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Evaluation of empirical models coupled with EUTROMOD for water quality prediction in Kansas reservoirs

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Abstract

Agricultural land use contributes high nutrient and sediment loads to nearby streams, lakes, and reservoirs, which can lead to excessive algal growth and increased siltation. Future intensification of agricultural production could further aggravate water quality concerns. To objectively evaluate the effects of agricultural intensification on future water quality, modeling tools must be able to quantitatively predict the degree to which land use change will affect the trophic state of water bodies. This study evaluated the water quality model EUTROMOD as well as several national and regional in-lake empirical water quality models as predictive tools for analyzing and estimating water quality in 28 Kansas reservoirs of varying size and watershed land use. Model-predicted nutrient loading was used with several in-lake empirical models to predict values for total nitrogen (TN), total phosphorus (TP), and chlorophyll *a* concentrations. Predicted values were then compared to long-term water quality measurements obtained from the Kansas lake and reservoir monitoring program. All models over-predicted concentrations of TN and TP in Kansas reservoirs; however, predictions from the Bachmann TN and Canfield-Bachmann TP in-lake empirical models were most closely coupled to observed trends and had the least error. Two possible sources of model bias were identified: the sedimentation coefficient in the in-lake empirical models and the nutrient loading estimates from the watershed model. Areas of further research are suggested for determining the dominant source of model bias and improving quantitative predictions of water quality in the Midwest, USA.

Key words: EUTROMOD, Kansas, Midwest, modeling, reservoirs, water quality

Introduction

The cultural eutrophication of surface waters is a major water quality problem worldwide (Carpenter et al. 1998, Smith 2000). Agricultural land use, such as row crop production, contributes high nutrient and sediment loads to nearby streams, lakes, and reservoirs, which can lead to excessive algal growth, high turbidity levels, and increased siltation (Isermann 1990, Arbuttle and Downing 2001). The degradation of water quality associated with agricultural land use in the US Central

Great Plains has been documented by numerous studies (Jones et al. 2004, 2008, Carney 2009). Future intensification of agricultural production to fuel the growing demand for biofuel feedstocks could potentially aggravate water quality concerns (Simpson et al. 2008, Dominguez-Faus et al. 2009).

Various watershed models have been used to evaluate the changes in runoff, soil erosion, and nutrient loads related to various land-use scenarios, including intensification of corn in crop rotations in Iowa (Secchi et al. 2011), extensification versus intensification of 18

different crop rotations in Michigan (Love and Nejadhashemi 2011), and the effects of corn stover removal in Indiana (Cibin et al. 2011); however, the water quality impacts to lakes and reservoirs from agricultural land-use change scenarios are not frequently analyzed. Future changes in reservoir water quality are of immense concern to water managers in Kansas because Kansas reservoirs serve approximately 80% of the state's population for drinking water and industrial needs. The hypothesis that an intensification in corn-based rotations would increase the export of nitrogen (N), phosphorus (P), and sediment into Kansas reservoirs cannot be tested without an accurate tool for predicting reservoir water quality changes.

This study was designed to evaluate the performance of EUTROMOD, a spatially lumped watershed model for the prediction of eutrophication-related reservoir water quality, using a diverse set of Kansas reservoirs as a case study. EUTROMOD (Reckhow et al. 1992) was examined because it can incorporate regional empirical models, be modified to incorporate various land use inputs, conduct basic land use change scenarios, provide a coupled watershed-reservoir model, and offer a transparent modeling platform that can be used for outreach and education efforts with the general public.

Additionally, several empirical models of in-lake total nitrogen (TN), in-lake total phosphorus (TP), and regional models of chlorophyll *a* (Chl-*a*) were tested using the EUTROMOD-predicted nutrient loading to determine which models best characterize the in-lake response to nutrient loading in Kansas. Global and regional empirical models have demonstrated strong merit in the understanding and management of lake and reservoir water quality and can be valuable tools to estimate changes in trophic state of a waterbody due to changes in nutrient loading (Peters 1986). General empirical models that are calibrated and validated for a group of lakes, rather than a lake-specific model, may have greater applicability for studying future lake conditions that may fall outside of the range of current lake conditions (Bryhn and Håkanson 2007). Additionally, general empirical models requiring few inputs may be useful tools for communicating water quality changes to local policy makers, farmers, and citizens. Yet, if predictions from general models consistently diverge from empirical observations, as was the case in this study, then the model validity may be contested; therefore, a secondary motivation of this study is to improve the use of empirical models for water quality management, with a focus on Midwestern man-made reservoirs.

In this study, 28 Kansas reservoirs and watersheds of varying size and dominant land use were analyzed using

EUTROMOD to determine nutrient loading estimates, which were subsequently used to determine in-lake TN, TP, and Chl-*a* concentrations using 7 TP empirical models, 6 TN empirical models, and 3 Chl-*a* models. The model-predicted values of TN, TP, and Chl-*a* were then compared to equivalent whole-lake average values of long-term water quality obtained from the Kansas Department of Health and Environment (KDHE) ambient lake monitoring program (KDHE 2010).

