta Lake CE-QUAL-W2 model

Constituent	ABSE[a]	r^2	Reliability index (RI)	n
Water temperature (°C)	0.743	0.97	1.08	3059
Dissolved oxygen (g m^{-3})	0.692	0.75	1.18	3059
Nitrate-nitrite (g m^{-3})	0.029	0.38	3.24	222
Soluble reactive phosphorus (g m^{-3})	0.009	0.27	2.25	222
Ammonium (g m^{-3})	0.003	0.16	2.54	222
Phytoplankton (g m^{-3})	0.134	0.08	3.05	595

[a] Units of ABSE are units of constituent
Source: Bartholow et al. (in review)

The RI for all constituents in Table 18 were better than those reported by Martin (1988) in his application of CE-QUAL-W2 to DeGray Lake, a reservoir in Arkansas. In another comparison of the one-dimensional CE-QUAL-R1 with CE-QUAL-W2, the RI

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for dissolved oxygen predictions averaged 1.46 for 805 comparisons, and 1.53 for 497 comparisons, respectively (Martin and Wlosinski 1986). Thus, the performance of CE-QUAL-W2 was comparable to or better than previously reported applications of the model for water quality modeling.

A comparison of predicted water temperatures in 1995 between a run simulating water temperature only, and a run simulating the full suite of water quality constituents and phytoplankton concentrations was made. Overall, summary statistics calculated for both runs in comparison to measured water temperatures for the seven days in which measurements were made in 1995, and at the six locations sampled in the reservoir, were better for the run simulating all of the constituents (Appendix C). Thus, adding the nutrient and phytoplankton simulation enhanced the water temperature profile predictions in 1995.

5.6 Applications of CE-QUAL-W2 for Shasta Lake TCD assessment

Alternative hydrologic conditions were simulated using the calibrated water temperature model to assess the effects of TCD operation on thermal patterns in the reservoir. A total of five historical years were simulated, which were hydrologically characterized as wet (1995 and 1974), above normal (1975), below normal (1968), and critically dry (1976) (USBR 1990). 'Baseline' simulations incorporating historical release operations were compared with simulations of hypothetical TCD operations directed towards meeting release temperature targets for the propagation of downstream chinook salmon (Table 19). Multi-year runs that artificially simulated two identical hydrologic years were also performed to investigate if the thermal effects of TCD

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operations persist or were magnified from year to year. To determine if the downstream temperature targets could be met under different hydrologic year conditions, Blair Hanna developed a pseudo-optimization program that was used with CE-QUAL-W2. This program enabled the simulation of operating the TCD so that water from multiple TCD outlets were mixed, and the release elevations were changed on a daily basis. Previous with-TCD simulations involved simpler TCD operations in which releases were only made through one TCD outlet at a time, and were changed on a seasonal basis (Hanna et al. 1999).

Table 19. Release temperature targets used in the simulations of the Shasta Lake temperature control device

Month	Target temperature (°C)	Additional criteria
January	13.9	or as warm as possible
February	13.9	or as warm as possible
March	12.8	or as warm as possible
April	12.8	
May	8.3	
June	8.3	
July	8.3	
August	8.3	
September	10.0	or as cool as possible
October	11.1	or as cool as possible
November	10.0	or as cool as possible
December	10.0	or as cool as possible

Sources: Fujitani (1997); USBR (1990)

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To evaluate the effects of the TCD on water quality and algal patterns in the reservoir, Bartholow et al. (in review) used a multivariable testing (MVT) approach to compare the TCD effects against the effects of six other variables that modeled reservoir dynamics were sensitive to: beginning of year reservoir volume, inflows, outflows, nutrient inputs, air temperature, and cloud cover. The effects of each of these variables were evaluated using a suite of 50 metrics that addressed: thermodynamic attributes such as changes in timing, duration, and strength of stratification; chemical attributes such as dissolved oxygen and nutrient concentrations in the epilimnion, hypolimnion and outfall; and biological attributes such as the strength of algal blooms and phytoplankton biomass (Bartholow et al. in review).

CE-QUAL-W2 runs performed for the assessment of thermal effects on fish using the bioenergetics model simulated with- and without-TCD conditions for the historical years (i.e., 1968, 1974-76, and 1995). For the pair of runs for each historical year, all input variables and coefficients remained the same except for the distribution of releases from the dam. Runs without the TCD maximized hydropower generation releases for the 1968 and 1974-76 runs, and used the bypass releases according to Table 9 for the 1995 run (Figures 13 and 14, respectively). Simulations with the TCD used the simplified TCD operations with seasonal releases through one TCD outlet at a time (Figure 15).

For linkage of CE-QUAL-W2 generated phytoplankton production predictions with the food web-energy transfer model, three scenarios were run using CE-QUAL-W2 with and without the TCD operating: 1) a hydrologically wet scenario; 2) a hydrologically dry scenario; and 3) calendar year 1995 conditions. The hydrologically wet scenario corresponded with Run #1 in Bartholow et al. (in review) in which the initial

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water surface at the beginning of the year was high, inflows were high, and outflows were low. Similarly, the hydrologically dry scenario corresponded with Run #4 in Bartholow et al. (in review), with low initial water surface and inflows, and high outflows. Note that these two scenarios were not exact opposites of each other, as there were other variables that were set either high or low for these runs (see Table 20). Bartholow et al. (in review) describe the definition of each of these settings and how the runs were set up. CE-QUAL-W2 runs without the TCD operating for the hydrologically wet and dry scenarios maximized hydropower releases (Figure 13), while the without-TCD run under 1995 conditions used the bypass releases as described in Table 9 (Figure 14). The pseudo-optimization model was used to develop the with-TCD distribution of releases for the three scenarios, with releases from one or more TCD elevations being changed on a daily basis (Figure 16). The different outlet distributions are illustrated in Figures 13, 14, 15 and 16 for 1995 input conditions. Descriptions of the outlet elevations are given in Table 21.

Table 20. CE-QUAL-W2 input variable settings for hydrologic scenarios used in assessment of effects of changes in algal production due to the TCD on reservoir fishery

Scenario	Initial water surface	Inflows	Outflows	Nutrient loading	Air temperature	Cloud cover
Hydrologically wet	*High*	*High*	*Low*	*High*	*Low*	*Low*
Hydrologically dry	*Low*	*Low*	*High*	*High*	*High*	*Low*

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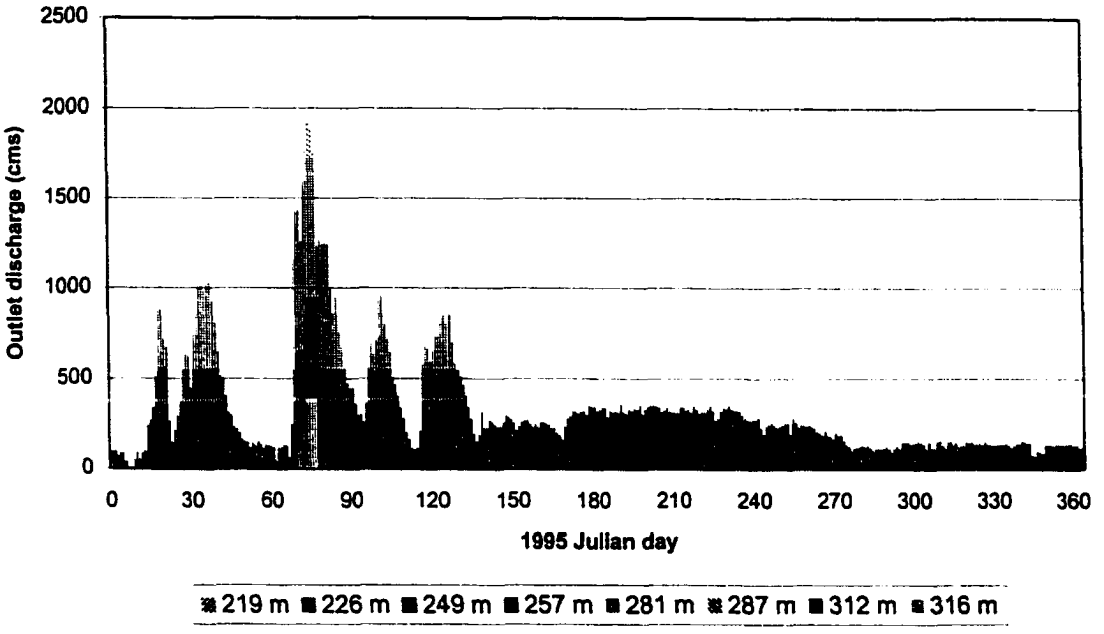

Figure 13. Without-TCD release distributions to maximize hydropower

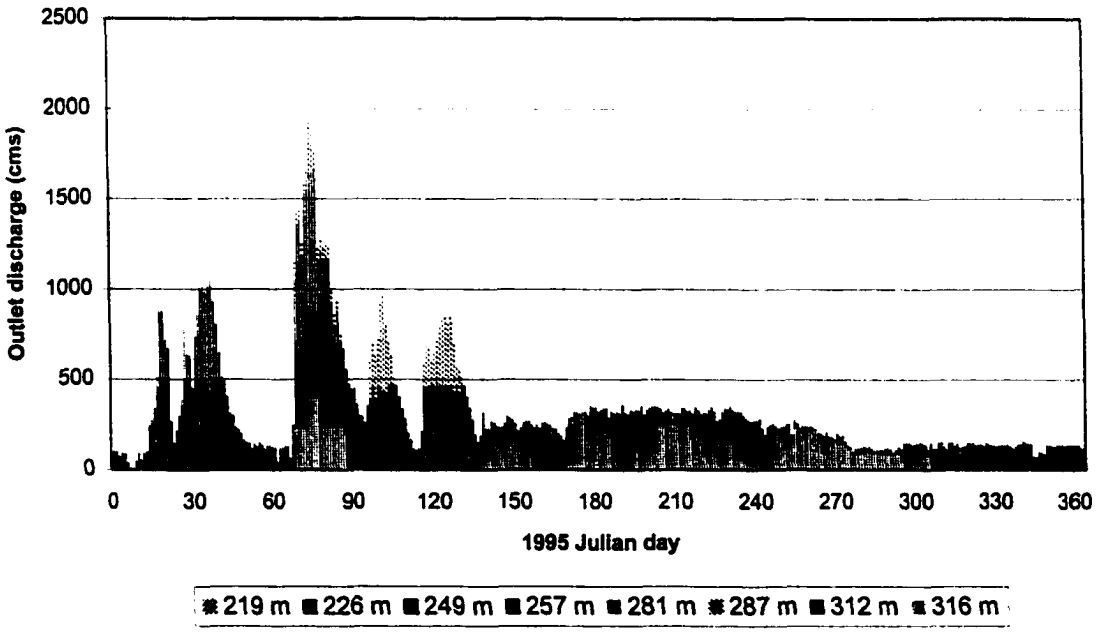

Figure 14. Without-TCD release distributions using bypass outlets

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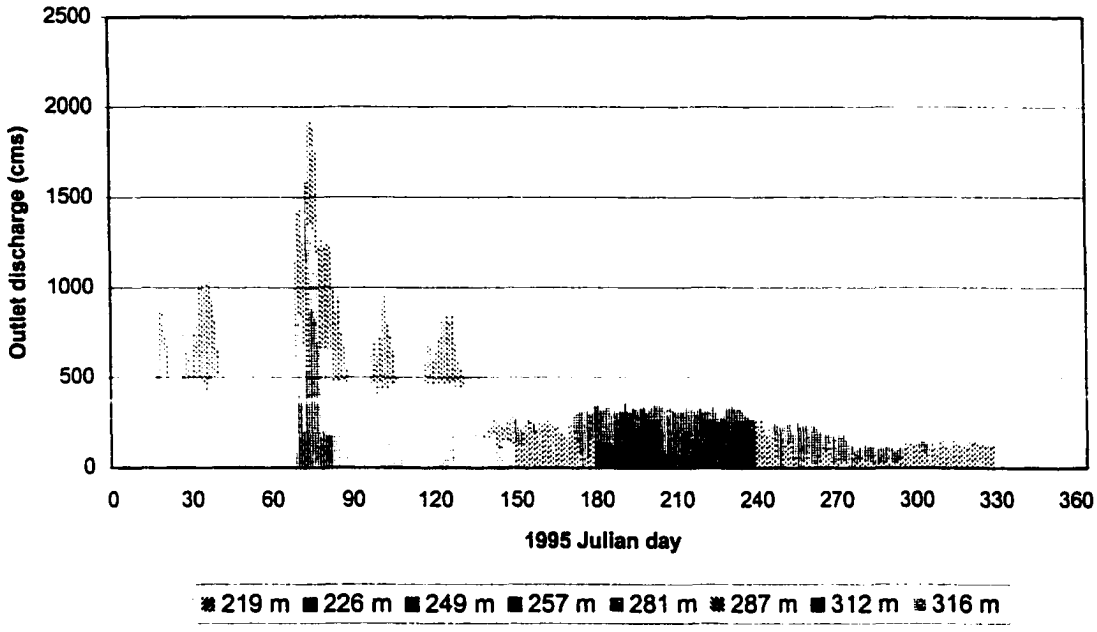

Figure 15. With-TCD simplified release distributions

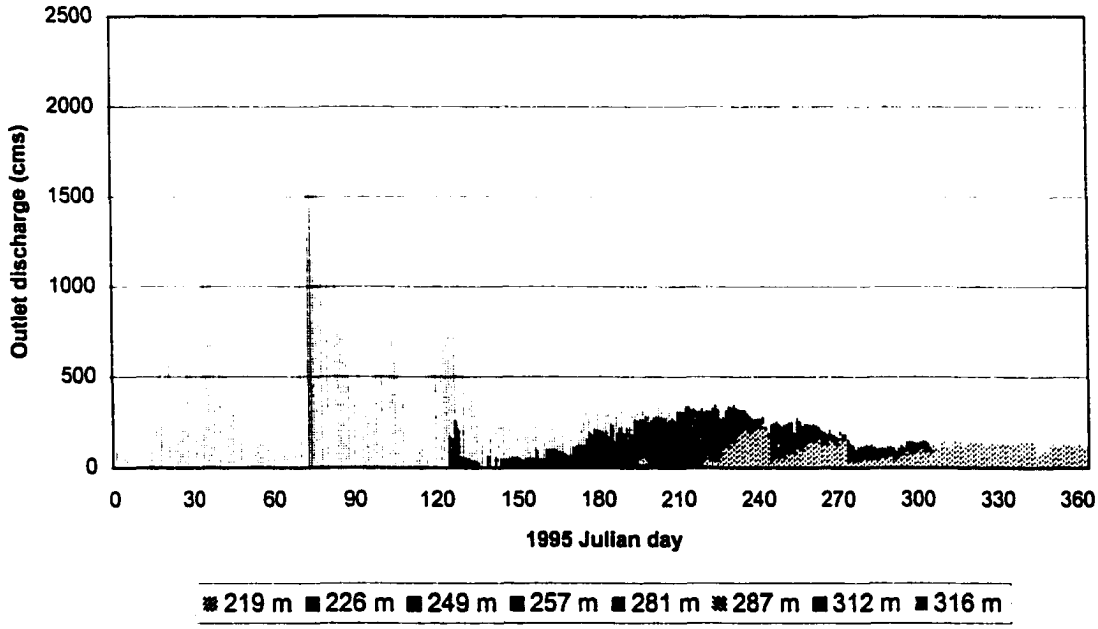

Figure 16. With-TCD pseudo-optimized release distributions

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Table 21. Description of release outlets with and without the TCD

Release elevation (m)	Pre-TCD description	With-TCD description
219 m	Did not exist	Low level through penstock
226 m	Low-level bypass outlets	Low-level bypass outlets
249 m	Penstocks to power plant	Low-level to penstocks
257 m	Mid-level bypass outlets	Mid-level bypass outlets
281 m	Did not exist	Mid-level to penstocks
287 m	High-level bypass outlets	High-level bypass outlets
312 m	Did not exist	High-level to penstocks
316 m	Spillway	Spillway

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Chapter 6. METHODS - BIOENERGETICS

6.1 Model development

Two programs were developed that use bioenergetics concepts to calculate 'scope for growth' for rainbow trout and smallmouth bass. Rainbow trout and smallmouth bass were selected because they were two of the most abundant sport fish species in Shasta Lake (CDFG 1991). The former is a cold water species, while the latter prefers warm water environments.

Scope for growth represents the opportunity for growth of a species, assuming that food is abundant and no other competitors or predators are present. It is the energy remaining for growth and reproduction after metabolic requirements are met (Hewett 1989; Wootton 1990). Because food is assumed abundant, scope for growth is solely dependent on water temperature.

Both of the programs were written using C++. They incorporate bioenergetics equations with corresponding coefficients from Hanson et al. (1997) to calculate scope for growth. Equations for anadromous steelhead were used for the primarily hatchery-stocked rainbow trout. Equations were added to convert terms for the scope for growth equation (Equation 68) into energy units in joules (J) because energy is the preferred model 'currency' so that energy density differences between fish and their prey can be accounted for (Hewett 1989). All equations and coefficients presented in this chapter are from Hanson et al. (1997) unless indicated with an asterisk (*). In addition to calculating

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scope for growth, the programs generate Tecplot grid files to enable graphical images of results. The programs were included in the Lakehab interface to facilitate the linkage between CE-QUAL-W2 water temperature outputs and scope for growth calculations. Output files from CE-QUAL-W2 were read by Lakehab, and then the user could choose different options to analyze the CE-QUAL-W2 data. When using Lakehab to do the scope for growth calculations, the user must specify which species of fish to analyze (i.e., rainbow trout or smallmouth bass), the wet weight of fish to analyze in grams, and the composition of the diet (i.e., proportions, digestibility and energy densities of prey).

The general procedure for calculating scope for growth is to first determine how much energy the fish consumes. This is done using a set of consumption equations. The general equations used for the Shasta Lake application were as follows:

$$C = C_{max} \cdot p \cdot f(T) \qquad \text{Equation 33}$$

$$C_{max} = CA \cdot W^{CB} \qquad \text{Equation 34}$$

$$CE = C \cdot PREYE \qquad \text{*Equation 35}$$

where

C = specific consumption rate (g g^{-1} d^{-1})

C_{max} = maximum specific feeding rate (g $g^{-1}d^{-1}$)

CE = specific consumption rate in terms of energy (J g^{-1} d^{-1})

p = proportion of maximum consumption

$f(T)$ = temperature dependence function

T = water temperature (°C) (input from CE-QUAL-W2)

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W = fish mass (g) (input by user)

CA = intercept of the allometric mass function for a 1 g fish at the optimum water temperature

= 0.628 for rainbow trout

= 0.303 for smallmouth bass

CB = slope of the allometric mass function

= -0.3 for rainbow trout

= -0.275 for smallmouth bass

$PREYE$= prey energy density (see Equation 67) (J g^{-1})

For rainbow trout, the temperature dependence function $f(T)$ for Equation 33 was:

$$f(T) = K_A K_B \qquad \text{Equation 36}$$

where

$$K_A = \frac{(CK1 \cdot L1)}{(1 + CK1 \cdot (L1 - 1))} \qquad \text{Equation 37}$$

$$L1 = e^{(G1 \cdot (T - CQ))} \qquad \text{Equation 38}$$

$$G1 = \frac{1}{(CTO - CQ)} \cdot \ln\left(0.98 \cdot \frac{(1 - CK1)}{(CK1 \cdot 0.02)}\right) \qquad \text{Equation 39}$$

$$K_B = \frac{(CK4 \cdot L2)}{(1 + CK4 \cdot (L2 - 1))} \qquad \text{Equation 40}$$

$$L2 = e^{(G2 \cdot (CTL - T))} \qquad \text{Equation 41}$$

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$$G2 = \frac{1}{(CTL - CTM)} \cdot \ln\left(0.98 \cdot \frac{(1 - CK4)}{(CK4 \cdot 0.02)}\right)$$ Equation 42

CQ = lower water temperature at which the temperature dependence is $CK1$ of the maximum rate

= 5°C

CTO = water temperature corresponding to 0.98 of the maximum consumption rate

= 20°C

$CK1$ = small fraction of the maximum consumption rate

= 0.33

CTM = water temperature ($\geq CTO$) at which dependence is still 0.98 of the maximum

= 20°C

CTL = water temperature at which dependence is $CK4$ of the maximum rate

= 24°C

$CK4$ = reduced fraction of the maximum consumption rate

= 0.2

For smallmouth bass, the temperature dependence function $f(T)$ for the consumption calculation (Equation 33) was:

$$f(T) = V^{X} \cdot e^{(X \cdot (1 - V))}$$ Equation 43

where

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$$V = \frac{(CTM - T)}{(CTM - CTO)}$$ Equation 44

$$X = \frac{\left(Z^2 \cdot \left(1 + \left(1 + \frac{40}{Y}\right)^{0.5}\right)^2\right)}{400}$$ Equation 45

$$Z = \ln(CQ) \cdot (CTM - CTO)$$ Equation 46

$$Y = \ln(CQ) \cdot (CTM - CTO + 2)$$ Equation 47

CTO = laboratory temperature preferendum

= 18°C

CTM = maximum water temperature above which consumption ceases

= 23°C

CQ = approximates a Q_{10} (the rate at which the function increases over relatively low water temperatures

= $2.3°C^{-1}$

Next, loss of energy due to waste losses such as egestion (i.e., fecal waste) and excretion (i.e., nitrogenous waste) were calculated. For rainbow trout, the following equations were used:

$$F = PF \cdot CE$$ Equation 48

$$U = UA \cdot T^{UB} \cdot e^{(UG \cdot P)} \cdot (CE - F)$$ Equation 49

where

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$$PF = \frac{(PE - 0.1)}{0.9} \cdot (1 - PFF) + PFF$$

$$PE = FA \cdot T^{FB} \cdot e^{(FG \cdot p)}$$

$$PFF = \sum (PREY[n] \cdot DIET[n]) \text{ for } n = 1 \text{ to number of prey}$$

F = egestion (J g^{-1} d^{-1})

FA = intercept of the proportion of consumed energy egested versus water temperature and ration

= 0.212

FB = coefficient of water temperature dependence on egestion

= -0.222

FG = coefficient for feeding level dependence (P-value) of egestion

= 0.631

U = excretion (J g^{-1} d^{-1})

UA = intercept of the proportion of consumed energy excreted versus water temperature and ration

= 0.0314

UB = coefficient of water temperature dependence on excretion

= 0.58

UG = coefficient for feeding level dependence (P-value) of excretion

= -0.299

$PREY[n]$= indigestible proportion of nth prey

$DIET[n]$= proportion of nth prey in diet

CE = specific consumption rate calculated by Equation 35 (J g^{-1} d^{-1})

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For smallmouth bass, the waste loss equations used were:

$$F = FA \cdot CE \qquad \text{Equation 50}$$

$$U = UA \cdot (CE - F) \qquad \text{Equation 51}$$

where

F = egestion (J g^{-1} d^{-1})

FA = proportion of consumption that is egestion

= 0.104

U = excretion (J g^{-1} d^{-1})

UA = proportion of assimilated energy that is excretion

= 0.068

CE = specific consumption rate calculated by Equation 35 (J g^{-1} d^{-1})

Loss of energy due to respiration was then calculated according to the following set of equations:

$$R = RA \cdot W^{RB} \cdot f(T) \cdot ACTIVITY \qquad \text{Equation 52}$$

$$RE = R \cdot OXYCAL \qquad \text{*Equation 53}$$

$$S = SDA \cdot (CE - F) \qquad \text{Equation 54}$$

where

R = specific rate of respiration (g g^{-1} d^{-1})

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RE = specific rate of respiration in terms of energy (J g^{-1} d^{-1})

W = fish mass (g)

RA = intercept of the allometric mass function (see specific definition under species equations)

RB = slope of the allometric mass function for standard metabolism

= -0.217 for rainbow trout

= -0.21 for smallmouth bass

f(T) = temperature dependence function

T = water temperature (°C)

OXYCAL= oxycalorific coefficient

= 13560 (J g O_2^{-1}) (Elliott and Davison 1975)

S = proportion of assimilated energy lost to specific dynamic action

SDA = specific dynamic action

= 0.172 for rainbow trout

= 0.16 for smallmouth bass

CE = specific consumption rate calculated by Equation 35 (J g^{-1} d^{-1})

F = specific egestion rate calculated by Equation 48 for rainbow trout or Equation 50 for smallmouth bass (g g^{-1} d^{-1})

The temperature dependence function *f(T)* and the *ACTIVITY* function for each species for the calculation of energy losses due to respiration were as follows:

Rainbow trout:

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$$f(T) = e^{(RQ \cdot T)}$$ Equation 55

$$ACTIVITY = e^{(RTO \cdot VEL)}$$ Equation 56

where

$$VEL = RK1 \cdot W^{RK4}, \text{ when } T > RTL, \text{ or}$$ Equation 57

$$VEL = ACT \cdot W^{RK4} \cdot e^{(BACT \cdot T)}, \text{ when } T \leq RTL$$ Equation 58

RA = specific weight of oxygen consumed by a 1 g fish at 0°C and zero swimming speed

= 0.00264 g O_2 g^{-1} d^{-1}

RQ = approximates the Q_{10} (rate at which the function increases over relatively low water temperatures)

= 0.06818°C^{-1}

RTO = coefficient for swimming speed dependence on metabolism

= 0.0234 s cm^{-1}

RTL = cutoff temperature at which the activity relationship changes

= 25°C

RK1 = intercept for swimming speed above the cutoff temperature

= 1 s cm^{-1}

RK4 = mass dependence coefficient for swimming speed at all water temperatures

= 0.13

ACT = intercept of the relationship for swimming speed versus mass at water temperatures less than *RTL*

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= 9.7 cm s^{-1} for a 1 gram fish at 0°C

BACT = water temperature dependence coefficient of swimming speed at water temperatures below *RTL*

= 0.0405°C^{-1}

Smallmouth bass:

$$f(T) = V^{X} \cdot e^{(X \cdot (1-V))} \quad \text{Equation 59}$$

$$ACTIVITY = ACT \quad \text{Equation 60}$$

where

$$V = \frac{(RTM - T)}{(RTM - RTO)} \quad \text{Equation 61}$$

$$X = \frac{\left(Z^2 \cdot \left(1 + \left(1 + \frac{40}{Y}\right)^{0.5}\right)^2\right)}{400} \quad \text{Equation 62}$$

$$Z = \ln(RQ) \cdot (RTM - RTO) \quad \text{Equation 63}$$

$$Y = \ln(RQ) \cdot (RTM - RTO + 2) \quad \text{Equation 64}$$

ACT = activity multiplier

= 2

RTO = optimum temperature for respiration

= 30°C

RTM = maximum (lethal) water temperature

= 37°C

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RQ = approximates the Q_{10} (the rate at which the function increases over relatively low water temperatures)

= $3.3°C^{-1}$

RA = number of grams of oxygen consumed by a 1 g fish at RTO

= 0.009 g O_2 g^{-1} d^{-1}

Predator and prey energy densities were needed to convert between wet weight and energy. To calculate predator energy density for rainbow trout, the following equation was used:

$$ED = \alpha + \beta W \qquad \text{Equation 65}$$

where

ED = predator energy density (J g^{-1} wet mass)

α = intercept of the allometric mass function (J g^{-1})

β = slope of the allometric mass function

W = fish mass (g) (input by user)

For rainbow trout, the predator energy density was defined using two size ranges. A *cutoff* weight was specified, with α_1 and β_1 used at or below the *cutoff* weight, and α_2 and β_2 used at all other weights. The coefficients were as follows (Hanson et al. 1997):

α_1 = 5764 J g^{-1}

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β_1 = 0.9862

cutoff = 4000 g

α_2 = 7602 J g^{-1}

β_2 = 0.5266

The predator energy density for smallmouth bass was the same for all sizes:

ED = 4186 J g^{-1} wet mass Equation 66

The programs calculated prey energy density according to the following equation:

$$PREYE = \sum(PREY[n] \cdot DIET[n] \cdot PREYEN[n]) \text{ for } n = 1 \text{ to number of prey}$$ *Equation 67

where

PREYE = overall prey energy density (J g^{-1} wet mass)

PREY[*n*] = indigestible proportion of *n*th prey (input by user)

DIET[*n*] = proportion of *n*th prey in diet (input by user)

PREYEN[*n*] = energy density of *n*th prey in diet (J g^{-1} wet mass) (input by user)

The programs included a check to ensure that 100 % of the diet was accounted for (i.e., $\Sigma DIET[n] = 1$. For all of the Shasta Lake simulations, the diet of rainbow trout and smallmouth bass were assumed to be 100% fish, with prey energy densities of 4200 J g^{-1}, and the prey were assumed to be 100% digestible.

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The scope for growth for both species was calculated according to the following equations:

$$SFGE = CE - F - S - U - RE$$ *Equation 68

$$SFG = \frac{SFGE}{ED}$$ *Equation 69

where

$SFGE$ = scope for growth in terms of energy (J g^{-1} d^{-1})

SFG = scope for growth in terms of mass (g g^{-1} d^{-1})

The programs allowed output in terms of scope for growth (g g^{-1} d^{-1}) or as a percent of maximum scope for growth. If percent of maximum scope for growth was the selected output, the programs calculated the maximum scope for growth for the species and weight of fish that were entered by the user. An algorithm determined the maximum scope for growth and temperature at which it occurred, and stored the scope for growth value for use in the following equation:

$$SFGP = \frac{SFG}{MAXSFG} \cdot 100\%$$ Equation 70

where

$MAXSFG$ = maximum scope for growth for the given species and weight (g g^{-1} d^{-1})

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No model calibration was performed because there was no measured fish data to compare model output with. Model verification was done by running the Windows-based bioenergetics model of Hanson et al. (1997) for specific temperatures, masses of predators, and diet compositions to check that the same numbers were calculated as with the Shasta Lake bioenergetics model.

6.2 Application of bioenergetics model for TCD assessment

Water temperature predictions from the CE-QUAL-W2 runs for calendar years 1968, 1974, 1976 and 1995 under with-TCD and without-TCD conditions were used with the bioenergetics model to calculate scope for growth in the reservoir through time and space. The difference in scope for growth for a 100 g fish of each species was calculated as with-TCD minus without-TCD and was plotted to determine the location, timing and magnitude of changes in scope for growth.

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Chapter 7. METHODS – FOOD WEB – ENERGY TRANSFER MODEL

7.1 General model background

One of the major constraints in making predictions about the fishery impacts in Shasta Lake was the scarcity of available quantified fish data at the reservoir. There also had never been any formal food web analysis of the structure of the aquatic ecosystem at Shasta. For these reasons, a modeling approach was selected in which the algal predictions in each CE-QUAL-W2 model cell were converted to available net production energy, and this available energy was transferred through a food web to specific fish species. The transferred energy was interpreted as growth potential and was compared between conditions with and without the TCD operating to assess its effects through changes in primary algal production.

In order to transfer the available energy from phytoplankton to fish growth potential, an estimate of the food web at Shasta Lake was needed. Kling et al. (1992) have argued that stable isotopes can be used to determine the strength of trophic interactions and to trace energy flows through ecosystems, and this was the approach selected to estimate the food web.

7.2 Stable isotope analysis

A chemical element can have several isotopes, all of which have the same number of protons but different numbers of neutrons. Because of the difference in numbers of

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neutrons, isotopes of the same element will have different mass numbers. Stable isotope analysis involves the comparison of the measured ratio of naturally occurring heavy and light isotopes of a sample with the ratio of a primary standard of the same element. The ratios, represented by the letter *R*, are calculated in terms of the ratio of the heaviest isotope (higher mass number) to the lighter one. The ratios are measured using an isotope ratio mass spectrometer, with typical per sample charges averaging $10-100, depending on the sample matrix and the type of isotope being analyzed (Lajtha and Michener 1994). For university projects, the Natural Resources Ecology Laboratory at Colorado State University (CSU) charges about $6 per sample analysis.

The standard notation used is (Lajtha and Michener 1994):

$$\delta(‰) = \frac{R_{sa} - R_{std}}{R_{std}} \times 1000 = \left(\frac{R_{sa}}{R_{std}} - 1 \right) \times 1000 \qquad \text{Equation 71}$$

where

$\delta(‰)$ = per mil difference (parts per thousand)

R_{sa} = isotope ratio of a sample

R_{std} = isotope ratio of a standard

Typical ratios used in ecological studies and the associated primary standards are shown in Table 22. 'Enriched' or 'heavier' samples are those that contain more of the heavier isotope than other samples, while those that contain less of the heavy isotope in comparison with other samples are considered 'depleted' or 'lighter' (Lajtha and Michener 1994).

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Table 22. Stable isotope ratios and associated primary standards

Isotope	Ratio	Primary standard
Carbon	$^{13}C/^{12}C$	Pee Dee Belemnite
Nitrogen	$^{15}N/^{14}N$	Atmospheric air (N_2 gas)
Sulfur	$^{34}S/^{32}S$	Troilite standard of the Canyon Diablo meteorite
Oxygen	$^{18}O/^{16}O$	Vienna Standard Ocean Water
Hydrogen	$^{2}H/^{1}H$	Vienna Standard Ocean Water

Source: Lajtha and Michener (1994)

The significance of stable isotopes in ecosystem studies is that they incorporate two kinds of information: origin and fractionation. The isotopic signature of an individual will reflect the signature of the source of the isotopes (i.e., where the isotope first entered the food web) and the change in the isotopic signature due to isotopic fractionation by physical and chemical reactions as the isotopes are consumed and metabolized in the food web (Peterson and Fry 1987).

The most significant stable isotope ratios in aquatic food web studies are the carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) ratios. There appears to be a slight (0.2 – 1.1 ‰) enrichment in $\delta^{13}C$ in animals relative to their diet due to: 1) the preferential loss of ^{12}C through carbon dioxide (CO_2) during respiration; 2) the preferential uptake of ^{13}C through digestion of foods containing ^{13}C compounds; or 3) metabolic fractionation of carbon during the synthesis of different tissue types (France and Peters 1997; Michener and Schell 1994; Peterson and Fry 1987). The very slight enrichment means that the carbon isotopic composition transfer is fairly conservative, and the $\delta^{13}C$ signature of the primary producer (first organic food source) is likely to be preserved through several trophic

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levels (Michener and Schell 1994; Yoshioka et al. 1994). Thus, carbon isotope analysis can be used to distinguish the influence of different primary food sources if the isotopic signatures of those food sources are distinctive enough (Forsberg et al. 1993; Michener and Schell 1994).

The enrichment of $\delta^{15}N$ with trophic level is much greater than with carbon, mostly due to the excretion of ^{14}N in urine (Peterson and Fry 1987). The $\delta^{15}N$ values of animals tend to reflect the nitrogen isotope ratios in the primary producers with a gain of about 3-4 ‰ per trophic level (Michener and Schell 1994). Because of this phenomenon, the nitrogen isotope ratio has been used as an indicator of the trophic level of an animal (Fry 1988; Kling et al. 1992; Yoshioka et al. 1994).

There are several advantages to applying stable isotope analysis in aquatic food web research over other methods such as analyzing stomach contents, direct observation of predator/prey interactions, and radiolabeling with radioactive isotopes. Michener and Schell (1994) provide a good discussion of these advantages, and the primary points they make are summarized here. First, isotopes build up in body tissues over time, so a one-time analysis provides a time-integrated measure of the diet (Fry and Sherr 1984; Hesslein et al. 1993; Rounick and Winterbourn 1986; Vander Zanden et al. 1998). This is an advantage over stomach analysis in which a one-time sample only gives a 'snapshot' of the diet, and important diet items can be easily missed. Differences in digestibility of prey may also make identification of stomach contents difficult, especially because some prey lose their morphological characteristics quickly (France 1997b; Hamilton et al. 1992; Kling et al. 1992; Michener and Schell 1994; Rounick and Hicks 1985). However, diet analysis can complement isotope analysis in complex food webs

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because of its taxonomic resolution (Hecky and Hesslein 1995; Hobson and Welch 1995; Vander Zanden et al. 1998).

Another advantage of using stable isotopes in aquatic food web studies is that only a small sample of body tissue is needed, so larger fish do not have to be killed in order to measure their isotopic signatures. Also, if different tissues of an organism are analyzed, isotopic analysis can be used to identify seasonal or life stage diet shifts because the assimilation of isotopes varies in different tissue types (France and Steedman 1996; Fry and Sherr 1984; Peterson and Fry 1987; Tieszen et al. 1983). Finally, isotope signatures can be measured in all levels of the food web, including phytoplankton, zooplankton, and aquatic insects (Michener and Schell 1994). This makes isotope analysis very useful for detecting omnivory in food webs (France 1997a; Vander Zanden and Rasmussen 1996).

The costs of stable isotope analysis are competitive with these other methods. As with stomach analysis, sampling is accomplished by methods such as electrofishing, gill netting, trap netting, and dip netting. The major analysis cost involves the use of the mass spectrometer, and the semi-automation of the analysis procedure is making this process more economical and time efficient (Michener and Schell 1994).

In applying stable isotope analysis to aquatic food web studies, many researchers advocate the use of multiple tracers because of the distinct behavior of different isotopes in food webs (France 1995b; France and Steedman 1996; Fry and Sherr 1984; Harrigan et al. 1989; Hobson and Welch 1995; Michener and Schell 1994; Peterson and Fry 1987). In addition, DeNiro and Epstein (1978, 1981) and Tieszen et al. (1983) recommend that several different tissue types should be analyzed when estimating isotopic ratios for

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animals and fish because the isotopic composition of different tissue types within an organism can vary. On the other hand, many studies have selectively sampled fish muscle tissue (Forsberg et al. 1993; France and Steedman 1996; Fry 1986; Fry 1988; Gu et al. 1996; Hamilton et al. 1992; Hobson and Welch 1995; Rau et al. 1983), or have analyzed whole bodies of smaller fish (Bootsma et al. 1996; Yoshioka et al. 1994). Estep and Vigg (1985) found that both fish scales and muscle tissue could be used to determine diet, while Rounick and Hicks (1985) and Pinnegar and Polunin (1999) analyzed several types of fish tissues and concluded that the white muscle tissue was the most appropriate for studies of aquatic food webs. The intermediate signatures of muscle tissue have also been observed in studies of gerbils and birds (Mizutani et al. 1991; Tieszen et al. 1983).

7.3 Measurement of isotopes

During the week of July 6, 1998, over 300 samples were collected from the Pit River, Sacramento River, and McCloud River arms and the main body of Shasta Lake. Sampled items included fish, aquatic insects, terrestrial insects, and zooplankton. All sampling in July 1998 was done using boats provided by the CDFG, USBR or the USGS (Table 23). Additional samples were obtained from a reconnaissance trip in June 1998, and from Ray Bark of the USBR, whose samples were collected from the entrainment nets at Shasta Dam in April, June, and July 1998 (Table 24).

Fish were collected using one of the following methods: electrofishing, gill netting, trap netting, hook-and-line fishing, or grab sampling. The insects were gathered using hand-held nets or by hand. Zooplankton were sampled by taking a 30-0 m (or

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depth at site to 0 m) vertical haul with a 64 μm or a 500 μm net. See Appendix D for more details on sampling methods.

Table 23. Summary of individual items obtained from July 1998 sampling

Species	# of individual items at each location			
	Main lake	McCloud River arm	Pit River arm	Sacramento River arm
Fish				
Alabama spotted bass	27	40	37	41[a]
Black crappie	0	0	3	0
Bluegill	7	19	29[b]	1
Brown bullhead	1	0	0	0
Brown trout	1	0	0	0
Carp	0	1	1	0
Channel catfish	1	0	6	1
Chinook salmon	7	0	0	0
Golden shiner	1	0	0	0
Green sunfish	4	2	4	4
Largemouth bass	0	2	1	0
Rainbow trout	5	0	0	0
Sacramento squawfish	3	1	0	1
Smallmouth bass	9	2	0	0
Threadfin shad	1	0	13	0
White catfish	0	1	4	0
White crappie	0	0	2	0
Aquatic and terrestrial insects				
Ants	1	0	0	1
Beetles	0	0	0	2
Caddis fly	0	0	0	1
Chironomid larvae	0	0	0	3[c]
Crane fly	0	0	0	1
Mayfly	0	0	0	1
Moth	1	0	0	0
Water striders	2[c]	0	3[c]	3[c]
Other				
Zooplankton	1	1	2	1

[a]Includes one hybrid individual that was not processed for analysis
[b]Two bluegill larvae were combined into one individual sample
[c]Individual items of the same species were combined into one individual sample

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Table 24. Summary of individual items obtained from entrainment study (April, June, and July 1998) and reconnaissance trip (June 1998)

Species	# of individual items at each location			
	Main lake entrainment[a]	Main lake	Pit River arm	Sacramento River arm
Fish				
Alabama spotted bass	0	1	1	2
Bluegill	1	0	0	0
Chinook salmon	1	0	0	0
Golden shiner	4	0	0	0
Sculpin	12	0	1	0
Sucker	1	0	0	0
Threadfin shad	5	0	0	0
White catfish	1	0	0	0
Aquatic insects				
Water striders	0	0	0	2
Other				
Zooplankton	0	1	2	1
Crayfish	1	0	0	0

[a] Collected from entrainment nets in April, June and July 1998

Dorsal muscle samples were taken from all fish larger than 67 millimeters (mm) total length (TL) using either a 5 mm diameter biopsy needle or by removing 0.5-1.0 cm^3 of dorsal muscle tissue with a filet knife. For smaller fish (< 67 mm), whole individuals

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were collected for analysis. Species, TL and wet weight were recorded for fish samples. Aquatic and terrestrial insects were placed whole in sample bags, labeled, and recorded. Water from each zooplankton haul was reduced as much as possible, and samples were then put in a sample bottle. All samples were immediately placed on ice in a cooler on the boat, and then frozen as soon as possible after return to shore. All samples were shipped overnight from Redding, California to Fort Collins, Colorado in coolers with dry ice, except the entrainment samples, which were shipped to the USBR in Denver. These were later shipped to Fort Collins from Denver in July 1998.

In addition to the aquatic samples, six bald eagle feathers were obtained from Ron Jackman, a consultant for the USFS. Although the feathers were prepared and analyzed for their isotopic signatures, they were not included in the food web analysis because of inadequate information about the terrestrial portion of their diet. For details on the eagle feather isotope analysis, please see Appendix E.

Sample processing was completed predominantly at the fish ecology laboratory in the Fishery and Wildlife Biology Department at CSU. Sample processing involved preparing and preserving fish, stomach, and tissue samples; preparing samples other than fish; drying samples; grinding samples; and preparing composite samples.

Stomachs were removed from all Alabama spotted bass from which dorsal muscle tissue was taken except for one angler-caught fish that was released after the dorsal sample was taken. Stomachs were also removed from some of the larger bluegill. Stomachs were not removed from other species except for one chinook salmon obtained from entrainment sampling. All stomachs were preserved with a 10% formalin solution.

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In general, fish smaller than 67 mm TL were prepared as whole fish samples. All whole fish and dorsal muscle tissue samples were placed individually in sample tins for drying. Zooplankton samples were thawed and placed in sample tins using a pipette. Aquatic and terrestrial insect samples were placed in sample tins as well, although some samples of the same species were combined in one tin to ensure that enough biomass would remain after drying for grinding purposes. The tail meat of the one crayfish obtained from entrainment sampling was placed in a sample tin. Samples were dried for at least 24 hours in a 60°C oven in the Fishery and Wildlife Biology Department at CSU.

Dried samples were ground to a homogeneous powder using a mortar and pestle as much as possible. Samples with high lipid content were ground with liquid nitrogen, which was very effective for improving the homogeneity of oily samples without affecting the measured signatures (Reuss 1998).

Individual fish samples were pooled into composite samples according to TL measurements (Manji 1998; Moyle 1976) and reservoir locations for Alabama spotted bass, smallmouth bass, chinook salmon, rainbow trout, black crappie, white crappie, bluegill, green sunfish, Sacramento squawfish, sculpins, threadfin shad, channel catfish, and white catfish. For detailed information on the determination of these composite samples, see Appendix F. Composite samples of fish species were prepared by measuring an equal amount (usually 10 milligrams (mg)) of each individual sample into a mortar, where the combined samples were ground into a homogeneous powder with a pestle. The only non-fish composite sample was of two water strider (*Hemiptera* spp.) samples.

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Composite samples of more than one individual of the same species were prepared in part because limited funding precluded the analysis of all of the individual samples separately, and because individuals of the same species needed to be combined in order to have enough biomass to do carbon and nitrogen isotopic analyses. In addition, researchers have often used composite samples of more than one individual (Fry 1988; Hobson and Welch 1995; Yoshioka et al. 1994) because the isotopic composition between different individuals of the same species eating the same diet can vary (DeNiro and Epstein 1978, 1981).

To test the hypothesis that the composite signatures were representative of the mean of the individual signatures of the composite sample, a one-way analysis of variance (ANOVA) analysis was done using the statistical package SAS (Release 6.12) to compare the mean signatures of individual samples with their corresponding composite signatures. For both carbon and nitrogen, the null hypothesis that the mean individual signatures were the same as the corresponding composite signatures was strongly confirmed (see Appendix G).

Approximately 1-2 mg of each sample to be analyzed was placed in a tin cup and processed for $\delta^{13}C$ and $\delta^{15}N$ ratios in a VG Isochrom mass spectrometer with Carlo-Erba NA1500 elemental analyzer at the Natural Resources Ecology Lab at CSU. Separate tins were prepared for the measurement of carbon and nitrogen signatures to improve the accuracy of the mass spectrometer.

The general estimate of machine error of the mass spectrometer used at CSU is 0.8 ‰ for both carbon and nitrogen signatures, based on repeating measurements on the same sample 30 times (Reuss 1998). The analytical error was independently estimated

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by making replicate measurements of 9 of the Shasta samples using two different analysts. The replicate measurements were compared by doing a two-way ANOVA analysis using SAS. The mean squared error (MSE) from this analysis was used to estimate the standard error of the mass spectrometer. The SAS results are included in Appendix H and are summarized in Table 25 below. For both carbon and nitrogen, the differences between replicate measurements were not significant at $\alpha = 0.05$. The estimated standard error of the mass spectrometer was 0.1162‰ and 0.4789‰ for carbon and nitrogen, respectively. Because these values were estimated using the Shasta data, they were considered more representative of the analytical error for the project than the general value of 0.8‰.

Table 25. Results of two-way ANOVA of 9 replicate carbon and nitrogen isotope signatures

Isotope	*H_0: replicates are the same* p-value	MSE[a] $(‰)^2$	Standard error ‰
Carbon	0.1569	0.0135	0.1162
Nitrogen	0.9202	0.2293	0.4789

[a] MSE = mean squared error

7.4 Construction of aquatic food web

As noted in Table 23 and Table 24, 19 fish species were collected in addition to zooplankton, one crayfish, and several species of aquatic and terrestrial insects. Most of the sample collections were made at four locations throughout the reservoir, and some

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additional samples were obtained from entrainment sampling at the dam. Because the primary objective of the food web-energy transfer model was to assess the impacts of TCD operations on popular sport fish in the reservoir, analyses were done to determine if the aquatic food web could be simplified to focus on species of interest. In particular, comparisons were made between locations to determine if locational differences were significant, and species and size class groupings were assumed. Additionally, isotopic signatures of some components at the base of the food web that had not been sampled were estimated. The following species were not included in these analyses because they were assumed to be irrelevant to a food web dominated by trout, salmon, and black bass: catfish (brown bullhead, channel catfish, and white catfish), carp, and sucker.

7.4.1 Locational and species grouping

Two sets of two-way ANOVAs were run in SAS of those species and size classes that were collected at more than one location to determine if differences in mean carbon or nitrogen signatures were significant between locations. In the first ANOVA, only data gathered during the July 1998 sampling was used. In the second ANOVA, additional data was used from the June 1998 preliminary (reconnaissance) sampling. Table 69 and Table 70 in Appendix I list the species and size classes that were used for the comparisons. Results were evaluated using $\alpha = 0.008333$ ($\alpha = 0.05$ adjusted for the six comparisons made). Table 26 and Table 27 show the p-values from each of the two-way ANOVAs for carbon and nitrogen signatures.

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Table 26. p-values for statistical analysis of mean differences between locations for July 1998 sampling data only

Location	Mean[a]	p-values			
		Main Lake	McCloud River arm	Pit River arm	Sacramento River arm
Carbon isotopes					
Main Lake	-29.23	--	0.1815	0.0004[b]	0.4149
McCloud River arm	-28.74	0.1815	--	0.0111	0.7100
Pit River arm	-27.56	0.0004[b]	0.0111	--	0.0098
Sacramento River arm	-28.49	0.4149	0.7100	0.0098	--
Nitrogen isotopes					
Main Lake	7.58	--	0.9877	0.0001[b]	0.6880
McCloud River arm	8.18	0.9877	--	0.0001[b]	0.6756
Pit River arm	9.13	0.0001[b]	0.0001[b]	--	0.0001[b]
Sacramento River arm	7.77	0.6880	0.6756	0.0001[b]	--

[a] Mean adjusted for unbalanced sample design
[b] Indicates adjusted mean isotope signatures are significantly different at $\alpha = 0.05/6 = 0.008333$

In general, both the carbon and nitrogen signatures were considerably higher for species collected from the Pit River arm as compared to corresponding collections from other arms. This observation was confirmed by the statistical analysis. Based upon these results, the carbon and nitrogen signatures for the Main Lake, Sacramento River arm, and the McCloud River arm were grouped together, and the signatures measured in the Pit River arm were not used in further analyses. This eliminated black crappie and white

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crappie from the estimated Shasta Lake food web. Sculpins and golden shiners were also removed from further analyses because of low reported abundance in Shasta Lake (see Table 1).

Table 27. p-values for statistical analysis of mean differences between locations for June and July 1998 sampling data

Location	Mean[a]	p-values			
		Main Lake	McCloud River arm	Pit River arm	Sacramento River arm
Carbon isotopes					
Main Lake	-29.67	--	0.1514	0.0001[b]	0.5178
McCloud River arm	-29.02	0.1514	--	0.0046[b]	0.4951
Pit River arm	-27.73	0.0001[b]	0.0046[b]	--	0.0010[b]
Sacramento River arm	-29.36	0.5178	0.4951	0.0010[b]	--
Nitrogen isotopes					
Main Lake	8.01	--	0.7108	0.0025[b]	0.6798
McCloud River arm	7.90	0.7108	--	0.0013[b]	0.9513
Pit River arm	8.89	0.0025[b]	0.0013[b]	--	0.0020[b]
Sacramento River arm	7.88	0.6798	0.9513	0.0020[b]	--

[a] Mean adjusted for unbalanced sample design
[b] Indicates adjusted mean isotope signatures are significantly different at $\alpha = 0.05/6 = 0.008333$

Species and size classes were then grouped by likely feeding habits of different species and size classes (see Appendix J for calculations of grouped signatures):

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- Bass species: It was assumed that the year class 1 individuals (0-113 mm TL) of all of the bass species (Alabama spotted bass, largemouth bass, and smallmouth bass) had similar diets. Diets of medium-sized bass were assumed to be similar for Alabama spotted bass and largemouth bass, but likely to be different for smallmouth bass. 'Medium bass' were assumed to be year classes 2 and 3 (114-290 mm). Only Alabama spotted bass individuals were collected that were larger than 290 mm, and all of these signatures were combined into a grouping for 'big bass.' The assumed diet similarities and differences between bass species were confirmed by literature review (Applegate and Mullan 1967; Coble 1975; Curtis 1949; Edwards et al. 1983; Emig 1966b, 1966c; Goodson 1965; Heidinger 1975; Hirst and DeVries 1994; McGinnis 1984; McKechnie 1966; McMahon et al. 1984; Miller and Kramer 1971; Moyle 1976; Turner 1966a; Vogele 1975; Wang 1986; Wanjala et al. 1986)

- Threadfin shad: all size classes of threadfin shad were assumed to eat similar diets (Baker and Schmitz 1971; Burns 1966b; Davis and Foltz 1991; DeVries et al. 1991; Hirst and DeVries 1994; Kampa 1984; Moyle 1976; Turner 1966b; Van Den Avyle and Petering 1988; Von Geldern and Mitchell 1975)

- Bluegill: all size classes of bluegill were assumed to eat similar diets (DeVries et al. 1991; Emig 1966a; Goodson 1965; Moyle 1976; Siefert 1972; Stuber et al. 1982a; Turner 1966a; Wang 1986)

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7.4.2 Estimation of signatures of primary producers

Phytoplankton signatures were not measured because they are known to be highly variable over time (France et al. 1997; Michener and Schell 1994). Because the phytoplankton life cycle is so short, Carpenter et al. (1985) point out that measured algal production will not correlate with biomasses of zooplankton and higher species collected at the same time. Similarly, isotopic signatures of predators will likely reflect signatures of phytoplankton that were present in the reservoir weeks before the higher species were sampled.

Variability of phytoplankton signatures can also occur because different species of phytoplankton exhibit different carbon and nitrogen signatures due to different metabolic pathways and processes, nutrient preferences, and nutrient assimilation (Boon and Bunn 1994; France et al. 1997; Goericke et al. 1994; Gu et al. 1997; Kling et al. 1992). In fact, some researchers have suggested that nitrogen isotopes can be indicators of nitrogen fixation amongst different species of phytoplankton (Alexander and Gu 1997). Variable phytoplankton signatures were expected at Shasta because, although Shasta is dominated by diatoms (*Melosira* sp.), a total of 134 phytoplankton species were collected during limnological sampling from spring 1995 through fall 1997 (Lieberman and Horn 1998).

Finally, different nutrient sources can affect phytoplankton signature variability. Kline et al. (1990) described a dependence of algae nitrogen signatures on the availability of ^{14}N in the dissolved nitrogen pool because of their preference to consume it over ^{15}N. This would cause algae signatures to show a shift once the available ^{14}N were consumed and they would be forced to consume ^{15}N. Likewise, some researchers suggest that

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phytoplankton carbon signatures will tend to become more enriched as lakes become more productive because of increased reliance on ^{13}C-enriched allochthonous organic matter or atmospheric CO_2 instead of respiratory CO_2 that is usually depleted in ^{13}C (France et al. 1997; Schindler et al. 1997). Keough et al. (1998) proposed that observed differences between carbon signatures of phytoplankton from offshore Lake Superior sites and nearby wetland areas were due to the fixation of atmospheric CO_2 by lake phytoplankton versus the use of a ^{13}C-depleted source in the more highly organic wetland waters and sediment.

In addition to the variability of measured phytoplankton signatures, it is difficult to obtain clean samples of phytoplankton that are not confounded with detritus, microzooplankton and bacteria (France et al. 1997; Kling et al. 1992; Michener and Schell 1994; Riera and Richard 1996). For all of these reasons, the phytoplankton signatures were estimated from the measured zooplankton signatures instead of measuring them directly. This 'baseline signature' approach (Vander Zanden et al. 1997) was based on the assumption that zooplankton were longer-lived primary consumer organisms of phytoplankton, and their signatures should therefore reflect integrated phytoplankton signatures. Zooplankton were assumed to consume 100% phytoplankton, and this assumption was used to estimate the phytoplankton signatures as follows:

$$\delta^{13}C'_{phyto} = \delta^{13}C_{zoo} - \Delta'_C \qquad \text{Equation 72}$$

$$\delta^{15}N'_{phyto} = \delta^{15}N_{zoo} - \Delta'_N \qquad \text{Equation 73}$$

where

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$\delta^{13}C'_{phyto}$ = estimated phytoplankton carbon signature (‰)

$\delta^{15}N'_{phyto}$ = estimated phytoplankton nitrogen signature (‰)

$\delta^{13}C_{zoo}$ = measured zooplankton carbon signature (‰)

$\delta^{15}N_{zoo}$ = measured zooplankton nitrogen signature (‰)

Δ'_C = estimated trophic enrichment of carbon signature (‰)

Δ'_N = estimated trophic enrichment of nitrogen signature (‰)

When diet proportions were estimated (Section 7.4.3), it was evident that phytoplankton were not the only primary organic food source in the Shasta Lake aquatic food web because of difficulty in matching estimated signatures with measured signatures. Although macrophytes were not likely to be a significant food source due to the frequent water level fluctuations at Shasta Lake (Brett 1999; Duffy 1999; Manji 1999), other primary organic food sources such as periphyton and detritus could contribute to the aquatic food web. These food sources were not expected to have the same signature as the phytoplankton (Hamilton and Lewis 1992). Unfortunately, samples were not gathered of these primary organic food sources, so it was necessary to estimate these signatures. A literature review was done of reported signatures for periphyton and detritus in north American temperate freshwater systems (see Appendix K). Species not present in Shasta Lake were eliminated from the synthesis. This review showed a wide variety of reported signatures that was likely due to species differences, and differences in source carbon and nitrogen signatures due to localized geology and sediment characteristics, fractionation processes, and anthropogenic inputs (Cabana and

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Rasmussen 1996; France 1995a, 1995b; Fry and Sherr 1984). Table 28 shows the resulting averages and corresponding ranges of reported signatures.

Table 28. Averages and ranges of reported periphyton and detritus carbon and nitrogen isotope signatures

Category	$\delta^{13}C$ (‰) reported values			$\delta^{15}N$ (‰) reported values		
	Average	Range	n	Average	Range	n
Periphyton	-23.23	-34.6 to -16.6	16	3.15	-0.7 to 6.6	15
Detritus	-23.32	-28.0 to -12.4	11	1.85	0 to 3.7	2

Because of the likelihood of local influences on the base signatures of the Shasta Lake food web, it was not appropriate to use the average of these literature-based values for periphyton and detritus. Instead, the relationships between periphyton or detritus signatures with other food web components were replicated for the Shasta Lake food web. This was done as follows:

- *Periphyton*: Angradi (1994) reported carbon and nitrogen signatures for chironomids, adult rainbow trout, and *Cladophora* with epiphytes 1-2 km downstream of Glen Canyon Dam on the Colorado River. Periphyton signatures were estimated using the signatures of chironomids that were included in the Shasta Lake food web as follows:

$$\delta^{13}C_{\text{peri, Shasta}} = \delta^{13}C_{\text{chir, Shasta}} - \left(\delta^{13}C_{\text{chir, Angradi (1994)}} - \delta^{13}C_{\text{epiphy, Angradi (1994)}}\right) \quad \text{Equation 74}$$

$$\delta^{15}N_{\text{peri, Shasta}} = \delta^{15}N_{\text{chir, Shasta}} - \left(\delta^{15}N_{\text{chir, Angradi (1994)}} - \delta^{15}N_{\text{epiphy, Angradi (1994)}}\right) \quad \text{Equation 75}$$

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where

$\delta^{13}C_{peri,\ Shasta}$ = estimated periphyton carbon signature (‰)

$\delta^{15}N_{peri,\ Shasta}$ = estimated periphyton nitrogen signature (‰)

$\delta^{13}C_{chir,\ Shasta}$ = measured chironomid carbon signature at Shasta (‰)

$\delta^{15}N_{chir,\ Shasta}$ = measured chironomid nitrogen signature at Shasta (‰)

$\delta^{13}C_{chir,\ Angradi\ (1994)}$ = measured chironomid carbon signature in Angradi (1994) (‰)

$\delta^{15}N_{chir,\ Angradi\ (1994)}$ = measured chironomid nitrogen signature in Angradi (1994) (‰)

$\delta^{13}C_{epiphy,\ Angradi\ (1994)}$ = measured *Cladophora* with epiphytes carbon signature in Angradi (1994) (‰)

$\delta^{15}N_{epiphy,\ Angradi\ (1994)}$ = measured *Cladophora* with epiphytes nitrogen signature in Angradi (1994) (‰)

Estimates were also made using the rainbow trout signatures from Shasta Lake and the Colorado River (Angradi 1994). Similar comparisons were made with published data from other studies, as shown in Table 29. See Appendix K for the associated calculations.

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Table 29. Comparisons used to estimate periphyton carbon and nitrogen signatures at Shasta Lake

Citation	Location	Estimated signatures		Shasta species used
		$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	
Angradi (1994)	1-2 km downstream of Glen Canyon Dam on Colorado River	-29.50	--	Chironomids
Angradi (1994)	1-2 km d/s of Glen Canyon Dam on Colorado River	-34.13	10.87	Rainbow trout
Bunn et al. (1989)	Alik Lake, Quebec	-24.60	--	Chironomids
Bunn et al. (1989)	Alik Lake, Quebec	-16.57	--	Tipula
Estep and Vigg (1985)	Pyramid Lake, Nevada	-28.30	--	Chironomids
Fry (1991a)	Hubbard Brook, Mirror Lake, NH	-23.72	-0.37	Crayfish
Fry (1991a)	Hubbard Brook, Mirror Lake, NH	-18.31	-0.24	Medium smallmouth bass
Fry (1991a)	Konza Prairie, KS	-32.41	4.32	Mayfly
Gu et al. (1997)	Smith Lake, Alaska	-32.60	--	Chironomids
Gu et al. (1997)	Smith Lake, Alaska	-28.92	4.68	Mayfly
Hecky and Hesslein (1995)	Canadian Shield lake	-22.25	1.04	Crayfish
Junger and Planas (1994)	Laflamme Creek just downstream of Lac Laflamme, Quebec	-27.60	--	Chironomid
Junger and Planas (1994)	De l'Aqueduc Creek, Quebec	-27.85	--	Chironomid
Peterson et al. (1993)	Tundra river ecosystem (control)	-32.7	--	Chironomid
Peterson et al. (1993)	Tundra river ecosystem (control)	-21.42	0.28	Mayfly
Rau (1980)	Lake Findlay, Washington	-23.30	--	Chironomids
Average		-26.51	2.94	

- *Detritus*: A process similar to that mentioned above for periphyton was used to estimate the detritus carbon and nitrogen signatures at Shasta Lake (Table 30 and Appendix K).

