Final Report
Evaluation of Chloride Management Alternatives: Application of the Comprehensive Aquatic Ecosystem Model (CASM) to Lake Texoma

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Contents

Introduction

Natural sources of total dissolved solids (TDS) are located on tributary streams to the Red River in the upper Red River Basin. Lake Texoma (Oklahoma, Texas) waters are high in ionic content as a result of inflows from the naturally saline or brackish Red River. The elevated concentrations of mainly chloride, sulfate, sodium, and calcium pose problems for domestic, agricultural, and industrial uses of Lake Texoma water (Schroeder and Toro 1996). Expensive and complex potable water treatment systems are often required to remedy these circumstances and facilitate use of Lake Texoma water.

The U.S. Army Corps of Engineers manages a chloride reduction program aimed at reducing chloride concentrations in major tributaries to Lake Texoma in order to improve water supplies. Concerns have been raised that decreases in total dissolved solids as a result of chloride management will reduce sedimentation rates within the reservoir. Corresponding increases in concentrations of suspended sediments might reduce light availability for primary production mainly by phytoplankton. A major issue concerns the propagation of potentially reduced levels of primary production throughout the Lake Texoma food web, ultimately expressed as reductions in the biomass of striped bass – a valuable recreational fishery in Lake Texoma.

The purpose of this modeling study was to evaluate the potential ecological and food web implications of managing chloride concentrations in Lake Texoma using the Comprehensive Aquatic Systems Model (CASM). The CASM is a flexible aquatic food web/ecosystem modeling platform that has been used to address both basic (e.g., DeAngelis *et al.* 1989) and applied (e.g., Bartell 2003, Bartell *et al.* 1999) questions in a variety of freshwater and coastal aquatic ecosystems. A version of the CASM has been developed to assess potential impacts (i.e., risks) of chloride management alternatives on light availability, primary production, and food web dynamics for selected locations within Lake Texoma. The Lake Texoma CASM importantly projects changes in the abundance of striped bass populations in relation to chloride management.

The Comprehensive Aquatic Systems Model (CASM) has been used to address challenges in theoretical ecology (DeAngelis *et al.* 1989) and assess potential risks posed by chemical contaminants in aquatic ecosystems (Bartell *et al.* 1999; Bartell *et al.* 2000; Naito *et al.* 2002). Theoretical foundations for modeling toxic effects and estimating risk in aquatic ecosystems derive from earlier peer-reviewed work based on the Standard Water Column Model (SWACOM) (O'Neill *et al.* 1982; O'Neill *et al.* 1983).

The following report describes the development of the Lake Texoma CASM and its application in assessing the future ecological production dynamics both without chloride management and for selected chloride management alternatives.

Modeling Objectives

The principal modeling objectives were to (1) develop a Lake Texoma version of the CASM that simulated ecological production dynamics of producer and consumer populations consistent with measured production, (2) use the Lake Texoma CASM to examine the potential food web implications of alternative chloride management scenarios, and (3) examine the responses of modeled populations to chloride management in relation to annual environmental variability and longer-term loss of storage capacity of this large reservoir.

The first modeling objective involves the collation and integration of existing data to develop an aquatic food web structure and define trophic interactions representative of Lake Texoma. Values of bioenergetics parameters are estimated for each of the modeled populations. This objective also entails the acquisition of necessary environmental input data required by the CASM. Repeated simulations using adjusted values of the model parameters are used to calibrate the model to measured values of population biomass. The calibrated model serves as a hypothesis concerning trophic dynamics of Lake Texoma populations.

To meet the second objective, the calibrated model will be used to assess possible effects of chloride management on Lake Texoma food webs. Speculations are that reduced total dissolved solids will reduce setting rates of suspended inorganic particles and correspondingly reduce light availability and decrease algal primary production. Decreased primary production is anticipated to propagate throughout the Lake Texoma food web and ultimately reduce populations of striped bass. An additional contention is that alteration of light availability might promote the growth of *Prymnesium parvum*. Alternative chloride management scenarios will be transformed into associated time series of total dissolved solids. The CASM will simulate the effects of these time series on the biomass of each of the modeled populations of producers and consumers. In pursuing this modeling objective, the same environmental input conditions will be used in all simulations.

The third modeling objective examines the relative impacts of chloride management, environmental variability, and reservoir aging on the production of modeled populations. To address this objective, the same chloride management scenarios developed for the second objective are simulated but with different environmental input values. The environmental inputs are varied by amounts determined by year-to-year variations observed for Lake Texoma. The simulations including environmental variability are then repeated using time-dependent decreases in water depths for the modeled locations within Lake Texoma. The modeled trends in decreased depth reflect projected longer-term losses in volume or storage due to sedimentation of Lake Texoma. The third objective examines the importance (i.e., signal:noise) of chloride management within the context of environmental variability and reservoir aging.

Model Description

CASM is a complex aquatic ecosystem model that considers water chemistry characteristics, spatial and temporal scales, and food web structure. Water chemistry parameters modeled within CASM include dissolved inorganic nitrogen, dissolved inorganic phosphorus, dissolved silica, dissolved oxygen, dissolved organic matter, particulate organic matter, and water temperature. CASM simulates daily concentrations of water quality parameters based on an overall mass-balance approach that includes external inputs to the aquatic ecosystem, internal uptake, and recycling of nutrients. Moreover, CASM simulates daily production dynamics of each producer and consumer population for a period of one or multiple years. Production dynamics are computed as grams of carbon (g-C) and integrated over the water column as defined by the system depth. The resulting biomass is expressed as g-C/m^2. Parameters that define water chemistry are expressed as volumetric units (e.g., mg/L).

Physical, chemical, biological, and ecological characteristics that represent the aquatic system of interest are determined based on site-specific data published in peer-reviewed literature and government documents. Community structure, population characteristics, and representative species within each population are selected based on ecological characteristics native to the aquatic system of interest and surrounding environment. Bioenergetics parameters are defined for each producer and consumer population. For producers, biomass production is dependent on initial biomass, growth rates, optimal temperature, light saturation, half-saturation constants (nitrogen, phosphorous, and silica), and sinking rates. For consumers, biomass production is dependent on initial biomass, maximum consumption, optimal temperature, respiration, excretion, and mortality. Biomass values of modeled populations, as well as water chemistry parameters, are determined by differential equations (implemented as difference equations) that are solved using a daily time-step (Bartell *et al.* 1999; Bartell *et al.* 2000; DeAngelis *et al.* 1989). Based on these equations, the model conserves biomass and other modeled materials (i.e., nutrients, dissolved carbon, and particulate carbon), where new carbon fixed via photosynthesis remains in the form of plant biomass, propagates through the food web, and finally enters the particulate organic matter pool. Consumer biomass exists as standing biomass, enters the food web, and then enters the particulate organic matter pool. Producer- or consumer-derived particulate organic matter may re-enter the food web, decompose, or accumulate.

The following sections describe the adaptation of the CASM to produce a version of the model capable of addressing the three principal modeling objectives.

Physical Model Structure

Lake Texoma results from the impoundment of the Red River and the Washita River by the Denison Dam. As a result, Lake Texoma features areas that are essentially riverine, transition areas to more lake-like conditions, and the limnetic characteristics of conditions near the dam. In this initial "proof-of-concept" application of the CASM, the complex physical structure of Lake Texoma was approximated using six locations distributed among the major hydrographic features of the reservoir (Figure 1).

The six locations were based on previous studies of phytoplankton and zooplankton productivity and community structure (e.g., Waller *et al.* 2002). The locations are described as follows with reference to the corresponding sampling station in Waller *et al.* (2002):

- CASM_LT01: Red River riverine location corresponds to station 1,
- CASM_LT02: Red River transition location corresponds to station 7,
- CASM_LT03: Red River main lake location corresponds to station 9,
- CASM_LT04: Main lake location near dam corresponds to station 17,
- CASM_LT05: Washita River transition location corresponds to station 20,
- CASM_LT06: Washita River riverine location corresponds to station 24.

A single CASM water column was modeled at each of the six locations. No attempt was made to explicitly model hydrodynamics or transport within the reservoir. However, results of a previous physical modeling effort for Lake Texoma performed by the University of North Texas (Acevedo *et al.* 2004) were used to approximate daily transport as it affected values of transportable state variables (e.g., nutrients, plankton, suspended particulate matter, TSS) in the Lake Texoma CASM.

Whole-reservoir scale values of selected CASM outputs were constructed using the results simulated for the six locations multiplied by the surface area of the reservoir segments represented by each location. GIS analysis of current aerial coverage's of Lake Texoma produced estimates of the total surface area represented by each of the modeled locations. Average segment lengths and widths are defined for each corresponding location in an input data set (LakeTexoma_geo.dat) used by the Lake Texoma CASM.

Food Web Structure

Given the stated modeling objectives, specifying a relevant food web structure was an important step in developing a version of the CASM for Lake Texoma. A combined taxonomic/functional approach was used to describe the key aquatic food web components in evaluating the potential impacts of chloride management alternatives (Figure 2). The Lake Texoma CASM food web structure reflects a consensus opinion based on a workshop held at the Tulsa District offices in October 2007. Participants from the USACE, the U.S. Fish and Wildlife Service, the Oklahoma Water Resources Board, the Oklahoma Department of Wildlife Conservation, and the Texas Parks and Wildlife Department reviewed and commented on an initial draft food web structure.

Phytoplankton

A broad taxonomic approach was used in specifying the phytoplankton assemblage included in the Lake Texoma CASM. The aggregated state variables are based on the detailed analyses of Lake Texoma phytoplankton provided by Waller *et al.* (2002).

However, even though the phytoplankton compartments are broadly defined, parameter estimation reflects the important species within these aggregated state variables. For example, the cyanophyte (blue green algae) compartment is almost entirely based on *Microcystis incerta*, the most abundant alga in the Waller *et al.* (2002) study. The second most abundant class of phytoplankton includes chloropohyte (green algae) species. Microflagellates constitute the third most abundant phytoplankton group in Lake Texoma (Waller *et al.* 2002). Diatoms are seasonally (i.e., spring) abundant in Lake Texoma (Waller *et al.* 2002). The dominant diatom species aggregated as 'diatoms' in the Lake Texoma food web are *Stephanodiscus hantzschii* and *Thalassiosira pseudonana*.

Blooms of the haptophyte *Prymnesium parvum* have been observed to cause substantial fish kills (Johansson and Graneli 1999). Because of their ecological importance apart from trophic interactions, *P. parvum* was represented as a separate species-specific phytoplankton population in the Lake Texoma CASM.

Zooplankton

Waller *et al.* (2002) described 59 species of zooplankton inhabiting Lake Texoma. Apart from copepod nauplii, the rotifer *Keratella quadrata_f_testuda* was the most abundant species. Other common rotifers included *K. cochlearis* and *Polyartha dolichoptera*. The Cladocera was dominated by the common reservoir species *Bosmina longirsotris*. *Diaphanosoma bergei* was also a common member of the Lake Texoma zooplankton assemblage. Cyclopoid copepods (e.g., *Mesocyclops edax*) were more abundant than calanoid copepods in this study.

Given the results of the Waller *et al.* (2002) analyses, the Lake Texoma CASM food web includes four zooplankton compartments: rotifers, cladocerans, cyclopoids, and calanoids. Parameter estimation and specification of trophic interactions focus on the dominant species associated with these taxonomic groups.

Benthos

The principal modeling focus in this study was on propagation of potential impacts of chloride management on production dynamics within the water column. However, contributions of benthic organisms to overall system productivity can be important in certain locations within large reservoirs. In addition, it might be possible that chloride management (e.g., decreased sedimentation) could reduce food supplies to benthic organisms and indirectly their production. Sublette (1957) provided a detailed qualitative description of the littoral-benthic community characteristic of Lake Texoma. Based on this description, the Lake Texoma CASM specifies three functional guilds of benthic invertebrates: mollusks, amphipods, and aquatic insects. These benthic invertebrates also serve as food supplies for several fishes represented in the model (e.g., freshwater drum, smallmouth buffalo, and blue catfish).

Fish

Based on intensive year-round sampling in 1954, 1981-1984, and 1996-1997, Gido *et al.* (2000) concluded that the fish assemblage in Lake Texoma is relatively stable. The dominant offshore species based on samples collected by Gido and Matthews (2000) in 1981-1984 are gizzard shad (*Dorosoma cepedianum* 34.1%), striped bass (*Morone saxatilis* 20.9%), threadfin shad (*D. petenense* 9.5%), and freshwater drum (*Aplodinotus grunniens* 9.5%). Other common species collected throughout the 43-y sampling period include white bass (*Morone chrysops*), smallmouth buffalo (*Ictiobus bubalus*), blue catfish (*Ictalurus furcatus*), and channel catfish (*Ictalurus punctatus*) (Gido *et al.* 2000). Goldeye (*Hiodon alosoides*) and white crappie (*Pomoxis annularis*) were important members of this assemblage in 1954, however their abundance has decreased during the near half-century of sampling.

Other species in the Lake Texoma fish community described by Gido *et al.* (2000) are common carp (*Cyprinus carpio*), river carpsucker (*Carpiodes carpio*), longnose gar (*Lepisosteus osseus*), largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), bigmouth buffalo (*Ictiobus cyprinellas*), and black crappie (*Pomoxis nigromaculatus*). However, these species are either characteristic of nearshore habitats or are sufficiently small in numbers to be addressed by the CASM application developed for Lake Texoma. In addition, smaller fishes (e.g., inland silversides *Menidia beryllina*) and minnows were not effectively sampled by the gill nets used in these studies (Gido and Matthews 2000).

The potential indirect impacts of chloride management on the production dynamics of recreationally fished striped bass are of key concern for Lake Texoma. Therefore, the Lake Texoma CASM includes three populations that define young-of-year (YOY), juvenile, and adult striped bass. These three age-classes are modeled separately because of differences in dietary preferences and reproduction by adults. YOY striped bass feed primarily on zooplankton, while juvenile and adult individuals feed heavily on gizzard shad and threadfin shad.

One possible effect of chloride management might be to change the relative proportion of primary production that enters deeper waters in the form of sinking phytoplankton or sedimented POC. Therefore, smallmouth buffalo, freshwater drum, and blue catfish were added as separate state variables in the modeled Lake Texoma fish assemblage.

Because of the potential competition with striped bass for shad, the white bass was included along with freshwater drum in the modeled food web.

Governing Equations

CASM was originally constructed as a set of coupled ordinary differential equations based on nonlinear and time-varying terms (DeAngelis *et al.* 1989). One equation is defined for each of the modeled state variables, including all aquatic plant or consumer populations and water quality parameters. The number of water quality parameters has remained consistent with previous CASM applications. However, the number of modeled populations varies according to the specified modeling objectives and food web structure.

The governing equations are based on a biomass (carbon) equivalent of energy – in other words, the equations fundamentally describe input-output bioenergetics that impact the growth rate of each modeled population. Growth rates are integrated using a simple difference approach.

Primary Producer Population Dynamics

Biomass production for primary producer populations is governed by a bioenergetics-based equation that describes changes in plant biomass (carbon) as a function of photosynthesis minus losses to catabolic processes (e.g., photorespiration, respiration, non-grazing or 'natural' mortality) and grazing. In the generic CASM framework, photosynthesis is a nonlinear function of light, temperature, and nutrient (nitrogen, phosphorous, silica) availability. Growth is additionally modified by habitat factors, including accumulated degree days, water depth, current velocity, and salinity (for estuarine applications). Within the revised version of CASM, daily biomass (g-C/m^2) for each modeled population (B_i) of periphyton and macrophytes is determined by the following equation:

dB_i/B_idt = photosynthesis – photorespiration – dark respiration – sinking – natural mortality – grazing

or

$$
\begin{aligned}
dB_i/B_idt = \; & [Pm_i \; \{h(T), f(I), g(N), hmod\}(1-presp_i)] \\
& - dresp_i \; h(T) - (s_i + m_i) \\
& - \sum[h(T)\, B_j\, C_{ij}\, w_{ij}\, a_{ij}\, h_{ij}\, B_i\,)/\, (B_j + \sum w_{ij}\, a_{ij}\, h_{ij}\, B_i)]
\end{aligned}
$$

where, for population i,

B_i	biomass of population	gC/m^2
Pm_i	maximum photosynthetic rate of growth	1/d
$h(T)$	temperature dependence of photosynthesis	unitless
$f(I)$	light dependence of photosynthesis	unitless
$g(N)$	nutrient dependence of photosynthesis	unitless
$hmod$	habitat quality modifier of growth	unitless
$presp_i$	photorespiration rate	unitless
$dresp_i$	dark respiration rate	1/d
s_i	sinking rate (phytoplankton only)	1/d
m_i	mortality rate	1/d
C_{ij}	rate of consumption by grazer j	gC/m^2/d
w_{ij}	preference of consumer *j* for prey i	unitless
a_{ij}	assimilation of producer i by consumer j	unitless

h_{ij} handling efficiency unitless

The above bioenergetics equation describes growth dynamics for primary producer populations during optimal environmental conditions. The complex life history characteristics (e.g., overwintering stages, phenology) that partially determine opportunities for growth were represented using a degree-day approximation that defined the onset of population growth for each producer population. Estimates of population-specific degree-days for phytoplankton growth initiation were derived from a temporal analysis of population-specific productivity in Lake Texoma. The simple adoption of a degree-day approach permits the inclusion of life history traits that are not readily formulated in terms of bioenergetics.

Factors Regulating Photosynthesis

In CASM, photosynthesis (growth rate) is a nonlinear function of available light, water temperature, and inorganic macronutrients. In addition, effects of other physical-chemical factors in determining overall habitat quality and plant growth are modeled as an additional constraint on photosynthesis. The following sections describe these functional aspects of photosynthesis in CASM.

Light Saturation

The effects of light availability and light saturation on photosynthesis are modeled using a formulation derived by Thomann and Mueller (1987).

$$f_I(I) = I / I_s \cdot \exp(- (I / I_s) + 1.0)$$

where,

I = light intensity eins/m^2/d

I_s = light saturation intensity eins/m^2/d

Population-specific values of I_s define the relationship between light intensity and the value of $f_I(I)$ for the modeled plant populations. Daily values of $f_I(I)$ range between zero and 1.0 for each modeled population. For all modeled plant populations, the value of I is attenuated surface light intensity, I_0. While not included in the Lake Texoma CASM, emergent aquatic plants defined in other CASM applications are assumed to be exposed to unattenuated I_0.

In the field, the following factors play a role in producing complex underwater light regimes:

1) Variations in surface light intensity.
2) Wavelength-specific reflectance.
3) Attenuation of direct and diffuse light by water.
4) Absorption and reflectance of light by both dissolved and particulate organic and inorganic compounds.

In CASM, daily values of I_0 are attenuated by water color, suspended particulate organic carbon, phytoplankton biomass, and suspended inorganic matter. Constant attenuation coefficients (m^{-1} per mg/L) are defined for each of these particulates. A separate coefficient (m^{-1}) is defined for water color. The sum of these coefficients is used to attenuate surface light with increasing depth using a negative exponential function (i.e., Beer's Law). Light attenuation varies in relation to changing concentrations of phytoplankton, particulate organic carbon, and inorganic sediments.

