JOURNAL OF ENVIRONMENTAL SCIENCES 67 (2018) 54–66

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JES JOURNAL OF ENVIRONMENTAL SCIENCES

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# Development of methods for establishing nutrient criteria in lakes and reservoirs: A review

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ARTICLE INFO

*Article history:*
Received 17 March 2017
Revised 23 June 2017
Accepted 17 July 2017
Available online 25 July 2017

*Keywords:*
Nutrient criteria
Statistical analysis approach
Paleolimnological reconstruction approach
Stressor-response model
Model inference approach

ABSTRACT

Nutrient criteria provide a scientific foundation for the comprehensive evaluation, prevention, control and management of water eutrophication. In this review, the literature was examined to systematically evaluate the benefits, drawbacks, and applications of statistical analysis, paleolimnological reconstruction, stressor-response model, and model inference approaches for nutrient criteria determination. The developments and challenges in the determination of nutrient criteria in lakes and reservoirs are presented. Reference lakes can reflect the original states of lakes, but reference sites are often unavailable. Using the paleolimnological reconstruction method, it is often difficult to reconstruct the historical nutrient conditions of shallow lakes in which the sediments are easily disturbed. The model inference approach requires sufficient data to identify the appropriate equations and characterize a waterbody or group of waterbodies, thereby increasing the difficulty of establishing nutrient criteria. The stressor-response model is a potential development direction for nutrient criteria determination, and the mechanisms of stressor-response models should be studied further. Based on studies of the relationships among water ecological criteria, eutrophication, nutrient criteria and plankton, methods for determining nutrient criteria should be closely integrated with water management requirements.



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http://dx.doi.org/10.1016/j.jes.2017.07.013

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## Introduction

The majority of freshwater resources worldwide have suffered from eutrophication caused by excessive inputs of nitrogen and phosphorus (Huo et al., 2013a, 2013b; Janssen et al., 2017). The determination of numeric nutrient criteria is considered important for controlling cultural eutrophication and protecting water quality in lakes and reservoirs (Hawkins et al., 2010; Kelly et al., 2015; Ma et al., 2016). Nutrient criteria are the maximum acceptable concentrations that cause ecological effects in water without threatening the function of the waterbody; thus, they represent the trophic state of surface waters in the absence of significant human disturbance (Bouleau and Pont, 2015). Due to geographical differences in catchments (*e.g.*, species biogeography, geology, and elevation) and lake factors (*e.g.*, depth, area, and water color), regional nutrient criteria must be developed to better protect water quality (Cardoso et al., 2007; Carvalho et al., 2008). Many countries have derived ecoregions to establish regional nutrient criteria and prevent eutrophication-related designated use impairments (Huo et al., 2014a, 2014b, 2014c, 2014d; Solheim, 2005).

Nitrogen and phosphorus are not toxic to aquatic organisms and humans at low concentrations and generally do not interfere with designated uses (Lamon and Qian, 2008; Stow et al., 2014). However, excessive nutrient levels can lead to the overgrowth of phytoplankton and aquatic plants, resulting in the depletion of dissolved oxygen, fluctuations in the water pH, changes in the taxonomic composition and structure of aquatic communities, the release of toxins from phytoplankton, and disinfectant byproducts in treated drinking water (Huo et al., 2013a, 2013b). Nutrient criteria are ecological criteria, not toxicological criteria, and are not derived by simple dose–response relationships in laboratory studies (US EPA, 2010). Hence, a statistical method based on large amounts of monitoring data would provide the theory and a foundational approach for the establishment of nutrient criteria. The objectives of this paper are as follows: (1) to review the research progress in nutrient criteria determination, (2) to summarize and evaluate the established methods of nutrient criteria determination, and (3) to describe the future developments and challenges for improving nutrient criteria determination in lakes and reservoirs.

## 1. Research on nutrient criteria determination in lakes and reservoirs

The United States (US) was the first country to develop nutrient criteria. The National Nutrient Strategy for the Development of Regional Nutrient Criteria was established by the US Environmental Protection Agency (US EPA) in 1998 (US EPA, 1998). Based on waterbody characteristics, a series of technical guidance documents was released for developing nutrient criteria for different waterbodies, such as lakes and reservoirs, rivers and streams, estuarine and coastal marine waters, and wetlands (US EPA, 2000a, 2000b, 2001a, 2008). Similarly, the Water Framework Directive issued by the European Union tasked member nations with developing nutrient criteria strategies for controlling water eutrophication (Solheim, 2005). In recent years, researchers have initiated studies of nutrient criteria determination in China, and a Regional Nutrient Criteria Research Plan was created in 2008 to develop region-specific nutrient criteria (Huo et al., 2014a).

The earliest approaches used by the US EPA to establish nutrient criteria were statistical analysis methods, model prediction or extrapolation, paleolimnological reconstruction of past conditions and expert judgments (US EPA, 2000a). In 2010, three types of approaches were recommended for scientifically determining numeric criteria: the reference condition approach, mechanistic modeling, and stressor-response analysis (US EPA, 2010; Hausmann et al., 2016). The continental US was divided into 14 separate lake ecoregions with similar geographical characteristics based on perceived patterns of causal and integrative factors, including land use, land surface form, potential natural vegetation, and soils. The US EPA suggested an ecoregion-based national strategy for establishing nutrient criteria, and a statistical analysis approach was applied to determine the nutrient criteria in 14 ecoregions (US EPA, 2001b; Omernik, 1987). In Europe, waterbodies have been divided into classes based on geographical differences in catchments and lake factors, and type-specific nutrient criteria were derived to reach the appropriate ecological quality (Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010). Based on a spatial cluster analysis that considered the boundaries of water resources and provincial administration boundaries, China has been divided into eight lake ecoregions to develop ecoregional nutrient criteria (Huo et al., 2014a).