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Table 30. Comparisons used to estimate detritus carbon and nitrogen signatures at Shasta Lake

Citation	Location	Estimated signatures		Shasta species used
		$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	
Angradi (1994)	1-2 km downstream of Glen Canyon Dam on Colorado River	-23.60	--	Chironomids
Angradi (1994)	1-2 km d/s of Glen Canyon Dam on Colorado River	-28.23	7.56	Rainbow trout
Bunn et al. (1989)	Alik Lake, Quebec	-33.30	--	Chironomids
Bunn et al. (1989)	Alik Lake, Quebec	-25.27	--	Tipula
Fry (1991a)	Konza Prairie, KS	-34.21	4.05	Crayfish
Fry (1986)	Green Lake, New York	-28.89	--	Crayfish
Gu et al. (1997)	Smith Lake, Alaska	-28.40	--	Chironomids
Gu et al. (1997)	Smith Lake, Alaska	-29.80	--	Chironomids
Gu et al. (1997)	Smith Lake, Alaska	-24.72	-1.62	Mayfly
Gu et al. (1997)	Smith Lake, Alaska	-26.12	-5.12	Mayfly
Harrington et al. (1998)	Bingo Brook, Vermont	--	-0.90	Mayfly
Harrington et al. (1998)	West Branch, Vermont	--	-1.20	Mayfly
Harrington et al. (1998)	Bethel-Gilead tributary, Vermont	--	-2.72	Mayfly
Harrington et al. (1998)	Peavine tributary, Vermont	--	-0.05	Mayfly
Harrington et al. (1998)	First Branch, Vermont	--	-4.80	Mayfly
Harrington et al. (1998)	Third Branch, Vermont	--	-6.99	Mayfly
Rau (1980)	Lake Findlay, Washington	-26.00	--	Chironomids
Average		-28.04	-1.18	

7.4.3 Estimation of proportions of diet

Several different methods have been applied to estimate diet proportions using stable isotopes. Many of these approaches were concerned with two-source diets

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(Forsberg et al. 1993; Gu et al. 1996). Cermelj et al. (1996) also applied a two-source mixing model to estimate composition of lake sediments using carbon isotope ratios. Junger and Planas (1994) applied a two-source mixing model with biomass measurements to discriminate between lake-derived food sources and those that were autochthonous or allochthonous. For more than two sources, Fry and Sherr (1984) noted that the determination of diet proportions required more information than the measurement of only one kind of isotope. Kline et al. (1993) used a multiple-isotope method to determine the contributions of three food sources that was based on the plotted distance of extrapolated prey signatures from the predator's carbon and nitrogen signatures. An unsuccessful attempt was made to replicate this method. Instead, an approach very similar to that applied by Harrigan et al. (1989) was applied, as detailed below.

A spreadsheet was set up in Excel with a matrix of the final grouped species. The carbon and nitrogen signatures of each species were estimated based on the proportions of consumed species in its diet:

$$\delta^{13}C_i' = \sum_{j=1}^{n}\left[f_j'\left(\delta^{13}C_j + \Delta_C'\right)\right] \qquad \text{Equation 76}$$

$$\delta^{15}N_i' = \sum_{j=1}^{n}\left[f_j'\left(\delta^{15}N_j + \Delta_N'\right)\right] \qquad \text{Equation 77}$$

$$\sum_{j=1}^{n} f_j' = 1 \qquad \text{Equation 78}$$

where

$\delta^{13}C_i$ = estimated carbon signature of species i (‰)

$\delta^{15}N_i$ = estimated nitrogen signature of species i (‰)

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$\delta^{13}C_j$ = measured carbon signature of species j (‰)

$\delta^{15}N_j$ = measured nitrogen signature of species j (‰)

Δ'_C = estimated trophic enrichment of carbon signature (‰)

Δ'_N = estimated trophic enrichment of nitrogen signature (‰)

f'_j = estimated fraction of diet of species i that is species j

n = total number of species in diet

The assumption of Equation 78 implied that all possible food sources for each species have been included in the analysis. Corresponding f_j values were set to zero if a species was not 'allowed' by the model to consume a particular species. These rules were established from the stomach analysis of Alabama spotted bass and bluegill and a literature review. Note that the above equations allowed cannibalism (i.e., a species eating itself).

All year class 2 Alabama spotted bass stomachs from the Main Lake and McCloud River locations were analyzed. Half of the stomachs of Alabama spotted bass at other locations and bluegill stomachs at all locations were randomly selected for analysis. The diet composition of each stomach was partitioned into zooplankton, invertebrates and fish by percent volume of contents. The percent volumes of invertebrates and fish were separated by species where possible. Percent volumes of zooplankton were separated into cladocerans or copepods. The results of the stomach analyses by species, size class and location are shown in Tables 31 and 32.

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Table 31. Results of stomach analysis for Alabama spotted bass

Year class	Location	Number analyzed	% average diet composition by volume		
			Zooplankton	Invertebrate	Fish
1	McCloud River arm	5	66.6	32.0	1.4
	Pit River arm	2	5.0	95.0	0.0
1	Whole lake (average)	7	49.0	50.0	1.0
2	Main lake	9	56.0	37.0	7.0
	McCloud River arm	9	46.1	16.1	37.8
	Pit River arm	9	17.2	76.4	6.3
	Sacramento River arm	6	13.5	35.8	50.7
2	Whole lake (average)	33	35.6	41.7	22.7
3	Main lake	6	17.5	65.8	16.7
	McCloud River arm	3	3.3	63.3	33.3
	Pit River arm	2	0.0	52.5	47.5
	Sacramento River arm	6	22.7	46.0	31.3
3	Whole lake (average)	17	14.8	56.8	28.4
4	Main lake	1	0.0	10.0	90.0
	McCloud River arm	3	0.0	4.0	96.0
	Pit River arm	3	0.0	66.7	33.3
	Sacramento River arm	2	2.5	5.0	92.5
4	Whole lake (average)	9	0.6	25.8	73.7
5	Pit River arm	2	10.0	0.0	90.0
All	Whole lake (average)	68	26.5	43.0	30.5

Note: Five Alabama spotted bass stomachs and the one chinook salmon stomach were empty and are not included in the results shown

Almost all of the zooplankton found in the fish stomachs were cladocerans, with very few copepods found. Invertebrates included hymenoptera, chironomid, hemiptera, orthoptera, culicidae, coleoptera, caddis, ephermeroptera, amphipod, gastropod, and oligochaeta. Most fish found in the stomachs were unidentifiable. The majority of identifiable fish parts were those of threadfin shad. Some crayfish were also found, all in Alabama spotted bass stomachs. Appendix L contains the stomach data along with pie charts of the results.

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Table 32. Results of stomach analysis for bluegill

Year class	Location	Number analyzed	% average diet composition by volume		
			Zooplankton	Invertebrate	Fish
ALL	Main lake	2	2.5	97.5	0.0
	McCloud River arm	4	47.5	47.5	5.0
	Pit River arm	10	27.8	72.2	0.0
ALL	Whole lake (average)	16	29.6	69.2	1.2

Table 33 is a matrix of each of the fish species and zooplankton that indicates what items it was allowed to eat in the Shasta Lake food web model, based on a literature review (Appendix M). Note that although the literature review did not indicate that medium black bass species eat zooplankton, the stomach analysis for Shasta fish did show considerable amounts of zooplankton in the Alabama spotted bass diet, so medium bass were allowed to eat zooplankton.

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Table 33. Diet rules for species in Shasta Lake food web

Prey	Consumer												
	CHS	BRT	BBS	MSMB	MBS	RBT	TFS	YC1BS	CRA	GSF	SQF	BLG	ZOO
CHS		X							X				
BRT									X				
BBS	X	X							X				
MSMB	X	X	X		X				X				
MBS	X	X	X		X				X				
RBT	X	X	X		X				X				
TFS	X	X	X	X	X	X		X	X			X	
YC1BS	X	X	X	X	X	X		X	X			X	
CRA			X	X	X			X	X			X	
GSF	X	X	X	X	X	X			X				
SQF	X	X	X	X	X	X			X	X			
BLG	X	X	X	X	X	X		X	X	X			
ZOO	X	X	X		X[a]	X	X	X		X		X	
TINS	X	X	X	X	X	X		X	X	X	X	X	
AINS		X	X	X	X	X	X	X	X	X	X	X	
PHY			X		X	X	X		X			X	X
PER						X	X		X	X		X	
DET							X		X				

AINS = aquatic insects
BBS = big bass (.290 mm Alabama spotted bass only)
BLG = bluegill (all sizes)
BRT = brown trout (510 mm)
CHS = chinook salmon (320-538 mm)
CRA = crayfish
DET = detritus (signature estimated from literature)
GSF = green sunfish (83-125 mm)
MBS = medium bass (114-290 mm, Alabama spotted bass and largemouth bass)
MSMB = medium smallmouth bass (114-290 mm)
PER = periphyton (signature estimated from literature)
RBT = rainbow trout (290-410 mm)
SQF = Sacramento squawfish (75-122 mm)
TFS = threadfin shad (all sizes)
TINS = terrestrial insects
YC1BS = year class 1 bass (0-113 mm, all bass species)
ZOO = zooplankton
X indicates that the species in that column was allowed to eat the species in that row
[a] Zooplankton were included in medium bass diet because of stomach analysis; all other rules from literature review

Excel Solver was used to estimate the proportions of diet of all species. Solver is a module included in Microsoft® Excel that allows the user to design an optimization

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model that finds an optimal value for a formula in a target cell by adjusting a group of cells that are related to the formula in that target cell. The following options were used in Solver (Microsoft 1999):

- Nonlinear optimization (generalized reduced gradient)
- Target cell (i.e., sum of squares of differences between measured and estimated values) set to a value of zero
- Precision of solutions was set at 0.000001
- The percentage by which the target cell can differ from the optimal value and still be considered acceptable was set at 5%
- The minimum amount of relative change in the target cell before Solver stops was set to 0.001
- Linear extrapolation from a tangent vector was used to obtain initial estimates from the basic variables in each one-dimensional search
- Forward differencing was used to estimate partial derivatives of the objective and constraint functions
- The Newton method was used at each iteration to determine the direction to search

The objective function for the optimization algorithm was that the sum of the squared differences between estimated and measured signatures be equal to zero:

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$$\sum_{i=1}^{n}\left[\left(\delta^{13}C_i - \delta^{13}C_i'\right)^2 + \left(\delta^{15}N_i - \delta^{15}N_i'\right)^2\right] = 0.0 \qquad \text{Equation 79}$$

where

$\delta^{13}C'_i$ = estimated carbon signature of species i (‰)

$\delta^{15}N'_i$ = estimated nitrogen signature of species i (‰)

$\delta^{13}C_i$ = measured carbon signature of species i (‰)

$\delta^{15}N_i$ = measured nitrogen signature of species i (‰)

n = number of species for which signatures were estimated

This objective formulation was used rather than minimizing the sum of squares of the differences because Solver iterated better by setting the objective function to zero. The cells that were adjusted to meet the objective function were those that contained the proportions of diet (f'_j) according to the diet rules in Table 33 (i.e., those that are indicated with an 'X'), and the estimated enrichment in signature per trophic level (Δ'). Constraints were placed on the diet proportion cells to ensure that the estimated proportions were greater than or equal to zero. Two additional constraints were placed to limit the amount of phytoplankton consumed: 1) the maximum allowable phytoplankton diet proportion for bluegill was 5%, and 2) the maximum allowable phytoplankton diet proportion for crayfish was 15%. Although bluegill and crayfish are known to consume phytoplankton (Emig 1966a; Hobbs 1991; Moyle 1976; Siefert 1972), it is highly unlikely that either of these species consumes very much phytoplankton.

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The proportions of diet of all species were estimated simultaneously such that the estimated signatures most closely matched the measured grouped signatures. Initial conditions were that all f_j's were zero, and Δ's were 0.0‰ for both carbon and nitrogen. These values were allowed to change independently. Solver did not reach a feasible solution, but did iterate from an initial objective function value of 11762.46‰^{2} to 4.12‰^{2}. A copy of the Solver setup and estimated food web is included in Appendix N.

7.5 Estimation of trophic position

Some researchers have assigned discrete trophic levels to species based on literature review or stomach analyses and observed the variation in nitrogen signatures (Forsberg et al. 1993; France 1997a; Fry 1991a, 1991b). Hecky and Hesslein (1995) assigned discrete trophic levels to species in different lakes using nitrogen isotope measurements and an assumed nitrogen trophic enrichment of approximately 3‰. However, Peters (1977) noted that many species are omnivorous and do not fit into discrete trophic levels. In fact, France (1997a) suggested that the increased variation in nitrogen signatures with discrete trophic level could be an indication of increased omnivory with trophic height.

Other studies have suggested that because of omnivory, stable isotopes can be used to determine 'trophic position' on a continuous scale (Kling et al. 1992; Vander Zanden et al. 1997; Vander Zanden and Rasmussen 1996). This trophic position could reflect the relative energetic importance of trophic connections, which Paine (1988) emphasized as a critical element of food web study. Thus, many studies have used nitrogen isotope signatures to estimate trophic positions of species by using assumed

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trophic enrichments (Fry et al. 1999; Hobson and Welch 1995; Kline et al. 1993). Some studies have also assumed a lower trophic level with which to reference other species' signatures (Fry 1988; Hobson et al. 1995; Sydeman et al. 1997; Vander Zanden et al. 1997; Vander Zanden and Rasmussen 1996). Most of the assumed trophic enrichments have been derived from experiments that measured changes in signatures between predators and their diets (e.g., DeNiro and Epstein 1978, 1981; Hesslein et al. 1993; Hobson and Clark 1992; Mizutani et al. 1991; Ostrom et al. 1997; Pinnegar and Polunin 1999; Tieszen et al. 1983).

The estimation of the Shasta Lake food web indicated that there was a high degree of omnivory present. Thus, trophic positions for Shasta Lake species were computed rather than discrete trophic levels. There are at least two approaches for estimating continuous trophic positions using stable isotopes. An approach used by Sydeman et al. (1997) and Vander Zanden et al. (1997) requires the assumption of trophic level of one component of the food web, and relating the trophic level of all remaining components using nitrogen isotope signatures:

$$TP_i' = TP_{known} + \frac{\left(\delta^{15}N_i - \delta^{15}N_{known}\right)}{\Delta_N} \qquad \text{Equation 80}$$

where

TP'_i = estimated trophic position of species i

TP_{known} = assumed trophic position of 'known' component

$\delta^{15}N_i$ = measured nitrogen signature of species i (‰)

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$\delta^{15}N_{known}$ = measured nitrogen signature 'known' component (‰)

Δ_N = trophic enrichment of nitrogen signature (‰)

Another approach requires information about diet proportions in order to estimate trophic position (Vander Zanden et al. 1997; Vander Zanden and Rasmussen 1996):

$$TP_i' = \sum_{j=1}^{n}\left(a_j TP_j' + 1\right) \qquad \text{Equation 81}$$

where

TP'_i = estimated trophic position of species i

a_j = proportion of diet of species i due to species j

TP'_j = estimated trophic position of species j

n = total number of species in diet of species i

The second approach was used at Shasta Lake because trophic position depends on the number of energy-transfer steps and is related to *function* in a food web. Primary producers are assigned to the first trophic position, primary consumers to the second trophic position, and so on (Begon et al. 1996; Krebs 1994). Equation 80 assumes that all species at trophic position 1 would have the same nitrogen signature, although considerable variability in source $\delta^{15}N$ has been observed (France 1995c, 1995d). Alternatively, the assumption of Equation 80 would imply that although phytoplankton, periphyton and detritus are likely primary producers, only one of them could be at trophic

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position 1. In contrast, Equation 81 allows several primary producers to have different nitrogen signatures, or several primary consumers to have different nitrogen signatures. It is likely that different primary producers would have different nitrogen signatures, since they may utilize different nitrogen sources (i.e., nitrate, ammonium, molecular nitrogen, etc.) (Fisher et al. 1994) or exhibit different fractionation rates. Primary consumers would then have different nitrogen signatures depending on their consumption of different primary producers with different nitrogen signatures. In applying Equation 81 for the trophic position estimation at Shasta, phytoplankton, periphyton and detritus were assumed to be primary producers (i.e., trophic position 1), while aquatic and terrestrial insects were assumed to be primary consumers (i.e., trophic position 2). Zooplankton were assumed to consume 100% phytoplankton, which automatically placed them at trophic position 2. The diets for each of these taxa were not estimated with the Solver algorithm. Because cannibalism was allowed in the food web, the solution procedure was iterative.

7.6 Estimation of energy transfer and phytoplankton dependency

Food web pathways that were linked to the phytoplankton were determined using the estimated diet proportions. For example, zooplankton ate 100% phytoplankton, which resulted in one pathway: zooplankton → phytoplankton. Likewise, bluegill ate 34% zooplankton, 5% phytoplankton, and 61% periphyton. This resulted in two pathways to phytoplankton (periphyton and detritus were assumed to have no trophic linkage with phytoplankton):

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bluegill → zooplankton → phytoplankton

bluegill → phytoplankton

Only one species group (medium bass) was estimated as being cannibalistic. This group included Alabama spotted bass and largemouth bass ranging in size from 114 mm to 290 mm. It was assumed that the cannibalism involved the larger bass eating the smaller ones in this size group, so only one iteration of cannibalism was modeled. In other words, the cannibalism was represented in the food web by having the medium bass eat medium bass that had eaten everything the diet analysis had resulted in except other medium bass. In all, the estimated diet proportions resulted in 398 food web pathways that were linked to phytoplankton (Appendix O).

To calculate energy transfer, it was assumed that only 10% of the available productivity at a particular trophic level was available at the next trophic level (Begon et al. 1996; Christensen and Pauly 1993a; DeAngelis 1992; Slobodkin 1968):

$$\text{Trophic transfer efficiency} = \frac{P_n}{P_{n-1}} \approx 0.10 \qquad \text{Equation 82}$$

where

P_n = total productivity available at trophic level n (J)

P_{n-1} = total productivity available at trophic level n-1 (J)

Applying the assumed 10% trophic transfer efficiency along the food web pathways with the diet proportions, the energy transferred from the phytoplankton to the

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other species in the food web was estimated. By transferring the same amount of phytoplankton energy through the food web (i.e., 100 J), the relative energy derived from phytoplankton for each species was determined (see Appendix O).

Harrigan et al. (1989) presented another approach for estimating the dependency of different species on primary organic food sources using measured isotope signatures. They estimated carbon and nitrogen fractionation for each species by examining stomach contents and measuring signatures of both predator and prey. They also assumed integer trophic levels for their consumers. Using the assumed trophic levels and measured fractionations, the measured isotope signatures for consumer organisms were extrapolated back to primary organic source (POS) signatures. The POS signatures were then used to solve for the percent contribution of one to three POS, which included primary producers, sedimentary organic matter and particulate organic matter from the water column (POM). A similar approach was applied by Ostrom et al. (1997) to estimate the diet contributions of alfalfa, wheat and maize to the ladybird beetle *Coleomegilla maculata lengi*. In their case, however, they also estimated the trophic fractionation between *C. maculata* and the plant mixture with a diet experiment.

A similar approach to that used by Harrigan et al. (1989) was applied with the Shasta Lake isotope signatures, but the Solver-estimated trophic enrichments and the continuous trophic positions estimated using Equation 81 were used to apply the technique. In other words, the measured signatures for each species were used to estimate the combined signature of primary organic food sources by back-calculating according to the following equations:

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$$\delta^{13}C'_{\text{prim},i} = \delta^{13}C_i - (TL_i - 1)\Delta'_C \quad \text{Equation 83}$$

$$\delta^{15}N'_{\text{prim},i} = \delta^{15}N_i - (TL_i - 1)\Delta'_N \quad \text{Equation 84}$$

where

$\delta^{13}C'_{\text{prim},i}$ = estimated carbon signature of all primary organic sources for species i (‰)

$\delta^{15}N'_{\text{prim},i}$ = estimated nitrogen signature of all primary organic sources for species i (‰)

$\delta^{13}C_i$ = measured carbon signature for species i (‰)

$\delta^{15}N_i$ = measured nitrogen signature for species i (‰)

TL_i = estimated trophic level for species i

Δ'_C = estimated trophic enrichment of carbon signature from Excel Solver (‰)

Δ'_N = estimated trophic enrichment of nitrogen signature from Excel Solver (‰)

The primary organic source signatures of each species were estimated in a spreadsheet according to the following three-source mass balances (Harrigan et al. 1989):

$$\delta^{13}C''_{\text{prim},i} = f'_{\text{phyto},i}\left(\delta^{13}C'_{\text{phyto}}\right) + f'_{\text{peri},i}\left(\delta^{13}C'_{\text{peri}}\right) + f'_{\text{det},i}\left(\delta^{13}C'_{\text{det}}\right) \quad \text{Equation 85}$$

$$\delta^{15}N''_{\text{prim},i} = f'_{\text{phyto},i}\left(\delta^{15}N'_{\text{phyto}}\right) + f'_{\text{peri},i}\left(\delta^{15}N'_{\text{peri}}\right) + f'_{\text{det},i}\left(\delta^{15}N'_{\text{det}}\right) \quad \text{Equation 86}$$

$$f'_{\text{phyto},i} + f'_{\text{peri},i} + f'_{\text{det},i} = 1 \quad \text{Equation 87}$$

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where

$\delta^{13}C''_{prim,i}$ = estimated carbon signature of primary organic sources for species i (‰)

$\delta^{15}N''_{prim,i}$ = estimated nitrogen signature of primary organic sources for species i (‰)

$\delta^{13}C'_{phyto}$ = estimated carbon signature of phytoplankton (see Table 36)(‰)

$\delta^{15}N'_{phyto}$ = estimated nitrogen signature of phytoplankton (see Table 36) (‰)

$\delta^{13}C'_{peri}$ = estimated carbon signature of periphyton (see Table 36) (‰)

$\delta^{15}N'_{peri}$ = estimated nitrogen signature of periphyton (see Table 36) (‰)

$\delta^{13}C'_{det}$ = estimated carbon signature of detritus (see Table 36) (‰)

$\delta^{15}N'_{det}$ = estimated nitrogen signature of detritus (see Table 36) (‰)

$f'_{phyto,i}$ = estimated fraction of diet of species i that is phytoplankton

$f'_{peri,i}$ = estimated fraction of diet of species i that is periphyton

$f'_{det,i}$ = estimated fraction of diet of species i that is detritus

These equations were simultaneously solved using Excel Solver with the objective that the sum of the squares of the differences between $\delta^{13}C'_{prim,i}$ and $\delta^{13}C''_{prim,i}$, and between $\delta^{15}N'_{prim,i}$ and $\delta^{15}N''_{prim,i}$ be zero:

$$\sum_{i=1}^{n}\left[\left(\delta^{13}C'_{prim,i}-\delta^{13}C''_{prim,i}\right)^2+\left(\delta^{15}N'_{prim,i}-\delta^{15}N''_{prim,i}\right)^2\right]=0.0 \qquad \text{Equation 88}$$

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All cells containing f' values were adjusted to meet the objective function. These cells were constrained to being greater than or equal to zero, and the sum of the fractions from primary sources for each species was constrained to equal 1 according to Equation 87. For the initial condition with all $f' = 0.0$, the objective function value was $12912.37(‰)^2$. Solver did not reach a feasible solution, but iterated to an objective function value of $0.39(‰)^2$. The constraint of Equation 81 was not satisfied for rainbow trout, green sunfish, and terrestrial and aquatic insects, but none of these species were expected to consume phytoplankton (see Appendix P for the Solver setup and solution).

7.7 Linkage of food web-energy transfer model with CE-QUAL-W2

In order to investigate the effects of changes in primary productivity due to TCD operations on sport fish species, the food web-energy transfer model needed to be linked with the algae predictions from CE-QUAL-W2. Because energy transfer was defined in terms of production (see Equation 82), the net primary production due to phytoplankton needed to be estimated. At the primary production level (i.e., trophic position 1), the total productivity available is the net production, or the gross production minus plant respiration (Begon et al. 1996; Goldman 1968).

In CE-QUAL-W2, the change in algae concentration for each timestep and each cell in the model is calculated according to Equation 21 in Chapter 5 (Cole and Buchak 1995). The algae concentration is in terms of grams organic matter (i.e., ash-free) dry weight per m^3 (Cole 1999). Typically, CE-QUAL-W2 outputs Φ_a for each cell at the desired output intervals, but to estimate net production, the model program was modified

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to also output $K_{ag}\Phi_a$, and $K_{ar}\Phi_a$ so that net algal production in grams organic matter dry weight for each cell was estimated as:

$$\text{Net algal production} = \left(K_{ag}\Phi_a - K_{ar}\Phi_a\right)\Delta t V \qquad \text{Equation 89}$$

where

Φ_a = algal concentration ($g\ m^{-3}$)

K_{ag} = algal growth rate (s^{-1})

K_{ar} = algal dark respiration rate (s^{-1})

Δt = length of time interval of output (s)

V = volume of cell (m^3)

Net algal production was calculated for the whole reservoir (i.e., the net production was summed over all active model cells), and for the epilimnetic region of the reservoir. Cells were assigned to the epilimnion if they were above the computed thermocline for Segment 19. This segment was considered representative of the main lake area of the reservoir. The thermocline was defined as the highest point in the water column at which the water temperature differed from the surface water temperature by 1°C (Bartholow et al., in review).

Wet and dry hydrologic scenarios and the 1995 conditions were run both with the TCD operating and without. For each of the six runs, the following outputs were generated eight times per day (i.e., Δt = 10800 s):

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- Thermocline depth (m)
- Whole lake average algae concentration (g m^{-3})
- Number of cells used to compute the whole lake average algae concentration
- Epilimnetic average algae concentration (g m^{-3})
- Number of cells used to compute the epilimnetic average algae concentration
- Top 30-m average algae concentration (g m^{-3})
- Number of cells used to compute the top 30-m average algae concentration
- Segment 19 average algae concentration (g m^{-3})
- Number of cells used to compute the Segment 19 average algae concentration
- Whole lake total mass of algae (g)
- Whole lake volume (m^3)
- Epilimnetic total mass of algae (g)
- Epilimnetic lake volume (m^3)
- Top 30-m total mass of algae (g)
- Top 30-m lake volume (m^3)
- Whole lake areal algae (g m^{-2})
- Whole lake areal chlorophyll *a* (g m^{-2})
- Whole lake gross production (μg s^{-1})*
- Whole lake dark respiration (μg s^{-1})*
- Whole lake net production (μg s^{-1})*
- Epilimnetic gross production (μg s^{-1})*
- Epilimnetic dark respiration (μg s^{-1})*
- Epilimnetic net production (μg s^{-1})*

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- Surface area (m^2)
- Whole lake areal net production ($\mu g\ m^{-2}\ s^{-1}$)*
- Whole lake volumetric net production ($\mu g\ m^{-3}\ s^{-1}$)*
- Epilimnetic areal net production ($\mu g\ m^{-2}\ s^{-1}$)*
- Epilimnetic volumetric net production ($\mu g\ m^{-3}\ s^{-1}$)

Each of the items designated with an asterisk(*) were numerically integrated over each day using Simpson's 1/3 rule (Chapra and Canale 1988) to get daily values of algal production. The daily results of the with-TCD runs for each pair of conditions were subtracted from the corresponding results for the without-TCD runs to calculate the change in algal production.

Effects of changes in phytoplankton energy due to TCD operation were only evaluated over the period in which the reservoir was stratified for each run because this is the period in which the largest changes in production were expected to occur. Also, the lower water temperatures during the mixing period correspond with slower growth rates for rainbow trout and smallmouth bass (see Figures 20 and 21 in Chapter 8). Thus, fish were expected to experience the most growth during the stratified period. The stratified period was defined as a continuous period in which the thermocline depth was always less than 50 m deep (Bartholow et al. in review). For each of the six runs, this period was identified, and the daily net production was summed over the corresponding period.

To convert the net primary production in grams to energy units, a factor of 5080 calories per gram ash-free dry weight was applied to the changes in net production that were predicted by CE-QUAL-W2 over the stratified period. This factor was determined

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by averaging reported conversion values in Cummins and Wuycheck (1971) for algae species reported to be at Shasta by Lieberman and Horn (1998). The spreadsheet used to calculate the conversion factor is included in Appendix Q.

The estimated energy available due to phytoplankton net production during the stratified period was used to estimate the wet biomass of fish that could be supported by using the following equation:

$$\text{Biomass}_i = \Delta E_{\text{phyto}} \left(4.168 \text{ J cal}^{-1}\right) \frac{ET_i}{ED_i} \qquad \text{Equation 90}$$

where

Biomass_i = estimated wet biomass of species i (g)

ΔE_{phyto} = change in energy due to phytoplankton net production (cal)

ET_i = energy transfer coefficient from phytoplankton to species i (see Table 37)

ED_i = energy density of species i (J g^{-1})

Estimates of energy density were obtained from Hanson et al. (1997). For chinook salmon and brown trout, this required the use of the following equation (Hanson et al. 1997):

$$ED_i = \alpha_i + \beta_i W_i \qquad \text{Equation 91}$$

where

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α_i = intercept of allometric mass function (J g^{-1})

β_i = slope of allometric mass function

W_i = fish mass (g)

Unfortunately, the weight of all of the chinook salmon (TL range: 320-539 mm) or the brown trout (TL 510 mm) was not measured. The weight was measured for two of the smaller chinook salmon, so the fish mass for chinook salmon was estimated as the average of these measured weights (299 g). The weight for the brown trout was estimated as 2000 g based on measured weights of Alabama spotted bass of similar size. Therefore, Table 34 shows the values used to calculate the energy densities of chinook salmon and brown trout:

Table 34. Coefficients, weight and calculated energy densities for brown trout and chinook salmon

Species	α_i (J g^{-1})[a]	β_i[a]	W_i (g)	ED_i (J g^{-1})
Brown trout	5764	0.9862	299	6059
Chinook salmon	5764	0.9862	2000	7736

[a]Coefficient values obtained from Hanson et al. (1997)

Energy densities for all bass species and size classes and bluegill were set at 4186 J g^{-1} (Hanson et al. 1997), and the zooplankton energy density used was the average for cladocera (2514 J g^{-1}) given in Hanson et al. (1997). Rainbow trout were not included in the analysis because they are not energetically dependent on phytoplankton. Other

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species were not included because energy density values were not available for them in Hanson et al. (1997). Calculations are included in Appendix R.

No calibration was done for the food web model because data was not available for comparisons with model predictions.

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Chapter 8. RESULTS

8.1 Thermal effects of the TCD

Conclusions of the thermal analysis with CE-QUAL-W2 were that changes in in-reservoir thermal patterns due to TCD operation were not significantly affected by different hydrologic conditions. As shown in Figure 17 for Segment 19 under 1968 hydrologic year conditions, epilimnetic effects of TCD operation were typically negligible, but hypolimnetic waters were about 1°C cooler in the summer and up to 5°C warmer in the later fall with TCD operation when compared to without-TCD conditions. Segment 19 is representative of conditions in the main reservoir. In-reservoir thermal effects of TCD operation persisted up to 20 km upstream of the dam (Figure 18). The multi-year analyses demonstrated that in-reservoir temperature changes did not carry over into subsequent years with continued TCD use. Using the pseudo-optimization model to generate complex TCD operation schemes that mixed discharges from multiple shutter outlets on a daily basis, required outlet temperatures were met throughout the year for some hydrologic conditions. Simpler adjustments of release elevations were not able to meet all of the discharge temperature targets, although the number of days in which they were met was increased with TCD operation (Hanna et al. 1999).

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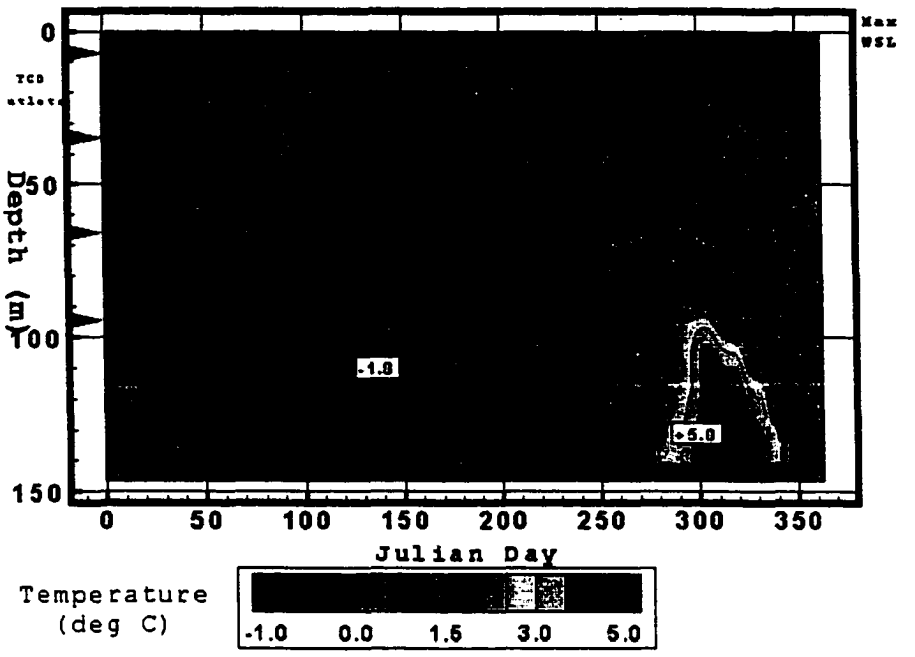

Figure 17. 1968 temperature differences (with TCD – without TCD) at Segment 19.

Reservoir depths are from maximum water surface level (Max WSL) for Shasta Lake. Depths of TCD outlets are indicated by red triangles.

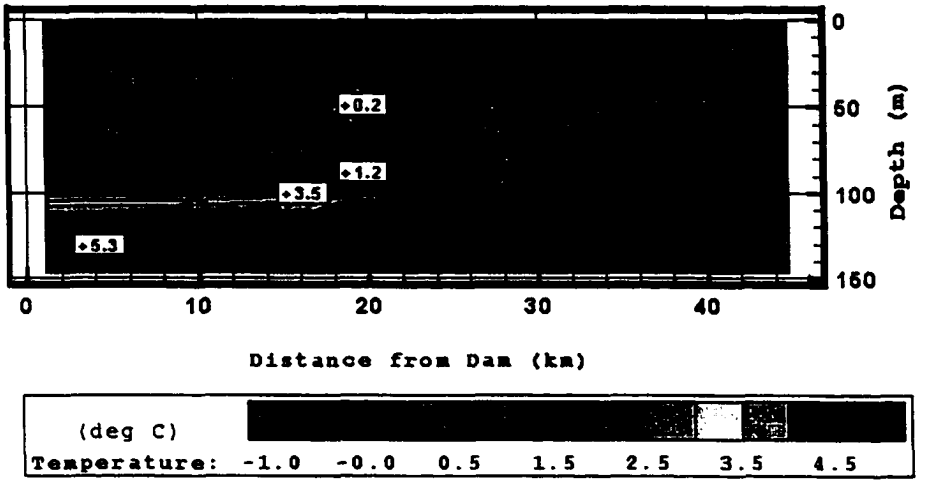

Figure 18. 1968 differences in temperatures (with TCD – without TCD) along Pit River arm on Julian day 312 (November 7).

Shasta Dam is at left, and uppermost segments on the Pit River arm are at right

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8.2 Nutrient and phytoplankton effects of the TCD

The MVT analysis of the seven variables (initial storage, inflows, outflows, nutrient loading, air temperature, cloud cover and TCD operation) revealed that overall, the TCD had the least effect on the 50 metrics evaluated (Figure 19). The changes in the 50 metrics due to TCD operation are shown in Table 35 (Bartholow et al. in review). See Bartholow et al. (in review) for further details on the application of CE-QUAL-W2 to assessing TCD effects on nutrients and phytoplankton.

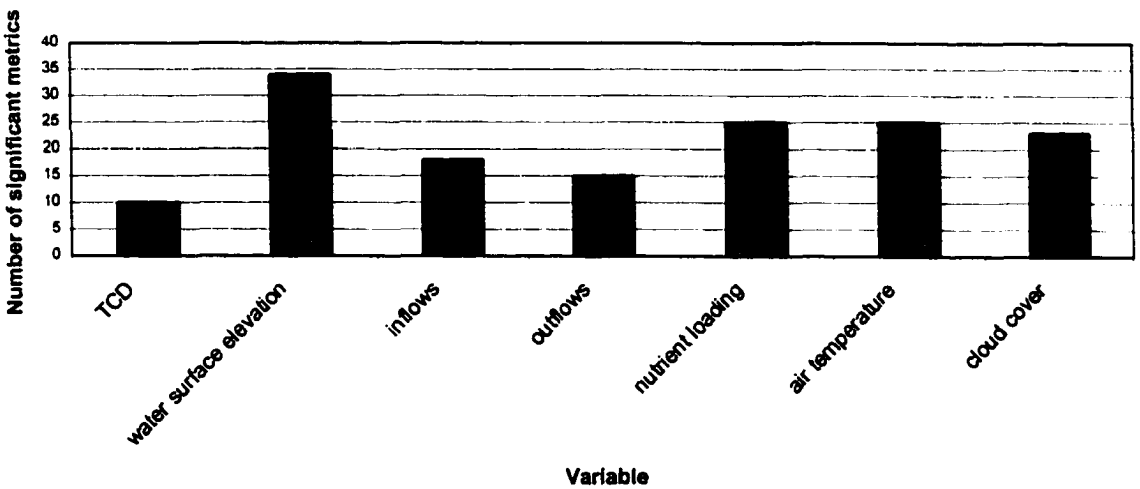

Figure 19. Number of significant metrics for each of the variables analyzed

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Table 35. Effects of TCD for 50 metrics

Metric	TCD effect
Stratification	
Start of stratification (JD)	-5.00
End of stratification (JD)	-16.38[a]
Number of stratified days (d)	-11.38
Degree days over target temperature (dd)[c]	
Thermocline depth (m)	
Before stratification [JD 70-140]	-1.72
While stratified [JD 141-260]	-0.02
After stratification [JD 261-350]	+12.87[a]
Mean annual	-0.69
Summer stratification strength (maximum RTR)[b]	-1.76
Vertical mixing strength (minimum RTR after JD 10)[b]	-0.45
Top 30 m average temperature (°C)	
Maximum	0.00
Minimum	-0.06[a]
Average annual	0.00
Whole lake average temperature (°C)	
Maximum	+0.35
Minimum	-0.09[a]
Annual average	+0.02
Degree days over target (dd)[c]	
Entire year	-135.3[a]
September-October	-43.9
Minimum whole lake average dissolved oxygen (% change)	-0.36
Minimum annual outfall dissolved oxygen (% change)	-2.77
Average epilimnetic nitrate-nitrite (% change)	
Jan-Apr [JD 0-120]	-1.10[a]
May-Oct [JD 121-300]	+6.72
Nov-Dec [JD 300-360]	+2.32
Average epilimnetic soluble reactive phosphorus (% change)	
Jan-Apr [JD 0-120]	+0.23
May-Oct [JD 121-300]	+1.87
Nov-Dec [JD 300-360]	+0.22
Average epilimnetic ammonium (% change)	
Jan-Apr [JD 0-120]	+3.40[a]
May-Oct [JD 121-300]	+1.91
Nov-Dec [JD 300-360]	+2.21
Average hypolimnetic nitrate-nitrite (% change)	
Jan-Apr [JD 0-120]	+5.69
May-Oct [JD 121-300]	-0.45
Nov-Dec [JD 300-360]	-16.48
Average hypolimnetic soluble reactive phosphorus (% change)	
Jan-Apr [JD 0-120]	+7.23
May-Oct [JD 121-300]	-0.51
Nov-Dec [JD 300-360]	-7.72
Average hypolimnetic ammonium	
Jan-Apr [JD 0-120]	+10.19
May-Oct [JD 121-300]	+0.54
Nov-Dec [JD 300-360]	-3.68
Average outfall nitrite-nitrate (% change)	
Jan-Apr [JD 0-120]	-2.22
May-Oct [JD 121-300]	+1.11
Nov-Dec [JD 300-360]	+0.90
Average outfall soluble reactive phosphorus	
Jan-Apr [JD 0-120]	-1.36
May-Oct [JD 121-300]	+1.12
Nov-Dec [JD 300-360]	+1.88
Average outfall ammonium (% change)	
Jan-Apr [JD 0-120]	-5.21
May-Oct [JD 121-300]	+20.10[a]
Nov-Dec [JD 300-360]	+20.46[a]
Average outfall algal biomass (% change)	
Spring: Mar-Jun [JD 61-182]	+1648.4[a]
Fall: Sep-Nov [JD 245-335]	-42.46
Maximum whole lake average algal biomass concentration (% change)	
Spring: Mar-Jun [JD 61-182]	-3.54
Fall: Sep-Nov [JD 245-335]	+10.61[a]

[a] Statistically significant difference ($p < 0.05$)

[b] RTR = relative thermal resistance = $\frac{(\rho_{bottom} - \rho_{top})}{(\rho_{4°C} - \rho_{5°C})}$, where ρ = density (g m^{-3}) (Wetzel 1983)

[c] One degree day (dd) is equivalent to being 1°C over temperature target for one day

Source: Bartholow et al. (in review)

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8.3 Bioenergetics results

All predictions of scope for growth were made for 100 g rainbow trout or smallmouth bass, with the assumption that they ate only fish that were 100% digestible with energy densities of 4200 J g^{-1}. Scope for growth curves in relation to temperature were constructed for each species to verify the equations and coefficients used in the model with predictions from Hanson et al. (1997). Maximum scope for growth for a 100 g rainbow trout (298 J g^{-1} d^{-1} or 0.071 g g^{-1} d^{-1}) was predicted to occur at about 15°C (Figure 20). In contrast, the smallmouth bass prefers warm water habitats, and the maximum scope for growth for a 100 g bass (84 J g^{-1} d^{-1} or 0.020 g g^{-1} d^{-1}) was predicted to occur at about 28°C (Figure 21).

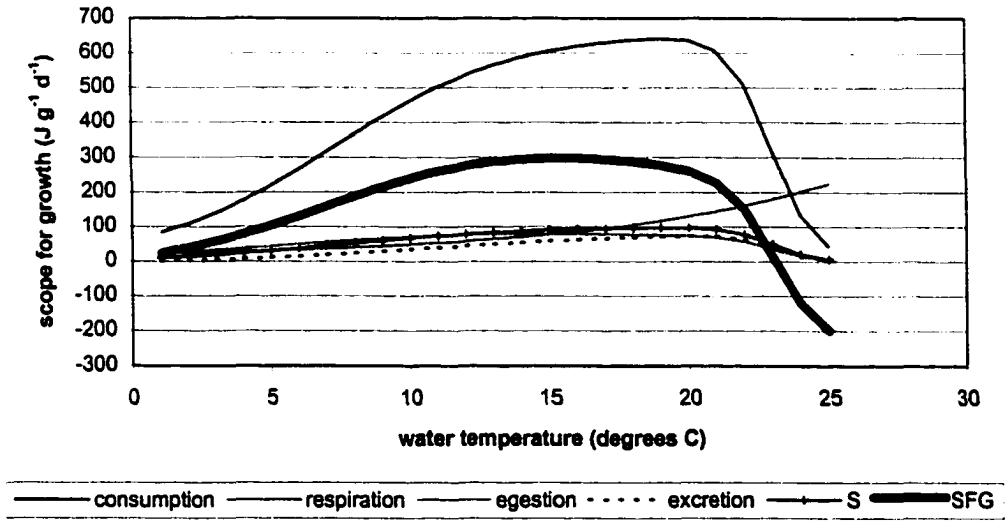

Figure 20. Scope for growth versus water temperature for 100 g rainbow trout

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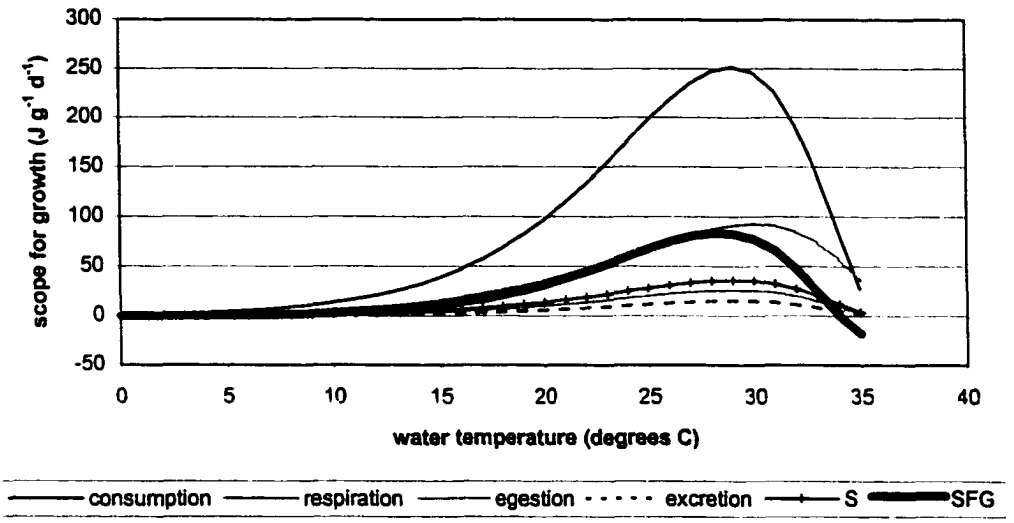

Figure 21. Scope for growth versus water temperature for 100 g smallmouth bass

After linking the water temperatures from CE-QUAL-W2 with the bioenergetics model, changes in scope for growth were assessed by subtracting without-TCD results from with-TCD results. Little change in scope for growth for both rainbow trout (Figure 22) and smallmouth bass was predicted between with- and without-TCD conditions until the lowest TCD gate was opened around Julian day 250. The largest difference in percentage of maximum scope for growth occurred around Julian day 312 (November 9) at a depth of about 100 m. In the 1968 simulations, TCD operations resulted in increasing the percentage of maximum scope for growth by as much as 32 % for rainbow trout, and almost 7 % for smallmouth bass. This maximum change was observed within about 10 km of the dam (Figure 23) (Hanna et al. 1999). Similar effects were seen in the other calendar years evaluated.

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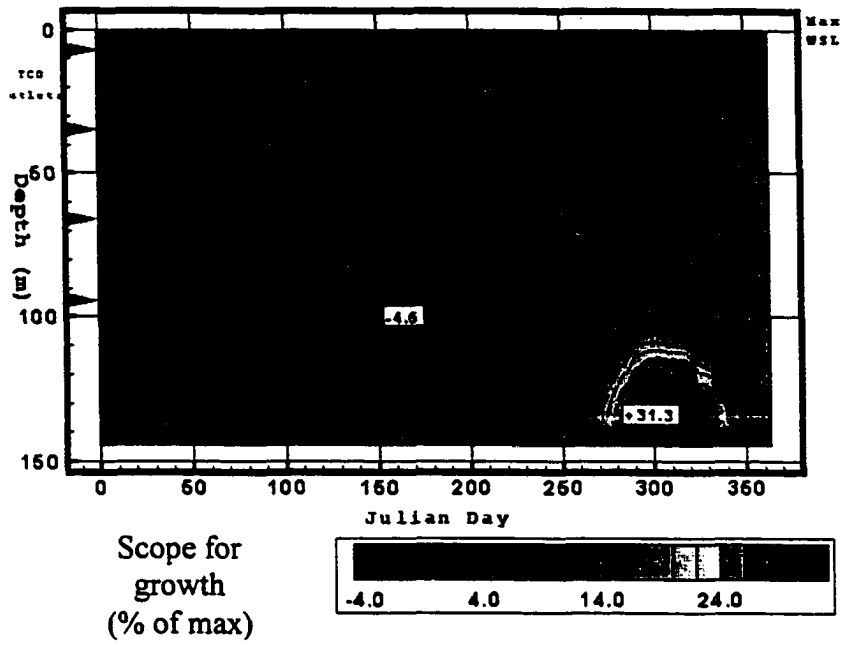

Figure 22. Difference (with TCD – without TCD) in 1968 percent of maximum scope for growth at Segment 19 for a 100 gram rainbow trout.

Reservoir depths are from maximum water surface level (Max WSL) for Shasta Lake. Depths of TCD outlets are indicated by red triangles.

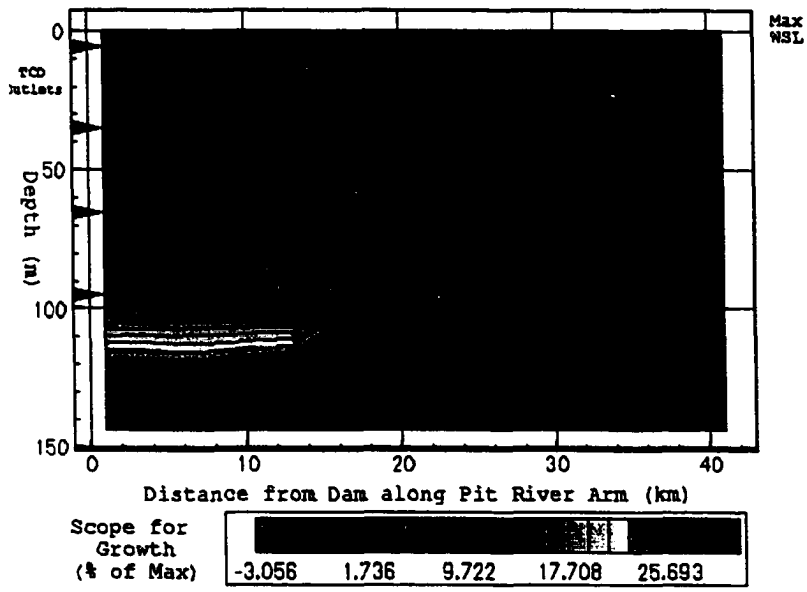

Figure 23. Difference (with TCD – without TCD) in 1968 percent of maximum scope for growth along the Pit River arm for a 100 gram rainbow trout on Julian day 312 (November 7).

Shasta Dam is at left and uppermost segments on Pit River arm are at right.

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8.4 Estimated food web

Table 36 and Figure 24 show the resulting estimated signatures along with the corresponding measured signatures and estimated trophic positions.

Table 36. Measured and estimated carbon and nitrogen signatures, and estimated trophic positions for grouped species at Shasta Lake

Species	$\delta^{13}C$ (‰)				$\delta^{15}N$ (‰)				Estimated trophic position
	Measured range	n	Avg.	Esti-mated	Measured range	n	Avg.	Esti-mated	
Chinook salmon (320-538 mm)	-28.22 to -27.60	2	-27.91	-27.91	11.60 to 12.10	2	11.85	11.86	4.4
Big bass (ASB > 290 mm)	-28.70 to -25.86	1	-27.32	-27.35	9.87 to 11.33	6	10.37	10.37	4.0
Brown trout (510 mm)	-29.02	6	-29.02	-29.02	10.54	1	10.54	10.52	3.9
Medium SMB (114-290 mm)	-27.69 to -26.99	2	-27.34	-27.32	9.71 to 10.13	2	9.92	9.91	3.9
Rainbow trout (290-410 mm)	-24.33	1	-24.33	-25.36	9.46	1	9.46	9.38	3.9
Medium bass (ASB, LMB 114-290 mm)	-29.44 to -27.88	9	-28.58	-28.56	9.03 to 10.52	9	9.71	9.70	3.7
Crayfish	-31.09	1	-31.09	-30.02	7.32	1	7.32	7.60	3.2
Green sunfish (83-125 mm)	-26.17 to -25.49	3	-25.84	-25.85	6.74 to 7.51	3	7.23	7.23	3.1
Threadfin shad (all sizes)	-29.94 to -31.21	3	-30.41	-30.81	8.92 to 9.67	3	9.18	8.13	3.0
Year class 1 bass (0-113 mm)	-32.01 to -27.23	9	-30.35	-30.11	5.74 to 8.69	9	7.34	7.86	3.0
Sacramento squawfish (75-122 mm)	-27.33 to -25.67	2	-26.50	-26.34	6.45 to 7.49	2	6.97	6.94	3.0
Bluegill (all sizes)	-32.11 to -24.55	7	-29.34	-29.16	5.81 to 7.19	7	6.69	7.10	2.3
Zooplankton[a]	-35.43 to -32.52	5	-34.31	-34.31	5.69 to 6.97	5	6.30	6.30	2.0
Terrestrial insects[a]	-29.01 to -24.59	4	-26.75	-26.75	2.79 to 5.77	4	3.94	3.94	2.0
Aquatic insects[a]	-27.41 to -22.52	6	-25.49	-25.49	1.68 to 4.52	4	2.96	2.96	2.0
Phytoplankton[a,b]	nm		nm	-34.72	nm		nm	3.30	1.0
Periphyton[b, c]	nm		nm	-26.51	nm		nm	2.94	1.0
Detritus[b, c]	nm		nm	-28.05	nm		nm	-1.18	1.0

ASB = Alabama spotted bass
LMB = largemouth bass
SMB = smallmouth bass

$\Delta'_C = 0.41$ ‰ ; $\Delta'_N = 3.0$ ‰

[a] These signatures and trophic levels were not estimated using Solver
[b] Phytoplankton signature was calculated from the zooplankton measured signature according to the estimated changes in signature per trophic level
[c] Signatures for periphyton and detritus were estimated from literature review

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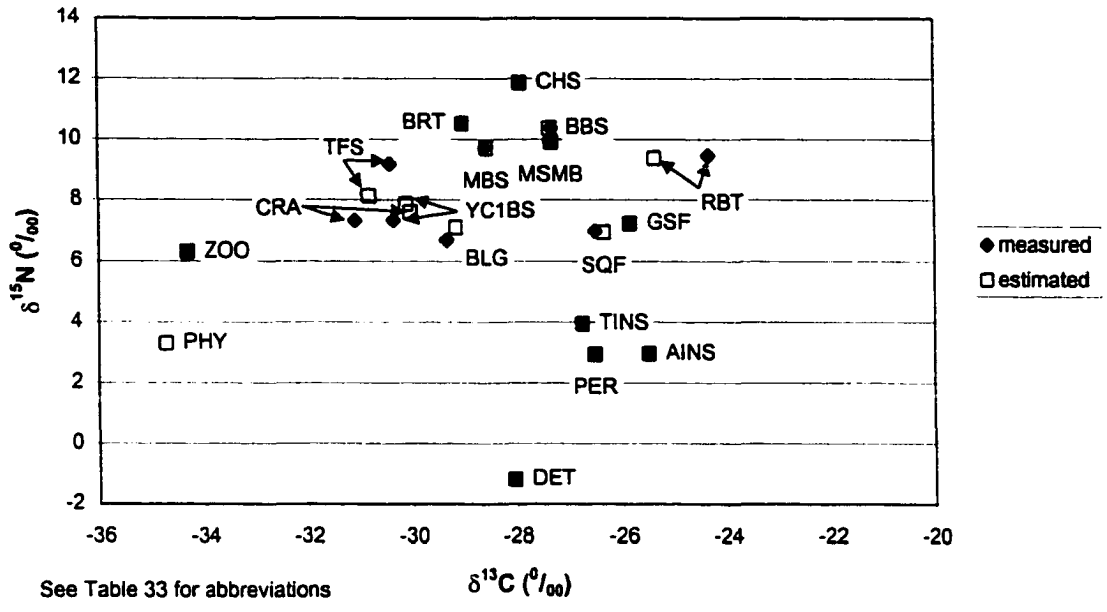

Figure 24. Measured and estimated signatures for Shasta Lake grouped species

8.5 Phytoplankton dependency

The energy available to Shasta Lake food web species from phytoplankton is shown in Table 37 and Figure 25. It is evident that the rainbow trout are not dependent on phytoplankton energy at all. In comparing energy derived from 100 J of phytoplankton energy, the trophic position of species should be considered. Species at higher computed trophic positions were estimated to have more trophic transfers of energy to get the phytoplankton energy than those at lower trophic positions. Thus, comparisons of energy dependence on phytoplankton can only be made for species with the same (or similar) calculated trophic position. The food web model therefore predicts that amongst the sport fish species, the bass are the most dependent on phytoplankton-derived energy.

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Table 37. Energy available to Shasta species from phytoplankton

Species	Energy available per 100 J phytoplankton energy (J)	Trophic position
Zooplankton	10.00	2.0
Bluegill	0.84	2.3
Sacramento squawfish	0.00	3.0
Year class 1 bass	0.57	3.0
Threadfin shad	0.65	3.0
Green sunfish	0.00	3.1
Crayfish	1.53	3.2
Medium bass	0.92	3.7
Rainbow trout	0.00	3.9
Medium smallmouth bass	0.03	3.9
Brown trout	0.23	3.9
Big bass	0.65	4.0
Chinook salmon	0.12	4.4

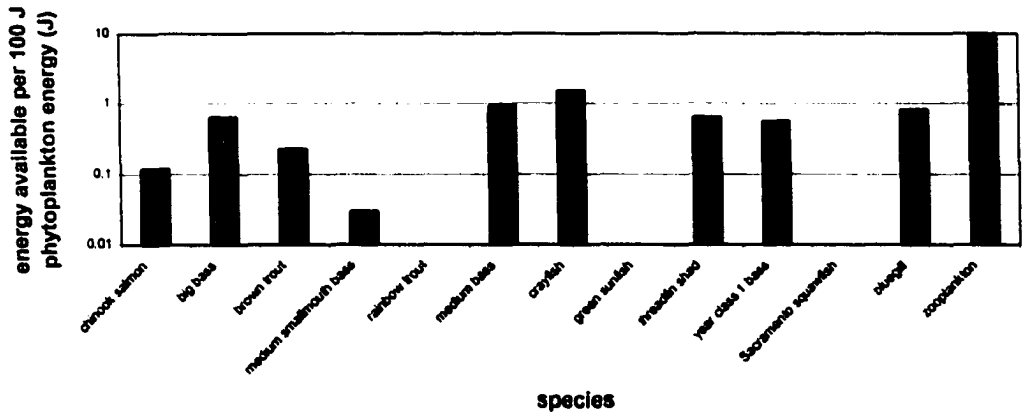

Figure 25. Energy available to Shasta species from phytoplankton

The estimates of proportions of species' diets derived from phytoplankton as calculated by the Harrigan et al. (1989) method are shown in Figure 26.

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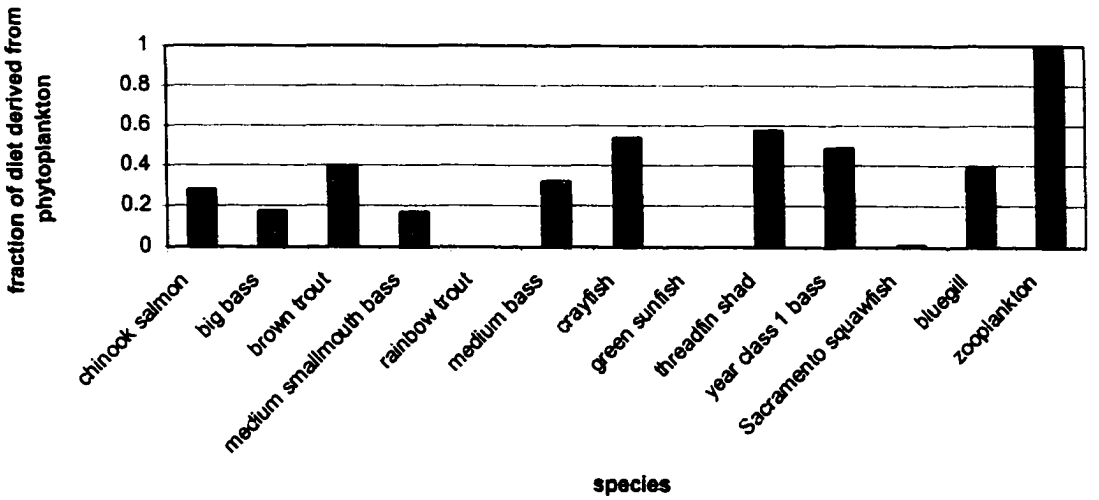

Figure 26. Fraction of species' diets due to phytoplankton-derived sources

8.6 Linkage of food web-energy transfer model with CE-QUAL-W2

A complete set of figures showing CE-QUAL-W2 and the linked modeling results is included in Appendix R. Figures 27 and 28 illustrate the hydrologic and thermal differences predicted by CE-QUAL-W2 between the three runs under without-TCD conditions. As shown, 1995 started out with hydrologic conditions similar to the dry scenario, but an early storm quickly brought the reservoir volume closer to hydrologically wet conditions. The hydrologically dry conditions caused the reservoir volume to fall in the latter part of the year, while a high reservoir volume was maintained throughout the year under wetter conditions. In addition, both 1995 and the hydrologically wet scenario showed an increase in epilimnetic water later in the year as the reservoir mixed, but such water was not available under hydrologically dry conditions.

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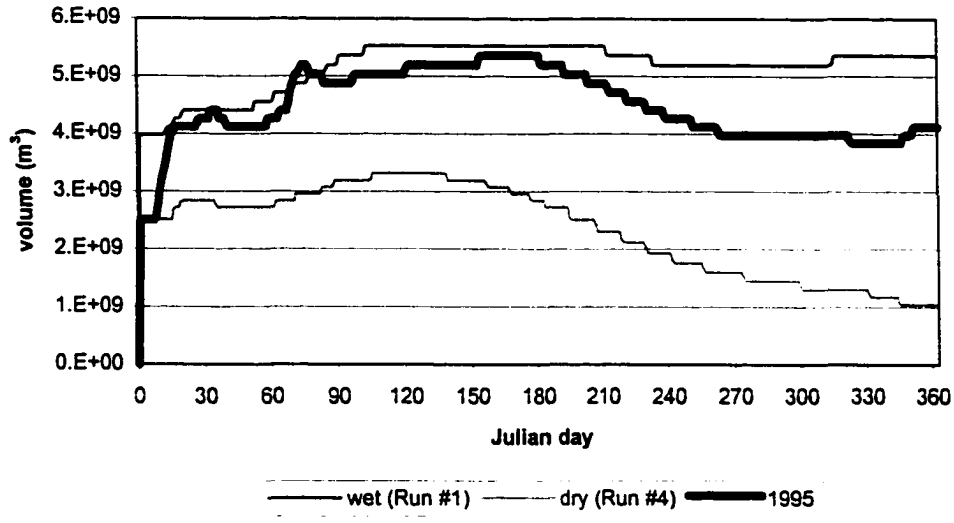

Figure 27. CE-QUAL-W2 predictions of whole lake volumes without TCD operating

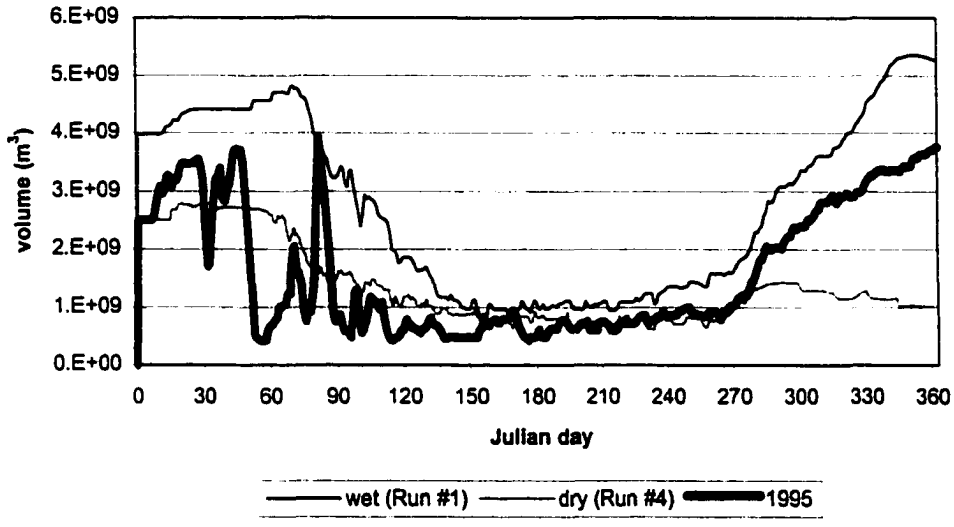

Figure 28. CE-QUAL-W2 predictions of epilimnetic volumes without TCD operating

CE-QUAL-W2-predicted changes in estimated daily algal net production in the epilimnion (Figures 29 and 30) were calculated as with-TCD conditions minus without-TCD conditions. This means that negative values are *losses* in net production with the

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TCD operating, while positive values correspond with *gains* in net production. Figure 29 shows that TCD operation under the wet scenario resulted in overall greater net production in the epilimnion, while the opposite was true for the dry scenario. The effects under 1995 conditions were much less throughout the year than either of the other two scenarios.