Nutrient Limitation

CASM addresses the effects of nutrient availability on photosynthesis. The model includes effects of nitrate, phosphate, and silica (diatoms only). A multiplier of photosynthesis (0-1) is calculated using a simple Monod expression for each potentially limiting nutrient:

$$f_{NO3} = NO_3/(k_N + NO_3)$$

$$f_{PO4} = PO_4/(k_P + PO_4)$$

$$f_S = Si/(k_S + Si)$$

The values of NO_3, PO_4, and Si are the dissolved concentrations (mg/L) of nitrate, phosphate, and silica respectively. The k_N, k_P, and k_S values define the Monod (or 'half-saturation') constants, where f_{NO3}, f_{PO4}, and f_{Si} are equal to 0.5.

The overall effect of nutrient limitation is defined using a Liebig minimum approach:

$$f_N = \min(f_{NO3}, f_{PO4}, f_{Si})$$

Nutrient limitation is assumed to only affect photosynthesis rates of phytoplankton and periphyton. Population-specific values of k_N (algae and diatoms), k_P (algae and diatoms), and k_S (diatoms only) are required as input parameters to CASM. It is assumed that submerged aquatic and emergent vegetation derive nutrients from sediments and that these plants are seldom nutrient-limited; therefore, half-saturation constants are not required as input parameters for macrophyte or emergent plant populations modeled in the CASM.

CASM does not directly simulate accumulation of dissolved nutrients by modeled producers. Rather, the summed products of population-specific gross photosynthetic rates (gC/m^2/d) are converted to equivalent nutrient concentrations using a constant C:N:P stoichiometry (140:16:1) in order to derive the necessary conversions. It is recognized that *in situ* plant N:P ratios can vary in relation growth and nutrient availability, but the model does not currently address these variations. Decomposition and nutrient remineralzation by consumers are similarly governed by this assumed stoichiometry, with the understanding that variations in these ratios occur in aquatic ecosystems (e.g., Cross *et al.* 2005). These conversions to nutrient equivalents are used in calculating changes in available nutrients and preserving the mass-balance of NO_3, PO_4, and dissolved Si during model simulations.

Photorespiration

Photorespiration occurs in the presence of light, resulting in excretion of dissolved organic matter (carbon) and release of carbon dioxide. Photorespiration applies to modeled populations of phytoplankton, periphyton, and submerged aquatic plants in CASM. Photorespiration is modeled as a complement to light limitation of photosynthesis (Park *et al.* 2004) and is expressed as a population-specific fraction of photosynthesis:

$$\text{photorespiration} = (1.0 - \text{presp}_i) \cdot \text{Ps}_i$$

where,

presp_i is the photorespiration coefficient for population I (unitless).

Carbon excreted via photorespiration is added to the dissolved organic carbon pool. CASM does not simulate the complex aquatic chemistry of carbon dioxide. It is assumed that dissolved carbon dioxide concentrations are sufficient to support photosynthesis and that carbon dioxide produced as a result of respiration is simply returned to the dissolved pool or lost from the modeled system.

Dark Respiration

Dark respiration requires oxygen for the production of maintenance energy, a catabolic process that releases carbon dioxide. The biomass-equivalent of respiration is modeled within CASM as a nonlinear function of temperature. The formulation is

$$\text{dark respiration} = \text{dresp}_i\ \text{bp}_i$$

where,

dresp_i	= dark respiration rate	1/d
bp_i	= biomass of producer	g-C/m^2

Producer Habitat Quality

The bioenergetics-based approach underlying CASM describes ecological production in terms of physiological ecology and food web interactions. CASM addresses complex life histories and other biological/ecological aspects of plant growth by specifying additional habitat quality modifiers. Daily values of water depth, current velocity, and salinity define habitat quality for plants modeled by CASM. For example, rooted aquatic plants are not commonly observed where current velocities exceed 1 m/s. Moreover, in addition to light attenuation, water depth influences habitat quality for rooted aquatic plants and emergent aquatic plants in the CASM.

Simple trapezoidal functions are used to define the habitat quality modifiers (water depth, current velocity, salinity). The values of each function ranges from zero (uninhabitable) to 1.0 (optimal habitat). For each population, thresholds that define lower and upper values are estimated for each factor. Within ranges defined by the threshold values, two additional parameters define the range for which growth is optimal. In CASM, the habitat factors are calculated separately for depth, velocity, and salinity. As CASM was calibrated to Lake Texoma, the salinity modifier was equal to 1.0 (= no salinity) for all modeled populations. Salinity in this sense should not be confused with TDS or chloride as used in discussions among resource agencies and the public concerning “salinity” in relation to chloride management for Lake Texoma. Actual salinity values in Lake Texoma are generally around 1 ppt. The geometric mean value of these three habitat quality factors defines the overall habitat modifier (hmod) used in the equation for photosynthesis.

Periphyton Sloughing

In lotic systems, physical sloughing is an important process in determining periphyton biomass. Sloughing has been modeled in substantial detail within other ecological models (e.g., Park *et al.* 2004). The net result of these detailed formulations is a need for extensive site-specific parameter estimation. To include periphyton sloughing within CASM, it was assumed that periphyton is largely adapted to average current velocities in the inhabited system. As current velocity increases towards a value equal to its mean plus two standard deviations, a sloughing modifier increases exponentially in value from zero (no sloughing) to 1.0 (complete loss of biomass to sloughing). Current velocities used to calculate the sloughing parameter are input on a segment-specific basis using the results of a hydrodynamic simulation of the overall system. Similar approaches to modeling periphyton scouring based on deviations from average current velocity have also been used by Horner *et al.* (1990) and Saravia *et al.* (1998). To maintain mass balance, periphyton biomass lost due to sloughing enters the settled particulate organic carbon pool, which is subject to decomposition and advective transport.

The overall modeling objectives focus on production dynamics in pelagic areas of Lake Texoma. Therefore, periphyton, while included, was assumed to be a comparatively minor component of the Lake Texoma CASM. In addition, current velocities obtained from the University of North Texas modeling effort suggest that physical sloughing would not be commonly observed for Lake Texoma periphyton (Acevedo *et al.* 2004).

Consumer Population Dynamics

Biomass production for consumer populations modeled in the CASM is a function of consumption (ingestion and assimilation of food or prey) minus losses to respiration, excretion, natural mortality, and predation. These processes are nonlinear functions of temperature (respiration) and food availability (consumption, predation). Modeled consumer populations are distinguished by population-specific rates of feeding, respiration, excretion, locomotion costs, prey preferences, prey assimilation, and temperatures for maximum feeding and respiration. The daily value of biomass (g-C/m^2) for each modeled population (B_i) of zooplankton, aquatic insects, benthic invertebrates, and fish is determined by the following equation:

$dB_i/B_i dt$ = consumption – respiration – excretion – egestion – natural mortality – (specific dynamic action) – predation

or

$$\begin{aligned} dB_i/B_i dt = \; & \sum[(Cm_i\, h(T)\, w_{ij}\, a_i\, h_{ij}\, B_j)/(B_i + \sum w_{ij}\, a_{ij}\, h_{ij}\, B_j) - (u_i + f_i + rsda_i)] \\ & - r_i\, h(T) - m_i \\ & - \sum[(Cm_j\, h(T)\, w_{ij}\, a_{ij}\, h_{ij}\, B_j)/(B_j + \sum w_{ij}\, a_{ij}\, h_{ij}\, B_i)] \end{aligned}$$

where, for population i

B_i	biomass	gC/m^2
Cm_i	maximum consumption rate	1/d
$h(T)$	temperature dependence of consumption	unitless
w_{ij}	preference of consumer i for prey *j*	unitless
a_{ij}	assimilation efficiency for consumer *i* and prey *j*	unitless
h_{ij}	handling efficiency	unitless
r_i	standard respiration rate	1/d
u_i	excretion rate	1/d
f_i	egestion rate	unitless
m_i	natural mortality rate	1/d
$rsda_i$	specific dynamic action (fish only)	unitless

The consumption term in this equation calculates grazing or predation as a normalized product of predator and prey biomass. Predator and prey biomass are modified by the preference of predator *i* for prey *j* (w_{ij}), the assimilation of prey *j* by predator *i* (a_{ij}), and a handling efficiency for predator *i* consuming prey *j* (h_{ij}) (DeAngelis *et al.* 1975; DeAngelis *et al.* 1989). The food consumed by each population is either assimilated, excreted, or egested. The egested (or unused food) fraction is remineralized by a temperature-dependent decomposition process. Recycled nutrients are added to the dissolved nutrient pools by assuming a constant C:N:P stoichiometry for plant and animal organic matter.

The mathematical nature of the consumption term introduces a density dependence in the overall production of consumer populations in CASM (DeAngelis *et al.* 1975). At low predator abundances, the biomass of the available predator determines the flux of ingested material into the predator populations. As the predator biomass increases, the flux becomes increasingly determined by the biomass of the prey. Such density dependence can influence the growth of individual populations and structure of complex food webs in CASM.

CASM includes a standard respiration rate and specific dynamic action (fish only). The standard (or basal) respiration is a nonlinear function of water temperature; postulated increases in respiration as a function of crowding (Park *et al.* 2004, Kitchell *et al.* 1974) are not addressed in CASM. Specific dynamic action is equal to the energetic costs of digestion encumbered by fish. Total respiration is the sum of standard respiration and specific dynamic action:

$$Rs_j = [(f_T(T) \cdot Rmax_j) \cdot B_j] + RSDA_j$$

where,

$f_T(T)$	= temperature dependence of standard respiration	unitless
$Rmax_j$	= maximum standard respiration rate	gC/gC/d
$RSDA_j$	= specific dynamic action	unitless
B_j	= biomass of consumer population j	gC/m^2

and

$$RSDA_j = r_{SDAj} \Sigma (Cs_j - F_j)$$

Specific dynamic action is calculated as a fraction (r_{SDAj}) of consumption minus egestion (i.e., assimilation) for each fish population j.

Consumer metabolism includes the excretion of carbon and its equivalent in nitrogen, phosphorus, and silica. Excreted carbon enters the dissolved organic carbon pool, and excreted nutrients enter the respective dissolved nutrient pools. The fraction of biomass lost to excretion is determined by population-specific estimates of consumption and a population-specific constant excretion rate $r_{Uj.}$

$$U_j = r_{Uj} \cdot \Sigma (Cs_j - F_j)$$

The amounts of N and P added to the dissolved nutrient pools per unit carbon are based upon a constant stoichiometry of C:N:P (and Si).

Consumer Habitat Quality

Habitat quality preferences were also included in CASM for consumer populations. Environmental factors that determine habitat quality include dissolved oxygen concentration, water depth, current velocity, and salinity. Similar to the primary producer populations, trapezoidal functions and the defining parameters were used to quantify habitat quality [range of 0 to 1] in relation to daily values of all factors (except salinity in the case of Lake Texoma). Population-specific preferences and limiting values were derived from the technical literature to define these functions for the modeled consumer populations (e.g., Jowett and Richardson, 1990). In addition to temperature modifiers, the geometric mean of daily habitat modifiers was used to reduce consumption from its maximum rate.

Parameter Estimation

With a food web structure representative of Lake Texoma and determined appropriate for addressing the modeling objectives, the next step in developing the Lake Texoma CASM involves estimation of the necessary bioenergetics parameter values and other input data required by the model. The following sections describe the derivation of the Lake Texoma CASM inputs.

Abiotic Environmental Factors

In the CASM, production of biomass by producer and consumer populations is strongly influenced by environmental factors including surface light intensity, light extinction, water temperature (including thermal stratification), water depth, current velocity, nutrient availability, dissolved oxygen, total suspended solids, particulate organic matter (suspended and settled), salinity, and surface wind velocity. Development of the Lake Texoma CASM required values that defined the daily euphotic zone depth as an additional environmental factor.

The following sections describe the derivation of the individual abiotic factors as they were implemented for the Lake Texoma CASM. This discussion refers to the 365 daily values of used to develop the Lake Texoma CASM reference simulation.

Light

The relationships between surface light intensity and the availability of light as a function of water depth are key factors in determining the trophic dynamics of the Lake Texoma CASM. The important hypothesis concerning chloride management in Lake Texoma is that reduced chloride will lead to decreased sedimentation rates, increased turbidity, and reductions in primary productivity. Reductions in primary productivity will propagate throughout the Lake Texoma food web and ultimately reduce the production of striped bass, a valued recreational fishery in the reservoir. Therefore, particular attention was paid in the implementation of the components of the CASM that determine light availability to primary producers, mainly phytoplankton.

The CASM requires daily values of photosynthetic active radiation (PAR, wavelengths 400 to 900 nm) incident at the water surface, I_0. The units of PAR used in the model are einsteins/m^2/d. These values can derive from field measures, if available. Alternatively, the daily values can be calculated (external to the CASM) as a function of site latitude (i.e., solar declination) and corresponding time-varying day-length.

Surface Light Intensity

Thirty years (1961-1990) of hourly measures of surface light intensity were obtained for a location near Wichita Falls from the National Solar Radiation Database (http://rredc.nrel.gov/solar/old_data/nsrdb). The Wichita Falls station provided a long-term period of surface light intensities measured at a latitude similar to Lake Texoma. These data were used to calculate daily average surface light intensities required by the CASM (Figure 3). The same incident light values were used for all six modeled locations. The data compare favorably with the annual average daily value of 44 einsteins/m2/d reported by Doyle and Baugher (2002).

Light Attenuation

Light attenuation is modeled as a function of depth in the CASM according to

$$I_z = I_0 \exp(-\lambda z)$$

Where:

I_z	light intensity at depth z
I_0	surface light intensity
λ	light extinction coefficient (m^{-1})
z	water depth (m)

Several factors influence the value of λ including the light extinction properties of pure water, the color of water in the system of interest, and both organic (e.g., dissolved organic carbon, suspended particulate organic carbon, phytoplankton) and inorganic particulate matter (e.g., suspended inorganic sediments) (e.g., Lee and Rast, 1997). To simulate the contribution of these factors to light extinction, the previous equation can be re-written as

$$I_z = I_0 \exp(-\lambda_s[C_{DOC} + C_{POC} + C_{PHYTO} + C_{TIS} + C_{TDS}] + \lambda_w + \lambda_c)\ z$$

where,

λ_s	light extinction per unit concentration of attenuating constituent
C_{DOC}	concentration of dissolved organic carbon (gC/m^3)
C_{POC}	concentration of particulate organic carbon (gC/m^3)
C_{PHYTO}	concentration of phytoplankton carbon (gC/m^3)
C_{TIS}	concentration of total suspended inorganic solids (g/m^3)
C_{TDS}	concentration of total dissolved solids (g/m^3)
λ_w	light extinction coefficient of pure water (m^{-1})
λ_c	light extinction coefficient of water color (m^{-1}).

For applications to freshwater systems, the relative contribution to light extinction by pure water is considered negligible and this term is not included in the model. Extinction by water color is modeled as a system-specific constant value. Importantly, the concentrations of the other factors that influence light extinction are dynamic in space and time and modeled light extinction coefficients vary correspondingly in the CASM.

Implementation of the light extinction calculation requires both estimates of the varying concentrations of these components and the mass-specific attenuation by each component Based on Park *et al.* (2004), attenuation of surface light was determined by the following extinction coefficients:

- Dissolved organic carbon 0.03 (m^{-1} per mg/L),
- Particulate organic carbon 0.12 (m^{-1} per mg/L),
- Phytoplankton biomass 0.014 (m^{-1} per gC/m^2),
- Total suspended inorganic solids 0.03 (m^{-1} per mg/L), and
- Water color 0.04 – 0.10 (m^{-1}).

For each modeled day and location, the simulated values of these five parameters were used to calculate a light extinction coefficient (m-1).

Water Temperature

The key bioenergetics-based growth processes (e.g., photosynthesis, consumption, respiration) included in the model are influenced by water temperature. Therefore, realistic daily water temperatures are important for model performance. Monthly values of water temperature were measured at multiple depths during 1996-1997 at the modeled locations (Waller *et al.* 2001). Temperature values were averaged over the sample depths for each month and location. Daily temperature values required by the Lake Texoma CASM were interpolated from the monthly values.

To represent temporal variations in daily temperatures, values at the beginning and end of each month were used to define a uniform distribution of temperatures for each month. Daily values were sampled at random from these distributions. This approach appears to provide more realistic time series of temperature values than simple linear interpolation. Figure 4 illustrates the daily temperature values for the five locations included in the Lake Texoma CASM.

The 1996-1997 seasonal water temperatures ranged from ~5 – 30 °C (Figure 4). Variations in daily temperatures among the modeled locations were approximately 5 °C, depending on the locations and time of year.

Nutrients

Waller *et al.* (2001) was used as a source of nitrogen and phosphorus inputs required by the model. Surface and bottom samples were collected monthly at the modeled locations, unless thermal stratification was measured. During stratification, samples were collected 1 m above and below the chemocline. Lake Texoma does not exhibit a thermocline as defined in standard limnological terms (Matthews and Hill 1982).

The same approach used for developing water temperatures based on samples of uniform distributions of reported monthly concentrations (mg/L) was used to generate the daily dissolved inorganic N (DIN) and P concentrations (DIP) needed to execute the Lake Texoma CASM.

Dissolved Inorganic Nitrogen

Seasonal concentrations of DIN based on 1996-1997 data ranged ~0.4 – 1.1 (mg/L) across all modeled locations in Lake Texoma (Figure 5). The highest concentrations were measured for the Red River locations in late fall, winter, and late spring – early summer. Comparatively lower concentrations were reported for the Washita River locations, although the seasonal patterns were similar to the Red River values. In general, the seasonal low DIN values occurred during the early spring and mid-summer periods. Concentrations near the dam were essentially intermediate between the Red River and Washita River locations, except for late fall when the highest concentrations were measured near the dam. DIN concentrations for the modeled location in the Big Mineral Arm were generally similar to the values reported for the Washita River.

Dissolved Inorganic Phosphorus

Concentrations of DIP developed for the modeled locations demonstrated a seasonal pattern similar to DIN (Figure 6). Concentrations ranged from ~0.02 – 0.11 (mg/L). However, the differences between the seasonally higher and lower DIP concentrations were less pronounced compared to DIN. The DIP concentrations were typically higher for the Red River modeled locations. The DIP concentrations in the Washita River stations were generally the lowest, especially for station 24.

Silica

Concentrations of dissolved silica were developed from data reported for a single location in the Big Mineral Arm of Lake Texoma. The available silica data proved to be sparse. Therefore, an empirical relationship was developed between reported flows and silica concentrations in the Big Mineral Arm. Flows were determined from the University of North Texas physical model for the other Lake Texoma CASM locations (Acevedo *et al.* 2004). These flows were used to construct daily concentrations of silica for the remaining modeled locations. The resulting input values for silica appear unrealistically high. However, in relation to the modeling objectives, silica is important only for the simulated growth of diatoms. The implication of the limited availability of silica data is that the modeled growth of diatoms is not likely to be limited by silica. This shortcoming represents one of the model limitations. At the same time, diatoms do not contribute significantly to the overall biomass production of the Lake Texoma phytoplankton (Waller *et al.* 2001).

Total Suspended Inorganic Solids

Total suspended solids data were obtained from Waller *et al.* (2001). The average daily values of suspended total inorganic solids were estimated by assuming that a constant fraction (10%) of reported TSS data was organic carbon. This fraction was subtracted from each daily TSS value to estimate the concentration of total inorganic solids (TIS). TIS concentrations influence the attenuation of surface light intensity in the CASM.