## 2. Methods of establishing nutrient criteria in lakes and reservoirs

### 2.1. Statistical analysis approach

Three methods are proposed as statistical analysis approaches for the determination of numeric criteria to address nitrogen/phosphorus pollution: the reference lake method, the lake population distribution method, and the trisection method. The reference lake method is suitable for watersheds with little human disturbance, and reference lakes in the upper 25$^{th}$ percentile are commonly used to develop nutrient criteria (Fig. 1). This method fails when reference lakes or sites are not available, as in some agriculturally dominated regions of the mid-western US (US EPA, 2000a) or eastern China (Huo et al., 2014a). If reference lakes are lacking in a region, the lake

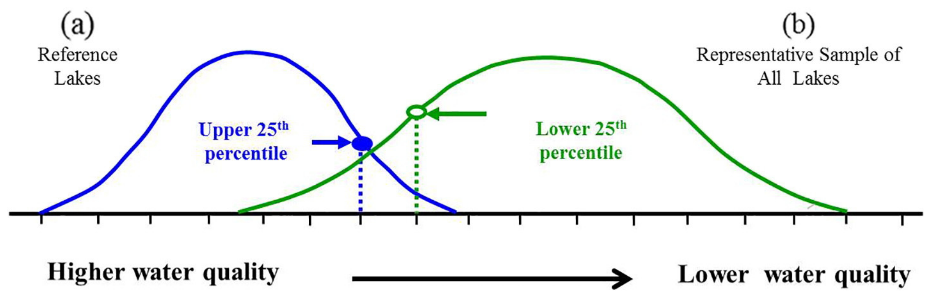


**Fig. 1 – Conceptual diagram of the reference lake and lake population distribution methods (US EPA, 2000a).**

population distribution method can serve as an alternative to the reference lake method. The lake population distribution method does not involve the identification of reference lakes and uses the entire population of lakes within a region to determine nutrient criteria. The lower 25$^{th}$ percentile of lake data has been suggested as a threshold for criteria determination. Fig. 1 shows a conceptual diagram of the reference lake method and lake population distribution method (US EPA, 2000a). These two statistical measures are thought to be relatively equal (Fig. 1); however, the final criteria could be between the two concentrations illustrated in the figure if a combination of these approaches is used (Evans-White et al., 2013). In regions that are extensively affected by human nutrient loading, 25$^{th}$ percentile criteria can lead to artificially large values for nutrient criteria (Smith et al., 2003) and possible failure to protect water quality (Dodds and Oakes, 2004). Therefore, a low percentile may be used to establish nutrient criteria for such regions. Similar to the lake population distribution method, the trisection method is a good alternative for determining nutrient criteria. The 50$^{th}$ percentile (median) values derived from the best one-third of lake data are defined as the nutrient criteria. However, the sensitivity of this approach to the proportion of impacted sites and the degree of regional influence should be carefully considered (Dodds et al., 2006).

The statistical analysis approach has been widely used to develop nutrient criteria in the US and Europe. In combination with the stressor-response relationships between anthropogenic land uses and nutrient concentrations (Dodds and Oakes, 2004), the reference lake and trisection methods were developed to determine ecoregional reference conditions for total nitrogen (TN), total phosphorus (TP), Secchi depth (SD) and chlorophyll a (Chl *a*) in Kansas lakes and reservoirs (Dodds et al., 2006). The reference site and lake population distribution methods were employed to determine nutrient criteria for rivers and streams in Montana and the Mediterranean (Suplee et al., 2007; Sánchez-Montoya et al., 2012). Nutrient concentrations based on the 75$^{th}$ percentile of reference data were matched to the corresponding concentrations in the general population (Suplee et al., 2007). Additionally, the impact of season on nutrient criteria should be considered, and the least restrictive values have been proposed as criteria to fully protect waterbodies, especially in unfavorable conditions (Sánchez-Montoya et al., 2012). Professional judgment combined with the trisection method was used to establish a nutrient baseline for tropical rivers and streams in the state of São Paulo (Cunha et al., 2011, 2012).

Implemented in 2008, the Regional Nutrient Criteria Research Plan of China preliminarily explored the feasibility of the statistical analysis approach to support the development of nutrient criteria in China using case studies. The reference lake, lake population distribution, and trisection methods were used to determine reference conditions for nutrients, Chl *a* and SD in the Yungui, Eastern, Northeast, and Xinjiang Lake ecoregions in China (Huo et al., 2012, 2013b, 2014b, 2015a). TP, TN, SD, and Chl *a* reference conditions in the Southeast ecoregion lakes were calculated using the lake population distribution and trisection methods; a multiple linear regression model, and a novel extrapolation approach by combining optimal map recognition with pattern recognition inverse mapping were developed to estimate reference nutrient concentrations (Huo et al., 2014d).

The statistical analysis approach was identified as one of the most straightforward methods of determining criteria because the data include natural variability (Solheim, 2005). An important assumption of the statistical analysis approach is that at least some lakes (sites) in the lake population are high-quality lakes (sites). However, lakes or sites that are minimally impacted by human disturbance are scarce in developed regions, and historical monitoring data are often few and sometimes problematic (Chambers et al., 2012). The scientific determination of reference lakes or sites can restrict the application of the statistical analysis approach, and the attainment of designated uses has not been considered by this approach in determining nutrient criteria. The establishment of criterion thresholds based on percentiles might be skewed by a bias in the data toward either pristine or highly influenced sites, which would introduce considerable uncertainty. Hence, more research should be conducted to quantify sources of uncertainty and account for uncertainty and related effects. Additionally, methods should be combined to determine nutrient criteria, which should be related to designated uses and the relevant causes of damage, such as excess algae and low biological diversity.

### 2.2. Paleolimnological reconstruction approach

A paleolimnological reconstruction approach has been applied to establish nutrient criteria by inferring the previous status using a sediment tracing approach. Paleolimnological techniques can be utilized to identify the pre-disturbance ecological conditions of lakes when the lake sediments remain undisturbed, and there are strong relationships between the water quality (mainly TP) and fossil assemblages obtained through the examination of sediment cores (Ter Braak and Juggins, 1993; Andersen et al., 2004; Bennion et al., 2004; Heinsalu et al., 2007). Due to the lack of historical data, samples of dated sediment cores should be collected based on information about the disturbance of sediments by the overlaying water. The chemical and biological properties of sediment core samples are analyzed to select appropriate proxies to infer past environmental conditions. The advantage of this technique is that it includes natural temporal variability and is site specific, *i.e.*, it does not require the selection of reference lakes based on an accurate sedimentary record (Solheim, 2005). A flowchart of the nutrient criteria development process using the paleolimnological reconstruction approach is shown in Fig. 2.