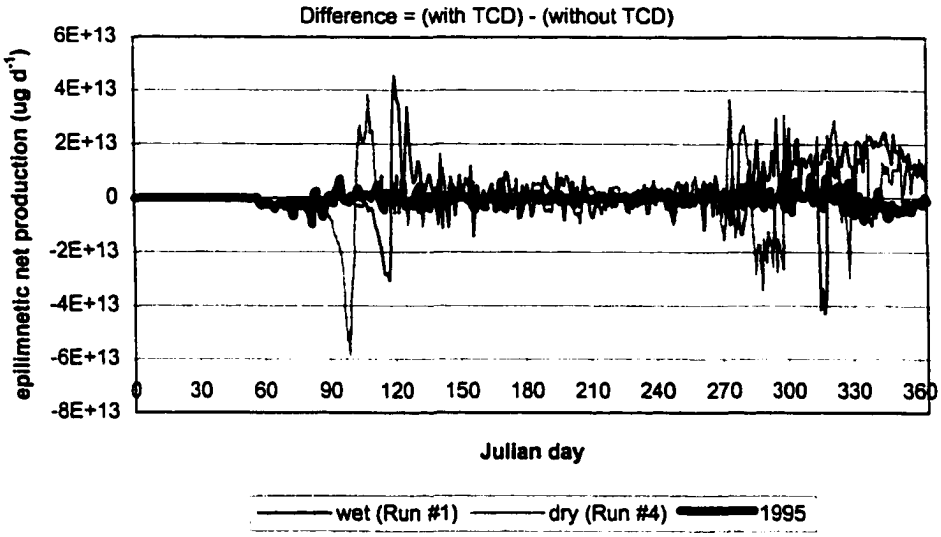

Figure 29. Epilimnetic differences in net production between without TCD and with TCD conditions as predicted by CE-QUAL-W2

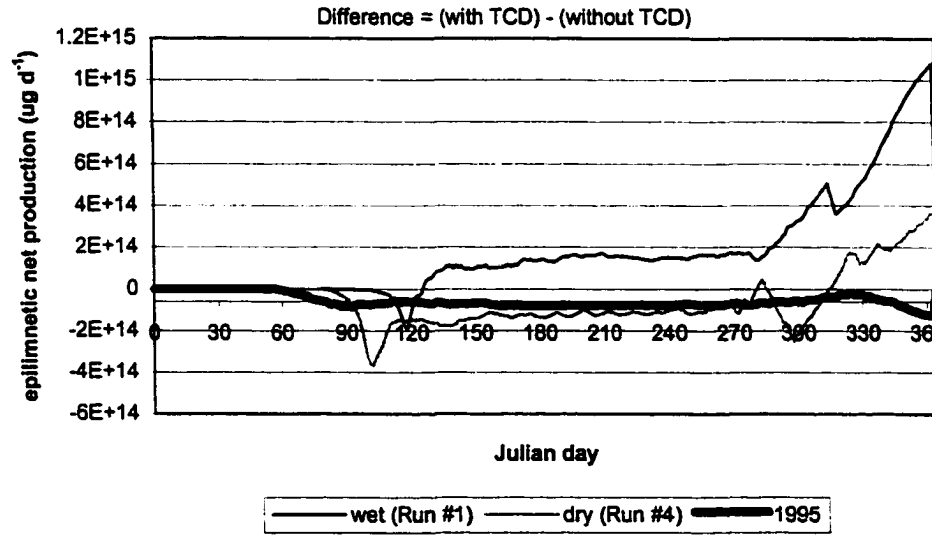

Figure 30. Cumulative epilimnetic differences in net production between with TCD and without TCD conditions as predicted by CE-QUAL-W2

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It is evident from Table 38 and Figure 31 that the hydrologically wet conditions resulted in increases in net production over the stratified period, while hydrological dry conditions resulted in less net production with the TCD operating. The results for 1995 showed an overall increase in net production in the whole lake with the TCD operating, but only a very slight increase in epilimnetic net production. In fact, all of the scenarios showed that hypolimnetic net production must increase with TCD operation, since gains in the whole lake were greater than gains in the epilimnion during the wet and 1995 scenarios, and losses in the whole lake were less than losses in the epilimnion during the dry scenario. This indicates that net production must increase in the hypolimnion under with-TCD conditions, possibly due to warmer hypolimnetic temperatures and increased mixing. Hanna et al. (1999) showed that hypolimnetic water temperatures were warmer later in the year with TCD operation, and these warmer temperatures could cause the model to predict increased algal production during this period. Also, the stratified period for both the hydrologically wet and hydrologically dry scenarios increased with TCD operation, so that the number of days over which the net production shown in Table 38 was calculated was greater.

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Table 38. Estimated net production over stratified period with and without the TCD operating for hydrologically wet, hydrologically dry and 1995 conditions as predicted by CE-QUAL-W2

Run	Start of stratified period (JD[a])	End of stratified period (JD[a])	Number of stratified days (d)	Whole lake net production (g)	Epilimnetic net production (g)	Epilimnetic areal net production (g m^{-2})
Hydrologically wet						
Without TCD	84	321	238	8.86×10^{9}	9.83×10^{9}	89.6
With TCD	83	325	243	9.66×10^{9}	1.06×10^{10}	96.5
Difference[b]			5	8.09×10^{8}	7.53×10^{8}	6.9
Percent difference[c]			2.10%	9.13%	7.66%	7.74%
Hydrologically dry						
Without TCD	71	278	208	3.37×10^{9}	4.02×10^{9}	56.7
With TCD	65	276	212	3.27×10^{9}	3.86×10^{9}	55.6
Difference[b]			4	-9.82×10^{7}	-1.53×10^{8}	-1.1
Percent difference[c]			1.92%	-2.91%	-3.82%	-1.06%
1995						
Without TCD	85	328	244	5.97×10^{9}	5.68×10^{9}	58.7
With TCD	84	326	243	6.17×10^{9}	5.71×10^{9}	59.0
Difference[b]			-1	1.98×10^{8}	3.05×10^{7}	0.3
Percent difference[c]			-0.41%	3.32%	0.54%	0.48%

[a]JD = Julian day

[b]Difference was calculated as (with TCD condition) – (without TCD condition)

[c]Percent difference was calculated as $\frac{\text{Difference}}{(\text{without TCD condition})} \times 100\%$

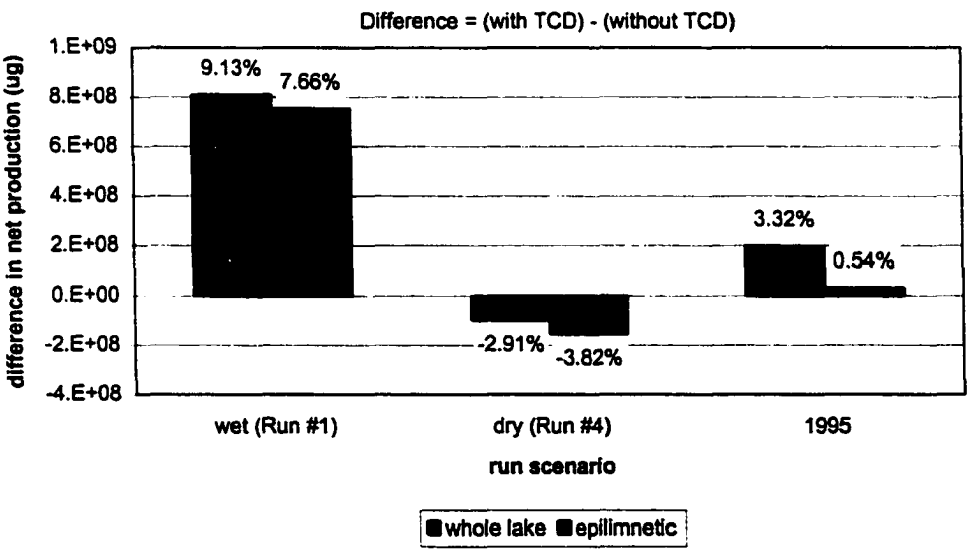

Figure 31. Difference in net production over stratified period as predicted by CE-QUAL-W2

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The estimated changes in species' wet weight due to TCD operation for the hydrologically wet, dry and 1995 scenarios are shown in Figures 32, 33 and 34, respectively. Notice that for the hydrologically wet and 1995 scenarios, all changes are shown as *gains* in wet weight due to TCD operation, while for the hydrologically dry scenario, all changes are *losses* in wet weight.

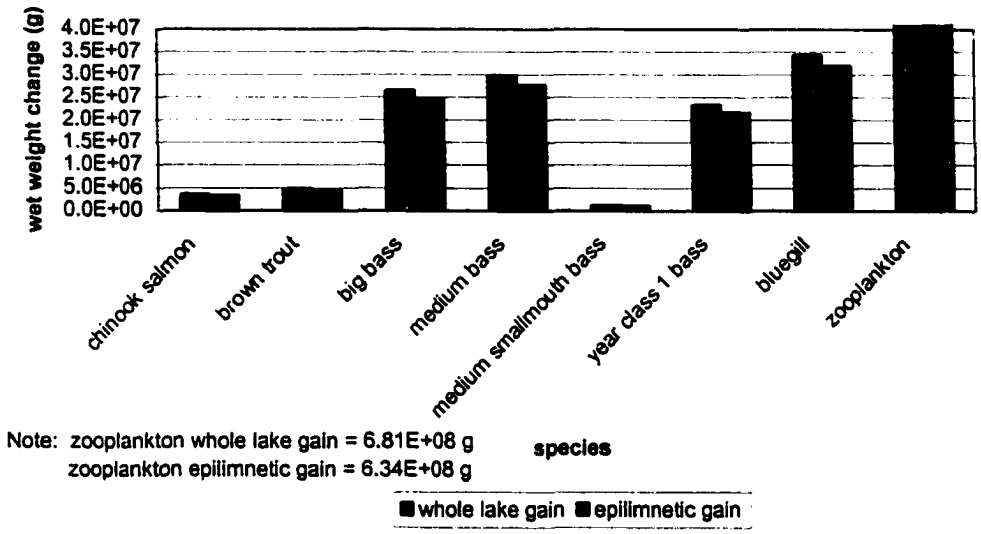

Figure 32. Changes in species' wet weight due to TCD operation for wet (Run #1) scenario

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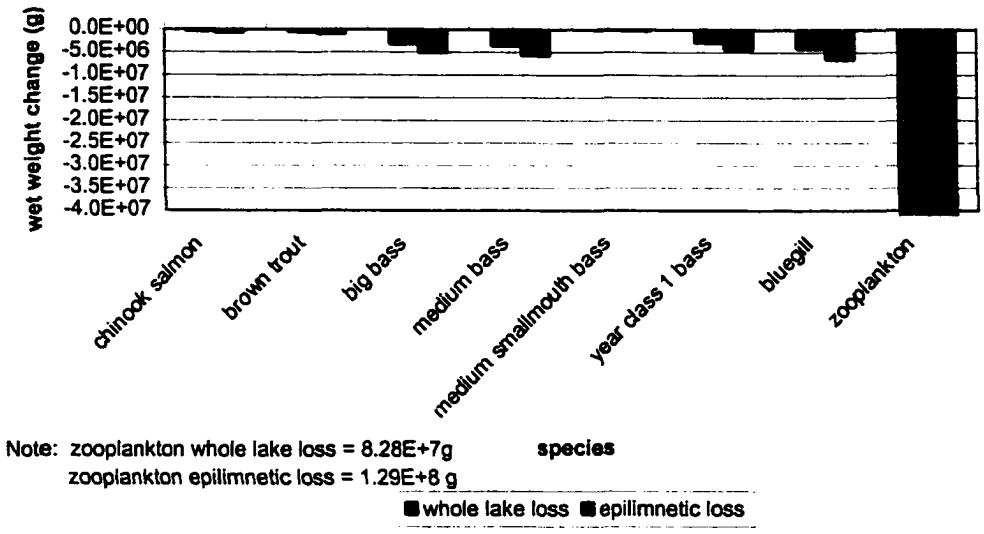

Figure 33. Change in species' wet weight due to TCD operation for dry (Run #4) scenario

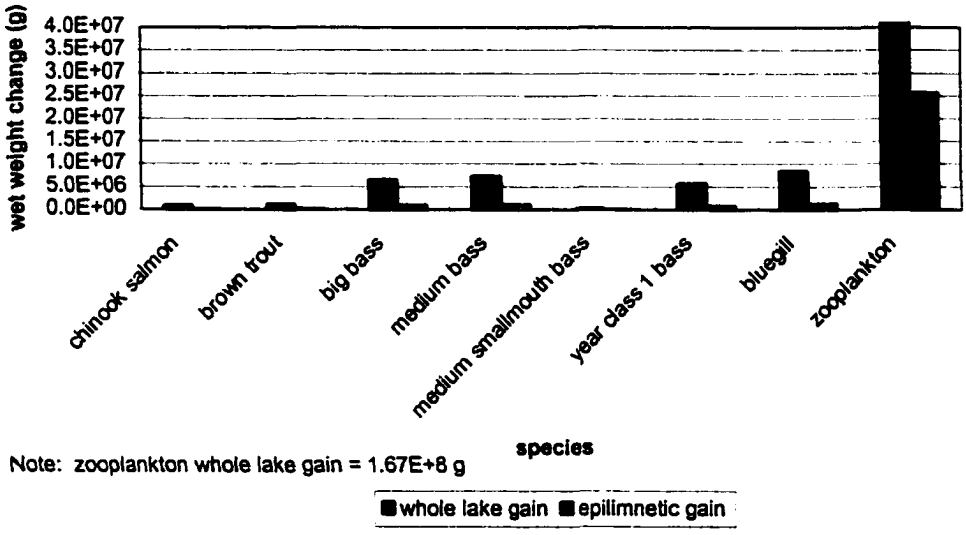

Figure 34. Change in species' wet weight due to TCD operation for 1995

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Chapter 9. DISCUSSION

Thermal calibration and validation of CE-QUAL-W2 showed that model performance increased with the quality of the input and calibration data, which agrees with an assessment of thermal models by Imberger (1985). The most input data was available for water temperature modeling in 1995 and 1996, with at least daily values of all input variables available most of the year. In the historical years, input data was much more scarce, which meant that more of the input data had to be estimated. In addition, measured water temperatures were available through time and space for 1995 calibration modeling, while water temperatures were only available at the dam for the historical years.

The amount of data available was adequate to rigorously calibrate CE-QUAL-W2 for its water temperature predictions, and it performed well in predicting thermal patterns in the reservoir. Model runs indicated that the internal modeling of the hydrodynamic processes in the reservoir was appropriate. The average error of less than 1°C throughout the reservoir for all years was comparable to or better than previous applications of CE-QUAL-W2 models (Garvey et al. 1998; Giorgino and Bales 1998; Martin 1988). Thus, the model was used to evaluate the thermal effects of the TCD with confidence.

The correlation between quality of model performance and input and calibration data was also evident in the nutrient and phytoplankton modeling. Inflow water quality data were not available for the calibration time period, so they were either estimated with

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methods from literature (Chapra 1997) or extrapolated from historical monthly nutrient measurements (Bartholow et al. in review). Measured data for calibration was much sparser than for the water temperature calibration, which made it more difficult to determine the sources of error when performing the calibration. Because the chlorophyll *a* observations were made with site-specific periodic samples, precise details of peak bloom timing or certainty of representativeness of collected samples was not possible due to algal patchiness from internal seiches and microscale variations in light availability (Paul 1987; Reynolds 1984). In addition to having to estimate much of the input data, and infer model performance between measured calibration data points, it was difficult to determine if internal model processes were not performing adequately. For example, CE-QUAL-W2 only includes one compartment of algae, so that all algae in the reservoir was modeled with one set of growth, respiration, and other such parameters. However, different algal types and species have very different nutrition requirements, growth, metabolism and palatability (Canale et al. 1976), so it was possible that the characterization of all algae into one compartment was inappropriate. In addition, coefficients such as the light extinction coefficient (EXH2O), fraction absorbed at the water surface (BETA), and factor for converting measured chlorophyll *a* concentrations in g m^{-3} to algal biomass were set at constant values for the duration of the model run even though they were likely to change over time. It was also assumed that biological and chemical rates were appropriately adjusted for temperature with the Thornton and Lessum (1978) algorithm:

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$$\lambda_{ii} = \frac{K_1 e^{\gamma_1 (T - T_1)}}{1 + K_1 e^{\gamma_1 (T - T_1)} - K_1} \frac{K_4 e^{\gamma_2 (T_4 - T)}}{1 + K_4 e^{\gamma_2 (T_4 - T)} - K_4}$$ Equation 92

where

λ_{ii} = temperature adjustment coefficient for cell *ii* that is multiplied by maximum reaction or biological rate

$$\gamma_1 = \frac{1}{T_2 - T_1} \ln \frac{K_2 (1 - K_1)}{K_1 (1 - K_2)}$$ Equation 93

$$\gamma_2 = \frac{1}{T_4 - T_3} \ln \frac{K_3 (1 - K_4)}{K_4 (1 - K_3)}$$ Equation 94

T_1 = lower mortality temperature limit (°C)

T_2 = lower optimum temperature limit (°C)

T_3 = upper optimum temperature limit (°C)

T_4 = upper mortality temperature limit (°C)

$K_{1,2,3,4}$ = rate multipliers at T_1, T_2, T_3 and T_4, respectively

Because of all of these uncertainties, model calibration focused more on matching trends in modeled water quality components than in obtaining high goodness-of-fit statistics. Bartholow et al. (in review) also noticed that the model performed better at predicting hypolimnetic or epilimnetic averages of water quality constituents rather than depth-specific measures, particularly for algae. Therefore, estimates of net algal production were summarized over the stratified period and across the whole lake and in the epilimnion for application with the food web model.

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Because CE-QUAL-W2 was calibrated to measured data in 1995 and 1996, conclusions of the two limnological studies (Brett et al. (1998) and Lieberman and Horn (1998)) overall corresponded with model simulations. TCD withdrawals of deeper, colder water in the late summer and fall resulted in a warmer hypolimnion until the reservoir mixed in the winter. After analyzing 50 years of historical water temperature profiles at Shasta Dam, Brett et al. (1998) concluded that TCD operation would reduce the reservoir's cool water pool until a wet year provided enough storage to replenish the pool. CE-QUAL-W2 simulations also indicated that storage affected the amount of available cold water and the ability of the TCD to meet downstream temperature targets, but thermal carryover effects of TCD operation into subsequent years were not predicted.

Spring and fall peaks of algal biomass were observed by Brett et al. (1998) and Lieberman and Horn (1998), and these were also predicted by the model. Because both of the limnological studies only spanned three years of monitoring, with only one year of data collected after the TCD began operating, effects of TCD operation based on observations were inconclusive (Brett et al. 1998; Lieberman and Horn 1998). Model predictions for 1995 showed very slight increases in net algal production with the TCD operating because TCD operations were so similar to the bypass operations used in 1995. In addition, the MVT analysis of model sensitivity to input variables indicated that, overall, hydrologic and meteorologic variables had greater effects on 50 thermal and water quality metrics than TCD operation. Both of these model conclusions indicate that the observed variability of water quality and biological parameters in the limnological studies could be largely due to hydrologic and climatic differences between the study years, rather than due to TCD operation.

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The ability to isolate the effects of the TCD operations from other variables is one of the reasons that models can be useful for alternative management assessments. Another utility is the ability to simulate conditions of concern, such as wet or dry year scenarios. It is often difficult to make such assessments with field observations alone, particularly when they are limited in time span. For example, although 1995 overall was classified as a hydrologically wet year based on total inflow, comparison of its storage pattern with storage patterns of averaged hydrologically wet and dry years (Figure 27) showed that it actually resembled a wet year during the first half of the year, and a dry year during the second half of the year. Thus, model simulations allowed evaluation of the effects of TCD operations under hydrologic extremes. The simulation of conditions in 1995 and 1996 also permitted evaluation of 'real year' operations.

As the models moved more into the realm of ecosystem effects, the uncertainties increased, and more assumptions had to be made. Ney (1993) pointed out four major areas of potential deficiencies in bioenergetics model applications: 1) unknown activity costs; 2) extrapolation of allometric functions; 3) unjustified species borrowing; and 4) inadequate estimation of external variables such as diet composition, energy densities, and temperature effects. The Shasta Lake bioenergetics application incorporates all of these deficiencies to a certain extent. It was assumed that the equations presented in Hanson et al. (1997) were appropriate for the species at Shasta, and the equations for anadromous steelhead were applicable for the primarily hatchery-stocked rainbow trout. The diet of rainbow trout and smallmouth bass were also assumed to be simple and constant. The use of the water temperature predictions from CE-QUAL-W2 improved the uncertainty that many bioenergetics applications have had regarding extrapolated water

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temperatures over space and time (Brandt et al. 1992; Goyke and Brandt 1993; Hewett 1989; Mason et al. 1995). Other key assumptions of the bioenergetics model were that food was abundant, and there were no other competitors or predators present. In other words, interspecific and intraspecific dynamics were not considered.

The uncertainties of the bioenergetics model were not quantifiable. For example, the error involved in using the steelhead bioenergetics coefficients from Hanson et al. (1997) for the Shasta Lake rainbow trout could not be assessed without experimentation. Railsback (1997) demonstrated that this can be a considerable error when consumption experiments were performed on rainbow trout for that study. To compensate for the inability to quantify these errors, the linked models were only interpreted in terms of assessing *changes* in relative scope for growth between with-TCD and without-TCD conditions. This approach assumed that errors in applying the bioenergetics equations would be consistent for both conditions, and observed differences in results between the conditions would be due to the effects of the TCD.

The application of the linked phytoplankton and food web models involved even more uncertainty than the linked bioenergetic modeling. In constructing the aquatic food web, it was apparent that estimated signatures matched well with measured signatures except for rainbow trout, threadfin shad, crayfish, and, to a lesser extent, year class 1 bass and bluegill (Table 36 and Figure 24). The inability to match the measured signatures more closely may involve one or more of the following critical assumptions that were made in this approach:

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- The approach assumed that all relevant pathways in the food web were accounted for and appropriately enabled (i.e., $f_i \neq 0$ for all possible items that can be consumed). This assumed that the literature review observations were appropriate for Shasta and also that there were no missing components of the food web. It is highly likely that one or more food web components were missing from the food web, and could compose significant portions of the diet of the threadfin shad, crayfish, and rainbow trout. Another related error could be the omission of one or more other carbon sources that have signatures (especially $\delta^{13}C$ values) similar to phytoplankton (Fry and Sherr 1984). This error would be harder to detect because diet proportions would be attributed to phytoplankton that should have been attributed to the omitted source.

- The detritus and periphyton signatures were estimated, and it is very possible that the estimated signatures were not correct. As noted previously, reported signatures vary widely, and the localized signatures at Shasta Lake could have been very different than those used in the food web.

- The enrichment of the carbon and nitrogen signatures (Δ') was assumed to be constant throughout the food web. In other words, the amount of enrichment did not depend on predator or prey. However, Gannes et al. (1997) point out that the efficiency of assimilation of dietary components varies with species, and several studies have suggested that variation in lipid content between animals may affect the trophic shift between prey and predator (Focken and Becker 1998; France 1995a; Kling et al. 1992). Hobson et al. (1995) observed greater carbon signature enrichment at lower

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trophic levels than at higher trophic levels, which could be an indication of strong microbial loop influences or inconsistent fractionation processes with different trophic linkages. Kline et al. (1998) presented an approach for normalizing carbon isotope signatures for lipid content effect in different fish species using nitrogen isotopes, but this approach was not applied at Shasta. Hesslein et al.'s (1993) study indicated that different rates of fish growth can affect the contribution of diet to changes observed in measured fish signatures, which also suggests that different fish species could have different trophic enrichment rates, especially if diet varies with time. Tan and Strain (1983) observed variation in carbon isotope fractionation factors over regions in an estuary, indicating possible spatial variability of enrichment factors. Also, nutritional stress could cause changes in fractionation rates (Gannes et al. 1997; Hobson and Clark 1992).

- Another assumption incorporated in the food web development was that the analysis of samples of muscle tissue and whole fish yielded data that was representative of the diet composition of the fish. However, isotopes in different food sources are routed differently to tissues and body parts, so that tissues may not reflect the isotopic composition of the bulk diet, but of the diet component that may make up most of that particular tissue growth (Gannes et al. 1997). Although many studies have shown that different body parts and tissues do exhibit different isotopic signatures, muscle tissue has had an intermediate signature in most organisms (DeNiro and Epstein 1978, 1981; Mizutani et al. 1991; Rounick and Hicks 1985; Tieszen et al. 1983). Therefore, Rounick and Hicks (1985) recommend muscle tissue as an appropriate

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body tissue for stable isotope analysis. Most recently, Pinnegar and Polunin (1999) performed laboratory investigations of differential fractionation of rainbow trout tissues and concluded that white muscle tissue was the most appropriate for ecotrophic studies.

- It was assumed that zooplankton consumed only phytoplankton. However, zooplankton may also consume detritus and planktonic heterotrophs (France et al. 1997).

- Equations 76 and 77 assume all diet items contribute linearly to the predator's signatures according to the proportions each makes up of the diet. However, Hobson et al. (1999) suggest that organisms that eat both plants and animals will tend to have isotopic signatures that are biased towards the animal source. They argue that while plants contribute both carbohydrates and proteins to herbivores, carnivores tend to get their dietary protein from animal sources, and carbohydrates from plant sources. The plant carbohydrates are respired as CO_2, while the animal protein ends up in tissue carbon in the predator (Krueger and Sullivan 1984). Therefore, Hobson et al. (1999) caution against using isotopes to trace carbohydrates or energy in food webs that include omnivores.

- It was assumed that the measured rainbow trout signatures reflected their in-reservoir diets even though they were hatchery-reared fish that were stocked in the reservoir. Trout are stocked in the spring (CDFG no date), and all of the trout collected were

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much larger than the size at which they are stocked (approximately 280 mm (11 inches) according to Manji (1998)). However, without measuring the signatures of the food provided at the hatchery, it was not possible to verify that the measured rainbow trout signatures were not affected by their past hatchery diet.

The estimate of the Shasta Lake food web was also based on an essentially one-time sample. Thus, the use of the estimated food web in evaluating effects over the period of stratification assumed that this sample captured the food web at Shasta during the stratified period. In reality, the food web is likely to be more dynamic. In terms of providing an 'average representation' of the food web, however, the one-time sampling was appropriate because the isotopes reflect time-integrated diet composition, and muscle tissue has a half-time for metabolic replacement of over 1 year (Hesslein et al. 1993). Aquatic studies that have sampled more than once throughout the year have found little difference in measured signatures within species (Fry et al. 1999; Gu et al. 1997).

The method of estimating the linkages in the food web to phytoplankton also involved some important assumptions. First, it was assumed that detritus was completely separate from phytoplankton when estimating energy and diet derived from phytoplankton. It was also assumed that no diet shifts occur when phytoplankton abundance changes. Because the food web model was not dynamic, diet proportions could not be adjusted based on changes in food abundance. Therefore, the model approach assumed that increases or decreases in phytoplankton abundance were accompanied by corresponding increases or decreases in other food web items so that diet proportions remained the same.

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The application of the Excel Solver approach to estimate diet proportions resulted in estimates of 0.41 ‰ and 3.0 ‰ for carbon and nitrogen isotope enrichment with trophic level, respectively. These results were encouraging because both of these values were well within the range of enrichment observed in other freshwater pelagic environments (France and Peters 1997; Fry 1991b).

There were a number of processes and dynamics that were not modeled by the linked modeling approaches. Diel movement of fish and zooplankton within the water column was not addressed, although such movement has been demonstrated and predicted (Carter and Gouldie 1986; Stockwell and Johnson 1997). The direct effects of dissolved oxygen impacts on fish populations were not assessed, although oxygen is known to affect fish physiologically (Coutant 1985) and has been assessed for reservoir fisheries through linked modeling approaches before (Chang et al. 1992). Both of the ecological models applied were primarily process-functional methods because they were concerned with energy flows. Interspecific and intraspecific competition, compensatory growth, and density dependence effects were therefore not addressed, although these dynamics are known to affect food consumption and growth (Henderson and Brown 1985; Majkowski and Waiwood 1981; Rudstam and Magnuson 1985; Whitledge et al. 1998). O'Neill et al. (1986) note that the process-functional approach can overlook dominant or key species that can strongly influence overall system function. It has been thought that threadfin shad could affect sport fish species abundance at Shasta Lake (Healey 1981), exerting a 'middle-out' control that has been observed in other studies (DeVries and Stein 1992). However, because the food web model was static, such influences of shad could not be assessed.

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This discussion has shown that only a portion of the error involved in the linked model predictions was due to data or procedures with known probabilities. As an example, Winter (1981) estimated the error associated with flow data from USGS gages as being about 5%, and suggested that with increased numbers of observation, the error would decrease if it was truly random. On the output side, prediction error of the CE-QUAL-W2 water temperature modeling could be estimated from the calibration results as being less than 1°C.

However, quantification of the error in the linked model predictions was not possible because the rest of the error was due to uncertainty associated with unknown probabilities (Costanza 1993). Beck (1987) described four problem areas associated with uncertainty: 1) uncertainty about model structure; 2) uncertainty in the values of parameters, coefficients, and inputs into the model; 3) uncertainty in predictions of future behavior; 4) and the design of experiments to reduce model uncertainty. All of these problems were encountered to varying degrees in the linked modeling effort.

The issue of uncertainty is common to ecosystem management issues because of the incomplete knowledge of the complexities of real ecosystems. Models are essential to addressing ecosystem issues such as sensitivity of system components to management change or in evaluating the health and functioning of a system (Christensen et al. 1996), partly because they enable the application of 'what if' scenarios. As this linked modeling effort has shown, it is critical that model predictions are made and applied in an appropriate manner that recognizes the uncertainties, known and unknown, within the simplified model representations of the complex ecosystem. As Johnson (1998) noted,

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the small sample sizes in most ecological analyses make it difficult to distinguish between natural variability and impact-induced variability.

To overcome this uncertainty, model interpretations were restricted in terms of resolution and comparison. Although the modeling approaches enabled calculation of quantifiable predictions through space and time, assessments with the linked bioenergetics modeling approach were restricted to comparisons between with-TCD and without-TCD conditions. The food web model interpretations were further restricted to assessing aggregated net production over the stratified period in the whole reservoir and epilimnetic region.

While the results presented here were dependent on numerous assumptions that were made because of data limitations, the application of stable isotope analysis with the linked modeling has much promise for future application in aquatic ecosystem studies. This experience has provided insight into how to improve sampling design to enable appropriate analyses. For example, it was evident that it was important to sample the entire food web, especially primary organic sources. Also, because signatures of higher species are not expected to change significantly over a season (Fry et al. 1999; Gu et al. 1997; Hesslein et al. 1993), another sampling effort could have provided calibration data for checking the estimated diet proportions in the food web model.

The isotope signatures of the Pit River arm were significantly different from the signatures measured at the other four locations in the reservoir, which was an interesting coincidence with similar findings from the limnological investigation. Lieberman and Horn (1998) consistently found lower Secchi transparencies, higher chl *a* concentrations, and greater phytoplankton biovolume in the Pit River arm than in the other locations

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sampled throughout the lake. Each of these factors indicates higher productivity on the Pit River arm. There is a potential to relate the observed isotopic variability with water quality data because isotopes have been used extensively to trace water sources and water quality constituents such as nitrate (Kendall 1998; Macko and Ostrom 1994). In a comparison study between streams in forested and agricultural watersheds, Harrington et al. (1998) observed elevated nitrogen isotope signatures throughout the aquatic food web in the streams with higher nitrate concentrations and greater agricultural and residential water quality impacts. Similarly, Peterson et al. (1993) performed a fertilization study in which they observed changes in the isotopic composition of consumers in response to fertilization of epilithic algae, and Schindler et al. (1997) observed higher carbon isotopic signatures in algae, zooplankton and fish in a manipulation experiment in which nutrient loading rates were increased to two out of four lakes. Wainright et al. (1996) hypothesized that the significant signature differences they observed in striped bass in the Delaware estuary could have been due to anthropogenic inputs that were affecting the signatures at the base of the food web. Further investigation at Shasta Lake could reveal similar impacts of water quality on the stable isotope signatures observed, and help to trace the sources of productivity inputs to the Pit River arm. Such data could indicate the influence of groundwater or anthropogenic inputs, which could enhance water quality and fisheries management strategies on the upper Pit River.

Although the stable isotopes were not applied to a dynamic food web, they could be used in the future to apply insight into food web interactions over time and space. For example, France and Steedman (1996) examined habitat preference between species by measuring isotopic signatures in the littoral and pelagic zones of four Canadian Shield

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lakes. Garvey et al. (1998) suggested that shad may link benthic and limnetic food webs by feeding on organic benthic sediments, and transporting nutrients to the limnetic zone via excretion. Examination of isotopic signatures of benthic food webs, shad, and pelagic food webs could verify this hypothesis.

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Chapter 10. CONCLUSIONS

The thermal analyses indicated that the TCD's effects in the upper 20-30 m of the reservoir were negligible. With simplified TCD operations (i.e., release through one outlet at a time on a seasonal basis), the reservoir was generally about 1°C cooler in the summer, while the hypolimnion was up to 5°C warmer in the fall. These changes were consistent regardless of hydrologic year, and were predicted to persist several km upstream of the dam. Because changes in epilimnetic temperatures were predicted to be negligible, scope for growth predictions for rainbow trout and smallmouth bass also indicated little change in growth potential. The greatest changes in fish scope for growth occurred about 100 m below the water surface where neither fish species was expected to be (Hanna et al. 1999).

The simulated impact of the TCD was not as great for 1995 as for the other hydrologic years. This was because the historical operation of the reservoir in 1995 used the bypass outlets to meet downstream temperature targets, while the historical operations of the other years maximized hydropower generation. Thus, 1995 historical operations were similar to modeled TCD operations. Although the hypolimnetic warming occurred later in the calendar year, multi-year simulations indicated that the next year's cool water reserve was not impacted because the annual fall turnover ensured that the winter isothermal conditions 'reset' the hypolimnetic water temperatures (Hanna et al. 1999).

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Although the simple TCD operations improved the ability to meet the downstream temperature targets, the targets could only be met throughout the year for some of the hydrologic years with complex TCD operations that involved blending releases from more than one outlet and changing the release elevations on a daily basis (Hanna et al. 1999). In hydrologically dry years, it was not possible to meet the downstream temperature targets with TCD operation because there was not enough cool water storage.

Model predictions of the CE-QUAL-W2 nutrient and phytoplankton simulations indicated that hydrologic and meteorological variables had much more significant effects on in-reservoir dynamics and outfall water quality than TCD operation. Although TCD operation would improve the ability of reservoir managers to meet downstream temperature targets, other variables such as having high storage at the beginning of the calendar year, or high inflows and low outflows throughout the year, would have a greater effect on meeting the targets (Bartholow et al. in review).

The food web analysis indicated that rainbow trout would be little affected by changes in phytoplankton, while bass species would be most affected. The linked modeling with CE-QUAL-W2 phytoplankton predictions indicated that in wet years, net algal production should increase over the stratified period, which may lead to increased fish biomass. Predictions under dry year conditions were opposite for the whole lake and epilimnetic regions of the reservoir.

The linked modeling shown in this dissertation demonstrated that interdisciplinary efforts can enhance the value of information obtained from ecosystem management studies. The approach used the strengths of the different modeling approaches to address

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concerns that none of the models could have addressed alone: 1) CE-QUAL-W2 could not have addressed quantitatively what happened to the fish; and 2) bioenergetics and food web models could not have looked at changes in temperatures or phytoplankton availability due to reservoir operations. The application of stable isotope analysis facilitated an estimate of the food web with minimal sampling in a short time frame. Future applications of the linked modeling approach with stable isotope analysis could provide more quantifiable predictions for systems with simpler food webs such as Blue Mesa or Horsetooth Reservoirs in Colorado. In addition, more comprehensive sampling of limnological data and components for stable isotope analysis could have yielded better estimation of model uncertainties. The application of the linked modeling approach at lakes and reservoirs with more comprehensive data sets such as Crater Lake in Oregon (Larson 1996) could generate more quantifiable predictions of ecosystem effects.

The work presented in this dissertation increased knowledge about the aquatic ecosystem at Shasta Lake. Although the predictions from the modeling effort were very generalized because of assumptions and lack of data, they could guide further monitoring or modeling efforts. Several hypotheses arose out of the work, including that:

- Rainbow trout are not tied to the phytoplankton primary production

- Net algal production over the stratified period increases with TCD operation in wet years, but decreases in dry years

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- There is increased net algal production in the hypolimnion with TCD operation under all hydrologic conditions

Another contribution of the modeling project was increased knowledge about the effects of TCD operations on reservoir ecosystems. In addition to Shasta Dam, selective withdrawal structures are currently functioning on Hungry Horse and Libby Dams in Montana, Flaming Gorge Dam in Utah, and Cachuma, Casitas, and Folsom Dams in California. Installation of TCDs are being considered at several other reservoirs, including Glen Canyon Dam, which impounds Lake Mead, one of the largest reservoirs in the country (Lieberman and Horn 1998). While the linked modeling provided insight into the Shasta Lake ecosystem, caution should be used in extrapolating the results to other reservoirs. Because Shasta is such a deep and large reservoir, the results of the Shasta Lake modeling effort presented in this dissertation indicated that changes in thermal and algal production patterns due to TCD operations would have little effect on the reservoir fishery. In smaller reservoirs with shorter detention times, it is likely that regions affected by altered thermal or productivity patterns would coincide with areas inhabited by fish and their prey. In addition, Shasta's TCD operations are dominated by the release and storage of cool water from low level outlets, while operations at other reservoirs may have different objectives. Thus, the utility of this research is not necessarily the extrapolation of the modeling results to other reservoirs, but the usefulness of the linked modeling approach with stable isotope analysis for addressing the ecosystem issues raised by the operation of selective withdrawal structures.

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The old engineering paradigm of optimizing technological performance (Wiedenhoeft 1999) would have focused primarily on finding reservoir operations that maximized hydropower generation. Consideration of the downstream ecological concerns of the endangered salmon still fit somewhat into this paradigm because the Endangered Species Act imposed a legal requirement to meet downstream temperature targets that meant that optimized technological performance needed to be multipurpose, a technical challenge that engineering modelers have addressed for years (Fontane et al. 1981; Ko et al. 1992; Lence et al. 1992). But the new engineering paradigm of stewardship and sustainability opens the Pandora's box of the ecosystem, and engineers can no longer hope to put scientific knowledge to practical use without actively working with other disciplines.

In the case of the Shasta Lake project, the new paradigm could be looked at in the context of the concerns over the downstream chinook salmon fishery that originally motivated the change in reservoir operations and installation of the TCD. Four types of community and ecosystem responses that could be expected due to the salmon habitat alterations are (Minns et al. 1996):

1. *Changes in total fish biomass and production*: This is the desired outcome of the change in reservoir operations at Shasta Lake. Specifically, the objective is to increase winter-run chinook salmon populations downstream of the dam.

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2. *Changes in the fish assemblage*: In addition to affecting chinook salmon populations, altered reservoir operations could change the species richness, biomass distribution, and production of chinook salmon and other species.

3. *Changes in some or all elements of the fish assemblage over time and/or space*: Fish species could be affected both upstream and downstream of the dam, and effects could be immediate or take some time to occur.

4. *Changes in the non-fish biotic elements of the ecosystem*: The ecosystem is an interconnected, complex system that includes other biotic components that are affected by changes in TCD operation and chinook salmon abundance.

The sixteen possible combinations of these outcomes (Table 39) provide a framework for looking at how ecosystem management has changed. The first hypothesis (H_0) is the null hypothesis that nothing responds, and the other symbols represent the 15 alternate hypotheses. If the only concern was whether or not the chinook salmon population increased abundance, and all other possible responses were ignored (which some would equate with assuming that they would not change), then alternate hypothesis II would be tested. In reality, the interconnectedness and uncertainties prevalent in the ecosystem imply that hypothesis VIII should be tested. The linked modeling approach of this dissertation was an intermediate step towards testing overall ecosystem effects because responses in phytoplankton (non-fish biota) and other fish species (fish

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assemblage composition) were investigated upstream of the dam (fish distribution patterns).

Table 39. Hypothesis matrix for fish assemblage response to ecosystem alteration

Change in fish assemblage composition?	Change in fish distribution patterns?	Change in total fish biomass and production?			
		No		Yes	
		Change in non-fish biota?		Change in non-fish biota?	
		No	Yes	No	Yes
No	No	H_0	I	IIa	IIb
	Yes	IIIa	IIIb	IVa	IVb
Yes	No	Va	Vb	VIa	VIb
	Yes	VIIa	VIIb	VIIIb	VIIIb

Source: Minns et al. (1996)

Was this interdisciplinary modeling effort a unique, one-time event? There are many reasons why it was not. The first reason is that it provided insight into the reservoir ecology at Shasta Lake that will be useful to Shasta's water and fishery managers in the future. On a broader scale, there is also increasing evidence that management is merging with science and will continue to do so.

The creation of the National Biological Service (NBS) in 1993 was a governmental effort to consolidate biological research, inventory and monitoring, and information transfer into an agency (NBS 1994). Research within the agency was intended to be less fragmented (Reichmann and Pulliam 1996) and provide a link between science and management mandates of the U.S. Department of the Interior and

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other agencies and institutions (Naimann et al. 1995). The NBS was incorporated into the Biological Resources Division (BRD) of the USGS in 1996 and the ecosystem management orientation of its research will hopefully be preserved. Interdisciplinary projects such as instream flow habitat modeling (Stalnaker 1994) and the efforts at Shasta Lake detailed in Chapter 2, including the modeling project highlighted in this dissertation, are examples of the kinds of programs pursued by the BRD.

In addition, ecologists have been climbing onto the bandwagon of adaptive management for ecosystems. Hilborn et al. (1996) define adaptive management as the use of deliberate management changes as experiments. This requires management to be flexible so that management policies can be altered as new knowledge is gained (Christensen et al. 1996). Critics of this approach point out that the gathering of new knowledge involves costly monitoring and that changing policy directions is not easy once vested interests have been created (Yaffee 1996). However, all management actions can be thought of as experiments, as there are going to be changes due to their action. Ward and Stanford (1987) considered regulated streams as excellent sites for testing ecological theory because of the manipulative ability of controlling downstream conditions. In fact, the highly-publicized release of flood-level waters from Glen Canyon Dam in 1995 was an example of merging management and science by altering reservoir management to perform a large-scale experiment (Collier et al. 1996; Wuethrich 1995).

Finally, there is a tremendous interest among disciplines to interact. At a recent ASCE conference, a discussion of interdisciplinary water resources modeling issues was filled to overflowing, and the interest was so great that a follow-on session is planned at next year's conference.

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The ultimate message of this dissertation is that no one can know everything about ecosystems, but together all of these disciplines know a lot more than they do separately. Rather than focus on the differences and discrepancies between disciplines, interdisciplinary efforts should focus on the similar goals the disciplines share. In water resources, the most interdisciplinary and sustainable viewpoint for water resources is probably the fundamental concept stated long ago in the Koran: *Water is the source of all life*. Further interdisciplinary efforts such as the one presented in this dissertation must be applied to preserve this precious resource.

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REFERENCES

Ahrens CD. 1991. Meteorology today: an introduction to weather, climate, and the environment. 4th ed. St. Paul: West Publishing Company. 577 p.

Alexander V, Gu B. 1997. The limnology of Smith Lake. In: Milner AM, Oswood MW, editors. Freshwaters of Alaska: ecological syntheses. New York: Springer-Verlag. p 131-153.

Allen MA, Hassler TJ. 1986. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Southwest) – chinook salmon. Washington (DC): US Department of the Interior, Fish and Wildlife Department. Biological Report 82(11.49), TR EL-82-4. 26 p.

[ASCE] American Society of Civil Engineers, Committee on Social and Environmental Objectives. 1984. Social and environmental objectives in water resources planning and management. New York: American Society of Civil Engineers. 44 p.

[ASCE] American Society of Civil Engineers, Task Committee on Sustainability Criteria. 1998. Sustainability criteria for water resource systems. Reston: American Society of Civil Engineers. 253 p.

Angradi TR. 1993. Stable carbon and nitrogen isotope analysis of seston in a regulated Rocky Mountain river, USA. Regul Rivers 8:251-270.

Angradi TR. 1994. Trophic linkages in the lower Colorado River: multiple stable isotope evidence. J North Am Benthol Soc 13(4):479-495.

Applegate RL, Mullan JW. 1967. Food of young largemouth bass, *Micropterus salmoides*, in a new and old reservoir. Trans Am Fish Soc 96(1):74-77.

Baker CD, Schmitz EH. 1971. Food habits of adult gizzard and threadfin shad in two Ozark reservoirs. In: Hall GE, editor. Reservoir fisheries and limnology. Washington (DC): American Fisheries Society. American Fisheries Society Special Publication No. 8. p 3-11.

Barnhart RA. 1986. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Southwest) – steelhead. Washington (DC): US

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Department of the Interior, Fish and Wildlife Department. Biological Report 82(11.60), TR EL-82-4. 21 p.

Bartholow JM. 1989. Stream temperature investigations: field and analytic methods. Washington (DC): US Department of the Interior, Fish and Wildlife Service. Instream Flow Information Paper nr 13, Biological Report nr 89(17). 139 p.

Bartholow J, Hanna B, Saito L, Lieberman D, Horn M. (in review). Simulated limnological effects of the Shasta Lake temperature control device. Environ Manage. Forthcoming.

Bartholow JM, Laake JL, Stalnaker CB, Williamson SC. 1993. A salmonid population model with emphasis on habitat limitations. Rivers 4(4):265-279.

Beck MB. 1983. A procedure for modeling. In: Orlob GT, editor. Mathematical modeling of water quality: streams, lakes, and reservoirs. Chichester: John Wiley and Sons. p 11-41.

Beck MB. 1987. Water quality modeling: a review of the analysis of uncertainty. Wat Resourc Res 23(8):1393-1442.

Begon M, Harper JL, Townsend CR. 1996. Ecology: individuals, populations and communities. Oxford: Blackwell Science Ltd. 1068 p.

Bender MD, Hauser GE, Shiao MC, Proctor WD. 1990. BETTER: a two-dimensional reservoir water quality model: technical reference manual and user's guide. Norris (TN): Tennessee Valley Authority Engineering Laboratory. Report nr WR28-2-590-152. 160 p.

Boon PI, Bunn SE. 1994. Variations in the stable isotope composition of aquatic plants and their implications for food web analysis. Aquat Bot 48:99-108.

Bootsma HA, Hecky RE, Hesslein RH, Turner GF. 1996. Food partitioning among Lake Malawi nearshore fishes as revealed by stable isotope analysis. Ecology 77(4):1286-1290.

Bosserman RW, Ragade RK. 1983. Ecosystem analysis using fuzzy set theory. Ecol Modell 16:191-208.

Brandt SB, Hartman KJ. 1993. Innovative approaches with bioenergetics models: future applications to fish ecology and management. Trans Am Fish Soc 122(5):731-735.

Brandt SB, Mason DM, Patrick EV. 1992. Spatially-explicit models of fish growth rate. Fisheries 17(2):23-31.

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Brett MT. 1999. Personal communication on July 13. Assistant Professor for Department of Civil Engineering, University of Washington. Seattle, WA.

Brett MT, Sarsfield C, DeStaso J, Duffy S, Heyvaert AC. 1998. Physical forcing of phytoplankton bloom dynamics in Shasta Lake, California: a progress report on the study of limnological effects following installation of a temperature control device. Davis: University of California. 38 p.

Bunn SE, Barton DR, Hynes HBN, Power G, Pope MA. 1989. Stable isotope analysis of carbon flow in a tundra river system. Can J Fish Aquat Sci 46:1769-1775.

Burns, James W. 1966a. Sacramento squawfish. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 525-527.

Burns JW. 1966b. Threadfin shad. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 481-488.

Burns TP, Higashi M, Wainright SC, Patten BC. 1991. Trophic unfolding of a continental shelf energy-flow network. *Ecol Modell* 55:1-26.

Cabana G, Rasmussen JB. 1996. Comparison of aquatic food chains using nitrogen isotopes. Proc Natl Acad Sci USA 93:10844-10847.

Canale RP, DePalma LM, Vogel AH. 1976. A plankton-based food web model for Lake Michigan. In: Canale RP, editor. Modeling biochemical processes in aquatic ecosystems. Ann Arbor: Ann Arbor Science. p 33-74.

Canfield DE, Bachmann RW. 1981. Prediction of total phosphorus concentrations, chlorophyll *a*, and Secchi depths in natural and artificial lakes. Can J Fish Aquat Sci 38:414-423.

Cantor LW. 1985. Environmental impacts of water resources projects. Chelsea: Lewis Publishers. 352 p.

Carpenter SR. 1989. Replication and treatment strength in whole-lake experiments. Ecology 70(2):453-463.

Carpenter SR, Kitchell JF, Hodgson JR. 1985. Cascading trophic interactions and lake productivity. BioScience 35(10):634-639.

Carter JCH, Goudie KA. 1986. Diel vertical migrations and horizontal distributions of *Limnocalanus macrurus* and *Senecella calanoides* (Copepoda, Calanoida) in lakes of southern Ontario in relation to planktivorous fish. Can J Fish Aquat Sci 43:2508-2514.

[CDFG] California Department of Fish and Game. No date. Fisheries management: Shasta Lake. Redding: California Department of Fish and Game. 2 p.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

[CDFG] California Department of Fish and Game. 1991. Draft Shasta Lake management plan. Redding: California Department of Fish and Game. 33 p.

Cermelj B, Faganeli J, Ogorelec B, Dolenec T, Pezdic J, Smodis B. 1996. The origin and recycling of sedimented biogenenic debris in a subalpine eutrophic lake (Lake Bled, Slovenia). Biogeochemistry 32:69-91.

Chang LH, Railsback SF, Brown RT. 1992. Use of a reservoir water quality model to simulate global climate change effects on fish habitat. Climate Change. 20:277-296.

Chapra SC. 1997. Surface water-quality modeling. New York: McGraw-Hill. 844 p.

Chapra SC, Canale RP. 1988. Numerical methods for engineers. 2nd ed. New York: McGraw-Hill. 839 p.

Chen CW, Wells JT. 1976. Boise River ecological modeling. In: Canale RP, editor. Modeling biochemical processes in aquatic ecosystem. Ann Arbor: Ann Arbor Science. p 171-203.

Christensen NL, Bartuska AM, Brown JH, Carpenter S, D'Antonio C, Francis R, Franklin JF, MacMahan JM, Noss RF, Parsons DJ, Peterson CH, Turner MG, Woodmansee RG. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecol Appl 6(3):665-691.

Christensen V, Pauly D. 1992. ECOPATH II – a software for balancing steady-state ecosystem models and calculating network characteristics. Ecol Modell 61:169-185.

Christensen V, Pauly D, editors. 1993a. Trophic models of aquatic ecosystems. ICLARM Conference Proceedings 26. Flow characteristics of aquatic ecosystems; p 338-352.

Christensen V, Pauly D, editors. 1993b. Trophic models of aquatic ecosystems. ICLARM Conference Proceedings 26. 390 p.

Christie GC, Regier HA. 1988. Measures of optimal thermal habitat and their relationship to yields for four commercial fish species. Can J Fish Aquat Sci 45:301-314.

Clark GH. 1929. Sacramento-San Joaquin salmon (*Oncorhynchus tshawytscha*) fishery of California. Division of Fish and Game of California. Fish Bulletin nr 17. 73 p.

Clarke R. 1993. Water: the international crisis. Cambridge (MA): The MIT Press. 193 p.

Coble DW. 1975. Smallmouth bass. In: Clepper H, editor. Black bass biology and management. Washington (DC): Sport Fishing Institute. p 21-33.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Cochrane KL, Ashton PJ, Jarvis AC, Twinch AJ, Zohary T. 1987. An ecosystem model of phosphorus cycling in a warm, monomictic, hypertrophic impoundment. Ecol Modell 37:207-233.

Cole RA, Ward TJ, Ward FA, Deitner RA, Rodden RW, Bolton SM, Green-Hammond KA. 1995. RIOFISH: a comprehensive management system model for New Mexico sportfisheries. Las Cruces: NM Water Resources Research Institute. WRRI Technical Completion Report nr 291. 230 p.

Cole TM. 1998. Personal communication on January 13. Hydrologist for US Army Corps of Engineers, Waterways Experiment Station. Vicksburg, MS.

Cole TM. 1999. Personal communication on September 14. Hydrologist for US Army Corps of Engineers, Waterways Experiment Station. Vicksburg, MS.

Cole TM, Buchak EM. 1995. CE-QUAL-W2: a two-dimensional, laterally averaged, hydrodynamic and water quality model. Version 2.0. Vicksburg: US Army Corps of Engineers, Waterways Experiment Station. Instruction Report EL-95-. 59 p. + appendices.

Collier M, Webb RH, Schmidt JC. 1996. Dams and rivers: primer on the downstream effects of dams. Tucson: US Geological Survey. U.S. Geological Survey Circular 1126. 94 p.

Cone J, Ridlington S, editors. 1996. The northwest salmon crisis: a documentary history. Corvallis: Oregon State University Press. 374 p.

Costanza R. 1993. Developing ecological research that is relevant for achieving sustainability. Ecol Appl 3(4):579-581.

Costanza R, Sklar FH. 1985. Articulation, accuracy and effectiveness of mathematical models: a review of freshwater wetland applications. Ecol Modell 27:45-68.

Coutant CC. 1976. Thermal effects on fish ecology. In: Encyclopedia of environmental science and engineering. Volume 2. New York: Gordon and Breach Science Publishers. p 891-896.

Coutant CC. 1985. Striped bass, temperature, and dissolved oxygen: a speculative hypothesis for environmental risk. Trans Am Fish Soc 114:31-61.

Crockett P. 1994. Water-quality modeling. In: Calow P, Petts GE, editors. The rivers handbook: hydrological and ecological principles. Volume 2. Oxford: Blackwell Scientific Publications. p 213-226.

Cullen P. 1990. The turbulent boundary between water science and water management. Freshwater Biol 24:201-209.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Cummins KW, Wuycheck JC. 1971. Caloric equivalents for investigations in ecological energetics. Int Verein Theoret Angewandte Limnol 18:1-158.

Curtis B. 1949. The warm-water game fishes of California. Calif Fish Game 35(4):255-273.

Davis BM, Foltz JW. 1991. Food of blueback herring and threadfin shad in Jocassee Reservoir, South Carolina. Trans Am Fish Soc 120(5):605-613.

DeAngelis DL. 1988. Strategies and difficulties of applying models to aquatic populations and food webs. Ecol Modell 43:57-73.

DeAngelis DL. 1992. Dynamics of nutrient cycling and food webs. London: Chapman and Hall. 270 p.

DeAngelis DL, Bartell SM, Brenkert AL. 1989. Effects of nutrient recycling and food-chain length on resilience. Am Nat 134(5):778-805.

DeAngelis DL, Cushman RM. 1990. Potential application of models in forecasting the effects of climate changes on fisheries. Trans Am Fish Soc 119:224-239

del Giorgio PA, France RL. 1996. Ecosystem-specific patterns in the relationship between zooplankton and POM or microplankton $\delta^{13}C$. Limnol Oceanogr 41(2):359-365.

DeNiro MJ, Epstein S. 1978. Influence of diet on the distribution of carbon isotopes in animals. Geochim Cosmochim Acta 42:495-506.

DeNiro MJ, Epstein S. 1981. Influence of diet on the distribution of nitrogen isotopes in animals. Geochim Cosmochim Acta 45:341-351.

DeVries DR, Stein RA. 1992. Complex interactions between fish and zooplankton: quantifying the role of an open-water planktivore. Can J Fish Aquat Sci 49:1216-1227.

DeVries DR, Stein RA, Miner JG, Mittelbach GG. 1991. Stocking threadfin shad: consequences for young-of-year fishes. Trans Am Fish Soc 120(3):368-381.

Duffy S. 1999. Personal communication on July 12. Regional Support Manager for Microm Bioresources, Inc. Auburn, CA.

Edwards EA, Gebhart G, Maughan OE. 1983. Habitat suitability information: smallmouth bass. Washington (DC): US Department of the Interior, Fish and Wildlife Service. FWS/OBS-82/10.36. 47 p.

Elliot JM, Davison W. 1975. Energy equivalents of oxygen consumption in animal energetics. Oecologia (Berl.) 19:195-201.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Emig JW. 1966a. Bluegill sunfish. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 375-392.

Emig JW. 1966b. Largemouth bass. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 332-353.

Emig JW. 1966c. Smallmouth bass. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 354-366.

Estep MF, Vigg S. 1985. Stable carbon and nitrogen isotope tracers of trophic dynamics in natural populations and fisheries of the Lahontan lake system, Nevada. Can J Fish Aquat Sci 42:1712-1719.

Fagerström T. 1987. On theory, data and mathematics in ecology. Oikos 50(2):258-261.

Ferrara RA. 1986. Introduction to the modeling of lakes, reservoirs, and impoundments. In: Gray WG, editor. Physics-based modeling of lakes, reservoirs, and impoundments. New York: American Society of Civil Engineers. p 1-16.

Fisher CR, Childress JJ, Macko SA, Brooks JM. 1994. Nutritional interactions in Galapagos Rift hydrothermal vent communities: inferences from stable carbon and nitrogen isotope analyses. Mar Ecol Prog Series 103:45-55.

Flint RW. 1986. Hypothesized carbon flow through the deepwater Lake Ontario food web. J Great Lakes Res 12(4):344-354.

Flug M. 1997. Ecosystems considerations in reservoir management. Wat Resourc Update 108:43-56.

Focken U, Becker K. 1998. Metabolic fractionation of stable carbon isotopes: implications of different proximate compositions for studies of the aquatic food webs using $\delta^{13}C$ data. Oecologia 115:337-343.

Fontane DG, Labadie JW, Loftis B. 1981. Optimal control of reservoir discharge quality through selective withdrawal. Wat Resourc Res 17(6):1594-1604.

Forsberg BR, Arauja-Lima CARM, Marinelli LA, Victoria RL, Bonassi JA. 1993. Autotrophic carbon sources for fish of the central Amazon. Ecology 74(3):643-652.

France RL. 1995a. Carbon isotopic variability in the composite pelagic foodweb of four oligotrophic lakes: feeding diversity or metabolic fractionations? J Plankton Res 17(10):1993-1997.

France R. 1995b. Critical examination of stable isotope analysis as a means for tracing carbon pathways in stream ecosystems. Can J Fish Aquat Sci 52:651-656.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

France RL. 1995c. Source variability in $\delta^{15}N$ of autotrophs as a potential aid in measuring allochthony in freshwaters. Ecography 18(3):318-320.

France R. 1995d. Stable nitrogen isotopes in fish: literature synthesis on the influence of ecotonal coupling. Estuar Coast Shelf Sci 41:737-742.

France RL. 1997a. $\delta^{15}N$ examination of the Lindeman-Hutchinson-Peters theory of increasing omnivory with trophic height in aquatic food webs. Res Pop Ecol 39(2):121-125.

France RL. 1997b. Stable carbon and nitrogen isotopic evidence for ecotonal coupling between boreal forests and fishes. Ecol Freshwater Fish 6:78-83.

France RL, del Giorgio PA, Westcott KA. 1997. Productivity and heterotrophy influences on zooplankton $\delta^{13}C$ in northern temperate lakes. Aquat Microbial Ecol 12:85-93.

France RL, Peters RH. 1997. Ecosystem differences in the trophic enrichment of ^{13}C in aquatic food webs. Can J Fish Aquat Sci 54:1255-1258.

France R, Steedman R. 1996. Energy provenance for juvenile lake trout in small Canadian shield lakes as shown by stable isotopes. Trans Am Fish Soc 125:512-518.

Fry B. 1986. Sources of carbon and sulfur nutrition for consumers in three meromictic lakes of New York State. Limnol Oceanogr 31(1):79-88.

Fry B. 1988. Food web structure on Georges Bank from stable C, N, and S isotopic compositions. Limnol Oceanogr 33(5):1182-1190.

Fry B. 1991a. Long term ecological research (LTER) stable isotope workshop. Ecological Society of America. ESA Supplementary Publication Service Document No. ESPS-9102. Ecology 72(1991). 32 p.

Fry B. 1991b. Stable isotope diagrams of freshwater food webs. Ecology 72(6):2293-2297.

Fry B, Mumford PL, Tam F, Fox DD, Warren GL, Havens KE, Steinman AD. 1999. Trophic position and individual feeding histories of fish from Lake Okeechobee, Florida. Can J Fish Aquat Sci 56:590-600.

Fry B, Sherr EB. 1984. $\delta^{13}C$ measurements as indicators of carbon flow in marine and freshwater ecosystems. Contrib Mar Sci 27:13-47.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Fujitani P. 1997. US Department of the Interior, Bureau of Reclamation, Mid-Pacific Region, Central Valley Operations Coordinating Office. Personal communication in January and February 1997.

Fukushima T, Muraoka K. 1985. Simple model to predict water quality in 90 Japanese lakes. Verh Int Verein Limnol 23:812-827.

Gannes LZ, O'Brien DM, Martínez del Rio C. 1997. Stable isotopes in animal ecology: assumptions, caveats, and a call for more laboratory experiments. Ecology 78(4):1271-1276.

Garvey E, Tobiason JE, Hayes M, Wolfram E, Reckhow DA, Male JW. 1998. Coliform transport in a pristine reservoir: modeling and field studies. Wat Sci Technol 37(2):137-144.

Giorgino MJ, Bales JD. 1997. Rhodhiss Lake, North Carolina: analysis of ambient conditions and simulation of hydrodynamics, constituent transport, and water-quality characteristics, 1993-94. Raleigh: US Geological Survey. US Geological Survey Water Resources Investigations Report 97-4131. 62 p.