Total Dissolved Solids

The modeling objectives focus on the food web implications of reductions in TDS as the result of chloride management actions. Therefore, the quality and availability of TDS data that describe pre-project conditions (i.e., prior to aggressive chloride management in the watershed) were central to the modeling effort. Pre-project conditions were developed based on total dissolved solids (TDS) data for various sampling dates in 1975-1978 for relevant Lake Texoma stations obtained from the USEPA STORET database (www. epa.gov/STORET). STORET data from these years were presumed to usefully represent pre-project TDS concentrations in Lake Texoma in support of the modeling effort.

Values obtained from the STORET database were compared with independent TDS values obtained at USGS locations near Terral, OK and Gainesville, TX. Data were obtained for 1966 at the Terral station and for 1966-67 at the Gainesville station. These data were presumed to represent pre-project conditions at these locations. As might reasonably be expected, the concentrations observed at Terral and Gainesville were on average approximately twice the concentrations reported for the riverine arm of the Red River in Lake Texoma (i.e., Station 1). The concentrations measured at Terral and Gainesville ranged between 1 and 3 times the values developed for Station 1.

Figure 7 illustrates the temporal pattern of pre-project TDS concentrations developed for the modeled locations in Lake Texoma. Concentrations ranged from ~600 – 1,900 (mg/L) across all the modeled locations. Concentration in the Red River and the Big Mineral Arm are greatest in the early spring and late fall. The concentrations for the area near the dam appear to reflect the seasonal contributions from the upriver locations and reach their seasonal maximum values slightly after the peaks in the Red River and Washita River locations. Values in the Big Mineral Arm are greatest during the late summer months when they exceed the TDS values for the other locations. The greatest range of variation in daily values is apparent for Station 1 in the Red River, especially between model year days ~230-260. The lowest TDS concentrations are consistently observed for Station 24 in the Washita River.

Suspended Particulate Organic Carbon

The allochthonous input of particulate organic carbon (POC) were derived by assuming that approximately 10 percent of the total suspended solids (TSS) constituted particulate organic carbon. POC as a percentage of TSS can vary substantially (Jones and Knowlton

2005, Cole *at al.* 1990). Values ranging from ~2 – 35% have been reported for rivers and reservoirs (e.g., Kao and Liu 1996, Trefry *et al.* 1994). The value of 10% used in developing the CASM is similar to the 8.5% value determined by Shantz *et al.* (2004) and appears as a reasonable value in relation to the overall modeling objectives. CASM includes biogenic sources of POC generated during model simulation as the result of egestion by consumers and plankton mortality. Particulate organic carbon serves as a food source for certain consumer populations in the CASM. Suspended POC also contributes to the attenuation of surface light intensity in the model. Suspended and sedimented POC are subject to decomposition by modeled planktonic and sediment bacteria populations.

Euphotic Zone Depth

Monthly measures of the depth of the euphotic zone were reported for each of the modeled locations (Waller *et al.* 2001). These data were used to derive daily values of the depth of the euphotic zone. Daily phytoplankton production was calculated at 1-m depth intervals from the water surface to the daily input euphotic zone depth at each location.

Surface Wind Velocity

Surface wind velocities are used by the CASM to calculate external inputs of oxygen across the air:surface water interface. Hourly values of 2007 surface wind velocities were obtained for a location near Wichita Falls from the National Climatic Data Center (http://www.ncdc.noaa.gov/oa/ncdc.htm). The Wichita Falls data were chosen mainly in terms of convenience and electronic accessibility given that this data source was also used to generate daily values of light intensity. The hourly wind data were used to calculate daily average wind velocities used by the Lake Texoma CASM.

Biotic Components

The technical ecological literature was used to derive values of the bioenergetics parameters and define trophic interactions for the modeled populations of producers and consumers in the Lake Texoma CASM. Derivation of model bioenergetics parameter values was based on the assumption that physiological process rates reported for other lakes and reservoirs applied to the populations included in the Lake Texoma CASM. Similarly, the trophic interactions including prey preferences observed in other aquatic systems were assumed to relevant for Lake Texoma.

The overall strategy used to estimate model parameters was to select values within reported ranges that produced biomass values consistent with available data for corresponding Lake Texoma producers and consumers. Initial values of were developed for each modeled population based on published information. The initial values of individual parameters were systematically increased or decreased in repeated simulations to determine the combination that provided the best attainable match to the limited available data (e.g., Waller *et al.* 2001, 2002). The resulting model calibration to Lake Texoma data represents a hypothesis concerning trophic dynamics within this reservoir. A usefully calibrated Lake Texoma CASM provides a mechanistic basis for examining the implications of chloride management on subsequent changes in Lake Texoma food webs.

Phytoplankton

Derivation of the growth parameters for the modeled phytoplankton populations was based on published values of photosynthesis (growth rate), optimal temperatures, light saturation, photorespiration, dark respiration, nutrient limitation, sinking, and mortality (Table 1).

Photosynthetic Rates

Maximum photosynthetic (growth) rates vary widely among phytoplankton, although values within a range of ~1-3 d^{-1} are commonly reported (e.g., Saravio *et al.* 1998, Robarts1987, Lehman *et al.* 1975, Goldman and Carpenter 1974, Sorokin and Kraus, 1958). In calibrating the CASM to Lake Texoma phytoplankton biomass, selected growth rates ranged from 1.00 – 2.15 for the phytoplankton. Chlorophytes and cyanophytes had the higher growth rates, followed by microflagellates, and diatoms (Table 1).

Optimal Temperatures

The seasonal pattern of water temperatures (Figure 4) and general observations of temperature tolerances for the modeled taxa were used to define temperature regimes conducive to growth (Table 1). Diatoms were assumed to grow at cooler temperatures corresponding to their observed periods (e.g., early spring, late fall) of maximum growth. Cyanophytes were assumed to grow optimally under warmer conditions associated with summer bluegreen algal blooms. Optimal temperature regimes for chlorophytes and microflagellates were defined as intermediate to the regimes developed for diatoms and bluegreen algae.

The bioenergetics-based governing equations have proven useful in describing patterns of plant population growth (e.g., Park *et al.* 2004). However, these equations cannot address life history traits that are not easily formulated in terms of energetics (e.g., sexual reproduction, overwintering stages, cysts, etc.). Such traits can influence the timing of entry of a seasonally nascent population into an actively growing phytoplankton community. To account for these kinds of life history characteristics and their influence on growth, the concept of accumulated degree-days was used to model phytoplankton production in the Lake Texoma CASM. The degree-day concept has proven useful in simulating the phenology of aquatic plants (Best *et al.* 2005). In CASM, degree-days are calculated as the simple running sum of daily water temperature values. Threshold values of accumulated degree-days were developed for each modeled population: 200 for diatoms, 1,300 for both microflagellates and bluegreens, and 2,000 for chlorophytes.

When the accumulated degree-days exceed a population threshold, the population is permitted to grow according to its bioenergetics parameters (Table 1). Until that time, it is assumed that the population is seasonally dormant. Specification of these population-specific thresholds permits more realistic description of the temporal pattern of phytoplankton growth using the CASM without having to model complex life histories that are not readily defined in terms of energy (biomass).

Light Saturation Constants

Values were derived for the modeled phytoplankton populations based on Robarts (1987) and Lehman *et al.* (1975). Diatoms were assumed to exhibit comparatively lower light saturation (3 einsteins/m^2/d), commensurate with the lower incident light availability in early spring and late fall. Bluegreen algae were assigned the highest value of 6 einsteins/m^2/d, which is characteristic of higher light intensities associated with their generally observed seasonally maximum growth and vertical distribution nearer the surface of the water column. A slightly lower value (5 einsteins/m^2/d) was derived for microflagellates and chlorophytes.

Photorespiration

Birmingham *et al.* (1982) provided estimates of photorespiration for diatoms, green algae, blue-green algae, and other algae. Their estimates ranged from 1.3 to 8.7% of gross photosynthesis. Photorespiration values ranging from 3.5 – 5.5% were determined to provide the best agreement between modeled and measured phytoplankton biomass in the calibration to Lake Texoma data (Table 1). These values were well within the range reported by Birmingham *et al.* (1982).

Dark Respiration

Buzzelli *et al.* (2000) used basal (dark) respiration rate of 0.1/d in a model of algal production in the Florida Everglades. Burris (1977) measured somewhat higher rates of dark respiration of 0.11/d to 0.42/d of gross photosynthesis for several marine algae. Values of 0.2 – 0.35 d-1 assigned to modeled populations contributed to the calibration of phytoplankton to Lake Texoma phytoplankton biomass data (Table 1).

Nutrient Limitation

Values of half-saturation (Monod) constants were assigned to model algae populations based on reported values for silica (Paasche, 1973), nitrogen (Son and Fujino, 2003; Reuter *et al.* 1986; Lehman *et al.* 1975; Carpenter and Guillard, 1971), and phosphorus (Son and Fujino, 2003; Bothwell, 1985; Bothwell, 1988; Buzzelli *et al.* 2000; Lehman *et al.* 1975).

Sinking Rates

Phytoplankton sinking rates were derived from values reported in Lehman *et al.* (1975) and Titman and Kilham (1976) for a variety of freshwater taxa. The greatest rate was defined for the modeled diatom population. Microflagellates and cyanophytes were assigned lower sinking rates on the basis of smaller size and gas vacuoles (cyanophytes). Chlorophytes were assigned a sinking rate greater than cyanophytes, but less than diatoms (Table 1).

Prymnesium parvum

Because of its specific importance in the original modeling objectives, parameter estimation is described separately for *P. parvum* (Table 1).

Cooler water temperatures, lower light intensities, higher salinities, and excessive nutrient concentrations appear to favor the growth of *P. parvum* (e.g., OWRB 2006). The growth parameters derived for the modeled golden algae reflect these observations. Water temperatures that permitted growth were defined as a range of 2-18 °C, with optimal growth occurring between 10 and 14 °C.

Reported requirements for higher nutrient conditions were used to derive a comparatively high half-saturation constant for N (0.60 mg N/L) assigned to this modeled algal population.

Highest cell densities were reported for lower light availabilities (e.g., Secchi depths of 0.1 – 0.3 m). Correspondingly, the light saturation constant was defined as 1 einstein PAR/m2/d for modeled *P. parvum*.

Heterotrophy has been reported for marine populations of *P. parvum* (e.g., Martin-Cereceda *et al.* 2003). However, data sufficient to describe grazing by *P. parvum* on potential prey items, including bacteria and small diatoms were not available. It was assumed that heterotrophy was not a significant contributor to modeled growth of *P. parvum*.

In addition, it was assumed that *P. parvum* was not subject to grazing by zooplankton or herbivorous fish. This assumption required the specification of a lower maximum growth rate of 0.5 d^{-1} (compared to the other phytoplankton) in order to avoid unrealistically high biomass values for this population.

Consumer Populations

The same approach to estimating phytoplankton model parameters was used to derive population-specific values of parameters for the zooplankton, benthic invertebrates, and fish included in the Lake Texoma CASM (Table 2).

Consumption Rates

Zooplankton feeding rates were derived primarily from Peters and Downing (1984), Schindler (1968), and Richman and Dodson (1983). Consumption rates for benthic invertebrates were based on allometric interpolations derived from zooplankton feeding rates. Maximum consumption rates for modeled populations of freshwater drum and smallmouth buffalo were estimated using data reported by Hanson *et al.* (1997). Consumption rates for the modeled silversides was derived from results obtained for smelt (Lantry and Stewart 1993). Values estimated for herring by Rudstam (1989) were used to derive consumption rates for threadfin and gizzard shad. The work by Hartman and Brandt (1995) on striped bass was used to estimate consumption rates for modeled YOY, juvenile, and adult striped bass (Table 2). The striped bass equations (i.e., Hartman and Brandt 1995) were also used to estimate maximum consumption for white bass. Blanc and Margraf (2002) provided the information used to estimate maximum consumption rate for blue catfish.

Optimal Temperatures

The basic ecology of modeled consumer populations and the seasonal pattern of daily water temperature input to CASM were used to define temperature preferences for the consumer populations (Table 2). Preferences were defined in such a way that commonly observed seasonal patterns of biomass resulted for the modeled consumers.

Standard Respiration Rates

Respiration rates for zooplankton were derived using results from Schneider (1992) and Schindler (1968). Benthic invertebrate respiration rates were estimated based on Hamburger and Dall (1990). Standard respiration rates for modeled freshwater drum and smallmouth buffalo were estimated using data reported by Hanson *et al.* (1997). Respiration rates for the modeled silversides was derived from results obtained for smelt (Lantry and Stewart 1993). Values estimated for herring by Rudstam (1989) were used to derive respiration rates for threadfin and gizzard shad. The work by Hartman and Brandt (1995) on striped bass was used to estimate respiration rates for modeled YOY, juvenile, and adult striped bass (Table 2). The striped bass equations (i.e., Hartman and Brandt 1995) were also used to estimate maximum respiration rate for white bass. Blanc and Margraf (2002) provided the information used to estimate maximum respiration rate for blue catfish.

Specific Dynamic Action

Values of specific dynamic action for modeled fish populations were estimated using data reported by Hanson *et al.* (1997). The previously identified studies used to estimate maximum rates of consumption and respiration for the modeled fish populations in the Lake Texoma CASM were also used to estimate rates of specific dynamic action for these same populations.

Egestion Rates

Estimates of egestion rates for modeled fish populations were derived using data reported by Hanson *et al.* (1997). The previously identified studies used to estimate maximum rates of consumption and respiration for the modeled fish populations in the Lake Texoma CASM were also used to estimate egestion rates of for these same populations.

Excretion Rates

Zooplankton excretion rates were estimated using the results of Mayzaud (1976), Hargrave and Geen (1968), and Wen and Peters (1994). Excretion rates for modeled fish populations were estimated using data reported by Hanson *et al.* (1997). Excretion by benthic invertebrates was interpolated using the rates obtained for zooplankton. The previously identified studies used to estimate maximum rates of consumption and respiration for the modeled fish populations in the Lake Texoma CASM were also used to estimate excretion rates for these same populations.

Mortality Rates

Zooplankton mortality rates were based on studies by Eiane *et al.* (2002) and Ohman and Hirche (2001). Benthic invertebrate mortality rates were extrapolated from zooplankton mortality rates. Mortality rates for modeled fish populations were based on estimates for 30 lotic and lentic fish species used for modeling effects of commercial navigation on fish entrainment (Bartell and Campbell, 2000).

Trophic Interactions

CASM permits specification of grazing and predator-prey interaction among the modeled populations of aquatic producers and consumers (DeAngelis *et al.* 1989). A non-zero prey preference value, w_{ij}, is specified for each trophic interaction between prey i and consumer j. The magnitudes of the w_{ij} values is somewhat arbitrary, because the calculation of each trophic interaction involves a normalization across all prey items consumed by a grazer or predator. However, in practice, values of w_{ij} range from 0 to 1, with higher values indicating increased preference. If available, diet composition data can be used to derive w_{ij} values. Qualitative descriptions of diet preferences can be used with professional judgment to develop these input parameters as well. Current versions of the CASM assume that specified prey preferences are constant; ontogenetic shifts in diet are not included, unless different life stages of consumers are modeled explicitly.

Information similar to the following reports was used in combination with professional judgment to develop the prey preference data used in the Lake Texoma CASM:

- Gizzard shad feed preferentially on zooplankton, while the threadfin shad diet consists of approximately half phytoplankton and half zooplankton (Etnier and Starnes 1993). Gizzard shad can switch to detritus as an alternate food source (Gido and Matthews 2000).

- Younger striped bass feed on zooplankton, while juvenile and adult striped bass feed primarily on shad (Etnier and Starnes 1993).

- Freshwater drum are bottom-oriented in their feeding habits (Etnier and Starnes 1993). While substantial amounts of fish (e.g., shad) are consumed, the drum emphasize aquatic insects, amphipods, and mollusks in their diet.

Assimilation Efficiencies

CASM also permits specification of assimilation efficiency for each grazing or predator-prey interaction. The a_{ij} parameters are analogous to the prey preference parameters, except these parameters quantify the fraction of consumer prey converted to new consumer biomass. In CASM, the assimilation efficiency multiplies the calculated amount of food ingested minus losses to egestion. The a_{ij} parameters characterize food quality derived from published values (Richman and Dodson 1983; Lampert 1987). The a_{ij} parameters are constant during a CASM simulation.

Prey Handling Efficiencies

CASM also permits specification of a prey handling efficiency for each grazing or predator-prey interaction. The h_{ij} parameters are analogous to the prey preference parameters, exceptt these parameters quantify the fraction of prey not ingested as a result of behavioral or mechanical inefficiencies that make the prey more difficult to ingest. In CASM, the assimilation efficiency multiplies the calculated amount of food ingested. Handling efficiencies were not considered in modeling the trophic interactions among Lake Texoma consumers and their prey items. It was recognized that smaller individuals of the inland silversides might encounter difficulties in preying upon the invasive *Daphnia lumholtzi* (Lienesch and Gophen 2005). However, it was determined that this cladoceran is only seasonally abundant and not a major contributor to overall zooplankton production in Lake Texoma (Work and Gophen 1999). All values of h_{ij} were therefore set equal to 1.0, which means that handling efficiencies were 100% - that is, no prey biomass was lost to ingestion because of difficulties associated with ingestion.

The prey preferences, assimilation efficiencies, and handling efficiencies are included in a single input file for the Lake Texoma CASM: web_casmLT.dat.

Water Quality Parameters

CASM also simulates daily changes in concentrations (mg/L) of dissolved oxygen, dissolved inorganic phosphorus, dissolved inorganic nitrogen, dissolved organic carbon, and particulate organic carbon. The equations describing the dynamics of these model state variables are:

Dissolved oxygen (DO)

dDO/dt = input from plant photosynthesis – uptake by plant and consumer respiration ± advective transfer ± air:water surface exchange

Dissolved inorganic phosphorous (DIP)

dDIP/dt = allochthonous input + remineralization by consumers – uptake by plants ± advective transfer

Dissolved inorganic nitrogen (DIN)

dDIN/dt = allochthonous input + remineralization by consumers – uptake by plants ± advective transfer

Dissolved organic carbon (DOC)

dDOC/dt = release from plants + excretion by consumers + decomposition of POC – decomposition of DOC ± advective transfer

Particulate organic carbon (POC)

dPOC/dt = plant mortality + consumer mortality + egestion by consumers – decomposition of POC ± advective transfer

Temperature-Dependent Processes

One recent revision to CASM was the replacement of the previously used temperature function (i.e., Bartell *et al.* 1999; Bartell *et al.* 2000) with a more generalized function developed by Thornton and Lessem (1978). The original function derived from O'Neill *et al.* (1972) overestimated physiological rates at low temperatures for populations with low values of Topt – a defining parameter for the temperature function. While requiring more parameters, the formulation developed by Thornton and Lessem (1978) results in more accurate predictions over broader ranges of preferred temperatures.

The function is a product of two sigmoid relationships: one curve (gcta) defines increasing functional response to increasing temperature; the second curve (gctb) describes decreasing response for sub-optimal temperatures.

$$f_T(T) = \text{gcta} \cdot \text{gctb}$$

where,

$$\text{gcta} = (\text{xk1} \cdot \text{t4}) / (1.0 + \text{xk1} \cdot (\text{t4} - 1.0))$$

$$\text{gctb} = (\text{xk4} \cdot \text{t6}) / (1.0 + \text{xk4} \cdot (\text{t6} - 1.0))$$

and,

$t4 = \exp[t5 \cdot (T - te1)]$

$t5 = tt5 \cdot \ln[xk2 \cdot (1.0 - xk1) / (0.02 \cdot xk1)]$

$t6 = \exp[t7 \cdot (te4 - T)]$

$t7 = tt7 \cdot \ln[xk3 \cdot (1.0 - xk4) / (0.02 \cdot xk4)]$

$tt5 = 1.0 / (te2 - te1)$

$tt7 = 1.0 / (te4 - te3)$

For each population, four pairs of temperature and temperature response values define the overall shape of the f_T: (te1, xk1), (te2, xk2), (te3, xk3), and (te4, xk4). Values of te1, te2, te3, and te4 are in units of °C. Values of xk1, xk2, xk3, and xk4 range from zero to 1.0.