Methods

EUTROMOD

EUTROMOD (Reckhow et al. 1992) is a collection of spreadsheet-based watershed, nutrient loading, and lake response models. It incorporates the Universal Soil Loss Equation (USLE), empirical nutrient loading concentrations for multiple land use categories, empirically derived nutrient trapping factors, as well as empirical lake response models developed for regional application. The EUTROMOD Midwest regional model was used in this study, which is applicable for Kansas. This modeling framework requires aggregated watershed data inputs, including the surface areas associated with different watershed land uses, multiple soil-related parameters, and nutrient loading derived from both nonpoint and point sources. In addition, it requires lake-specific data inputs such as lake area, mean depth, and average annual evaporation. Based on these 2 sets of data inputs, EUTROMOD uses empirical water quality relationships to predict growing season (Jun–Sep) average concentrations of TN, TP, and Chl-*a* on a steady state basis, indicating no consideration of change or fluctuations over time (Reckhow et al. 1992). Thus, EUTROMOD can be used to predict long-term changes in lake trophic state but not short-term events or seasonal or spatial dynamics (Reckhow et al. 1992).

Empirical lake response models

Canfield and Bachmann (1981) developed a set of empirical models to predict TP concentrations in 704 natural and artificial lakes, based on the generalized loading model proposed by Vollenweider (1969):

$$TP = \frac{L_p}{z(\sigma + \rho)}, \quad (1)$$

where TP is the model-predicted annual mean whole lake concentration of TP (mg/m³), L_p is the annual P loading per unit of lake surface area (mg/m²/yr), z is mean depth of the lake (m), σ is the P sedimentation coefficient (1/yr), and ρ is the flushing rate (1/yr). The flushing rate

was determined from the hydraulic retention time calculated in EUTROMOD (Lake Volume/Stream Runoff * Watershed Area). In this study, we applied 4 different models (Table 1) from Canfield and Bachmann (1981) as well as models developed by Reckhow (1979a), and Jones and Bachmann (1976). These models were chosen because they were highly referenced in the literature.

To integrate the empirical models into our overall modeling framework, EUTROMOD's estimate for TP mass loading (kg/yr) was converted to a lake-specific areal TP load (L_p , mg/m²/yr). The EUTROMOD-estimated values for areal P load (L_p), mean depth (z), flushing rate (ρ), and the sedimentation coefficient (σ) were used to predict annual mean P concentrations in each of the reservoirs studied. The P sedimentation coefficient is determined empirically, based on areal P loading and mean depth (Table 1). In the Reckhow (1979a) model, q_s refers to the surface overflow rate (m/yr) and is equivalent to the mean depth divided by the hydraulic retention time (z/τ).

The Bachmann (1980) TN model (equation 2) is similar in structure to the Canfield-Bachmann (1981) P model, except that an empirically determined N attenuation coefficient (α) replaces the P sedimentation coefficient (σ):

$$TN = \frac{L_n}{z(\alpha + \rho)}, \quad (2)$$

where TN is the predicted annual whole lake mean concentration of TN (mg/m³), L_n is the annual N loading per unit lake surface area (mg/m²/yr), and α is the N attenuation coefficient (1/yr). The N attenuation coefficient represents the loss or change of TN in the water column due to sedimentation losses, denitrification, N fixation, and internal loading (Bachmann 1980). In Bachmann (1980) the N attenuation coefficient can be determined empirically 3 ways, based on volumetric N loading (L/z), areal N loading (L), or hydraulic flushing rate (ρ). Bachmann's models (1980) were determined for artificial lakes and natural lakes separately, as well as together, resulting in 9 total models. In this study, 5 models for N attenuation were tested with equation 2 to determine the best possible model for the Kansas reservoirs analyzed in this study (Table 1).

In addition to the embedded EUTROMOD model, (Table 1) for Chl-*a*, 2 additional predictive models for algal biomass were tested. The first model (Dodds et al. 2006) was derived from Kansas reservoir monitoring data:

$$Chl-a_{log} = -0.421 + (0.96 * TP_{log}). \quad (3)$$

We also tested the model of Jones et al. (2008), which was derived from lake and reservoir monitoring data from Missouri and southern Iowa:

$$Chl-a_{log} = -0.59 + (1.09 * TP_{log}). \quad (4)$$

Because these empirical models were both developed in the central Midwest region, they were the best candidates for modeling in-lake Chl-*a* concentrations in Kansas. Both equations were used to estimate mean Chl-*a* concentrations from all in-lake predicted TP concentrations from all 7 TP models, which resulted in 14 Chl-*a* predictions for each reservoir studied.

Data collection

The input parameters for EUTROMOD were gathered using watershed and land cover maps created by the KDHE. These watershed maps were coded for land cover (pasture/rangeland, row crop, forest, urban, feedlot), and the sub-basins of all small ponds and depressions were delineated. Online soil maps from the United States Department of Agriculture, National Resources Conservation Services soil survey website (USDA-NRCS 2010) were then used to obtain data for key watershed properties: hydrologic soil group, soil erodibility (K), and the average slope for each dominant soil type in the watershed. Soil erosion losses (SE, t/ha/yr) were estimated using the metric version of the USLE, $SE = 1.29 * RE * K * LS * C * P$ (Reckhow et al. 1992). The rainfall erosivity term (RE) was interpolated for each lake location from average annual values provided in Wischmeier and Smith (1978) and converted to metric units. Soil erodibility (K) and the topographic factor combining slope length and steepness (LS) were averaged within each watershed land use category (Wischmeier and Smith 1978, USDA-NRCS 2010).