Gleick PH, editor. 1993. Water in crisis: a guide to the world's fresh water resources. New York: Oxford University Press. An introduction to global fresh water issues; p 3-12.

Goericke R, Montoya JP, Fry B. 1994. Physiology of isotopic fractionation in algae and cyanobacteria. In: Lajtha K, Michener RH, editors. Stable isotopes in ecology and environmental science. Oxford: Blackwell Scientific Publications. p 187-221.

Goldman CR. 1968. Aquatic primary production. Am Zool 8:31-42.

Goodson LF. 1965. Diets of four warmwater game fishes in a fluctuating, steep-sided, California reservoir. Calif Fish Game 51(4):259-269.

Goyke AP, Brandt SB. 1993. Spatial models of salmonine growth rates in Lake Ontario. Trans Am Fish Soc 122(5):870-883.

Gu B, Alexander V, Schell DM. 1997. Stable isotopes as indicators of carbon flows and trophic structure in the benthic food web in a subarctic lake. Arch Hydrobiol 138(3):327-344.

Gu B, Schelske CL, Hoyer MV. 1996. Stable isotopes of carbon and nitrogen as indicators of diet and trophic structure of the fish community of a shallow hypereutrophic lake. J Fish Biol 49:1233-1243.

Halfon E, Schito N, Ulanowicz RE. 1996. Energy flow through the Lake Ontario food web: conceptual model and an attempt at mass balance. Ecol Modell 86:1-36.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Hall CAS, Day JW, editors. 1977. Ecosystem modeling in theory and practice. New York: John Wiley and Sons. Chapter 1, Systems and models: terms and basic principles; p 6-36.

Hamilton DP, Schladow SG. 1997. Prediction of water quality in lakes and reservoirs: part I – model description. Ecol Modell 96:91-110.

Hamilton SK, Lewis WM. 1992. Stable carbon and nitrogen isotopes in algae and detritus from the Orinoco River floodplain, Venezuela. Geochim Cosmochim Acta 56:4237-4246.

Hamilton SK, Lewis WM, Sippel SJ. 1992. Energy sources for aquatic animals in the Orinoco River floodplain: evidence from stable isotopes. Oecologia 89:324-330.

Hanna RB, Saito L, Bartholow JM, Sandelin J. 1999. Results of simulated temperature control device operations on in-reservoir and discharge water temperatures using CE-QUAL-W2. Lake Reserv Manage 15(2):87-102.

Hanson JM, Leggett WC. 1982. Empirical prediction of fish biomass and yield. Can J Fish Aquat Sci 39:257-263.

Hanson PC, Johnson TB, Schindler DE, Kitchell JF. 1997. Fish bioenergetics 3.0 for Windows. University of Wisconsin-Madison. 86 p.

Harrigan P, Zieman JC, Macko SA. 1989. The base of nutritional support for the gray snapper (*Lutjanus griseus*): an evaluation based on a combined stomach content and stable isotope analysis. Bull Mar Sci 44(1):65-77.

Harrington RR, Kennedy BP, Chamberlain CP, Blum JD, Holt CL. 1998. ^{15}N enrichment in agricultural catchments: field patterns and applications to tracking Atlantic salmon (*Salmo salar*). Chem Geol 147:281-294.

Hayes DB, Ferreri CP, Taylor WW. 1996. Linking fish habitat to their population dynamics. Can J Fish Aquat Sci 53(Suppl. 1):383-390.

Healey MC. 1994. Variation in the life history characteristics of chinook salmon and its relevance to conservation of the Sacramento winter run of chinook salmon. Conserv Biol 8(3):876-877.

Healey TP. 1981. Shasta Lake trout fishery investigations, 1971-1974. California Department of Fish and Game. Administrative Report nr 81-4. 29 p.

Hecky RE, Hesslein RH. 1995. Contributions of benthic algae to lake food webs as revealed by stable isotope analysis. J North Am Benthol Soc 14(4):631-653.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Heidinger RC. 1975. Life history and biology of the largemouth bass. In: Clepper H, editor. Black bass biology and management. Washington (DC): Sport Fishing Institute. p. 11-20.

Heidtke TM, Auer MT, Canale RP. 1986. Microcomputers and water quality models: access for decision makers. J Wat Poll Control Fed 58(10):960-966.

Heizer RF, editor. 1973. Notes on the McCloud River Wintu and selected excerpts from Alexander S. Taylor's *Indianology of California*. Berkeley: University of California. 79 p.

Henderson BA, Brown EH. 1985. Effects of abundance and water temperature on recruitment and growth of alewife (*Alosa pseudoharengus*) near South Bay, Lake Huron, 1954-82. Can J Fish Aquat Sci 42:1608-1613.

Hesslein RH, Hallard KA, Ramlal P. 1993. Replacement of sulfur, carbon and nitrogen in tissue of growing broad whitefish (*Coregonus nasus*) in response to a change in diet traced by $\delta^{34}S$, $\delta^{13}C$, and $\delta^{15}N$. Can J Fish Aquat Sci 50:2071-2076.

Hewett SW. 1989. Ecological applications of bioenergetics models. Am Fish Soc Sympos 6:113-120.

Heymans JJ, McLachlan A. 1996. Carbon budget and network analysis of a high-energy beach/surf-zone ecosystem. Estuar Coast Shelf Sci 43:485-505.

Hilborn R, Ludwig D. 1993. The limits of applied ecological research. Ecol Appl 3(4):550-552.

Hilborn R, Walters CJ, Ludwig D. 1995. Sustainable exploitation of renewable resources. Annu Rev Ecol Syst 26:45-67.

Hildén M, Kettunen J. 1985. Models for the assessment of environmental impacts on fish populations and fisheries. Finn Fish Res 6:35-42.

Hirst SC, DeVries DR. 1994. Assessing the potential for direct feeding interactions among larval black bass and larval shad in two southeastern reservoirs. Trans Am Fish Soc 123(2):173-181.

Hobbs HH. 1991. Decapoda. In: Thorps JH, Covich AP, editors. Ecology and classification of north American freshwater invertebrates. San Diego: Academic Press. p 823-258.

Hobson KA, Ambrose WG, Renaud PE. 1995. Sources of primary production, benthic-pelagic coupling, and trophic relationships within the Northeast Water Polynya: insights from $\delta^{13}C$ and $\delta^{15}N$ analysis. Mar Ecol Prog Series 128:1-10.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Hobson KA, Clark RG. 1992. Assessing avian diets using stable isotopes II: factors influencing diet-tissue fractionation. Condor 94:189-197.

Hobson KA, Drever MC, Kaiser GW. 1999. Norway rats as predators of burrow-nesting seabirds: insights from stable isotope analysis. J Wildl Manage 63(1):14-25.

Hobson KA, Welch HE. 1995. Cannibalism and trophic structure in a high Arctic lake: insights from stable-isotope analysis. Can J Fish Aquat Sci 52:1195-1201.

Holliman J. 1974. Consumer's guide to the protection of the environment. London: Pan/Ballantine. 254+ p.

Hovecamp S, Sarsfield C, DeStaso J. 1996. Shasta Dam temperature control device: ecological study design. Shasta Lake: US Department of the Interior, Bureau of Reclamation, Northern California Area Office.

Huston M, DeAngelis D, Post W. 1988. New computer models unify ecological theory. BioScience 38(10):682-691.

Imberger J. 1985. Thermal characteristics of standing waters: an illustration of dynamic processes. Hydrobiologia 125:7-29.

Imberger J, Patterson J, Hebbert B, Loh I. 1978. Dynamics of a reservoir of medium size. J Hydraul Div 104(HY5):725-743.

Jenkins RM. 1982. The morphoedaphic index and reservoir fish production. Trans Am Fish Soc 111:133-140.

Johnson BH. 1981. A review of numerical reservoir hydrodynamic modeling. Vicksburg: US Army Corps of Engineers, Waterways Experiment Station. Technical Report E-81-2. 182 p.

Johnson BM, Stockwell JD, Bonfantine K. 1997. Ecological effects of reservoir operations on Blue Mesa Reservoir: annual progress report May 1, 1996 – April 30, 1997. Fort Collins: Colorado State University. 144 p.

Johnson RK. 1998. Spatiotemporal variability of temperate lake macroinvertebrate communities: detection of impact. Ecol Appl 8(1):61-70.

Jones JR, Knowlton MF, Kaiser MS. 1998. Effects of aggregation on chlorophyll-phosphorus relations in Missouri reservoirs. Lake Reserv Manage 14(1):1-9.

Jørgensen SE. 1983. Mathematical modeling of water quality: streams, lakes, and reservoirs. Chichester: John Wiley and Sons. Chapter 4, Modeling the ecological processes; p 116-149.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Jørgensen SE. 1988. Fundamentals of ecological modelling. Elsevier: Amsterdam. 391 p.

Jørgensen SE, Patten BC, Straskraba M. 1992. Ecosystems emerging: toward an ecology of complex systems in a complex future. Ecol Modell 62:1-27.

Junger M, Planas D. 1994. Quantitative use of stable carbon isotope analysis to determine the trophic base of invertebrate communities in a boreal forest lotic system. Can J Fish Aquat Sci 51:52-61.

Kampa JM. 1984. Density-dependent regulation of gizzard shad populations in experimental ponds. [MS thesis]. Columbia (MO): University of Missouri.

Kendall C. 1998. Tracing nitrogen sources and cycling in catchments. In: Kendall C, McDonnell JJ, editors. Isotope tracers in catchment hydrology. Elsevier Science. p 519-576.

Keough JR, Hagley CA, Ruzycki E, Sierszen M. 1998. $\delta^{13}C$ composition of primary producers and role of detritus in a freshwater coastal ecosystem. Limnol Oceanogr 43(4):734-740.

Kilgour DM, McCauley RW, Kwain W. 1985. Modeling the lethal effects of high temperature on fish. Can J Fish Aquat Sci 42:947-951.

Kline TC, Goering JJ, Mathisen OA, Poe PH, Parker PL. 1990. Recycling of elements transported upstream by runs of Pacific Salmon: I. $\delta^{15}N$ and $\delta^{13}C$ evidence in Sashin Creek, southeastern Alaska. Can J Fish Aquat Sci 47:136-144.

Kline TC, Goering JJ, Mathisen OA, Poe PH, Parker PL, Scalan RS. 1993. Recycling of elements transported upstream by runs of Pacific salmon: II. $\delta^{15}N$ and $\delta^{13}C$ evidence in the Kvichak River watershed, Bristol Bay, southwestern Alaska. Can J Fish Aquat Sci 50:2350-2365.

Kline TC, Wilson WJ, Goering JJ. 1998. Natural isotope indicators of fish migration at Prudhoe Bay, Alaska. Can J Fish Aquat Sci 55:1494-1502.

Kling GW, Fry B, O'Brien WJ. 1992. Stable isotopes and planktonic trophic structure in arctic lakes. Ecology 73(2):561-566.

Ko SK, Fontane DG, Labadie JW. 1992. Multiobjective optimization of reservoir systems operation. Wat Resourc Bull 28(1):111-127.

Krebs CJ. 1994. Ecology: the experimental analysis of distribution and abundance. 4th ed. New York: Harper Collins College Publishers. 801 p.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Krenkel PA, French RH. 1982. State-of-the-art of modeling surface water impoundments. Wat Sci Technol 14:241-261.

Krenkel PA, Novotny V. 1980. Water quality management. New York: Academic Press. 671 p.

Krueger HW, Sullivan CH. 1984. Models for carbon isotope fractionation between diet and bone. In: Turnlund JR, Johnson PE, editors. Stable isotopes in nutrition. Washington (DC): American Chemical Society. p 205-220.

Lackey RT, Powers JE, Zuboy JR. 1975. Modeling to improve management of bass fisheries. In: Clepper H, editor. Black bass biology and management. Washington (DC): Sport Fishing Institute. p 430-435.

Lajtha K, Michener RH, editors. 1994. Stable isotopes in ecology and environmental science. Oxford: Blackwell Scientific Publications. 316 p.

Larson GL. 1996. Overview of the limnology of Crater Lake. Northwest Science 70(2):39-47.

Leach JH, Herron RC. 1992. A review of lake habitat classification. In: Bush WDN, Sly PG, editors. The development of an aquatic habitat classification system for lakes. Boca Raton: CRC Press. p 27-57.

Leggett RW, Williams LR. 1981. A reliability index for models. Ecol Modell 13:303-312.

Lence BJ, Latheef MI, Burn DH. 1992. Reservoir management and thermal power generation. J Wat Resourc Planning Manage 118(4):388-405.

Lieberman DM, Horn MJ. 1998. 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. Denver: US Geological Survey. US Geological Survey Open-File Report 98-251. 126 p.

Lindeman RL. 1942. The trophic-dynamic aspect of ecology. Ecology 23(4):399-418.

Loucks DP. 1983. Models for management applications. In: Orlob GT, editor. Mathematical modeling of water quality: streams, lakes, and reservoirs. Chichester: John Wiley and Sons. p 468-502.

Lynch DR. 1986. Basic hydrodynamic equations for lakes. In: Gray WG, editor. Physics-based modeling of lakes, reservoirs, and impoundments. New York: American Society of Civil Engineers. p 17-53.

Lynott ST, Bryan SD, Hill TD, Duffy W. 1995. Monthly and size-related changes in the diet of rainbow trout in Lake Oahe, South Dakota. J Freshwater Ecol 10(4):399-407.

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Macko SA, Ostrom NE. 1994. Pollution studies using stable isotopes. In: Lajtha K, Michener RH, editors. Stable isotopes in ecology and environmental science. Oxford: Blackwell Scientific Publications. p 45-62.

Magnuson JJ, Meisner JD, Hill DK. 1990. Potential changes in the thermal habitat of Great Lakes fish after global climate warming. Trans Am Fish Soc 119:254-264.

Majkowski J, Waiwood KG. 1981. A procedure for evaluating the food biomass consumed by a fish population. Can J Fish Aquat Sci 38:1199-1208.

Manji NT. 1998. Personal communication on July 27 and July 31. Fishery biologist for the California Department of Fish and Game. Redding, CA.

Manji NT. 1999. Personal communication on July 13. Fishery biologist for the California Department of Fish and Game. Redding, CA.

Marotz BL, Althen C, Lonon B, Gustafson D. 1996. Model development to establish integrated operational rule curves for Hungry Horse and Libby Reservoirs – Montana: final report 1996. Kalispell: Montana Department of Fish, Wildlife and Parks. 114 pp. + appendices.

Martin JL. 1988. Application of two-dimensional water quality model. J Environ Engin 114(2):317-336.

Martin JL, Wlosinski JH. 1986. A comparison of reservoir oxygen predictions from one- and two-dimensional models. Lake Reserv Manage 2:98-103.

Mason DM, Goyke A, Brandt SB. 1995. A spatially explicit bioenergetics measure of habitat quality for adult salmonines: comparison between Lakes Michigan and Ontario. Can J Fish Aquat Sci 52:1572-1583.

Mauersberger P. 1983. General principles of deterministic water quality modeling. In: Orlob GT, editor. Mathematical modeling of water quality: streams, lakes, and reservoirs. Chichester: John Wiley and Sons. p 42-115.

McAfee WR. 1966. Rainbow trout. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 192-215.

McGauhey PH. 1968. Engineering management of water quality. New York: McGraw-Hill. 295 p.

McGinnis SM. 1984. Freshwater fishes of California. Berkeley: University of California Press. 316 p.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

McKechnie RJ. 1966. Spotted bass. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 366-370.

McKechnie RJ, Tharratt R. 1966. Green sunfish. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 399-401.

McMahon TE, Gebhart G, Maughan OE, Nelson PC. 1984. Habitat suitability index models and instream flow suitability curves: spotted bass. Washington (DC): US Department of the Interior, Fish and Wildlife Service. FWS/OBS-82/10.72. 41 p.

Michener RH, Schell DM. 1994. Stable isotope ratios as tracers in marine aquatic food webs. In: Lajtha K, Michener RH, editors. Stable isotopes in ecology and environmental science. Oxford: Blackwell Scientific Publications. p 138-157.

Microsoft. 1999. Microsoft® Excel 2000 Help. Available with software.

Miller KD, Kramer RH. 1971. Spawning and early life history of largemouth bass (*Micropterus salmoides*) in Lake Powell. In: Hall GE, editor. Reservoir fisheries and limnology. Washington (DC): American Fisheries Society. American Fisheries Society Special Publication nr 8. p 73-83.

Minns CK, Kelso JRM, Randall RG. 1996. Detecting the response of fish to habitat alterations in freshwater ecosystems. Can J Fish Aquat Sci 53(Suppl.1):403-414.

Mizutani H, Kabaya Y, Wada E. 1991. Nitrogen and carbon isotope compositions relate linearly in Cormorant tissues and its diet. Isotopenpraxis 27(4):166-168.

Monaco ME, Ulanowicz RE. 1997. Comparative ecosystem trophic structure of three U.S. mid-Atlantic estuaries. Mar Ecol Prog Series 161:239-254.

Morel-Seytoux HJ. 1990. Physical hydrology (ensemble of lecture notes and class handouts developed since 1977): interim set. Fort Collins: Hydrowar Reports Division. Report nr 86.5.

Morrissey WA. 1996. Science policy and federal ecosystem-based management. Ecol Appl 6(3):717-720.

Moyle PB. 1976. Inland fishes of California. Berkeley: University of California Press. 405 p.

Müller F. 1997. State-of-the-art in ecosystem theory. Ecol Modell 100:135-161.

Naimann RJ, Magnuson JJ, McKnight DM, Stanford JA. 1995. The freshwater imperative: a research agenda. Washington (DC): Island Press. 165 p.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

[NBS] National Biological Survey. 1994. The facts on the National Biological Survey: program summary and initiatives. US Department of the Interior. 13 p.

[NBS] National Biological Service. 1995. Research study plan: Shasta Lake limnology and fish habitat. Fort Collins: Midcontinent Ecological Science Center.

Ney JJ. 1993. Bioenergetics modeling today: growing pains on the cutting edge. Trans Am Fish Soc 122:736-748.

Niquil N, Arias-González JE, Delesalle B, Ulanowicz RE. 1999. Characterization of the planktonic food web of Takapoto Atoll lagoon, using network analysis. Oecologia 118:232-241.

Odum HT. 1957. Trophic structure and productivity of Silver Springs, Florida. Ecol Monogr 27(1):55-112.

Oksanen L, Fretwell SD, Arruda J, Niemelä P. 1981. Exploitation ecosystems in gradients of primary productivity. Am Nat 118(2):240-261.

O'Neill RV, DeAngelis DL, Waide JB, Allen TFH. 1986. A hierarchical concept of ecosystems. Princeton: Princeton University Press. 253 p.

Orlob GT, editor. 1983a. Mathematical modeling of water quality: streams, lakes, and reservoirs. Chichester: John Wiley and Sons. Future directions; p 503-509.

Orlob GT, editor. 1983b. Mathematical modeling of water quality: streams, lakes, and reservoirs. Chichester: John Wiley and Sons. One-dimensional models for simulation of water quality in lakes and reservoirs; p 227-273.

Orlob GT, editor. 1983c. Mathematical modeling of water quality: streams, lakes, and reservoirs. Chichester: John Wiley and Sons. Preface; p ix-x.

Ostrom PH, Colunga-Garcia M, Gage SH. 1997. Establishing pathways of energy flow for insect predators using stable isotope ratios: field and laboratory evidence. Oecologia 109(1):108-113.

Oziransky Y, Shteinman B. 1993. Assessment of turbulent processes in Lake Kinneret by the two-dimensional reservoir water quality model "BETTER." Wat Sci Technol 27(7/8):381-385.

Pace ML, Glasser JE, Pomeroy LR. 1984. A simulation analysis of continental shelf food webs. Mar Biol 82:47-63.

Paine RT. 1988. Food webs: road maps of interactions or grist for theoretical development? Ecology 69(6):1648-1654.

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Paul L. 1987. Influence of seiche-generated light field fluctuations on phytoplankton growth. Int Revue Gesam Hydrobiol 72(3):269-281.

Peters RH. 1977. The unpredictable problems of tropho-dynamics. Environ Biol Fish 2(2):97-101.

Peters RH. 1986. The role of prediction in limnology. Limnol Oceanogr 31(5):1143-1159.

Peterson BJ, Fry B. 1987. Stable isotopes in ecosystem studies. Annu Rev Ecol Syst 18:293-320.

Peterson B, Fry B, Deegan L, Hershey A. 1993. The trophic significance of epilithic algal production in a fertilized tundra river ecosystem. Limnol Oceanogr 38(4):872-878.

Pinnegar JK, Polunin NVC. 1999. Differential fractionation of $\delta^{13}C$ and $\delta^{15}N$ among fish tissues: implications for the study of trophic interactions. Funct Ecol 13:225-231.

Ploskey GR, Aggus LR, Bivin WM, Jenkins RM. 1986. Regression equations for predicting fish standing crop, angler use, and sport fish yield for United States reservoirs. Ann Arbor: US Fish and Wildlife Service. Administrative Report nr. 86-5. USFWS-GLFL/AR-86-5. 92 p.

Post JR, McQueen DJ. 1994. Variability in first-year growth of yellow perch (*Perca flavenscens*): predictions from a simple model, observations, and an experiment. Can J Fish Aquat Sci 51:2501-2512.

Puccia CJ, Levins RL. 1985. Qualitative modeling of complex systems: an introduction to loop analysis and time averaging. Cambridge (MA): Harvard University Press. 259 p.

Railsback SF. 1997. Guidance for using a bioenergetics model to assess effects of stream temperature. San Ramon: Pacific Gas and Electric Company. Report 001.8-97.8.

Raleigh RF, Hickman T, Soloman RC, Nelson PC. 1984. Habitat suitability information: rainbow trout. Washington (DC): US Department of the Interior, Fish and Wildlife Service. FWS/OBS-82/10.60. 64 p.

Rau GH. 1980. Carbon-13/carbon-12 variation in subalpine lake aquatic insects: food source implication. Can J Fish Aquat Sci 37:742-746.

Rau GH, Mearns AJ, Young DR, Olson RJ, Schafer HA, Kaplan IR. 1983. Animal $^{13}C/^{12}C$ correlates with trophic level in pelagic food webs. Ecology 64(5):1314-1318.

Reckhow KH. 1988. Empirical models for trophic state in southeastern U.S. lakes and reservoirs. Wat Resourc Bull 24(4):723-734.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Reckhow KH, Chapra SC. 1983. Engineering approaches for lake management. Boston: Butterworth Publishers. Volume 1, Data analysis and empirical modelling. 340 p.

Regier HA, Meisner JD. 1990. Anticipated effects of climate change on freshwater fishes and their habitat. Fisheries 15(6):10-15.

Reichman OJ, Pulliam HR. 1996. The scientific basis for ecosystem management. Ecol Appl 6(3):694-696.

Reuss DE. 1998. Personal communication. Coordinator for the Natural Resources Ecology Lab, Colorado State University , Fort Collins, CO.

Reynolds CS. 1984. The ecology of freshwater phytoplankton. Cambridge (MA): Cambridge University Press. 384 p.

Riera P, Richard P. 1996. Isotopic determination of food sources of *Crassostrea gigas* along a trophic gradient in the estuarine bay of Marennes-Oléron. Estuar Coast Shelf Sci 42:347-360.

Riley MJ, Stefan HG. 1988a. Development of the Minnesota lake water quality management model "MINLAKE." Lake Reserv Manage 4(2):73-83.

Riley MJ, Stefan HG. 1988b. MINLAKE: a dynamic lake water quality simulation model. Ecol Modell 43:155-182.

Rounds SA, Wood TM. 1998. Using CE-QUAL-W2 to assess the ammonia assimilative capacity of the Tualatin River, Oregon. In: Proceedings of the First Federal Interagency Hydrologic Modeling Conference. Las Vegas: Subcommittee on Hydrology of the Interagency Advisory Committee on Water Data. p 2-133 to 2-140.

Rounds SA, Wood TM, Lynch DL. 1998. Modeling discharge, temperature, and water quality in the Tualatin River, Oregon. Portland: US Geological Survey. US Geological Survey Water-Open File Report 98-186. 122 p.

Rounick JS, Hicks BJ. 1985. The stable carbon isotope ratios of fish and their invertebrate prey in four New Zealand rivers. Freshwater Biol 15:207-214.

Rounick JS, Winterbourn MJ. 1986. Stable carbon isotopes and carbon flow in ecosystems. BioScience 36(3):171-177.

Rudstam LG, Magnuson JJ. 1985. Predicting the vertical distribution of fish populations: analysis of cisco, *Coregonus artedii*, and yellow perch, *Perca flavescens*. Can J Fish Aquat Sci 42:1178-1188.

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Ryder RA. 1982. The morphoedaphic index – use, abuse, and fundamental concepts. Trans Am Fish Soc 111:154-164.

Scheffer M. 1991. Fish and nutrients interplay determines algal biomass: a minimal model. Oikos 62:271-282.

Schindler DE, Carpenter SR, Cole JJ, Kitchell JF, Pace ML. 1997. Influence of food web structure on carbon exchange between lakes and the atmosphere. Science 277(5323):248-251.

Seip KL, Ibrekk H. 1988. Regression equations for lake management – how far do they go? Verh Int Verein Limnol 23:778-785.

Siefert RE. 1972. First food of larval yellow perch, white sucker, bluegill, emerald shiner, and rainbow smelt. Trans Am Fish Soc 101(2):219-225.

Slobodkin LB. 1968. How to be a predator. Am Zool 8:43-51.

Smith CL. 1996. Commentary on hatcheries: an artificial solution. In: Cone J, Ridlington S, editors. The northwest salmon crisis: a documentary history. Corvallis: Oregon State University Press. p 72-74.

Staley J. 1966. Brown trout. In: Calhoun A, editor. Inland fisheries management. California Department of Fish and Game. p 233-242.

Stalnaker CB. 1994. Evolution of instream flow modeling. In: Calow P, Petts GE, editors. The rivers handbook: hydrological and ecological principles. Volume 2. Oxford (UK): Blackwell Scientific Publications. p 276-286.

Stanford JA, Ward JV, Liss WJ, Frissell CA, Williams RN, Lichatowich JA, Coutant CC. 1996. A general protocol for restoration of regulated rivers. Regul Rivers 12:391-413.

Stefan HG, Ambrose RB, Dortch MS. 1989. Formulation of water quality models for streams, lakes, and reservoirs: modeler's perspective. Vicksburg: US Army Corps of Engineers, Waterways Experiment Station. Miscellaneous Paper E-89-1. 82 p.

Stockwell JD, Johnson BM. 1997. Refinement and calibration of a bioenergetics-based foraging model for kokanee (*Oncorhynchus nerka*). Can J Fish Aquat Sci 54:2659-2676.

Stone, L. 1874. Report of the Commissioner of the United States Commission of Fish and Fisheries for 1872 and 1873. p 168-169, 180-182. (Cited in Heizer 1973).

Stone L. 1896. The artificial propagation of salmon on the Pacific coast of the United States, with notes on the natural history of the Quinnat salmon. Bull US Fish Commission 16:205-220 (Cited in Cone and Ridlington, eds. 1996).

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Straskraba M. 1994. Ecotechnological models for reservoir water quality management. Ecol Modell 74:1-38.

Strong M, Dahl B, Göranson G. 1993. Preface. In: Clarke R. Water: the international crisis. Cambridge (MA): The MIT Press. p ix-xii.

Stuber RJ, Gebhart G, Maughan OE. 1982a. Habitat suitability index models: bluegill. Washington (DC): US Department of the Interior, Fish and Wildlife Service. FWS/OBS-82/10.8. 26 p.

Stuber RJ, Gebhart G, Maughan OE. 1982b. Habitat suitability index models: green sunfish. Washington (DC): US Department of the Interior, Fish and Wildlife Service. FWS/OBS-82/10.15. 28 p.

Summer RM, Alonso CV, Young RA. 1990. Modeling linked watershed and lake processes for water quality management decision. J Environ Quality. 19:421-427.

Sydeman WJ, Hobson KA, Pyle P, McLaren EB. 1997. Trophic relationships among seabirds in central California: combined stable isotope and conventional dietary approach. Condor 99:327-336.

Takeuchi K, Hamlin M, Kunzewicz ZW, Rosbjerg D, Simonovic, editors. 1998. Sustainable reservoir development and management: report by the IAHS/ICWRS project team. International Association of Hydrological Sciences (IAHS) Publication No. 251. 190 p.

Taft AC, Murphy GI. 1950. The life history of the Sacramento squawfish (*Ptychocheilus grandis*). Calif Fish Game 36(2):147-164.

Tan FC, Strain PM. 1983. Sources, sinks and distribution of organic carbon in the St. Lawrence Estuary, Canada. Geochim Cosmochim Acta 47:125-132.

Terrell JW, McMahon TE, Inskip PD, Raleigh RF, Williamson KL. 1982. Habitat suitability index models: appendix A. Guidelines for riverine and lacustrine applications of fish HSI models with the habitat evaluation procedures. Fort Collins: US Fish and Wildlife Service. FWS/OBS-82/10.A. 54 p.

Theurer FD, Voos KA, Miller WJ. 1984. Instream water temperature model. US Fish and Wildlife Service. Instream Flow Information Paper nr 16. FWS/OBS-84/15.

Thomann RV. 1998. The future "golden age" of predictive models for surface water quality and ecosystem management. J Environ Engin 124(2):94-103.

Thomas JW. 1996. Forest Service perspective on ecosystem management. Ecol Appl 6(3):703-705.

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Thornton KW, Lessum AS. 1978. A temperature algorithm for modifying biological rates. Trans Am Fish Soc 107(2):284-287.

Tieszen LL, Boutton TW, Tesdahl KG, Slade NA. 1983. Fractionation and turnover of stable carbon isotopes in animal tissues: implications for $\delta^{13}C$ analysis of diet. Oecologia 57:32-37.

Turner JL. 1966a. Distribution and food habits of the centrarchid fishes in the Sacramento-San Joaquin Delta. In Turner JL, Kelley DW, editors. Ecological studies of the Sacramento-San Joaquin Delta, part II. Fishes of the Delta. California Department of Fish and Game. California Department of Fish and Game Bulletin 136. p 144-153.

Turner JL. 1966b. Distribution of threadfin shad, *Dorosoma petenense*; tule perch, *Hysterocarpus traskii*; sculpin spp. and crayfish spp. in the Sacramento-San Joaquin Delta. In Turner JL, Kelly DW, editors. Ecological studies of the Sacramento-San Joaquin Delta, part II. Fishes of the Delta. California Department of Fish and Game. California Department of Fish and Game Bulletin 136. p 160-168.

Ulanowicz RE. 1986. Growth and development: ecosystems phenomenology. New York: Springer-Verlag. 203 p.

[USACE] US Department of the Army, Corps of Engineers. 1995a. CE-QUAL-R1: a numerical one-dimensional model of reservoir water quality: user's manual. Washington (DC): Waterways Experiment Station. Instruction Report E-82-1. 427 p. + appendices.

[USACE] US Department of the Army, Corps of Engineers. 1995b. CE-QUAL-W2 workshop. Boulder: Waterways Experiment Station. 457 p.

[USBR] US Department of the Interior, Bureau of Reclamation. 1983. Central Valley Project: Shasta/Trinity River Divisions. US Department of the Interior, Bureau of Reclamation, Mid-Pacific Region. GPO 583-058/11019. 16 p.

[USBR] US Department of the Interior, Bureau of Reclamation. 1990. Appendices to Shasta outflow temperature control, planning report/environmental statement. Appendices A and B. US Department of the Interior, Bureau of Reclamation, Mid-Pacific Region. Revised May 1991. 307 p.

[USBR] US Department of the Interior, Bureau of Reclamation. 1994. Fact sheet: Shasta Dam, Shasta Powerplant, Shasta Lake, Keswick Dam, Powerplant, and Reservoir. Sacramento: US Department of the Interior, Bureau of Reclamation, Mid-Pacific Region. GPO-683-062/21026. 5 p.

Van Den Avyle MJ, Petering RW. 1988. Inundated timber as nursery habitat for larval gizzard and threadfin shad in a new pumped storage reservoir. Trans Am Fish Soc 117:84-89.

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

Vander Zanden MJ, Cabana G, Rasmussen JB. 1997. Comparing trophic position of freshwater fish calculated using stable nitrogen isotope ratios ($\delta^{15}N$) and literature dietary data. Can J Fish Aquat Sci 54:1142-1158.

Vander Zanden MJ, Hulshof M, Ridgway MS, Rasmussen JB. 1998. Application of stable isotope techniques to trophic studies of age-0 smallmouth bass. Trans Am Fish Soc 127:729-739.

Vander Zanden MJ, Rasmussen JB. 1996. A trophic position model of pelagic food webs: impact on contaminant bioaccumulation in lake trout. Ecol Monogr 66(4):451-477.

Van Winkle W, Rose KA, Chambers RC. 1993a. Individual-based approach to fish population dynamics: an overview. Trans Am Fish Soc 122:397-403.

Van Winkle W, Rose KA, Shuter BJ, Jager HI, Holcomb BD. 1997. Effects of climatic temperature change on growth, survival, and reproduction of rainbow trout: predictions from a simulation model. Can J Fish Aquat Sci 54:2526-2542.

Van Winkle W, Rose KA, Winemiller KO, DeAngelis DL, Christensen S, Otto RG, Shuter BJ. 1993b. Linking life history theory, environmental setting, and individual-based modeling to compare responses of different fish species to environmental change. Trans Am Fish Soc 122:459-466.

Vermeyen T. 1997. Pre-temperature control device selective withdrawal evaluation for Shasta Lake for the years 1995 and 1996. Denver: U.S. Bureau of Reclamation, Water Resources Research Laboratory. 13 p.

Vermeyen TB. 1998a. First year performance of the Shasta Dam temperature control device. Denver: USBR Power Operations and Maintenance Workshop. 10 p.

Vermeyen T. 1998b. US Bureau of Reclamation, Water Resources Research Laboratory. Personal communication on August 10, 1998.

Vogele LE. 1975. The spotted bass. In: Clepper H, editor. Black bass biology and management. Washington (DC): Sport Fishing Institute. p 34-45.

Von Geldern C, Mitchell DF. 1975. Largemouth bass and threadfin shad in California. In: Clepper H, editor. Black bass biology and management. Washington (DC): Sport Fishing Institute. p 436-449.

Wainwright SC, Fuller CM, Michener RH, Richards RA. 1996. Spatial variation of trophic position and growth rate of juvenile striped bass (*Morone saxatilis*) in the Delaware River. Can J Fish Aquat Sci 53:685-692.

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Walker WW. 1986. Models and software for reservoir eutrophication assessment. Lake Reserv Manage 2:143-148.

Walters C, Christensen V, Pauly D. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Rev Fish Biol Fish 7:139-172.

Wang JCS. 1986. Fishes of the Sacramento-San Joaquin estuary and adjacent waters, California: a guide to the early life histories. A cooperative study by the California Department of Water Resources, California Department of Fish and Game, U.S. Bureau of Reclamation and U.S. Fish and Wildlife Service. Technical Report 9. FS/B10-4ATR 86-9.

Wanjala BS, Tash JC, Matter WJ, Ziebell CD. 1986. Food and habitat use by different sizes of largemouth bass (*Micropterous salmoides*) in Alamo Lake, Arizona. J Freshwater Ecol 3(3):359-369.

Ward JV, Stanford JA. 1987. The ecology of regulated streams: past accomplishments and directions for future research. In: Craig JF, Kemper JB, editors. Regulated streams: advances in ecology. New York: Plenum Press. p 391-409.

Watanabe M, Harleman DRF, Vasiliev OF. 1983. Two- and three-dimensional mathematical models for lakes and reservoirs. In: Orlob GT, editor. Mathematical modeling of water quality: streams, lakes, and reservoirs. Chichester: John Wiley and Sons. p 274-336.

Webster's new world dictionary of the American language. 1984. New York: Warner Books, Inc. 696 p.

Wetzel RG. 1983. Limnology. 2nd ed. New York: Harcourt Brace College Publishers. 767 p.

Whipple S. 1998. Path-based network unfolding: a solution for the problem of mixed trophic and non-trophic processes in trophic dynamic analysis. J Theoret Biol 190:263-276.

Whitledge GW, Hayward RS, Noltie DB, Wang N. 1998. Testing bioenergetics models under feeding regimes that elicit compensatory growth. Trans Am Fish Soc 127:740-746.

Wiedenhoeft RV. 1999. Ecology and engineering: changing paradigms. J Engin Educ 88(1):15-18.

Willis AJ. 1997. The ecosystem: an evolving concept viewed historically. Funct Ecol 11:268-271.

Wilson JA, French J, Kleban P, McKay SR, Townsend R. 1991. Chaotic dynamics in a multiple species fishery: a model of community predation. Ecol Modell 58:303-322.

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Winter TC. 1981. Uncertainties in estimating the water balance of lakes. Wat Resourc Bull 17(1):82-115.

Wootton RJ. 1990. Ecology of teleost fishes. London: Chapman and Hall. 404 p.

Wuethrich B. 1995. Deliberate flood renews habitats. Science 272:344-345.

Yaffee SL. 1996. Ecosystem management in practice: the importance of human institutions. Ecol Appl 6(3):724-727.

Yoshioka T, Wada E, Hayashi H. 1994. A stable isotope study on seasonal food web dynamics in a eutrophic lake. Ecology 75(3):835-846.

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ABBREVIATIONS

1D	=	one-dimensional
2D	=	two-dimensional
3D	=	three-dimensional
ABSE	=	absolute mean error (see Equation 31)
ADCP	=	acoustic Doppler current profiler
ADP	=	acoustic Doppler profiler
AINS	=	aquatic insects
ALK	=	alkalinity
ANOVA	=	analysis of variance
ASCE	=	American Society of Civil Engineers
BENTH	=	benthos
BETA	=	fraction of incident solar radiation absorbed at the water surface (parameter in CE-QUAL-W2)
BOD	=	biochemical oxygen demand
BRD	=	Biological Resources Division (of the USGS)
C	=	Celsius
CAR	=	carbon species (inorganic, CO_2, bicarbonate, or carbonate)
CBHE	=	coefficient of bottom heat exchange
CDFG	=	California Department of Fish and Game
cfs	=	cubic feet per second
CHLA	=	chlorophyll *a*
CL	=	chloride
cm	=	centimeters
cms	=	cubic meters per second
COD	=	chemical oxygen demand
COLI	=	coliforms or fecal coliform
CRA	=	crayfish
CSU	=	Colorado State University
d	=	day
$\delta^{13}C$	=	carbon isotope ratio
$\delta^{15}N$	=	nitrogen isotope ratio
dd	=	degree day
DET	=	detritus
DO	=	dissolved oxygen
DOM	=	dissolved organic matter
DS	=	dissolved solids
EXH2O	=	light extinction for water (parameter in CE-QUAL-W2)

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FE	=	iron
g	=	gram
ha	=	hectare
Hg	=	mercury
HSI	=	habitat suitability index
IAHS	=	International Association of Hydrological Sciences
J	=	joules
JD	=	Julian day
K	=	Kelvin
km	=	kilometers
LARM	=	laterally averaged reservoir model
m	=	meters
MAC	=	macrophytes
MEI	=	morphoedaphic index
mg	=	milligrams
mm	=	millimeters
MN	=	manganese
MSE	=	mean squared error
MVT	=	multivariable testing
NBS	=	U.S. Department of the Interior, National Biological Service
nm	=	no measurements
NPP	=	net primary productivity
NUTS	=	nutrients (any components of nitrogen and phosphorus)
POM	=	particulate organic matter
POS	=	primary organic source
PQC	=	place QIN control (switch for CE-QUAL-W2)
RI	=	reliability index (see Equation 32)
RMSE	=	root mean squared error (see Equation 30)
RTR	=	relative thermal resistance
s	=	seconds
SAL	=	salinity
SD	=	Secchi disk measurements
SI	=	silica
SS	=	suspended solids
SU	=	sulfur species
TCD	=	temperature control device
TDS	=	total dissolved solids
TH	=	thermal habitat
THA	=	thermal habitat area
THV	=	thermal habitat volume
TINS	=	terrestrial insects
TL	=	total length
TRST	=	trophic state
TSED	=	ground temperature (parameter in CE-QUAL-W2)
TSS	=	total suspended solids
USACE	=	U.S. Department of the Army, Corps of Engineers

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USBR	=	U.S. Department of the Interior, Bureau of Reclamation
USFS	=	U.S. Department of Agriculture, Forest Service
USGS	=	U.S Department of the Interior, Geological Survey
W	=	watt
WSC	=	wind sheltering coefficient (parameter in CE-QUAL-W2)
WT	=	water temperature
ZOO	=	zooplankton

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Appendix A. Table of summary statistics for calibration runs and figures comparing *Oldbath* and *Newcalib* with measured water temperatures

Tables 40 through 45 summarize the r^2 values, root mean squared error (RMSE), and absolute mean error (ABSE) for the different runs according to segment number and Julian day. See Table 10 for description of each run and Figure 12 for location of each segment. Shading indicates maximum r^2 or minimum RMSE or ABSE for each segment or Julian day.

Table 40. r^2 values for *Oldbath*, *Newbath*, *Newflows*, *Newtemps*, *Newout*, and *Newcalib* according to segment number

Segment	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
13	0.949	0.970	0.971	0.970	0.970	0.981
16	0.952	0.971	0.972	0.972	0.971	0.984
19	0.953	0.972	0.973	0.973	0.973	0.981
41	0.953	0.972	0.971	0.971	0.971	0.980
54	0.942	0.965	0.965	0.965	0.965	0.984

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Table 41. RMSE (°C) for *Oldbath*, *Newbath*, *Newflows*, *Newtemps*, *Newout*, and *Newcalib* according to segment number

Segment	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
13	1.11	0.75	0.74	0.74	0.73	0.67
16	1.19	0.81	0.80	0.80	0.78	0.62
19	1.38	0.91	0.90	0.90	0.87	0.64
41	1.17	0.76	0.76	0.76	0.76	0.74
54	1.17	0.95	0.94	0.94	0.93	0.71

Table 42. ABSE (°C) for *Oldbath*, *Newbath*, *Newflows*, *Newtemps*, *Newout*, and *Newcalib* according to segment number

Segment	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
13	0.85	0.58	0.57	0.57	0.57	0.57
16	0.92	0.62	0.62	0.61	0.59	0.51
19	1.10	0.71	0.70	0.70	0.67	0.51
41	0.88	0.57	0.58	0.58	0.57	0.59
54	1.07	0.73	0.72	0.72	0.69	0.53

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Table 43. r^2 values for *Oldbath*, *Newbath*, *Newflows*, *Newtemps*, *Newout*, and *Newcalib* according to Julian day

Julian day	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
130	0.959	0.954	0.954	0.955	0.956	**0.969**
170	0.971	**0.979**	**0.979**	**0.979**	0.978	0.977
206	0.968	0.987	0.987	0.987	0.986	**0.989**
241	0.965	0.980	0.980	0.980	0.980	**0.988**
264	0.958	0.968	0.968	0.968	0.968	**0.976**
291	0.938	0.953	0.953	0.953	0.953	**0.975**
317	0.896	0.907	0.904	0.904	0.903	**0.928**

Table 44. RMSE (°C) for *Oldbath*, *Newbath*, *Newflows*, *Newtemps*, *Newout*, and *Newcalib* according to Julian day

Julian day	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
130	**0.67**	0.75	0.74	0.74	0.75	0.79
170	0.67	0.55	0.54	0.54	**0.53**	0.67
206	1.31	0.74	0.73	0.73	0.71	**0.62**
241	1.50	0.95	0.94	0.94	0.92	**0.69**
264	1.61	1.08	1.07	1.07	1.05	**0.75**
291	1.38	0.90	0.90	0.90	0.87	**0.70**
317	1.18	0.80	0.80	0.80	0.78	**0.54**

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Table 45. ABSE (°C) for *Oldbath*, *Newbath*, *Newflows*, *Newtemps*, *Newout*, and *Newcalib* according to Julian day

Julian day	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
130	0.46	0.47	0.47	0.47	0.48	0.67
170	0.50	0.39	0.38	0.38	0.36	0.54
206	1.06	0.58	0.57	0.57	0.54	0.50
241	1.25	0.78	0.77	0.77	0.74	0.54
264	1.41	0.93	0.91	0.91	0.88	0.56
291	1.11	0.71	0.71	0.71	0.69	0.59
317	0.86	0.65	0.66	0.66	0.64	0.41

Tables 46 through 50 summarize the maximum differences between modeled water temperatures and measured water temperatures for the different runs according to segment number and Julian day. See Table 10 for description of each run and Figure 12 for location of each segment. Shading indicates the smallest maximum differences from measured temperatures for each Julian day.

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Table 46. Maximum temperature differences (°C) between modeled and measured water temperature for Segment 13 (Pit River arm)

Julian day	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
130.6	2.14	2.22	2.18	2.19	2.18	**1.76**
170.6	2.03	1.63	**1.62**	**1.62**	1.64	2.09
206.6	2.83	1.30	**1.29**	1.30	**1.29**	1.76
241.6	2.64	1.66	1.64	1.65	1.62	**1.29**
264.6	2.38	1.79	1.76	1.77	1.76	**1.62**
291.6	2.34	1.68	1.66	1.66	1.65	**1.27**
317.7	2.15	1.55	1.55	1.54	1.53	**1.21**

Table 47. Maximum temperature differences (°C) between modeled and measured water temperature for Segment 16 (McCloud River confluence with Pit River)

Julian day	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
130.6	2.285	2.355	2.325	2.325	2.325	**1.745**
171.6	1.695	1.415	**1.405**	**1.405**	1.425	1.965
206.6	3.19	1.64	1.67	1.65	1.69	**1.25**
241.6	2.885	1.79	1.78	1.78	1.77	**1.485**
264.6	2.72	1.98	1.94	1.95	1.94	**1.32**
291.6	2.40	1.81	1.8	1.79	1.67	**1.29**
317.7	1.90	1.20	1.19	1.19	1.18	**0.91**

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Table 48. Maximum temperature differences (°C) between modeled and measured water temperature for Segment 19 (Main Lake)

Julian day	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
170.6	2.13	1.61	1.61	1.61	1.61	2.28
207.6	2.965	1.64	1.63	1.62	1.61	1.88
242.6	2.94	2.03	2.01	2.01	1.99	1.11
264.6	2.61	1.93	1.92	1.91	1.77	1.38
291.6	2.76	2.19	2.17	2.17	2.04	1.45
317.7	2.91	2.31	2.30	2.29	2.18	1.57

Table 49. Maximum temperature differences (°C) between modeled and measured water temperature for Segment 41 (McCloud River arm)

Julian day	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
130.6	1.225	1.125	1.135	1.135	1.155	1.49
172.6	1.975	1.545	1.615	1.615	1.615	2.395
206.6	2.83	1.34	1.26	1.24	1.25	1.48
241.6	2.77	2.07	2.01	2.02	2.01	2.14
264.6	2.69	2.31	2.29	2.29	2.29	2.28
291.6	2.60	1.98	1.91	1.92	1.91	1.57
317.7	2.24	1.63	1.63	1.63	1.61	1.25

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Table 50. Maximum temperature differences (°C) between modeled and measured water temperature for Segment 54 (Sacramento River arm)

Julian day	*Oldbath*	*Newbath*	*Newflows*	*Newtemps*	*Newout*	*Newcalib*
130.6	2.26	2.46	2.45	2.46	2.51	2.35
170.6	1.71	1.51	1.51	1.50	1.47	0.855
207.6	3.89	2.47	2.48	2.49	2.49	2.55
242.6	2.985	2.225	2.225	2.235	2.235	2.185
264.6	2.85	2.105	2.075	2.075	2.075	1.775
291.6	3.08	1.83	1.81	1.81	1.79	1.57
317.7	2.05	1.61	1.60	1.61	1.60	1.23

Figure 35 shows water temperature profiles for the run prior to calibration process (*Oldbath*), the calibrated run (*Newcalib*), and measured water temperatures with depth for Segment 16 (confluence of McCloud River with Pit River) on Julian days 130, 171, 206, 241, 264, 291 and 317. Figures 36 and 37 are isothermal plots of predicted temperatures with *Oldbath* and *Newcalib*, respectively, as compared with measured isotherms for Segment 16 in 1995.

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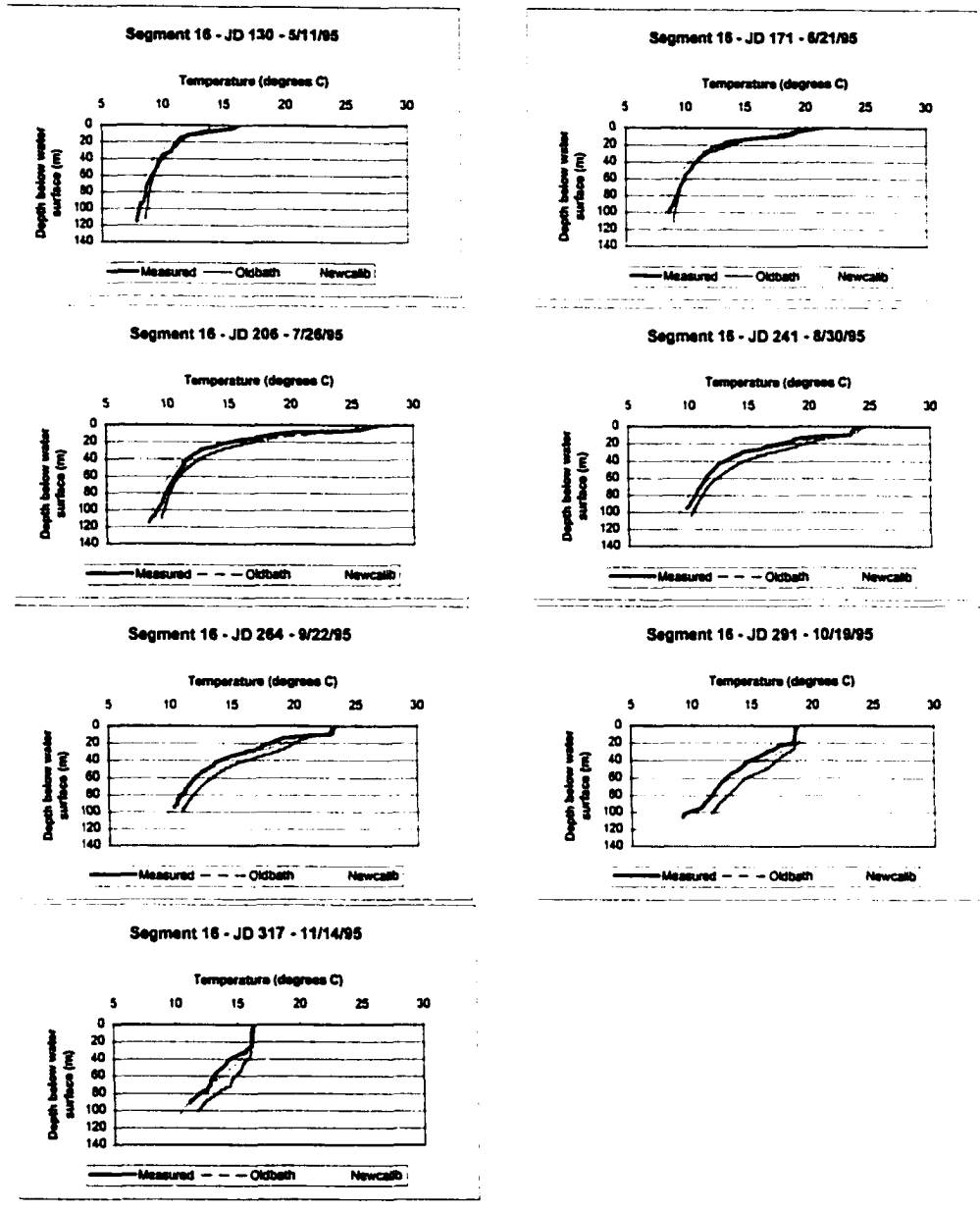

Figure 35. Water temperature profiles for *Oldbath, Newcalib* and measured water temperatures with depth for Segment 16

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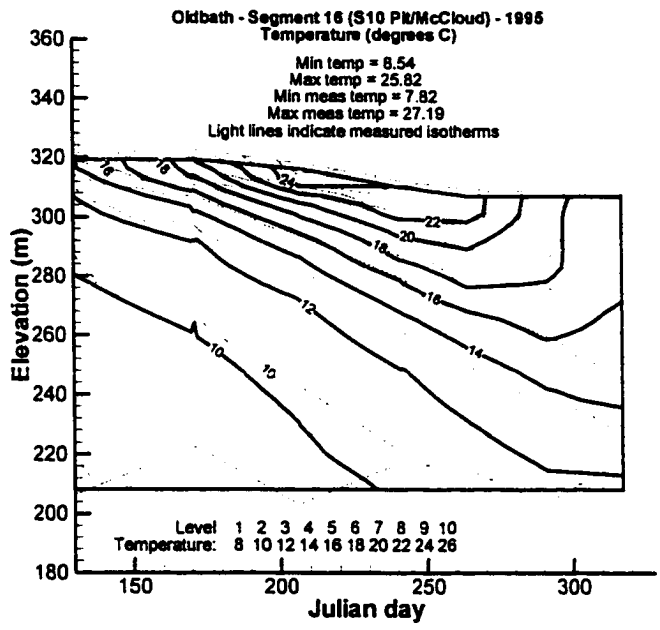

Figure 36. Isothermal plot for *Oldbath* and measured temperatures in 1995

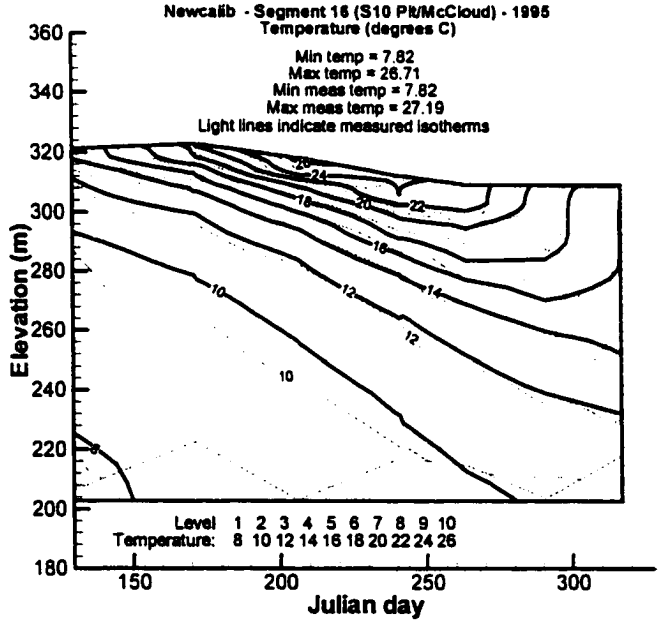

Figure 37. Isothermal plot for *Newcalib* and measured temperatures in 1995

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APPENDIX B. Table of summary statistics from all runs performed for water temperature validation and sensitivity analysis, and figures of initial and 'best' validation runs for 1968, 1974, 1976 and 1995.

Table 51 shows summary statistics for all water temperature validation and sensitivity analysis runs. Shading on figures indicates runs for a particular year with the best statistics for the goodness-of-fit indicator in that column.

Figures 38 through 45 show initial and 'best' validation runs for 1968, 1974, 1976 and 1995 as plotted against measured water temperatures at Shasta Dam in corresponding years.

