The Salton Sea Centennial Symposium

Developments in Hydrobiology 201

Series editor
K. Martens

The Salton Sea Centennial Symposium

Proceedings of a Symposium Celebrating a Century of Symbiosis Among Agriculture, Wildlife and People, 1905–2005, held in San Diego, California, USA, March 2005

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Reprinted from Hydrobiologia, Volume 604 (2008)

Springer

Library of Congress Cataloging-in-Publication Data

A C.I.P. Catalogue record for this book is available from the Library of Congress.

ISBN-13: 978-1-4020-8805-6

Published by Springer,
P.O. Box 17, 3300 AA Dordrecht, The Netherlands

Cite this publication as Hydrobiologia vol. 604 (2008).

Cover illustration: The Salton Sea in January 2008, as seen from North Shore, California. Photo credit: Doug Barnum, US Geological Survey.

Printed on acid-free paper

Printed in the Netherlands

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DOI 10.1007/s10750-008-9314-3

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Preface

Stuart H. Hurlbert

The Salton Sea is the largest lake in California and occupies a below sea level depression in the desert just north of the border with Mexico. The current lake formed accidentally in 1905 as a result of a breaching of diversion structures on the Colorado River. Ever since then it has persisted, despite the hot desertic climate, as 1.6 billion cubic meters of agricultural wastewaters flow into it every year. From its earliest days, this lake was prime habitat for fish and wildlife. Over time, residential communities, marinas, a wildlife refuge, and a state park developed along its shoreline, and it became a mecca for sport fishing, boating, water sports, birdwatching, and camping. Over the last three decades, however, fluctuating lake levels, rising salinity and continued eutrophication have caused increasing problems for both wildlife and man.

In the late 1990s, in response mainly to large fish kills and heavy bird mortalities at the Salton Sea, significant funding finally became available for scientific analysis of the ecology of the Sea and of possible solutions to its problems. Much of the resultant new scientific information on the Salton Sea was published in four earlier volumes of *Hydrobiologia* (Zheng et al., 1998; Melack et al., 2001; Barnum et al., 2002; Melack, 2007).

In order to stimulate synthesis and publication of new findings on the Salton Sea ecosystem, the Salton Sea Centennial Symposium was convened in San Diego in March 2005, under the auspices of the U.S. Geological Survey (USGS) Salton Sea Science Office and San Diego State University's Center for Inland Waters, with support from the Water Education Foundation, the SDSU President's Leadership Fund, and the California Department of Water Resources. Additional support for publishing the symposium papers was provided by the Salton Sea Authority. Planning of the scientific program was carried out by Douglas Barnum (USGS Salton Sea Science Office) and the editor. The present volume contains papers based on 14 of the 35 oral presentations made at that symposium. A companion set of symposium papers with new biological findings on the Salton Sea is being published simultaneously as a special issue of the journal *Lake and Reservoir Management* (23(5), 2007).

Several major speakers and events at the symposium are not represented in this collection of papers. Experts on three other large, saline, aquatic ecosystems were keynote speakers at the symposium. Enrique Bucher (Professor of Ecology, National University of Cordoba, Cordoba, Argentina) talked on Mar Chiquita, a 5,000 km^2 lake in northern Argentina. Philip Micklin (Professor of Geography, Western Michigan University, Kalamazoo, Michigan) presented a talk on the Aral Sea. Jose Campoy Favela

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(Director, Northern Gulf of California and Colorado River Delta Biosphere Reserve, San Luis Rio Colorado, Sonora, Mexico) gave a talk on wetlands in the lower portion of the Colorado River delta in Mexico. We were also honored by a brief visit and discussions with a delegation of 16 scientists and officials from Uzbekistan, Turkmenistan, Kazakhstan, and Tajikistan, countries bordering the Aral Sea. Finally, Rita Schmidt Sudman (Executive Director, Water Education Foundation) organized and moderated a lively panel discussion on the topic, *The Link Between Water Supply, Science, Restoration and The Law.*

The future of the Salton Sea ecosystem is uncertain, but it certainly is headed toward uncharted waters. The ecosystem analyzed in these studies already has changed. Water inflows are declining. The fish are essentially gone except for the hardy tilapia. Salinity is presently 47–48 g l^{-1}, matching the previous historic high of the mid-1930s, and continues to rise. We may intervene or not, but the Salton Sea ecosystem of the last half century is no more, and just as we understood it better than ever before!

Much of what we have learned about it has proved useful in at least developing plans for a brighter future. After many years of study and discussion involving large numbers of stakeholders, the California Resources Agency has put forward its preferred alternative for a "Salton Sea Ecosystem Restoration Program" (available at http://www.saltonsea.water.ca.gov/). This ambitious and complex plan defies concise description. It envisions use of dikes, berms, canals, and other elements to create: a narrow, 182 km^2, horseshoe-shaped salt lake, stabilized at 30–40 g l^{-1}, around the perimeter of the northern two-thirds of the present lake; a 251 km^2 complex of tiered, shallow, saline (20–200 g l^{-1}) wetlands, mostly around the southern end of the present lake; and a large central area that eventually will consist of exposed lakebed or playa (429 km^2) and two very shallow brine lakes (69 km^2). Aquatic habitat diversity will be greatly increased in the region, though total area of aquatic habitat will be about half that of the present lake (930 km^2). Project capital costs are estimated at USD 8.9 billion, with post-construction operating and maintenance costs estimated at USD142 million per year. Many large technical and financial issues concerning the proposed project are not yet resolved. The state of California is in a fiscal crisis at this time; its population continues to grow, water demand is up, and climate models predict increasing aridity over large portions of the American Southwest. Whatever will be the consequences of these colliding plans and forces for the Salton Sea, they will—at a minimum—be "interesting" over the next decades.