The cropping factor (C) in the USLE reflects local land use characteristics such as crop choice, crop rotation, and land management practices (Stewart et al. 1975). Local crop choices and land use practices differed both within individual watersheds and across watersheds within the state. Because detailed information on crop choices and land management were not available for each study site, a consistent single value of C for each possible land use category was chosen and applied uniformly to all 28 watersheds. For example, in the case of row crop land use, the chosen cropping factor assumed that all row crops were planted as a corn-soybean rotation because literature values for C do not differ greatly for corn (0.19) and soybean in a corn-soy rotation (0.18) with similar land management practices (Stewart et al. 1975). In the row crop land use category, we also assumed a 30–40%

Table 1. Mathematical equations for all predictive models used in this study (Jones and Bachmann 1976, Reckhow 1979a, Bachmann 1980, Canfield and Bachmann 1981, Reckhow et al. 1992 [EUTROMOD], Dodds et al. 2006, Jones et al. 2008); legend: TN = total nitrogen; TP = total phosphorus; Chl-*a* = chlorophyll *a*; L = areal load (mg/m²/yr); z = mean depth (m); ρ = flushing rate (per yr); HRT = hydraulic residence time (yr); α = attenuation coefficient; q_s = surface overflow rate (m/yr).

Empirical Model	Predicted Parameter	Model Equation
EUTROMOD	TN	$TN_{in-lake} = \frac{TN_{input}}{1 + (0.459 * HRT^{0.253} * z^{0.218} * TN_{input}^{0.955})} * 10^{0.0058}$
Bachmann-All	TN	$TN = \frac{L_n}{z(\alpha + \rho)}$
Bachmann-1	α	$\ln \alpha = -6.430 + 0.709 \ln (L_n)$
Bachmann-2	α	$\ln \alpha = -4.34 + 0.618 \ln (L_n/z)$
Bachmann-3	α	$\ln \alpha = -4.144 + 0.594 \ln (L_n/z)$
Bachmann-4	α	$\ln \alpha = -0.291 + 0.5821 \ln (\rho)$
Bachmann-5	α	$\ln \alpha = -0.367 + 0.554 \ln (\rho)$
EUTROMOD	TP	$TP_{in-lake} = \frac{TP_{input}}{1 + (10.767 * HRT^{0.395} * z^{0.009} * TP_{input}^{0.821})} * 10^{0.024}$
Canfield-Bachmann-1	TP	$TP = \frac{0.8L}{z \left(0.0569 \left(\frac{L}{z} \right)^{0.639} + \rho \right)}$
Canfield-Bachmann-2	TP	$TP = \frac{0.8L}{z \left(0.114 \left(\frac{L}{z} \right)^{0.589} + \rho \right)}$
Canfield-Bachmann-3	TP	$TP = \frac{0.8L}{z \left(0.129 \left(\frac{L}{z} \right)^{0.549} + \rho \right)}$
Canfield-Bachmann-4	TP	$TP = \frac{0.49L}{z \left(0.0926 \left(\frac{L}{z} \right)^{0.510} + \rho \right)}$
Reckhow (1979a)	TP	$TP = \frac{L_p}{11.6 + 1.2q_s}$
Jones-Bachmann	TP	$TP = \frac{0.84L_p}{z(0.65 + \rho)}$
EUTROMOD	Chl- <i>a</i>	$Chl-a = 10^{1.985 + 0.51 * \log\left(\frac{TP}{10^{0.024}}\right) - 0.352 * \log(z) + 0.234 * \log(HRT)} * 10^{0.025}$
Jones et al.	Chl- <i>a</i>	$Chl-a_{log} = -0.59 + (1.09 * TP_{log})$
Dodds et al.	Chl- <i>a</i>	$Chl-a_{log} = -0.421 + (0.96 * TP_{log})$

residue coverage left on the field, that the field was chiseled plowed for corn, and that no-till was utilized for soybeans; these soil practices were assumed based on discussion with a soil scientist at Kansas State University and survey data of field management practices in Kansas (Charles Rice, Kansas State University, pers. comm.). The supporting practice factor (P) describes the effect of soil conservation practices such as contouring and terracing on cropland erosion (Stewart et al. 1975). A constant supporting practice ($P = 1.0$; no contouring or terracing) was applied to all land uses within the 28 watersheds because detailed information on soil conservation practices was not available.

The average annual runoff coefficient for each hydrologic soil group was used to estimate overland runoff (Chow et al. 1988). The annual runoff coefficient indicates a fixed ratio of runoff to rainfall, which can provide an adequate approximation for long-term annual runoff estimates (Reckhow et al. 1992). Watershed area, lake area, mean depth, and long-term site-specific precipitation and evaporation values were from a dataset developed by the KDHE. Mean annual precipitation was specifically derived from interpolating long-term annual average maps from the Kansas Department of Agriculture (KDA 2000), and mean annual evaporation was derived from interpolating long-term annual lake evaporation (Hjelmfelt and Cassidy 1975). In EUTROMOD, hydraulic retention time is calculated by dividing lake volume (10^6 m^3) by the product of estimated stream runoff (m/yr) and watershed area (km^2). Estimated stream runoff is determined using the rational method in the watershed portion of the model (see Supplemental Materials for lake-specific water budget values).