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Table 51. Summary statistics for water temperature validation and sensitivity analysis

Validation runs									[redacted]						
Run	TSED	CBHE	EXH20	BETA	Tin	Tair	Wind	extra laye	Max+diff	Max-diff	MaxRMS	AvgRMS	ABSE	r-squared	Comments
1995	15.9	7.00E-08	0.4	0.45	nc	nc	nc	no	2.791	-2.228	1.245	0.858	0.627	0.9408	bottom too warm; bottom layer off 5/11, 5/25, 6/19
1995b	15.9	7.00E-09	0.4	0.45	nc	nc	nc	no	2.791	-2.198	1.165	0.812	[redacted]	0.9464	bottom better; bottom layer off all months
1995c	10	8.00E-08	0.4	0.45	nc	nc	nc	no	2.831	-2.128	[redacted]				bottom layer off all months
1995d	15.9	7.00E-08	0.45	0.45	nc	nc	nc	no	3.071	-2.192	1.178	0.845	0.616	0.9422	about same
1995e	15.9	7.00E-08	0.4	0.36	nc	nc	nc	no	2.588	-2.338	1.299	0.866	0.634	0.9406	bottom too warm, bottom layer off 5/11, 5/25, 6/19
1995f	15.9	7.00E-08	1.5	0.72	nc	nc	nc	no	5.151	[redacted]	1.371	0.928	0.621	0.9408	bottom too warm; surface looks better
1995g	15.9	7.00E-08	1	0.61	nc	nc	nc	no	4.751	-2.092	1.258	0.887	0.607	0.9422	bottom too warm; bottom layer off 5/11, 5/25, 6/19
1995h	15.9	7.00E-08	0.5	0.42	nc	nc	nc	no	3.251	-2.182	1.140	0.838	0.610	0.9431	about same
1995i	15.9	7.00E-08	0.2	0.18	nc	nc	nc	no	3.152	-4.538	2.334	1.314	0.912	0.8961	
1995j	15.9	7.00E-08	0.3	0.28	nc	nc	nc	no	[redacted]	-3.098	1.642	0.985	0.712	0.9298	bottom layer too cole 5/11, 5/25, 6/19
1995k	15.9	7.00E-08	0.4	0.45	nc	nc	nc	yes	2.891	-2.208	1.225	0.845	0.620	0.9424	bottom too warm; top too cold
Run	TSED	CBHE	EXH20	BETA	Tin	Tair	Wind	extra laye	Max+diff	Max-diff	MaxRMS	AvgRMS	ABSE	r-squared	Comments
1976	15.9	7.00E-08	0.4	0.45	nc	nc	nc	no	5.854	-4.344	2.030	1.349	0.961	0.9379	bottom OK; bottom layer too cold all months'
1976b	6.09	7.00E-08	0.4	0.45	nc	nc	nc	no	5.894	-4.324	1.985	1.329	1.009	0.9425	bottom too cold; bottom layer more off
1976c	15.9	7.00E-07	0.4	0.45	nc	nc	nc	no	5.954	-4.814	2.887	2.227	1.879	0.8740	bottom too warm; bottom layer OK
1976d	15.9	7.00E-09	0.4	0.45	nc	nc	nc	no	5.864	-4.304	2.011	1.349	1.010	0.9402	bottom too cold; bottom layer off all months
1976e	10	8.00E-08	0.4	0.45	nc	nc	nc	no	5.864	-4.334	1.989	1.321	0.969	0.9413	bottom layer off all months; worse than 1976
1976f	15.9	7.00E-08	0.4	0.45	-2.5 all	nc	nc	no	8.178	[redacted]	4.382	3.687	2.976	0.8508	everything too low
1976g	15.9	7.00E-08	0.45	0.45	nc	nc	nc	no	5.784	-4.224	1.993	1.323	0.943	0.9401	bottom layer too cold, a little better
1976h	15.9	7.00E-08	0.4	0.45	-1 br1&3	nc	nc	no	7.698	-1.719	4.097	3.322	2.735	0.8666	everything too low
1976i	15.9	7.00E-08	0.4	0.36	nc	nc	nc	no	6.014	-4.444	2.089	1.380	0.980	0.9351	bottom layer too cold
1976j	15.9	7.00E-08	1	0.61	nc	nc	nc	no	5.094	-3.586	1.772	1.180	0.848	0.9516	better; bottom layer off all months
1976k	15.9	7.00E-08	1.5	0.72	nc	nc	nc	no	[redacted]	-3.336	[redacted]			0.9545	bottom looks good, bottom layer off
1976l	15.9	7.00E-08	0.4	0.45	nc	-1	nc	no	6.324	-4.244	2.063	1.352	0.963	0.9354	looks good; bottom layer great!
1976m	15.9	7.00E-08	0.4	0.45	nc	nc	-1	no	5.124	-3.947	1.866	1.187	0.881	0.9586	looks good; bottom layer better
1976n	15.9	7.00E-08	0.4	0.45	nc	-0.5	-0.5	no	5.674	-4.064	1.903	1.239	0.903	0.9499	bottom layer a little cold
1976o	15.9	7.00E-08	0.4	0.45	nc	nc	-0.5	no	5.454	-4.084	1.916	1.234	0.903	[redacted]	bottom layer a little cold; surface a little better
1976p	15.9	7.00E-08	0.2	0.18	nc	nc	nc	no	6.854	-4.581	2.462	1.674	1.196	0.9058	looks better; bottom layer looks ok
1976q	15.9	7.00E-08	0.3	0.28	nc	nc	nc	no	6.354	-4.724	2.253	1.491	1.061	0.9249	bottom layer too cold,
1976r	15.9	7.00E-08	0.4	0.45	nc	nc	nc	yes	6.214	-4.394	2.144	1.360	0.968	0.9359	bottom looks good, bottom layer off
Run	TSED	CBHE	EXH20	BETA	Tin	Tair	Wind	extra laye	Max+diff	Max-diff	MaxRMS	AvgRMS	ABSE	r-squared	Comments
1974	15.9	7.00E-08	0.4	0.45	nc	nc	nc	no	3.777	-4.013	1.995	1.363	1.099	0.9197	bottom too wwarm, top ok
1974b	15.9	7.00E-08	0.4	0.45	nc	nc	-0.5	no	[redacted]						surface better, bottom too warm
1974c	15.9	7.00E-08	0.4	0.36	nc	nc	nc	no	3.907	-3.943	2.046	1.393	1.121	0.9149	bottom too warm
1974d	15.9	7.00E-08	0.3	0.28	nc	nc	nc	no	4.347	-4.328	2.353	1.544	1.218	0.8922	bottom too warm; surface too cold
1974e	10	7.00E-08	0.4	0.45	nc	nc	nc	no	3.797	-3.983	1.865	1.272	1.013	0.9264	bottom too warm; surface too cold
1974f	15.9	7.00E-08	0.4	0.45	nc	nc	nc	yes	3.887	-3.793	2.002	1.359	1.104	0.9184	bottom too warm, surface too cold
Run	TSED	CBHE	EXH20	BETA	Tin	Tair	Wind	extra laye	Max+diff	Max-diff	MaxRMS	AvgRMS	ABSE	r-squared	Comments
1968	15.9	7.00E-08	0.4	0.45	nc	nc	nc	no	5.043	-3.526	1.946	1.495	1.192	0.8907	bottom too warm, top too cold
1968b	15.9	7.00E-09	0.4	0.45	nc	nc	nc	no	5.063	-3.466	1.776	1.403	1.070	0.9017	bottom better; bottom layer too cold, profile shape worse
1968c	15.9	7.00E-08	0.4	0.45	nc	-1	nc	no	5.493	[redacted]	1.959	1.543	1.170	0.8853	not much difference
1968d	15.9	7.00E-08	0.4	0.45	nc	-0.5	-0.5	no	4.953	-3.286	1.746	1.361	[redacted]	0.9112	looks about same as 1968
1968e	15.9	7.00E-08	0.4	0.45	nc	nc	-0.5	no	[redacted]	-3.376	[redacted]		1.055	[redacted]	surface looks better
1968f	15.9	7.00E-08	0.4	0.36	nc	nc	nc	no	4.963	-3.626	1.946	1.496	1.197	0.8899	bottom too warm, top too cold
1968g	15.9	7.00E-08	0.3	0.28	nc	nc	nc	no	4.876	-4.286	2.040	1.547	1.242	0.8794	bottom too warm, top too cold
1968h	10	7.00E-08	0.4	0.45	nc	nc	nc	no	5.063	-3.456	1.794	1.417	1.095	0.9007	bottom ok but shape wrong
1968i	15.9	7.00E-08	0.4	0.45	nc	nc	nc	yes	5.026	-3.436	1.973	1.509	1.194	0.8887	bottom too warm; top too cold

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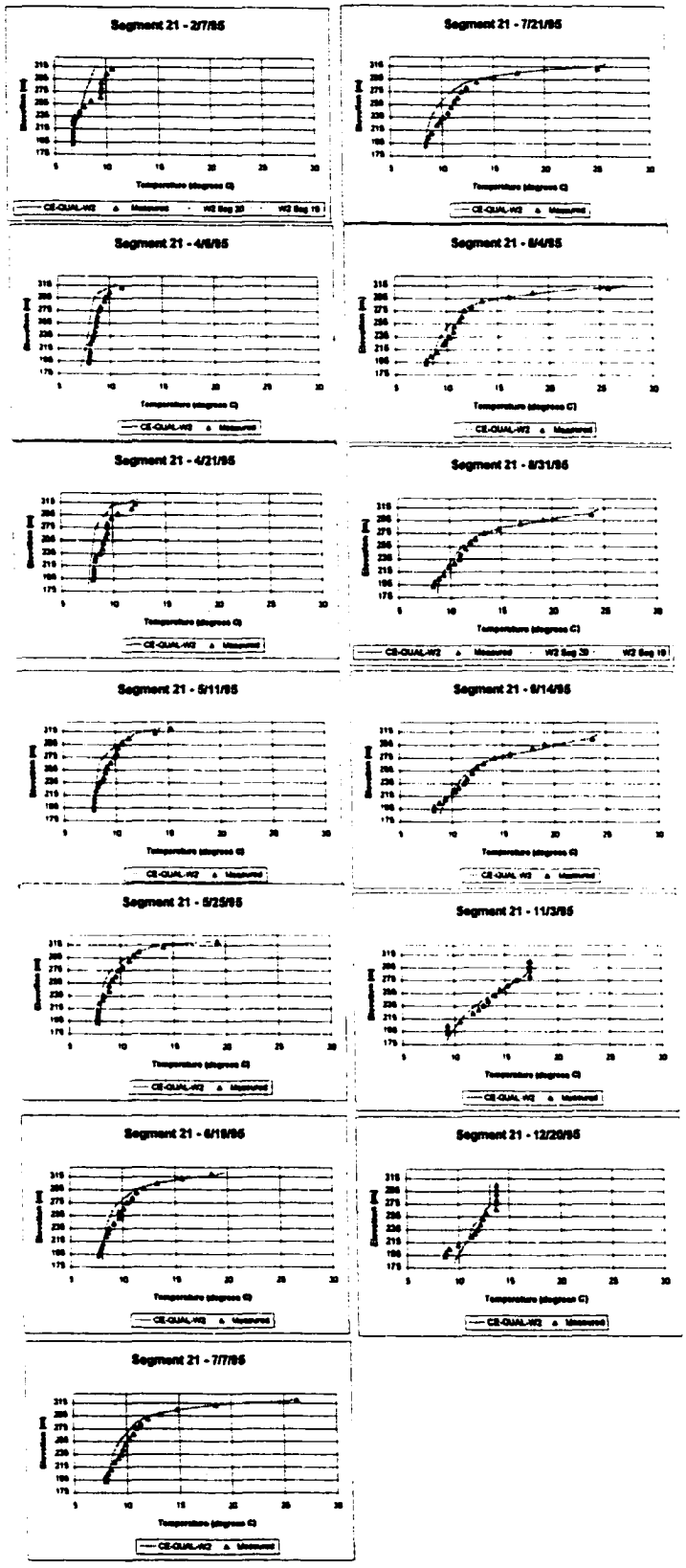

Figure 38. Results of initial validation run for 1995 data

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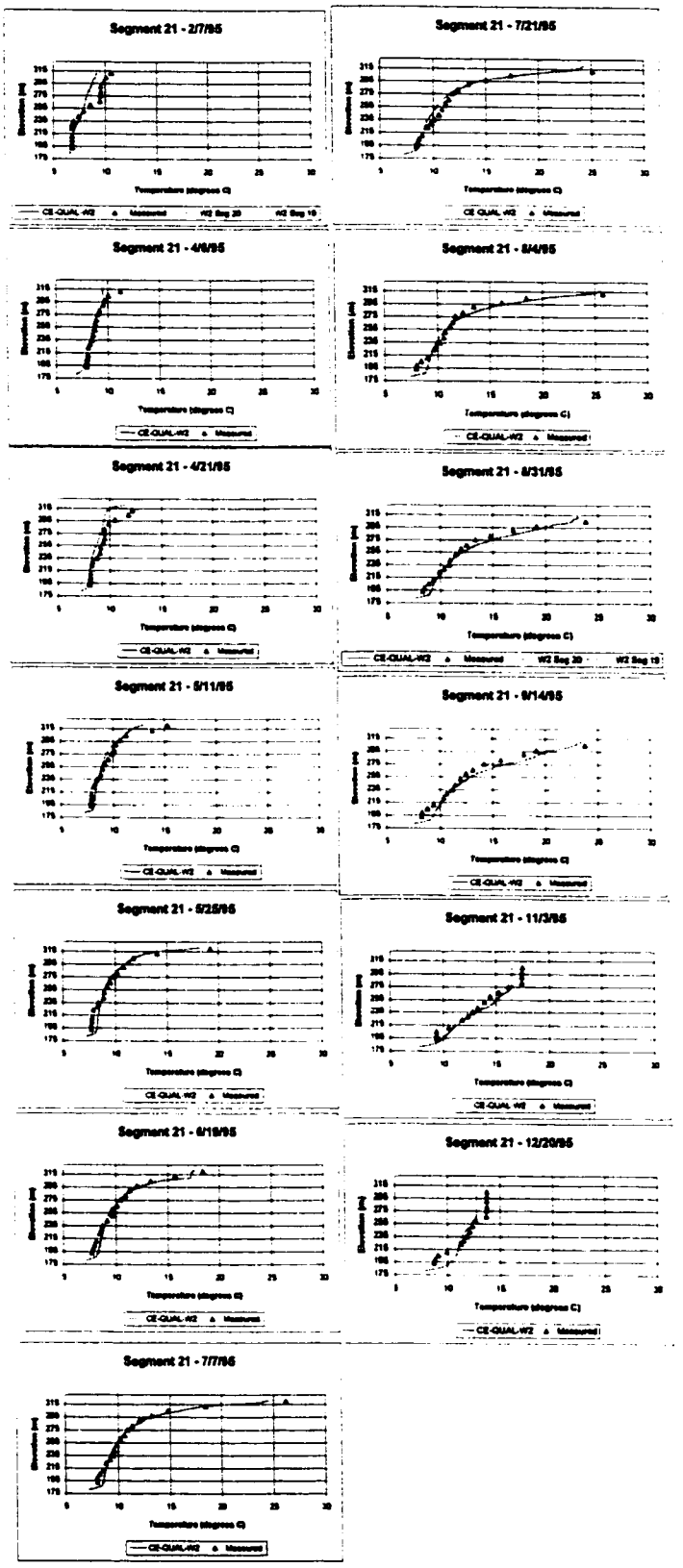

Figure 39. Results of 'best' validation run for 1995 data

(1995c: TSED = 10°C; CBHE = 8.0E-8 W m^{-2} s^{-1})

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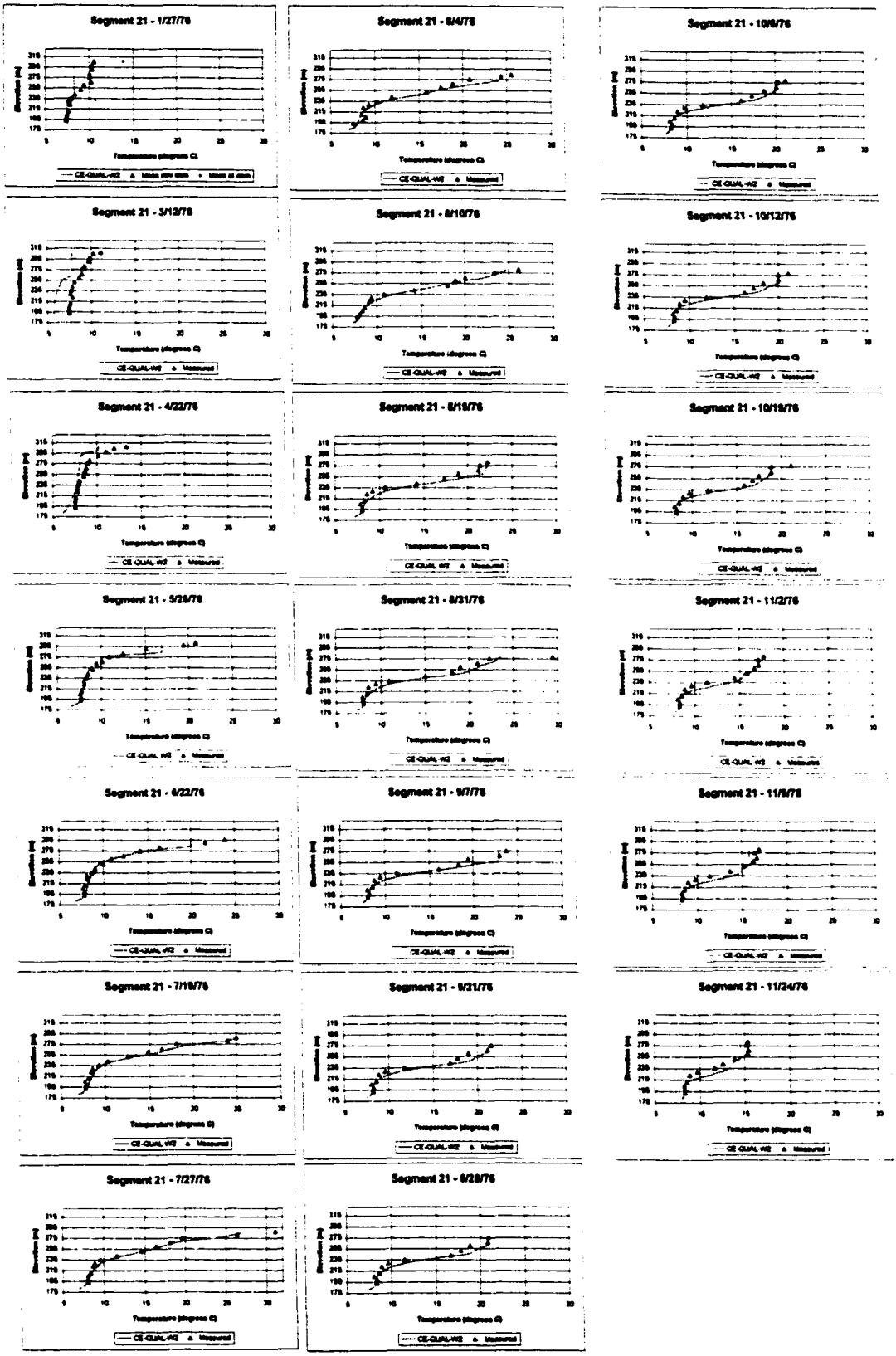

Figure 40. Results of initial validation run for 1976 data

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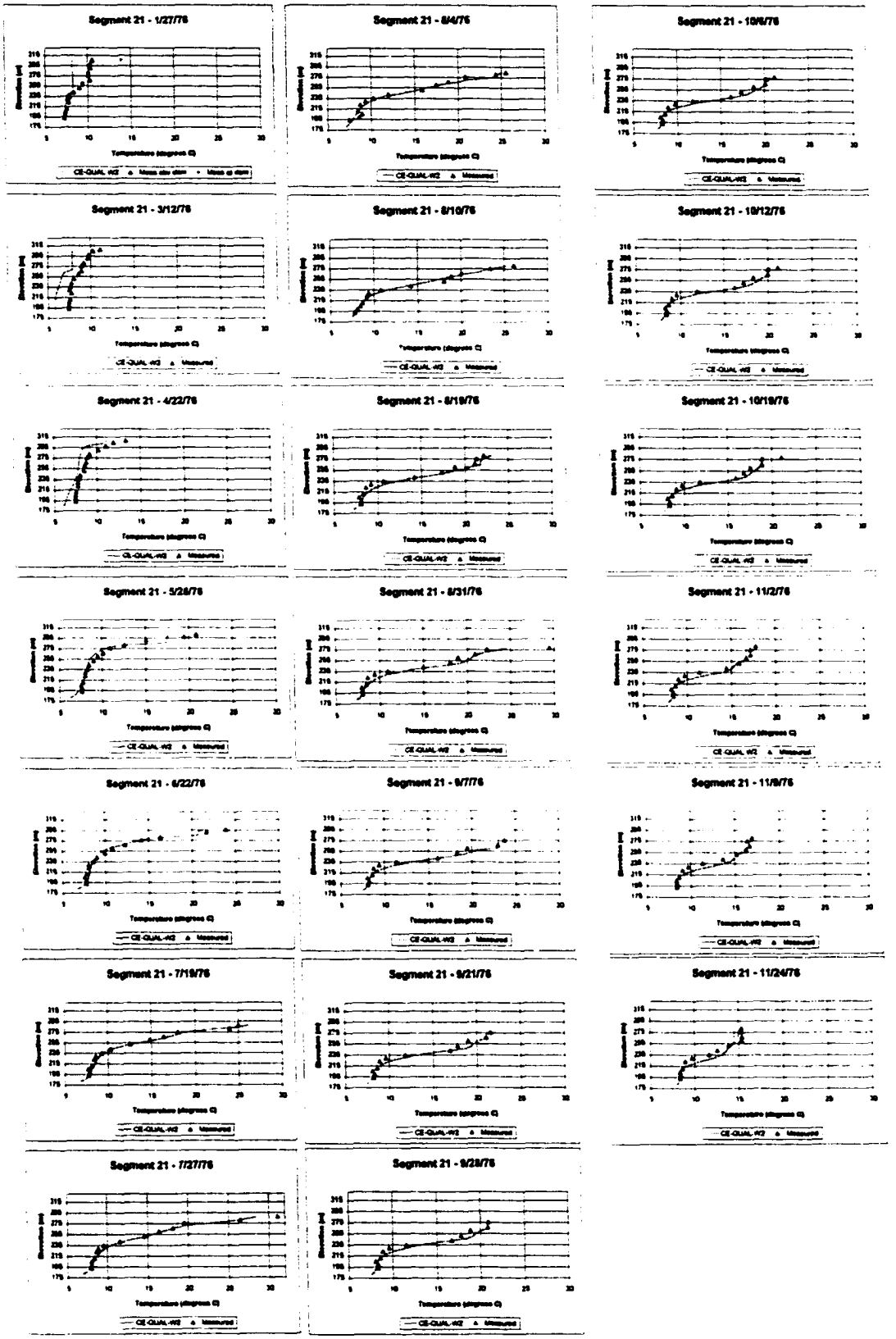

Figure 41. Results of 'best' validation run for 1976 data

(1976k: EXH2O = 1.5 m^{-1}, BETA = 0.72)

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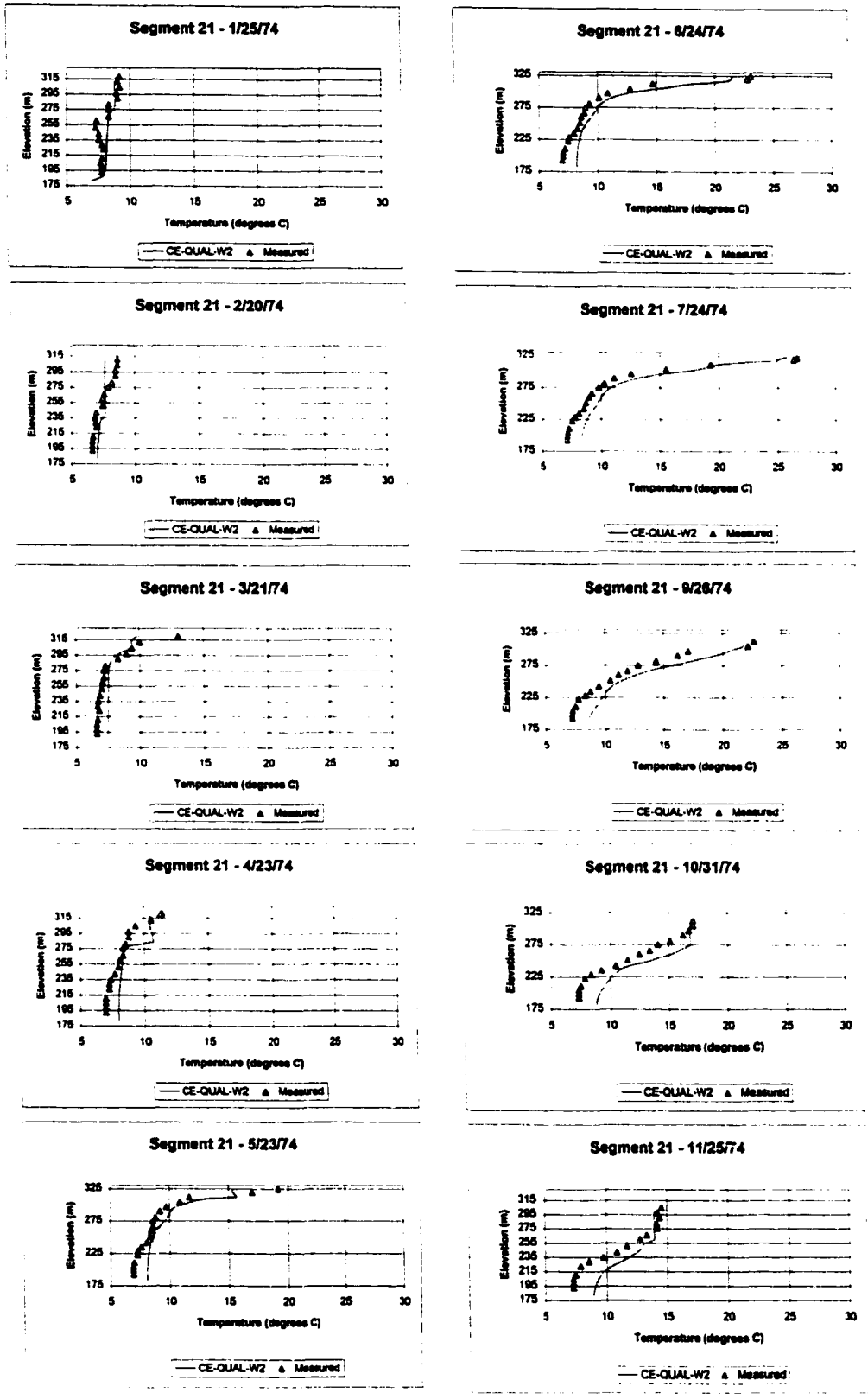

Figure 42. Results of initial validation run for 1974 data

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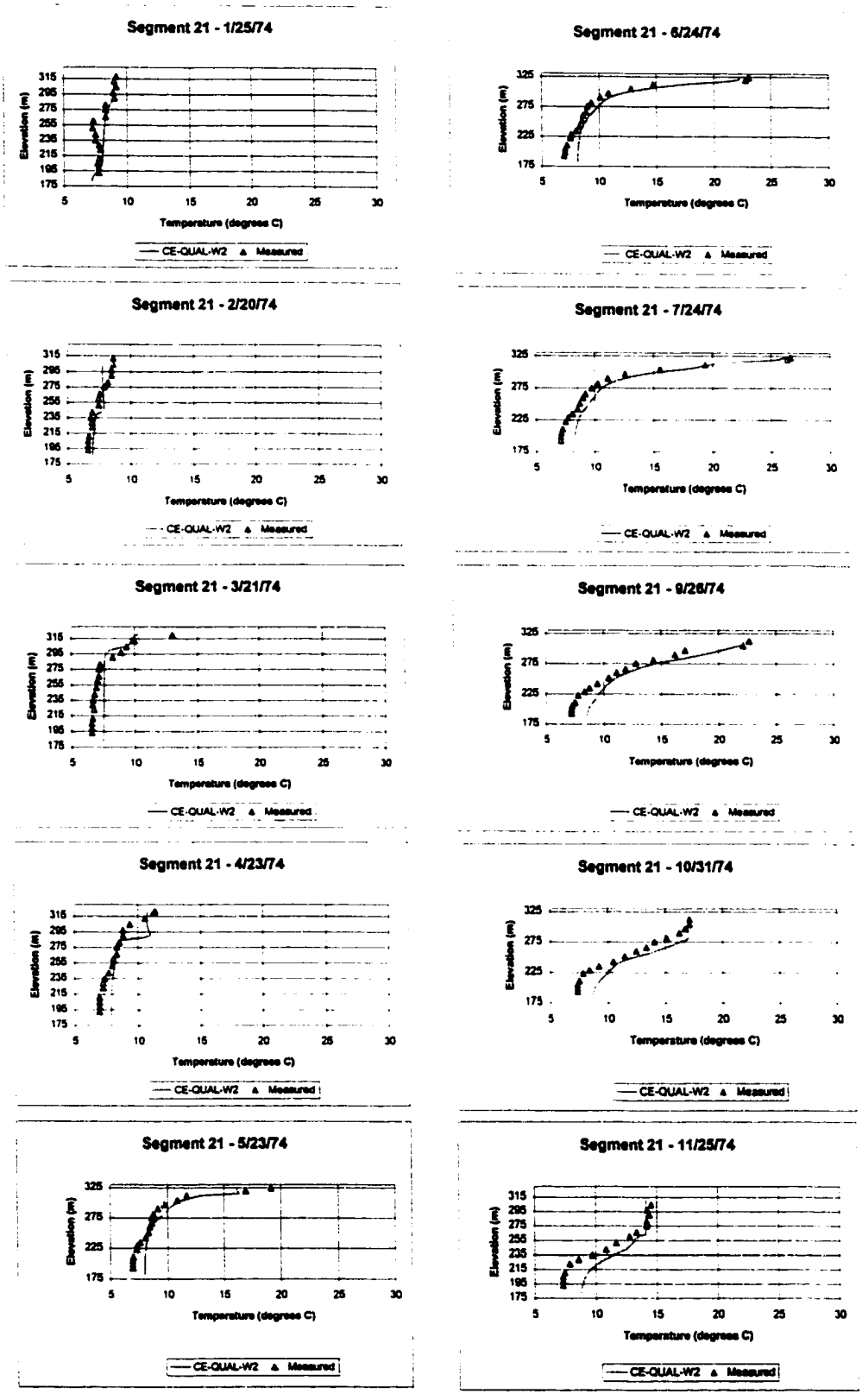

Figure 43. Results of 'best' validation run for 1974 data

(1974b: wind speed reduced by 0.5 m s^{-1})

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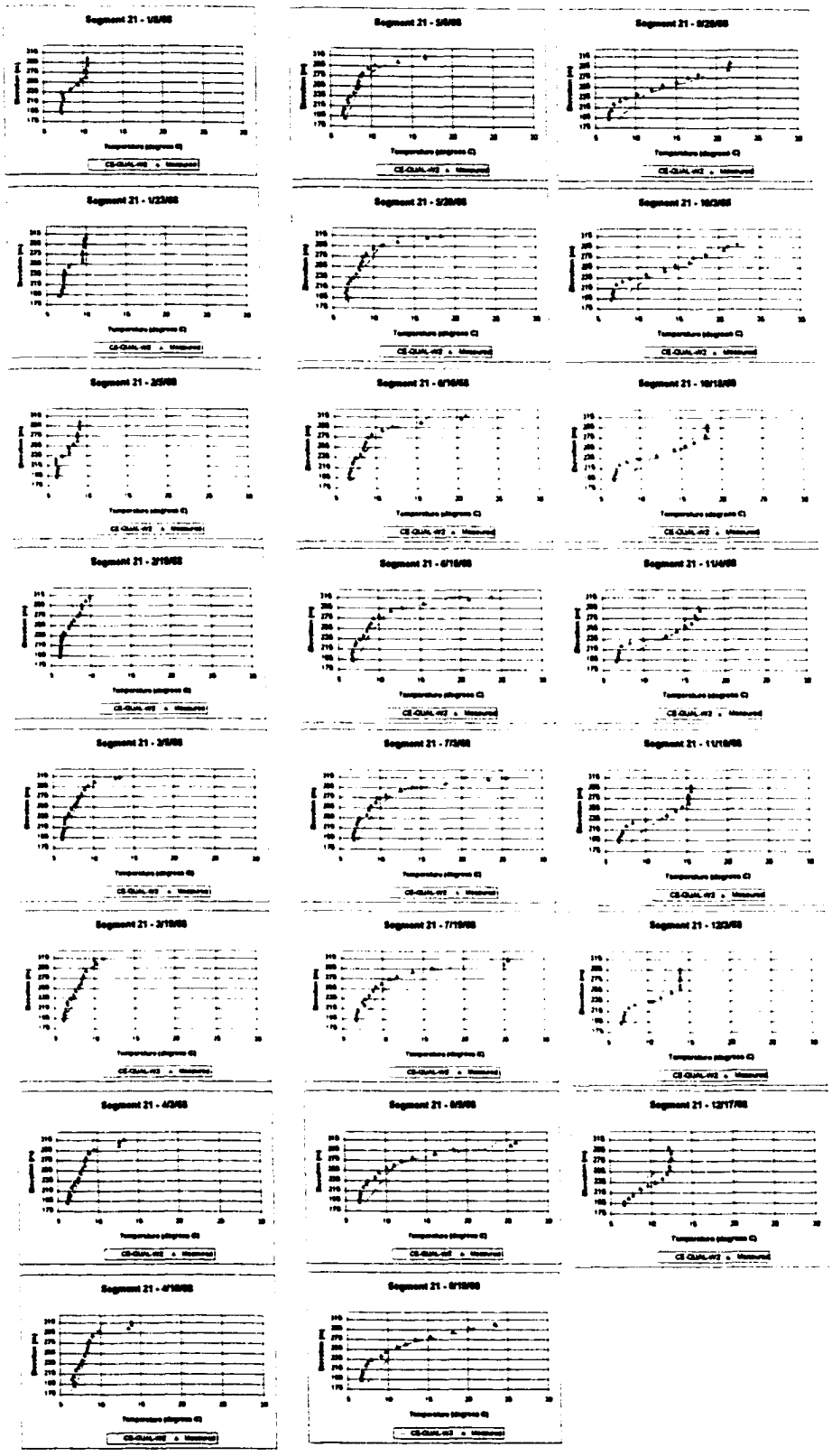

Figure 44. Results of initial validation run for 1968 data

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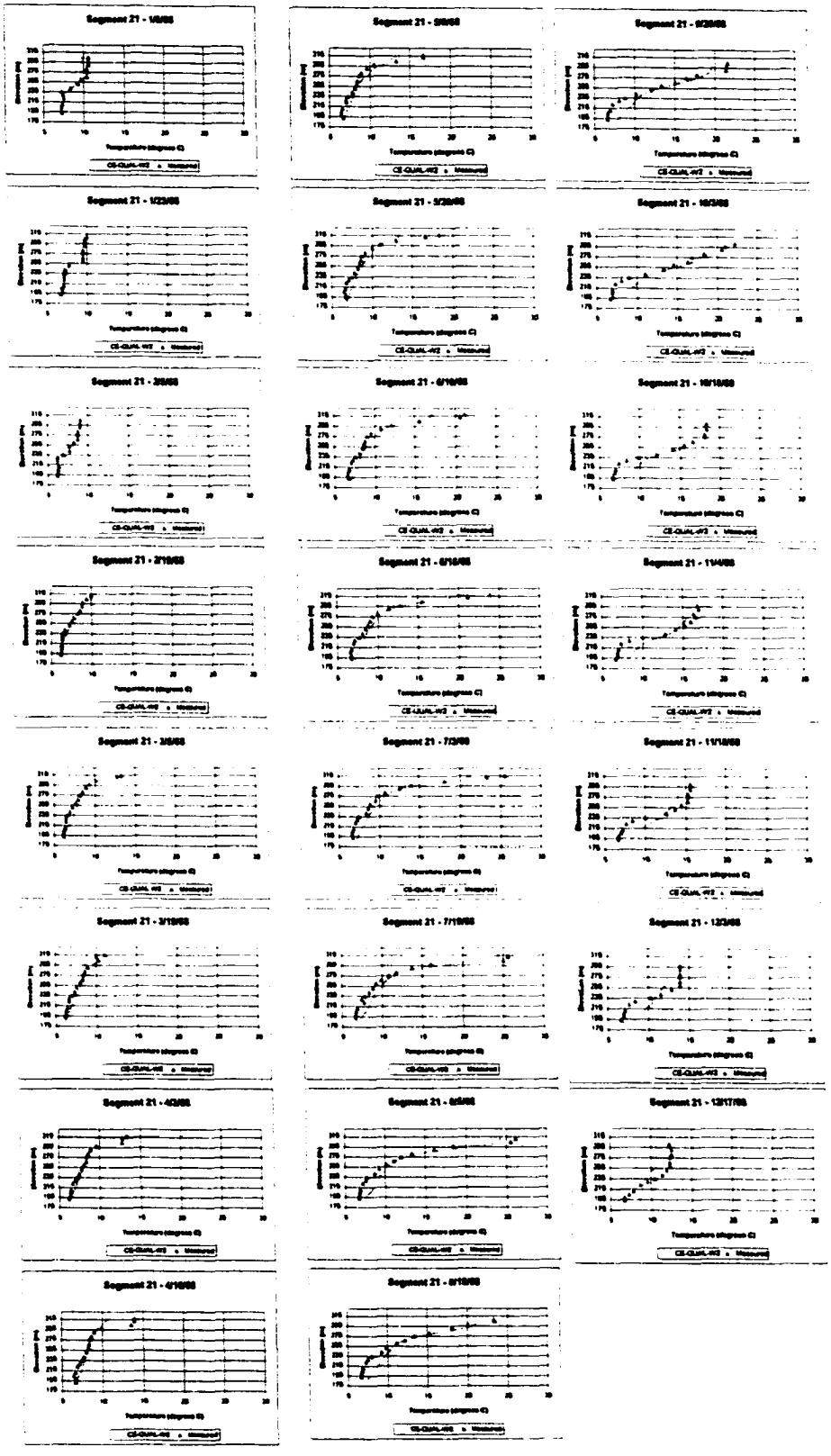

Figure 45. Results of 'best' validation run for 1968 data

(1968e: wind speed reduced 0.5 m s^{-1})

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APPENDIX C. Tables of summary statistics comparing 1995 run simulating water temperature only (*temponly*), and 1995 run simulating nutrients and phytoplankton as well (*phyto*).

Tables 52 through 57 summarize the r^2 values, root mean squared error (RMSE), and absolute mean error (ABSE) for the 1995 run simulating water temperature only (*temponly*), and 1995 run simulating nutrients and phytoplankton as well (*phyto*). Shading indicates maximum r^2 or minimum RMSE or ABSE for each segment or Julian day.

Table 52. r^2 values for *temponly* and *phyto* according to segment number

Segment	*temponly*	*phyto*
13	0.982	0.983
16	0.983	0.986
19	0.980	0.981
41	0.980	0.982
54	0.984	0.986
All	**0.980**	**0.982**

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Table 53. r^2 values for *temponly* and *phyto* according to Julian day

Julian day	*temponly*	*phyto*
130	0.973	0.969
170	0.978	0.974
206	0.987	0.991
241	0.987	0.989
264	0.977	0.977
291	0.974	0.977
317	0.925	0.934
All	**0.980**	**0.982**

Table 54. RMSE (°C) for *temponly* and *phyto* according to segment number

Segment	*temponly*	*phyto*
13	0.68	0.68
16	0.65	0.59
19	0.68	0.61
41	0.79	0.71
54	0.74	0.64
All	**0.71**	**0.65**

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Table 55. RMSE (°C) for *temponly* and *phyto* according to Julian day

Julian day	*temponly*	*phyto*
130	**0.75**	0.82
170	**0.67**	0.73
206	0.69	**0.59**
241	0.74	**0.59**
264	0.78	**0.69**
291	0.76	**0.58**
317	0.56	**0.50**
All	**0.71**	**0.65**

Table 56. ABSE (°C) for *temponly* and *phyto* according to segment number

Segment	*temponly*	*phyto*
13	**0.57**	0.58
16	0.53	**0.49**
19	0.54	**0.48**
41	0.61	**0.60**
54	0.55	**0.48**
All	**0.56**	**0.52**

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Table 57. ABSE (°C) for *temponly* and *phyto* according to Julian day

Julian day	*temponly*	*phyto*
130	0.63	0.70
170	0.52	0.60
206	0.54	0.48
241	0.59	0.48
264	0.59	0.54
291	0.64	0.49
317	0.42	0.41
All	**0.56**	**0.52**

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APPENDIX D. Sampling methods

D.1 Electrofishing

Electrofishing was done at all four reservoir locations . Shoreline surveys were conducted at night using a CDFG electrofishing boat fitted with a pulsed-DC system. Output from the generator was continually adjusted in response to changes in local water chemistry such that the minimum electrical shock sufficient to obtain samples was applied. The boat was operated by Neil Manji, a CDFG fishery biologist. Two people served as "dippers," collecting fish from the bow of the boat. Collected fish were placed in an aerated livewell on the boat. After an hour or two of electrofishing, the fish were processed. Those fish not needed for isotope sampling were released alive.

D.2 Gill netting

Horizontal experimental gill nets were only set at the Pit River, Sacramento River, and main body sampling locations. The nets were set at two or more locations each evening and retrieved the next morning using a boat provided by the CDFG. The nets were also provided by the CDFG. They consisted of variable mesh sizes and were 30.5 m (100 feet) long. Nets were set perpendicular to shore. Fish caught in the gill nets were picked from the nets and processed after all of the nets were retrieved.

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D.3 Trap netting

Trap nets were set at the Pit River, Sacramento River, and main body sampling locations. Nets were set at three locations each evening and retrieved the next morning using a CDFG boat. CDFG provided trap nets that were approximately 30.5 m (100 feet) long and 15.3 m (50 feet) deep. Trap nets were set perpendicular to shore. Fish caught in the nets were put into buckets and were processed after all nets were retrieved.

D.4 Hook-and-line fishing

Most hook-and-line samples were gathered from anglers near Shasta Dam or in the Dry Fork Inlet area on July 7 and 9, 1998. Anglers were approached and queried as to their success in catching trout or salmon. If the anglers were agreeable, two or three of their catch were transferred to the CDFG or USBR boat. Each fish was measured for its total length in mm, and one or two muscle plugs were removed from each fish before it was returned to the angler. One large Alabama spotted bass was caught by Kevin Gale of the CDFG using hook-and-line fishing on July 8, 1998 at the Sugarloaf Creek Inlet on the Sacramento River arm. The length and weight of the fish was recorded, and muscle plugs and scales were taken from the fish before it was returned to the reservoir.

D.5 Grab sampling

A USBR boat was used on July 9, 1998 to gather aquatic and terrestrial insect samples throughout the reservoir. While gathering these samples, some very small fish were collected using hand-held nets. These fish were placed whole in sample bags and placed on ice until the boat returned to shore, after which they were frozen.

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APPENDIX E. Preparation and isotopic analyses of eagle feathers at Shasta Lake

Ron Jackman, a consultant for the USFS, provided eagle feathers obtained from the following three locations in June and July 1998: Flume Creek on the Squaw Creek arm, Frost Gulch on the Main Lake area, and Jones Valley on the Pit River arm. Table 58 summarizes the descriptions of the feather samples.

Table 58. Summary of eagle feathers obtained

Description	number of individual items at each location		
	Main lake	Pit River arm	Squaw Creek arm
Alive adult flight feather	0	1	1
Alive adult tail feather	1	0	0
Alive nestling down feather	0	1	0
Dead nestling flight feather	1	0	1

The feather samples were prepared by washing them in water and rinsing with acetone. The feathers were then cut into narrow strips from 1 to 2 mm in width and placed in sample tins. They were then dried in a 60°C oven for 4-8 hours. Feather samples were ground with 20 mg of 40-grain silica sand. This method was very effective at creating a homogeneous mixture for the feather samples.

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To test the applicability of using silica sand to facilitate grinding of bald eagle feathers, isotope measurements were made of a dove feather gathered on the CSU campus with and without 40 grain silica sand. In addition, measurements were made of the sand alone, but no trace of ^{14}C or ^{15}N were detected, even when sample sizes were increased to ten times the size normally run through the mass spectrometer. The resulting measured signatures of the feather with and without sand are shown in Table 59.

Table 59. Results of isotope analyses of dove feather with and without 40 grain silica sand

Replicate number	$\delta^{13}C$ (‰)		$\delta^{15}N$ (‰)	
	Feather only	Feather and sand	Feather only	Feather and sand
1	-21.57	-20.98	5.38	5.64
2	-21.55	-21.13	5.42	5.54

Since the measurement error of the mass spectrometer is approximately 0.8 ‰ (Reuss 1998), these results combined with the inability to detect ^{14}C and ^{15}N in the sand only samples led us to conclude that grinding the bald eagle feather samples with 40 grain silica sand would not significantly alter the feather signatures.

The measured signatures of the eagle feathers are shown in Table 60. Although we did not include the eagle feathers in our food web analysis, their signatures indicate that they are eating a mixture of aquatic and terrestrial items (refer to Table 36).

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Table 60. Measured signatures of eagle feathers at Shasta Lake

Description	Main lake		Pit River		Squaw Creek	
	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Alive adult flight feather	--	--	-23.59	8.16	-23.66	12.05
Alive adult tail feather	-24.15	12.07	--	--	--	--
Alive nestling down feather	--	--	-23.53	10.89	--	--
Dead nestling flight feather	-25.18	12.03	--	--	-24.80	11.70

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APPENDIX F. Determination of composite samples for fish species

F.1 Black bass

The term "black bass" is used for all members of the genus *Micropterus* (Moyle 1976). Thus, the black bass species present at Shasta Lake include largemouth bass (*M. salmoides*), smallmouth bass (*M. dolomieui*), and Alabama spotted bass (*M. punctulatus henshalli*). Alabama spotted bass are undoubtedly the most abundant sport game fish at Shasta Lake, as evidenced by recent reservoir fish sampling (including the one documented here) and bass tournament results. The CDFG has done scale analyses on Alabama spotted bass in 1972, 1975, 1986 and 1996 and has broken down their growth as shown in the following table (Manji 1998):

Table 61. Alabama spotted bass size breakdowns

Year class	Size
1	0-113 mm TL
2	114-203 mm TL
3	204-290 mm TL
4	292-370 mm TL
5+	371+ mm TL

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Because samples smaller than 67 mm TL were generally dried as whole fish while larger samples consisted of only muscle tissue, the first year class was further broken into two size groupings at 67 mm TL. This resulted in the composite samples shown in Table 62 for Alabama spotted bass (note that if there is only one individual sample in a grouping, it is not considered a composite sample):

Table 62. Composite samples of Alabama spotted bass

Year class	Size (mm TL)	# of individual samples at each location			
		Main lake	McCloud River arm	Pit River arm	Sacramento River arm
1A	0-66	1	7	1	10
1B	67-113	1	9	6	0
2	114-203	10	11	19	12
3	204-290	11	6	3	13
4	291-370	4	5	6	3
5+	371+	0	1	2	2
Total individual samples		27	39	37	40
Total composite samples		3	5	5	5

In addition to the composite samples noted above, a composite sample was made of the two Alabama spotted bass caught during the preliminary sampling in June 1998 on the Sacramento River arm.

The growth patterns for largemouth bass and smallmouth bass were assumed to be similar to those for Alabama spotted bass. The three individual largemouth bass samples

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were in different size classes or at different locations, so none of them were combined into a composite sample. Smallmouth bass were only gathered at the main reservoir and McCloud River arm locations. Their composite sample groupings were as shown in Table 63.

Table 63. Composite samples for smallmouth bass

Year class	Size (mm TL)	# of individual samples at each location	
		Main lake	McCloud River arm
1A	0-66	3	2
1B	67-113	0	0
2	114-203	3	0
3	204-290	3	0
4	291-370	0	0
5+	371+	0	0
Total individual samples		9	2
Total composite samples		3	1

F.2 Black crappie (*Pomoxis nigromaculatus*) and white crappie (*P. annularis*)

Only three individual black crappie samples were obtained in the lake sampling, and all of these samples were on the Pit River arm. They ranged in size from 144 mm TL to 162 mm TL. Based on information provided in Moyle (1976), all of these fish fell into

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the second year class for black crappie. They were therefore combined into one composite sample.

Two individual white crappie samples were obtained, also only on the Pit River arm. According to Moyle (1976), the white crappie measured at 159 mm TL is probably in the second year class, while the other individual measured at 211 mm TL is probably in the fourth year class. However, a composite sample had already been made of the two individual samples prior to consulting Moyle (1976).

F.3 Bluegill (*Lepomis macrochirus*)

According to Moyle (1976), bluegill usually grow to about 40-50 mm TL in the first year, and then grow 20-50 mm per year. Visual inspection of the whole fish samples (i.e., samples <67 mm TL) indicated that bluegill that were 50 mm TL or larger tended to have ungrindable scales, whereas the smaller fish were much more easily ground. The size breakdown between year classes 1 and 2 was therefore set at 50 mm TL. The size breakdown between year classes 2 and 3 was set at 67 mm TL because that was the general dividing line between whole fish and dorsal muscle tissue sampling.

In addition to grouping composite samples according to year class, two composite samples were made by combining all individual samples in year classes 1 and 2 on the McCloud River and Pit River arms. This was done because the composites had been made before it was decided that the variation in the appearance of the individual samples above and below the 50 mm TL threshold might indicate another size breakdown. Composite groupings of bluegill did not include bluegill that were larger than 140 mm

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TL because such fish are not likely to be eaten by the game fish. The summary of composite samples for bluegill is shown in Table 64.

Table 64. Composite samples for bluegill

Year class	Size (mm TL)	# of individual samples at each location			
		Main lake	McCloud River arm	Pit River arm	Sacramento River arm
1	0-49	0	5	2	0
2	50-66	2	1	1	0
3+	67-140	5	9	23	1
Total individual samples		7	15	26	1
Total composite samples		2	4[a]	4[a]	0

[a]Additional composite sample made combining year classes 1 and 2

F.4 Channel catfish (*Ictalurus punctatus*) and white catfish (*I. catus*)

Moyle (1976) gives estimates of growth of channel catfish under good habitat conditions. Based on these estimates, the composite groupings of channel catfish were as shown in Table 65.

White catfish generally grow faster that brown bullhead, but slower than channel catfish (Moyle 1976). Using growth data for white catfish in Clear Lake given in Moyle (1976), it was estimated that the one individual sampled in the McCloud River arm (320 mm TL) was over six years old. Four white catfish were sampled in the Pit River arm

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and ranged in size from 256 mm TL to 312 mm TL. Although these catfish would likely be of different year classes, their diet was most likely very similar (Manji 1998), so all four were grouped into one composite sample.

Table 65. Composite samples for channel catfish

Year class	Size (mm TL)	# of individual samples at each location			
		Main lake	McCloud River arm	Pit River arm	Sacramento River arm
1	0-100	0	0	0	0
2	101-200	0	0	2	0
3	201-350	0	0	3	1
4	351-400	0	0	0	0
5	401-450	0	0	0	0
6+	451+	1	0	1	1
Total individual samples		1	0	6	2
Total composite samples		0	0	2	0

F.5 Golden shiner (*Notemigonus crysoleucas*)

Golden shiners can reach up to 76 mm TL in one year in warmer waters, but only 36 to 46 mm TL in colder waters (Moyle 1976). All of the golden shiners obtained at the reservoir were between 78 mm and 90 mm TL, so they were all considered to be year class 2 fish. The four golden shiners caught in the April 1998 entrainment sampling were composited into one sample. Because of the limited amount of individual sample

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available for one of the shiners, only 1 mg of each individual sample was used to make the composite sample instead of the customary 10 mg.

F.6 Green sunfish (*L. cyanellus*)

In California reservoirs, green sunfish usually reach up to 30 to 50 mm standard length (SL) in their first year, 50 to 100 mm SL in their second year, and 80 to 130 mm SL in their third year (Moyle 1976). This size breakdown of the year classes indicates that all of the green sunfish obtained at all of the reservoir locations were probably second or third year class fish. Both of these year classes were combined into composite samples at each location. Table 66 summarizes the composition of the green sunfish composite samples.

Table 66. Composite samples for green sunfish

Year class	Size (mm TL)	# of individual samples at each location			
		Main lake	McCloud River arm	Pit River arm	Sacramento River arm
1	0-49	0	0	0	0
2	50-100	1	2	3	3
3	101-130	3	0	1	1
Total individual samples		4	2	4	4
Total composite samples		1[a]	1	1[a]	1[a]

[a]Composite sample made by combining year classes 2 and 3

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F.7 Sacramento squawfish (*Ptychocheilus grandis*)

One composite sample was made of three Sacramento squawfish sampled in the main reservoir body that ranged in size from 90 mm to 103 mm. These fish were estimated to be in the second year class (Moyle 1976).

F.8 Sculpin spp. (*Cottus spp.*)

Species identification of sculpin samples was not done. In general, sculpins in the vicinity of the project area tend to reach 57-72 mm SL in the second year class, 59-89 mm SL in the third year class, and 75-104 mm SL in the fourth year class (Moyle 1976). All sculpins found during the Shasta sampling were between 22 mm TL and 88 mm TL. Composite samples of sculpins were therefore made as shown in Table 67.

Table 67. Composite samples for sculpins

Year class	Size (mm TL)	# of individual samples at Main Lake
1	0-50	3
2	51-72	2[a]
3	73-89	7[b]
Total individual samples		12
Total composite samples		2

[a]Individual samples were not composited
[b]2 mg of each individual was used to make composite sample

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F.9 Threadfin shad (*Dorosoma petenense*)

Moyle (1976) stated that threadfin shad usually reach 40-60 mm TL during their first year under normal conditions, and 60-100 mm TL by the end of their second year. He observed that threadfin shad rarely live longer than two years or grow to lengths greater than 100 mm TL. However, more than half of the threadfin shad obtained in the July 1998 reservoir sampling were greater than 100 mm TL, and the smallest shad sampled was 85 mm TL. Because it was expected that larval and first year class shad are a primary food source for piscivorous fish, samples of smaller shad were obtained from entrainment sampling done in April 1998. Thus the five smallest individual shad on the main lake in Table 68 are from the entrainment sampling.

Table 68. Composite samples for threadfin shad

Year class	Size (mm TL)	# of individual samples at each location	
		Main lake	Pit River arm
1	0-60	4	0
2	61-100	1	4
3+	101+	1	9
Total individual samples		6	13
Total composite samples		1	2

Note that individual samples of the first year class of threadfin shad were also analyzed.

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APPENDIX G. Comparison of composite signatures versus mean individual signatures

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species	Yr-class	location	comp/ind	$\delta^{13}C$	$\delta^{15}N$
ASB	2	MAIN	COMP	-28.65	9.10
ASB	3	MAIN	COMP	-28.22	10.14
ASB	4	MAIN	COMP	-28.26	10.07
ASB	2	MAIN	IND	-28.27	9.11
ASB	3	MAIN	IND	-27.93	10.06
ASB	4	MAIN	IND	-27.91	9.94
BLG*	1/2	PIT	COMP	-28.49	8.30
BLG	3	MAIN	COMP	-29.67	7.19
BLG*	1/2	PIT	IND	-28.35	8.23
BLG	3	MAIN	IND	-29.27	6.99
CHS	2/3	MAIN	COMP	-28.22	11.60
CHS	2/3	MAIN	IND	-28.10	11.33
GSH	2	MAIN	COMP	-19.70	7.25
GSH	2	MAIN	IND	-20.16	8.22
RBT	2/3	MAIN	COMP	-24.33	9.46
RBT	2/3	MAIN	IND	-24.28	9.44
SCU	3	MAIN	COMP	-24.07	7.56
SCU	3	MAIN	IND	-24.76	7.79
TFS	1	MAIN	COMP	-31.21	8.92
TFS	2	PIT	COMP	-29.89	9.79
TFS	1	MAIN	IND	-31.30	8.76
TFS	2	PIT	IND	-29.63	8.81
WCR*	2	PIT	COMP	-26.27	10.95
WCR*	2	PIT	IND	-26.33	11.12

*indicates only two individuals in sample

Paired t-test

carbon: H_0: $\mu_{comp} = \mu_{ind}$
p-value = 0.5703
do not reject

nitrogen: H_0: $\mu_{comp} = \mu_{ind}$
p-value = 0.7344
do not reject

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OBS	CCOM	CIND	CDIFF
1	-28.65	-28.27	-0.38
2	-28.22	-27.93	-0.29
3	-28.26	-27.91	-0.35
4	-28.49	-28.35	-0.14
5	-29.67	-29.27	-0.40
6	-28.22	-28.10	-0.12
7	-19.70	-20.16	0.46
8	-24.33	-24.28	-0.05
9	-24.07	-24.76	0.69
10	-31.21	-31.30	0.09
11	-29.89	-29.63	-0.26
12	-26.27	-26.33	0.06

Univariate Procedure

Variable=CDIFF

Moments

N	12	Sum Wgts	12
Mean	-0.0575	Sum	-0.69
Std Dev	0.340431	Variance	0.115893
Skewness	1.21847	Kurtosis	0.907684
USS	1.3145	CSS	1.274825
CV	-592.054	Std Mean	0.098274
T:Mean=0	-0.5851	Pr>|T|	0.5703
Num ^= 0	12	Num > 0	4
M(Sign)	-2	Pr>=|M|	0.3877
Sgn Rank	-11	Pr>=|S|	0.4238

Quantiles(Def=5)

100% Max	0.69	99%	0.69
75% Q3	0.075	95%	0.69
50% Med	-0.13	90%	0.46
25% Q1	-0.32	10%	-0.38
0% Min	-0.4	5%	-0.4
		1%	-0.4
Range	1.09		
Q3-Q1	0.395		
Mode	-0.4		

Extremes

Lowest	Obs	Highest	Obs
-0.4(	5)	-0.05(	8)
-0.38(	1)	0.06(	12)
-0.35(	3)	0.09(	10)
-0.29(	2)	0.46(	7)
-0.26(	11)	0.69(	9)

Variable	N	Mean	Std Dev
CCOM	12	-27.2483333	3.1838908
CIND	12	-27.1908333	2.9559138

Analysis Variable : CDIFF

Mean	Std Dev	T	Prob>|T|	Lower 95.0% CLM	Upper 95.0% CLM
-0.057	0.340	-0.585	0.5703	-0.274	0.159

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OBS	NCOM	NIND	NDIFF
1	9.10	9.11	-0.01
2	10.14	10.06	0.08
3	10.07	9.94	0.13
4	8.30	8.23	0.07
5	7.19	6.99	0.20
6	11.60	11.33	0.27
7	7.25	8.22	-0.97
8	9.46	9.44	0.02
9	7.56	7.79	-0.23
10	8.92	8.76	0.16
11	9.79	8.81	0.98
12	10.95	11.12	-0.17

Univariate Procedure

Variable=NDIFF

Moments

N	12	Sum Wgts	12	100% Max	0.98	99%	0.98
Mean	0.044167	Sum	0.53	75% Q3	0.18	95%	0.98
Std Dev	0.43972	Variance	0.193354	50% Med	0.075	90%	0.27
Skewness	-0.30354	Kurtosis	3.70546	25% Q1	-0.09	10%	-0.23
USS	2.1503	CSS	2.126892	0% Min	-0.97	5%	-0.97
CV	995.5927	Std Mean	0.126936			1%	-0.97
T:Mean=0	0.347944	Pr>|T|	0.7344	Range	1.95		
Num ^= 0	12	Num > 0	8	Q3-Q1	0.27		
M(Sign)	2	Pr>=|M|	0.3877	Mode	-0.97		
Sgn Rank	11	Pr>=|S|	0.4238				

Extremes

Lowest	Obs	Highest	Obs
-0.97(	7)	0.13(	3)
-0.23(	9)	0.16(	10)
-0.17(	12)	0.2(	5)
-0.01(	1)	0.27(	6)
0.02(	8)	0.98(	11)

Variable	N	Mean	Std Dev
NCOM	12	9.1941667	1.4241071
NIND	12	9.1500000	1.3026337

Analysis Variable : NDIFF

Mean	Std Dev	T	Prob>|T|	Lower 95.0% CLM	Upper 95.0% CLM
0.044	0.440	0.348	0.7344	-0.235	0.324

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```
/* Laurel Saito
   Shasta Lake isotope analyses
   Composite versus individual analysis

   Data for 12 composite samples are compared with the
   averages of the individual measurements. A paired t-test
   is done.                                                  */

* Set line and page lengths;

options linesize=85 pagesize=50;

* Measured data;

data isotope;
   input sampleid $ ccom cind ncom nind;
   cdiff = ccom - cind;
   ndiff = ncom - nind;
cards;
asb2mai -28.65 -28.27  9.10  9.11
asb3mai -28.22 -27.93 10.14 10.06
asb4mai -28.26 -27.91 10.07  9.94
blg1pit -28.49 -28.35  8.30  8.23
blg3mai -29.67 -29.27  7.19  6.99
chs2mai -28.22 -28.10 11.60 11.33
gsh2mai -19.70 -20.16  7.25  8.22
rbt2mai -24.33 -24.28  9.46  9.44
scu3mai -24.07 -24.76  7.56  7.79
tfs1mai -31.21 -31.30  8.92  8.76
tfs2pit -29.89 -29.63  9.79  8.81
wcr2pit -26.27 -26.33 10.95 11.12
;

proc print;
   var ccom cind cdiff;
   title 'Shasta Lake carbon isotope composite versus individual analysis';

proc univariate;
   var cdiff;

proc means n mean std;
   var ccom cind;

proc means mean std t prt clm alpha=0.05 maxdec=3;
   var cdiff;

proc print;
   var ncom nind ndiff;
   title 'Shasta Lake nitrogen isotope composite versus individual analysis';

proc univariate;
   var ndiff;

proc means n mean std;
   var ncom nind;

proc means mean std t prt clm alpha=0.05 maxdec=3;
   var ndiff;

run;
```


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APPENDIX H. SAS output for two-way ANOVA comparison of replicate measurements of Shasta data

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Sample Number	SPP code	1 $\delta^{13}C$ (‰)	2 $\delta^{13}C$ (‰)	diff (‰)	abs diff (‰)	1 $\delta^{15}N$ (‰)	2 $\delta^{15}N$ (‰)	diff (‰)	abs diff (‰)
4	ASB					10.14	9.47	0.67	0.67
6	ASB	-27.88	-27.9	0.02	0.02				
25	BCR					9.96	10.51	-0.55	0.55
32	BLG	-29	-28.88	-0.12	0.12	7.09	7.75	-0.66	0.66
34	BLG	-27.45	-27.19	-0.26	0.26				
35	BLG	-28.49	-28.77	0.28	0.28				
36	BLG	-28.21	-28.01	-0.20	0.20				
37	BLG	-27.55	-27.36	-0.19	0.19	8.94	7.43	1.51	1.51
38	BLG	-29.55	-29.5	-0.05	0.05				
45	CCF	-27.69	-27.49	-0.20	0.20	9.06	9.39	-0.33	0.33
83	SQF					7.49	7.71	-0.22	0.22
86	TFS	-31.11	-31.06	-0.05	0.05	9.28	9.2	0.08	0.08
98	WCR					11.36	11.53	-0.17	0.17
103	ZOO					8.28	8.4	-0.12	0.12
				avg	0.15			avg	0.48

2-way ANOVA: H_0: analysts same

carbon: p = 0.1569
MSE = 0.013501
std. err. = 0.116195

nitrogen: p = 0.9202
MSE = 0.229325
std. err. = 0.478879

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Analysis Variable : CARBON

ANALYST	N Obs	N	Mean	Std Dev	Minimum	Maximum
ja	9	9	-28.5477778	1.1871055	-31.1100000	-27.4500000
ls	9	9	-28.4622222	1.2446865	-31.0600000	-27.1900000

General Linear Models Procedure
Class Level Information

Class	Levels	Values
SAMPLENO	9	cs09 cs32 cs34 cs35 cs36 cs37 cs38 cs45 cs86
ANALYST	2	ja ls

Number of observations in data set = 18

General Linear Models Procedure

Dependent Variable: CARBON

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	9	23.59263889	2.62140432	194.16	0.0001
Error	8	0.10801111	0.01350139		
Corrected Total	17	23.70065000			

R-Square	C.V.	Root MSE	CARBON Mean
0.995443	-0.407632	0.1161955	-28.505000

Source	DF	Type I SS	Mean Square	F Value	Pr > F
SAMPLENO	8	23.55970000	2.94496250	218.12	0.0001
ANALYST	1	0.03293889	0.03293889	2.44	0.1569

Source	DF	Type III SS	Mean Square	F Value	Pr > F
SAMPLENO	8	23.55970000	2.94496250	218.12	0.0001
ANALYST	1	0.03293889	0.03293889	2.44	0.1569

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```
/* Laurel Saito
   Shasta Lake isotope analyses
   Carbon isotope replicate analysis

   Data for 9 carbon isotope samples are compared between two
   analysts (John Arterburn and Laurel Saito).  A two-way ANOVA
   is done.                                                    */

* Set line and page lengths;

options linesize=80 pagesize=50;

* Measured data;

data isotope;
   input sampleno $ analyst $ carbon;
cards;
cs09  ja -27.88
cs09  ls -27.90
cs32  ja -29.00
cs32  ls -28.88
cs34  ja -27.45
cs34  ls -27.19
cs35  ja -28.49
cs35  ls -28.77
cs36  ja -28.21
cs36  ls -28.01
cs37  ja -27.55
cs37  ls -27.36
cs38  ja -29.55
cs38  ls -29.50
cs45  ja -27.69
cs45  ls -27.49
cs86  ja -31.11
cs86  ls -31.06
;

proc means;
   var carbon;
   class analyst;
   title 'Shasta Lake carbon isotope replicate analysis';

proc glm;
   class sampleno analyst;
   model carbon=sampleno analyst;
run;
```


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Analysis Variable : NITROGEN

ANALYST	N Obs	N	Mean	Std Dev	Minimum	Maximum
ja	9	9	9.0666667	1.3361793	7.0900000	11.3600000
ls	9	9	9.0433333	1.3751636	7.4300000	11.5300000

General Linear Models Procedure
Class Level Information

Class	Levels	Values
SAMPLENO	9	cs04 cs103 cs25 cs32 cs37 cs45 cs83 cs86 cs98
ANALYST	2	ja ls

Number of observations in data set = 18

General Linear Models Procedure

Dependent Variable: NITROGEN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	9	27.57945000	3.06438333	13.36	0.0006
Error	8	1.83460000	0.22932500		
Corrected Total	17	29.41405000			

R-Square	C.V.	Root MSE	NITROGEN Mean
0.937628	5.288558	0.4788789	9.0550000

Source	DF	Type I SS	Mean Square	F Value	Pr > F
SAMPLENO	8	27.57700000	3.44712500	15.03	0.0005
ANALYST	1	0.00245000	0.00245000	0.01	0.9202

Source	DF	Type III SS	Mean Square	F Value	Pr > F
SAMPLENO	8	27.57700000	3.44712500	15.03	0.0005
ANALYST	1	0.00245000	0.00245000	0.01	0.9202

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```
/* Laurel Saito
   Shasta Lake isotope analyses
   Nitrogen isotope replicate analysis

   Data for 9 nitrogen isotope samples are compared between two
   analysts (John Arterburn and Laurel Saito).  A two-way ANOVA
   is done.                                                    */

* Set line and page lengths;

options linesize=80 pagesize=50;