Acknowledgments

The sponsoring and funding agencies are acknowledged above. Their moral support and financial contributions were critical to the success of the symposium and were obtained primarily through the efforts of Douglas Barnum of the USGS Salton Sea Science Office. Rita Schmidt Sudman, Judy Maben, and Sue McClurg of the Water Education Foundation did superb jobs of overseeing the registration process and all arrangements for facilities and services at the symposium site, moderating the panel discussion, and producing a summary of the abstracts of all talks given (WEF, 2005). Joan Dainer and Jim Zimmer helped with various logistical matters. A Cooperative Agreement between the USGS Salton Sea Science Office and the SDSU Research Foundation supported the editor's involvement in planning of the scientific program and the early editorial work on this volume and its companion in *Lake and Reservoir Management.*

Each manuscript submitted for this volume was reviewed by 2–4 referees plus the editor, and most manuscripts underwent extensive revision before being accepted. Many thanks to all those referees whose constructive suggestions helped authors to "put their best foot forward": K.K. Bertine, J. Bloesch, M. Bowes, C. Brauner, M.T. Brett, M.J. Cohen, D.M. Dexter, M. Dittrich, J. Fram, C. Franson, E. Furlong, R. Gersberg, M. Gurol, K. Hoef-Emden, G.C. Holdren, K.A. Hovel, A.H. Hurlbert, R. Jellison, S. Kelly, L.Y. Lewis, T.W. Lyons, L. Majewski, A.K. Miles, S. McIntyre, E. McNaughton, M.F. Moreau, C.R. Phillips, T.S. Presser, K.M. Reifel, D.M. Robertson, T.E. Rocke, D. Schlenk, R.A. Schroeder, J.L. Scott, J.G. Setmire, P. Smith, V.H. Smith, B.K. Swan, M.A. Tiffany, D. Trolle, D.L. Valentine, J.M. Watts, E. Welch, W.A. Wurtsbaugh. A final thanks to the authors themselves for their contributions, their patience, and their stoic courtesy in putting up with unending editorial suggestions.

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Hydrobiologia (2008) 604:5–19
DOI 10.1007/s10750-008-9321-4

SALTON SEA

Response in the water quality of the Salton Sea, California, to changes in phosphorus loading: an empirical modeling approach

Dale M. Robertson · S. Geoffrey Schladow

Abstract Salton Sea, California, like many other lakes, has become eutrophic because of excessive nutrient loading, primarily phosphorus (P). A Total Maximum Daily Load (TMDL) is being prepared for P to reduce the input of P to the Sea. In order to better understand how P-load reductions should affect the average annual water quality of this terminal saline lake, three different eutrophication programs (BATHTUB, WiLMS, and the Seepage Lake Model) were applied. After verifying that specific empirical models within these programs were applicable to this saline lake, each model was calibrated using water-quality and nutrient-loading data for 1999 and then used to simulate the effects of specific P-load reductions. Model simulations indicate that a 50% decrease in external P loading would decrease near-surface total phosphorus concentrations (TP) by 25–50%. Application of other empirical models demonstrated that this decrease in loading should decrease near-surface chlorophyll *a* concentrations (Chl *a*) by 17–63% and increase Secchi depths (SD) by 38–97%. The wide range in estimated responses in Chl *a* and SD were primarily caused by uncertainty in how non-algal turbidity would respond to P-load reductions. If only the models most applicable to the Salton Sea are considered, a 70–90% P-load reduction is required for the Sea to be classified as moderately eutrophic (trophic state index of 55). These models simulate steady-state conditions in the Sea; therefore, it is difficult to ascertain how long it would take for the simulated changes to occur after load reductions.

Keywords Eutrophication · TMDL · Chlorophyll · Secchi depth · Saline lake

Guest editor: S. H. Hurlbert
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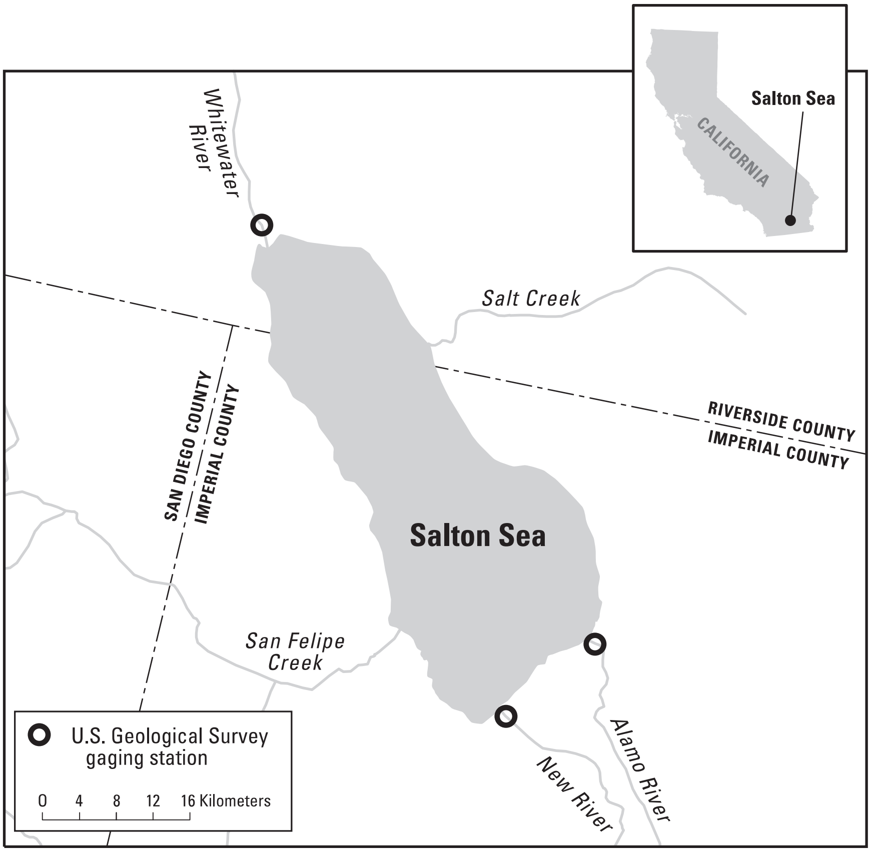
Introduction