Nutrient loading estimates

Dissolved nutrient inputs were calculated using average edge-of-field dissolved nutrient concentrations from the EUTROMOD manual and calculated runoff volumes (Reckhow et al. 1992). For row crop land use categories, dissolved N and P export values from corn with conventional tillage were used (2.9 mg/L N and 0.26 mg/L P; Reckhow et al. 1992). For pasture land use, nutrient export values for conventional practice were used (3.0 mg/L N and 0.15 mg/L P; Reckhow et al. 1992). No literature values were reported for rangeland, so an average between hay with conventional practice and forest was used (1.43 mg/L N and 0.08 mg/L P) because rangelands are expected to contain mixtures of grasses and woody vegetation. For forest land use, mean values for the Central United States were used (0.06 mg/L N and 0.009 mg/L P; Reckhow et al. 1992).

Sediment-attached nutrient loads (kg/yr N or P) were

determined using USLE-based estimates of soil loss ($\text{t}/\text{ha}/\text{yr}$) coupled with average soil nutrient concentrations (mg/kg N or P) obtained from United States N and P content soil maps developed by Mills et al. (1985). EUTROMOD also allows the user to specify several different sediment trapping factors by area for each land use type. Trapping factors were determined using delivery ratios from Stewart et al. (1975), based on the contiguous drainage area of each land use within the watershed.

Model evaluation

Model-predicted average concentrations of TN, TP, and Chl-*a* for each reservoir were compared against their observed long-term average values provided by the KDHE lake monitoring database for each of the 28 reservoirs analyzed. Long-term averages include samples taken during the 1975–2007 period on a 3–5 year rotating basis. Least squares regression was then used to determine relative model accuracy (regression slope and y-intercept) and precision (r^2 , the coefficient of determination; Tedeschi 2006). The mean absolute error (MAE) and the model average percent error (PE) were also used to compare the predictive accuracy of the different models and to determine the best-performing empirical models for estimating in-lake concentrations of TN, TP, and Chl-*a* (Willmott and Matsuura 2005). Additionally, the sensitivity of several parameters in the USLE were tested, such as the cropping factor (C) and the length-slope factor (LS) in relation to predicted TP load from the watershed and predicted in-lake TP estimated with the EUTROMOD and Canfield-Bachmann (1981) models. This sensitivity analysis was conducted for 3 reservoirs of varying nutrient levels: Lone Star Lake, Brown County State Fishing Lake, and Woodson County State Fishing Lake.

Graphical plots were used to determine whether the residual model error for in-lake TN, TP, and Chl-*a* was related to key parameters such as watershed area, lake area, watershed to lake area ratio, lake mean depth, land use type, and model outputs, such as predicted sediment load into the lake (data not shown).

To further evaluate the model-estimated sedimentation coefficients for TP and TN, the optimal P sedimentation coefficient and N attenuation coefficient were calculated for each reservoir by rearranging equation 1 and 2, respectively, using the long-term TP or TN observed concentrations and the EUTROMOD-estimated areal TP or TN load, and then solving for the sedimentation coefficient. The optimal N attenuation and P sedimentation coefficients provide reference points to analyze if the actual model-estimated attenuation coefficients are reasonable or if they need to be adjusted.

Results

Total nitrogen

TN concentrations were over-predicted by the unaltered EUTROMOD framework for 23 of the 28 reservoirs (Fig. 1). Note, however, the large variation in the measured data, with standard deviations ranging from 129 to 10 230 $\mu\text{g/L}$ N for the long-term averages. The slope of the regression line for the EUTROMOD-predicted values and the observed means was 0.50 (95% confidence interval 0.40–0.60), which is significantly less than one.

Using the various Bachmann (1980) TN models, the predicted values are greater than the observed TN concentrations in all reservoirs; however, the overall trend of predicted to observed improved using the Bachmann models, with slopes ranging from 0.62 to 0.78 compared to 0.5 obtained with EUTROMOD. The average model percent error ranges from 33 to 98%, and the MAE ranges from 425 to 782. Bachmann models 2 and 3 had the lowest MAE and percent error, but the lowest slopes. Bachmann models 4 and 5 both had the highest slopes, but these models also had greater MAE scores and percent error (Table 2).