* Measured data;

data isotope;
   input sampleno $ analyst $ nitrogen;
cards;
cs04 ja 10.14
cs04 ls 9.47
cs25 ja 9.96
cs25 ls 10.51
cs32 ja 7.09
cs32 ls 7.75
cs37 ja 8.94
cs37 ls 7.43
cs45 ja 9.06
cs45 ls 9.39
cs83 ja 7.49
cs83 ls 7.71
cs86 ja 9.28
cs86 ls 9.20
cs98 ja 11.36
cs98 ls 11.53
cs103 ja 8.28
cs103 ls 8.40
;

proc means;
   var nitrogen;
   class analyst;
   title 'Shasta Lake nitrogen isotope replicate analysis';

proc glm;
   class sampleno analyst;
   model nitrogen=sampleno analyst;
run;
```


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APPENDIX I. Carbon and nitrogen isotope signatures used in the statistical analysis of location differences

Table 69 and Table 70 respectively list the carbon and nitrogen isotope signatures used in the statistical comparison of location differences.

Table 69. Carbon isotope signatures used in statistical analysis of differences between reservoir locations

Species code[a,c]	Year-class[b]	$\delta^{13}C$ (‰)			
		Main Lake	Sacramento River arm	McCloud River arm	Pit River arm
ASB	1A	-32.010	-30.150	-31.540	-27.400
ASB	1B	-27.230	ne	-29.600	-27.660
ASB	2	-28.650	-29.190	-28.770	-25.960
ASB	3	-28.220	-28.390	-27.880	-24.990
ASB	4	-28.260	-28.640	-26.390	-24.820
ASB	5+	ne	-26.075	-25.860	-27.970
BLG	1	ne	ne	-32.110	-29.240
BLG	2	-30.980	ne	-29.520	-27.450
BLG	3+	-29.670	-29.550	-29.000	-27.880
GSF	2	-26.170	-25.490	-25.870	-26.450
HEM	--	-25.590	-25.995	ne	-25.460
LMB	2	ne	ne	-28.180	-27.370
SMB	1A	-30.670	ne	-29.450	ne
SQF	2	-27.330	ne	-25.670	ne
TFS	3+	-29.940	ne	ne	-30.140
ZOO	--	-35.250	-32.980	-32.520	-33.005
PASB	3	-28.490	-29.440	ne	-24.880
PSCU	1	-31.190	ne	ne	-28.210
PZOO	--	-35.430	-35.350	ne	-34.680

[a]See Table 33 for species codes
[b]See Appendix F for descriptions of year-classes
[c]A "P" at the beginning of the species code indicates samples gathered during the preliminary sampling in June 1998
ne = not evaluated

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Table 70. Nitrogen isotope signatures used in statistical analysis of differences between reservoir locations

Species code[a,c]	Year-class[b]	$\delta^{15}N$ (‰)			
		Main Lake	Sacramento River arm	McCloud River arm	Pit River arm
ASB	1A	7.440	5.910	7.110	8.740
ASB	1B	8.030	ne	8.690	9.810
ASB	2	9.100	9.260	9.030	10.530
ASB	3	10.140	9.890	10.190	10.350
ASB	4	10.070	9.930	9.980	10.540
ASB	5+		11.330	11.020	11.140
BLG	1	ne	ne	6.900	7.490
BLG	2	5.810	ne	6.350	8.970
BLG	3+	7.190	7.050	7.090	8.515
GSF	2	7.510	7.430	6.740	9.290
HEM	--	2.720	2.900	ne	3.400
LMB	2	ne	ne	10.520	10.880
SMB	1A	7.440	ne	7.340	ne
SQF	2	6.450	ne	7.490	ne
TFS	3+	9.670	ne	ne	9.500
ZOO	--	6.970	6.190	6.030	8.635
PASB	3	9.510	9.790	ne	10.230
PSCU	1	10.400	ne	ne	7.240
PZOO	--	5.690	6.630	ne	7.450

[a]See Table 33 for species codes
[b]See Appendix F for descriptions of year-classes
[c]A "P" at the beginning of the species code indicates samples gathered during the preliminary sampling in June 1998
ne = not evaluated

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location	species	Yr-class	$\delta^{13}C$	$\delta^{15}N$
MAIN	ASB	1A	-32.010	7.440
MAIN	ASB	1B	-27.230	8.030
MAIN	ASB	2	-28.650	9.100
MAIN	ASB	3	-28.220	10.140
MAIN	ASB	4	-28.260	10.070
MAIN	BLG	2	-30.980	5.810
MAIN	BLG	3+	-29.670	7.190
MAIN	GSF	2	-26.170	7.510
MAIN	HEM	0	-25.590	2.720
MAIN	SMB	1A	-30.670	7.440
MAIN	SQF	2	-27.330	6.450
MAIN	TFS	3+	-29.940	9.670
MAIN	ZOO	0	-35.250	6.970
MCC	ASB	1A	-31.540	7.110
MCC	ASB	1B	-29.600	8.690
MCC	ASB	2	-28.770	9.030
MCC	ASB	3	-27.880	10.190
MCC	ASB	4	-26.390	9.980
MCC	ASB	5+	-25.860	11.020
MCC	BLG	1	-32.110	6.900
MCC	BLG	2	-29.520	6.350
MCC	BLG	3+	-29.000	7.090
MCC	GSF	2	-25.870	6.740
MCC	LMB	2	-28.180	10.520
MCC	SMB	1A	-29.450	7.340
MCC	SQF	2	-25.670	7.490
MCC	ZOO	0	-32.520	6.030
PIT	ASB	1A	-27.400	8.740
PIT	ASB	1B	-27.660	9.810
PIT	ASB	2	-25.960	10.530
PIT	ASB	3	-24.990	10.350
PIT	ASB	4	-24.820	10.540
PIT	ASB	5+	-27.970	11.140
PIT	BLG	1	-29.240	7.490
PIT	BLG	2	-27.450	8.970
PIT	BLG	3+	-27.880	8.515
PIT	GSF	2	-26.450	9.290
PIT	HEM	0	-25.460	3.400
PIT	LMB	2	-27.370	10.880
PIT	TFS	3+	-30.140	9.500
PIT	ZOO	0	-33.005	8.635
SAC	ASB	1A	-30.150	5.910
SAC	ASB	2	-29.190	9.260
SAC	ASB	3	-28.390	9.890
SAC	ASB	4	-28.640	9.930
SAC	ASB	5+	-26.075	11.330
SAC	BLG	3+	-29.550	7.050
SAC	GSF	2	-25.490	7.430
SAC	HEM	0	-25.995	2.900
SAC	ZOO	0	-32.980	6.190
MAIN	PASB	3	-28.490	9.510
MAIN	PSCU	1	-31.190	10.400
MAIN	PZOO	0	-35.430	5.690
PIT	PASB	3	-24.880	10.230
PIT	PSCU	1	-28.210	7.240
PIT	PZOO	0	-34.680	7.450
SAC	PASB	3	-29.440	9.790
SAC	PZOO	0	-35.350	6.630

results including preliminary data

alpha = 0.05 adjusted alpha = 0.05/6 = 0.008333

carbon

Location	lsmean	mai	mcc	pit	sac
mai	-29.6687302		0.1514	0.0001	0.5178
mcc	-29.0267224	0.1514		0.0046	0.4951
pit	-27.7311498	0.0001	0.0046		0.001
sac	-29.36446	0.5178	0.4951	0.001	

nitrogen

Location	lsmean	mai	mcc	pit	sac
mai	8.01115899		0.7108	0.0025	0.6798
mcc	7.90263962	0.7108		0.0013	0.9513
pit	8.89154126	0.0025	0.0013		0.002
sac	7.88264488	0.6798	0.9513	0.002	

results for just July 1998 data

alpha = 0.05 adjusted alpha = 0.05/6 = 0.008333

carbon

Location	lsmean	mai	mcc	pit	sac
mai	-29.2783879		0.1815	0.0004	0.4149
mcc	-28.6560284	0.1815		0.0111	0.71
pit	-27.4586131	0.0004	0.0111		0.0098
sac	-28.8493062	0.4149	0.71	0.0098	

nitrogen

Location	lsmean	mai	mcc	pit	sac
mai	7.88134986		0.9877	0.0001	0.688
mcc	7.88500984	0.9877		0.0001	0.6756
pit	9.02740288	0.0001	0.0001		0.0001
sac	7.77270476	0.688	0.6756	0.0001	

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Analysis Variable : CARBON

LOCATION	N Obs	N	Mean	Std Dev	Minimum	Maximum
mai	13	13	-29.2284615	2.6192106	-35.2500000	-25.5900000
mcc	14	14	-28.7400000	2.2860413	-32.5200000	-25.6700000
pit	14	14	-27.5567857	2.1802537	-33.0050000	-24.8200000
sac	9	9	-28.4955556	2.3879587	-32.9800000	-25.4900000

General Linear Models Procedure
Class Level Information

Class	Levels	Values
LOCATION	4	mai mcc pit sac
SPECIES	16	asb1a asb1b asb2 asb3 asb4 asb5 blg1 blg2 blg3 gsf2 hem lmb2 smb1a sqf2 tfs3+ zoo0

Number of observations in data set = 50

General Linear Models Procedure
Dependent Variable: CARBON

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	18	237.98403129	13.22133507	10.29	0.0001
Error	31	39.84809921	1.28542256		
Corrected Total	49	277.83213050			

R-Square	C.V.	Root MSE	CARBON Mean
0.856575	-3.979281	1.1337648	-28.491700

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	3	20.15680869	6.71893623	5.23	0.0049
SPECIES	15	217.82722260	14.52181484	11.30	0.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	3	22.59806954	7.53268985	5.86	0.0027
SPECIES	15	217.82722260	14.52181484	11.30	0.0001

General Linear Models Procedure
Least Squares Means

LOCATION	CARBON LSMEAN	Pr > |T| HO: LSMEAN(i)=LSMEAN(j) i/j	1	2	3	4
mai	-29.2783879	1	.	0.1815	0.0004	0.4149
mcc	-28.6560284	2	0.1815	.	0.0111	0.7100
pit	-27.4586131	3	0.0004	0.0111	.	0.0098
sac	-28.8493062	4	0.4149	0.7100	0.0098	.

NOTE: To ensure overall protection level, only probabilities associated with pre-planned comparisons should be used.

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Analysis Variable : NITROGEN

LOCATION	N Obs	N	Mean	Std Dev	Minimum	Maximum
mai	13	13	7.5800000	2.0048358	2.7200000	10.1400000
mcc	14	14	8.1771429	1.6870978	6.0300000	11.0200000
pit	14	14	9.1278571	1.9510823	3.4000000	11.1400000
sac	9	9	7.7655556	2.6067322	2.9000000	11.3300000

General Linear Models Procedure
Class Level Information

Class	Levels	Values
LOCATION	4	mai mcc pit sac
SPECIES	16	asb1a asb1b asb2 asb3 asb4 asb5 blg1 blg2 blg3 gsf2 hem lmb2 smb1a sqf2 tfs3+ zoo0

Number of observations in data set = 50

General Linear Models Procedure

Dependent Variable: NITROGEN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	18	197.20849206	10.95602734	31.98	0.0001
Error	31	10.61985794	0.34257606		
Corrected Total	49	207.82835000			

R-Square	C.V.	Root MSE	NITROGEN Mean
0.948901	7.125639	0.5853000	8.2140000

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	3	18.74625635	6.24875212	18.24	0.0001
SPECIES	15	178.46223571	11.89748238	34.73	0.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	3	13.01804622	4.33934874	12.67	0.0001
SPECIES	15	178.46223571	11.89748238	34.73	0.0001

General Linear Models Procedure
Least Squares Means

LOCATION	NITROGEN LSMEAN	Pr > |T| HO: LSMEAN(i)=LSMEAN(j) i/j	1	2	3	4
mai	7.88134986	1	.	0.9877	0.0001	0.6880
mcc	7.88500984	2	0.9877	.	0.0001	0.6756
pit	9.02740288	3	0.0001	0.0001	.	0.0001
sac	7.77270476	4	0.6880	0.6756	0.0001	.

NOTE: To ensure overall protection level, only probabilities associated with pre-planned comparisons should be used.

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```
/* Laurel Saito
   Shasta Lake isotope analyses
   Spatial isotope analysis

   Carbon and nitrogen isotope signatures for species collected
   at four locations at Shasta Lake are compared.              */

* Set line and page lengths;

options linesize=80 pagesize=50;

* Measured data;

data isotope;
   input location $ species $ carbon nitrogen;
cards;
mai  asb1a -32.010  7.440
mai  asb1b -27.230  8.030
mai  asb2  -28.650  9.100
mai  asb3  -28.220 10.140
mai  asb4  -28.260 10.070
mai  blg2  -30.980  5.810
mai  blg3  -29.670  7.190
mai  gsf2  -26.170  7.510
mai  hem   -25.590  2.720
mai  smb1a -30.670  7.440
mai  sqf2  -27.330  6.450
mai  tfs3* -29.940  9.670
mai  zoo0  -35.250  6.970
mcc  asb1a -31.540  7.110
mcc  asb1b -29.600  8.690
mcc  asb2  -28.770  9.030
mcc  asb3  -27.880 10.190
mcc  asb4  -26.390  9.980
mcc  asb5  -25.860 11.020
mcc  blg1  -32.110  6.900
mcc  blg2  -29.520  6.350
mcc  blg3  -29.000  7.090
mcc  gsf2  -25.870  6.740
mcc  lmb2  -28.180 10.520
mcc  smb1a -29.450  7.340
mcc  sqf2  -25.670  7.490
mcc  zoo0  -32.520  6.030
pit  asb1a -27.400  8.740
pit  asb1b -27.660  9.810
pit  asb2  -25.960 10.530
pit  asb3  -24.990 10.350
pit  asb4  -24.820 10.540
pit  asb5  -27.970 11.140
pit  blg1  -29.240  7.490
pit  blg2  -27.450  8.970
pit  blg3  -27.880  8.515
pit  gsf2  -26.450  9.290
pit  hem   -25.460  3.400
pit  lmb2  -27.370 10.880
pit  tfs3* -30.140  9.500
pit  zoo0  -33.005  8.635
sac  asb1a -30.150  5.910
sac  asb2  -29.190  9.260
sac  asb3  -28.390  9.890
sac  asb4  -28.640  9.930
sac  asb5  -26.075 11.330
sac  blg3  -29.550  7.050
sac  gsf2  -25.490  7.430
sac  hem   -25.995  2.900
sac  zoo0  -32.980  6.190
;

proc means;
   var carbon;
   class location;
   title 'Shasta Lake carbon isotope spatial analysis';

proc glm;
   class location species;
   model carbon=location species;
   lsmeans location/pdiff;

proc means;
   var nitrogen;
   class location;
   title 'Shasta Lake nitrogen isotope spatial analysis';

proc glm;
   class location species;
   model nitrogen=location species;
   lsmeans location/pdiff;
run;
```


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Analysis Variable : CARBON

LOCATION	N Obs	N	Mean	Std Dev	Minimum	Maximum
mai	16	16	-29.6925000	2.8487412	-35.4300000	-25.5900000
mcc	14	14	-28.7400000	2.2860413	-32.5200000	-25.6700000
pit	17	17	-27.8567647	2.7225660	-34.6800000	-24.8200000
sac	11	11	-29.2045455	2.9658864	-35.3500000	-25.4900000

General Linear Models Procedure
Class Level Information

Class	Levels	Values
LOCATION	4	mai mcc pit sac
SPECIES	19	asb1a asb1b asb2 asb3 asb4 asb5 blg1 blg2 blg3 gsf2 hem lmb2 pasb3 pscu1 pzoo0 smb1a sqf2 tfs3+ zoo0

Number of observations in data set = 58

General Linear Models Procedure

Dependent Variable: CARBON

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	21	380.09109017	18.09957572	14.23	0.0001
Error	36	45.80025681	1.27222936		
Corrected Total	57	425.89134698			

R-Square	C.V.	Root MSE	CARBON Mean
0.892460	-3.912084	1.1279314	-28.831983

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	3	29.66097720	9.88699240	7.77	0.0004
SPECIES	18	350.43011298	19.46833961	15.30	0.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	3	33.00144527	11.00048176	8.65	0.0002
SPECIES	18	350.43011298	19.46833961	15.30	0.0001

General Linear Models Procedure
Least Squares Means

LOCATION	CARBON LSMEAN	Pr > |T| HO: LSMEAN(i)=LSMEAN(j) i/j	1	2	3	4
mai	-29.6687302	1	.	0.1514	0.0001	0.5178
mcc	-29.0267224	2	0.1514	.	0.0046	0.4951
pit	-27.7311498	3	0.0001	0.0046	.	0.0010
sac	-29.3644600	4	0.5178	0.4951	0.0010	.

NOTE: To ensure overall protection level, only probabilities associated with pre-planned comparisons should be used.

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Analysis Variable : NITROGEN

LOCATION	N Obs	N	Mean	Std Dev	Minimum	Maximum
mai	16	16	7.7587500	2.0489115	2.7200000	10.4000000
mcc	14	14	8.1771429	1.6870978	6.0300000	11.0200000
pit	17	17	8.9829412	1.8828885	3.4000000	11.1400000
sac	11	11	7.8463636	2.4428765	2.9000000	11.3300000

General Linear Models Procedure
Class Level Information

Class	Levels	Values
LOCATION	4	mai mcc pit sac
SPECIES	19	asb1a asb1b asb2 asb3 asb4 asb5 blg1 blg2 blg3 gsf2 hem lmb2 pasb3 pscu1 pzoo0 smb1a sqf2 tfs3+ zoo0

Number of observations in data set = 58

General Linear Models Procedure

Dependent Variable: NITROGEN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	21	211.09061660	10.05193412	17.98	0.0001
Error	36	20.12998168	0.55916616		
Corrected Total	57	231.22059828			

R-Square	C.V.	Root MSE	NITROGEN Mean
0.912940	9.080249	0.7477741	8.2351724

Source	DF	Type I SS	Mean Square	F Value	Pr > F
LOCATION	3	14.84738007	4.94912669	8.85	0.0002
SPECIES	18	196.24323652	10.90240203	19.50	0.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
LOCATION	3	10.31538915	3.43846305	6.15	0.0017
SPECIES	18	196.24323652	10.90240203	19.50	0.0001

General Linear Models Procedure
Least Squares Means

LOCATION	NITROGEN LSMEAN	Pr > |T| HO: LSMEAN(i)=LSMEAN(j) i/j	1	2	3	4
mai	8.01115899	1	.	0.7108	0.0025	0.6798
mcc	7.90263962	2	0.7108	.	0.0013	0.9513
pit	8.89154126	3	0.0025	0.0013	.	0.0020
sac	7.88264488	4	0.6798	0.9513	0.0020	.

NOTE: To ensure overall protection level, only probabilities associated with pre-planned comparisons should be used.

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```
/* Laurel Saito
   Shasta Lake isotope analyses
   Spatial isotope analysis

   Carbon and nitrogen isotope signatures for species collected
   at four locations at Shasta Lake are compared. This
   comparison includes preliminary data.                    */

* Set line and page lengths;

options linesize=80 pagesize=50;

* Measured data;

data isotope;
   input location $ species $ carbon nitrogen;
cards;
mai  asb1a -32.010  7.440
mai  asb1b -27.230  8.030
mai  asb2  -28.650  9.100
mai  asb3  -28.220 10.140
mai  asb4  -28.260 10.070
mai  blg2  -30.980  5.810
mai  blg3  -29.670  7.190
mai  gsf2  -26.170  7.510
mai  hem   -25.590  2.720
mai  psab1 -28.490  9.510
mai  pscu1 -31.190 10.400
mai  proo0 -35.120  5.690
mai  smb1a -30.670  7.440
mai  sqf2  -27.330  6.450
mai  tfa1* -29.940  9.670
mai  zoo0  -35.250  6.970
mcc  asb1a -31.540  7.110
mcc  asb1b -29.600  8.690
mcc  asb2  -28.770  9.010
mcc  asb3  -27.880 10.190
mcc  asb4  -26.190  9.980
mcc  asb5  -25.860 11.020
mcc  blg1  -32.110  6.900
mcc  blg2  -29.520  6.150
mcc  blg3  -29.000  7.090
mcc  gsf2  -25.870  6.740
mcc  lmb2  -28.180 10.520
mcc  smb1a -29.450  7.340
mcc  sqf2  -25.670  7.490
mcc  zoo0  -32.520  6.030
pit  asb1a -27.400  8.740
pit  asb1b -27.660  9.810
pit  asb2  -25.960 10.530
pit  asb3  -24.990 10.350
pit  asb4  -24.820 10.540
pit  asb5  -27.970 11.140
pit  blg1  -29.240  7.490
pit  blg2  -27.450  8.970
pit  blg3  -27.880  8.515
pit  gsf2  -26.450  9.290
pit  hem   -25.460  3.400
pit  lmb2  -27.370 10.680
pit  psab1 -24.880 10.230
pit  pscu1 -28.210  7.240
pit  proo0 -34.680  7.450
pit  tfa1* -30.140  9.500
pit  zoo0  -33.005  8.635
sac  asb1a -30.150  5.310
sac  asb2  -29.190  9.260
sac  asb3  -28.190  9.890
sac  asb4  -28.640  9.930
sac  asb5  -26.075 11.330
sac  blg3  -29.550  7.050
sac  gsf2  -25.490  7.430
sac  hem   -25.095  2.900
sac  psab1 -29.440  9.790
sac  proo0 -35.350  6.630
sac  zoo0  -32.980  6.190


proc means;
   var carbon;
   class location;
   title 'Shasta Lake carbon isotope spatial analysis with preliminary data';

proc glm;
   class location species;
   model carbon=location species;
   lsmeans location/pdiff;

proc means;
   var nitrogen;
   class location;
   title 'Shasta Lake nitrogen isotope spatial analysis with preliminary data';