To reduce the number of required model parameters while maintaining a realistic description of water temperature effects on plant respiration, the following equation was used:

$$h(T) = V^x * \exp(x[1 - V])$$

where $V = (T_m - T)/(T_m - T_o)$. T_m defines an upper lethal temperature specified for each population and process rate; T_o specifies the temperature at which the maximum rate (i.e., photosynthesis, respiration, and feeding) is observed for each population. The value of x is a non-linear scalar of the ln $Q_{10}\,(T_m - T_o)$, where the value of $Q_{10} = 2.0$.

Long-Term Environmental Scenarios

The longer term implications of chloride management on food web dynamics within Lake Texoma were examined within a framework of projected environmental variability over a 50-y planning horizon. Historical data were used to construct 50-y data sets that contained daily values of environmental input parameters required by the model. These initial scenarios were intended to examine the implications of historical patterns of environmental variability and do not address changes anticipated in relation to climate change.

Longer-term implications of chloride management need also to be addressed within the ongoing sedimentation that is gradually filling-in Lake Texoma and correspondingly reducing the availability valued aquatic habitat. Therefore, scenarios of sedimentation were constructed and included in the overall assessment of chloride management on food web dynamics, and particularly striped bass production, within the impoundment.

Physical-Chemical Factors

Average daily light surface intensity was (ein/m2/d) calculated for years 1994 – 2007. Inter-annual variability was defined as the percent deviation (+/-) of the daily average for each year from the overall average of 69.48 ein/m2/d determined for the 1994 – 2007 period. The annual deviations ranged from ~ (+/-) 1 to 8 percent of the 1994-2007 average. The percent deviations were used to define a data set that was randomly sampled (with replacement) to generate 49 multipliers (i.e., percent deviations). These multipliers were used to modify the light intensity and other baseline environmental factors (except sedimentation) to generate 49 different annual environmental scenarios. These 49 scenarios were added in sequence to the baseline environmental scenario to produce a 50-y time series of daily values of environmental input parameters required by the CASM.

In developing the environmental scenarios, it was assumed that the following CASM input parameters were positively correlated with surface light intensity:

- depth of the euphotic zone
- surface wind velocity
- water temperature – epilimnion and hypolimnion

The baseline year values of these parameters were multiplied by the same factor that adjusted daily surface light intensity.

Years with above average light intensity were assumed to be drier with corresponding reductions in flows and material inputs to Lake Texoma. The following environmental inputs were negatively correlated with light intensity:

- total depth
- current velocity
- salinity
- concentrations of total dissolved solids, chloride, particulate organic carbon, total inorganic solids, dissolved inorganic N, dissolved inorganic P, and silica

Daily values of these input parameters were proportionally decreased by higher than average light surface light intensity or increased in relation to lower than average light intensity according to (1.0 + (1.0 – x)), where x is the multiplier determined for surface light intensity.

Sedimentation

Accretion of sediments since completion of Lake Texoma in 1944 has resulted in loss of aquatic surface area, morphometric changes, fragmentation of lacustrine habitats, isolation of coves, and establishment of terrestrial vegetation on deposited lands (Patton and Lyday 2008). Aquatic habitat modifications as the result of sedimentation can impact fish community structure and fish productivity.

Scenarios for simulating accretion of sediments and corresponding losses of aquatic habitat were developed for Lake Texoma. Sediments are not explicitly modeled in the CASM. Sedimentation of the impoundment was modeled simply as a time-varying reduction in total depth and segment area for each of the six modeled locations. A constant annual percentage reduction in depth and area (0.36 % y^{-1}) was estimated from changes in Lake Texoma storage capacity projected for 2044 based on the original reservoir capacity and results of a 2002 survey (D. Tomlinson, USACE, pers. comm.).

Reference Simulation

The values of the baseline abiotic environmental variables were used in combination with the modeled aquatic populations and their associated calibrated bioenergetics parameters to simulate one year of ecological production dynamics for each of the locations included in the Lake Texoma CASM. The resulting simulation served as a reference for evaluating future production in Lake Texoma (1) without chloride management and (2) for selected chloride management scenarios (e.g., USACE 2003).

The usefulness of the reference simulation for assessing the probable food web effects of chloride (i.e., TDS) management in Lake Texoma depends on the accuracy of the model results. The following sections present comparisons of the CASM reference simulation with corresponding data reported for Lake Texoma. Given the mechanism whereby reductions in TDS can indirectly affect light attenuation, comparisons were made between modeled and observed measures of light extinction. Additional comparisons addressed the accuracy of simulated population biomass for aquatic producers and consumers.

Light Extinction

Modeled values of light extinction (m^{-1}) are determined by the complex dynamics of phytoplankton production; input, generation, decomposition, and settling of POC and DOC; and inputs and settling of TIS. The model also approximates the physical transport of these constituents among the segments represented in the Lake Texoma CASM. Given the number of factors and their spatial-temporal variability, agreement between modeled and measured extinction coefficients would importantly demonstrate the usefulness of the model for assessing the implications of chloride management alternatives.

Annual average extinction coefficients computed by the CASM compare reasonably with values reported by Clyde (2004) for Stations 9, 17, and 24 (Table 3). The model mean value of 1.13 (m^{-1}) for Station 9 is nearly identical to the value of 1.14 determined by Clyde (2004). The computed mean value of 1.46 (m^{-1}) for Station 24 compares reasonably with the measured value of 1.89. The model average value of 1.98 (m^{-1}) for Station 17, however, is more than double the average value of 0.83 (Clyde 2004), although the minimum model (0.48) and measured (0.54) values for this station are quite similar.

The modeled annual minimum and maximum values bracket the corresponding reported values for all three stations (Table 3). Corresponding modeled and measured values are all well within the same order of magnitude. Given the number of interrelated factors involved in simulating light extinction coefficients, the results of the CASM appear to compare favorably with reported values for Lake Texoma. It is important to note that the modeled light extinction values for the reference simulation result from calibrating phytoplankton production to phytoplankton data available for Lake Texoma. The corresponding degree of similarity between modeled and measured light extinction coefficients suggests that the model realistically describes the relationship between light extinction and phytoplankton production. Furthermore, the overall results appear to support the modeled contributions of POC, DOC, and TIS to light extinction.

Secchi Depth

Secchi depth is calculated for each location and day modeled by the CASM in Lake Texoma. The model computes the daily Secchi depth (m) as 1.65 divided by the calculated extinction coefficient (Giesen *et al.* 1990). This formulation indicates that the computed Secchi depths depend on the computed light extinction coefficients. Thus, the comparisons of modeled and measured Secchi depths are not offered as an independent evaluation of model performance (Table 4). However, the degree of similarity between modeled and measured values of this metric underscores the generality of the Giesen *et al.* (1990) formulation to Lake Texoma. This generality may become important in the continued evaluation of the Lake Texoma model because Secchi depths are more easily and routinely measured than light extinction coefficients.

Phytoplankton Production

As emphasized previously, the reference simulation represents calibration of modeled phytoplankton production to available data through adjustments of phytoplankton bioenergetics parameters (i.e., Table 1). Doyle and Baugher (2002) measured net annual phytoplankton productivity at four locations within Lake Texoma – two locations are identical to the Stations 9 and 17 included in the model (Table 5). Measured values for Stations 3 and 22 have been compared with model results for Stations 1 and 24. The total phytoplankton productivity was apportioned among the modeled taxonomic populations using the results of Waller *et al.* (2001). The percentages determined by Waller *et al.* (2001) were used to calculate annual production by cyanophytes, chlorophytes, diatoms, and microflagellates (Table 5). These values were used in calibrating the Lake Texoma CASM.

The calibrated model results show that it is possible to simulate a phytoplankton community dominated by cyanophytes using environmental input data developed for the Lake Texoma CASM and technically defensible estimates of phytoplankton parameter values. The model also shows diatoms to be the least productive component of the phytoplankton community. Chlorophytes and microflagellates are more abundant than diatoms, but well below an order of magnitude lower than bluegreens in terms of overall productivity. These qualitative features of the calibration are consistent with descriptions of the Lake Texoma phytoplankton community (Waller *et al.* 2001, 2002).

The results demonstrate that it is possible to calibrate the CASM simultaneously to 16 different values of phytoplankton (Table 5). However, the model overestimates cyanophyte production at all four locations; the most notable discrepancy occurs for Station 9, the closest agreement for bluegreen algae is for model Station 1. The model otherwise shows no consistent bias in its calibration to the remaining taxa –some values are slightly overestimated, others are underestimated. The closest agreement across all four taxonomic groups was obtained for modeled Station 1 compared to values derived for Station 3.

Perhaps a fair conclusion is that the Lake Texoma CASM represents a reasonable calibration to a limited set of measured phytoplankton productivity data. Importantly, the data are not sufficient to characterize any spatial-temporal variability associated with the single-value estimates derived from Doyle and Baugher (2002) or Waller *et al.* (2001). Spatial-temporal heterogeneity in phytoplankton production is commonly measured in lakes and reservoirs. Distributions of measured phytoplankton productivity characterized by realistic heterogeneities might easily include the modeled values listed in Table 5.

Zooplankton Production

The model results indicate a zooplankton community dominated by copepods. Annual modeled production per square meter by rotifers ranged across the modeled locations from 22.1 – 208.3 g-C/m^2/y. Cladoceran production correspondingly ranged from 13.1 – 357.2 g-C/m^2/y. Production by cyclopoid copepods ranged from14.4 – 306.8 g-C/m^2/y and calanoid copepod production ranged from 24.7 – 514.0 g-C/m^2/y.

Highest zooplankton production per square meter was simulated for the main lake Station 17, followed by Station 9 in the Big Mineral Arm of Lake Texoma. Lower modeled zooplankton production resulted for the riverine areas of the Red River and Washita River.

Limited zooplankton data for Lake Texoma are available for comparison with model results. However, Franks (2001) reported abundances of rotifers, cladocerans, cyclopoid copepods, and calanoid copepods for sample locations representative of the river arms and main lake portions of the reservoir. Based on numbers of organisms per liter, rotifers and cladocerans (mainly *Bosmina*) dominated the zooplankton community. The data suggest that rotifers and cladocerans might be more abundant in the riverine portions of the reservoir, especially the Red River arm, while the copepods appear more uniformly distributed throughout the sample locations. Converting these numbers to carbon suggests that cladocerans dominated the zooplankton biomass (~4.4-12.4 g-C/m^2). Calanoid copepods ranged from ~4.2 – 6.6 g-C/m^2, followed by cyclopoids (1.4 – 3.0 g-C/m^2) and rotifers (0.8 – 2.7 g-C/m^2).

The modeled annual production values cannot be directly compared with the limited samples of Franks (2001). However, examination of daily model values for the four populations in the reference simulation shows that the model results and the data are similar in magnitude for many of the model days. For example, for modeled Station 1, daily cladoceran biomass ranged from 2.3 – 3.5 g-C/m^2, calanoid copepods ranged from 3.0 – 8.6 g-C/m^2, cyclopoid biomass was between 0.3 – 4.8 g-C/m^2, and rotifer biomass ranged from 0.1 – 8.2 g-C/m^2. Similar to the modeled phytoplankton results, the main discrepancy appears to be an overestimation of zooplankton biomass in the main lake Stations 9 and 17.

Benthic Invertebrate Production

Data were not identified for comparison with modeled values of benthic invertebrate production. The model results indicated that amphipods (0.4 – 2.6 g-C/m^2) and aquatic insects (0.9 – 1.2 g-C/m^2) dominated the benthic invertebrate communities in the shallower river arm stations (Stations 1, 7, 20, 24). Mollusks were comparatively less productive (0.06 – 0.17 g-C/m^2) and dominated the deeper locations (Stations 9, 17). Given the modeling objectives, less attention was directed at the benthic invertebrate community. These populations mainly provide portions of the diet for freshwater drum, smallmouth buffalo, blue catfish, and white bass.

Fish Production

Direct estimates of fish production or standing stocks were not available in a format relevant for evaluating modeled fish biomass in Lake Texoma. However, Jenkins (1968) provides empirical relationships between selected physical-chemical characteristics of reservoirs and associated fish standing stock. These relationships were used to generate estimates of fish standing stocks (lbs/acre) for total fish and selected species in Lake Texoma (G.A. Clyde, Jr., USACE, personal communication, 2008). The empirical estimates lack the spatial resolution of the model. These estimates were converted to g-C/m^2 and compared with corresponding outputs from the Lake Texoma CASM. Thus, the comparisons were made using daily averaged model results across all the stations. The limited comparisons demonstrate some cases where the model biomass values are substantially greater than the empirical estimates (e.g., freshwater drum: 0.01 g-C/m^2 based on Jenkins (1968) versus 0.4 g-C/m^2 model average value). In contrast, if channel catfish are used as a surrogate for the blue catfish, Jenkins (1968) estimates 0.06 g-C/m^2 compared to the model results of 0.02 g-C/m^2. However, the predicted (0.091 g-C/m^2) and average modeled (0.092 g-C/m^2) values for white bass biomass are essentially identical.

In addition to the limited comparisons between model results and data, there are some additional qualitative aspects of fish production that can be used to evaluate the model performance. For example, production of forage fishes by the model shows that the biomass of silversides, threadfin shad, and gizzard shad are consistently greater than biomass of the more piscivorous adult striped bass and white bass. Modeled freshwater drum are approximately as abundant (biomass) as the combined shad populations, but less abundant than adult striped bass or white bass.

Chloride Management and Settling Rates

A principal concern associated with chloride management in Lake Texoma is that reduced concentrations of total dissolved solids (TDS) will correspondingly reduce rates of settling of suspended inorganic particulate matter in the reservoir (Schroeder and Toro 1996). Reduced rates of sedimentation will in turn decrease water transparency and negatively impact phytoplankton primary productivity. Decreased primary productivity might then propagate throughout the Lake Texoma food web (i.e., trophic cascade) and ultimately reduce biomass of a valued striped bass recreational fishery. Central to testing this hypothesis with the Lake Texoma CASM is the derivation of a quantitative relationship between TDS and settling rate (h^{-1}) relevant to this reservoir. In the terminology of ecological risk assessment, this relationship defines an exposure-response function, where the TDS concentration is the exposure and the corresponding settling rate is the response (USEPA 1998).

To date, the seminal study of the effects of TDS concentrations on settling rates in Lake Texoma was performed by Schroeder and Toro (1996). These investigators returned water samples collected from Lake Texoma to the laboratory for controlled studies of settling under varying physical-chemical conditions, TDS concentrations, and initial turbidity (i.e., 8 and 16 NTUs). Statistical analysis of the experimental results (2-way ANOVA) demonstrated that differences in TDS concentrations accounted for more than 90 percent of the variance in sedimentation rate. The analysis also showed that sedimentation rates were not significantly different for the different initial turbidity values.

Using the results reported by Schroeder and Toro (1996) for the initial condition of 8 NTUs, a simple linear relationship was determined between TDS and a first-order sedimentation rate (Figure 8). This relationship was programmed into the Lake Texoma CASM. For each simulated day and model station, the sedimentation rate is calculated based on the corresponding TDS concentration. In the model, TDS concentrations vary in space and time according to available data (Figure 8). It was assumed in the model that the sedimentation rate applied to the concentrations of total inorganic solids (TIS) and particulate organic carbon (POC). Application of the equation defined by Figure 8 would translate a decrease in TDS to reduced rates of settling of TIS and POC. Both TIS and POC affect light extinction. Reduced rates of settling of TIS and POC will reduce light availability in Lake Texoma. However, increased retention of POC in the euphotic zone also provides additional food for modeled zooplankton and forage fish (e.g., silversides and shad).

Future Without-Project Results

The reference simulation represents a model hypothesis that describes the overall ecological production dynamics of Lake Texoma based on pre-project concentrations of TDS. One projection of management interest is the future ecological production in the absence of chloride manipulations – that is, the future "without-project" management alternative. In the absence of chloride (TDS) management, the two principal factors that will influence production in Lake Texoma are year-to-year environmental variability and the longer-term sedimentation of the reservoir. To project the future without-project, the current conditions for Lake Texoma were simulated over a 50-y planning horizon based on two assumptions (1) annual variation in environmental conditions, and (2) environmental variations in combination with ongoing overall sedimentation of Lake Texoma (i.e., D. Tomlinson, USACE, personal communication). The following model results are presented for total production of phytoplankton and striped bass. Similar patterns were obtained for the other model food web components.

Total Phytoplankton

Figure 9 illustrates the projected total phytoplankton (minus P. parvum) based on the previously described longer-term environmental variability. Essentially all of the varied physical-chemical parameters (e.g., light, temperature, nutrients) directly influence phytoplankton production. Phytoplankton production for the entire reservoir averaged 267,000 metric tons of carbon over the 50-y simulation. Total annual production ranged from 243,000 to 309,000 tons. Annual variation in production was approximately 7% of 50-y average for this scenario.

Long-term sedimentation of the reservoir reduces the total phytoplankton habitat and correspondingly reduces productivity in the model (Figure 10). The 50-y projected average phytoplankton production was 238,000 metric tons of carbon for a future that included both simulated environmental variability and observed sedimentation rates. Modeled phytoplankton production ranged from 186,000 to 298,000 tons for this scenario. Annual variation in phytoplankton production was approximately 12% of the 50-y average.

Total Striped Bass

Striped bass production averaged 17,300 metric tons of carbon over the 50-y simulation that addressed only annual environmental variability (Figure 11). Variations in temperature and depth directly influence modeled striped bass production. Any other model effects associated with environmental variability result from alterations in the food web. Striped bass production ranged from 13,200 to 21,600 over this modeled period. Annual variation in striped bass production was 17.5% of the 50-y mean value.

Environmental variability and sedimentation resulted in the same 50-y average value for striped bass production, 17,300 metric tons (Figure 12). Production ranged from 12,100 to 20,400 when the effects of sedimentation were combined with environmental variability. Annual variations in striped bass production were approximately 12% of the mean value.

The simulated future without-project scenarios characterize the variations in phytoplankton and striped bass production that might be expected in relation to environmental variability and longer-term filling-in of the reservoir. The results demonstrate the importance of these factors, especially sedimentation of the reservoir. The results of these scenarios provide a basis for longer-term evaluation of the incremental effects of chloride management on production dynamics within Lake Texoma.

Future With-Project Results

To assess the implications of chloride management on ecological production within Lake Texoma, simulations were performed using adjusted daily input concentrations of TDS defined by different management alternatives.