The Salton Sea, the largest lake in California, USA, is located in the southeastern desert region (Fig. 1). The current Salton Sea was formed during a 17-month period from October 1905 to February 1907 following a high-flow event in the Colorado River that resulted in the river breaching a levee of an early irrigation diversion channel and flooding the low-lying desert of Imperial and Riverside counties. The Salton Sea is a terminal water body (lakes with inlets

Fig. 1 Salton Sea, California, with major tributaries and U.S. Geological Survey gaging stations identified



but no outlets) that has no natural tributaries and would have completely evaporated if it were not for receiving agricultural discharges from the Coachella (Whitewater River), Imperial (Alamo and New rivers), and Mexicali valleys along with municipal and industrial effluent from Mexicali, Mexico (New River). One of the major functions of the Sea, since the mid 1920s, is to serve as a sump for agricultural wastewater for the Imperial and Coachella Valleys (Redlands Institute, 2002). As a result of the Salton Sea being a terminal water body in an area with very high evaporation, the salinity of the sea has increased to over 44 g/l in 2000 (Schroeder et al., 2002).

As a result of the high external loading of nutrients, the Salton Sea has become a eutrophic to hypereutrophic waterbody characterized by high nutrient concentrations, high algal biomass as demonstrated by high chlorophyll *a* concentrations (Chl *a*), high fish productivity, low clarity, frequently very low dissolved oxygen concentrations, massive fish kills, and noxious odors (Holdren & Montaño, 2002a). Unlike many other saline systems, its eutrophic condition is believed to be controlled or limited by the phosphorus (P) concentrations in the Sea rather than nitrogen (N) (Holdren & Montaño, 2002a; Robertson et al., 2008, this issue). Most of the P input to the Salton Sea on an annual basis is from tributary loading (Robertson et al., 2008, this issue). Nutrient and sediment concentrations and loads in the tributaries to the Sea are very high because they carry agricultural discharge and, in addition, the New River carries municipal and industrial effluent from Mexicali, Mexico. Although tributary loading is believed to be the major source of nutrients, it is uncertain how important the release of P from the sediments of the Sea is to driving short-term algal blooms and anoxic conditions. In many lakes, P is released from the sediments in deep areas of the lake during low redox conditions or released during with the resuspension of sediment in shallow areas of the lake associated with strong winds and waves. In order to reduce the P loading to the Sea, a P Total Maximum Daily Load (TMDL; a detailed plan to reduce P loading to a specified value; USEPA, 2000) is being prepared by the Colorado River Regional Water Quality Control Board (CRRWQCB, 2006), that, if implemented, is intended to improve the water quality and reduce the problems associated with eutrophication.

In order to aid in developing the TMDL for the Salton Sea, a better understanding is needed of how nutrient-load reductions will affect the Sea's water quality. One way to determine how much of the P that enters a waterbody needs to be eliminated to improve water quality or to determine how increases in P loading may degrade water quality, is through the use of empirical or semi-empirical models that relate P loading with measured water quality (USEPA, 2000; National Research Council, 2001; Minnesota Pollution Control Agency, 2006). These types of models have been developed on the basis of comparisons between hydrologic and nutrient loads determined for many different waterbodies and specific measures describing water quality, such as near-surface total phosphorus (TP), Chl *a*, and Secchi depths (Walker, 1986; Chapra, 1997). Several of these empirical models are contained within the Wisconsin Lakes Modeling Suite (WiLMS; Panuska & Kreider, 2002) and the semi-empirical models within BATHTUB (Walker, 1996). In BATHTUB, steady-state water- and nutrient-balance calculations are performed in a spatially segmented hydraulic network to account for advective and diffusive transport, and nutrient sedimentation. Changes in water-quality conditions are then predicted using empirical relationships derived from assessments of data in various lakes and reservoirs (Walker, 1985, 1986). Most of these empirical models were developed with data from non-terminal, freshwater lakes, and reservoirs, which typically differ both chemically and biologically from their saline counterparts. No empirical eutrophication models have been specifically developed from data on terminal saline lakes. Therefore, it is important to determine if the existing empirical models are applicable to saline lakes, specifically the Salton Sea, prior to using them to determine how a saline lake should respond to changes in nutrient loading.

Although dynamic or deterministic models are useful for understanding processes that occur in a waterbody during the year(s) examined, specific processes could dramatically change in response to changes in nutrient concentrations, such as those associated with changes in specific species or populations of fish. As these changes cannot usually be foreseen, the altered processes that would not be incorporated into the dynamic model make long-term predictions questionable. Therefore, it is advantageous to use empirical models, based on the differences that have been documented in a wide range of systems, to predict the long-term effects of nutrient-load reductions, rather than using more complicated dynamic models.

Two empirical modeling studies have been previously conducted on the Salton Sea to determine if its water quality responds to nutrient loading like other lakes and reservoirs (Setmire et al., U.S. Geological Survey (USGS), pers. comm.) and to determine how the Sea should respond to incremental changes in P loading (Anderson, 2003). Both these studies were primarily based on the water quality measured in the Sea and the estimated tributary P loading in 1999 by Holdren & Montaño (2002a, b). Setmire et al. (USGS, pers. comm.) applied several of the empirical models contained in WiLMS to the Salton Sea and concluded that the TP in the Sea was lower than what would be expected for a lake with the Sea's morphometry and 1999 P-loading rates.