proc glm;
   class location species;
   model nitrogen=location species;
   lsmeans location/pdiff;
run;
```


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APPENDIX J. Calculation of grouped stable isotope data

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Zooplankton

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
7/98 w/64um net	ZOO	-32.52	6.03	MCC
7/98 w/505um net	ZOO	-35.25	6.97	MAI
7/98 w/505um net	ZOO	-32.98	6.19	SAC
6/98 w/505um net	PZOO	-35.43	5.69	MAI
6/98 w/505um net	PZOO	-35.35	6.83	SAC
	average	-33.583	6.396666667	3
	paverage	-34.305	6.302	5
	std dev	1.46169	0.502924779	
	psid dev	1.43113	0.503706264	

Insects

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
herb/det/carnv	CHI midges	-28.3	--	SAC
	HEM	-25.59	2.72	MAI
terr/aq insects	HEM	-25.995	2.9	SAC
algae, det	HEX mayfly	-23.12	1.68	SAC
herb/det/carnv	TIP cranefly	-22.52	--	SAC
	Beetle	-24.59	2.79	SAC
	Flying beetle	-26.25	5.77	SAC
	HYM	-27.15	3.09	SAC
	MOTH	-29.01	4.1	MAI
	PHEM	-27.41	4.52	SAC
	average	-25.836	3.292857143	9/7
	paverage	-25.994	3.44625	10/8
	std dev	2.18101	1.301636333	
	psid dev	2.11565	1.280802822	
	aq ins avg	-25.489	2.955	6/4
	aq ins std	2.29283	1.173754659	
	terr ins avg	-26.75	3.9375	4
	terr ins std	1.84239	1.34405791	

Bluegill

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
0-49 mm	BLG1	-32.11	6.9	MCC
50-66 mm	BLG2	-30.98	5.81	MAI
50-66 mm	BLG2	-29.52	6.35	MCC
67+ mm	BLG3+	-29.67	7.19	MAI
67+ mm	BLG3+	-29	7.09	MCC
67+ mm	BLG3+	-29.55	7.05	SAC
67+ mm	PBLG3+	-24.55	6.45	MAI
	average	-30.136	6.731666667	6
	paverage	-28.34	6.691428571	7
	std dev	1.19857	0.54097751	
	psid dev	2.3667	0.505187377	

Threadfin shad

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
0-60 mm	ETFS1	-31.21	6.92	MAI
61-100 mm	ETFS2	-30.07	8.95	MAI
101+ mm	TFS3+	-29.94	9.67	MAI
	average	-29.94	9.67	1
	saverage	-30.407	9.18	3
	psid dev	0.69874	0.424617475	

Green sunfish

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
50-100 mm	GSF2/3	-28.17	7.51	MAI
50-100 mm	GSF2/3	-25.87	6.74	MCC
50-100 mm	GSF2/3	-25.49	7.43	SAC
	average	-26.843	7.226666667	3
	std dev	0.34078	0.423359579	

Crayfish

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
	ECRA	-31.09	7.32	MAI

Sacramento squawfish

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
75-142 mm	SQF2	-27.33	6.45	MAI
75-142 mm	SQF2	-25.67	7.49	MCC
	average	-26.5	6.97	2
	std dev	1.1738	0.735391052	

estimated phytoplankton from zooplankton with $\Delta\delta^{13}C = 1^{0}/_{00}$ and $\Delta\delta^{15}N = 3^{0}/_{00}$

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
		-36.308	3.302	

Year class 1 ASB

Size	Species	$\delta^{13}C$	$\delta^{15}N$	n
0-66 mm	ASB1A	-32.01	7.44	MAI
0-66 mm	ASB1A	-31.54	7.11	MCC
0-66 mm	ASB1A	-30.15	5.91	SAC
67-113 mm	ASB1B	-27.23	8.03	MAI
67-113 mm	ASB1B	-29.6	8.69	MCC
0-66 mm	PASB1A	-31.01	5.74	SAC
	average	-30.108	7.436	5
	paverage	-30.257	7.183333333	6
	std dev	1.86487	1.044068983	
	psid dev	1.7258	1.162525985	

YC2 ASB

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
114-203 mm	ASB2	-28.65	9.1	MAI
114-203 mm	ASB2	-28.77	9.03	MCC
114-203 mm	ASB2	-29.19	9.26	SAC
	average	-28.87	9.13	3
	std dev	0.26365	0.117898281	

YC3 ASB

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
204-290 mm	ASB3	-28.22	10.14	MAI
204-290 mm	ASB3	-27.88	10.19	MCC
204-290 mm	ASB3	-28.39	9.89	SAC
204-290 mm	PASB3	-28.49	9.51	MAI
204-290 mm	PASB3	-29.44	9.79	SAC
	average	-28.183	10.07333333	3
	paverage	-28.484	9.904	5
	std dev	0.25968	0.160727513	
	psid dev	0.56252	0.276550178	

Big ASB (>290 mm)

Size	Species	$\delta^{13}C$	$\delta^{15}N$	n
291-370 mm	ASB4	-28.26	10.07	MAI
291-370 mm	ASB4	-28.39	9.98	MCC
291-370 mm	ASB4	-28.64	9.93	SAC
371+ mm	ASB5+	-25.88	11.02	MCC
371+ mm	ASB5+	-26.075	11.33	SAC
291-370 mm	PASB4	-28.7	9.87	SAC
	average	-27.045	10.466	5
	paverage	-27.321	10.36666667	6
	std dev	1.3033	0.658354008	
	psid dev	1.34736	0.637139441	

Year class 1 smallmouth bass

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
0-66 mm	SMB1A	-30.67	7.44	MAI
0-66 mm	SMB1A	-29.45	7.34	MCC
	average	-30.06	7.39	2
	std dev	0.86267	0.070710678	

Medium smallmouth bass

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
114-203 mm	SMB2	-27.69	9.71	MAI
204-290 mm	SMB3	-26.99	10.13	MAI
	average	-27.34	9.92	2
	std dev	0.49497	0.296984848	

Brown trout

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
411+ mm	BRT5+	-29.02	10.54	MAI

Chinook salmon

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
231-460 mm	CHS2	-28.22	11.6	MAI
231-460 mm	ECHS2	-27.6	12.1	MAI
	average	-28.22	11.6	1
	saverage	-27.91	11.85	2

Rainbow trout

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
291-460 m	RBT3	-24.33	9.46	MAI

Catfish

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
201-280 m	BBH4	-29.26	10.01	MAI
201-350 m	CCF3	-26.07	7.8	SAC
451+ mm	CCF6+	-25.45	8.75	MAI
451+ mm	CCF6+	-27.69	9.05	SAC
301+ mm	WCF7+	-26.56	11	MAI

Feathers

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
adult slave tail feather	FEAA	-24.15	12.07	MAI
ng dead flight feather	FEAN	-25.16	12.03	MAI
	average	-24.655	12.05	2

Sculpins

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
0-50 mm	PSCU1	-31.19	10.4	MAI
51-72 mm	PSCU2	-28.92	7.935	MAI
51-72 mm	ESCU2	-26.645	6.93	MAI
73-89 mm	ESCU3	-24.07	7.56	MAI
	paverage	-30.055	9.1675	2
	saverage	-25.3575	7.245	2
	totaverag	-27.7063	8.20625	4
	totstd dev	3.052773	1.52015	

Year class 1 bass

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
0-66 mm	ASB1A	-32.01	7.44	MAI
0-66 mm	ASB1A	-31.54	7.11	MCC
0-66 mm	ASB1A	-30.15	5.91	SAC
67-113 m	ASB1B	-27.23	8.03	MAI
67-113 m	ASB1B	-29.6	8.69	MCC
0-66 mm	PASB1A	-31.01	5.74	
0-66 mm	LBM1A	-31.52	8.39	MCC
0-66 mm	SMB1A	-30.67	7.44	MAI
0-66 mm	SMB1A	-29.45	7.34	MCC
	average	-30.3633	7.343333	6
	std dev	1.46737	1.004913	

Big bass (> 290 mm)

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
291-370 m	ASB4	-28.26	10.07	MAI
291-370 m	ASB4	-28.39	9.98	MCC
291-370 m	ASB4	-28.64	9.93	SAC
371+ mm	ASB5+	-25.88	11.02	MCC
371+ mm	ASB5+	-26.075	11.33	SAC
291-370 m	PASB4	-28.7	9.87	SAC
	average	-27.3208	10.39667	6
	std dev	1.347362	0.637139	

Medium bass (ASB and LMB)

Size	Species	$\delta^{13}C$	$\delta^{15}N$	LOC
114-203 m	ASB2	-28.65	9.1	MAI
114-203 m	ASB2	-28.77	9.03	MCC
114-203 m	ASB2	-29.19	9.26	SAC
204-290 m	ASB3	-28.22	10.14	MAI
204-290 m	ASB3	-27.88	10.19	MCC
204-290 m	ASB3	-28.39	9.89	SAC
204-290 m	PASB3	-28.49	9.51	MAI
204-290 m	PASB3	-29.44	9.79	SAC
114-203 m	LMB2	-28.16	10.52	MCC
	average	-28.5789	9.714444	9
	std dev	0.497062	0.522865	

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APPENDIX K. Data and calculations for estimation of Shasta detritus and periphyton signatures

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Carbon isotopes

category	at Shasta?	δ¹³C (‰) Average	Range	Number	Description	Source	Notes
PHY	Y	-33.2	~0.11	2	Phytoplankton (< 250 um)	Angradi (1993)	Island Park Reservoir, ID
PERI	Y	-17.8		3	epilithon scraped from rocks about 35 cm deep	Bunn et al. (1989)	Allt Lake, Quebec
PHY	Y		-33.49 to -24.19	28 x 2	microplankton (> 20 um, < 250 um)	del Giorgio and France (1996)	Canadian lakes
PHY	Y	-17*			Unpalatable blue green algae	Estep and Vigg (1985)	Lahontan lake system, NV
PHY	Y	-23.3			green algae (Cladophora)	Estep and Vigg (1985)	Pyramid Lake, NV
PHY	Y	-23.3			green algae (Cladophora)	Estep and Vigg (1985)	Pyramid Lake, NV
PERI	Y	-17	~2		epilithon (mostly diatoms) scraped from sheltered rock s	France (1995d)	Canadian Shield lakes
PERI	Y	-27	~3		epiphytic and metaphytic filamentous algae	France (1995d)	Canadian Shield lakes
PERI	Y	-17	-21 to -15		Benthic epilithon	France (1997a)	Boreal Canadian Shield lakes
PHY	Y	-25.36			filamentous green alga, Prasiola sp.	Fry (1991a)	LTER site: Andrews Forest, OR
PHY	Y	-25.53			benthic algae, surface film, mostly diatoms	Fry (1991a)	LTER site: Andrews Forest, OR
PHY	Y	-18.41			filamentous algae (Mugiochia?)	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
PHY	Y	-29.44			phytoplankton	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
PERI	Y	-19.36			epilithiphyton	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
MAC	Y	-28.44			macrophyte, Lobelia	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
MAC	Y	-24.29			aquatic plant, Utricularia	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
PHY	Y	-32.71			phytoplankton, 0.5 m depth on filter	Fry (1991a)	LTER site: North temperate lakes, WI (Crystal Bog)
PHY	Y	-27.3			phytoplankton, 3.0 m depth on filter	Fry (1991a)	LTER site: North temperate lakes, WI (Trout Lake)
PHY	Y		-8 to -27		Macroalgae	Fry and Sherr (1984)	Lit review
PHY	Y		-35 to -45		Phytoplankton in small lakes	Fry and Sherr (1984)	Lit review
PHY	Y	-28.1	SD 6.0	6	Macroalgae (Cladophora)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	-24.5	SD 0.1	5	Macroalgae (Mougeotia)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	-12.7	SD 0.2	8	Macroalgae (Rivularia)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	-13.2	SD 0.1	4	Macroalgae (Spirogyra)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	-28	SD 0.3	4	Macroalgae (Zygnema)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	-32.6	SD 0.4	3	Microalgae (Chlorococcum)	Gu et al. (1997)	Smith Lake, AK
PERI	Y	-30.8	SD 1.8	13	unidentified epiphytic algae	Gu et al. (1997)	Smith Lake, AK
PHY	Y	-33	SD 3	8	phytoplankton (POM)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	-14.1			Diatoms	Gu et al. (1998)	Lake Apopka, FL
PHY	Y	-13.4			Small cyanobacteria	Gu et al. (1998)	Lake Apopka, FL
PHY	Y	-3			Microcystis	Gu et al. (1998)	Lake Apopka, FL
PERI	Y	-16.62			Epilithon	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
PERI	Y	-17.24			Epilithon	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
PERI	Y	-19.35			Epiphyton	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
PERI	Y	-20.13			Epiphyton	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
MAC	Y	-12.37			Potamogeton	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
MAC	Y	-11.4			Potamogeton	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
MAC	Y	-18.1			Potamogeton sp. (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
MAC	Y	-20.67			Potamogeton sp. (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
MAC	Y	-21.51			Potamogeton sp. (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
MAC	Y	-18.53			Ranunculus aquatilis (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
MAC	Y	-18.28			Ranunculus aquatilis (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
PHY	Y	-20.9			Algae	Hobson and Welch (1995)	Arctic lakes
PERI	Y		-29.2 to -27.8		biofilm (epilithon) samples	Junger and Planas (1994)	Laflamme Creek just d/s of Lac Laflamme, Quebec
PHY	Y	-28.1			stream seston	Junger and Planas (1994)	Laflamme Creek just d/s of Lac Laflamme, Quebec
MAC	Y	-21.8	-22.5 to -21.2	5	pondweed (submersed), *Potamogeton foliosus*	Keough et al. (1998)	Allouez Bay wetland (Lake Superior)
PHY	Y	-31.2	-32.6 to -30.3	12	phytoplankton, 53-163 μm, diatoms	Keough et al. (1998)	Allouez Bay wetland (Lake Superior)
PERI	Y	-27.8	-28.5 to -26.4	4	epiphytes on *Potamogeton*, diatoms	Keough et al. (1998)	Allouez Bay wetland (Lake Superior)
PHY	Y	-27.3	-27.9 to -26.7	3	phytoplankton, 53-163 μm, diatoms	Keough et al. (1998)	Lake Superior
PHY	Y	-26.5	-26.5 to -26.3	2	phytoplankton, <53 μm, *chrysophytes*	Keough et al. (1998)	Lake Superior
PHY	Y	-28.5	-32 to -21.6		POM and algae	King et al. (1992)	Arctic lakes
PHY	Y	-45.9			plankton	Rau (1980)	Lake Findley, WA
PERI	Y	-23			Periphyton	Estep and Vigg (1985)	Lahontan lake system, NV
PERI	Y	-30.1			Periphyton	Hecky and Hesslein (1995)	South Lake (arctic lake)
PERI	Y	-27.66			Periphyton	Hecky and Hesslein (1995)	South Lake (arctic lake)
PERI	Y	-28.76			Periphyton	Hecky and Hesslein (1995)	South Lake (arctic lake)
PERI	Y	-28.46			Periphyton	Hecky and Hesslein (1995)	South Lake (arctic lake)
PERI	Y	-34.8			periphyton	Rau (1980)	Lake Findley, WA
[illegible]							

averages

category	Average	Description
PHY	-25.05	phytoplankton only
MAC	-15.883	macrophytes
PER	-23.23	periphyton
DET	-23.3182	detritus

Nitrogen isotopes

category	at Shasta?	δ¹⁵N (‰) Average	Range	Number	Description	Source	Notes
PHY	Y	6	7.0 to 7.0	2	Cladophora with epiphytes	Angradi (1994)	Colorado River 2 km d/s of dam (AZ)
PHY	Y	6.7			Cladophora with epiphytes 25d in situ	Angradi (1994)	Colorado River 2 km d/s of dam (AZ)
PHY	Y	6.6			Cladophora with epiphytes 53d in situ	Angradi (1994)	Colorado River 2 km d/s of dam (AZ)
PHY	Y	5.2	5.2 to 5.2	2	Cladophora without epiphytes	Angradi (1994)	Colorado River 2 km d/s of dam (AZ)
PHY	Y	9			Ulothrix tenuissima (green alga)	Angradi (1994)	Colorado River 1 km d/s of dam (AZ)
PHY	Y		1 to 3		Unpalatable blue green algae	Estep and Vigg (1985)	Lahontan lake system, NV
PHY	Y	6			green algae (Cladophora)	Estep and Vigg (1985)	Pyramid Lake, NV
PHY	Y	4.2			green algae (Cladophora)	Estep and Vigg (1985)	Pyramid Lake, NV
PHY	Y	8.6			algae	Estep and Vigg (1985)	Lahontan Reservoir (Aug 1981)
PHY	Y	8.2			algae	Estep and Vigg (1985)	Lahontan Reservoir (Oct 1982)
PHY	Y	-1.43			filamentous green alga, Prasiola sp.	Fry (1991a)	LTER site: Andrews Forest, OR
PHY	Y	-1.08			benthic algae, surface film, mostly diatoms	Fry (1991a)	LTER site: Andrews Forest, OR
PHY	Y	2.03			filamentous algae (Mugiochia?)	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
PERI	Y	-0.7			epilithiphyton	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
MAC	Y	2.37			macrophyte, Lobelia	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
MAC	Y	2.93			aquatic plant, Utricularia	Fry (1991a)	LTER site: Hubbard Brook, Mirror Lake, NH
PHY	Y	0.5			phytoplankton, 0.5 m depth on filter	Fry (1991a)	LTER site: North temperate lakes, WI (Crystal Bog)
PHY	Y	2.75			phytoplankton, 3.0 m depth on filter	Fry (1991a)	LTER site: North temperate lakes, WI (Trout Lake)
PHY	Y	9.5	SD 0.5	2	Macroalgae (Cladophora)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	4.2	SD 0.1	3	Macroalgae (Mougeotia)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	0.7	SD 1.7	9	Macroalgae (Rivularia)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	3.6	SD 1.2	4	Macroalgae (Spirogyra)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	14.5	SD 0.8	2	Macroalgae (Zygnema)	Gu et al. (1997)	Smith Lake, AK

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PHY	Y	4.6	SD 0.5	4	Microalgae (Chlorococcum)	Gu et al. (1997)	Smith Lake, AK
PERI	Y	5	SD 1.2	13	unidentified epiphytic algae	Gu et al. (1997)	Smith Lake, AK
PHY	Y	1.7	SD 0.2	8	phytoplankton (POM)	Gu et al. (1997)	Smith Lake, AK
PHY	Y	4			Diatoms	Gu et al. (1995)	Lake Apopka, FL
PHY	Y	4.2			Microcystis	Gu et al. (1995)	Lake Apopka, FL
PHY	Y	5.1			Small cyanobacteria (bulk plankton)	Gu et al. (1995)	Lake Apopka, FL
PERI	Y	1.15			Epilithon	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
PERI	Y	0.9			Epilithon	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
PERI	Y	0.74			Epiphyton	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
PERI	Y	0.67			Epiphyton	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
MAC	Y	1.78			Potamogeton	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
MAC	Y	-0.01			Potamogeton	Hecky and Hesslein (1995)	Canadian Shield lake ELA 240
MAC	Y	-1.46			Potamogeton sp. (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
MAC	Y	0.66			Potamogeton sp. (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
MAC	Y	2.93			Potamogeton sp. (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
MAC	Y	1.15			Ranunculus aquatilis (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
MAC	Y	0.23			Ranunculus aquatilis (macrophyte)	Hecky and Hesslein (1995)	South Lake (arctic lake)
PHY	Y	2.3		1.6 to 3.2	POM and algae	Kling et al. (1992)	Arctic lakes
PERI	Y			4 to 6	Periphyton	Estep and Vigg (1985)	Lahontan lake system, NV
PERI	Y	4.03			Periphyton	Hecky and Hesslein (1995)	South Lake (arctic lake)
PERI	Y	2.74			Periphyton	Hecky and Hesslein (1995)	South Lake (arctic lake)
PERI	Y	2.39			Periphyton	Hecky and Hesslein (1995)	South Lake (arctic lake)
PERI	Y	3.9	SD 3.2		Periphyton (drift input from salmon) (annual)	Kline et al. (1993)	Bristol Bay, AK
PERI	Y	6.3	SD 3.4		Periphyton (drift input from salmon) (Aug - Sep)	Kline et al. (1993)	Bristol Bay, AK
PERI	Y	3.9	SD 1.6		Periphyton (low salmon spawning density) (annual)	Kline et al. (1993)	Bristol Bay, AK
PERI	Y	3.6	SD 1.6		Periphyton (low salmon spawning density) (Aug - Sep)	Kline et al. (1993)	Bristol Bay, AK
PERI	Y	6.1	SD 2.7		Periphyton (high salmon spawning density) (annual)	Kline et al. (1993)	Bristol Bay, AK
PERI	Y	8.6	SD 4.2		Periphyton (high salmon spawning density) (Aug - Sep)	Kline et al. (1993)	Bristol Bay, AK

averages

PHY	4.7068
MAC	1.173333
PER	3.154667
DET	1.85

phytoplankton only
macrophytes
periphyton
detritus

Yellow shading indicates values used for phytoplankton signature estimate
Green shading indicates values used for periphyton signature estimate
Blue shading indicates values used for macrophyte signature estimate

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Angradi (1994): 1-2 km d/s of Glen Canyon Dam on CO River

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Chironomids	-32.3	10.3
Rainbow trout adult	-23.7	5.6

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
DIC	-7	
Cladophora w/epiphytes	-33.5	7
tamarisk litter	-27.6	3.7
Δ (chironomids) - per	1.2	3.3
Δ (chironomids) - det	-4.7	6.6
Δ (trout) - per	9.8	-1.4
Δ (trout) - det	3.9	1.9

Bunn et al. (1989): Alik Lake, Quebec

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Chironomids	-21.5	
Tipula	-23.75	

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
decomposing leaves	-26.5	
epilithon	-17.8	
Δ (chironomids) - det	5	
Δ (chironomids) - per	-3.7	
Δ (tipula) - det	2.75	
Δ (tipula) - per	-5.95	

Estep and Vigg (1985): Pyramid Lake, NV

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Chironomids	-23	6.5

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-23	5
Δ (chironomids)	0	1.5

France (1995a) recommends -28 ‰ for detritus in streams

Fry (1991a): Hubbard Brook, Mirror Lake, NH

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
crawfish	-26.73	6.99
smallmouth bass	-28.395	9.46

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
epilithiphyton	-19.36	-0.7
Δ (crawfish)	-7.37	7.69
Δ (smallmouth bass)	-9.035	10.16

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomids	-28.3	--
rainbow trout	-24.33	9.46

if our chironomids are like CO river chironomids

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-29.5	--
detritus	-23.6	--

if our rainbow trout is like CO river rainbow

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-34.13	10.86
detritus	-28.23	7.56

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomids	-28.3	--
tipula	-22.52	--

if our chironomid is like Alik Lake chironomid

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
detritus	-33.3	--
periphyton	-24.6	--

if our tipula is like Alik Lake tipula

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
detritus	-25.27	--
periphyton	-16.57	--

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomids	-28.3	--

if our crayfish is like Green Lake crayfish

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-28.3	--

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
crayfish	-31.09	7.32
smallmouth bass (YC1)	-30.06	7.39
smallmouth bass (medium)	-27.34	9.92

if our crayfish is like Hubbard Brook crayfish

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-23.72	-0.37

if our YC1 SMB is like Hubbard Brook SMB

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-21.025	-2.77

if our medium SMB is like Hubbard Brook SMB

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-18.305	-0.24

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Fry (1991a): Konza Prairie, KS small perennial stream

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
small crayfish	*-24.73*	*3.8*

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
stream leaf detritus	*-27.85*	*0.53*
stream pool sediment	*-19.215*	*0.37*
periphyton	*-26.05*	*0.8*
Δ (crayfish) - leaf det	3.12	3.27
Δ (crayfish) - sed	-5.515	3.43
Δ (crayfish) - peri	1.32	3

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
crayfish	*-31.09*	*7.32*

if our crayfish is like Konza Prairie crayfish

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
detritus	-34.21	4.05
periphyton	-32.41	4.32

Fry (1986): Green Lake, NY

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Orconectes obscurus	*-27*	

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
leaf detritus	*-24.8*	
Δ (crayfish)	-2.2	

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
crayfish	*-31.09*	*7.32*

if our crayfish is like Green Lake crayfish

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
leaf detritus	-28.89	

Gu et al. (1997): Smith Lake, AK

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Chironomus	*-26.5*	*-1.3*
Trichorythodes (mayfly)	*-25*	*2*

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
unID'd epiphytic algae (also detritus)	*-30.8*	*5*
sediments (FPOM)	*-29.2*	*0.2*
littoral alder leaves (Alnus)	*-26.6*	*-1.3*
littoral birch leaves (Betula)	*-28*	*-4.8*
Δ (chironomus) -peri	4.3	-6.3
Δ (chironomus) -FPOM	2.7	-1.5
Δ (chironomus) -alder	0.1	0
Δ (chironomus) -birch	1.5	3.5
Δ (mayfly) - peri	5.8	-3
Δ (mayfly) - FPOM	4.2	1.8
Δ (mayfly) - alder	1.6	3.3
Δ (mayfly) - birch	3	6.8

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomids	*-28.3*	--
mayfly (HEX)	*-23.12*	*1.68*

if our chironomid is like Smith Lake chironomid

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-32.6	--
FPOM	-31	--
alder det	-28.4	--
birch det	-29.8	--

if our mayfly is like Smith Lake mayfly

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-28.92	4.68
FPOM	-27.32	-0.12
alder det	-24.72	-1.62
birch det	-26.12	-5.12

Harrington et al. (1998): tributaries in Vermont

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Bingo Brook mayfly		*2.37*
West Branch mayfly		*3.07*
Bethel-Gilead mayfly		*4.62*
Peavine mayfly		*6.43*
First Branch mayfly		*8.39*
Third Branch mayfly		*10*

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Bingo Brook detritus		*-0.21*
West Branch detritus		*0.19*
Bethel-Gilead detritus		*0.22*
Peavine detritus		*4.7*
First Branch detritus		*1.91*
Third Branch detritus		*1.33*
Δ (Bingo Brook mayfly)		2.58
Δ (West Branch mayfly)		2.88
Δ (Bethel-Gilead mayfly)		4.4
Δ (Peavine mayfly)		1.73
Δ (First Branch mayfly)		6.48
Δ (Third Branch mayfly)		8.67

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
mayfly (HEX)	*-23.12*	*1.68*

If our mayfly is like VT tributaries mayfly

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
detritus (Bingo Brook)		-0.9
detritus (West Branch)		-1.2
detritus (Bethel-Gilead)		-2.72
detritus (Peavine)		-0.05
detritus (First Branch)		-4.8
detritus (Third Branch)		-6.99

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Hecky and Hesslein (1995): Canadian Shield Lake ELA 240

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Orconectes (small)	*-26.58*	*5.82*
Orconectes (small)	*-19.35*	*4.7*
Orconectes (small)	*-24.03*	*5.39*
Orconectes (small)	*-24.21*	*5.12*
average	-23.5425	5.2575

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Orconectes (large)	*-29.42*	*6.83*
Orconectes (large)	*-27.34*	*8.24*
Orconectes (large)	*-26.05*	*6.43*
Orconectes (large)	*-25.88*	*7.08*
average	-27.1725	7.145

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Epilithon	*-16.62*	*1.15*
Epilithon	*-17.24*	*0.9*
average	-16.93	1.025
Δ (sml crayfish)	-6.6125	4.2325
Δ (lg crayfish)	-10.2425	6.12

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Epiphyton	*-19.35*	*0.74*
Epiphyton	*-20.13*	*0.67*
average	-19.74	0.705
Δ (sml crayfish)	-3.8025	4.5525
Δ (lg crayfish)	-7.4325	6.44

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Epilithon	*-16.62*	*1.15*
Epilithon	*-17.24*	*0.9*
Epiphyton	*-19.35*	*0.74*
Epiphyton	*-20.13*	*0.67*
average	-18.335	0.865
Δ (sml crayfish)	-5.2075	4.3925
Δ (lg crayfish)	-8.8375	6.28

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Potamogeton	*-12.37*	*1.76*
Potamogeton	*-11.4*	*-0.01*
average	-11.885	0.875
Δ (sml crayfish)	-11.6575	4.3825
Δ (lg crayfish)	-15.2875	6.27

Junger and Planas (1994): Quebec rivers

Laflamme Creek just d/s of Lac Laflamme

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomid pupae	*-29.2*	

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
biofilm (epilithon)	*-28.5*	
Δ (chironomids)	-0.7	

De l'Aqueduc Creek, Quebec

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomid pupae	*-25.6*	

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
biofilm (epilithon)	*-25.15*	
Δ (chironomids)	-0.45	

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
crayfish	*-31.09*	*7.32*

if our crayfish is like small crayfish in Canada:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Epilithon	-24.4775	3.0875

if our crayfish is like large crayfish in Canada:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Epilithon	-20.8475	1.2

if our crayfish is like small crayfish in Canada:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Epiphyton	-27.2875	2.7675

if our crayfish is like large crayfish in Canada:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Epiphyton	-23.6575	0.88

if our crayfish is like large crayfish in Canada:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Periphyton	-22.2525	1.04

if our crayfish is like small crayfish in Canada:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Potamogeton	-19.4325	2.9375

if our crayfish is like large crayfish in Canada:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Potamogeton	-15.8025	1.05

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomid	*-28.3*	

if our chironomid is like Laflamme Creek chironomid

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-27.6	

if our chironomid is like De l'Aqueduc Creek chironomid

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-27.85	

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Peterson et al. (1993): tundra river ecosystem

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Baetis (mayfly) - control	*-30.9*	*3.3*
Baetis (mayfly) - P fert	*-29.8*	*3.2*
Baetis (mayfly) - N+P fert	*-28.6*	*-2.5*
average	-29.76667	1.333333

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Orthocladius (chironomid) - control	*-24.8*	*1.9*
Orthocladius (chironomid) - P fert	*-20.9*	*2.4*
Orthocladius (chironomid) - N+P fert	*-19*	*-4.6*
average	-21.56667	-0.1

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Epilithon - control	*-29.2*	*1.9*
Epilithon - P fert	*-27.7*	*2.5*
Epilithon - N+P fert	*-28*	*0*
average	-28.3	1.466667
Δ epilithon-control(mayfly-control)	1.7	-1.4
Δ epilithon-P fert (mayfly-P fert)	2.1	-0.7
Δ epilithon-N+P fert (mayfly-N+P fert)	0.6	2.5
Δ epilithon-avg (mayfly-avg)	1.466667	0.133333
Δ epilithon-control(chiro-control)	-4.4	0
Δ epilithon-P fert (chiro-P fert)	-6.8	0.1
Δ epilithon-N+P fert (chiro-N+P fert)	-9	4.6
Δ epilithon-avg (chiro-avg)	-6.733333	1.566667

Rau (1980): Lake Findlay, WA

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Chironomids	*-29.6*	

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	*-34.6*	
forest detritus	-27.3	
Δ (chironomids) - per	5	
Δ (chironomids) - det	-2.3	

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomids	*-28.3*	--
mayfly (HEX)	*-23.12*	*1.68*

if our chironomid is like chironomid in tundra river ecosystem:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	
periphyton	-32.7	--	control
periphyton	-35.1	--	p-fert
periphyton	-37.3	--	n+p fert
periphyton	-35.03333	--	avg

if our mayfly is like mayfly in tundra river ecosystem

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	
periphyton	-21.42	0.28	control
periphyton	-21.02	0.98	p-fert
periphyton	-22.52	4.18	n+p fert
periphyton	-21.65333	1.813333	avg

Shasta data:

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
chironomid	*-28.3*	

if our chironomid is like Lake Findlay chironomid

species	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
periphyton	-23.3	
detritus	-26	

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summary 1 - all possible comparisons

species (study)	estimated $\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	Shasta source	location
periphyton (Angradi 1994)	-29.5		chironomid	1-2 km d/s of Glen Canyon Dam on CO River
periphyton (Angradi 1994)	-34.13	10.87	rainbow trout	1-2 km d/s of Glen Canyon Dam on CO River
periphyton (Bunn et al. 1989)	-24.6		chironomid	Alik Lake, Quebec
periphyton (Bunn et al. 1989)	-16.57		tipula	Alik Lake, Quebec
periphyton (Estep and Vigg 1985)	-28.3		crayfish	Pyramid Lake, NV
periphyton (Fry 1991a)	-23.72	-0.37	crayfish	Hubbard Brook, Mirror Lake, NH
periphyton (Fry 1991a)	-18.305	-0.24	med smallmouth bas	Hubbard Brook, Mirror Lake, NH
periphyton (Fry 1991a)	-32.41	4.32	crayfish	Konza Prairie, KS perennial stream
periphyton (Gu et al. 1997)	-32.6		chironomid	Smith Lake, AK
periphyton (Gu et al. 1997)	-28.92	4.68	mayfly	Smith Lake, AK
periphyton (Hecky and Hesslein 1995)	-22.25	1.04	crayfish	Canadian Shield Lake ELA 240
periphyton (Junger and Planas 1994)	-27.6		chironomid	Laflamme Creek just d/s of Lac Laflamme, Quebec
periphyton (Junger and Planas 1994)	-27.85		chironomid	De l'Aqueduc Creek, Quebec
periphyton (Peterson et al. 1993)	-32.7		chironomid	tundra river ecosystem (control)
periphyton (Peterson et al. 1993)	-21 42	0 28	mayfly	tundra river ecosystem (control)
periphyton (Rau 1980)	-23.3		chironomid	Lake Findlay, WA
average	-26.51094	2.94		
detritus (Angradi 1994)	-23.6		chironomid	1-2 km d/s of Glen Canyon Dam on CO River
detritus (Angradi 1994)	-28.23	7.56	rainbow trout	1-2 km d/s of Glen Canyon Dam on CO River
detritus (Bunn et al. 1989)	-33.3		chironomid	Alik Lake, Quebec
detritus (Bunn et al. 1989)	-25.27		tipula	Alik Lake, Quebec
detritus (Fry 1991a)	-34.21	4.05	crayfish	Konza Prairie, KS perennial stream
leaf detritus (Fry 1986)	-28.89		crayfish	Green Lake, NY
alder detritus (Gu et al. 1997)	-28.4		chironomid	Smith Lake, AK
alder detritus (Gu et al. 1997)	-24.72	-1.62	mayfly	Smith Lake, AK
birch detritus (Gu et al. 1997)	-29.8		chironomid	Smith Lake, AK
birch detritus (Gu et al. 1997)	-26.12	-5.12	mayfly	Smith Lake, AK
detritus (Harrington et al. 1998)		-0.9	mayfly	Bingo Brook, VT
detritus (Harrington et al. 1998)		-1.2	mayfly	West Branch, VT
detritus (Harrington et al. 1998)		-2.72	mayfly	Bethel-Gilead tributary, VT
detritus (Harrington et al. 1998)		-0.05	mayfly	Peavine tributary, VT
detritus (Harrington et al. 1998)		-4.8	mayfly	First Branch, VT
detritus (Harrington et al. 1998)		-6.99	mayfly	Third Branch, VT
detritus (Rau 1980)	-26		chironomid	Lake Findlay, WA
average	-28.04909	-1.179		
overall average	-27.13759	0.517059		

LTER data only

species (study)	estimated $\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	Shasta source	location
periphyton (Fry 1991a)	-23.72	-0.37	crayfish	Hubbard Brook, Mirror Lake, NH
periphyton (Fry 1991a)	-18.305	-0.24	med smallmouth bas	Hubbard Brook, Mirror Lake, NH
periphyton (Fry 1991a)	-32.41	4.32	crayfish	Konza Prairie, KS perennial stream
average	-24.81167	1.236667		
detritus (Fry 1991a)	-34.21	4.05	crayfish	Konza Prairie, KS perennial stream

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APPENDIX L. Results of stomach analyses of Alabama spotted bass and bluegill

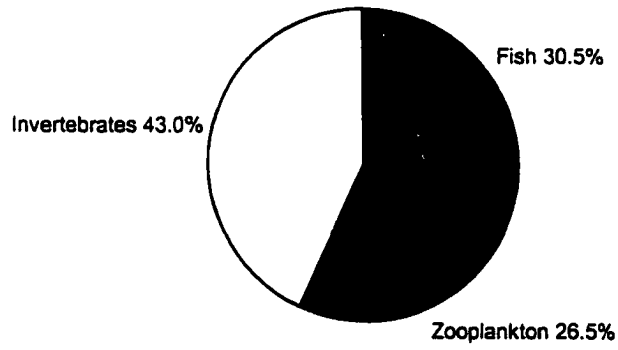

Figure 46. Mean % diet of Alabama spotted bass from Shasta Lake from stomach analysis

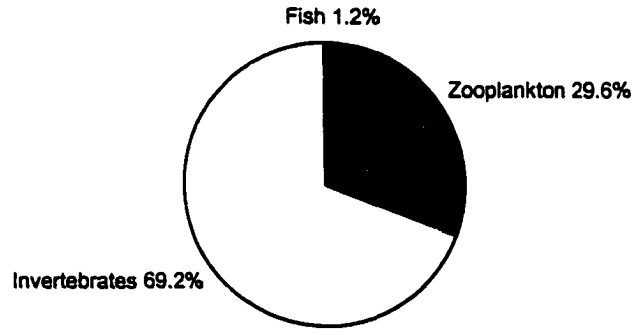

Figure 47. Mean % diet of bluegill from Shasta Lake from stomach analysis

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SHASTA FISH DIET DATA

Processed on ____/____/____ Processed by _____

Notes 1) If more than one entry for a sample number (i.e., more than one measured fish in sample) then diet %'s only get entered on first line, missing on rest
2) If stomach is empty, then ZP = MT, INV = 0, FISH = 0

Misc. codes
CCM = complete backbone length
PAR = partial backbone length
MT = Empty (in %ZP column)
UNK = unknown species

Fish spp codes
ASB = Alabama spotted bass
BCR = black crappie
BLG = bluegill
GSF = green sunfish
GSH = golden shiner
LMB = largemouth bass
RBT = rainbow trout
SMB = smallmouth bass
SOF = Sacramento squawfish
TFS = threadfin shad
WCR = white crappie

Macroinvertebrate codes as needed
AMP = amphipod
CAD = caddis
CHI = chironomid
COL = coleoptera
CRA = crayfish
EPH = mayfly
CUL=culicidae
OLI=oligocaet
HEM = hemiptera
HYM = hymenoptera
ODO = odonata
TRI = stonefly
ORT = orthoptera
TIP = tipulidae
GAS=Gastropod

OBS	Sample ID	Location	Sample date	Date processed	Predator spp	Total length (mm)	diet composition by volume, by eye: % ZP	% INV	% FSH	Fish prey species (see codes)	Backbone length (mm)	Complete or partial?	Percent composition of stomach (redundant with ZP/INV/FSH): Clado-ceran	Cope-pod	HYM	CHI	HEM	ORT	CUL	COL	Other insect code	Other insect count	units or comments
1	0708BAI011	MCC	07/08/1998	10/27/98	ASB	110	20	80	0				20			15	65						
2	0708TUR006	MCC	07/08/1998	10/27/98	ASB	101	25	75	0				25			75							very little in stomach
3	0708TUR017	MCC	07/08/1998	10/27/98	ASB	111	100	0	0				100										
4	0708TUR002	MCC	07/08/1998	10/27/98	ASB	94	98	2	0				98			2							
5	0708TUR009	MCC	07/08/1998	10/27/98	ASB	110	93	3	7	TFS	6	partial	80								UNK	3	
6	0707PIN020	PIT	07/07/1998	10/27/98	ASB	105	10	90	0				10		90								
7	0707PIN002	PIT	07/07/1998	10/27/98	ASB	78	0	100	0						85	15					OLI/UNK	20	
8	0707DGB011	MAI	07/07/1998	10/13/1998	ASB	120	100	0	0				100										only one found, stomach basically empty
9	0708BAC012	MAI	07/08/1998	10/13/1998	ASB	121	50	50	0				50			20			30				
10	0708BLA002	MAI	07/08/1998	10/13/1998	ASB	121	90	10	0				90			10							
11	0708BAC010	MAI	07/08/1998	10/13/1998	ASB	135	10	20	70	UNK	4-6 mm	partial	10		5	15							not much in stomach
12	0708BLA001	MAI	07/08/1998	10/15/1998	ASB	135	100	0	0				100								UNK	20%	
13	0708BLA003	MAI	07/08/1998	10/15/1998	ASB	142	80	20	0				80										
14	0707DGB005	MAI	07/07/1998	10/15/1998	ASB	180	20	80	0				20			30			50				
15	0707DGB007	MAI	07/07/1998	10/15/1998	ASB	180	10	90	0				10						80	80	10		
16	0708BLA006	MAI	07/08/1998	10/15/1998	ASB	199	0	100	0							20				80			
17	0708BLA005	MAI	07/08/1998	10/15/1998	ASB	202	100	0	0														not much in stomach
18	0708BAI027	MCC	07/08/1998	10/15/1998	ASB	130	95	5	0				95			3					OLI	2	oligocaet
19	0708BAI028	MCC	07/08/1998		ASB	130	20	0	80		13	partial	20										many digested fish
20	0708TUR010	MCC	07/08/1998	10/20/98	ASB	138	60	0	40		16	partial	58	1									
21	0708BAI029	MCC	07/08/1998	10/20/98	ASB	138	20	0	80		12	partial	20										
22	0708TUR005	MCC	07/08/1998	10/20/98	ASB	141	60	0	40		14	partial	58	1									
23	0708TUR007	MCC	07/08/1998	10/20/98	ASB	141	60	0	40		16	partial	60										
25	0708TUR020	MCC	07/08/1998	10/20/98	ASB	170	100	0	0				100										very little in stomach
26	0708TUR021	MCC	07/08/1998	10/20/98	ASB	162	0	100	0						100								
27	0708BAI006	MCC	07/08/1998	10/20/98	ASB	188	0	40	60		34	partial									CAD	40%	
28	0708BAI007	MCC	07/08/1998	10/20/98	ASB	187	0	100	0												UNK	100	very little in stomach
29	0707PIN028	PIT	07/07/1998	10/20/98	ASB	122	0	100	0														
30	0707PIN021	PIT	07/07/1998	10/20/98	ASB	165	5	95	0				5			25			70				
31	0707PIN006	PIT	07/07/1998	10/20/98	ASB	189	2	98	0				2		33	5			60				
32	0707PIN024	PIT	07/07/1998	10/20/98	ASB	178	50	50	0				50			40					UNK	10	stomach had rubber grub in it
33	0707ARB010	PIT	07/07/1998	10/20/98	ASB	121	95	5	0				92	3		5							
34	0708ARB003	PIT	07/08/1998	10/20/98	ASB	191	0	100	0												UNK	100	very little in stomach
35	0707PIN035	PIT	07/07/1998	10/20/98	ASB	130	3	48	57		5	partial	2	1		15					UNK	25	
36	0707PIN013	PIT	07/07/1998	10/20/98	ASB	170	0	100	0							15			70	15			
37	0707ARB008	PIT	07/07/1998	11/3/98	ASB	176	0	100	0										50		50		
38	0708GNK006	SAC	07/08/1998	11/3/98	ASB	125	5	95	0				5			1			85				
40	0708SUG008	SAC	07/08/1998	11/3/98	ASB	182	0	0	100	UNK	28	partial											
41	0708GNK012	SAC	07/08/1998	11/3/98	ASB	165	60	40	0				60			40							
42	0708SUG007	SAC	07/08/1998	11/3/98	ASB	125	1	4	95	UNK	30	complete	1								UNK	4	Boat fish sample so far but unidentifiable
43	0708SUG008	SAC	07/08/1998	11/3/98	ASB	162	0	5	95	UNK	45	complete			5								
44	0708GNK013	SAC	07/08/1998	11/3/98	ASB	167	15	80	5	UNK	6	partial	15		30	10				40			
45	0708SLA007	MAI	07/08/1998	11/3/98	ASB	220	5	95	0				5							35	CRA	60	
46	0707DGB009	MAI	07/07/1998	11/3/98	ASB	205	100	0	0				100										very little in stomach
47	0708BAC013	MAI	07/08/1998	11/3/98	ASB	262	0	100	0							2			98				large good sample
48	0708BAC015	MAI	07/08/1998	11/3/98	ASB	230	0	100	0						90	10							
49	0708BAC014	MAI	07/08/1998	11/3/98	ASB	280	0	100	0											100			very little in stomach
50	0708BAC016	MAI	07/08/1998	11/5/98	ASB	287	0	0	100	yellow perch	106	complete											
51	0708TUR027	MCC	07/08/1998	11/5/98	ASB	215	0	5	100	TFS	32	partial											
52	0708BAI005	MCC	07/08/1998	11/5/98	ASB	283	0	100	0												UNK	100	
53	0708BAI008	MCC	07/08/1998	11/5/98	ASB	217	10	90	0				10		80	30							rubber grub used for fishing
54	0707ARB005	PIT	07/07/1998	11/10/98	ASB	234	0	100	0						80		10						very little in stomach
55	0707ARB014	PIT	07/07/1998	11/10/98	ASB	242	0	5	95		12	partial				5							
56	0708GNK011	SAC	07/08/1998	11/10/98	ASB	204	100	0	0				100								CRA	7	
57	0708SUG012	SAC	07/08/1998	11/10/98	ASB	243	0	10	90		18	complete			3								
58	0708SUG005	SAC	07/08/1998	11/10/98	ASB	287	2	0	98		22	partial	2										
59	0708SUG014	SAC	07/08/1998	11/10/98	ASB	255	0	100	0						10		10	80			CRA	40	
60	0708GNK009	SAC	07/08/1998	11/10/98	ASB	277	30	70	0				30		30								
61	0708GNK008	SAC	07/08/1998	11/10/98	ASB	287	4	96	0				4				6	80					

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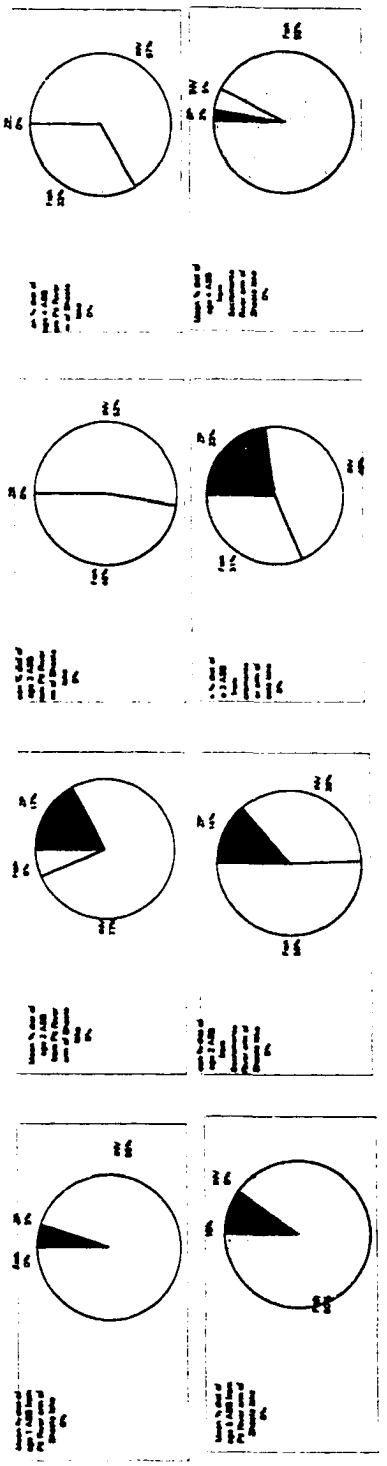

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SHASTA FISH DIET DATA Processed on ____/____/____ Processed by _____.

Notes 1) If more than one entry for a sample number (i.e., more than one measured fish in sample) then diet %'s only get entered on first line, weighing on next
2) If stomach is empty, then ZP = MT, INV = 0, FISH = 0

Misc. codes
COM = complete backbone length
PAR = partial backbone length
MT = Empty (in %ZP column)
UNK = unknown species

Fish spp codes
ASB = Alabama spotted bass
BCR = black crappie
BLG = bluegill
GSF = green sunfish
GSH = golden shiner
LMB = largemouth bass
RBT = rainbow trout
SMB = smallmouth bass
SQF = Sacramento squawfish
TFS = threadfin shad
WCR = white crappie

Macroinvertebrate codes as needed
AMP = amphipod
CAD = caddis
CHI = chironomid
COL = coleoptera
CRA = crayfish
EPH = mayfly
CUL=culicidae
OLI=oligocaet
HEM = hemiptera
HYM = hymenoptera
ODO = odonata
TRI = stonefly
ORT = orthoptera
TIP = tipulidae
GAS=Gastropod

OBS	Sample ID	Location	Sample date	Date processed	Predator spp	Total length (mm)	diet composition by volume, by eye: % ZP	% INV	% FISH	Fish prey species code	Backbone length (mm)	Complete or partial?	Percent composition of stomach (redundant with ZP/INV/FISH): Clado-cera	Cope-pod	HYM	CHI	HEM	ORT	CUL	COL	Other insect code	Other insect count	units or comments
75	0708BAC001	MAI	07/08/1998	10/13/1998	BLG	135	0	100	0						5	75					CAD	20%	
76	0708BAC002	MAI	07/08/1998	01/02/1999	BLG	67	5	95	0				5			95							
77	0708BAI030	MCC	07/08/1998	01/02/1999	BLG	111	30	70	0				30		40	30							
78	0708TUR018	MCC	07/08/1998	01/02/1999	BLG	71	100	0	0				86	14									1 this 1% is zoo-plankton (very small)
79	0708TUR015	MCC	07/08/1998	01/02/1999	BLG	102	30	70	0				30			60					EPH	10	
80	0708BAI023	MCC	07/08/1998	01/02/1999	BLG	101	30	50	20	TFS	5	partial	30			20				10			
81	0707ARB017	PIT	07/07/1998	01/04/1999	BLG	91	2	98	0				2		25	8	25			40			
82	0707PIN022	PIT	07/07/1998	01/04/1999	BLG	78	0	100	0						50	20	20		10				
83	0707ARB019	PIT	07/07/1998	01/04/1999	BLG	95	20	80	0				20			45			25	10			
84	0707PIN037	PIT	07/07/1998	01/04/1999	BLG	89	0	100	0						20	30	10		10	30			
85	0707PIN014	PIT	07/07/1998	01/04/1999	BLG	84	0	100	0							15	50				CAD	35	
86	0707PIN015	PIT	07/07/1998	01/04/1999	BLG	68	60	40	0				60		5	10	15		10				
87	0707PIN016	PIT	07/07/1998	01/04/1999	BLG	128	0	100	0								5				AMP/GAS	20/75	
88	0707ARB022	PIT	07/07/1998	01/04/1999	BLG	76	96	4	0				96			2			2				
89	0707ARB020	PIT	07/07/1998	01/04/1999	BLG	101	95	5	0				95			2	3						
90	0707PIN040	PIT	07/07/1998	01/04/1999	BLG	136	5	95	0				5		10	75	5			5			

ZP stats

Column1	
Mean	29 5625
Standard Error	9 368803
Median	12 5
Mode	0
Standard Devi	37 42721
Sample Varian	1400 796
Kurtosis	-0 23443
Skewness	1 118722
Range	100
Minimum	0
Maximum	100
Sum	473
Count	16
Confidence Le	19 94366

INV stats

Column1	
Mean	69 1875
Standard E	9 443789
Median	87 5
Mode	100
Standard D	37 77516
Sample Var	1426 963
Kurtosis	-0 53282
Skewness	-0 88763
Range	100
Minimum	0
Maximum	100
Sum	1107
Count	16
Confidence	20 12887

FISH stats

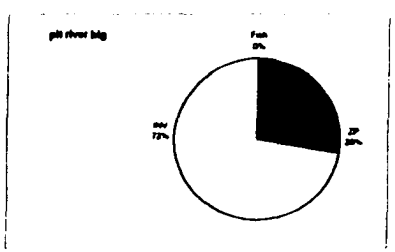
Column1	
Mean	1 25
Standard	1 25
Median	0
Mode	0
Standard	5
Sample V	25
Kurtosis	16
Skewness	4
Range	20
Minimum	0
Maximum	20
Sum	20
Count	16
Confidenc	2 664314

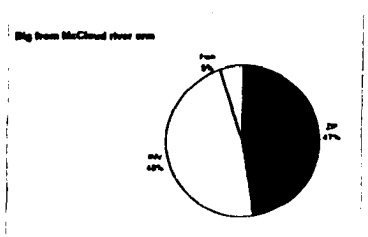
Mean % diet of BLG from Shasta lake: % ZP 29 5, % INV 69 2, %Fish 1 2; n= 16, mean length 95 625

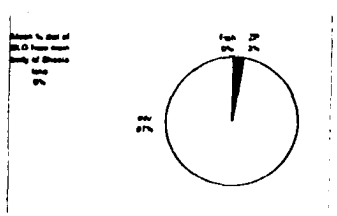
Mean % diet of BLG from Pit River arm of Shasta lake: ZP 27 8, INV 72 2, Fish 0; n= 16, mean length 84 3

Mean % diet of BLG from McCloud River arm of Shasta lake: ZP 47 5, INV 47 5, Fish 5; n= 4, mean length 98 25

Mean % diet of BLG from main body of Shasta lake: ZP 2 5, INV 97 5, Fish 0; n= 2, mean length 101







APPENDIX M. Literature review of diets of food web species

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Bluegill (all year classes)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	
Med bass(2/3)	
RBT2/3	
TFS	Emig (1966a); Goodson (1965)
age1bass	eat small fish -- Emig (1966a); Moyle (1976)
CRA	Emig (1966a); Moyle (1976); Turner (1966a)
GSF2/3	
SQF2	
BLG	
ZOO	DeVries et al. (1991); Emig (1966a); Goodson (1965); Moyle (1976); Siefert (1972); Stuber et al. (1982a); Turner (1966a); Wang (1986)
Terr insects	Emig (1966a); Goodson (1965); Moyle (1976); Siefert (1972); Stuber et al. (1982a); Turner (1966a); Wang (1986)
Aq insects	Emig (1966a); Goodson (1965); Moyle (1976); Siefert (1972); Stuber et al. (1982a); Turner (1966a); Wang (1986)
Phytopl	Emig (1966a); Moyle (1976); Siefert (1972)
Periphyton	Emig (1966a); Moyle (1976)
Macrophytes	Emig (1966a); Moyle (1976); Stuber et al. (1982a)
Detritus	
Terr Plants	

Fishes:	Emig (1966a): eat small fish, threadfin shad, fish eggs
	Goodson (1965): eat fish eggs, threadfin shad
	Moyle (1976): eat small fish, threadfin shad, fish eggs, crayfish
	Turner (1966a): ate unidentified fishes

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Sacramento squawfish (YC2: 75-122 mm)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	
Med bass(2/3)	
RBT2/3	
TFS	
age1bass	
CRA	
GSF2/3	
SQF2	
BLG	
ZOO	
Terr insects	Burns (1966a); Taft and Murphy (1950); Wang (1986)
Aq insects	Burns (1966a); Moyle (1976); Taft and Murphy (1950); Wang (1986)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

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Green sunfish (YC2/3 - 83-125 mm)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	
Med bass(2/3)	
RBT2/3	
TFS	
age1bass	
CRA	
GSF2/3	
SQF2	McKechnie and Tharratt (1966); Moyle (1976); Stuber et al. (1982b)
BLG	McKechnie and Tharratt (1966); Moyle (1976); Stuber et al. (1982b)
ZOO	
Terr insects	McKechnie and Tharratt (1966); Moyle (1976); Stuber et al. (1982b)
Aq insects	McKechnie and Tharratt (1966); Moyle (1976); Stuber et al. (1982b)
Phytopl	
Periphyton	McKechnie and Tharratt (1966)
Macrophytes	McKechnie and Tharratt (1966)
Detritus	
Terr Plants	

Fish: McGinnis (1984): adults partially piscivorous
McKechnie and Tharratt (1966): prefer larger animal foods such as fish; fish < 10% of diet
Moyle (1976): > 8 cm SL, eat small sunfish (including own young)
Stuber et al. (1982b): juveniles and adults feed on fish

Note: although McKechnie and Tharratt (1966), Moyle (1976), Stuber et al. (1982b) indicate that GSF eat their own young, it is assumed that they would not eat this size of GSF

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Crayfish

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	
Med bass(2/3)	
RBT2/3	
TFS	
age1bass	
CRA	
GSF2/3	
SQF2	
BLG	
ZOO	
Terr insects	Hobbs (1991)
Aq insects	Hobbs (1991)
Phytopl	Hobbs (1991)
Periphyton	Hobbs (1991)
Macrophytes	Hobbs (1991)
Detritus	Hobbs (1991)
Terr Plants	

Fish: Hobbs (1991): *predation by crayfishes on both invertebrates and vertebrates is well documented*

Hobbs, III, H.H. 1991. Decapoda. Pp. 823-858 in Thorp, James H. and Alan P. Covich (eds.). Ecology and Classification of North American Freshwater Invertebrates. Academic Press, Inc. San Diego. 911 pp.

Year class 1 bass (ASB, LMB, SMB: 6-113 mm)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	
Med bass(2/3)	
RBT2/3	
TFS	Applegate and Mullan (1967), Emig (1966b), McKechnie (1966), McMahon et al (1984), Miller and Kramer (1971), Moyle (1976) Vogele (1975), Wang (1986)
age1bass	Applegate and Mullan (1967), Heidinger (1975), Miller and Kramer (1971), Moyle (1976), Vogele (1975)
CRA	Edwards et al (1983), McMahon et al (1984), Moyle (1976)
GSF2/3	
BOF2	
BLG	Edwards et al (1983), Emig (1966c), Heidinger (1975), McKechnie (1966), McMahon et al (1984), Miller and Kramer (1971) Moyle (1976), Vogele (1975)
ZOO	Applegate and Mullan (1967), Coble (1975), Curtis (1949), Edwards et al (1983), Emig (1966b), Emig (1966c), Heidinger (1975), Hirst and DeVries (1994), McGinnis (1984), McKechnie (1966), McMahon et al (1984), Miller and Kramer (1971), Moyle (1976), Vogele (1975), Wang (1986)
Terr insects	Coble (1975), Curtis (1949), Edwards et al (1983), Emig (1966b), Emig (1966c), Heidinger (1975), Hirst and DeVries (1994), McKechnie (1966), McMahon et al (1984), Moyle (1976), Vogele (1975), Wang (1986)
Aq insects	Applegate & Mullan (1967), Coble (1975), Curtis (1949), Edwards et al (1983), Emig (1966b), Emig (1966c), Heidinger (1975), Hirst & DeVries (1994), McGinnis (1984), McKechnie (1966), McMahon et al (1984), Miller & Kramer (1971), Moyle (1976), Turner (1966a), Vogele (1975), Wang (1986)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish	
	Applegate and Mullan (1967) LMB at > 40 mm TL, larval shad and unidentified fish fry dominant food
	Edwards et al (1983) SMB Juveniles eat larger insects, crayfish, fish
	Emig (1966b) LMB < 8.0 in, mostly TFS and cladocerans
	Emig (1966c) SMB 1-2in most of diet is insects and small fish
	Heidinger (1975) LMB young bass eat fingerling fishes, including each other - only fingerlings we have are BLG and BASS
	McKechnie (1966) ASB by end of first summer, fish 50% of diet
	McMahon et al (1984) ASB > 75 mm TL, feed on insects, fish (centrarchids, clupeids) and crayfish
	Miller and Kramer (1971) LMB 20-29 mm 3.7% occ, 30-39 mm 21.62% occ, 40-49 mm 8.98% occ, 50-59 mm 9.09% occ 60-69 mm 3.09% occ, 70-79 mm 14.63% occ, 80-89 mm 14.63% occ, 90-99 mm 4.55% occ >100 mm 40% occ
	Moyle (1976) ASB > 50 mm TL, fed heavily on fish commonly feed on own young and other bass species
	Moyle (1976) LMB 50-60 mm SL feed largely on fish fry, including own species, 100-125 mm SL usually subsist primarily on fish, mostly TFS , GSH, BLG
	Moyle (1976) SMB at 5cm TL, start feeding heavily on fish
	Vogele (1975) ASB all sizes except smallest eat fish
	Wang (1986) ASB larger juveniles found small TFS in stomachs

Note: although literature review indicates YC1 bass probably eat green sunfish and Sacramento squawfish, the specimens we collected were probably too large to be eaten by the YC1 bass

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Threadfin shad (all sizes)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	
Med bass(2/3)	
RBT2/3	
TFS	
age1bass	
CRA	
GSF2/3	
SQF2	
BLG	
ZOO	Baker and Schmitz (1971), Burns (1966b), Davis and Foltz (1991), DeVries et al (1991), Hirst and DeVries (1994), Kampa (1984), Moyle (1976), Turner (1966b), Van Den Avyle and Pettering (1988), Von Geldern and Mitchell (1975)
Terr insects	
Aq insects	Kampa (1984), Van Den Avyle and Pettering (1988), Von Geldern and Mitchell (1975)
Phytopl	Baker and Schmitz (1971), Burns (1966b), Davis and Foltz (1991), Moyle (1976), Turner (1966b)
Periphyton	Turner (1966b)
Macrophytes	Turner (1966b)
Detritus	Baker and Schmitz (1971), Burns (1966b), Moyle (1976), Turner (1966b)
Terr Plants	

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Rainbow trout (YC2/3: 290-410 mm)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	
Med bass(2/3)	
RBT2/3	
TFS	McAfee (1966); Moyle (1976); Raleigh et al (1984)
age1bass	Moyle (1976); Raleigh et al. (1984)
CRA	
GSF2/3	Moyle (1976); Raleigh et al. (1984)
SQF2	Moyle (1976); Raleigh et al. (1984)
BLG	Moyle (1976); Raleigh et al. (1984)
ZOO	Lynott et al. (1995); McAfee (1966); Moyle (1976)
Terr insects	Barnhart (1986); Lynott et al. (1995); McAfee (1966); Moyle (1976)
Aq insects	Barnhart (1986); Lynott et al. (1995); McAfee (1966); Moyle (1976); Raleigh et al. (1984)
Phytopl	McAfee (1966)
Periphyton	McAfee (1966)
Macrophytes	McAfee (1966)
Detritus	
Terr Plants	

Fish: Lynott et al. (1995): 201-330 mm TL: misc prey in diet in low amts; 331-460 mm TL: misc prey in low amts
McAfee (1966): being eating shad at 11-12 inches; fish one of most significant foods; adults feed heavily on shad in Shasta Lake
Moyle (1976): lake RBT eat more fish than streamdwelling RBT
Raleigh et al (1984): 30 cm when begin preying on other fish species

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Medium bass (ASB, LMB: 114-290 mm)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	McKechnie (1966); Moyle (1976); Vogele (1975)
Med bass(2/3)	Emig (1966b); Goodson (1965); Heidinger (1975); McKechnie (1966); Moyle (1976); Vogele (1975)
RBT2/3	McKechnie (1966); Moyle (1976); Vogele (1975)
TFS	Curtis (1949); Emig (1966b); Goodson (1965); Heidinger (1975); McKechnie (1966); Moyle (1976); Turner (1966a); Vogele (1975); Wanjala et al. (1986)
age1bass	Curtis (1949); Emig (1966b); Goodson (1965); Heidinger (1975); McKechnie (1966); Moyle (1976); Turner (1966a); Vogele (1975)
CRA	Curtis (1949); Emig (1966b); Goodson (1965); Heidinger (1975); McKechnie (1966); McMahon et al. (1984); Turner (1966a); Vogele (1975)
GSF2/3	Curtis (1949); Emig (1966b); Goodson (1965); Heidinger (1975); McKechnie (1966); McMahon et al. (1984); Moyle (1976); Turner (1966a); Vogele (1975)
SQF2	Curtis (1949); Emig (1966b); Goodson (1965); Heidinger (1975); McKechnie (1966); Moyle (1976); Turner (1966a); Vogele (1975)
BLG	Curtis (1949); Emig (1966b); Goodson (1965); Heidinger (1975); McKechnie (1966); McMahon et al. (1984); Moyle (1976); Turner (1966a); Vogele (1975); Wanjala et al. (1986)
ZOO	
Terr insects	Curtis (1949); Emig (1966b); McKechnie (1966); McMahon et al. (1984); Turner (1966a); Vogele (1975)
Aq insects	Curtis (1949); Emig (1966b); Goodson (1965); McKechnie (1966); McMahon et al. (1984); Vogele (1975); Wanjala et al. (1986)
Phytopl	McKechnie (1966)
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish:

Curtis (1949): LMB as young fish grow older, eat small fish; adults eat larger food such as big fish and crayfish
Emig (1966b): LMB: 6.0-9.9 in: mostly TFS and GSH; > 9.9 in, almost exclusively shad; also eat GSF, centrarchids, LMB
Goodson (1965): LMB: 6-9.9 in: TFS, unidentified fish; > 10 in: TFS, unidentified fish, LMB, BLG
Heidinger (1975): LMB: adults eat fishes and crayfishes
McKechnie (1966): ASB: BLG, crayfish, unid'd fish remains, adult insects, white crappie, channel catfish, algae, black bass in order of pref
McMahon et al. (1984): ASB: > 75 mm TL, feed on insects, fish (centrarchids, clupeids) and crayfish
Moyle (1976): ASB: > 50 mm TL; fed heavily on fish; commonly feed on own young and other bass species
Moyle (1976): LMB: >100-125 mm SL: usually subsist primarily on fish; mostly TFS , GSH, BLG
Turner (1966a): LMB: 16-49 cm: BLG, TFS, unidentified fishes
Vogele (1975): ASB: all sizes except smallest eat fish
Wanjala et al. (1986): LMB: 351-305 mm TL: 3% BLG; 60.4% shad; 27.3% aquatic insects;

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Medium SMB (114-290 mm)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	
Med bass(2/3)	
RBT2/3	
TFS	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
age1bass	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
CRA	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
GSF2/3	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
SQF2	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
BLG	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