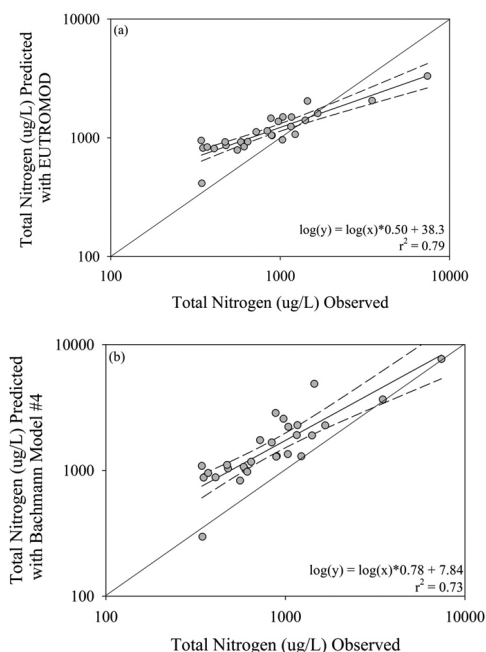


Fig. 1. Observed TN concentrations from the KDHE database for all 28 reservoirs compared to (a) predicted TN concentrations from the unaltered EUTROMOD model, and (b) predicted TN concentrations from EUTROMOD's TN loading calculation coupled with the Bachmann (1981) in-lake empirical model #4. Regression results are indicated by a solid line, 95% confidence intervals are shown with a dashed line, and the one-to-one predicted to observed relationship is shown with a solid line across the graph.

N attenuation coefficients ranged from approximately 1 to 6 for Bachmann (1980) models 1, 4, and 5, while Bachmann models 2 and 3 had coefficients that ranged from 2 to 7/yr, with an outlier value near 20/yr; however, the majority of the coefficients were $<3/\text{yr}$. The optimal calculated N attenuation coefficients ranged from 2 to 10/yr (Fig. 2). Of the 384 artificial lakes included in the original 1980 Bachmann paper, N attenuation coefficients ranged from -50 to 392/yr with a mean of 8.7/yr; therefore, attenuation coefficients estimated in this model are predominantly below the average coefficients developed from the original study (Bachmann 1980).

Total phosphorus

TP concentrations were also over-predicted in EUTROMOD for 24 of the 28 reservoirs (Fig. 3). As was found with TN concentrations, the standard deviations were high for observed values of TP, ranging from 5.8 to 808.5 $\mu\text{g/L}$ P. When EUTROMOD-predicted TP was compared to observed concentrations, the percent error was 119% with an MAE of 53. In addition, the slope of the regression of the EUTROMOD-predicted values and the observed means was 0.46 (95% confidence interval 0.35–0.57), which is significantly less than one.

Using the various in-lake TP models, the predicted TP concentrations are still greater than the observed concentrations, regardless of which empirical model is used; however, the overall relationship of predicted values to observed values improved using the Canfield-Bachmann models (1981), the Reckhow model (1979a), and the Jones-Bachmann model (1976) with slopes ranging from 0.58 to 0.93, compared to 0.45 obtained with

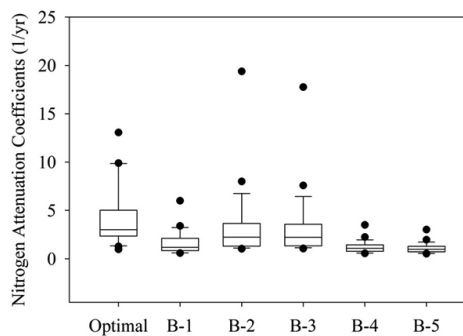


Fig. 2. Box plots of the TN attenuation coefficients estimated from each Bachmann (1980)-TN empirical model (B = Bachmann), as well as the optimal coefficient, which is calculated from the empirical model using observed TN concentration for each reservoir as an input. In the box plot, the lines of the box indicate the 25th percentile, median, and 75th percentile; the whiskers above and below the box indicate the 10th and 90th percentiles, respectively; and the dark circles represent the outlying points.

Table 2. Results of total nitrogen predictive models plotted against observed total nitrogen concentrations for 28 reservoirs in Kansas (Bachmann 1980, Reckhow et al. 1992 [EUTROMOD]).

Model	Intercept	Slope	95% Confidence Interval	r ²	Mean Absolute Error	Average Percent Error
Bachmann-1	11.7	0.71	0.57–0.85	0.80	578	75%
Bachmann-2	16.1	0.62	0.48–0.76	0.76	431	33%
Bachmann-3	14.1	0.64	0.50–0.79	0.76	425	33%
Bachmann-4	7.84	0.78	0.59–0.97	0.73	722	91%
Bachmann-5	8.10	0.78	0.59–0.97	0.73	782	98%
EUTROMOD	38.3	0.50	0.40–0.60	0.79	481	43%

EUTROMOD. The average model percent error ranges from 101 to 540%, and the mean absolute error ranges from 49 to 317. The Jones-Bachmann model overpredicts TP concentrations by more than a 5-fold increase (mean percent error 540%), yet has a high slope in the predicted-observed regression (0.89). The fourth Canfield-Bachmann model is the most promising, with the lowest mean absolute error (49) and the lowest percent error (101%) of all the models, but also with a reasonably high slope (0.71; Table 3).

The P sedimentation coefficients for Canfield-Bachmann (1981) models 1, 2, and 3 had similar values that ranged from ~1 to 7/yr, each with an outlier near 40–60/yr. The fourth Canfield-Bachmann model had a much narrower range, with the majority of values <3/yr and an outlier near 21/yr. The optimal P sedimentation coefficients ranged from 2 to 71/yr (Fig. 4). Of the 433 artificial lakes included in the Canfield-Bachmann study, P sedimentation coefficients ranged from –290 to 490/yr with a mean of 145 ± 55/yr (1981). Attenuation coefficients estimated for Kansas reservoirs are therefore much lower than the average coefficients developed from the original study (Canfield and Bachmann 1981).