Anderson (2003) applied several of the P (first- and second-order P-settling algorithms), Chl *a*, and Secchi depth models within BATHTUB, and a coupled sediment/water-column model that was developed for the Salton Sea to determine how the Sea's water quality should respond to incremental changes in P loading. The coupled sediment/water-column model separately apportioned loading to external sources and internal recycling, and allowed for P removal via settling. In all the P models, internal loading of P from the sediments was included at a rate of 3.2 $mg/m^2/day$ (for the 1999 base-case scenario), based on results of sediment core incubations in the laboratory. This resulted in the internal P load from the sediment being slightly less than the external P load from the watershed. Anderson found that the coupled sediment/water-column model and BATHTUB models accurately simulated the TP measured in the Salton Sea in 1999, but to do so the models required large calibration factors. Based on the results of these models, a 50% decrease in external P loading should cause TP in the Sea to decrease by $\sim$30% (second-order model) to 50% (first-order and coupled sediment/water-column model). Based on the coupled sediment/water-column model, it was concluded that a 25% reduction in external TP loading would reduce the total P loading to the lake by 50% and that complete responses to changes in external loading should occur in as little as 3 years.

There are several problems with the way the empirical models were applied and the conclusions that were made in the previous studies. First, several of the models used by Setmire et al. (USGS, pers. comm.) were not applicable to terminal lakes. Second, based on the water-quality data measured in this saline lake, there is little or no net internal P loading from the deep sediments (Holdren & Montaño, 2002a; Robertson et al., 2008, this issue); however, there may be substantial input of P associated with the resuspension of shallow sediment. Even if there was internal P loading, as assumed by Anderson (2003), a 25% reduction in external P loading would reduce external loading by 25% and indirectly reduce part of the internal loading, and thus would only reduce the total P loading by ≤25%. Finally, if extensive internal P loading was present in the Sea, then internal P loading would persist much longer than 3 years and possibly reduce the immediate effects of external P-load reductions. Internal P loading has been found in many lakes to delay the effects of P-load reductions (for example, Shagawa Lake, Minnesota; Larsen et al., 1981).

Methods

In order to predict how the average whole-lake water quality of the Salton Sea should respond to changes in P loading, models in three eutrophication programs were applied to the Sea with the conditions measured in 1999 (Holdren & Montaño, 2002a), which was the base case for calibration and the case with which to compare other simulations. The eutrophication programs included BATHTUB (Walker, 1996), WiLMS (Panuska & Kreider, 2002), and the Wisconsin Department of Natural Resource's (WDNR's) Seepage Lake Model (J. Panuska, WDNR, pers. comm.). Models within each of these programs were calibrated, if necessary, and then applied with changes in P loading ranging from reductions in P loadings from tributaries and drainage canals by 10–100% and to increases in loading from these sources by 10–100%. 1999 was used as a base case and for calibration because it is the only year that had sufficient coinciding information on external loading and in-Sea water quality. Since the Sea is believed to be potentially limited by P (Holdren & Montaño, 2002a; Robertson et al., 2008, this issue), only the effects of alterations to P loads (TP and orthophosphate-P (OP) concentrations) were simulated; loadings of all other constituents were kept similar to those measured in 1999. Although there is considerable temporal and spatial variability in the water quality of the Salton Sea, these models were used to predict changes in the average whole-lake near-surface water quality.

Model input

Four types of data are required as input for these empirical eutrophication models: morphometric and physical characteristics, hydrologic, nutrient-loading, and water-quality data (Tables 1 and 2). The time period for the hydrologic and nutrient-loading data used in BATHTUB is dependent on the P-turnover ratio or the number of times the P mass in the lake is displaced during the averaging period. BATHTUB should be run for a period that results in a P-turnover ratio >2. Annual simulations resulted in the P-turnover ratio of 2.02. Annual water-loading rates and volumetrically weighted mean concentrations for 1999 were obtained from Robertson et al. (2008, this issue) except those for the Whitewater River. Volumetrically weighted concentrations for the Whitewater River for 1999, based on the long-term calibration of a loading model, appeared to overestimate the concentrations for 1999. Given that only 2 of 18 TP concentrations measured in the Whitewater River by Reclamation in 1999 exceeded 1.0 mg/l (Holdren & Montaño, 2002b), an average concentration of 1.004 mg/l appeared to be too high. Therefore, a P load for the Whitewater River for 1999 was obtained by multiplying the monthly or biweekly TP concentrations (measured by Reclamation) by the coinciding total streamflows (measured by the USGS), and then a volumetrically weighted concentration was obtained by dividing the total P load by the total annual flow. This resulted in an average TP concentration of 0.88 mg/l for the Whitewater River. Average TP, OP, total nitrogen (TN), and inorganic N in the tributaries to the Salton Sea for 1999 are listed in Table 1. The total annual P load to the Salton Sea in 1999 was $\sim$1,440,000 kg, which equates to a unit-area yield from the basin of $\sim$67 kg/km^2 and an areal loading to the Sea of $\sim$1.5 g/m^2/year.