Description of Management Alternatives

Management alternatives were represented changes in the daily values of TDS for all six modeled locations within Lake Texoma. Based on descriptions of likely alternatives (e.g., FEIS 2004, Schroeder and Toro 1996), it was assumed that the direct results of chloride management would reduce TDS concentrations in the Red River as it enters Lake Texoma. As the result of transport and dilution, reduced concentrations in the Red River would subsequently influence TDS concentrations throughout Red River portions (i.e., riverine, transition zone) and main lake areas of the reservoir. For purposes of this initial assessment, it was assumed that management alternatives would not influence concentrations of TDS in the Washita River portions of the reservoir, although the model structure permits adjustments to TDS values for the Washita River areas (i.e., Stations 20 and 24) if relevant in future assessments.

Detailed projections of TDS reductions in relation to chloride management remain to be developed for Lake Texoma. However, it has been suggested that reductions on the order of 8% of pre-project TDS concentrations might result from chloride management. For purposes of this initial model assessment, a series of percentage reductions (0, 4, 8, 12 and 16) in daily TDS values was examined.

Simulation of Management Alternatives

The proposed management scenarios were implemented in the Lake Texoma CASM by adjusting the pre-project TDS concentrations (i.e., Figure 7) for all the modeled locations except the Washita River Stations 20 and 24. Examination of the TDS values among the modeled stations indicated a spatial gradient within the reservoir. The spatial differences presumably result from the advective transport and dilution of TDS that enter mainly from the Red River. The observed pattern of dilution was used in defining the management scenarios. It was assumed that the TDS values for Station 1 would be directly reduced by the management scenario. For example, the 4% scenario would result in a 4% reduction in each of the daily values for Station 1. There is an approximate 23% difference between pre-project annual average TDS values for Station 1 and Station 17 (Dennison Dam). Thus, to more accurately characterize the effects of a management reduction of 4% at Station 1 on corresponding values for Station 17, the daily TDS values for Station 17 were calculated as 23% (i.e., dilution) of 4% - or approximately a 0.92% reduction in the baseline Station 17 values. It was assumed that the same dilution processes that determined pre-project baseline differences in TDS values among the stations would similarly affect the impacts of reductions in TDS associated with chloride management that directly affected Station 1.

Annual Effects of Chloride Management

To assess the potential short-term effects of chloride management on ecological production within Lake Texoma, a series of 1-y simulations was performed using the reference conditions and TDS reductions of 4, 8, 12, and 16%. The results of these simulations were examined in detail across modeled locations and populations of phytoplankton, zooplankton, forage fish, striped bass, and other fish (Tables 6-10).

Phytoplankton

Phytoplankton demonstrated a mixed response to reduced TDS (Table 6). The values of annual production of modeled microflagellates and chlorophytes were slightly reduced for Stations 1, 7, 9, and 17 across the TDS scenarios. Corresponding production of cyanophytes, diatoms, and *P. parvum* generally remained unaffected or slightly increased, except for Station 17 where responses varied among these taxa. Values of annual phytoplankton production for Stations 20 and 24 remained unaffected, which is consistent with the assumption that Washita River area TDS values are not influenced by chloride management in the Red River.

Zooplankton

Zooplankton growth is not directly affected by TDS in the model. Therefore, any changes in zooplankton production result from alterations in food web interactions. Rotifers increased slightly at Stations 1, 7, and 9 and more dramatically at Station 17 across the TDS scenarios (Table 7). Cladocerans decreased slightly at Stations 1 and 7, increased slightly at Station 9 and increased substantially at Station 17. Modeled production values for cyclopoid and calanoid copepods showed similar responses: slight increases or decreases occurred for Stations 1, 7, and 9, while production markedly increased at Station 17. Consistent with the phytoplankton, annual zooplankton production was not affected at Stations 20 and 24.

Fish

For Stations 1, 7, and 9, the modeled values of annual production of silversides, threadfin shad, and gizzard shad were minimally impacted or increased slightly by the changes in phytoplankton and zooplankton associated with alterations in TDS (Table 8). Forage fish production was not affected at Stations 20 and 24. However, significant reductions in forage fish production resulted for Station 17.

The model produced a similar pattern of results concerning the potential impacts of TDS reductions on striped bass production (Table 9). Across all three life stages, minimal impacts or slight increases were calculated for Stations 1, 7, and 9. Stations 20 and 24 were not affected. Yet, substantial decreases in striped bass production resulted for Station 17.

Freshwater drum demonstrated a similar pattern of response to TDS alterations as the forage fish and striped bass (Table 10). Minimal impacts or slight increases occurred for Stations 1, 7, and 9. Substantial decreases were obtained for Station 17.

Interestingly, the more benthivorous populations of smallmouth buffalo and blue catfish did not exhibit decreased annual production at Station 17 (Table 10). Minimal decreases were obtained for smallmouth buffalo at Stations 1, 7, and 9. Blue catfish were not impacted by TDS management.

Longer-term Effects of Chloride Management

The longer-term implications of the separate management alternatives were simulated using the 50-y without project environmental and sedimentation baseline projections for comparison. This permitted a more realistic assessment of the potential impacts of reduced TDS values on ecological production in relation to environmental variability and long-term filling-in of the reservoir.

Phytoplankton

Figure 13 presents the results of the 50-y simulations of total phytoplankton (not including *P. parvum*) production that examines phytoplankton response to TDS management in comparison with environmental variability. The results demonstrate that modeled phytoplankton production is influenced more by year-to-year environmental variability than by the 4-16% reductions in TDS. The 0% scenario is the same simulation as the future without-project that includes the same pattern of year-to-year environmental variability. The modeled influences of TDS on total phytoplankton production are evident for certain years (e.g., 11, 15, 31, and 46). However, the impacts of TDS are substantially less than the effects of variations in physical-chemical parameters that influence phytoplankton growth.

The combination of environmental variability and long-term sedimentation of the reservoir further outweighed the effects of the modeled TDS scenarios on phytoplankton production for Lake Texoma (Figure 14).

Striped Bass

A pattern similar to phytoplankton resulted for modeled total striped bass production in relation to the combined effects of alterations to TDS and environmental variability (Figure 15). Environmental variability clearly produced greater year-to-year fluctuations in striped bass biomass than the 4-16% reductions in TDS.

Including the effects of reservoir sedimentation with environmental variability and reductions in TDS further suggested comparatively minimal impacts of chloride management on overall striped bass production in Lake Texoma (Figure 16).

Discussion

It proved possible to develop a version of the comprehensive aquatic system model (CASM) to examine the potential effects of reduced TDS concentrations on food web production dynamics in Lake Texoma. The resulting Lake Texoma CASM was based on a consensus food web structure, available and relevant environmental data, and published values of bioenergetics parameters for the modeled populations of aquatic producers and consumers. It proved possible to usefully calibrate the model to existing data that described phytoplankton production within this large reservoir. Calibration to reported phytoplankton production values resulted in values of zooplankton and fish production that were reasonably similar to the few existing values for these consumer populations. The bioenergetics-based modeling framework appears capable of describing spatial-temporal patterns of ecological production similar to those measured in Lake Texoma. Other modeling outcomes were possible. For example, biologically unrealistic or infeasible bioenergetics parameter values might have been required to calibrate the model to the observed phytoplankton data. Plausible combinations of parameter values and available environmental data could have produced biomass results that were orders of magnitude greater or less than suggested by the existing biomass data. It could have proven possible to calibrate to the phytoplankton data, but corresponding estimates of consumer production might have been unrealistically high or low.

Nonlinear relationships between phytoplankton growth and environmental factors (e.g., light, temperature, nutrients) make it difficult to anticipate the effects of TDS reductions on phytoplankton production. In the simple case, reductions in TDS and settling rate for suspended particulate matter would be expected to reduce light intensity and correspondingly reduce phytoplankton growth. However, phytoplankton response depends upon ambient light intensity being greater or less than light saturation for the population of interest. If light is above saturation (e.g., near the water surface), reduced intensity can actually increase productivity; if light is less than the saturation value, reduced TDS and associated reduced light can reduce phytoplankton growth. Both responses were observed in modeling the effects of TDS on Lake Texoma phytoplankton. Such nonlinear growth responses to changing environmental conditions have been specifically formulated in the Lake Texoma CASM.

The model was necessarily developed using environmental input data from several sources, locations, and years. For example, many of the water quality parameters (e.g., DIN, DIP, temperature) were based on data from 1996-1997 (Waller *et al.* 2001). But the pre-project TDS data derived from samples collected from 1975-1978. Collation of data from these various sources might lead to unrealistic combinations of environmental inputs. However, examination of long-term flows recorded at Terral, Oklahoma shows similar flows for the 1996-1997 and 1975-1978 periods (Figure 17). This might help justify the use of these combinations of input data. At least the inputs are not from extremely different flows.

The empirical relationship between TDS concentrations and settling rate derived from Schroeder and Toro (1996) is central to the assessment of chloride management alternatives. Yet, this relationship is based on limited experimental results obtained under controlled laboratory conditions with samples of Lake Texoma water. However, it is not certain how these laboratory results extrapolate to the complex and varying physical-chemical conditions defined by Lake Texoma. In developing the Lake Texoma CASM, it was assumed that the settling rates influenced by TDS concentrations determined the sinking rate for both inorganic solids and particulate organic matter. Both of these constituents contribute to attenuation of light in the model. It remains uncertain whether TDS influences the settling rate of POC in Lake Texoma. As a result of this assumption, the CASM might overestimate the effects of TDS reductions on phytoplankton production. The model is conservatively biased in this respect.

The greatest discrepancy between model results, available data, and general knowledge of the Lake Texoma occurred for Station 17, which represents main lake conditions near the dam. The model indicates this area as highly productive, while the data suggest otherwise. Overestimation of phytoplankton productivity for Station 17 might result from integration of photosynthesis at 1-m increments over the input depth of the euphotic zone. Station 17 characteristically had greater euphotic zone depths than the shallower riverine stations included in the model. The model also currently distributes phytoplankton biomass equally throughout the water column, which might result in overestimation of productivity. Actual phytoplankton biomass tends to be concentrated near the metalimnion in stratified systems. In fact, high values of input euphotic zone depths might define areas of lower productivity. Extinction coefficients reported for Station 17 were less than values for other stations (Clyde 2004). Correspondingly higher Secchi depths were reported for this station as well. Integration of photosynthesis over the independently input euphotic zone depths may be a model assumption that requires revision to improve overall model performance.

Summary

Despite the assumptions and limitations inherent in developing a complex aquatic systems model for Lake Texoma, the resulting CASM application appears as a useful approach for assessing the ecological impacts of chloride management in this reservoir. The combination of environmental input data, food web structure, and literature-based bioenergetics parameters proved capable of usefully describing production dynamics measured in Lake Texoma. Using an experimentally determined relationship between TDS concentrations and particulate settling rates (Schroeder and Toro, 1996), the model translated various percentage reductions in TDS to possible effects on primary producers and consumers. The overall performance of the model supports its utility in the context of "proof of principle" in assessing risks posed by chloride management in Lake Texoma.

With the exception of the potentially anomalous results for the main lake Station 17, the TDS reductions representative of proposed management alternatives had minimal effects on the aquatic populations of phytoplankton, zooplankton, and fish included in the Lake Texoma CASM. Importantly, 4-16% reductions in daily TDS concentrations did not translate to substantive impacts on striped bass production, especially when compared to the effects of environmental variability or the long-term filling-in of Lake Texoma. Of similar importance, reductions in TDS concentrations did not result in dramatic increases in the modeled production of *P. parvum*.

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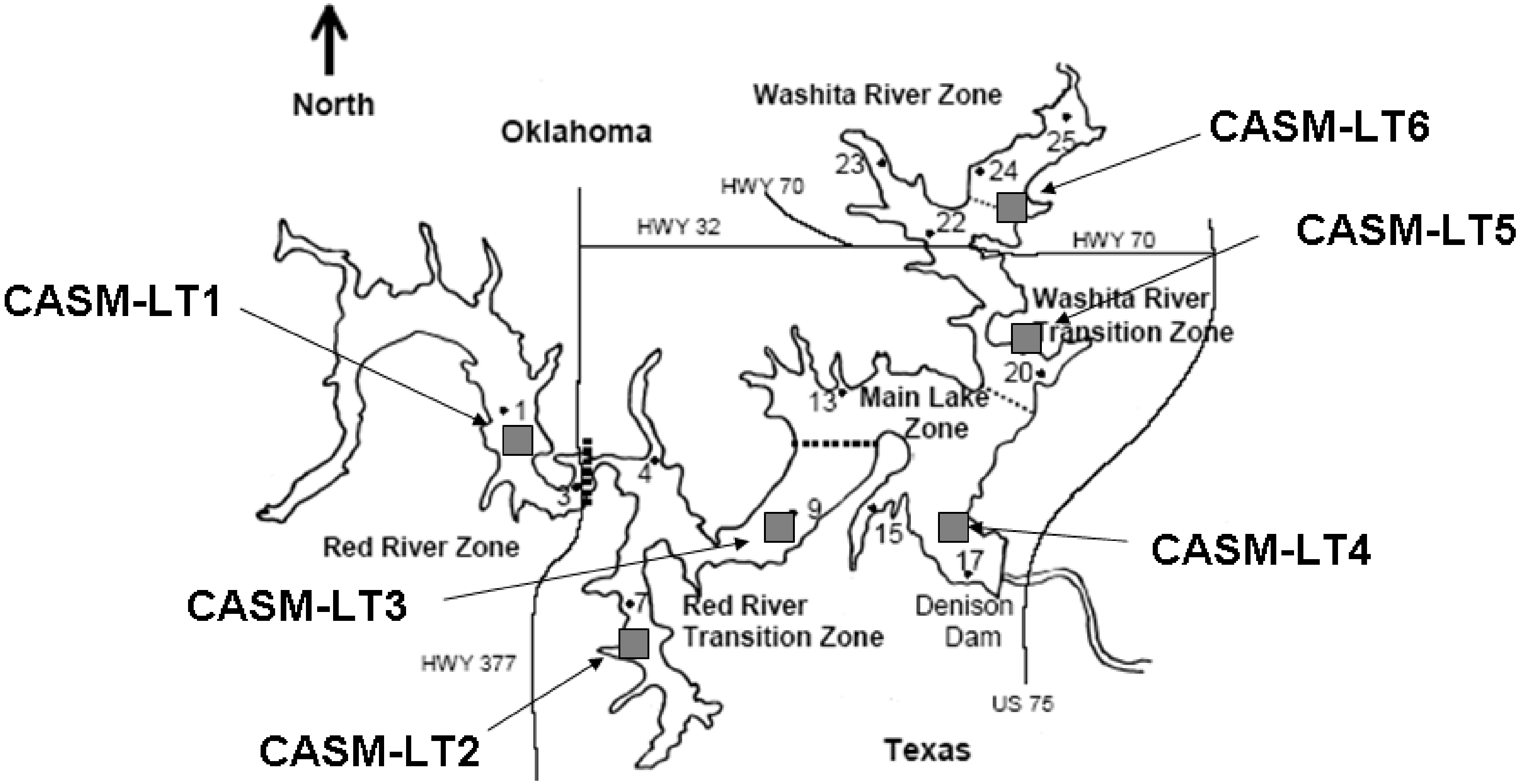

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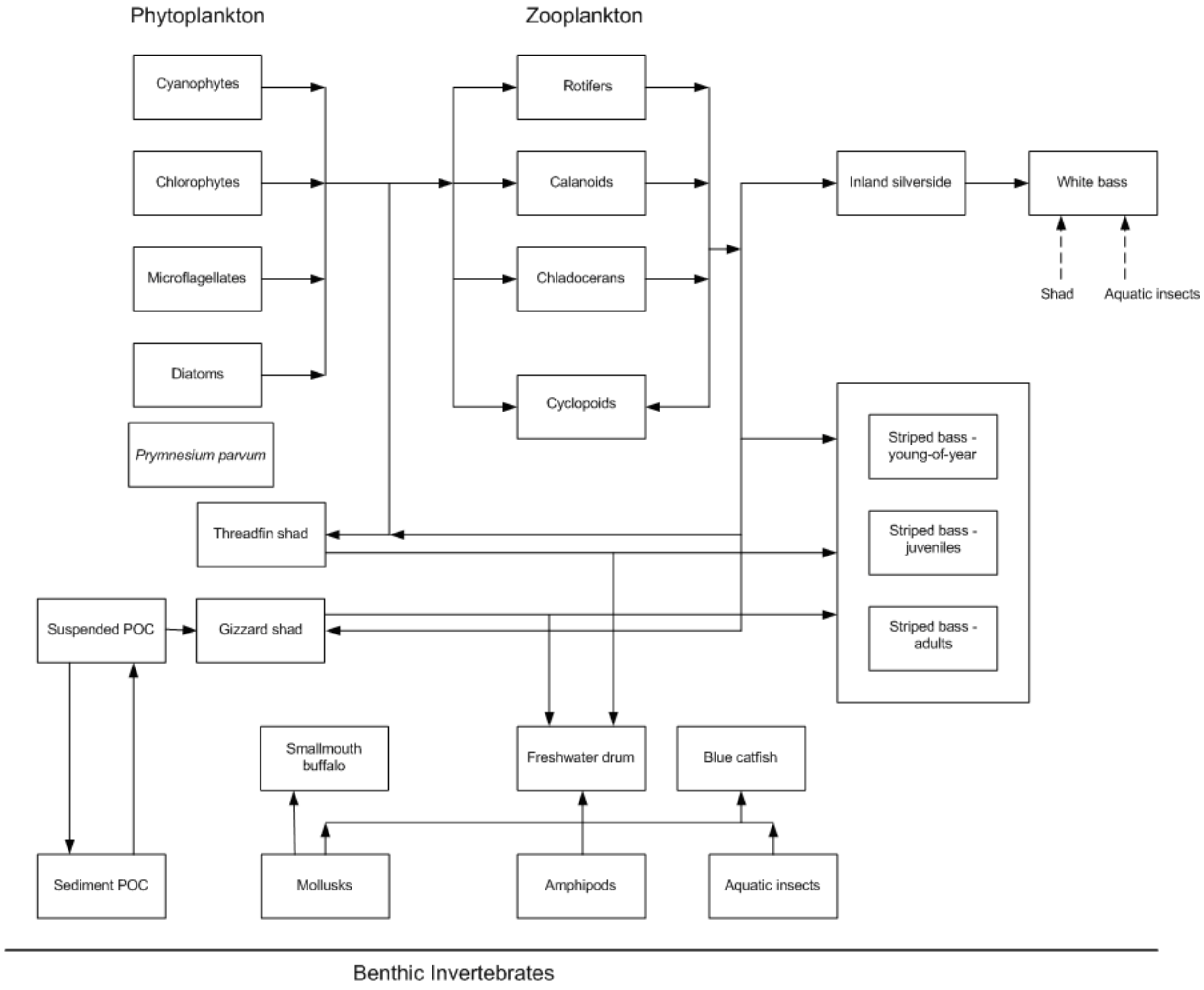