ZOO	
Terr insects	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); McGinnis (1984); Moyle (1976)
Aq insects	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); McGinnis (1984); Moyle (1976)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish: Coble (1975): adults eat mainly fish, crayfish, insects
Curtis (1949): as young fish grow older, eat small fish; adults eat larger food such as big fish and crayfish
Edwards et al. (1983): juveniles eat larger insects, crayfish, fish; adults eat primarily fish and crayfish
Emig (1966c): adults prefer crayfish, fish and insects; may prey on their own young and young of other spp.; GSF eaten to limited extent; eat sunfish
Moyle (1976): frequently cannibalistic

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Big bass (ASB: > 290 mm)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	
Med SMB(2/3)	McKechnie (1966); Moyle (1976); Vogele (1975)
Med bass(2/3)	McKechnie (1966); Moyle (1976); Vogele (1975)
RBT2/3	McKechnie (1966); Moyle (1976); Vogele (1975)
TFS	McKechnie (1966); Moyle (1976); Vogele (1975)
age1bass	McKechnie (1966); Moyle (1976); Vogele (1975)
CRA	McKechnie (1966); Vogele (1975)
GSF2/3	McKechnie (1966); Moyle (1976); Vogele (1975)
SQF2	McKechnie (1966); Moyle (1976); Vogele (1975)
BLG	McKechnie (1966); Moyle (1976); Vogele (1975)
ZOO	Vogele (1975)
Terr insects	McKechnie (1966); Vogele (1975)
Aq insects	McKechnie (1966); Vogele (1975)
Phytopl	McKechnie (1966)
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish: McKechnie (1966): BLG, crayfish, unid'd fish remains, adult insects, white crappie, channel catfish, algae, black bass in order of pref
Moyle (1976): > 50 mm TL; fed heavily on fish; commonly feed on own young and other bass species
Vogele (1975): all sizes except smallest eat fish

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Brown trout (adult - 510 mm (20 in))

Prey	
CHS2/3	Moyle (1976); Staley (1966)
BRT5+	
Big bass(4+)	Moyle (1976); Staley (1966)
Med SMB(2/3)	Moyle (1976); Staley (1966)
Med bass(2/3)	Moyle (1976); Staley (1966)
RBT2/3	Moyle (1976); Staley (1966)
TFS	Moyle (1976); Staley (1966)
age1bass	Moyle (1976); Staley (1966)
CRA	
GSF2/3	Moyle (1976); Staley (1966)
SQF2	Moyle (1976); Staley (1966)
BLG	Moyle (1976); Staley (1966)
ZOO	Staley (1966)
Terr insects	Staley (1966)
Aq insects	Staley (1966)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fishes:	Moyle (1976): adults feed mostly on fish
	Staley (1966): larger individuals feed more extensively on fish

Chinook salmon (YC2/3 - 320-538 mm)

Prey	
CHS2/3	
BRT5+	
Big bass(4+)	Allen and Hassler (1986); Moyle (1976)
Med SMB(2/3)	Allen and Hassler (1986); Moyle (1976)
Med bass(2/3)	Allen and Hassler (1986); Moyle (1976)
RBT2/3	Allen and Hassler (1986); Moyle (1976)
TFS	Allen and Hassler (1986); Moyle (1976)
age1bass	Allen and Hassler (1986); Moyle (1976)
CRA	
GSF2/3	Allen and Hassler (1986); Moyle (1976)
SQF2	Allen and Hassler (1986); Moyle (1976)
BLG	Allen and Hassler (1986); Moyle (1976)
ZOO	Allen and Hassler (1986); Moyle (1976)
Terr insects	Moyle (1976)
Aq insects	
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fishes:	Allen and Hassler (1986)
	Moyle (1976): adults feed mostly on fish

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Alabama spotted bass

Year class 1 (0-113 mm)

PREY	
Chinook	
Brown	
Big bass	
Med SMbass	
Med bass	
RBT	
Shad	McKechnie (1966); McMahon et al. (1984); Moyle (1976); Vogele (1975); Wang (1986)
age1bass	Moyle (1976); Vogele (1975)
crayfish	McMahon et al. (1984)
Greensun	McKechnie (1966); McMahon et al. (1984); Moyle (1976); Vogele (1975)
Squawfish	McKechnie (1966); Vogele (1975)
bluegill	McKechnie (1966); McMahon et al. (1984); Moyle (1976); Vogele (1975)
ZP	Hirst and DeVries (1994); McKechnie (1966); McMahon et al. (1984); Vogele (1975); Wang (1986)
Terr insects	Hirst and DeVries (1994); McKechnie (1966); McMahon et al. (1984); Moyle (1976); Vogele (1975); Wang (1986)
Aq insects	Hirst and DeVries (1994); McKechnie (1966); McMahon et al. (1984); Moyle (1976); Vogele (1975); Wang (1986)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish:	
	McKechnie (1966): by end of first summer, fish 50% of diet
	McMahon et al. (1984): > 75 mm TL, feed on insects, fish (centrarchids, clupeids) and crayfish
	Moyle (1976): > 50 mm TL; fed heavily on fish; commonly feed on own young and other bass species
	Vogele (1975): all sizes except smallest eat fish
	Wang (1986): larger juveniles: found small TFS in stomachs

Year class 2/3 (114-290 mm)

PREY	
Chinook	
Brown	
Big bass	
Med SMbass	McKechnie (1966); Moyle (1976); Vogele (1975)
Med bass	McKechnie (1966); Moyle (1976); Vogele (1975)
RBT	McKechnie (1966); Moyle (1976); Vogele (1975)
Shad	McKechnie (1966); Moyle (1976); Vogele (1975)
age1bass	McKechnie (1966); Moyle (1976); Vogele (1975)
crayfish	McKechnie (1966); McMahon et al. (1984); Vogele (1975)
Greensun	McKechnie (1966); McMahon et al. (1984); Moyle (1976); Vogele (1975)
Squawfish	McKechnie (1966); Moyle (1976); Vogele (1975)
bluegill	McKechnie (1966); McMahon et al. (1984); Moyle (1976); Vogele (1975)
ZP	
Terr insects	McKechnie (1966); McMahon et al (1984); Vogele (1975)
Aq insects	McKechnie (1966); McMahon et al (1984); Vogele (1975)
Phytopl	McKechnie (1966)
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish:	
	McKechnie (1966): BLG, crayfish, unid'd fish remains, adult insects, white crappie, channel catfish, algae, black bass in order of pref
	McMahon et al. (1984): > 75 mm TL, feed on insects, fish (centrarchids, clupeids) and crayfish
	Moyle (1976): > 50 mm TL; fed heavily on fish; commonly feed on own young and other bass species
	Vogele (1975): all sizes except smallest eat fish

Year class 4+ (>290 mm)

PREY	
Chinook	
Brown	
Big bass	
Med SMbass	McKechnie (1966); Moyle (1976); Vogele (1975)
Med bass	McKechnie (1966); Moyle (1976); Vogele (1975)
RBT	McKechnie (1966); Moyle (1976); Vogele (1975)
Shad	McKechnie (1966); Moyle (1976); Vogele (1975)
age1bass	McKechnie (1966); Moyle (1976); Vogele (1975)
crayfish	McKechnie (1966); Vogele (1975)
Greensun	McKechnie (1966); Moyle (1976); Vogele (1975)
Squawfish	McKechnie (1966); Moyle (1976); Vogele (1975)
bluegill	McKechnie (1966); Moyle (1976); Vogele (1975)
ZP	Vogele (1975)
Terr insects	McKechnie (1966); Vogele (1975)
Aq insects	McKechnie (1966); Vogele (1975)
Phytopl	McKechnie (1966)
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish:	
	McKechnie (1966): BLG, crayfish, unid'd fish remains, adult insects, white crappie, channel catfish, algae, black bass in order of pref
	Moyle (1976): > 50 mm TL; fed heavily on fish; commonly feed on own young and other bass species
	Vogele (1975): all sizes except smallest eat fish

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largemouth bass

Year class 1 (0-113 mm)

PREY	
Chinook	
Brown	
Big bass	
Med Stlbass	
Med bass	
RBT	
Shad	Applegate and Mullan (1967); Emig (1966b); Miller and Kramer (1971); Mayto (1978)
age1bass	Applegate and Mullan (1967);Heidinger (1975); Miller and Kramer (1971); Mayto (1978)
crayfish	Mayto (1978)
Greensun	
Squawfish	
Mosqfi	Mayto (1978)
ZP	Heidinger (1975); Miller and Kramer (1971); Mayto (1978)
Terr macrolnv	Applegate and Mullan (1967); Curtis (1949); Emig (1966b); Heidinger (1975); Hirst and DeVries (1994); McGinnis (1984); Miller and Kramer (1971); Mayto (1978)
Aq macrolnv	Emig (1966b); Curtis (1949); Heidinger (1975); Hirst and DeVries (1994)
Phytopl	Applegate and Mullan (1967); Curtis (1949); Emig (1966b); Heidinger (1975); Hirst and DeVries (1994); Miller and Kramer (1971); Turner (1966a)
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish: Applegate and Mullan (1967). at > 40 mm TL, larval shad and unidentified fish fry dominant food
Emig (1966b): < 6.0 m, mostly TFS and cladocerans
Heidinger (1975): young bass eat fingerling fishes, including each other - only fingerlings we have are BLG and BASS
Miller and Kramer (1971): 20-29 mm: 3.7% occ; 30-39 mm: 21.62% occ; 40-49 mm: 6.88% occ; 50-59 mm: 5.88% occ; 60-69 mm: 3.08% occ; 70-79 mm: 14.63% occ; 80-89 mm: 14.63% occ; 90-99 mm: 4.55% occ; >100 mm 40% occ
Mayto (1978): 50-60 mm SL: feed largely on fish fry, including own species; 100-125 mm SL: usually subsist primarily on fish; mostly TFS, GSH, BLG

Year class 2/3 (114-299 mm)

PREY	
Chinook	
Brown	
Big bass	
Med Stlbass	
Med bass	Emig (1966b); Goodson (1966); Heidinger (1975)
RBT	
Shad	Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a); Wanjala et al. (1986)
age1bass	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a)
crayfish	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Turner (1966a)
Greensun	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a)
Squawfish	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a)
Mosqfi	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a); Wanjala et al. (1986)
ZP	
Terr macrolnv	Curtis (1949); Emig (1966b); Turner (1966a)
Aq macrolnv	Curtis (1949); Emig (1966b); Goodson (1966); Turner (1966a); Wanjala et al. (1986)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish: Curtis (1949): as young fish grow older, eat small fish; adults eat larger food such as big fish and crayfish
Emig (1966b): 6.0-6.9 in: mostly TFS and GSH; > 6.9 in, almost exclusively shad; also eat GSF, centrarchids, LMB
Goodson (1966): 6-6.9 in: TFS, unidentified fish; > 10 in: TFS, unidentified fish, LMB, BLG
Heidinger (1975): adults eat fishes and crayfishes
Mayto (1978): >100-125 mm SL: usually subsist primarily on fish; mostly TFS, GSH, BLG
Turner (1966a): 16-40 cm: BLG, TFS, unidentified fishes
Wanjala et al. (1986): 261-305 mm TL: 3% BLG; 60.4% shad; 27.3% aquatic macrolnv;

Year class 4+ (>300 mm)

PREY	
Chinook	
Brown	
Big bass	
Med Stlbass	Curtis (1949)
Med bass	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975)
RBT	Curtis (1949)
Shad	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a); Von Geldern and Mitchell (1975); Wanjala et al. (1986)
age1bass	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a)
crayfish	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Turner (1966a)
Greensun	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a); Von Geldern and Mitchell (1975)
Squawfish	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a)
Mosqfi	Curtis (1949); Emig (1966b); Goodson (1966); Heidinger (1975); Mayto (1978); Turner (1966a); Von Geldern and Mitchell (1975); Wanjala et al. (1986)
ZP	
Terr macrolnv	Emig (1966b); Turner (1966a)
Aq macrolnv	Emig (1966b); Goodson (1966); Turner (1966a); Wanjala et al. (1986)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish: Curtis (1949): as young fish grow older, eat small fish; adults eat larger food such as big fish and crayfish
Emig (1966b): > 6.9 in, almost exclusively shad; also eat GSF, centrarchids, LMB
Goodson (1966): >10 in: TFS, unidentified fish, LMB, BLG
Heidinger (1975): adults eat fishes and crayfishes
Mayto (1978): >100-125 mm SL: usually subsist primarily on fish; mostly TFS, GSH, BLG
Turner (1966a): 16-40 cm: TFS, BLG
Von Geldern and Mitchell (1975): shad (56.3%); centrarchids (BLG and GSF) (4.4%)
Wanjala et al. (1986): 261-305 mm TL: 3% BLG; 60.4% shad; 27.3% aquatic macrolnv; 306-500 mm: 16% BLG, 57.7% shad; 14.6% aquatic macrolnv

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Smallmouth bass

Year class 1 (0-113 mm)

PREY	
Chinook	
Brown	
Big bass	
Med SMbass	
Med bass	
RBT	
Shad	Moyle (1976)
age1bass	
crayfish	Edwards et al. (1983)
Greensun	Moyle (1976)
Squawfish	Moyle (1976)
bluegill	Edwards et al. (1983); Emig (1966c); Moyle (1976)
ZP	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); McGinnis (1984); Moyle (1976)
Terr insects	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); McGinnis (1984); Moyle (1976)
Aq insects	Coble (1975), Curtis (1949); Edwards et al. (1983); Emig (1966c); McGinnis (1984); Moyle (1976)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish: Edwards et al. (1983): Juveniles eat larger insects, crayfish, fish
Emig (1966c): 1-2in most of diet is insects and small fish
Moyle (1976): at 5cm TL, start feeding heavily on fish

Year class 2/3 (114-290 mm)

PREY	
Chinook	
Brown	
Big bass	
Med SMbass	
Med bass	
RBT	
Shad	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
age1bass	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
crayfish	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
Greensun	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
Squawfish	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
bluegill	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
ZP	
Terr insects	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); McGinnis (1984); Moyle (1976)
Aq insects	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); McGinnis (1984); Moyle (1976)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish: Coble (1975): adults eat mainly fish, crayfish, insects
Curtis (1949): as young fish grow older, eat small fish; adults eat larger food such as big fish and crayfish
Edwards et al. (1983): juveniles eat larger insects, crayfish, fish, adults eat primarily fish and crayfish
Emig (1966c): adults prefer crayfish, fish and insects; may prey on their own young and young of other spp., GSF eaten to limited extent; eat sunfish
Moyle (1976): frequently cannibalistic

Year class 4+ (>290 mm)

PREY	
Chinook	
Brown	
Big bass	
Med SMbass	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
Med bass	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
RBT	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
Shad	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
age1bass	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
crayfish	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
Greensun	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
Squawfish	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
bluegill	Coble (1975); Curtis (1949); Edwards et al. (1983); Emig (1966c); Moyle (1976)
ZP	
Terr insects	Coble (1975); Emig (1966c); McGinnis (1984); Moyle (1976)
Aq insects	Coble (1975); Emig (1966c); McGinnis (1984); Moyle (1976)
Phytopl	
Periphyton	
Macrophytes	
Detritus	
Terr Plants	

Fish: Coble (1975): adults eat mainly fish, crayfish, insects
Curtis (1949): as young fish grow older, eat small fish; adults eat larger food such as big fish and crayfish; SMB eat more crayfish than LMB
Edwards et al. (1983): adults eat primarily fish and crayfish
Emig (1966c): adults prefer crayfish, fish and insects; may prey on their own young and young of other spp.; GSF eaten to limited extent; eat sunfish
Moyle (1976): frequently cannibalistic

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APPENDIX N. Solver structure and results for estimation of diet proportions

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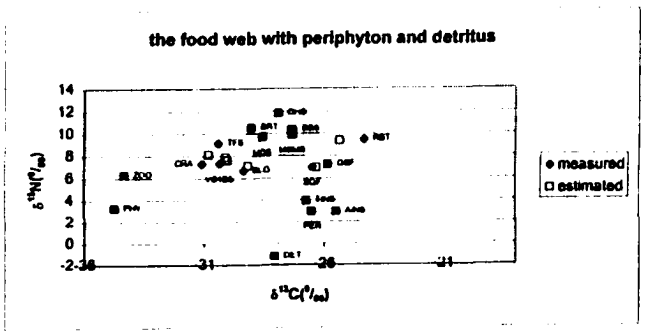
species	measured $\delta^{13}C$(‰)	measured $\delta^{15}N$(‰)	estimated $\delta^{13}C$(‰)	estimated $\delta^{15}N$(‰)	squared error $\delta^{13}C$(‰)	squared error $\delta^{15}N$(‰)	sum
chinook salmon (CHS 320-638 mm)	-27 91	11.85	-27.907	11 8638	9E-06	0 00019	0 000199
brown trout (BRT 510 mm)	-29 02	10.54	-29 0225	10 5239	6 25E-06	0 000259	0 000265
big bass - ASB (BBS > 290 mm)	-27 32	10.37	-27.3498	10 368	0 000888	4E-06	0 000892
medium smallmouth bass (MSMB 114-290 mm	-27 34	9.92	-27.3236	9 9124	0 000269	5 78E-05	0 000327
medium bass - ASB+LMB(MBS 114-290 mm)	-28 58	9.71	-28.5582	9 7043	0 000475	3 25E-05	0 000508
rainbow trout (RBT 290-410 mm)	-24 33	9 46	-25 36	9 376	1 0609	0 007056	1 067956
threadfin shad (TFS)	-30 41	9.18	-30.813	8 131	0 162409	1 100401	1 26281
YC1 bass (YC1BS 0-113 mm)	-30 35	7.34	-30 1074	7 8638	0 058855	0 274366	0 333221
crayfish (CRA)	-31 09	7 32	-30 0205	7.604	1 14383	0 080656	1 224486
green sunfish (GSF 83-125 mm)	-25 84	7 23	-25 8465	7 2289	4 22E-05	1 21E-06	4 35E-05
Sacramento squawfish (SQF 75-122 mm)	-26.5	6 97	-26 34	6 94	0 0256	0 0009	0 0265
bluegill (BLG)	-29 34	6 69	-29 1625	7 1004	0 031506	0 168428	0 199934
zooplankton (ZOO)	-34 31	6 3	-34 31	6 3	0	0	0
terrestrial insects (TINS)	-26 75	3 94	-26 75	3 94	0	0	0
aquatic insects (AINS)	-25 49	2 96	-25 49	2 96	0	0	0
phytoplankton (PHY)	-34 72	3 3	-34 72	3 3	0	0	0
periphyton (PER)*	-26 51	2 94	-26 51	2 94	0	0	0
detritus (DET)*	-28 05	-1 18	-28 05	-1 18	0	0	0
					2 48479	1 632353	4 117143

*indicates signatures that were estimated from literature

estimated change in signature per trophic level
carbon: 0 41 ‰
nitrogen: 3 ‰



proportion of diet

food sources	CHS	BRT	BBS	MSMB	MBS	RBT	TFS	YC1BS	CRA	GSF	SQF	BLG	ZOO	TINS*	AINS*	PHY*	PER*	DET*
chinook salmon (CHS 320-638 mm)	0	0 04	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
brown trout (BRT 510 mm)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
big bass - ASB (BBS > 290 mm)	0 46	0 07	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
medium smallmouth bass (MSMB 114-290 mm	0 02	0 06	0 22	0	0 07	0	0	0	0	0	0	0	0	0	0	0	0	0
medium bass - ASB+LMB(MBS 114-290 mm)	0 08	0 08	0 09	0	0 08	0	0	0	0	0	0	0	0	0	0	0	0	0
rainbow trout (RBT 290-410 mm)	0 01	0 01	0 12	0	0 06	0	0	0	0	0	0	0	0	0	0	0	0	0
threadfin shad (TFS)	0 14	0 12	0 06	0 14	0 08	0	0	0	0	0	0	0	0	0	0	0	0	0
YC1 bass (YC1BS 0-113 mm)	0 02	0 12	0 03	0 08	0 08	0	0	0	0 6	0	0	0	0	0	0	0	0	0
crayfish (CRA)	0	0	0 02	0 1	0 07	0	0	0	0	0	0	0	0	0	0	0	0	0
green sunfish (GSF 83-125 mm)	0 04	0 03	0 07	0 3	0 06	0 8	0	0	0	0	0	0	0	0	0	0	0	0
Sacramento squawfish (SQF 75-122 mm)	0 04	0 05	0 06	0 18	0 06	0	0	0	0	0 27	0	0	0	0	0	0	0	0
bluegill (BLG)	0 04	0 1	0 04	0 06	0 07	0	0	0	0	0	0	0	0	0	0	0	0	0
zooplankton (ZOO)	0 07	0 19	0 03	0	0 09	0	0 65	0 57	0	0 01	0	0 34	0	0	0	0	0	0
terrestrial insects (TINS)	0 08	0 06	0 09	0 07	0 09	0	0	0	0	0 16	1	0	0	0	0	0	0	0
aquatic insects (AINS)	0	0 07	0 11	0 07	0 11	0 2	0 35	0 43	0	0 36	0	0	0	0	0	0	0	0
phytoplankton (PHY)	0	0	0 06	0	0 08	0	0	0	0 15	0	0	0 05	1	0	0	0	0	0
periphyton (PER)*	0	0	0	0	0	0	0	0	0	0 2	0	0 61	0	0	0	0	0	0
detritus (DET)*	0	0	0	0	0	0	0	0	0 25	0	0	0	0	0	0	0	0	0
sum	1	1	1	1	1	1	1	1	1	1	1	1	1	0	0	0	0	0
constraint sum	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*0*	*0*	*0*	*0*	*0*
estimated trophic position*	***4.36116***	***3.85496***	***3.99672***	***3.8614***	***3.68876***	***3.856***	***3***	***3***	***3.2***	***3.07***	***3***	***2.34***	***2***	***2***	***2***	***1***	***1***	***1***
estimated trophic position*	***3.8548***	***3.40797***	***3.356***	***3.20413***	***3.13477***	***3.02533***	***2.61033***	***2.52127***	***2.43467***	***2.30963***	***2.21333***	***2.2668***	***2***	***1.213333***	***0.886667***	***1***	***0.88***	***-0.49333***

*Trophic position assumed for these species using approach b

[b]Using approach of Vander Zanden et al. (1997) and Vander Zanden and Rasmussen (1996)

[c]Using approach of Sydeman et al. (1997) and Vander Zanden et al. (1997)

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carbon signature calcs

food sources	CHS	BRT	BBS	MSMB	MBS	RBT	TFS	YC1BS	CRA	GSF	SQF	BLG	ZOO	TINS*	AINS*	PHY*	PER*	DET*
chinook salmon (CHS 320-538 mm)	0	-1.1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
brown trout (BRT 510 mm)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
big bass - ASB (BBS > 290 mm)	-12.3786	-1.8837	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
medium smallmouth bass (MSMB 114-290 mm	-0.5386	-1.6158	-5.9246	0	-1.8851	0	0	0	0	0	0	0	0	0	0	0	0	0
medium bass - ASB+LMB(MBS 114-290 mm)	-2.2536	-2.2536	-2.5353	0	-2.2536	0	0	0	0	0	0	0	0	0	0	0	0	0
rainbow trout (RBT 290-410 mm)	-0.2392	-0.2392	-2.8704	0	-1.4352	0	0	0	0	0	0	0	0	0	0	0	0	0
threadfin shad (TFS)	-4.2	-3.6	-1.8	-4.2	-2.4	0	0	0	0	0	0	0	0	0	0	0	0	0
YC1 bass (YC1BS 0-113 mm)	-0.5988	-3.5928	-0.8982	-2.3952	-2.3952	0	0	0	-17.964	0	0	0	0	0	0	0	0	0
crayfish (CRA)	0	0	-0.6136	-3.068	-2.1476	0	0	0	0	0	0	0	0	0	0	0	0	0
green sunfish (GSF 83-125 mm)	-1.0172	-0.7629	-1.7801	-7.629	-1.5258	-20.344	0	0	0	0	0	0	0	0	0	0	0	0
Sacramento squawfish (SQF 75-122 mm)	-1.0436	-1.3045	-1.5654	-4.6962	-1.5654	0	0	0	0	-7.0443	0	0	0	0	0	0	0	0
bluegill (BLG)	-1.1572	-2.893	-1.1572	-1.7358	-2.0251	0	0	0	0	0	0	0	0	0	0	0	0	0
zooplankton (ZOO)	-2.373	-6.441	-1.017	0	-3.051	0	-22.035	-19.323	0	-0.339	0	-11.526	0	0	0	0	0	0
terrestrial insects (TINS)	-2.1072	-1.5804	-2.3706	-1.8438	-2.3706	0	0	0	0	-4.2144	-26.34	0	0	0	0	0	0	0
aquatic insects (AINS)	0	-1.7556	-2.7588	-1.7556	-2.7588	-5.016	-8.778	-10.7844	0	-9.0288	0	0	0	0	0	0	0	0
phytoplankton (PHY)	0	0	-2.0586	0	-2.7448	0	0	0	-5.1465	0	0	-1.7155	-34.31	0	0	0	0	0
periphyton (PER)*	0	0	0	0	0	0	0	0	0	-5.22	0	-15.921	0	0	0	0	0	0
detritus (DET)*	0	0	0	0	0	0	0	0	-6.91	0	0	0	0	0	0	0	0	0
sum	-27.907	-29.0225	-27.3496	-27.3236	-28.5582	-25.36	-30.813	-30.1074	-30.0205	-25.8465	-26.34	-29.1625	-34.31	-26.75	-25.49	-34.72	-26.51	-28.05

nitrogen signature calcs

food sources	CHS	BRT	BBS	MSMB	MBS	RBT	TFS	YC1BS	CRA	GSF	SQF	BLG	ZOO	TINS*	AINS*	PHY*	PER*	DET*
chinook salmon (CHS 320-538 mm)	0	0.594	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
brown trout (BRT 510 mm)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
big bass - ASB (BBS > 290 mm)	6.1502	0.9359	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
medium smallmouth bass (MSMB 114-290 mm	0.2584	0.7752	2.8424	0	0.9044	0	0	0	0	0	0	0	0	0	0	0	0	0
medium bass - ASB+LMB(MBS 114-290 mm)	1.0168	1.0168	1.1439	0	1.0168	0	0	0	0	0	0	0	0	0	0	0	0	0
rainbow trout (RBT 290-410 mm)	0.1246	0.1246	1.4952	0	0.7476	0	0	0	0	0	0	0	0	0	0	0	0	0
threadfin shad (TFS)	1.7052	1.4616	0.7308	1.7052	0.9744	0	0	0	0	0	0	0	0	0	0	0	0	0
YC1 bass (YC1BS 0-113 mm)	0.2068	1.2408	0.3102	0.8272	0.8272	0	0	0	6.204	0	0	0	0	0	0	0	0	0
crayfish (CRA)	0	0	0.2064	1.032	0.7224	0	0	0	0	0	0	0	0	0	0	0	0	0
green sunfish (GSF 83-125 mm)	0.4092	0.3069	0.7161	3.069	0.6138	8.184	0	0	0	0	0	0	0	0	0	0	0	0
Sacramento squawfish (SQF 75-122 mm)	0.3988	0.4985	0.5982	1.7946	0.5982	0	0	0	0	2.6919	0	0	0	0	0	0	0	0
bluegill (BLG)	0.3876	0.969	0.3876	0.5814	0.6783	0	0	0	0	0	0	0	0	0	0	0	0	0
zooplankton (ZOO)	0.651	1.767	0.279	0	0.837	0	6.045	5.301	0	0.093	0	3.162	0	0	0	0	0	0
terrestrial insects (TINS)	0.5552	0.4164	0.6246	0.4858	0.6246	0	0	0	0	1.1104	6.94	0	0	0	0	0	0	0
aquatic insects (AINS)	0	0.4172	0.6556	0.4172	0.6556	1.192	2.086	2.5628	0	2.1456	0	0	0	0	0	0	0	0
phytoplankton (PHY)	0	0	0.378	0	0.504	0	0	0	0.945	0	0	0.315	6.3	0	0	0	0	0
periphyton (PER)*	0	0	0	0	0	0	0	0	0	1.188	0	3.6234	0	0	0	0	0	0
detritus (DET)*	0	0	0	0	0	0	0	0	0.455	0	0	0	0	0	0	0	0	0
sum	11.8638	10.5239	10.368	9.9124	9.7043	9.376	8.131	7.8638	7.604	7.2289	6.94	7.1004	6.3	3.94	2.96	3.3	2.94	-1.18

assumptions:

1. Part of insects' diets are terrestrial. Since we do not know the terrestrial signature, insect signature is not optimized
2. All of phytoplankton diets is nutrients. Phytoplankton signature is not optimized. "Measured" phytoplankton signature was derived from zooplankton signature

Shading indicates that species in that column was not allowed to eat species in that row

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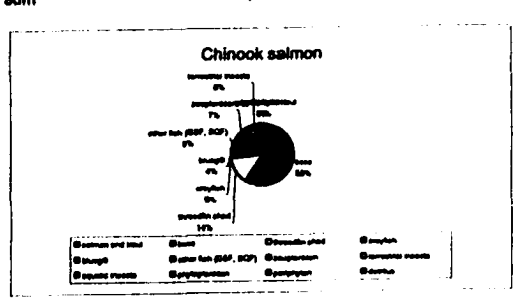
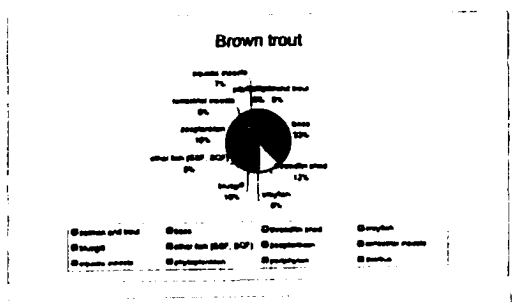
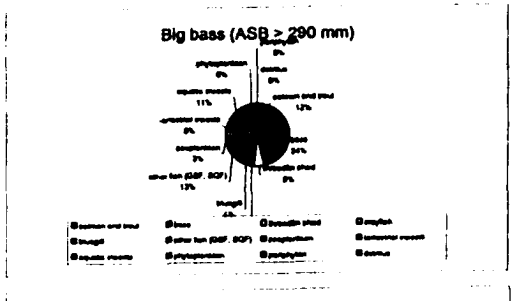
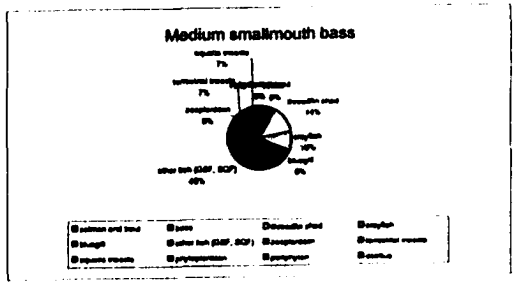
summary proportion of diet

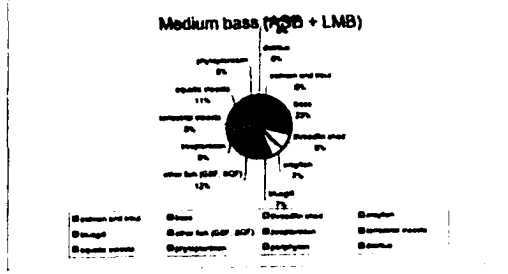
food sources	CHS	BRT	BBS	MSMB	MBS	RBT	TFS	YC1BS	CRA	GSF	SQF	BLG	ZOO	TINS[a]	AINS[a]	PHY[a]	PER[a]	DET[a]
salmon and trout	0.01	0 05	0 12	0	0 06	0	0	0	0	0	0	0	0	0	0	0	0	0
bass	0.58	0.33	0.34	0.08	0.23	0	0	0	0 6	0	0	0	0	0	0	0	0	0
threadfin shad	0.14	0.12	0.06	0.14	0.08	0	0	0	0	0	0	0	0	0	0	0	0	0
crayfish	0	0	0.02	0.1	0 07	0	0	0	0	0	0	0	0	0	0	0	0	0
bluegill	0.04	0.1	0 04	0.06	0.07	0	0	0	0	0	0	0	0	0	0	0	0	0
other fish (GSF, SQF)	0.08	0.08	0.13	0.48	0.12	0.8	0	0	0	0.27	0	0	0	0	0	0	0	0
zooplankton	0.07	0.19	0.03	0	0.09	0	0.65	0.57	0	0.01	0	0.34	0	0	0	0	0	0
terrestrial insects	0.08	0.06	0.09	0.07	0.09	0	0	0	0	0 16	1	0	0	0	0	0	0	0
aquatic insects	0	0.07	0.11	0.07	0.11	0.2	0.35	0.43	0	0.36	0	0	0	0	0	0	0	0
phytoplankton	0	0	0.06	0	0.08	0	0	0	0.15	0	0	0.05	1	0	0	0	0	0
periphyton	0	0	0	0	0	0	0	0	0	0.2	0	0.61	0	0	0	0	0	0
detritus	0	0	0	0	0	0	0	0	0.25	0	0	0	0	0	0	0	0	0
sum	1	1	1	1	1	1	1	1	1	1	1	1	1	0	0	0	0	0

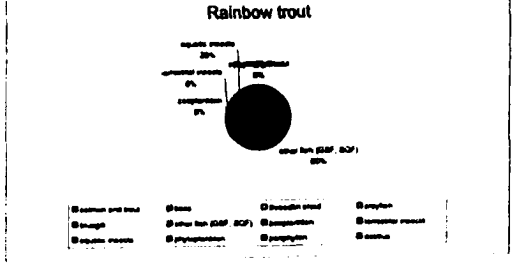












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=$B$34:$C$34=0
=$B$35:$B$39>=0
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=$C$26>=0
=$C$27=0
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```

APPENDIX O. Food web pathways and estimated energy transfer for Shasta species

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estimation of energy transferred using 10% transfer efficiency

TL	diet proportion	energy available per 100 J phyto (J)	Number of trophic transfers 10	9	8	7	6	5	4	3	2	1
Phytoplankton												
1	100%	100										
Zooplankton												
2	100%	10									ZOO	PHY
	total	10										
Bluegill												
3	34%	1								BLG	ZOO	PHY
2	5%	10									BLG	PHY
2.871795	total	0.84										
Green sunfish												
3	1%	1								GSF	ZOO	PHY
	total	0.01										
Year class 1 bass												
3	57%	1								YC1BS	ZOO	PHY
	total	0.57										
Threadfin shad												
3	65%	1								TFS	ZOO	PHY
	total	0.65										
Crayfish												
4	60%	0.057							CRA	YC1BS	ZOO	PHY
2	15%	10									CRA	PHY
3.6	total	1.5342										
Rainbow trout												
4	80%	0.001							RBT	GSF	ZOO	PHY
	total	0.0008										
Medium smallmouth bass												
4	14%	0.065							MSMB	TFS	ZOO	PHY
4	8%	0.057							MSMB	YC1BS	ZOO	PHY
4.6	10%	0.15342						MSMB	CRA	YC1BS	ZOO	PHY
										MSMB	CRA	PHY
4	30%	0.001							MSMB	GSF	ZOO	PHY
3.871795	6%	0.084							MSMB	BLG	ZOO	PHY
										MSMB	BLG	PHY
4.076923	total	0.034342										
Medium bass												
5.076923	7%	0.003434						MBS	MSMB	TFS	ZOO	PHY
								MBS	MSMB	YC1BS	ZOO	PHY
							MBS	MSMB	CRA	YC1BS	ZOO	PHY
								MBS	MSMB	CRA	PHY	
								MBS	MSMB	GSF	ZOO	PHY
								MBS	MSMB	BLG	ZOO	PHY
									MBS	MSMB	BLG	PHY
5	6%	0.00008						MBS	RBT	GSF	ZOO	PHY
4	8%	0.065							MBS	TFS	ZOO	PHY
4	8%	0.057							MBS	YC1BS	ZOO	PHY
4.6	7%	0.15342						MBS	CRA	YC1BS	ZOO	PHY
										MBS	CRA	PHY
4	6%	0.001							MBS	GSF	ZOO	PHY
3.871795	7%	0.084							MBS	BLG	ZOO	PHY
										MBS	BLG	PHY
3	9%	1								MBS	ZOO	PHY
2	8%	10									MBS	PHY
3.876379	8%	0.091668					MBS	MBS	MSMB	TFS	ZOO	PHY
							MBS	MBS	MSMB	YC1BS	ZOO	PHY
						MBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
								MBS	MBS	MSMB	CRA	PHY
							MBS	MBS	MSMB	GSF	ZOO	PHY
							MBS	MBS	MSMB	BLG	ZOO	PHY
								MBS	MBS	MSMB	BLG	PHY
							MBS	MBS	RBT	GSF	ZOO	PHY

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estimation of energy transferred using 10% transfer efficiency

TL	diet proportion	energy available per 100 J phyto (J)	Number of trophic transfers 10	9	8	7	6	5	4	3	2	1
								MBS	MBS	TFS	ZOO	PHY
								MBS	MBS	YC1BS	ZOO	PHY
							MBS	MBS	CRA	YC1BS	ZOO	PHY
									MBS	MBS	CRA	PHY
								MBS	MBS	GSF	ZOO	PHY
								MBS	MBS	BLG	ZOO	PHY
									MBS	MBS	BLG	PHY
									MBS	MBS	ZOO	PHY
										MBS	MBS	PHY
3.876379	total	0.924018										
Big bass												
5.076923	22%	0.003434						BBS	MSMB	TFS	ZOO	PHY
								BBS	MSMB	YC1BS	ZOO	PHY
							BBS	MSMB	CRA	YC1BS	ZOO	PHY
									BBS	MSMB	CRA	PHY
								BBS	MSMB	GSF	ZOO	PHY
								BBS	MSMB	BLG	ZOO	PHY
									BBS	MSMB	BLG	PHY
4.876379	9%	0.092402					BBS	MBS	MSMB	TFS	ZOO	PHY
							BBS	MBS	MSMB	YC1BS	ZOO	PHY
						BBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
								BBS	MBS	MSMB	CRA	PHY
							BBS	MBS	MSMB	GSF	ZOO	PHY
							BBS	MBS	MSMB	BLG	ZOO	PHY
								BBS	MBS	MSMB	BLG	PHY
							BBS	MBS	RBT	GSF	ZOO	PHY
								BBS	MBS	TFS	ZOO	PHY
								BBS	MBS	YC1BS	ZOO	PHY
							BBS	MBS	CRA	YC1BS	ZOO	PHY
									BBS	MBS	CRA	PHY
								BBS	MBS	GSF	ZOO	PHY
								BBS	MBS	BLG	ZOO	PHY
									BBS	MBS	BLG	PHY
									BBS	MBS	ZOO	PHY
										BBS	MBS	PHY
						BBS	MBS	MBS	MSMB	TFS	ZOO	PHY
						BBS	MBS	MBS	MSMB	YC1BS	ZOO	PHY
					BBS	MBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
							BBS	MBS	MBS	MSMB	CRA	PHY
						BBS	MBS	MBS	MSMB	GSF	ZOO	PHY
						BBS	MBS	MBS	MSMB	BLG	ZOO	PHY
							BBS	MBS	MBS	MSMB	BLG	PHY
						BBS	MBS	MBS	RBT	GSF	ZOO	PHY
							BBS	MBS	MBS	TFS	ZOO	PHY
							BBS	MBS	MBS	YC1BS	ZOO	PHY
						BBS	MBS	MBS	CRA	YC1BS	ZOO	PHY
								BBS	MBS	MBS	CRA	PHY
							BBS	MBS	MBS	GSF	ZOO	PHY
							BBS	MBS	MBS	BLG	ZOO	PHY
								BBS	MBS	MBS	BLG	PHY
								BBS	MBS	MBS	ZOO	PHY
									BBS	MBS	MBS	PHY
5	12%	0.00008						BBS	RBT	GSF	ZOO	PHY
4	6%	0.065							BBS	TFS	ZOO	PHY
4	3%	0.057							BBS	YC1BS	ZOO	PHY
4.6	2%	0.15342						BBS	CRA	YC1BS	ZOO	PHY
										BBS	CRA	PHY
4	7%	0.001							BBS	GSF	ZOO	PHY
3.871795	4%	0.084							BBS	BLG	ZOO	PHY
										BBS	BLG	PHY
3	3%	1								BBS	ZOO	PHY
2	6%	10									BBS	PHY
4.233336	total	0.65119										
Chinook salmon												
5.233336	46%	0.065119					CHS	BBS	MSMB	TFS	ZOO	PHY
							CHS	BBS	MSMB	YC1BS	ZOO	PHY
						CHS	BBS	MSMB	CRA	YC1BS	ZOO	PHY

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estimation of energy transferred using 10% transfer efficiency

TL	diet proportion	energy available per 100 J phyto (J)	Number of trophic transfers 10	9	8	7	6	5	4	3	2	1
								CHS	BBS	MSMB	CRA	PHY
							CHS	BBS	MSMB	GSF	ZOO	PHY
							CHS	BBS	MSMB	BLG	ZOO	PHY
								CHS	BBS	MSMB	BLG	PHY
						CHS	BBS	MBS	MSMB	TFS	ZOO	PHY
						CHS	BBS	MBS	MSMB	YC1BS	ZOO	PHY
					CHS	BBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
						CHS	BBS	MBS	MSMB	CRA	PHY	
						CHS	BBS	MBS	MSMB	GSF	ZOO	PHY
						CHS	BBS	MBS	MSMB	BLG	ZOO	PHY
						CHS	BBS	MBS	MSMB	BLG	PHY	
						CHS	BBS	MBS	RBT	GSF	ZOO	PHY
						CHS	BBS	MBS	TFS	ZOO	PHY	
						CHS	BBS	MBS	YC1BS	ZOO	PHY	
						CHS	BBS	MBS	CRA	YC1BS	ZOO	PHY
							CHS	BBS	MBS	CRA	PHY	
						CHS	BBS	MBS	MBS	GSF	ZOO	PHY
						CHS	BBS	MBS	MBS	BLG	ZOO	PHY
							CHS	BBS	MBS	BLG	PHY	
							CHS	BBS	MBS	BLG	ZOO	PHY
								CHS	BBS	MBS	MBS	PHY
					CHS	BBS	MBS	MBS	MSMB	TFS	ZOO	PHY
					CHS	BBS	MBS	MBS	MSMB	YC1BS	ZOO	PHY
				CHS	BBS	MBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
						CHS	BBS	MBS	MBS	MSMB	CRA	PHY
					CHS	BBS	MBS	MBS	MSMB	GSF	ZOO	PHY
					CHS	BBS	MBS	MBS	MSMB	BLG	ZOO	PHY
						CHS	BBS	MBS	MSMB	BLG	ZOO	PHY
					CHS	BBS	MBS	MBS	RBT	GSF	ZOO	PHY
						CHS	BBS	MBS	MBS	TFS	ZOO	PHY
						CHS	BBS	MBS	MBS	YC1BS	ZOO	PHY
					CHS	BBS	MBS	MBS	CRA	YC1BS	ZOO	PHY
							CHS	BBS	MBS	MBS	CRA	PHY
						CHS	BBS	MBS	MBS	GSF	ZOO	PHY
						CHS	BBS	MBS	MBS	BLG	ZOO	PHY
							CHS	BBS	MBS	MBS	BLG	PHY
							CHS	BBS	MBS	MBS	ZOO	PHY
								CHS	BBS	MBS	MBS	PHY
							CHS	BBS	RBT	GSF	ZOO	PHY
								CHS	BBS	TFS	ZOO	PHY
								CHS	BBS	YC1BS	ZOO	PHY
							CHS	BBS	CRA	YC1BS	ZOO	PHY
									CHS	BBS	CRA	PHY
								CHS	BBS	GSF	ZOO	PHY
								CHS	BBS	BLG	ZOO	PHY
									CHS	BBS	BLG	PHY
									CHS	BBS	ZOO	PHY
										CHS	BBS	PHY
5.076923	2%	0.003434						CHS	MSMB	TFS	ZOO	PHY
								CHS	MSMB	YC1BS	ZOO	PHY
							CHS	MSMB	CRA	YC1BS	ZOO	PHY
									CHS	MSMB	CRA	PHY
								CHS	MSMB	GSF	ZOO	PHY
								CHS	MSMB	BLG	ZOO	PHY
									CHS	MSMB	BLG	PHY
4.876379	8%	0.092402					CHS	MBS	MSMB	TFS	ZOO	PHY
							CHS	MBS	MSMB	YC1BS	ZOO	PHY
						CHS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
							CHS	MBS	MSMB	CRA	PHY	
							CHS	MBS	MSMB	GSF	ZOO	PHY
							CHS	MBS	MSMB	BLG	ZOO	PHY
								CHS	MBS	MSMB	BLG	PHY
							CHS	MBS	RBT	GSF	ZOO	PHY
								CHS	MBS	TFS	ZOO	PHY
								CHS	MBS	YC1BS	ZOO	PHY
							CHS	MBS	CRA	YC1BS	ZOO	PHY
									CHS	MBS	CRA	PHY
								CHS	MBS	GSF	ZOO	PHY
								CHS	MBS	BLG	ZOO	PHY

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estimation of energy transferred using 10% transfer efficiency

TL	diet proportion	energy available per 100 J phyto (J)	Number of trophic transfers 10	9	8	7	6	5	4	3	2	1
									CHS	MBS	BLG	PHY
									CHS	MBS	ZOO	PHY
										CHS	MBS	PHY
						CHS	MBS	MBS	MSMB	TFS	ZOO	PHY
						CHS	MBS	MBS	MSMB	YC1BS	ZOO	PHY
					CHS	MBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
							CHS	MBS	MBS	MSMB	CRA	PHY
						CHS	MBS	MBS	MSMB	GSF	ZOO	PHY
						CHS	MBS	MBS	MSMB	BLG	ZOO	PHY
							CHS	MBS	MBS	MSMB	BLG	PHY
						CHS	MBS	MBS	RBT	GSF	ZOO	PHY
							CHS	MBS	MBS	TFS	ZOO	PHY
							CHS	MBS	MBS	YC1BS	ZOO	PHY
						CHS	MBS	MBS	CRA	YC1BS	ZOO	PHY
								CHS	MBS	MBS	CRA	PHY
							CHS	MBS	MBS	GSF	ZOO	PHY
							CHS	MBS	MBS	BLG	ZOO	PHY
								CHS	MBS	MBS	BLG	PHY
								CHS	MBS	MBS	ZOO	PHY
									CHS	MBS	MBS	PHY
5	1%	0.00008						CHS	RBT	GSF	ZOO	PHY
4	14%	0.065							CHS	TFS	ZOO	PHY
4	2%	0.057							CHS	YC1BS	ZOO	PHY
4	4%	0.001							CHS	GSF	ZOO	PHY
3.871795	4%	0.084							CHS	BLG	ZOO	PHY
										CHS	BLG	PHY
3	7%	1								CHS	ZOO	PHY
4.819575	total	0.121056										
Brown trout												
5.819575	4%	0.012106				BRT	CHS	BBS	MSMB	TFS	ZOO	PHY
						BRT	CHS	BBS	MSMB	YC1BS	ZOO	PHY
					BRT	CHS	BBS	MSMB	CRA	YC1BS	ZOO	PHY
							BRT	CHS	BBS	MSMB	CRA	PHY
						BRT	CHS	BBS	MSMB	GSF	ZOO	PHY
						BRT	CHS	BBS	MSMB	BLG	ZOO	PHY
							BRT	CHS	BBS	MSMB	BLG	PHY
					BRT	CHS	BBS	MBS	MSMB	TFS	ZOO	PHY
					BRT	CHS	BBS	MBS	MSMB	YC1BS	ZOO	PHY
				BRT	CHS	BBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
						BRT	CHS	BBS	MBS	MSMB	CRA	PHY
					BRT	CHS	BBS	MBS	MSMB	GSF	ZOO	PHY
					BRT	CHS	BBS	MBS	MSMB	BLG	ZOO	PHY
						BRT	CHS	BBS	MBS	MSMB	BLG	PHY
					BRT	CHS	BBS	MBS	RBT	GSF	ZOO	PHY
						BRT	CHS	BBS	MBS	TFS	ZOO	PHY
						BRT	CHS	BBS	MBS	YC1BS	ZOO	PHY
					BRT	CHS	BBS	MBS	CRA	YC1BS	ZOO	PHY
							BRT	CHS	BBS	MBS	CRA	PHY
						BRT	CHS	BBS	MBS	GSF	ZOO	PHY
						BRT	CHS	BBS	MBS	BLG	ZOO	PHY
							BRT	CHS	BBS	MBS	BLG	PHY
							BRT	CHS	BBS	MBS	ZOO	PHY
								BRT	CHS	BBS	MBS	PHY
				BRT	CHS	BBS	MBS	MBS	MSMB	TFS	ZOO	PHY
				BRT	CHS	BBS	MBS	MBS	MSMB	YC1BS	ZOO	PHY
			BRT	CHS	BBS	MBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
					BRT	CHS	BBS	MBS	MBS	MSMB	CRA	PHY
				BRT	CHS	BBS	MBS	MBS	MSMB	GSF	ZOO	PHY
				BRT	CHS	BBS	MBS	MBS	MSMB	BLG	ZOO	PHY
					BRT	CHS	BBS	MBS	MBS	MSMB	BLG	PHY
				BRT	CHS	BBS	MBS	MBS	RBT	GSF	ZOO	PHY
					BRT	CHS	BBS	MBS	MBS	TFS	ZOO	PHY
					BRT	CHS	BBS	MBS	MBS	YC1BS	ZOO	PHY
				BRT	CHS	BBS	MBS	MBS	CRA	YC1BS	ZOO	PHY
						BRT	CHS	BBS	MBS	MBS	CRA	PHY
					BRT	CHS	BBS	MBS	MBS	GSF	ZOO	PHY
					BRT	CHS	BBS	MBS	MBS	BLG	ZOO	PHY
						BRT	CHS	BBS	MBS	MBS	BLG	PHY

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estimation of energy transferred using 10% transfer efficiency

TL	diet proportion	energy available per 100 J phyto (J)	Number of trophic transfers 10	9	8	7	6	5	4	3	2	1
						BRT	CHS	BBS	MBS	MBS	ZOO	PHY
							BRT	CHS	BBS	MBS	MBS	PHY
						BRT	CHS	BBS	RBT	GSF	ZOO	PHY
							BRT	CHS	BBS	TFS	ZOO	PHY
							BRT	CHS	BBS	YC1BS	ZOO	PHY
						BRT	CHS	BBS	CRA	YC1BS	ZOO	PHY
								BRT	CHS	BBS	CRA	PHY
							BRT	CHS	BBS	GSF	ZOO	PHY
							BRT	CHS	BBS	BLG	ZOO	PHY
								BRT	CHS	BBS	BLG	PHY
								BRT	CHS	BBS	ZOO	PHY
									BRT	CHS	BBS	PHY
							BRT	CHS	MSMB	TFS	ZOO	PHY
							BRT	CHS	MSMB	YC1BS	ZOO	PHY
						BRT	CHS	MSMB	CRA	YC1BS	ZOO	PHY
								BRT	CHS	MSMB	CRA	PHY
							BRT	CHS	MSMB	GSF	ZOO	PHY
							BRT	CHS	MSMB	BLG	ZOO	PHY
								BRT	CHS	MSMB	BLG	PHY
						BRT	CHS	MBS	MSMB	TFS	ZOO	PHY
						BRT	CHS	MBS	MSMB	YC1BS	ZOO	PHY
					BRT	CHS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
						BRT	CHS	MBS	MSMB	CRA	PHY	
						BRT	CHS	MBS	MSMB	GSF	ZOO	PHY
						BRT	CHS	MBS	MSMB	BLG	ZOO	PHY
							BRT	CHS	MBS	MSMB	BLG	PHY
						BRT	CHS	MBS	RBT	GSF	ZOO	PHY
							BRT	CHS	MBS	TFS	ZOO	PHY
							BRT	CHS	MBS	YC1BS	ZOO	PHY
						BRT	CHS	MBS	CRA	YC1BS	ZOO	PHY
							BRT	CHS	MBS	CRA	PHY	
						BRT	CHS	MBS	MSMB	GSF	ZOO	PHY
						BRT	CHS	MBS	MSMB	BLG	ZOO	PHY
							BRT	CHS	MBS	BLG	PHY	
							BRT	CHS	MBS	BLG	ZOO	PHY
								BRT	CHS	MBS	MBS	PHY
					BRT	CHS	MBS	MSMB	TFS	ZOO	PHY	
					BRT	CHS	MBS	MSMB	YC1BS	ZOO	PHY	
				BRT	CHS	MBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
					BRT	CHS	MBS	MBS	MSMB	CRA	PHY	
					BRT	CHS	MBS	MBS	MSMB	GSF	ZOO	PHY
					BRT	CHS	MBS	MBS	MSMB	BLG	ZOO	PHY
						BRT	CHS	MBS	MBS	MSMB	BLG	PHY
					BRT	CHS	MBS	MBS	RBT	GSF	ZOO	PHY
						BRT	CHS	MBS	MBS	TFS	ZOO	PHY
						BRT	CHS	MBS	MBS	YC1BS	ZOO	PHY
					BRT	CHS	MBS	MBS	CRA	YC1BS	ZOO	PHY
							BRT	CHS	MBS	MBS	CRA	PHY
						BRT	CHS	MBS	MBS	GSF	ZOO	PHY
						BRT	CHS	MBS	MBS	BLG	ZOO	PHY
							BRT	CHS	MBS	MBS	BLG	PHY
							BRT	CHS	MBS	MBS	ZOO	PHY
								BRT	CHS	MBS	MBS	PHY
							BRT	CHS	RBT	GSF	ZOO	PHY
								BRT	CHS	TFS	ZOO	PHY
								BRT	CHS	YC1BS	ZOO	PHY
								BRT	CHS	GSF	ZOO	PHY
								BRT	CHS	BLG	ZOO	PHY
									BRT	CHS	BLG	PHY
									BRT	CHS	ZOO	PHY
5.233336	7%	0.065119					BRT	BBS	MSMB	TFS	ZOO	PHY
							BRT	BBS	MSMB	YC1BS	ZOO	PHY
						BRT	BBS	MSMB	CRA	YC1BS	ZOO	PHY
								BRT	BBS	MSMB	CRA	PHY
							BRT	BBS	MSMB	GSF	ZOO	PHY
							BRT	BBS	MSMB	BLG	ZOO	PHY
								BRT	BBS	MSMB	BLG	PHY
						BRT	BBS	MBS	MSMB	TFS	ZOO	PHY
						BRT	BBS	MBS	MSMB	YC1BS	ZOO	PHY

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estimation of energy transferred using 10% transfer efficiency

TL	diet proportion	energy available per 100 J phyto (J)	Number of trophic transfers 10	9	8	7	6	5	4	3	2	1
					BRT	BBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
							BRT	BBS	MBS	MSMB	CRA	PHY
						BRT	BBS	MBS	MSMB	GSF	ZOO	PHY
						BRT	BBS	MBS	MSMB	BLG	ZOO	PHY
						BRT	BRT	BBS	MBS	MSMB	BLG	PHY
						BRT	BBS	MBS	RBT	GSF	ZOO	PHY
							BRT	BBS	MBS	TFS	ZOO	PHY
							BRT	BBS	MBS	YC1BS	ZOO	PHY
						BRT	BBS	MBS	CRA	YC1BS	ZOO	PHY
								BRT	BBS	MBS	CRA	PHY
							BRT	BBS	MBS	GSF	ZOO	PHY
							BRT	BBS	MBS	BLG	ZOO	PHY
								BRT	BBS	MBS	BLG	PHY
								BRT	BBS	MBS	ZOO	PHY
									BRT	BBS	MBS	PHY
					BRT	BBS	MBS	MBS	MSMB	TFS	ZOO	PHY
					BRT	BBS	MBS	MBS	MSMB	YC1BS	ZOO	PHY
				BRT	BBS	MBS	MBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY
						BRT	BBS	MBS	MBS	MSMB	CRA	PHY
					BRT	BBS	MBS	MBS	MSMB	GSF	ZOO	PHY
					BRT	BBS	MBS	MBS	MSMB	BLG	ZOO	PHY
						BRT	BBS	MBS	MBS	MSMB	BLG	PHY
					BRT	BBS	MBS	MBS	RBT	GSF	ZOO	PHY
						BRT	BBS	MBS	MBS	TFS	ZOO	PHY
						BRT	BBS	MBS	MBS	YC1BS	ZOO	PHY
					BRT	BBS	MBS	MBS	CRA	YC1BS	ZOO	PHY
						BRT	BBS	MBS	MBS	CRA	PHY
					BRT	BBS	MBS	MBS	GSF	ZOO	PHY
					BRT	BBS	MBS	MBS	BLG	ZOO	PHY
						BRT	BBS	MBS	MBS	BLG	PHY
						BRT	BBS	MBS	MBS	ZOO	PHY
							BRT	BBS	MBS	MBS	PHY
						BRT	BBS	RBT	GSF	ZOO	PHY
							BRT	BBS	TFS	ZOO	PHY
							BRT	BBS	YC1BS	ZOO	PHY
						BRT	BBS	CRA	YC1BS	ZOO	PHY
								BRT	BBS	CRA	PHY
							BRT	BBS	GSF	ZOO	PHY
							BRT	BBS	BLG	ZOO	PHY
								BRT	BBS	BLG	PHY
								BRT	BBS	ZOO	PHY
									BRT	BBS	PHY
5.076923	6%	0.003434						BRT	MSMB	TFS	ZOO	PHY
								BRT	MSMB	YC1BS	ZOO	PHY
							BRT	MSMB	CRA	YC1BS	ZOO	PHY
									BRT	MSMB	CRA	PHY
								BRT	MSMB	GSF	ZOO	PHY
								BRT	MSMB	BLG	ZOO	PHY
									BRT	MSMB	BLG	PHY
4.876379	8%	0.092402					BRT	MBS	MSMB	TFS	ZOO	PHY
							BRT	MBS	MSMB	YC1BS	ZOO	PHY
						BRT	MBS	MSMB	CRA	YC1BS	ZOO	PHY
								BRT	MBS	MSMB	CRA	PHY
							BRT	MBS	MSMB	GSF	ZOO	PHY
							BRT	MBS	MSMB	BLG	ZOO	PHY
								BRT	MBS	MSMB	BLG	PHY
							BRT	MBS	RBT	GSF	ZOO	PHY
								BRT	MBS	TFS	ZOO	PHY
								BRT	MBS	YC1BS	ZOO	PHY
							BRT	MBS	CRA	YC1BS	ZOO	PHY
									BRT	MBS	CRA	PHY
								BRT	MBS	GSF	ZOO	PHY
								BRT	MBS	BLG	ZOO	PHY
									BRT	MBS	BLG	PHY
									BRT	MBS	ZOO	PHY
										BRT	MBS	PHY
						BRT	MBS	MBS	MSMB	TFS	ZOO	PHY
						BRT	MBS	MBS	MSMB	YC1BS	ZOO	PHY
					BRT	MBS	MBS	MSMB	CRA	YC1BS	ZOO	PHY

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estimation of energy transferred using 10% transfer efficiency

TL	diet proportion	energy available per 100 J phyto (J)	Number of trophic transfers 10	9	8	7	6	5	4	3	2	1
							BRT	MBS	MBS	MSMB	CRA	PHY
						BRT	MBS	MBS	MSMB	GSF	ZOO	PHY
						BRT	MBS	MBS	MSMB	BLG	ZOO	PHY
							BRT	MBS	MBS	MSMB	BLG	PHY
						BRT	MBS	MBS	RBT	GSF	ZOO	PHY
							BRT	MBS	MBS	TFS	ZOO	PHY
							BRT	MBS	MBS	YC1BS	ZOO	PHY
						BRT	MBS	MBS	CRA	YC1BS	ZOO	PHY
								BRT	MBS	MBS	CRA	PHY
							BRT	MBS	MBS	GSF	ZOO	PHY
							BRT	MBS	MBS	BLG	ZOO	PHY
								BRT	MBS	MBS	BLG	PHY
								BRT	MBS	MBS	ZOO	PHY
									BRT	MBS	MBS	PHY
5	1%	0.00008						BRT	RBT	GSF	ZOO	PHY
4	12%	0.065							BRT	TFS	ZOO	PHY
4	12%	0.057							BRT	YC1BS	ZOO	PHY
4	3%	0.001							BRT	GSF	ZOO	PHY
3.871795	10%	0.084							BRT	BLG	ZOO	PHY
										BRT	BLG	PHY
3	19%	1								BRT	ZOO	PHY
4.123197	total	0.225712										
											total paths	398

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APPENDIX P. Estimation of fractions of species' diets from phytoplankton-derived sources

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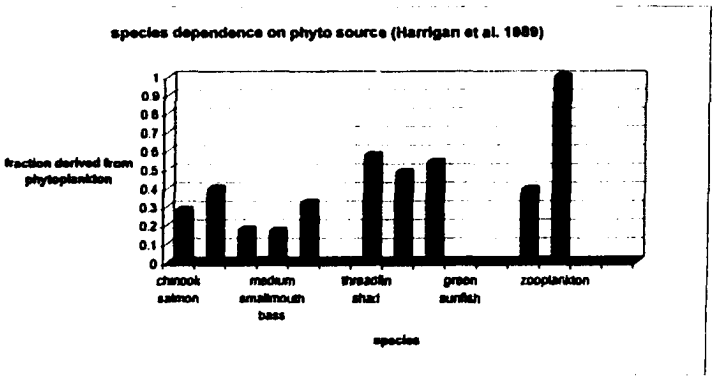
species	measured $\delta^{13}C$(‰)	measured $\delta^{15}N$(‰)	TL (est.)	primary source $\delta^{13}C$(‰)	primary source $\delta^{15}N$(‰)	estimated $\delta^{13}C$(‰)	estimated $\delta^{15}N$(‰)	error^2 $\delta^{13}C$(‰)	error^2 $\delta^{15}N$(‰)	sum
chinook salmon	-27.91	11.85	4.361178	-29.2881	1.766455	-29.2881	1.766454	7.11E-12	1.21E-10	1.28E-10
brown trout	-29.02	10.54	3.854962	-30.1905	1.975114	-30.1906	1.975107	2.21E-10	5.22E-11	2.73E-10
big bass	-27.32	10.37	3.996716	-28.5446	1.409851	-28.5446	1.409854	1.24E-11	1.05E-11	2.29E-11
medium smallmouth bass	-27.34	9.92	3.8614	-28.5132	1.3358	-28.5132	1.335807	5.08E-11	4.83E-11	9.91E-11
medium bass	-28.58	9.71	3.668759	-29.6824	1.643724	-29.6824	1.643723	1.2E-10	3.73E-13	1.2E-10
rainbow trout	-24.33	9.46	3.856	-25.501	0.892	-25.5191	0.898768	0.000328	4.58E-05	0.000374
threadfin shad	-30.41	9.18	3	-31.23	3.16	-31.2315	3.147031	2.1E-06	0.001087	0.001089
YC1 bass	-30.35	7.34	3	-31.17	1.34	-31.17	1.340016	2.27E-10	2.67E-10	4.94E-10
crayfish	-31.09	7.52	3.2	-31.992	0.72	-31.6459	1.235257	0.119765	0.26549	0.385255
green sunfish	-25.84	7.23	3.07	-26.6867	1.02	-26.7068	1.02677	0.000328	4.58E-05	0.000374
Sacramento squawfish	-26.5	6.97	3	-27.32	0.97	-27.32	0.969991	8.96E-14	8.83E-11	8.84E-11
bluegill	-29.34	6.69	2.34	-29.8894	2.67	-29.8894	2.66997	2.51E-10	8.94E-10	1.14E-09
zooplankton	-34.31	6.3	2	-34.72	3.3	-34.72	3.3	0	0	0
terrestrial insects	-26.75	3.94	2	-27.18	0.94	-27.1781	0.94677	0.000328	4.58E-05	0.000374
aquatic insects	-25.49	2.96	2	-25.9	-0.04	-25.9181	-0.03319	0.000328	4.64E-05	0.000374
phytoplankton	-34.72	3.3	1							
periphyton	-26.51	2.94	1							
detritus	-28.05	-1.18	1							
*indicates signatures that were estimated from literature								0.121078	0.266761	0.387839

estimated change in signature per trophic level
carbon: 0.41 ‰
nitrogen: 3 ‰

species	phyto	peri	det	sum	$\delta^{13}C$(‰)	$\delta^{15}N$(‰)
chinook salmon	0.280354	0.410306	0.309338	1	-29.2881	1.766454
brown trout	0.397647	0.333193	0.268961	1	-30.1905	1.975107
big bass	0.175277	0.438013	0.38671	1	-28.5446	1.409854
medium smallmouth bass	0.1682	0.427736	0.404064	1	-28.5132	1.335807
medium bass	0.322108	0.335117	0.342776	1	-29.6824	1.643723
rainbow trout	0	0.486361	0.450112	0.936473	-25.5191	0.898768
threadfin shad	0.575085	0.424915	0	1	-31.2315	3.147031
YC1 bass	0.486776	0.082345	0.430879	1	-31.17	1.340016
crayfish	0.53912	0	0.46088	1	-31.6459	1.235257
green sunfish	0	0.530247	0.450979	0.981226	-26.7068	1.02677
Sacramento squawfish	0.006825	0.512247	0.478929	1	-27.32	0.969991
bluegill	0.392888	0.50724	0.099871	1	-29.8894	2.66997
zooplankton	1	0	0	1	-34.72	3.3
terrestrial insects	0	0.515408	0.481805	0.997213	-27.1781	0.94677
aquatic insects	0	0.260683	0.677626	0.938309	-25.9181	-0.03319



APPENDIX Q. Caloric conversion estimate for phytoplankton

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From Cummins and Wuycheck (1971) - in calories

species	spp at Shasta?	per g dw	per g ash-free dw	per g ww	N	# avgd	std dev	notes
primary producers (terrestrial and aquatic)		4135			1070	342		
			4681		769	255		
aquatic primary productivity		3482			502	126		
			4628		359	89		
				611	171	40		
Chlorophyta		3850			89	22		
			4780		55	15		
				847	22	5		
Chlorophyta - Chlamydomonas reinhardi	Y	5289			12	1	95.6	lab
Chlorophyta - Pandorina morum	Y		4969		1	1		lab
Chlorophyta - Cladophora sp.	Y	2120			5			all year
Chlorophyta - Cladophora sp.	Y		5170	580	5			all year
Chlorophyta - Oocystaceae	Y?	5120			9	3		
Chlorophyta - Oocystaceae	Y?		5287		1	1		
Chlorophyta - Scenedesmus obliquus	Y	5158			1			
Chlorophyta - Scenedesmus obliquus	Y	5507			1			
Chlorophyta - Scenedesmus brasiliensis	Y	5453			1			
Chlorophyta - Spirogyra maxima	Y	2449			6			all year
Chlorophyta - Spirogyra spp.	Y	4204.3			1			fall
Chrysophyta - Navicula minima	Y	3218			6		201	lab
Chrysophyta - Navicula spp.	Y	4943			1			lab
Chrysophyta - Melosira spp.	Y		5150		2			summer
Chrysophyta - Nitzschia paradora	Y	3280			2			all year
Chrysophyta - Nitzschia paradora	Y		5470		2		160	all year
Cyanophyta - Microcystis spp.	Y		4781		1		812	august
Cyanophyta - Anabaena solitaria	Y		5410		2			summer
Cyanophyta - Oscillatonia terebriformis	Y	1249			7		96	lab
Cyanophyta - Oscillatonia terebriformis	Y		4405		7			lab
mixed blue-green algae			5175		4		350	summer
mixed algal species		4477			5	1	63	aug-nov
mixed algal species			4469		5	1	68	aug-nov
avg reported spp at Shasta (shaded values):		3999.192	5080.25					

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APPENDIX R. Figures and spreadsheet showing results of linked food web-energy transfer model and CE-QUAL-W2 assessment

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stratified period

energy conversion[a]: 5080 cal (g organic dry weight)$^{-1}$ 4.168 J cal^{-1}

[a] average of Shasta species from Cummins and Wuycheck (1971)

	stratified period begin (JD)	stratified period end (JD)	number of stratified days (JD)	whole lake gross prod (g)	whole lake dark resp (g)	whole lake net prod (g)	whole lake volume (m^3)	surface area (m^2)	whole lake areal net prod (g m^{-2})	whole lake volumetric net prod (g m^{-3})	whole lake phyto energy (J)
1995 without	85	328	244	1.41E+10	8.11E+09	5.97E+09	4.67E+09	1.01E+08	61.17888	1.378373	1.26E+14
1995 with	84	326	243	1.4E+10	7.81E+09	6.16E+09	4.67E+09	1.02E+08	63.04834	1.417757	1.31E+14
1995 difference			-1	-1.07E+08	-3.05E+08	1.98E+08	8093688	103568.2	1.869456	0.039384	4.2E+12
1995 percent difference			-0.41	-0.76	-3.76	3.32	0.17	0.10	3.06	2.86	3.32
Run #1 without	84	321	238	2.1E+10	1.22E+10	8.86E+09	5.36E+09	1.1E+08	80.8558	1.665949	1.88E+14
Run #1 with	83	325	243	2.23E+10	1.26E+10	9.66E+09	5.36E+09	1.1E+08	88.28435	1.819685	2.05E+14
Run #1 difference			5	1.28E+09	4.7E+08	8.09E+08	-19191111	-226721	7.428547	0.153736	1.71E+13
Run #1 percent difference			2.10	6.08	3.86	9.13	-0.36	-0.21	9.19	9.23	9.13
Run #4 without	71	278	208	8.24E+09	4.86E+09	3.37E+09	2.64E+09	70535956	47.68123	1.306713	7.14E+13
Run #4 with	65	276	212	7.78E+09	4.5E+09	3.27E+09	2.57E+09	69336450	47.04667	1.298875	6.93E+13
Run #4 difference			4	-4.57E+08	-3.59E+08	-98236637	-63920353	-1199507	-0.634561	-0.007839	-2.08E+12
Run #4 percent difference			1.92	-5.55	-7.38	-2.91	-2.42	-1.70	-1.33	-0.60	-2.91

	stratified period begin (JD)	stratified period end (JD)	number of stratified days (JD)	epilimnetic gross prod (g)	epilimnetic dark resp (g)	epilimnetic net prod (g)	epilimnetic volume (m^3)	surface area (m^2)	epilimnetic areal net prod (g m^{-2})	epilimnetic volumetric net prod (g m^{-3})	epilimnetic phyto energy (J)
1995 without	85	328	244	1.09E+10	5.18E+09	5.68E+09	1.12E+09	1.01E+08	58.74033	1.331265	1.2E+14
1995 with	84	326	243	1.07E+10	5.03E+09	5.71E+09	1.1E+09	1.02E+08	59.0223	1.337047	1.21E+14
1995 difference			-1	-1.24E+08	-1.55E+08	30536058	-12695748	103568.2	0.281966	0.005782	6.47E+11
1995 percent difference			-0.41	-1.14	-2.98	0.54	-1.14	0.10	0.48	0.43	0.54
Run #1 without	84	321	238	2.02E+10	1.04E+10	9.83E+09	1.83E+09	1.1E+08	89.61356	1.844841	2.08E+14
Run #1 with	83	325	243	2.14E+10	1.09E+10	1.06E+10	1.85E+09	1.1E+08	96.54645	1.98863	2.24E+14
Run #1 difference			5	1.22E+09	4.67E+08	7.53E+08	25691642	-226721	6.932887	0.143789	1.59E+13
Run #1 percent difference			2.10	6.03	4.49	7.66	1.41	-0.21	7.74	7.79	7.66
Run #4 without	71	278	208	7.88E+09	3.86E+09	4.02E+09	9.8E+08	70535956	56.67268	1.549639	8.51E+13
Run #4 with	65	276	212	7.45E+09	3.59E+09	3.86E+09	9.84E+08	69336450	55.57778	1.533423	8.18E+13
Run #4 difference			4	-4.29E+08	-2.75E+08	-1.53E+08	3202616	-1199507	-1.0949	-0.016416	-3.25E+12
Run #4 percent difference			1.92	-5.44	-7.13	-3.82	0.33	-1.70	-1.93	-1.06	-3.82

	whole lake phyto energy (J)	wet weight (g) chinook salmon	brown trout	big bass	medium bass	medium smallmouth bass	year class 1 bass	bluegill	zooplankton
energy transfer (J J^{-1})		0.0013	0.0022	0.0065	0.0073	0.0003	0.0057	0.0084	0.1
predator energy density (J (g wet mass)$^{-1}$)[a, b, c]		6058.874	7736.4	4186	4186	4186	4186	4186	2513.5
1995 without	1.26E+14	27105567	35924507	1.96E+08	2.2E+08	9053762	1.72E+08	2.54E+08	5.03E+09
1995 with	1.31E+14	28006471	37118525	2.03E+08	2.28E+08	9354681	1.78E+08	2.62E+08	5.19E+09
1995 difference	4.2E+12	900904.3	1194018	6519906	7322356	300918.7	5717456	8425725	1.67E+08
1995 percent difference	3.323687	3.323687	3.323687	3.323687	3.323687	3.323687	3.323687	3.323687	3.323687
Run #1 without	1.88E+14	40232013	53321712	2.91E+08	3.27E+08	13436239	2.55E+08	3.76E+08	7.46E+09
Run #1 with	2.05E+14	43906016	58191071	3.18E+08	3.57E+08	14665424	2.79E+08	4.11E+08	8.14E+09
Run #1 difference	1.71E+13	3674002	4869358	26589006	29861500	1227185	23316513	34361178	6.81E+08
Run #1 percent difference	9.132037	9.132037	9.132037	9.132037	9.132037	9.132037	9.132037	9.132037	9.132037
Run #4 without	7.14E+13	15319852	20304247	1.11E+08	1.25E+08	5117115	97225180	1.43E+08	2.84E+09
Run #4 with	6.93E+13	14873563	19712755	1.08E+08	1.21E+08	4966046	94392871	1.39E+08	2.76E+09
Run #4 difference	-2.08E+12	-446289.2	-591491.7	-3229825	-3627342	-149068.9	-2832308	-4173928	-82753433
-(Run #4 difference)	2.08E+12	446289.2	591491.7	3229825	3627342	149068.9	2832308	4173928	82753433
Run #4 percent difference	-2.913143	-2.913143	-2.913143	-2.913143	-2.913143	-2.913143	-2.913143	-2.913143	-2.913143

	epilimnetic phyto energy (J)	wet weight (g) chinook salmon	brown trout	big bass	medium bass	medium smallmouth bass	year class 1 bass	bluegill	zooplankton
energy transfer (J J^{-1})		0.0013	0.0022	0.0065	0.0073	0.0003	0.0057	0.0084	0.1
predator energy density (J (g wet mass)$^{-1}$)[a, b, c]		6058.874	7736.4	4186	4186	4186	4186	4186	2513.5
1995 without	1.2E+14	25809659	34206968	1.87E+08	2.1E+08	8620905	1.64E+08	2.41E+08	4.79E+09
1995 with	1.21E+14	25948384	34390829	1.88E+08	2.11E+08	8667242	1.65E+08	2.43E+08	4.81E+09
1995 difference	6.47E+11	138725.4	183860.4	1003965	1127530	46336.84	880400	1297432	25723231
1995 percent difference	0.537494	0.537494	0.537494	0.537494	0.537494	0.537494	0.537494	0.537494	0.537494
Run #1 without	2.08E+14	44640555	59164596	3.23E+08	3.63E+08	14910773	2.83E+08	4.18E+08	8.28E+09
Run #1 with	2.24E+14	48059282	63696624	3.48E+08	3.91E+08	16052692	3.05E+08	4.49E+08	8.91E+09
Run #1 difference	1.59E+13	3418727	4531028	24741582	27786678	1141918	21696447	31973711	6.34E+08
Run #1 percent difference	7.658344	7.658344	7.658344	7.658344	7.658344	7.658344	7.658344	7.658344	7.658344
Run #4 without	8.51E+13	18253127	24191879	1.32E+08	1.48E+08	6096883	1.16E+08	1.71E+08	3.38E+09
Run #4 with	8.18E+13	17555828	23267710	1.27E+08	1.43E+08	5863972	1.11E+08	1.64E+08	3.26E+09
Run #4 difference	-3.25E+12	-697298.8	-924168.7	-5046399	-5667495	-232910.7	-4425304	-6521501	-1.29E+08
-(Run #4 difference)	3.25E+12	697298.8	924168.7	5046399	5667495	232910.7	4425304	6521501	1.29E+08
Run #4 percent difference	-3.820161	-3.820161	-3.820161	-3.820161	-3.820161	-3.820161	-3.820161	-3.820161	-3.820161

Notes:

a. Predator energy densities from Hanson et al. (1997) unless otherwise stated

b. Average size of chinook salmon = 299 g (based on two measured weights - could be on small side)

c. Wet weight of brown trout assumed to be 2000 g based on size (510 mm)

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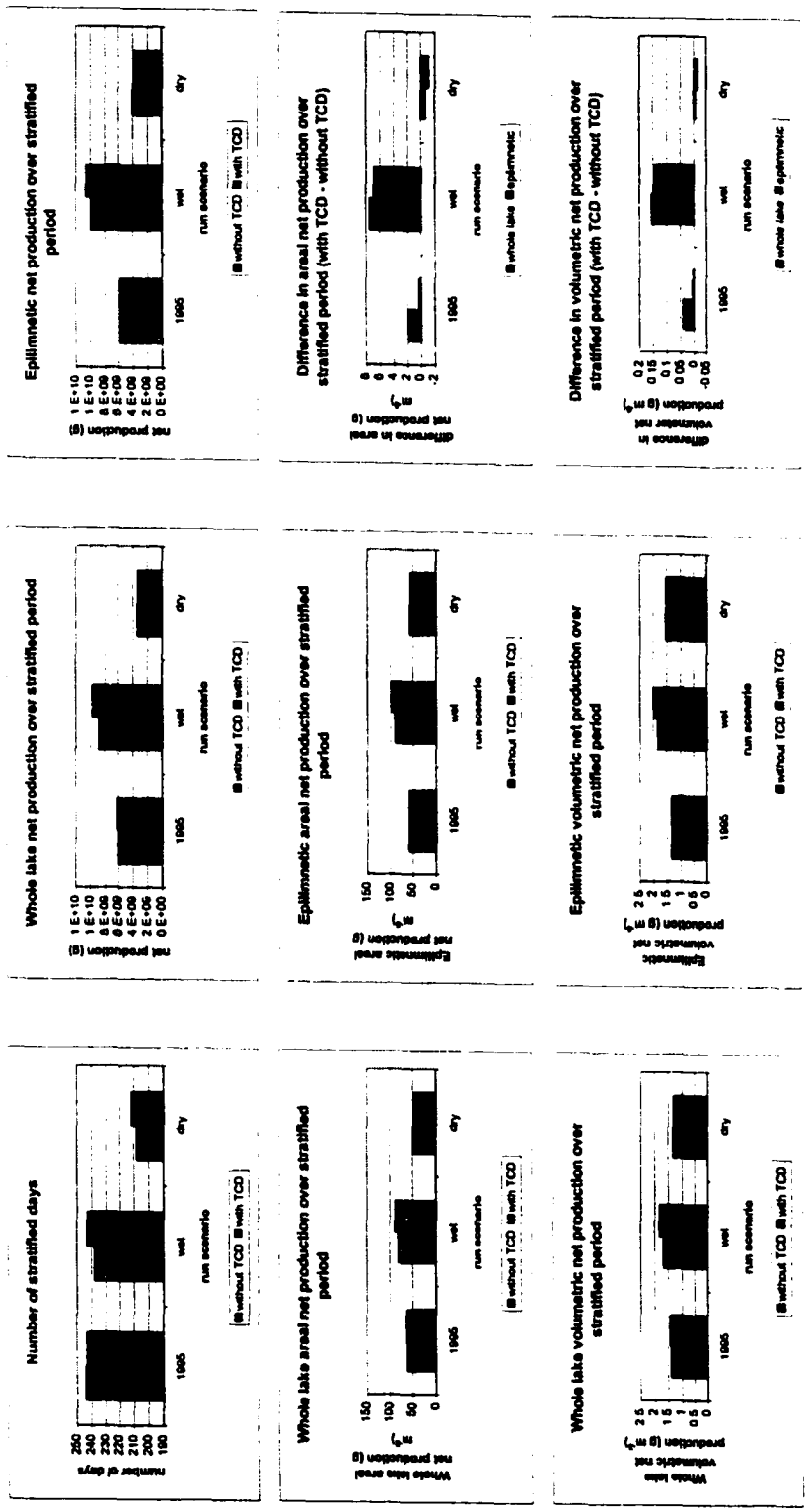

Figure 48. Column charts comparing number of stratified days and net production for wet, dry and 1995 scenarios

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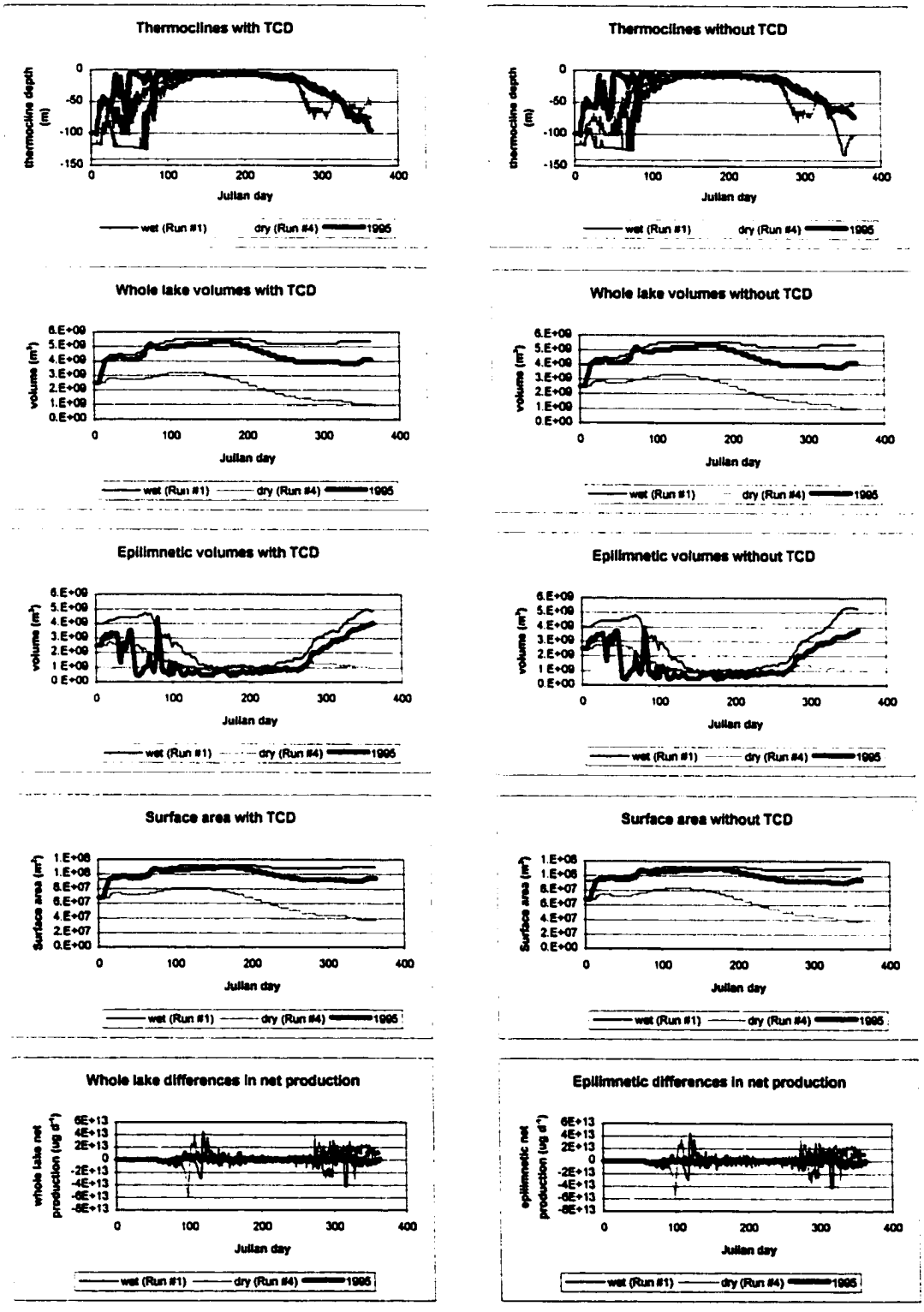

Figure 49. Comparisons of thermoclines, volumes, surface area, and whole lake differences in net production between wet, dry and 1995 scenarios

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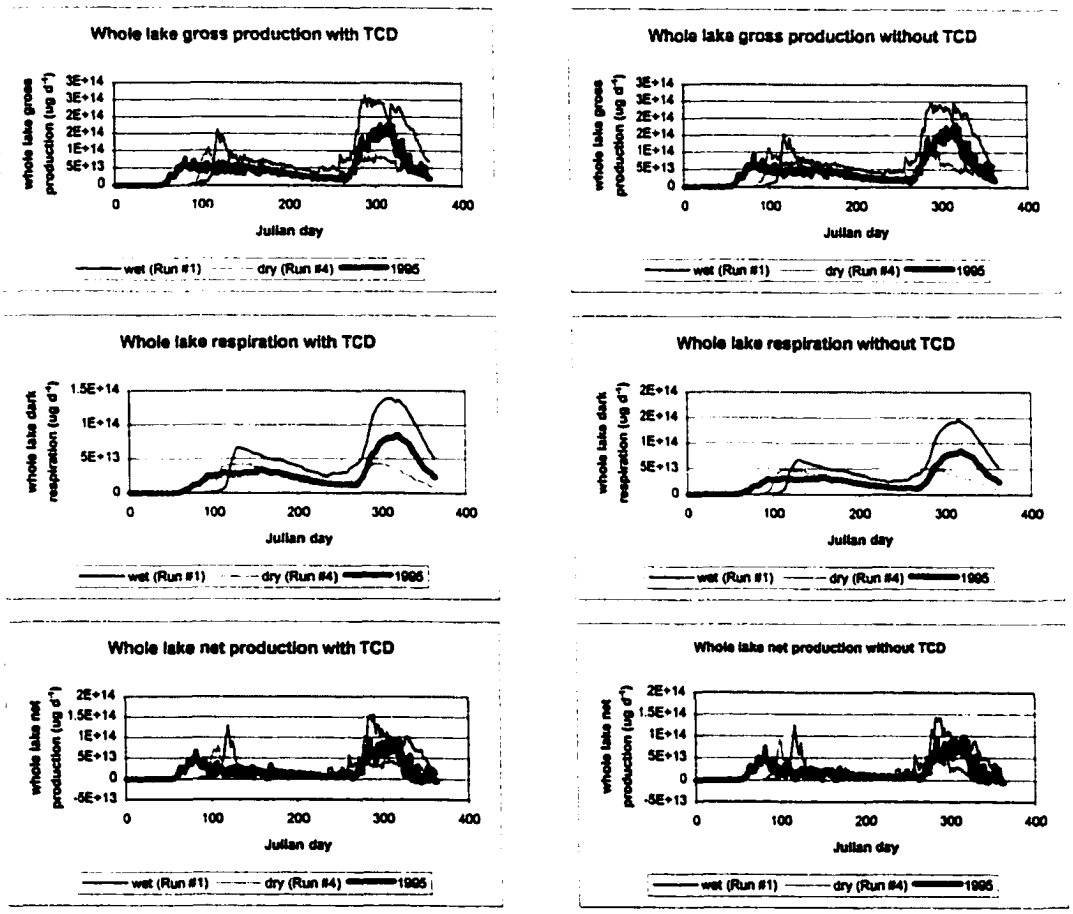

Figure 50. Comparison of whole lake production for wet, dry and 1995 scenarios

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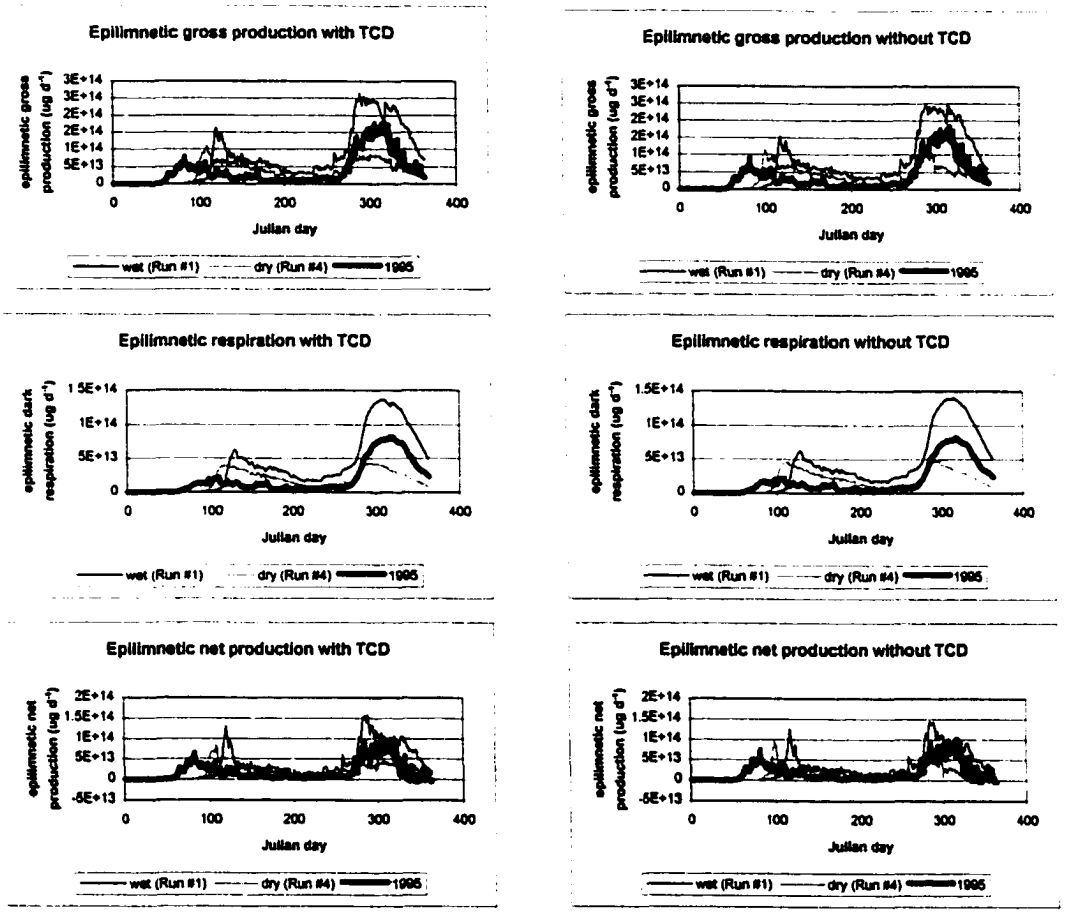

Figure 51. Comparison of epilimnetic production for wet, dry and 1995 scenarios

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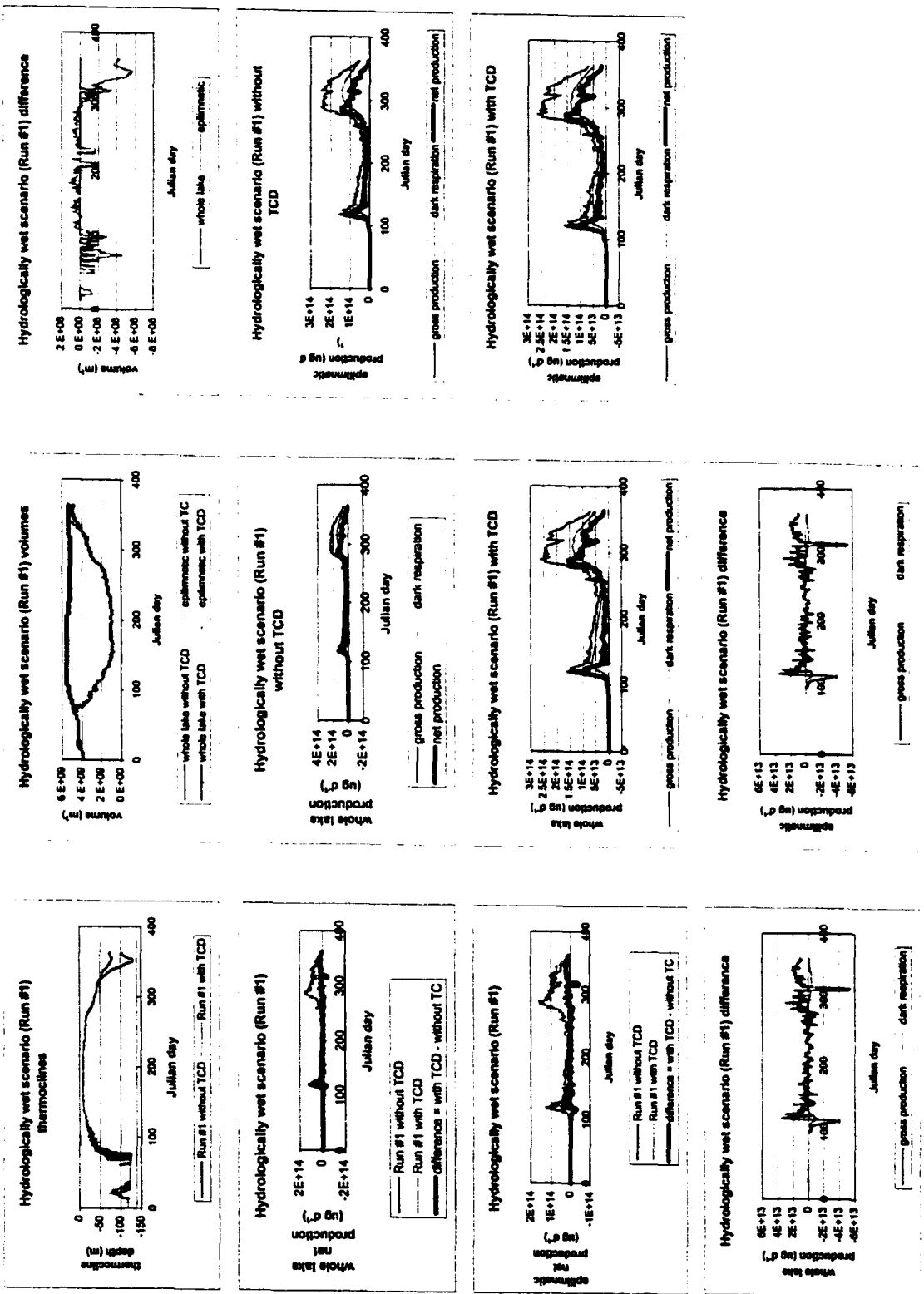

Figure 52. Plots for hydrologically wet scenario (Run #1)

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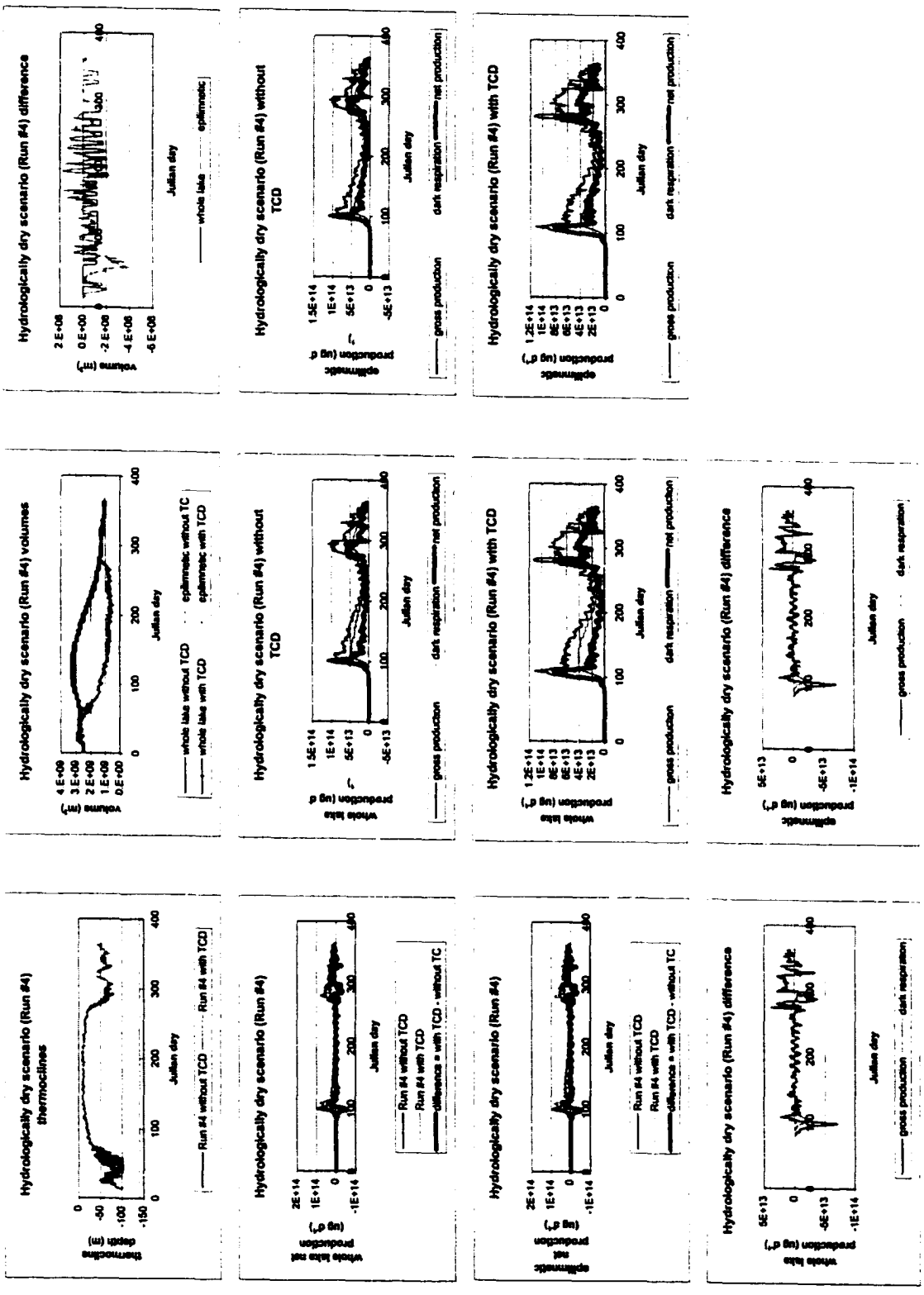

Figure 53. Plots for hydrologically dry scenario (Run #4)

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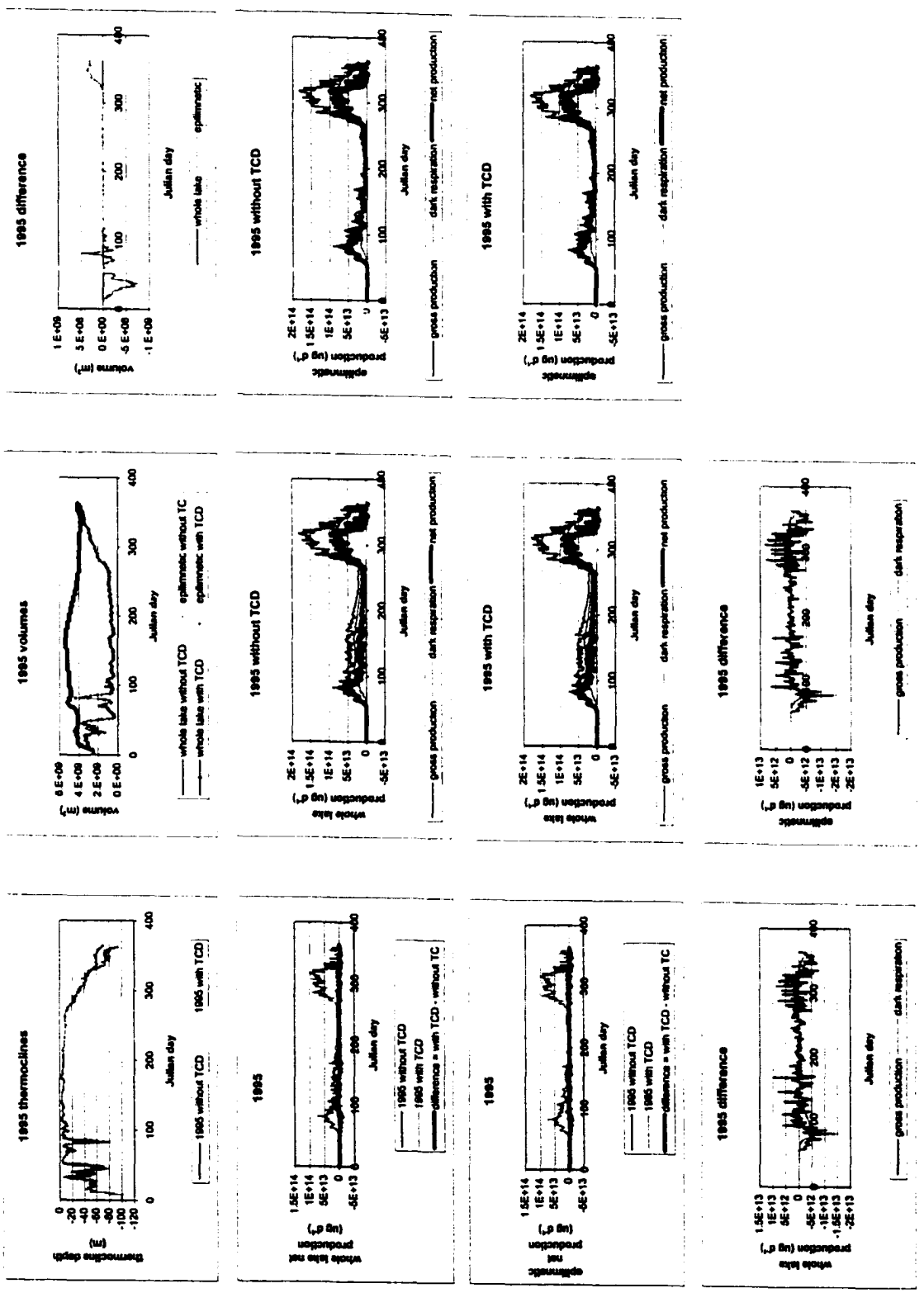

Figure 54. Plots for 1995 conditions

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