Even though loading data are summarized for the entire year, BATHTUB typically estimates water quality only for the growing season (May through

Table 1 Morphometric, hydrologic, and tributatry inputs for the empirical models for the Salton Sea, California, for 1999 (water quality for the base-case scenario)

Salton Sea morphometric and physical characteristics data				
Average water level[a]	69.4 m below sea level	Width	24 km	
Area[b]	980.8 km^2	Volume[b]	9.257×10^9 m^3	
Mean depth[b]	9.44 m	Mixed layer depth[c]	6.6 m	
Length	56 km	Non-algal turbidity factor	0.45	
Precipitation, evaporation, and hydraulic flushing				
Precipitation[d]	0.076 m	13.5 kg of total phosphorus per km^2/year		
Evaporation[d]	1.663 m			
Hydraulic flushing rate	0.18 $year^{-1}$			
Tributary inflow volumes and nutrient concentrations: loading information				
Parameter	Alamo River	New River	Whitewater River	Direct Drains[g]
Flow ($\times$ 10^6 m^3)[e]	761	603	65	128
Total phosphorus (mg/l)[e]	0.716	1.218	0.880	0.716
Orthophosphate-P (mg/l)[f]	0.409	0.697	0.710	0.409
Total nitrogen (mg/l)[f]	9.263	8.289	16.367	15.072
Inorganic nitrogen (mg/l)[f]	7.713	7.267	15.072	7.713

[a] Based on elevation data obtained from the Bureau of Reclamation (P. Weghorst, Bureau of Reclamation, unpubl. raw data)

[b] Based on bathymetric data obtained from the Bureau of Reclamation (P. Weghorst, Bureau of Reclamation, unpubl. raw data)

[c] Based on algorithms in BATHTUB (Walker, 1996) using the morphometry and water quality measured in the Salton Sea

[d] Based on estimated precipitation and evaporation from Robertson et al. (2008, this issue)

[e] From loading estimated by Robertson et al. (2008, this issue) using U.S. Geological Survey flow data for the gages identified in Fig. 1 and data collected by various agencies, except the Whitewater River that is described in the text

[f] Based on average monthly concentrations for 1999 (Holdren and Montaño, 2002b)

September in temperate lakes and reservoirs). The models within WiLMS and the Seepage Lake model also use annual hydrologic and P loadings, but typically simulate the water quality for different discrete seasons. Since algal growth occurs throughout the year in the Salton Sea, output from all the models were compared with the average monthly water quality for the entire 1999 year (Table 2). All these models estimate the water quality of the upper mixed layer of the water column; therefore, results of model simulations were compared with measured near-surface water-quality data. Additional P from internal loading (chemical diffusion of P from the deep sediments) was not included as a P source for BATHTUB, WiLMS, and the Seepage Lake Model because: (1) based on the water quality measured in the Sea, there was minimal net internal P loading (increased TP immediately above the bottom was not detected even during anoxic periods) and (2) even if there was internal P loading that was typical of other lakes and reservoirs (as suggested by Anderson and Amrhein, 2002), most empirical models (such as those used in this study) inherently incorporate this source and, therefore, additional P from internal

Table 2 Monthly average near-surface water quality for the Salton Sea in 1999

Constituent	Concentration/depth	Trophic state index value
Total phosphorus	0.077 mg/l	67
Chlorophyll *a*	33 μg/l	65
Secchi depth	0.81 m	63
Orthophosphate-P	0.022 mg/l	–
Total nitrogen	3.80 mg/l	–
Organic nitrogen	2.58 mg/l	–
Total suspended sediment	45.7 mg/l	–

All values were based on data collected by the Bureau of Reclamation (Holdren and Montaño, 2002b), except chlorophyll *a* concentrations that were obtained from San Diego State University (Tiffany et al., in press) and represent a whole-lake average value loading should only be added when a lake/reservoir has abnormally high internal P-loading rates (Walker, 1996, p. 4-24).

Algorithms and calibration of models

Phosphorus

Most empirical models used to estimate TP, including those in this study, are based on the Vollenweider eutrophication modeling approach (Vollenweider, 1975). With this approach, the average TP in the lake or reservoir is estimated as a function of the annual areal loading rate of P (L), the mean depth of the lake (z), the P-settling coefficient (σ), and the hydraulic flushing rate (ρ) (Eq. 1). The main difference in most of the algorithms used to estimate average TP is in how σ is estimated.

$$\mathrm{TP} = L/(z\ (\sigma + \rho)). \tag{1}$$

When applying BATHTUB, specific models (algorithms) must be selected to simulate each water-quality constituent. The algorithms chosen to simulate TP, Chl *a*, and Secchi depth (SD) are given in Table 3. For TP, three different algorithms were chosen: one first-order P-settling algorithm and two second-order P-settling algorithms (available-P and decay-rate algorithms). The first-order algorithm assumes that changes in particulate concentrations over time are directly proportional to the concentration of the particulates in the water column (σ is set equal to 1.0). The second-order algorithms assume that changes in concentrations over time are proportional to the square of the particulate concentration (TP is computed with Eq. 2). The second-order, available-P algorithm (σ computed with Eq. 3) is the default algorithm in BATHTUB and the one most generally applicable to lakes and reservoirs (Walker, 1996). This algorithm performs mass-balance calculations on "available P," a weighted sum of OP and non-OP, and places a higher emphasis on OP (the more biologically available component) (Walker, 1996). Walker (1996) states that the second-order, decay-rate algorithm (σ computed with Eq. 4) may be most applicable to lakes with long residence times, such as the Salton Sea. The first-order and second-order decay-rate algorithms were used by Anderson (2003) to simulate changes in TP in the Salton Sea in response to various changes in P loading.

$$\mathrm{TP} = \{-1 + [1 + 4\ \sigma L/(z\ \rho^2)]^{0.5}\}/(2\sigma/\rho) \tag{2}$$

where

$$\sigma = 0.17\ Q_s/(Q_s + 13.3) \tag{3}$$

or

Table 3 Models used to estimate changes in the water quality of the Salton Sea, California

Eutrophication program	Model (algorithm) description	Pre-calibration concentration, µg/l	Post-calibration concentration, µg/l	Calibration factor
Total phosphorus—measured 0.077 mg/l				
BATHTUB	First order P-settling	0.156	0.078	2.00
BATHTUB	Second order P-settling, available P	0.078	0.078	1.00
BATHTUB	Second order P-settling, decay rate	0.084	0.077	1.18
WiLMS	Canfield & Bachman (1981)	0.095	0.077	−23%
Seepage Lake model				$V_s = 19.0$
Chlorophyll a—measured 33 µg/l				
BATHTUB	Based on P and N concentrations, light, and flushing rate	19	32	1.65
WiLMS/Seepage Lake model	Carlson TSI equations	40	33	−20%
Secchi depth—measured 0.81 m				
BATHTUB	Based on P concentration and turbidity	0.70	0.80	1.25
WiLMS/Seepage Lake model	Carlson (1977) TSI equations	0.60	0.81	−25%

The pre- and post-calibration values and calibration factors used for each empirical model are listed. V_s is a settling velocity

$$\sigma = 0.056\ Q_s/[\mathrm{FOP}\ (Q_s + 13.3) \quad (4)$$

where Q_s is the maximum of the total depth $\times$ ρ or 4; FOP is fraction of tributary OP loading to TP loading.