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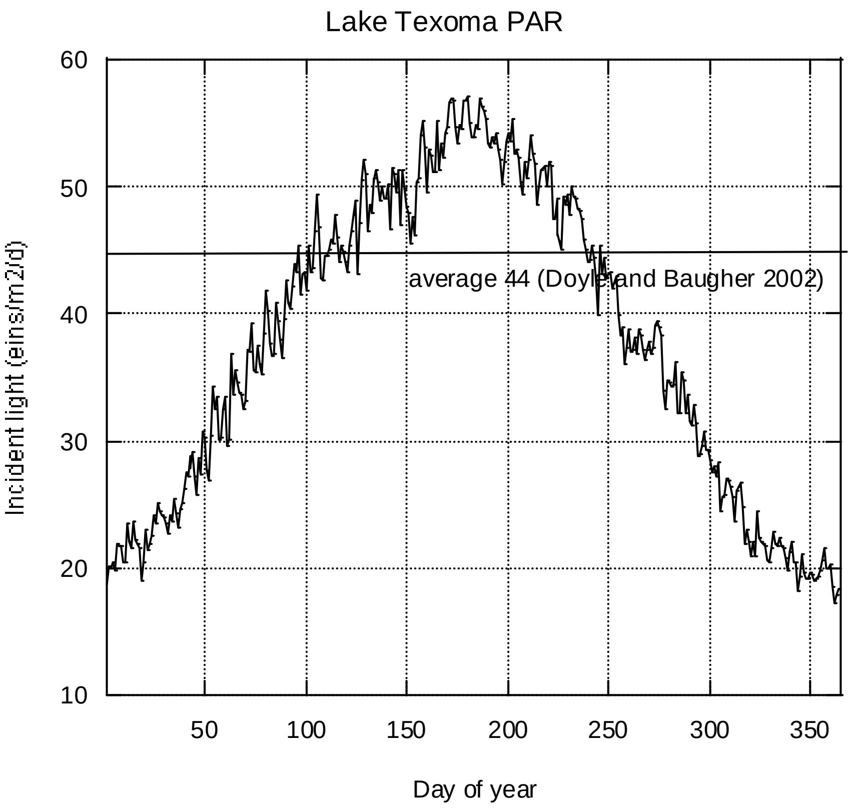

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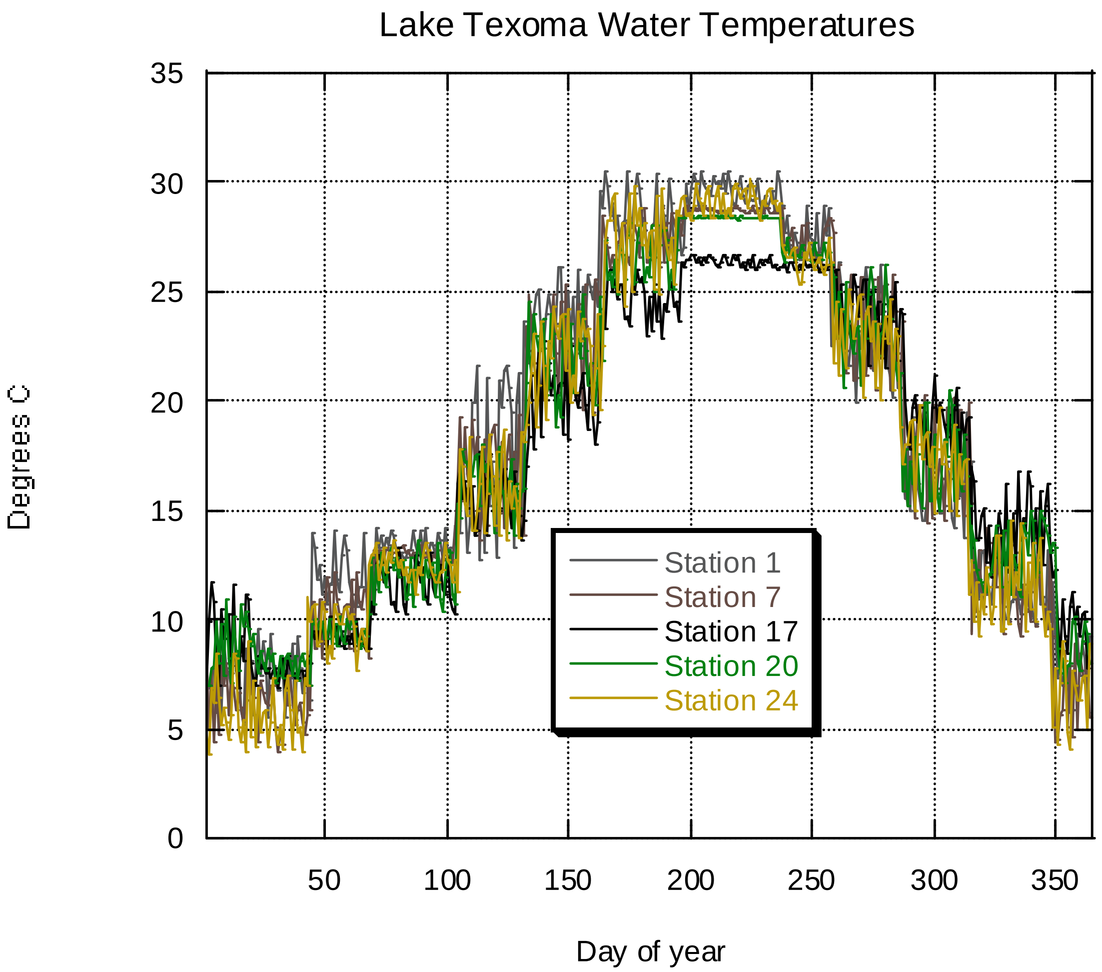

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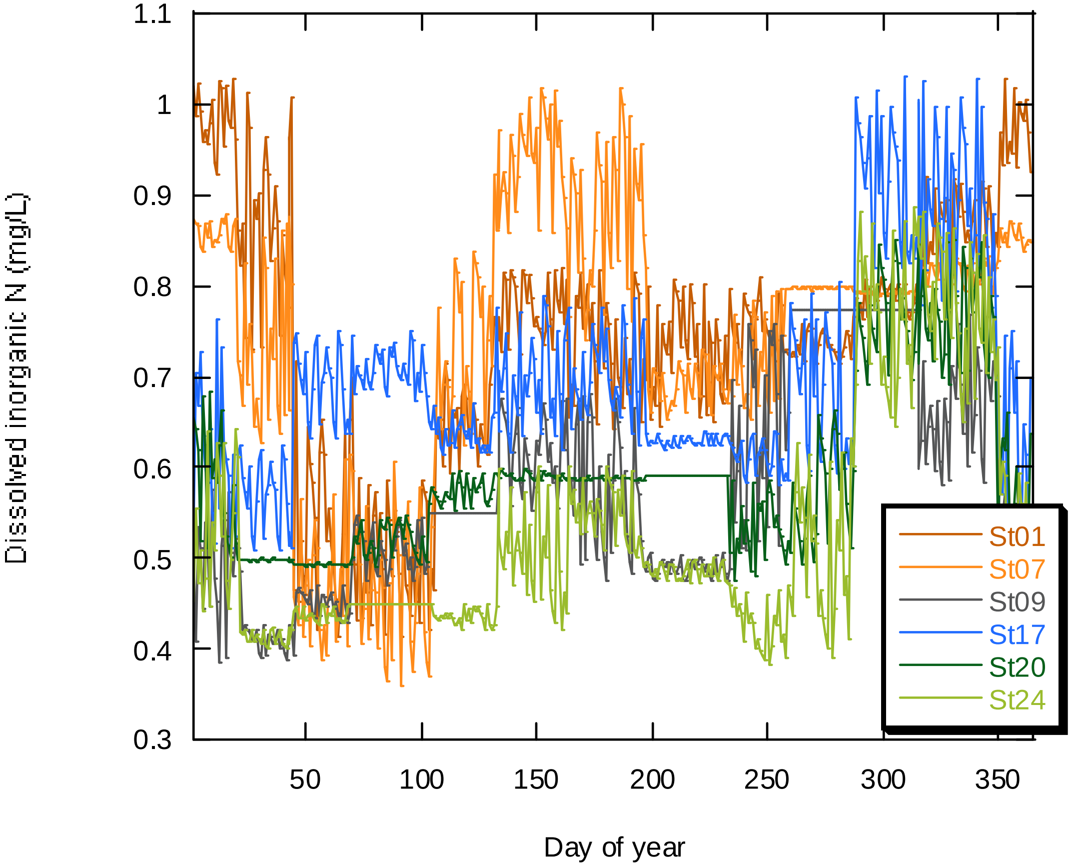

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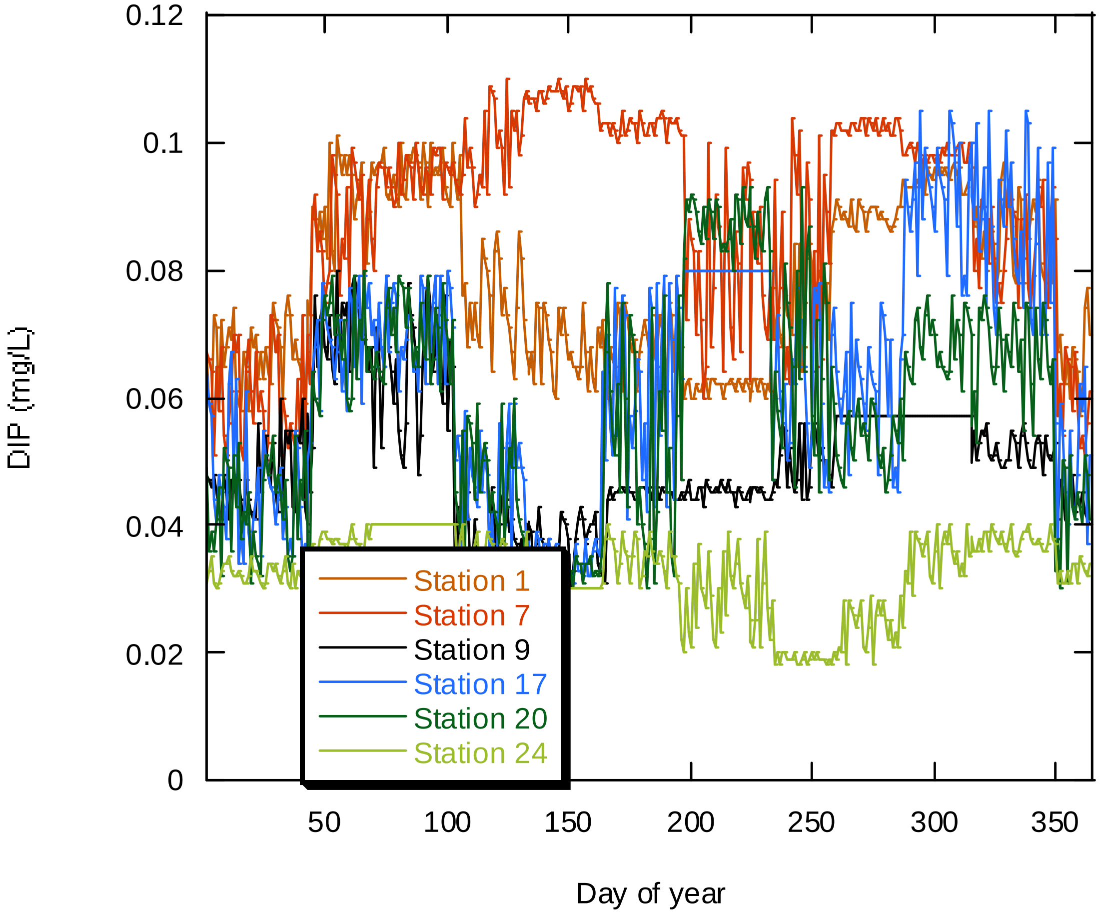

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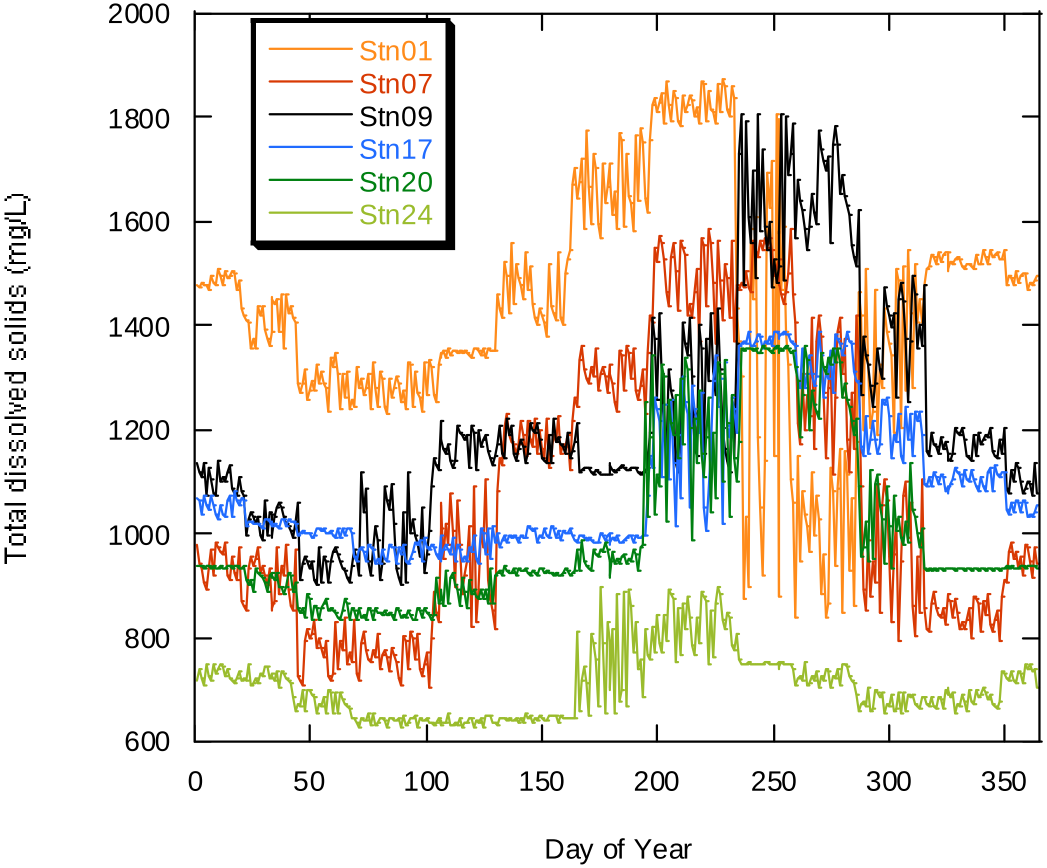

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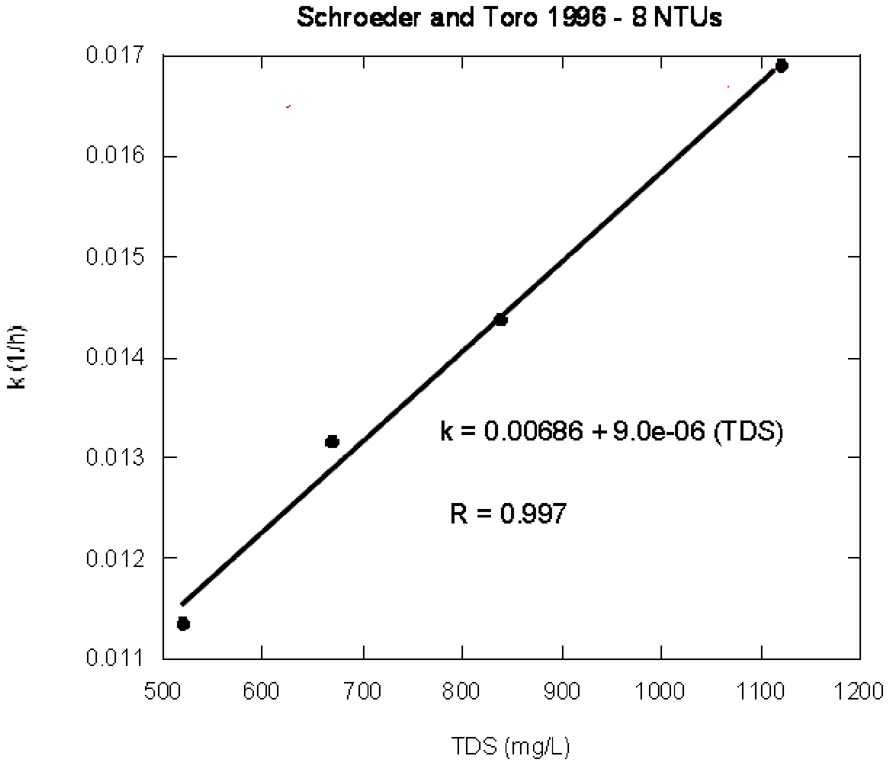

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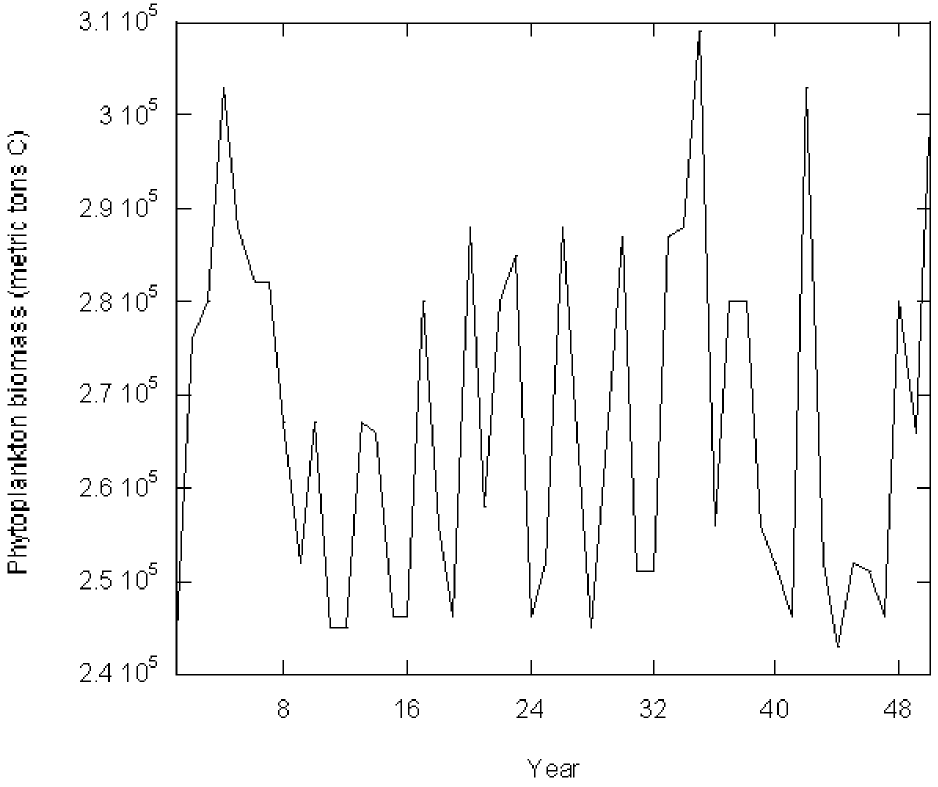