The diatom-TP transformation function, which is a research hotspot in the field of paleolimnology, can be used to reconstruct the background TP in lakes and predict variations in the TP concentration. The ecological and chemical reference conditions and degrees of flora change in Scottish freshwater lochs and nine enriched lakes in Europe were determined using fossil diatom assemblages in dated sediment cores and TP transfer functions (Bennion et al., 2004, 2011). The analogue matching technique and a training set of diatom taxa were used to identify appropriate reference sites for lakes impacted by eutrophication (Bennion et al., 2011). Heinsalu et al. (2007) used sediment diatom assemblages and the composition of pore-water dissolved organic matter to assess the recent eutrophication history of Lake Peipsi, Europe, and identify the possible reference conditions.

However, the preservation of organisms in the sediment is often poor, and the remains are restricted to a few organism groups. The components of diatoms decompose easily after death, leading to a difference between diatom fossils and active diatoms. There may be deviations in diatom fossil information, which make investigations of lake evolution difficult, especially for modern lakes. The method may also require complex data analysis and expert interpretation, and it is difficult to reconstruct the past nutrient conditions of shallow lakes in which sediments are easily disturbed. As a result, fluctuations in stable isotopes C and N and other indicators in the sedimentary record can be used to identify the major sources of phosphorus in various historical periods and to deduce the succession processes of lakes (US EPA, 2000a). Additionally, radiometric dating techniques ($^{210}Pb$ and $^{137}Cs$) can be used to establish a chronology for each core (Bennion et al., 2004). The paleolimnological approach is a direct measure of nutrient conditions before large-scale human intervention. Thus, the paleolimnological approach

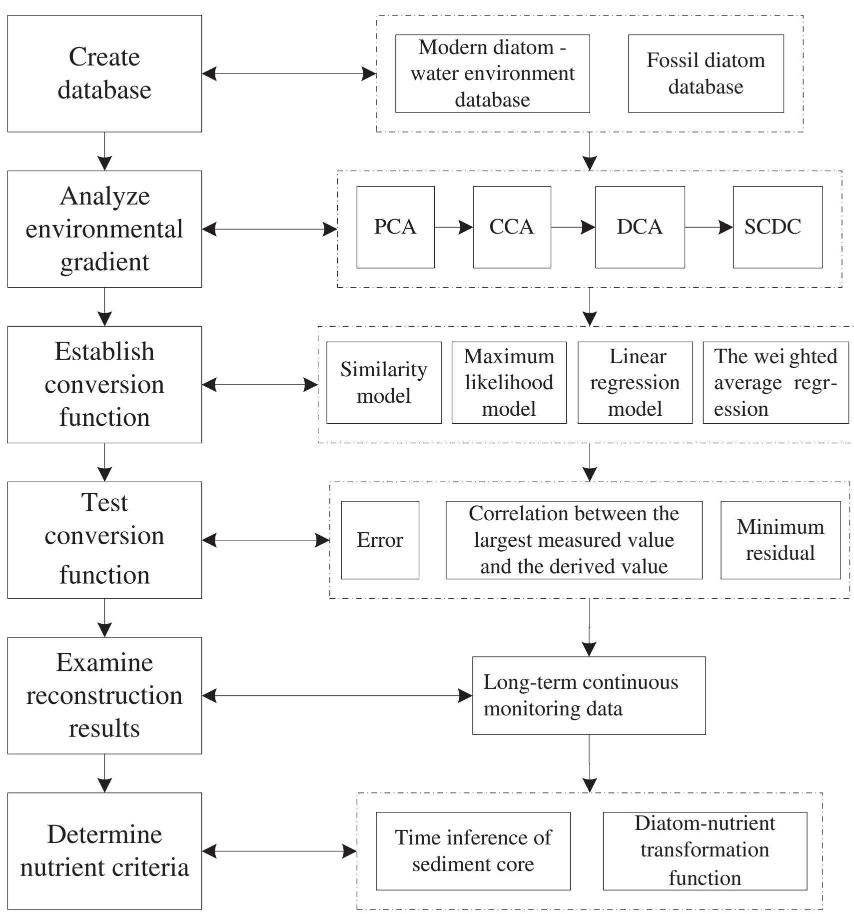


**Fig. 2 – Flowchart of nutrient criteria development using the paleolimnological reconstruction approach (PCA-principal component analysis; CCA-canonical correspondence analysis; DCA-detrended correspondence analysis; SCDC-squared chord distance coefficient).**

is limited to only natural deep lakes and cannot be applied in ecoregions with few natural lakes (Herlihy et al., 2013).

### *2.3. Stressor-response model*

If minimally affected sites cannot be identified and paleoecological or historic data are not available, stressor-response analysis is used to derive numerical criteria (Bowman and Somers, 2005; Stoddard et al., 2006; Huo et al., 2013a). Stressor-response models representing the most important known relationships between nutrient concentrations and primary productivity have been developed based on analyses of observational data collected in the field. Nutrient criteria can be established using these stressor-response models to prevent problems caused by algae blooms that affect the designated water uses and biodiversity.

In addition, appropriate response variables must be considered for various waterbodies. The Chl *a* concentration, which is strongly related to the algae biomass, can serve as an important response variable that is linked to nutrient concentrations when developing lake nutrient criteria for designated water uses (Huo et al., 2013a). For example, Lamon and Qian (2008) applied a Bayesian multilevel modeling approach to deduce nutrient criteria using $\log_{10}$Chl *a* as a response and $\log_{10}$TP and $\log_{10}$TN as predictors in lakes and reservoirs of the US. Freeman et al. (2009) developed a regional-scale Bayesian TREED model for lakes, ponds, and reservoirs to investigate the link between nutrients and Chl *a* and to determine nutrient criteria by quantifying uncertainty sources in an empirical water quality model. The nutrient effects were analyzed to estimate nutrient criteria using linear regression models of Chl *a* as a function of TP and/or TN in the seven lake ecoregions in China (Huo et al., 2014a, 2014c; Zhang et al., 2014). Diatoms have also been used to assess ecological conditions in lakes and reservoirs because they directly and sensitively respond to many physical, chemical, and biological changes in water ecosystems (Stevenson et al., 2010; Zhang et al., 2016a; Yang et al., 2016). In addition, diatoms are the source of many nuisance algae problems, such as the taste and odor impairment of drinking water, decreases in water clarity, and water filter clogging. Using the diatom index score as the response variable, Threshold Indicator Taxa ANalysis (TITAN) was used to derive thresholds of lake nutrient criteria in eastern China and a shallow eutrophic Chinese lake (Zhang et al., 2016a; Cao et al., 2016; Baker and King, 2010). Using diatoms to monitor stream biological integrity, Lavoie et al. (2014) established stream nutrient criteria for each trophic state in eastern Canada. Using diatom index data for stream sites in New Jersey and surrounding Mid-Atlantic states, Hausmann et al. (2016) structured a diatom-based conceptual model of the biological condition gradient to assess impairment and develop nutrient criteria for streams in the US.