Algal biomass (Chlorophyll a)

The EUTROMOD-embedded model (Table 1) uses HRT (hydraulic residence time), mean depth, and predicted in-lake TP to predict water column concentrations of Chl-a. The predicted values for Chl-a from the original EUTROMOD modeling framework fell within an exceptionally narrow range of 11 to 32 µg/L, while observed concentrations varied over almost 2 orders of magnitude, from 5 to 497 µg/L. A plot of predicted versus observed values for this model (Fig. 5) exhibited an extremely shallow slope (0.16, 95% confidence interval 0.10–0.22).

In contrast, the predictive capabilities of both the Dodds et al. (2006) model and the Jones et al. (2008) model were greatly superior to the embedded EUTROMOD equations for algal biomass, both of which require only TP concentration inputs to estimate Chl-a (Table 1). Output from these 2 models was generated using all of the predicted TP concentrations, generating 14 additional predictions for algal biomass for each reservoir (Table 4).

The regression results of predicted Chl-a values from the Dodds et al. (2006) model with EUTROMOD TP input and observed Chl-a generated a slope of 0.36 (95% confidence interval 0.26–0.46) with r² = 0.68, and the

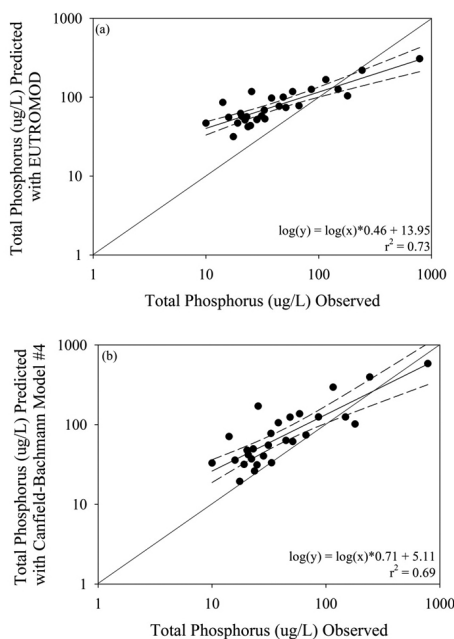


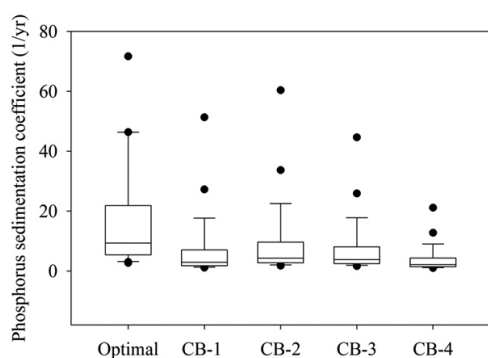
Fig. 3. Observed TP concentrations from the KDHE database for all 28 reservoirs compared to (a) predicted TP concentrations from the unaltered EUTROMOD model, and (b) predicted TP concentrations from EUTROMOD’s TP loading calculation coupled with the Canfield-Bachmann (1981) in-lake empirical model #4. Regression results are indicated by a solid line, 95% confidence intervals are shown with a dashed line, and the one-to-one predicted to observed relationship is demonstrated with a solid line across the graph.

Table 3. Results of total phosphorus predictive models plotted against observed total phosphorus concentrations for 28 reservoirs in Kansas (Jones and Bachmann 1976, Reckhow 1979a, Canfield and Bachmann 1981, Reckhow et al. 1992 [EUTROMOD]).

Model	Intercept	Slope	95% Confidence Interval	r ²	Mean Absolute Error	Average Percent Error
Canfield-Bachmann-1	10.8	0.58	0.42–0.74	0.69	65	162%
Canfield-Bachmann-2	7.35	0.61	0.45–0.77	0.70	49	102%
Canfield-Bachmann-3	6.85	0.66	0.49–0.83	0.70	53	123%
Canfield-Bachmann-4	5.11	0.71	0.52–0.90	0.69	49	101%
Reckhow (1979a)	3.12	0.93	0.64–1.21	0.63	125	213%
Jones-Bachmann	7.64	0.89	0.61–0.116	0.63	317	540%
EUTROMOD	13.95	0.46	0.35–0.57	0.73	53	119%

predicted values ranged from 10 to 93 µg/L. The Jones et al. (2008) Chl-*a* empirical model with EUTROMOD TP input performed similarly to the Dodds et al. model with EUTROMOD TP input (data not shown), with predicted values ranging from 11 to 132 µg/L and a slope of 0.41 (95% confidence interval, 0.30–0.52; Table 4).

When the Dodds et al. (2006) Chl-*a* model was combined with the predicted-TP input from the other 6 TP models (Table 1), there was a range in accuracy of the Chl-*a* predictions. The MAE ranged from 23 to 75, and the slope of the predicted to observed regression ranged from 0.47 to 0.79 (see Table 4 for the results of all model iterations). The results of the fourth Canfield-Bachmann (1981) model paired with the Dodds et al. (2006) Chl-*a* model have the lowest error (MAE = 23 and PE = 83%) and the highest slope (0.58) of all the Dodds et al. Chl-*a* model iterations (Table 4).