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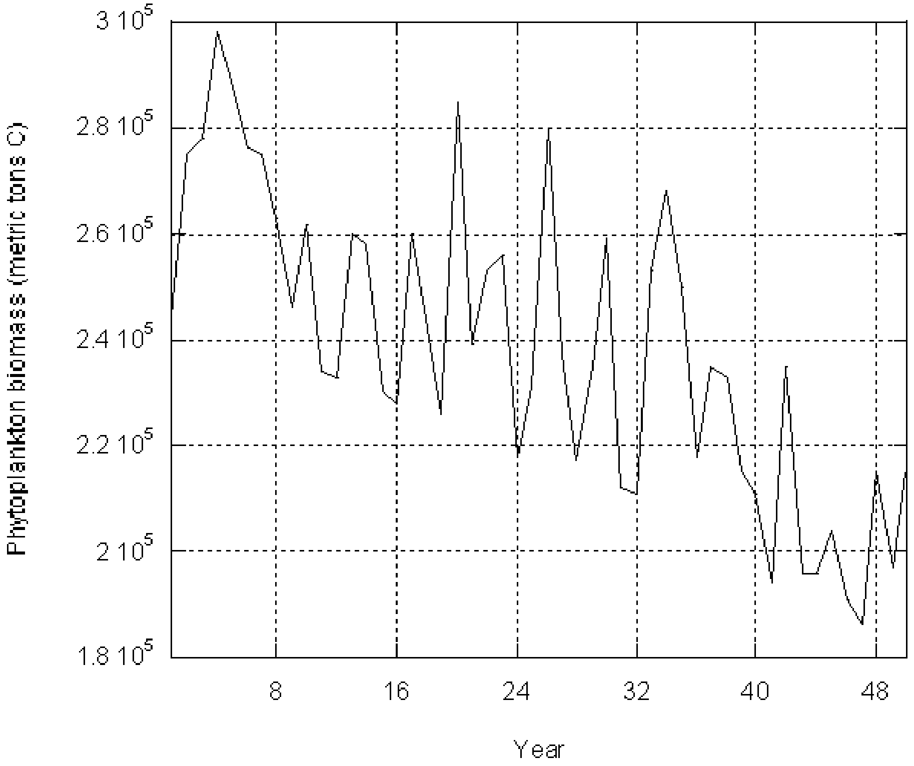

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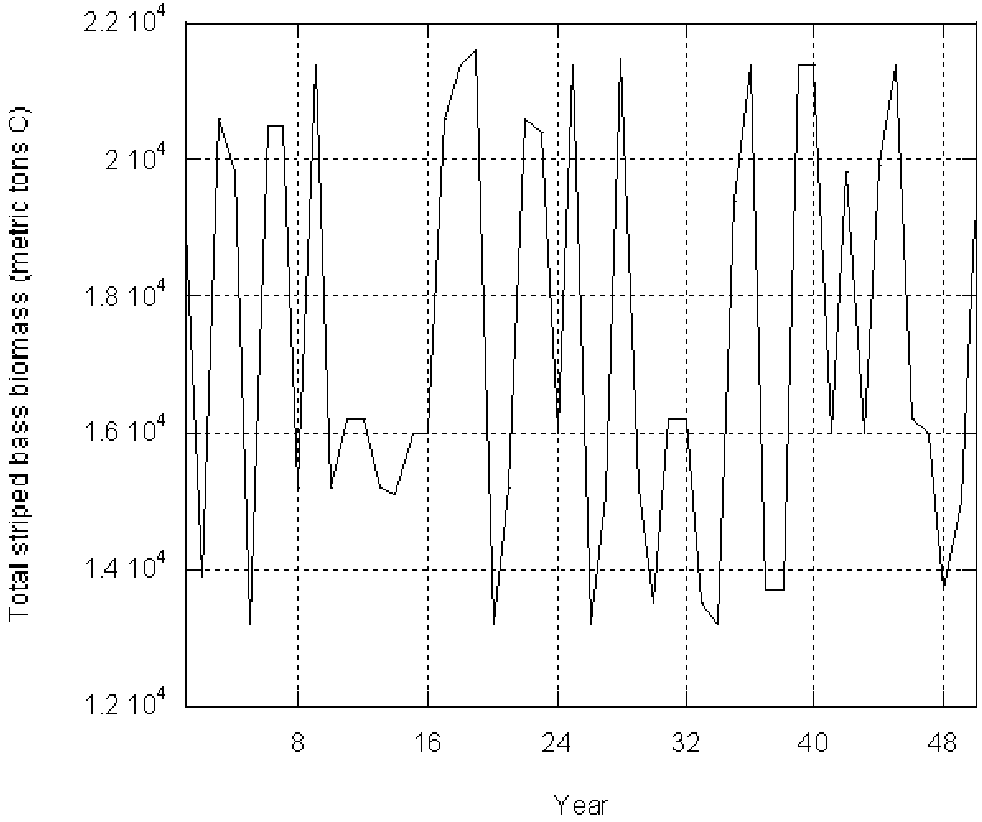

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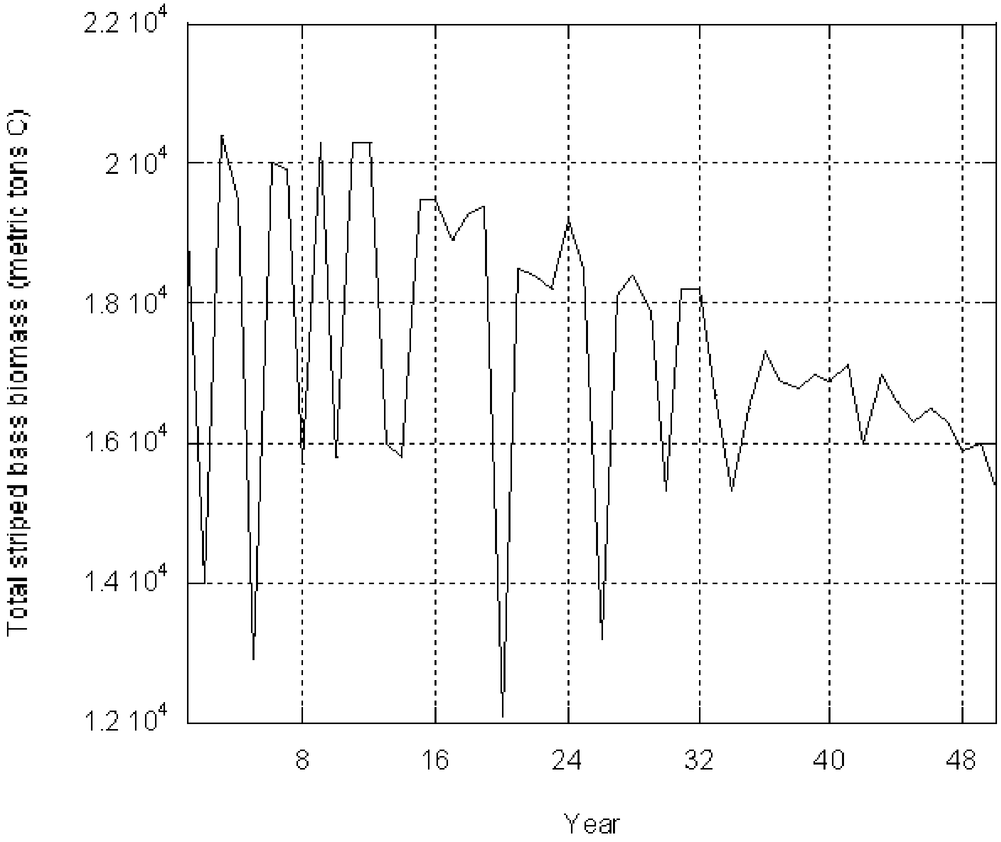

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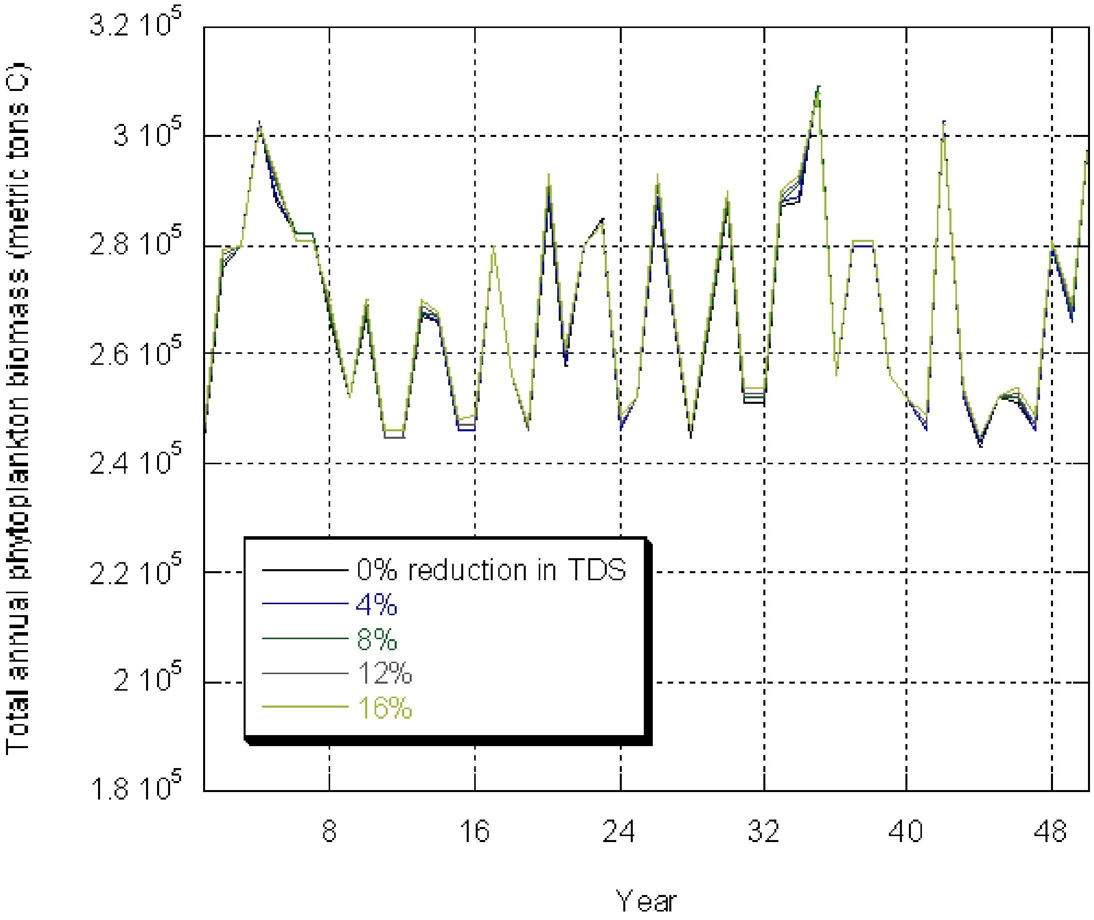

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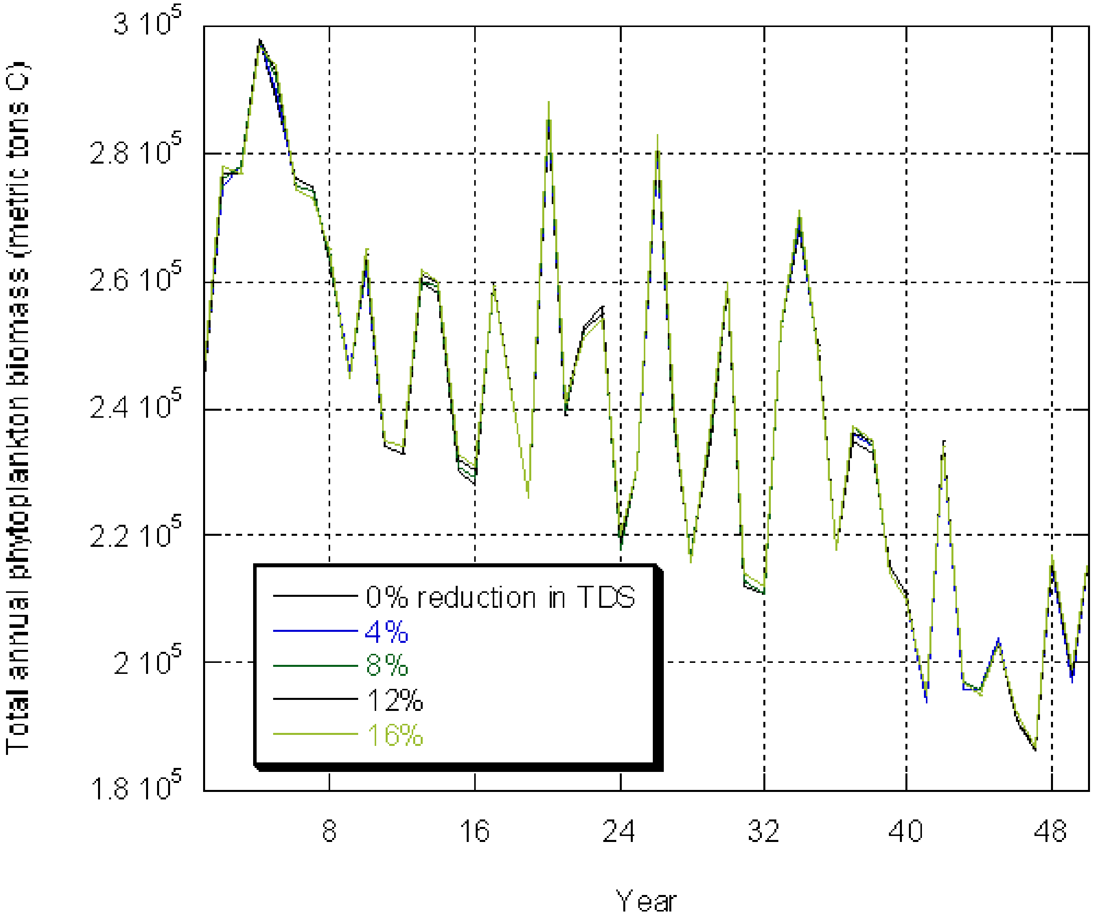

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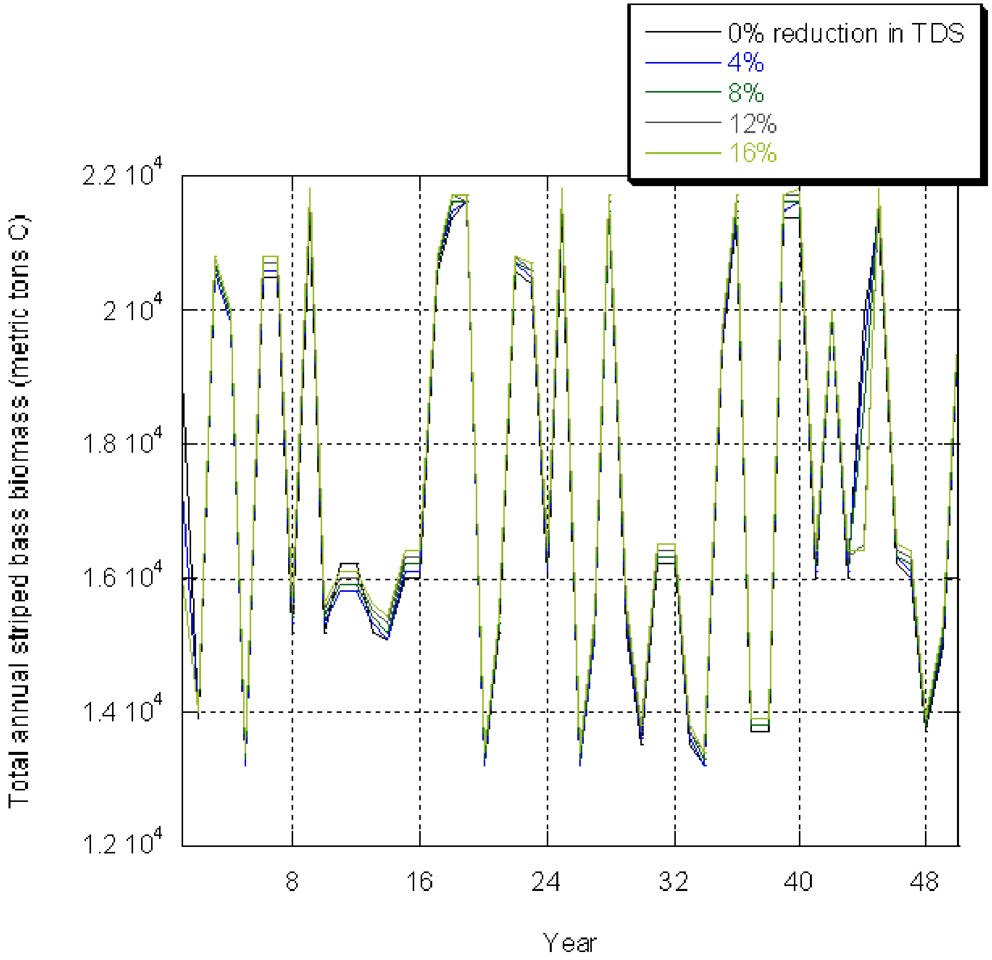

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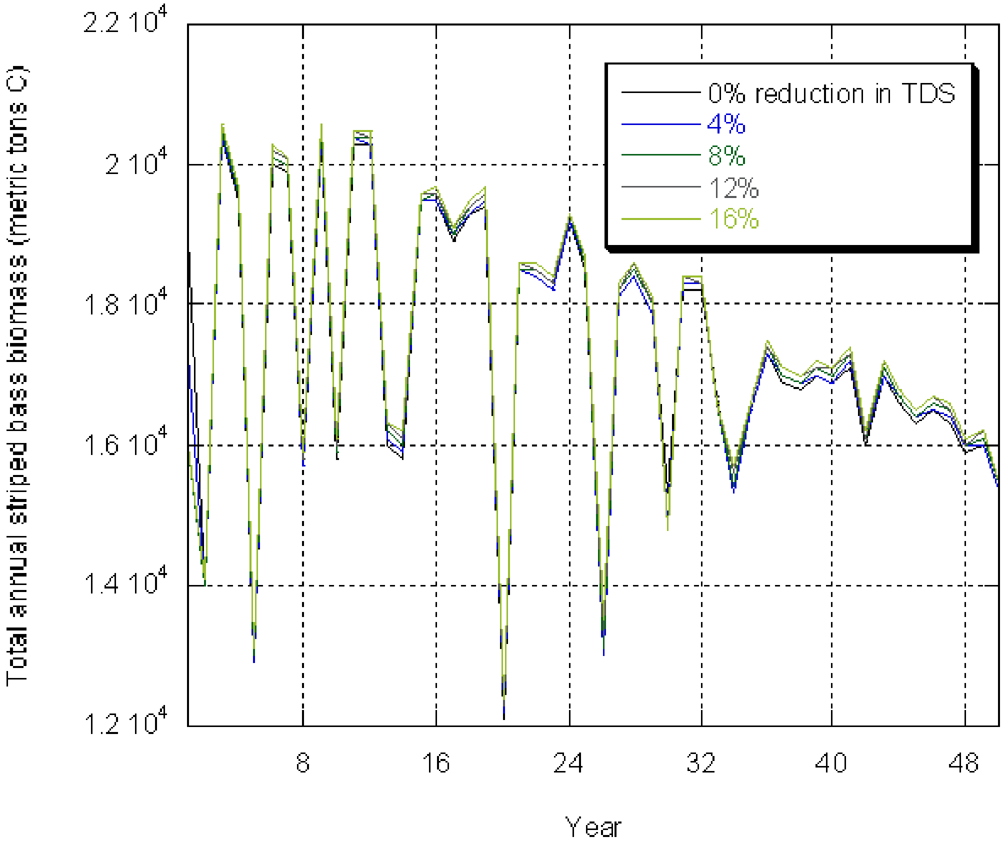

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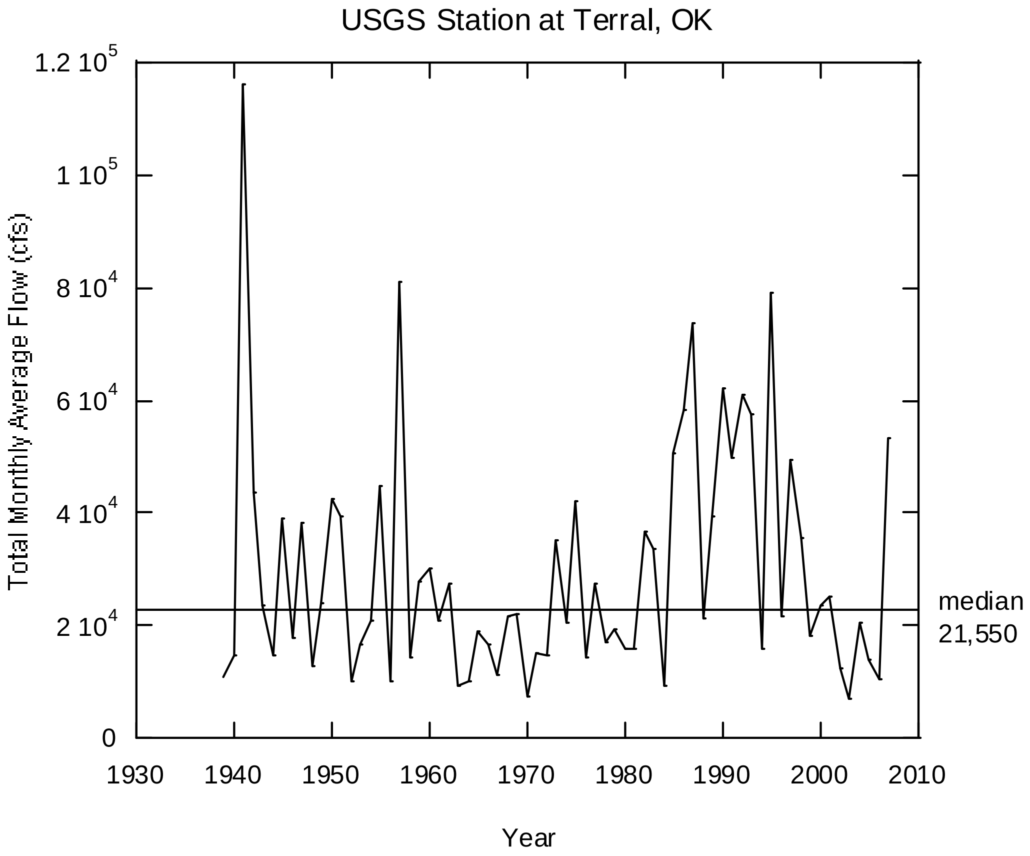

Figure 17. Average monthly flows from 1938 through 2007 based on flows measured at the USGS gage at Terral, OK.