Of the 13 empirical models contained within WiLMS, only three of the models were relatively insensitive to the residence time of water in the lake; most of the models are not capable of simulating water quality in terminal lakes, lakes with no outlets and therefore very long residence times. Simulated TP from the 10 other models increased dramatically as the residence time increased. Of the three models relatively insensitive to residence time, only the Canfield & Bachman (1981) natural-lake model (σ for Eq. 1 is computed with Eq. 5) was applicable to the hydrology, loading rates, and TP concentrations of the Salton Sea,

$$\sigma = 0.162\ (L/z)^{0.458}. \quad (5)$$

The Seepage Lake Model (J. Panuska, WDNR, pers. comm.), specifically developed for terminal lakes, estimates a lake's average TP as a function of areal P loading (L), areal water loading (Q_A), and an apparent settling velocity (V_s) for P (Eq. 6). The value of V_s required to accurately estimate the measured TP in a lake can be used to infer processes occurring in a lake that control TP. V_s values less than 0 m/year are indicative of high internal P loading, values between 0 and 6 m/year are indicative of moderate internal P loading, values between 7 and 12 m/year are indicative of low internal P loading, values between 12 and 20 m/year are indicative of no internal P loading and that marling (co-precipitation with calcium carbonate) or other removal mechanisms occur for P, and values more than 20 m/year are indicative of very high marling or extensive P removal (J. Panuska, WDNR, pers. comm.). A V_s value of 19 m/year was required to estimate the measured TP of 0.077 mg/l in the Salton Sea. Therefore, the Seepage Lake Model indicates that there is no net internal P loading and suggests that there are removal mechanisms occurring in the Sea.

$$\mathrm{TP} = L/(Q_A + V_s). \quad (6)$$

None of these P algorithms was specifically developed using data for terminal saline lakes, such as the Salton Sea; however, they were developed using data from lakes and reservoirs with a wide range of environmental and hydraulic conditions. Therefore, their applicability to the Salton Sea can be evaluated by the calibration coefficients required to accurately simulate the water quality measured in the Sea in 1999 (Table 3). Predictions made with the models prior to and following calibration are also shown in Table 3. All these coefficients are within the typical range applied for most lakes (Walker, 1996). By not including the internal P loading (as included in the models by Anderson, 2003), only the first-order model required much, if any, calibration for predicting TP. Therefore, these empirical P models appear to be able to simulate changes in TP in this terminal saline lake.

Chlorophyll a *and Secchi depth*

The TP estimated with BATHTUB, WiLMS, and the Seepage Lake Model, were then used to estimate Chl *a* and SD. Chl *a* was estimated with BATHTUB using an algorithm that is a function of TP, TN, light, and flushing rate (similar to that used by Anderson, 2003). SDs were predicted with BATHTUB using the algorithm that is a function of TP and a non-algal turbidity factor (0.45 m^{-1}, computed within BATHTUB). Non-algal turbidity is the portion of light extinction that is due to factors other than algae, such as inorganic suspended solids, dissolved organic matter, and color (Walker, 1996). The non-algal turbidity factor was held constant in all simulations. The calibration coefficients applied to accurately simulate the Chl *a* and SD for 1999 with BATHTUB are given in Table 3. These calibration coefficients are within the typical range applied for most lakes (Walker, 1996), which again indicates that these empirical models appear to be able to simulate changes in Chl *a* and SD in this terminal saline lake.

The TP concentrations predicted with WiLMS (Canfield & Bachman 1981 model) and the Seepage Lake Model were used to estimate Chl *a* and SD through the use of Carlson's (1977) trophic state index (TSI) equations. In other words, the Chl *a* and SD were computed that yielded similar TSI values as the predicted TP. There are no calibration factors when the Carlson's TSI equations are used to estimate Chl *a* and SD; however, the output can be adjusted to account for model biases by only interpreting the results as a percentage of change from present conditions. In other words, the percent bias found in predicting concentrations or depths for 1999 (Table 3) is removed from all the other predictions. Changes in Chl *a* were also estimated with algorithms by Jones & Bachman (1976) and Dillon & Rigler (1974), but the results (simulated Chl *a* concentrations of 37 and 40 μg/l, respectively) were similar to those of the Carlson TSI equations and therefore are not presented. The uncalibrated models were within 25% of the observed Chl *a* and SD, which is well within the variability found for non-saline lakes, which again suggests that the Salton Sea responds to changes in TP similar to other lakes.

In a previous study, Chl *a* was found to be unusually low in prairie saline lakes in Canada compared to other lakes with similar TP and lower than predicted with empirical models (Campbell & Prepas, 1986). However, the Chl *a* in the Salton Sea are not unusually low and are predicted well by the empirical models used in this study (Table 3). The unusually low concentrations measured in the prairie saline lakes may have been caused by those lakes being N limited compared to the Salton Sea which is usually P limited; therefore, the empirical models that are based on P limitation and used for the prairie saline lakes in Canada were not appropriate for that application.