The biological response to nutrient gradients can be subtle and often difficult to detect using linear regression analysis (Brian et al., 2013; Janssen et al., 2017). Moreover, the ecological responses to environmental gradients are often nonlinear, non-normal, and heterogeneous (Legendre and Legendre, 1998). Hence, various methods should be developed to explore the linear or nonlinear stressor-response relationships and determine the nutrient thresholds at which Chl *a* varies across nutrient concentrations (Huo et al., 2014c; Zhang et al., 2014; Heiskary and Bouchard, 2015). Linear regression models (LRMs) and Bayesian hierarchical regression models (BHRMs) can be used to determine the linear relationships associated with these gradients and parameters, whereas classification and regression trees (CARTs) and changepoint analysis (CPA) can be used to explore complex nonlinear relationships between stressor variables and response variables. CART analysis can be used to identify the most important stressor variables that affect response variable concentrations. The differences in the stressor-response models are illustrated in Table 1.

The use of LRM and BHRM to establish a stressor-response relationship requires the selection of a response threshold to derive a given nutrient concentration. The response threshold is established to protect the designated function, which introduces a degree of subjective bias. The response threshold may vary in different countries and regions due to differences in the response level of algae to nutrients and the production conditions of algal toxins. The US EPA (2010) suggested that a Chl *a* concentration of greater than 20 μg/L indicated an impaired aquatic condition. To protect the drinking water supply in China, the criterion range of the Chl *a* concentration was set to 2–5 μg/L (Huo et al., 2014a, 2014c; Zhang et al., 2014).

The CART and CPA methods do not require the establishment of a threshold for the response variable to determine a potential numerical criterion. A threshold or a changepoint refers to the position of a stressor variable that leads to an abrupt change in the statistical attributes of the ecological response variable, thereby providing a natural candidate for a nutrient criterion (Breiman et al., 1984; Qian et al., 2003). These methods eliminate the subjective bias of setting a response criterion value. Generally, the criteria values obtained using these two methods can be mutually verified to improve the accuracy of the nutrient criteria. For example, as CPA methods, nonparametric changepoint analysis (nCPA) and Bayesian hierarchical modeling (BHM) were applied to estimate the nonlinear relationships between stressor variables and response variables and detect environmental thresholds in an Everglades wetland (Qian et al., 2003). The CART, nCPA, and BHM methods were developed to determine the nutrient thresholds at which Chl *a* variation changes across nutrient concentrations, and to explore ecoregional differences and seasonal variations in nutrient criteria in the seven lake ecoregions, China (Huo et al., 2015c). Using similar methods, Zhang et al. (2016b) determined the nutrient criteria of nine typical lakes in China that differ in the degree of disturbance and trophic status and developed a conceptual model to address ecological succession along the gradient of disturbance. Using sestonic Chl *a* as the response variable, Haggard et al. (2013) applied regression models and CART to establish nutrient criterion thresholds in the Red River Basin in the US. By integrating the graphical representation of a Bayesian network model with an empirical model-developing approach, a continuous variable Bayesian network model was presented to establish nitrogen criteria for small rivers and streams in Ohio, US (Qian and Miltner, 2015).

Using data from 22 lake watersheds in the Yungui Lake ecoregion (China), Huo et al. (2015b) found that the ratio of

**Table 1 – Stressor-response models.**

| Model | Benefits | Drawbacks | Applications |
|---|---|---|---|
| LRM | • Quantitatively analyses the influence of a nutrient increase on the response variables<br>• Does not require identification of reference lakes or least-impacted lakes in the study region or the collection of historical data | • Must set the response criteria<br>• The data must be classified to eliminate the influence of confounding factors on the relationship<br>• The model extrapolation process increases uncertainty<br>• The land use-nutrient regression model cannot quantify all anthropogenic sources | • Heavily impacted by human activities and a good linear relationship between the stressor variable and response variable in the region |
| BHRM | • The influences of covariate variables can be adjusted on all levels to predict the variability of the output results<br>• Effectively overcomes missing data issues and measurement error to evaluate and compare the relative heterogeneity and avoid under- or overestimation | • Must set the response criteria<br>• The modeling dataset is generated randomly, which leads to slight differences in the calculated results | |
| CART | • A nonparametric method that makes no assumptions about the underlying distribution of values of the predictor variables<br>• Does not require the establishment of a response threshold value | • Model lacks robustness when the number of samples is small<br>• The accuracy of the results might not be guaranteed | • Heavily impacted by the existence of human activities; ecological responses to environmental gradients are nonlinear, non-normal, and heterogeneous<br>• The stressor-response relationship cannot be expressed by linear models in a region |
| CPA | • Evaluates the positions of thresholds or changepoints in binary relationships and provides natural candidates for nutrient criteria<br>• Does not require the establishment of a response threshold value | • Additional analyses are required to determine if the characteristics of the selected value are consistent with a protective target<br>• Requires estimation if the values of the response variable below the changepoint support the designated uses of waterbodies | |

land use to lake depth was strongly correlated with mean annual nutrient and Chl *a* concentrations in log–log models. The nutrient and Chl *a* criteria were determined using nutrient-land use/lake depth regression models. Land use is a strong predictor of the TN and TP yields because it is related to the intensity of soil erosion and is the primary non-point source of nutrient inputs (Huo et al., 2015b; Ma et al., 2016). Analyzing data from watersheds in four ecoregions in east-central Kansas, Dodds and Oakes (2004) established regression models for $\log_{10}$TP and $\log_{10}$TN yields as a function of the percentage of cropland and the percentage of urban land.