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TABLE 1 Bioenergetics Parameters for Primary Producer Populations

Producer Community	**CASM population**	**Growth Rate (1/d)**	**Te_1 (ºC)**	**Te_2 (ºC)**	**Te_3 (ºC)**	**Te_4 (ºC)**	**Light Saturation (Eins/m2/d)**	**Sinking Rate (1/d)**
Phytoplankton	microflagellates	1.40	14	20	30	32	5	0.02
	cyanophytes	2.05	18	22	30	32	6	0.02
	chlorophytes	2.15	16	20	28	32	5	0.05
	diatoms	1.00	6	12	16	20	3	0.15
	P. parvum	0.50	2	10	14	18	1	0.01
Periphyton	periphyton	1.75	12	22	32	36	10	-

TABLE 1 Bioenergetics Parameters for Primary Producer Populations (Continued)

Producer Community	**CASM population**	**k_P (mg/L)**	**k_N (mg/L)**	**k_S (mg/L)**	**Mortality (1/d)**	**Photo-respiration (unitless)**	**Dark respiration (1/d)**	
Phytoplankton	microflagellates	0.020	0.47	-	0.104	0.045	0.30	
	cyanophytes	0.030	0.20	-	0.102	0.045	0.20	
	chlorophytes	0.012	0.45	-	0.064	0.035	0.20	
	diatoms	0.008	0.45	0.60	0.082	0.055	0.35	
	P. parvum	0.006	0.60	-	0.040	0.020	0.20	
Periphyton	periphyton	0.016	0.22	-	0.040	0.350	0.10	

TABLE 2 Bioenergetics Parameters for Consumer Populations

Consumer Community	CASM Populations	Max Consumption (1/d)	Te_1 (ºC)	Te_2 (ºC)	Te_3 (ºC)	Te_4 (ºC)	Rsda (unitless)
Zooplankton	rotifers	2.15	10	16	30	38	-
	cladocerans	1.95	14	18	30	38	-
	cyclopoids	1.65	10	16	30	38	-
	calanoids	1.95	14	20	30	38	-
Pelagic omnivores	silversides	0.34	12	16	30	38	0.11
	threadfin shad	0.32	10	16	30	38	0.08
	gizzard shad	0.34	10	16	30	38	0.06
Pelagic piscivores	st. bass-yoy	0.16	10	16	32	38	0.123
	st. bass-juv	0.14	10	16	30	38	0.135
	st. bass-adult	0.12	10	16	28	38	0.08
	white bass	0.10	10	16	30	38	0.08
Benthic invertebrates	mollusks	0.90	12	16	30	38	-
	amphipods	2.05	12	16	30	38	-
	aq insects	2.00	14	18	32	38	-
Benthic omnivores	fr. drum	0.18	12	16	32	36	0.08
	smth buffalo	0.16	12	16	30	34	0.125
	blue catfish	0.10	10	14	30	34	0.125
Benthic piscivores	benthic piscivore	0.10	12	28	32	36	0.14

TABLE 2 Bioenergetics Parameters for Consumer Populations (Continued)

Consumer Community	CASM Populations	Rmax (1/d)	Tr_o (C)	Tr_m (C)	F (unitless)	U (unitless)	Mortality (1/d)
Zooplankton	Rotifers	0.18	32	37	0.040	0.05	0.10
	Cladocerans	0.16	28	37	0.025	0.06	0.10
	Cyclopoids	0.18	26	37	0.025	0.06	0.15
	Calanoids	0.16	30	37	0.035	0.04	0.10
Pelagic omnivores	Silversides	0.0017	34	40	0.058	0.025	0.015
	threadfin shad	0.0017	34	40	0.0104	0.068	0.010
	gizzard shad	0.0017	34	40	0.0104	0.068	0.010
Pelagic piscivores	st. bass-yoy	0.0004	32	38	0.104	0.068	0.0084
	st. bass-juv	0.0007	32	38	0.160	0.100	0.020
	st. bass-adult	0.0007	32	38	0.160	0.100	0.020
	white bass	0.0006	32	38	0.120	0.080	0.015
Benthic invertebrates	Mollusks	0.014	32	37	0.040	0.05	0.02
	Amphipods	0.008	30	37	0.040	0.04	0.02
	aq insects	0.018	32	37	0.060	0.06	0.038
Benthic omnivores	fr. Drum	0.0007	32	37	0.175	0.08	0.00045
	smth buffalo	0.0006	30	37	0.175	0.08	0.0010
	blue catfish	0.0020	30	37	0.175	0.08	0.0010
Benthic piscivores	benthic piscivores	0.0005	32	37	0.170	0.06	0.00003

TABLE 3. Comparison of Modeled and Measured Annual Light Extinction Coefficients

	Extinction coefficient (m^{-1})		
	Station 9	**Station 17**	**Station 24**
Clyde, 2004			
Mean	1.13	0.83	1.89
Minimum	0.82	0.54	1.11
Maximum	2.20	1.21	3.58
Lake Texoma CASM			
Mean	1.14	1.98	1.46
Minimum	0.49	0.48	0.40
Maximum	5.52	6.34	5.83

TABLE 4. Comparison of Modeled and Measured Secchi Depths

	Secchi depth (m)		
	Station 9	**Station 17**	**Station 24**
Clyde, 2004			
Mean	1.05	1.60	0.54
Minimum	0.45	0.65	0.15
Maximum	1.85	3.45	1.25
Lake Texoma CASM			
Mean	1.63	0.93	1.32
Minimum	0.30	0.26	0.28
Maximum	3.30	3.46	4.08

TABLE 5. Comparison of Reference Simulation and Measured Annual Phytoplankton Production

		Percent of total annual productivity[2]			
	Measured	82.9	6.7	2.6	4.4
	net annual				
Lake Texoma	**productivity**[1]	**cyanophytes**	**chlorophytes**	**diatoms**	**microflagellates**
Station	g-C/m^2/y	g-C/m^2/y	g-C/m^2/y	g-C/m^2/y	g-C/m^2/y
3	326	270	22	8	14
1		296	21	6	9
9	285	236	19	7	13
9		1,125	11	4	30
17	267	221	18	7	12
17		424	35	5	21
22	308	255	21	8	14
24		317	6	4	8

[1] Doyle and Baugher, 2002.
[2] Waller *et al.* 2001

TABLE 6. Simulated Effects of Chloride Management (i.e., TDS) on Phytoplankton Production

	Percent reduction in TDS concentrations (%)				
	0	**4**	**8**	**12**	**16**
	$gC/m^2/y$	$gC/m^2/y$	$gC/m^2/y$	$gC/m^2/y$	$gC/m^2/y$
Microflagellates					
Station 01 RRZ	9.4	9.4	9.3	9.2	9.0
Station 07 RTZ	2.4	2.5	2.5	2.4	2.5
Station 09 RMZ	30.7	30.7	30.3	29.9	29.5
Station 17 Dam	21.6	21.4	20.9	20.5	20.0
Station 20 WTZ	20.0	20.0	20.0	20.0	20.0
Station 24 WRZ	7.8	7.8	7.8	7.8	7.8
Cyanophytes					
Station 01 RRZ	296.5	297.2	297.5	297.4	296.9
Station 07 RTZ	293.2	294.3	295.1	296.3	297.0
Station 09 RMZ	1,125.0	1,126.0	1,127.0	1,129.0	1,130.0
Station 17 Dam	424.3	428.6	422.0	420.3	418.3
Station 20 WTZ	313.0	313.0	313.0	313.0	313.0
Station 24 WRZ	316.8	316.8	316.8	316.8	316.8
Chlorophytes					
Station 01 RRZ	20.6	20.6	20.2	19.7	19.3
Station 07 RTZ	1.9	1.9	1.9	1.9	1.9
Station 09 RMZ	11.1	11.1	10.9	10.8	10.7
Station 17 Dam	35.2	35.0	33.9	33.2	32.6
Station 20 WTZ	13.7	13.7	13.7	13.7	13.7
Station 24 WRZ	6.1	6.1	6.1	6.1	6.1

TABLE 6. Simulated Effects of Chloride Management (i.e., TDS) on Phytoplankton Production (Continued)

	Percent reduction in TDS concentrations (%)				
	0	**4**	**8**	**12**	**16**
	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y
Diatoms					
Station 01 RRZ	6.4	7.2	8.0	8.8	9.8
Station 07 RTZ	3.9	4.3	4.8	5.3	5.8
Station 09 RMZ	39.3	41.6	44.2	46.8	49.8
Station 17 Dam	4.6	4.5	4.4	4.4	4.3
Station 20 WTZ	19.1	19.1	19.1	19.1	19.1
Station 24 WRZ	3.8	3.8	3.8	3.8	3.8
P. parvum					
Station 01 RRZ	0.8	0.8	0.9	0.9	0.9
Station 07 RTZ	0.7	0.7	0.7	0.7	0.8
Station 09 RMZ	5.7	5.7	5.7	5.8	5.9
Station 17 Dam	3.7	3.7	3.7	3.6	3.6
Station 20 WTZ	2.8	2.8	2.8	2.8	2.8
Station 24 WRZ	1.2	1.2	1.2	1.2	1.2

TABLE 7. Simulated Effects of Chloride Management (i.e., TDS) on Zooplankton Production

	Percent reduction in TDS concentrations (%)				
	0	**4**	**8**	**12**	**16**
	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y
Rotifers					
Station 01 RRZ	28.4	28.5	28.6	28.7	28.7
Station 07 RTZ	38.9	39.0	39.1	39.1	39.1
Station 09 RMZ	105.1	105.5	105.6	105.6	105.6
Station 17 Dam	208.3	237.9	245.4	245.2	244.7
Station 20 WTZ	42.0	42.0	42.0	42.0	42.0
Station 24 WRZ	22.1	22.1	22.1	22.1	22.1
Cladocerans					
Station 01 RRZ	13.1	13.0	13.0	12.9	12.7
Station 07 RTZ	26.3	26.1	26.0	25.9	25.8
Station 09 RMZ	110.0	110.7	110.8	110.5	110.1
Station 17 Dam	357.2	445.9	494.1	496.9	495.4
Station 20 WTZ	30.5	30.5	30.5	30.5	30.5
Station 24 WRZ	25.5	25.5	25.5	25.5	25.5

TABLE 7. Simulated Effects of Chloride Management (i.e., TDS) on Zooplankton Production (Continued)

	Percent reduction in TDS concentrations (%)				
	0	**4**	**8**	**12**	**16**
	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y
Cyclopoids					
Station 01 RRZ	14.4	14.4	14.3	14.3	14.3
Station 07 RTZ	21.3	21.1	21.1	21.0	20.9
Station 09 RMZ	57.9	58.6	58.8	58.5	58.3
Station 17 Dam	306.8	395.2	452.7	456.9	455.9
Station 20 WTZ	29.1	29.1	29.1	29.1	29.1
Station 24 WRZ	20.1	20.1	20.1	20.1	20.1
Calanoids					
Station 01 RRZ	24.7	24.7	24.8	24.8	24.8
Station 07 RTZ	46.1	46.0	46.0	46.0	46.0
Station 09 RMZ	185.3	186.7	187.4	187.4	187.4
Station 17 Dam	514.0	622.4	681.4	685.3	684.0
Station 20 WTZ	43.9	43.9	43.9	43.9	43.9
Station 24 WRZ	36.5	36.5	36.5	36.5	36.5

TABLE 8. Simulated Effects of Chloride Management (i.e., TDS) on Forage Fish Production

	Percent reduction in TDS concentrations (%)				
	0	**4**	**8**	**12**	**16**
	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y
Silverside					
Station 01 RRZ	46.1	46.4	46.7	46.9	47.1
Station 07 RTZ	47.9	48.2	48.5	48.8	49.0
Station 09 RMZ	53.4	53.6	53.8	54.2	54.6
Station 17 Dam	31.5	12.5	2.7	2.0	2.1
Station 20 WTZ	133.9	133.9	133.9	133.9	133.9
Station 24 WRZ	89.0	89.0	89.0	89.0	89.0
Threadfin shad					
Station 01 RRZ	95.4	95.9	96.4	96.8	97.1
Station 07 RTZ	77.8	78.6	79.0	79.4	79.8
Station 09 RMZ	99.4	98.7	98.5	98.8	99.2
Station 17 Dam	43.4	12.7	2.7	2.0	2.1
Station 20 WTZ	165.7	165.7	165.7	165.7	165.7
Station 24 WRZ	109.6	109.6	109.6	109.6	109.6
Gizzard shad					
Station 01 RRZ	127.1	127.6	128.1	128.5	128.9
Station 07 RTZ	102.6	103.4	103.9	104.4	104.8
Station 09 RMZ	137.1	136.2	135.9	136.4	136.7
Station 17 Dam	77.9	29.4	7.2	5.4	5.5
Station 20 WTZ	223.9	223.9	223.9	223.9	223.9
Station 24 WRZ	158.2	158.2	158.2	158.2	158.2

TABLE 9. Simulated Effects of Chloride Management (i.e., TDS) on Striped Bass Production

	Percent reduction in TDS concentrations (%)				
	0	**4**	**8**	**12**	**16**
	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y
Young-of-year					
Station 01 RRZ	20.6	20.6	20.6	20.5	20.5
Station 07 RTZ	30.8	30.8	30.8	30.8	30.8
Station 09 RMZ	22.3	22.3	22.4	22.4	22.5
Station 17 Dam	12.8	6.9	3.5	3.4	3.4
Station 20 WTZ	34.2	34.2	34.2	34.2	34.2
Station 24 WRZ	20.7	20.7	20.7	20.7	20.7
Juveniles					
Station 01 RRZ	30.3	30.5	30.7	30.9	31.1
Station 07 RTZ	28.3	28.6	28.8	29.0	29.1
Station 09 RMZ	32.9	32.9	33.0	33.1	33.3
Station 17 Dam	26.3	13.3	4.3	3.4	3.5
Station 20 WTZ	47.0	47.0	47.0	47.0	47.0
Station 24 WRZ	25.9	25.9	25.9	25.9	25.9
Adults					
Station 01 RRZ	4.3	4.3	4.3	4.3	4.3
Station 07 RTZ	12.7	12.8	12.8	12.9	12.9
Station 09 RMZ	19.2	19.2	19.2	19.3	19.3
Station 17 Dam	18.6	10.0	3.2	2.4	2.5
Station 20 WTZ	12.6	12.6	12.6	12.6	12.6
Station 24 WRZ	4.8	4.8	4.8	4.8	4.8

TABLE 10. Simulated Effects of Chloride Management (i.e., TDS) on Other Fish Production

	Percent reduction in TDS concentrations (%)				
	0	**4**	**8**	**12**	**16**
	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y	gC/m^2/y
Freshwater drum					
Station 01 RRZ	123.5	124.1	124.7	125.1	125.6
Station 07 RTZ	128.3	129.1	129.9	130.6	131.2
Station 09 RMZ	184.5	184.2	184.4	185.3	186.2
Station 17 Dam	133.1	52.8	14.2	11.3	11.5
Station 20 WTZ	256.6	256.6	256.6	256.6	256.6
Station 24 WRZ	83.1	83.1	83.1	83.1	83.1
Smallmouth buffalo					
Station 01 RRZ	8.9	8.9	8.8	8.8	8.7
Station 07 RTZ	9.0	8.9	8.9	8.9	8.9
Station 09 RMZ	9.3	9.3	9.2	9.2	9.1
Station 17 Dam	1.6	1.6	1.6	1.5	1.5
Station 20 WTZ	5.4	5.4	5.4	5.4	5.4
Station 24 WRZ	4.4	4.4	4.4	4.4	4.4
Blue catfish					
Station 01 RRZ	4.2	4.2	4.2	4.2	4.2
Station 07 RTZ	4.4	4.4	4.4	4.4	4.4
Station 09 RMZ	4.2	4.1	4.1	4.1	4.1
Station 17 Dam	0.9	0.9	0.9	0.9	0.9
Station 20 WTZ	1.9	1.9	1.9	1.9	1.9
Station 24 WRZ	1.0	1.0	1.0	1.0	1.0

TABLE 10 Simulated Effects of Chloride Management (i.e., TDS) on Other Fish Production (Continued)

	Percent reduction in TDS concentrations (%)				
	0	**4**	**8**	**12**	**16**
	$gC/m^2/y$	$gC/m^2/y$	$gC/m^2/y$	$gC/m^2/y$	$gC/m^2/y$
White bass					
Station 01 RRZ	23.4	23.5	23.6	23.7	23.8
Station 07 RTZ	24.3	24.4	24.5	24.6	24.7
Station 09 RMZ	34.5	34.4	34.5	34.6	34.7
Station 17 Dam	29.4	14.8	4.5	3.5	3.6
Station 20 WTZ	51.2	51.2	51.2	51.2	51.2
Station 24 WRZ	37.6	37.6	37.6	37.6	37.6