Results

In order to simulate the effects of changes in P loading, the average TP and OP concentrations (or loads) for each of the tributaries in 1999 (Table 1) were decreased by 10–100% and increased by 10–100%; N concentrations were not modified (average monthly concentrations for 1999 were used in the models). Since the Salton Sea is P limited (Holdren & Montaño, 2002a; Robertson et al., 2008, this issue), altering N concentrations should and had little effect on the simulated results (these results are not presented).

Response in phosphorus concentration

On the basis of results of the first-order P-settling BATHTUB model and the Seepage Lake Model, TP in Salton Sea should have a linear response to a linear change in P loading (Fig. 2). On a percentage basis, results from both models indicate that the percent changes in TP are the same as the percent changes in P loadings, i.e., a 50% decrease in P loading should cause a 50% decrease in TP in the Sea.

On the basis of results of simulations with both second-order P-settling models in BATHTUB and the Canfield & Bachman (1981) model within WiLMS, TP in the Salton Sea should have a relatively linear response to a linear change in P loading, except with very large decreases in P loading for which the changes in TP in the Sea are predicted to be larger (Fig. 2). On a percentage basis, results from these three models indicate that the changes in TP are ~30–40% of the changes in P loadings, except with very large decreases in loading (>80% decreases) for which the decreases in TP in the Sea become non-linear and

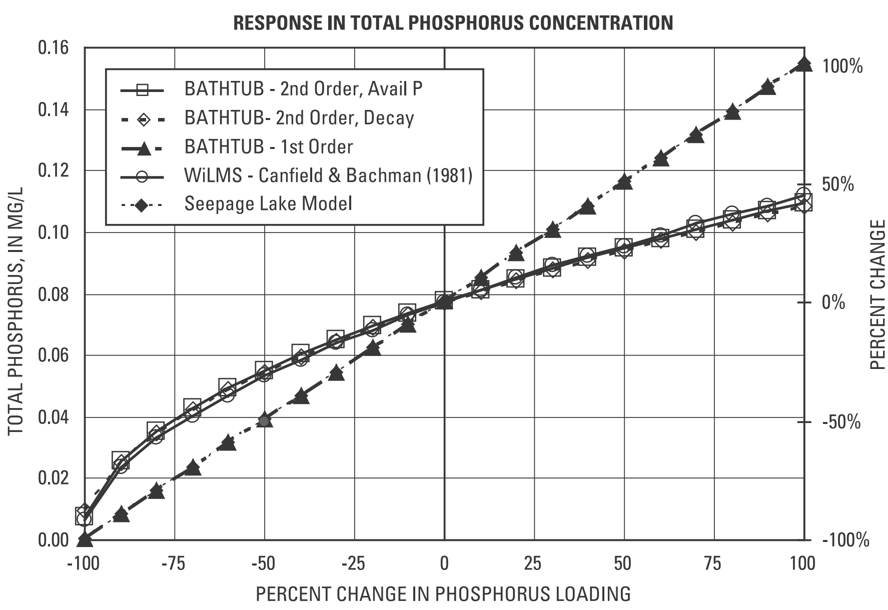


Fig. 2 Modeled changes in near-surface total phosphorus concentrations in response to changes in external phosphorus loading to the Salton Sea

should be larger. A 50% decrease in P loading should cause a ~30% decrease in TP in the Sea.

With a 50% decrease in P loading, results of these models indicate TP in the Salton Sea should decrease from 0.077 to ~0.039–0.055 mg/l (30–50% decrease). The second-order P-settling models in BATHTUB and the Canfield & Bachman model should be the most applicable models to the Salton Sea; therefore, a 50% decrease in P loading, should decrease TP from 0.077 to ~0.055 mg/l (a 30% decrease).

Response in chlorophyll *a* concentration

On the basis of the results of the first-order and the second-order P-settling models in BATHTUB (all three using the same Chl *a* algorithm), Chl *a* should have a larger response to decreases in P loading than to increases in P loading (Fig. 3). The differences in the results from these three models are caused only by the differences in the predicted TP. The largest relative response in Chl *a* occurs with P-load reductions >80%. Results from the second-order P-settling models indicate a smaller response in Chl *a* to changes in P loading than with the first-order P-settling model. Based on results from the BATHTUB models, a 50% decrease in P loading should cause a 17–34% decrease in Chl *a* in the Sea.

Results from the Canfield & Bachman (1981) model within WiLMS and the Carlson TSI equations, indicate a response similar to the first-order P-settling decay model in BATHTUB for decreases in P loading, but larger responses than any of the BATHTUB models for increases in P loading (Fig. 3). A 50% decrease in P loading should cause a ~40% decrease in Chl *a* in the Sea.

Results from the Seepage Lake Model and the Carlson TSI equation indicate a much larger response in Chl *a* than indicated from the other models (Fig. 3). The larger response in Chl *a* from this combination of models was primarily caused by the larger predicted changes in TP. With a 50% decrease in P loading, results from these models indicate a 63% decrease in Chl *a*. Results for 50% increase in P loading indicate an 80% increase in Chl *a*.