Stressor-response models are suitable for deriving the nutrient criteria of waterbodies affected by anthropogenic activities due to the widespread contamination of aquatic ecosystems by industrialization, urbanization, and agriculture (Huo et al., 2013a). Many studies have suggested that the relationship between nutrients and Chl *a* might be confounded by environmental factors such as salinity, light, temperature, water color, and suspended sediment (Lee et al., 1978; US EPA, 2010; Vincent, 2001; Wong and Chang, 2000; Yang et al., 2016). These factors should be identified or included in future models. In addition, in one study, a land use-nutrient regression model was not able to quantify all sources of anthropogenic influence due to that such data were not readily available (Huo et al., 2015b). Quantitative consideration of factors such as the atmospheric deposition of nitrogen, the distribution of animal farms, and point discharges would enhance the accuracy of such models (Huo et al., 2015b; Ma et al., 2016).

### 2.4. Model inference approach

The model inference approach represents ecological systems using equations that can empirically calibrate the relationship between ecological processes and water quality parameters using site-specific data. These models can be used to predict changes in the system based on changes in N and P concentrations (US EPA, 2010). The model inference approach can establish a continuous evaluation baseline, which is suitable for lakes influenced by intensive human activities. However, this approach requires sufficient data to identify the appropriate equations for characterizing a waterbody or group of waterbodies and to calibrate the parameters in these equations.

The model inference approach has been widely developed in the US, Europe, and other developed countries to determine nutrient criteria. The US EPA recommended using the morphoedaphic index (MEI) method and mass balance models to establish nutrient criteria (US EPA, 2000a). The MEI is the ratio of total dissolved solids (as measured by alkalinity or conductivity) in water to the mean depth of the waterbody

(US EPA, 2000a; Ryder, 1982; Vighi and Chiaudani, 1985). Because both alkalinity and conductivity are generally minimally impacted by anthropogenic activities in the catchment and biological activity within the lake (Ryder et al., 1974), the MEI has been extended to predict phosphorus concentrations resulting from natural background loading in undisturbed watersheds (Vighi and Chiaudani, 1985). Using data from 567 European lakes, Cardoso et al. (2007) applied an MEI model to derive TP criteria in five lake regions and compared the derived criterion values with those obtained using the statistical analysis approach. The relationships between the annual mean TP or Chl *a* and $MEI_{cond}$ were determined to deduce the TP criteria in undisturbed deep lakes in the Yungui Lake ecoregion of China (Huo et al., 2014c). However, the MEI model does not consider the entire TP load that reaches a lake; it only estimates the TP concentration using conductivity and alkalinity as proxies, and it lacks a component related to discharge (Salerno et al., 2014).

Mass balance models can be used to estimate nutrient concentrations based on lake loading and the hydrological conditions of the lake. A mass balance model by itself cannot be used to establish nutrient criteria, but it can predict nutrient concentrations given certain loading values. Thus, mass balance models require estimates of the natural background nutrient loading to derive nutrient criteria for a waterbody. A frequency distribution approach combined with loading and mass balance models was used to estimate the nutrient criteria of waterbodies in Ohio, US (US EPA, 2000a). This method would be appropriate for stream-fed lakes only and not lakes with substantial groundwater inputs (US EPA, 2000a).

A shallow lake can be transformed from a clear state with submerged macrophytes to a turbid state dominated by phytoplankton if the phosphorus load exceeds a critical value, causing eutrophication (Janse et al., 2008). The PCLake ecological model, a dynamic model of nutrient cycling, biota, such as phytoplankton and macrophytes, and a simplified food web, was used to calculate the critical nutrient thresholds of various lake types in Europe (Janse et al., 2008). The PCLake model was also used to simulate lake nutrient dynamics and investigate key factors that cause eutrophication, such as in-lake nutrient cycling, a long residence time and shallow depths (Laspidoua et al., 2017; Kong et al., 2017). Using data from the western Boreal Plain in Alberta, Canada, the TP threshold for the transition from clear to turbid conditions was 275 μg TP/L for lakes with high levels of submersed aquatic vegetation (SAV), whereas the TP threshold was 50 μg TP/L for lakes with low levels of SAV (Bayley et al., 2007). The PCLake model was developed to determine spatial patterns in critical nutrient loads in Lake Taihu, China (Janssen et al., 2017).

Many models such as MONERIS, AQUATOX, SPARROW, and SWAT were also widely used to determine nutrient criteria. Hirt et al. (2014) applied the MONERIS model to reconstruct four scenarios of nutrient release in rivers in the German Baltic Sea catchment and derive nutrient criteria. The results were in accordance with historical and calculated pristine nutrient concentrations from other rivers in similar catchments. The AQUATOX ecosystem model can be used to investigate impairment thresholds based on biological indexes that can be simulated (Carleton et al., 2009). A calibrated AQUATOX model was applied to predict the nutrient changes based on different species in a landscape lake, and the results provided important information for the management of the lake (Niu et al., 2016). The SPARROW model was used to estimate summer nutrient concentrations in northeastern lakes and provide useful information regarding the lake quality for the development of nutrient criteria (Milstead et al., 2013). The Soil and Water Assessment Tool (SWAT) model can remove the effects of anthropogenic-impacted land use and simulate natural conditions to predict nutrient criteria. Using the SWAT model, the reference nutrient conditions were determined for small and large streams in the Genesee River watershed (Makarewicz et al., 2015). Using monthly observed streamflow values from three flow stations, the SWAT model was calibrated and validated to accurately assess and predict hydrological processes for water quality improvement in the Lake Dianchi basin (Zhou et al., 2014). Through an assessment of the uncertainty and accuracy of the MEI models, export coefficients, and diatom- and pigment-inferred TP models of 35 subalpine lakes, a new process-based watershed approach was proposed to predict nutrient criteria with lower uncertainty than other methods (Salerno et al., 2014).