**Fig. 4.** Box plots of the TP sedimentation coefficients estimated from each Canfield-Bachmann (1981) TP empirical model (CB = Canfield-Bachmann), as well as the optimal coefficient, which is calculated from the empirical model using observed TP concentration for each reservoir as an input. In the box plot, the lines of the box indicate the 25th percentile, median, and 75th percentile; the whiskers above and below the box indicate the 10th and 90th percentiles, respectively; and the dark circles represent the outlying points.

Similarly, there was also a great range in accuracy for the Jones et al. (2008) Chl-*a* model combined with 6 TP models examined in this study. The MAE ranged from 22 to 146 and the slope of the predicted to observed regression ranged from 0.53 to 0.89. Again, the results of the fourth Canfield-Bachmann (1981) model with the Jones et al. (2008) Chl-*a* model have the lowest error (MAE = 22 and PE = 116%) and the highest slope (0.66) of all the Jones et al. (2008) Chl-*a* models (Table 4).

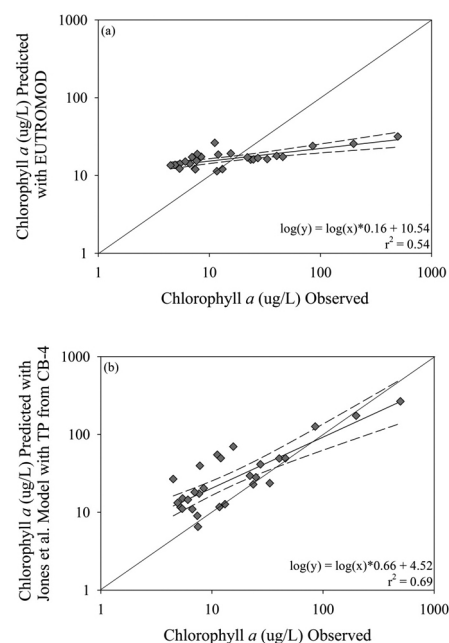
**Fig. 5.** Observed Chl-*a* concentrations from the KDHE database for all 28 reservoirs compared to (a) predicted Chl-*a* concentrations from the unaltered EUTROMOD model, and (b) predicted Chl-*a* concentrations from the Jones Chl-*a* model with TP input from the Canfield-Bachmann (1981) model #4 (CB = Canfield-Bachmann). Regression results are indicated by a solid line, 95% confidence intervals are shown with a dashed line, and the one-to-one predicted to observed relationship is demonstrated with a solid line across the graph.

Table 4. Results for chlorophyll *a* (Chl-*a*) predictive models plotted against observed Chl-*a* concentrations for 28 reservoirs in Kansas (Jones and Bachmann 1976, Reckhow 1979a, Canfield and Bachmann 1981, Reckhow et al. 1992 [EUTROMOD], Dodds et al. 2006, Jones et al. 2008).

Chl- <i>a</i> Model	TP Model	Intercept	Slope	95% Confidence Interval	r ²	Mean Absolute Error	Average Percent Error
Dodds et al.	Canfield-Bachmann-1	8.23	0.47	0.34–0.60	0.68	28	140%
Dodds et al.	Canfield-Bachmann-2	5.99	0.50	0.36–0.63	0.69	26	87%
Dodds et al.	Canfield-Bachmann-3	5.95	0.53	0.39–0.68	0.69	24	105%
Dodds et al.	Canfield-Bachmann-4	4.73	0.58	0.42–0.73	0.69	23	83%
Dodds et al.	Jones-Bachmann	8.53	0.74	0.52–0.95	0.66	75	422%
Dodds et al.	Reckhow (1979a)	3.61	0.79	0.57–1.0	0.69	28	155%
Dodds et al.	EUTROMOD	9.22	0.36	0.26–0.46	0.68	30	106%
Jones et al.	Canfield-Bachmann-1	8.46	0.53	0.39–0.68	0.68	30	191%
Jones et al.	Canfield-Bachmann-2	5.90	0.56	0.41–0.71	0.69	24	120%
Jones et al.	Canfield-Bachmann-3	5.85	0.61	0.44–0.77	0.69	24	144%
Jones et al.	Canfield-Bachmann-4	4.52	0.66	0.48–0.83	0.69	22	116%
Jones et al.	Jones-Bachmann	8.81	0.84	0.59–1.08	0.66	146	629%
Jones et al.	Reckhow (1979a)	3.32	0.89	0.65–1.14	0.69	49	225%
Jones et al.	EUTROMOD	9.63	0.41	0.30–0.52	0.68	30	144%
EUTROMOD	EUTROMOD	10.54	0.16	0.10–0.22	0.54	33	48%

Discussion

The primary goal of this study was to evaluate EUTROMOD and regional in-lake empirical models as predictive tools for water quality estimation in Kansas reservoirs to determine if future research could utilize EUTROMOD to predict water quality impacts of land use change scenarios in response to biofuel feedstock development. Unfortunately, all models used consistently overpredict in-lake TN and TP, which suggests that an overall driving mechanism is causing bias in the models. Model error could be primarily due to the in-lake empirical models, and especially the sedimentation coefficient. Other studies have concluded that in-lake TP empirical models overpredict P concentrations, often by almost 2-fold (Wagner 2010; M. Ernst, Tarrant Regional Water District, Dec 2011, pers. comm.). The overprediction of nutrient concentrations from empirical models thus is provisionally attributed to a low model-estimated P sedimentation coefficient.