With a 50% decrease in P loading, results of these models indicate Chl *a* in the Salton Sea should decrease from 33 to ~12–26 µg/l (a 17–63% decrease). Since the second-order P-settling models and the Canfield & Bachman model appear to be the most applicable models for the Salton Sea, it is expected that a 50% decrease in P loading should decrease average Chl *a* from 33 to ~19–26 µg/l (17–43% decrease). The changes in response to increased P loading based on the Carlson TSI equation may be larger than the other models because these equations

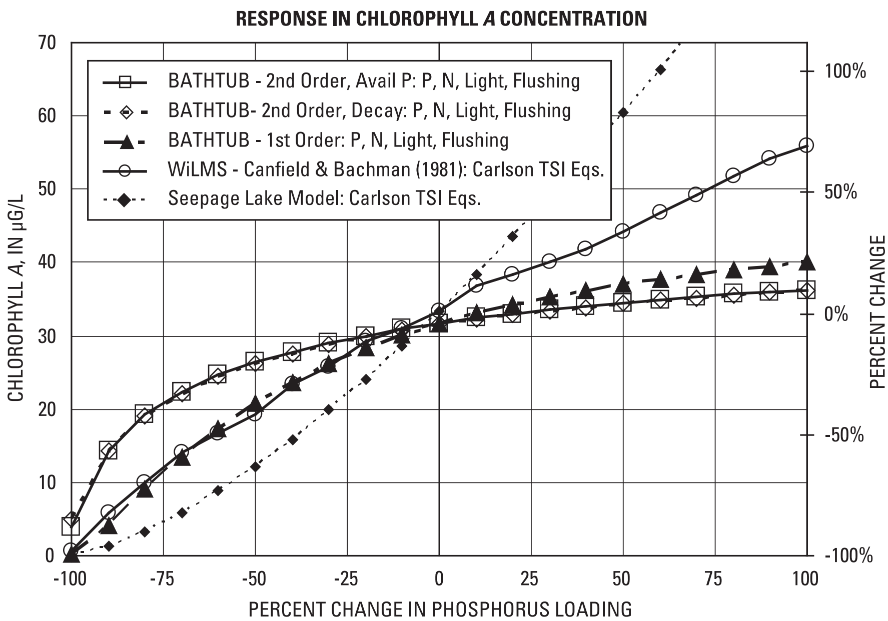


Fig. 3 Modeled changes in near-surface chlorophyll *a* concentrations in response to changes in external phosphorus loading to the Salton Sea

do not incorporate the effects of the high non-algal turbidity that occurs in the Salton Sea.

Response in Secchi depth

Results of all the models indicate that SD (water clarity) is much more responsive to decreases in P loading (especially to decreases in loading by >50%) than to increases in P loading (Fig. 4). Results of all the models indicate that with a 50% increase in P loading, the average SD should decrease by ~0.1–0.3 m. However, the responses are quite variable to decreases in P loading. Results of the BATHTUB models indicate the smallest response in SD, and results of the Seepage Lake Model indicate the largest response.

With a 50% decrease in P loading, results of these models indicate that SD in the Salton Sea should increase from ~0.8 to ~1.1–1.6 m (a 38–97% increase). Since the second-order P-settling models and the Canfield & Bachman model should be the most applicable models for the Salton Sea, it is expected that a 50% decrease in P loading should increase the average SD from ~0.8 to ~1.1–1.2 m (38–50% increase). The main reason for the wide range in the responses in SD with decreased P loading is that the algorithms in BATHTUB assume that the non-algal turbidity will remain constant, whereas, the Carlson TSI equations assume that the turbidity is primarily a function of the amount of algae in the water column.

Response in trophic status

TSI values indicate that Salton Sea is presently eutrophic to hypereutrophic (TSI values ranging from 63 to 67; Table 2). TSI values were computed from the results of the models for changes in P loading. On the basis of simulation results from BATHTUB, WiLMS, and the Seepage Lake Model, any increase in P loading should result in the Sea remaining or becoming more hypereutrophic (Fig. 5). Results from the Seepage Lake Model and Canfield & Bachman model indicate higher TSI values than do the results from the models in BATHTUB. Results of all the models indicate that decreases in P loading can convert the Salton Sea to a eutrophic system (TSI values from 50 to 60). The results indicate that P-load reductions of ~50% (Seepage Lake Model) to 80% (Canfield & Bachman model in WiLMS) are required for the Sea to be classified as moderately eutrophic (a TSI value of 55) with respect to TP; load reductions of ~50% (Seepage Lake Model) to 90% (second-order P-settling BATHTUB models) to be classified as moderately eutrophic with respect to Chl *a*; and P-load reductions of ~50% (Seepage Lake Model) to

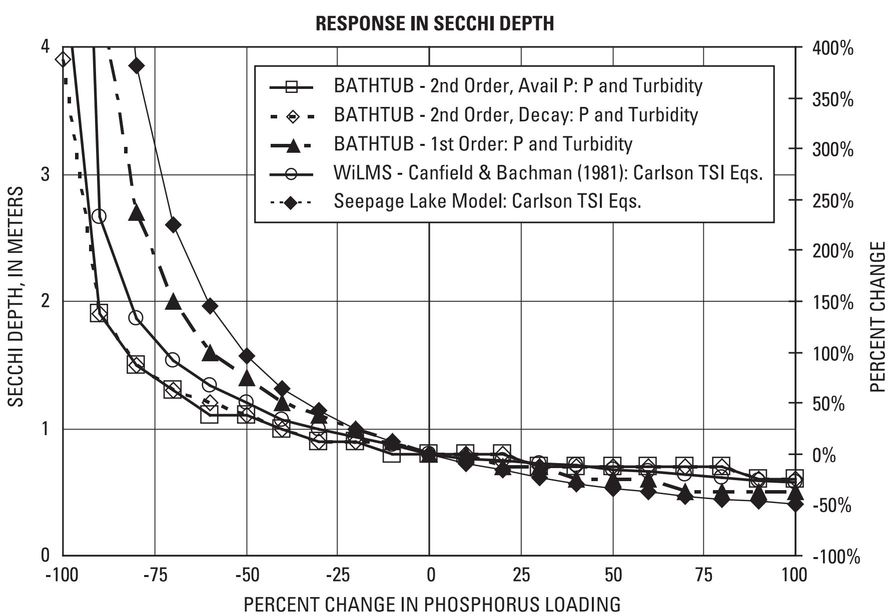


Fig. 4 Modeled changes in Secchi depth in response to changes in external phosphorus loading to the Salton Sea

Fig. 5 Modeled changes in the trophic state of the Salton Sea in response to changes in external phosphorus loading