### 2.5. Comparison with various approaches

More than one method can be used to establish nutrient criteria. It is important to cross-validate results from different methods to ensure high confidence in estimates of nutrient criteria in ecoregions where many approaches are appropriate (Dodds et al., 2006; Huo et al., 2014c). For example, numerical modeling and Bayesian analysis were integrated to establish water quality criteria (TP and Chl *a*) in Hamilton Harbor, Canada, and ecosystem restoration measures were proposed to increase the probability of meeting water quality goals (Ramin et al., 2011). Herlihy et al. (2013) calculated potential nutrient criteria for TP, TN, and Chl *a* using the 25th percentile of the population, the 75th percentile of the least-disturbed reference sites, diatom-based paleolimnological reconstruction, and stressor modeling and compared the results with existing US EPA draft criteria in national nutrient ecoregions. The results indicated that the ecoregional criteria derived using the different approaches were highly correlated at the national scale (Herlihy et al., 2013). Heatherly (2014) examined TN and TP criteria for rivers and streams in Nebraska using whole-population nutrient percentiles, reference stream percentiles, model-predicted estimates of the relationships between nutrients and anthropogenic land use, and stressor-response modeling of nutrients and invertebrate and fish assemblages and discussed the applicability of the criteria in agriculturally intensive streams in Nebraska, US.

Nutrient criterion thresholds or ranges determined using different methods in various countries and regions are listed in Table 2. There were no significant differences in the results obtained using the methods in the same region in China. The nutrient criteria determined using the CART, nCPA, and BHM methods were almost the same in the seven lake ecoregions of China and were within the ranges of criteria obtained using the LRM method (except in the Yungui Lake ecoregion) (Huo

**Table 2 – Criterion thresholds or ranges for TN, TP, Chl *a*, and SD determined using different methods in various countries and regions.**

| Country or region | TP (mg/L) | TN (mg/L) | Chl *a*(μg/L) | SD (m) | Method | Source |
|---|---|---|---|---|---|---|
| **US** | | | | | | |
| Ecoregion I | 0.055 | 0.660 | 4.88 | 2.55 | LPDM | US EPA, 2001b |
| Ecoregion II | 0.009 | 0.100 | 1.90 | 4.50 | LPDM | US EPA, 2001b |
| Ecoregion III | 0.017 | 0.400 | 3.40 | 2.70 | LPDM | US EPA, 2001b |
| Ecoregion IV | 0.020 | 0.440 | 2.00 | 2.00 | LPDM | US EPA, 2001b |
| Ecoregion V | 0.033 | 0.560 | 2.30 | 1.30 | LPDM | US EPA, 2001b |
| Ecoregion VI | 0.038 | 0.780 | 8.59 | 1.36 | LPDM | US EPA, 2001b |
| Ecoregion VII | 0.015 | 0.660 | 2.63 | 3.33 | LPDM | US EPA, 2001b |
| Ecoregion VII | 0.008 | 0.240 | 2.43 | 4.93 | LPDM | US EPA, 2001b |
| Ecoregion IX | 0.020 | 0.360 | 4.93 | 1.53 | LPDM | US EPA, 2001b |
| Ecoregion X | 0.060 | 0.570 | 5.50 | 0.80 | LPDM | US EPA, 2001b |
| Ecoregion XI | 0.008 | 0.460 | 2.79 | 2.86 | LPDM | US EPA, 2001b |
| Ecoregion XII | 0.010 | 0.520 | 2.60 | 2.10 | LPDM | US EPA, 2001b |
| Ecoregion XIII | 0.018 | 1.270 | 12.35 | 0.79 | LPDM | US EPA, 2001b |
| Ecoregion XIV | 0.008 | 0.320 | 2.90 | 4.50 | LPDM | US EPA, 2001b |
| Central Great Plains | 0.044–0.062 | 0.695 | 11.00 | 0.66–1.17 | RLM, TM and LRM | Dodds et al., 2006 |
| Central Irregular Plains | 0.020–0.027 | 0.362 | 8.00–11.00 | 1.09–1.30 | RLM, TM and LRM | Dodds et al., 2006 |
| Fint Hills | 0.019–0.023 | 0.301 | 5.00–9.00 | 1.12–1.49 | RLM, TM and LRM | Dodds et al., 2006 |
| Western High Plains | 0.025–0.027 | 0.201–0.658 | 13.00 | 0.93–1.14 | RLM, TM and LRM | Dodds et al., 2006 |
| Everglades | 0.011–0.014 | – | – | – | nCPA and BHM | Qian et al., 2003 |
| Red River Basin | 0.100–0.220 | 0.750–2.110 | – | – | LRM and CART | Haggard et al., 2013 |
| Streams, Genesee River watershed | 0.076 | – | – | – | SWAT Model | Makarewicz et al., 2015 |
| **Europe** | | | | | | |
| Atlantic(L-A1) | 0.009 | – | 2.60–3.80 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Atlantic(L-A2) | 0.008 | – | 2.60–3.80 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Atlantic(L-A3) | 0.010 | – | – | – | RLM, LRM, and MEI | Cardoso et al., 2007 |
| Atlantic(L-AX) | 0.008 | – | – | – | RLM, LRM, and MEI | Cardoso et al., 2007 |
| Alpine(L-AL3) | 0.005 | – | 2.00–2.80 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Alpine(L-AL4) | 0.008 | – | 2.70–3.30 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Alpine(L-ALX) | 0.012 | – | – | – | RLM, LRM, and MEI | Cardoso et al., 2007 |
| Central-Baltic(L-CB1) | 0.021 | – | 2.60–3.80 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Central-Baltic(L-CB2) | 0.018 | – | 6.20–7.50 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Central-Baltic(L-CB3) | 0.016 | – | 2.50–4.80 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Mediterranean(L-MX) | 0.017 | – | – | – | RLM, LRM, and MEI | Cardoso et al., 2007 |
| Nordic(L-N1) | 0.010 | – | 2.50–3.50 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Nordic(L-N2a) | 0.008 | – | 1.50–2.50 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Nordic(L-N2b) | 0.006 | – | 1.50–2.50 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Nordic(L-N3) | 0.013 | – | 4.10–13.80 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Nordic(L-N5) | 0.007 | – | 1.00–2.00 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |

*(continued on next page)*

**Table 2** (*continued*)

| Country or region | TP (mg/L) | TN (mg/L) | Chl *a*(μg/L) | SD (m) | Method | Source |
|---|---|---|---|---|---|---|
| **Europe** | | | | | | |
| Nordic(L-N6) | 0.009 | – | 2.00–3.80 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Nordic(L-N8) | 0.014 | – | 3.50–7.80 | – | RLM, LRM, and MEI | Cardoso et al., 2007; Carvalho et al., 2008; Poikāne et al., 2010 |
| Nordic(L-NX) | 0.011 | – | – | – | RLM, LRM, and MEI | Cardoso et al., 2007 |
| Nine European lakes | 0.013–0.067 | – | – | – | PRA | Bennion et al., 2011 |
| Shallow lakes, Sweden | 0.070–0.100 | 1.700 | – | – | MIA | Blindow and Haregy, 1993 |
| Shallow lakes, Netherlands | 0.030–0.050 | 1.000 | – | – | MIA | Hosper, 1998 |
| Rivers German, Baltic Sea watershed | 0.050 | 1.000 | – | – | MONERIS Model | Hirt et al., 2014 |
| **Canada** | | | | | | |
| Atlantic Maritime | 0.010–0.030 | 0.870–1.200 | – | – | LPDM, LRM | Chambers et al., 2012 |
| Montane Cordillera | 0.020 | 0.210 | – | – | LPDM, LRM | Chambers et al., 2012 |
| Mixedwood Plains | 0.030 | 1.100 | – | – | LPDM, LRM | Chambers et al., 2012 |
| Interior prairies of Canada | 0.100 | 0.390–0.980 | – | – | LPDM, LRM | Chambers et al., 2012 |
| Western Boreal Plain, Alberta | 0.050–0.275 | – | – | – | MIA | Bayley et al., 2007 |
| **China** | | | | | | |
| Eastern Plain ecoregion | 0.014–0.043 | 0.360–0.785 | 3.92 | 0.85 | LPDM | Huo et al., 2013b |
| Eastern Plain ecoregion | 0.029 | 0.670 | 1.78–4.73 | 0.68–1.21 | TM | Huo et al., 2013b |
| Eastern Plain ecoregion | 0.020–0.032 | 0.250–0.474 | – | – | LRM | Zhang et al., 2014 |
| North Lake ecoregion | 0.019 ± 0.0039 | 0.846 ± 0.473 | – | – | LRM | Huo et al., 2014a |
| North Lake ecoregion | 0.022 | 1.685 | – | – | CART | Huo et al., 2015c |
| North Lake ecoregion | 0.022 | 0.708 | – | – | nCPA | Huo et al., 2015c |
| North Lake ecoregion | 0.022 | 0.698 | – | – | BHM | Huo et al., 2015c |
| Mid-East Lake ecoregion | 0.0218 ± 0.0074 | 0.374 ± 0.139 | – | – | LRM | Huo et al., 2014a |
| Mid-East Lake ecoregion | 0.038 | 0.643 | – | – | CART | Huo et al., 2015c |
| Mid-East Lake ecoregion | 0.038 | 0.579 | – | – | nCPA | Huo et al., 2015c |
| Mid-East Lake ecoregion | 0.038 | 0.565 | – | – | BHM | Huo et al., 2015c |
| Southeast Lake ecoregion | 0.010–0.025 | 0.310–0.650 | – | – | LPDM | Huo et al., 2014d |
| Southeast Lake ecoregion | 0.020 | 0.550 | – | – | TM | Huo et al., 2014d |
| Southeast Lake ecoregion | 0.0293 ± 0.01 | – | – | – | LRM | Huo et al., 2014a |
| Southeast Lake ecoregion | 0.030 | 0.532 | – | – | CART | Huo et al., 2015c |
| Southeast Lake ecoregion | 0.030 | 0.511 | – | – | nCPA | Huo et al., 2015c |
| Southeast Lake ecoregion | 0.025 | 0.510 | – | – | BHM | Huo et al., 2015c |
| Northeast Lake ecoregion | 0.053 ± 0.021 | 0.863 ± 0.64 | – | – | LRM | Huo et al., 2014a |
| Northeast Lake ecoregion | 0.026 | 0.643 | – | – | CART | Huo et al., 2015c |
| Northeast Lake ecoregion | 0.026 | 0.670 | – | – | nCPA | Huo et al., 2015c |
| Northeast Lake ecoregion | 0.022 | 0.670 | – | – | BHM | Huo et al., 2015c |
| Ganxin Lake ecoregion | 0.016 ± 0.005 | 0.421 ± 0.293 | – | – | LRM | Huo et al., 2014a |
| Ganxin Lake ecoregion | 0.047 | 0.508 | – | – | CART | Huo et al., 2015c |
| Ganxin Lake ecoregion | 0.047 | 0.877 | – | – | nCPA | Huo et al., 2015c |
| Ganxin Lake ecoregion | 0.048 | 0.885 | – | – | BHM | Huo et al., 2015c |
| Ningmeng Lake ecoregion | 0.146 | 1.745 | – | – | CART | Huo et al., 2015c |
| Ningmeng Lake ecoregion | 0.159 | 1.745 | – | – | nCPA | Huo et al., 2015c |
| Ningmeng Lake ecoregion | 0.157 | 1.750 | – | – | BHM | Huo et al., 2015c |
| Yungui Lake ecoregion | 0.010 | 0.175 | 2.20 | 5.50 | RLM | Huo et al., 2012 |
| Yungui Lake ecoregion | 0.010 | 0.370 | 2.00 | 2.20 | LPDM | Huo et al., 2012 |
| Yungui Lake ecoregion | 0.010 | 0.210 | 1.59 | 3.50 | TM | Huo et al., 2012 |
| Yungui Lake ecoregion | 0.008 ± 0.0018 | 0.173 ± 0.049 | – | – | LRM | Huo et al., 2014a |
| Yungui Lake ecoregion | 0.019 | 0.494 | – | – | CART | Huo et al., 2015c |
| Yungui Lake ecoregion | 0.015 | 0.494 | – | – | nCPA | Huo et al., 2015c |
| Yungui Lake ecoregion | 0.015 | 0.300 | – | – | BHM | Huo et al., 2015c |
| Yungui Lake ecoregion | 0.010 | – | – | – | MEI | Huo et al., 2012 |
| **Brazil** | | | | | | |
| Tropical Rivers and Streams in São Paulo State | 0.010 | 0.350 | – | – | TM | Cunha et al., 2012 |