Phosphorus sedimentation in lakes can be achieved by several mechanisms: (1) sedimentation of P minerals from the watershed, which may settle rapidly in near-shore areas; (2) adsorption or precipitation of P with inorganic compounds, such as co-precipitation with iron and manganese, adsorption to clays, and co-precipitation with and/or adsorption to carbonates; and (3) sedimentation of

P associated with organic matter, both produced in the lake and in the watershed (Wetzel 2001, Esten and Wagner 2010). Many variables can therefore alter the P sedimentation rate in a particular lake. The long and dendritic basin morphologies of many reservoirs can contribute to high rates of P sedimentation and N attenuation in the shallow areas receiving high nutrient inflow from streams (David et al. 2006, Harrison et al. 2009). Therefore, models like EUTROMOD, as well as empirical models derived predominantly from lake studies, may have limited application to reservoirs because they do not account for spatial dynamics or the differing capacity of reservoirs to attenuate nutrients.

The N attenuation coefficient in the in-lake empirical models refers to fluxes of N due to internal loading, sedimentation losses, N fixation, and denitrification, all of which are factors that are challenging and time-intensive to measure *in situ*. The substantial literature on the denitrification potential of freshwater environments (Jansson et al. 1994, Peterson et al. 2001, Saunders and Kalff 2001, David et al. 2006, Piña-Ochoa and Álvarez-Cobelas 2006), however, concludes that increased N loading is the primary factor that leads to increased N attenuation due to high concentrations of nitrate available for denitrification. Long hydraulic retention times and high seasonal temperatures can also increase N attenuation in lake and reservoir ecosystems (Jansson et al. 1994, Saunders and Kalff 2001,

David et al. 2006, Piña-Ochoa and Álvarez-Cobelas 2006). Small streams can also be extremely effective with N processing and attenuation within the watershed (Peterson et al. 2001). The EUTROMOD loading model does not include a loss function for dissolved N processed within the watershed, however, and the entire dissolved load is assumed to reach the waterbody. This could be a source of error in overestimating the estimated N load and, therefore, in-lake N concentrations.

With the multitude of factors regulating P and N attenuation in lakes, some of the bias in this study is likely due to the empirical coefficients for P sedimentation and N attenuation. To test the potential error of these coefficients, the optimal P sedimentation and N attenuation coefficients were calculated by setting the TP and TN equal to observed concentrations in equations 1 and 2, respectively. These optimal coefficients are approximately double the coefficients determined using the empirical models in this study (Fig. 2 and 4); however, the optimal coefficients are still within the lower range of the coefficient values estimated in the original Bachmann (1980) and Canfield and Bachmann (1981) studies. To better estimate P sedimentation and N attenuation coefficients, a regional and reservoir-specific empirical model may be necessary to account for regional variation and differences in nutrient processing between lakes and reservoirs. A regional modeling effort to estimate P and N attenuation in reservoirs will require a collaborative effort across many institutions and would be a promising area for future study to improve the quantitative prediction of water quality in reservoirs.

Alternatively, the systematic overpredictions of TP, TN, and Chl-*a* may be caused by error in the nutrient loading estimate generated from the watershed model, EUTROMOD, and in the empirical nutrient concentrations, which are necessary inputs into the model. When in-lake nutrient concentrations are estimated from nutrient loading values indirectly derived from land use and landscape characteristics, then the uncertainty in these estimates can be quite large (Reckhow 1979a); therefore, the associated error from the indirect estimation of nutrient loading may be much greater than the empirical model standard error (Reckhow 1979b).

To test if the primary source of overall model error is generated in the loading estimates, other methods and models should be utilized to estimate nutrient loading, which can then be compared to the EUTROMOD estimates. Unfortunately, the study sites in this paper do not have the available stream gauge and nutrient data to calculate nutrient loading from a flow-weighted method; however, large reservoirs in Kansas and in the region may have these data available, which could be a topic for further study. Additionally, other watershed models such

as the Soil and Water Assessment Tool (SWAT) or the Hydrologic Simulation Program–Fortran (HSPF) could be options for generating comparable nutrient loading estimates.

In conclusion, there is still uncertainty as to the overall source of model bias and error. We suggest 2 areas of further research to improve the quantitative modeling of water quality conditions in the Midwest. First, future work could gather available data to create regional empirical models relating nutrient loading and in-lake nutrient concentrations; such tools would greatly benefit reservoir management efforts and future studies on regional water quality. Second, other methods and models should be used to generate nutrient loading estimates, which can be compared to the EUTROMOD estimates to determine if nutrient loading estimates are the dominant source of overall model error and if EUTROMOD is a reliable tool for predicting nutrient loads from agricultural watersheds.

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Supplementary Material

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website, <https://www.fba.org.uk/journals/index.php/IW>:

Supplementary Table 1. Parameters for calculating the water budget for lakes in study.