Note: RLM-reference lake method; LPDM-lake population distribution method; TM-trisection method; PRA-paleolimnological reconstruction approach; and MIA-model inference approach; TN: total nitrogen; TP: total phosphorus; Chl *a*: chlorophyll a; SD Secchi depth.

et al., 2014a, 2015c). In the Yungui Lake ecoregion, the criteria determined using the statistical analysis approach were consistent with those derived using the LRM and MEI methods (Huo et al., 2012, 2014a). Table 2 shows that there are significant regional differences in the nutrient criteria among various regions and that, in the absence of human activities, environmental factors, rather than TP and TN (*e.g.*, salinity, light, temperature, water color, and suspended sediment), promote or inhibit the growth of algae (Huo et al., 2015c; Yang et al., 2016). Furthermore, differences in the stressor-response relationships between nutrients and Chl *a* were found among lakes in various regions. High nutrient concentrations did not always correspond to a high Chl *a* concentration. For example, to achieve the same Chl *a* concentration (approximately 2.6–3.8 μg/L), there was a 0.08 mg/L TP criteria for the L-A2 type and 0.021 mg/L TP criteria for the L-CB1 type in Europe (Table 2). Although many studies have shown high correlations between Chl *a* and SD, lake depth, organic matter, dissolved color, and particulate matter affect the SD (Huo et al., 2012; Carlson and Havens, 2005). Hence, a correspondingly high SD is not necessary when the Chl *a* concentration is low (such as in ecoregions I and V in the US).

Due to the lack of data in some ecoregions, an order of preference in applying these different techniques should be considered when determining criteria. In regions with minimally impacted reference lakes, the reference lake method should be given preference to determine the regional nutrient criteria (Dodds et al., 2006; Huo et al., 2014c). In regions where undisturbed or nearly undisturbed conditions are uncommon but sufficient data are available, the population distribution and trisection methods should be used to determine the regional nutrient criteria (Huo et al., 2013b). For deep lakes with undisturbed sediments and strong relationships between biota and water quality, the paleolimnological reconstruction approach can be used to infer past chemical conditions (US EPA, 2000a). If degraded conditions prevail and appropriate data exist to adequately quantify the relationships between variables, the stressor-response relationship would be preferred to provide a statistically defensible method (Dodds et al., 2006; Lamon and Qian, 2008). If sufficient data are available to identify the appropriate equations and reliably calibrate the parameters in these equations, the model inference approach can be used to account for the site-specific effects of N and P enrichment and link changes in concentrations to the impairment of designated uses (US EPA, 2010). If only one method can be used to determine criteria due to data availability or type limitations, professional judgment and expertise are required.

## 3. Developments and challenges in nutrient criteria determination in China

The reference lake method is preferred for deriving nutrient criteria because a reference lake can reflect the original state of an ecoregion. However, reference sites have become less available due to climate change and atmospheric pollution. The paleolimnological reconstruction approach requires complex data analysis and expert interpretation, and it is difficult to reconstruct the previous nutrient conditions of shallow lakes in which sediments are easily disturbed. The model inference approach requires sufficient data to identify the appropriate equations and regulate the parameters in the equations (US EPA, 2010). Current studies of the mechanisms of eutrophication are lacking, and the model inference approach has not been widely applied (Huo et al., 2013a). Hence, the stressor-response model is a possible development direction for nutrient criteria determination in China.

The development of methods for nutrient criteria determination faces the following challenges:

(1) The mechanisms of the stressor-response models should be further researched. The influence of the nutrient transformation process on the criteria and the ecological effects caused by the combined actions of nutrients remain unclear. Limited systematic research is available regarding the environmental behavior and ecological effect of differences in the primary productivity (such as algae) on various nutrient levels. Furthermore, environmental factors such as the biogeographic properties of species, watershed area, salinity, color, and suspended solid concentration affect the migration and transformation of nitrogen, phosphorus and other nutrients, thereby complicating the stressor-response relationship between nutrients and algae. Therefore, research on stressor-response mechanisms must be improved, and the influences of complex factors on the response relationship should be eliminated. For cases in which a linear relationship is unsuitable for a given waterbody, nonlinear stressor-response relationships can be used to determine nutrient criterion thresholds.

(2) Research on the relationships between ecological criteria and eutrophication should be expanded. The purpose of nutrient criteria determination is to control water eutrophication, as the ecological effects of algae were different at the same nutrient level. For example, algae are vulnerable to mass proliferation and cause eutrophication in lakes/reservoirs with long hydraulic retention times (HRTs), whereas algae have difficulty aggregating in lakes/reservoirs with short HRTs. The response relationships between nutrients and plankton require further research considering various environmental factors, such as hydrological dynamics and conditions (Jones et al., 2008). Research on the determination of nutrient criteria and the mechanisms of algal blooms would be beneficial for the effective prevention and control of eutrophication.

(3) The relationships between nutrient criteria and plankton, especially the sensitive species, should be elucidated. For example, the relative abundance and richness of diatoms provide multiple sensitive indexes to assess changes in environmental stressors and can be used to describe the complexity, stability, and functionality of a lake ecosystem. Thus, diatoms can be applied to explore complex relationships, determine the nutrient criteria thresholds of ecological degradation, and validate derived nutrient criteria.

(4) The determination of nutrient criteria should be closely integrated with the management demand. The

translation of nutrient criteria to standards should be accelerated, and a management framework and nutrient reduction technology should be established based on the eutrophication standard. Management policies related to eutrophication control should be studied and developed, including the establishment of nutrient reduction systems, anti-degradation strategies, eutrophication control strategies, and nutrient reduction plans, as well as policies regarding downstream effects and trade policies.

## Acknowledgment

This study is supported by the National key research and development program of China (No. 2017YFA0605003) and the National Natural Science Foundation of China (No. 41521003).

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