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A Review of Stream Nutrient Criteria Development in the United States

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Elevated nutrients and sediments are the main factors contributing to the poor biological condition measured in over 40% of US waters, highlighting the need for criteria that can aid management efforts to protect or restore the quality of US waters. A large amount of literature on nutrient criteria has been generated since the USEPA called for their development in 1998. Our objective was to examine this peer-reviewed literature to evaluate two main approaches for criteria development in lotic ecosystems: percentile rank and bivariate predictive statistical analyses. The 25th percentile approach has been examined broadly across USEPA-aggregate nutrient ecoregions, and we found that USEPA-suggested criteria for these aggregate ecoregions were often more conservative than criteria estimated using more current regionally focused data based on our compiled data set. Furthermore, 25th percentile estimates were often less than 75th percentile estimates based on reference sites, suggesting that 75th percentile estimates were not more conservative than 25th percentile estimates. Predictive approaches have focused on establishing linear and nonlinear relationships between water quality and algae, macroinvertebrate, and fish communities; attributing causation; and determining whether threshold points exist that can aid in nutrient criteria development. Most of the predictive approaches have occurred at the state or watershed level and may not be directly comparable to USEPA aggregate ecoregions. However, percentile method estimates often fell within the confidence interval of biological threshold criteria estimates, suggesting overlap and some consensus between the two main approaches.

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Approximately 40% of the United States freshwater wadeable streams are considered impaired (i.e., not meeting designated use criteria) or have poor biological conditions (USEPA, 2002; USEPA, 2006). Elevated nitrogen (N), phosphorus (P), and fine sediments were identified as the main stressors affecting these ecosystems (USEPA, 2006; Paulsen et al., 2008), and stream total N (TN) and total P (TP) exceeded the 75th percentile of the reference condition more than 50% of the time (USEPA, 2006). The condition of wadeable streams is important because they account for approximately 90% of the total length of perennial streams and rivers in the United States and can have significant impacts on water quality downstream in larger rivers (Dodds and Oakes, 2006; Alexander et al., 2008; Dodds and Oakes, 2008; Mulholland et al., 2008).

The 1998 Clean Water Action Plan was a presidential initiative to provide a blueprint for establishing nutrient criteria that would aid in management efforts to protect and restore US waters. Each state or tribe was expected to derive nutrient criteria values using scientific methods. These methods could focus on estimating reference conditions or on statistical analysis that related nutrient variables to algal biomass or to changes in biological or ecological condition, indicating eutrophication. Since the first call for development of nutrient criteria, a large amount of peer-reviewed literature has been produced focusing on statistical approaches for nutrient criteria development, but an analysis examining the consensus outcomes of this large body of literature that could help guide future research and criteria development is lacking.

Nutrient criteria have been estimated using two main approaches: percentile analysis and predictive relationships. Percentile analysis identifies reference reaches based on analysis of data frequency distributions. Predictive relationships include those between nutrient variables and response variables representing ecological condition to establish desired levels for criteria. The objective of this report is to summarize and evaluate the use of the percentile analysis of nutrient or biological variables and the use of predictive statistical analyses to aid development of nutrient criteria for wadeable streams and rivers. Predictive analyses review focused on benthic and sestonic algae,

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Abbreviations: 2DKS, 2-dimensional Kolmogorov-Smirnov; AFDM, ash-free dry mass; ANE, aggregate nutrient ecoregion; chl-a, chlorophyll a; NAWQA, National Water Quality Monitoring Network; nCPA, nonparametric changepoint analysis; TN, total nitrogen; TP, total phosphorus.

benthic macroinvertebrates, and fishes. We chose not to focus on macrophytes, which can be indicators of interest in some streams (Demars and Edwards, 2009; Demars et al., 2012). Finally, we focused our efforts on peer-reviewed literature from the United States published after 1998.

Percentile Analysis of Nutrient or Algal Variables

The goal of percentile analysis approaches is to identify a reference condition for regions by using the 75th percentile of reference streams or the 25th percentile of a general population of streams within regions (USEPA, 2000). These two statistical measures were thought to be relatively equal, and each region was supposed to be a population of streams with similar biological, ecological, chemical, and physical features that determine water quality in the absence of human impact. In 2000, the USEPA provided an initial TN, TP, suspended chlorophyll a (chl-a), and turbidity criteria for streams (Tables 1–3). These criteria were based on percentile analysis of streams grouped into larger-scale combinations of similar Omernik Level III ecoregions called aggregate nutrient ecoregions (ANEs). Given the lack of reference streams in many ANEs, they reported the lower 25th percentile of a population of all streams within each region. Since this initial attempt by the USEPA to provide regional nutrient criteria, several studies have used a similar approach to define criteria and have compared their analyses with the USEPA's original proposed nutrient criteria. The results of these studies are reviewed in this section and are reported in Tables 1, 2, and 3. We had two main objectives when compiling these percentile data: (i) to determine whether 75th percentiles of reference streams were equal to or more conservative than 25th percentile estimates of a general population of streams and (ii) to compare individual study percentile estimates with USEPA percentile estimates (USEPA, 2000) to determine whether more focused regional studies resulted in more or less conservative estimates than those originally proposed by the USEPA.

The assumption that the 25th percentile estimates are equal or more conservative than the 75th percentile estimates does not appear to be valid based on previous studies or on our cross-study analysis. Two studies have directly compared 25th and 75th percentile estimates calculated from the same regional data set (Suplee et al., 2007; Herlihy and Sifneos, 2008). The reference sites used by Suplee et al. (2007) were identified using qualitative (Best Professional Judgment) and quantitative (watershed land use assessments and local water quality variables) approaches. The reference sites chosen by Herlihy and Sifneos (2008) were defined using several approaches, including an Environmental Monitoring and Assessment Program screening method and a GIS and site-specific information method to quantify human disturbance. These studies found that 75th percentile of reference stream estimates were only more conservative than 25th percentile general population estimates in region III (Xeric West) for TN and region II (Western Forested Mountains) for TP (Tables 1 and 2). In all other regions, the 75th percentile estimates ranged from 1.2 to 2.1 and from 1.3 to 8.5 times higher than the 25th percentile estimates for TN and TP, respectively. We computed the ratio of 75th percentile to 25th percentile (i.e., from the same study or from the USEPA) estimates from each study using measured water quality data (Rohm et al., 2002; Suplee et al., 2007; Herlihy and Sifneos, 2008) and found that the mean ratio was greater for TP (3.9 ± 0.66 [mean ± 1 SE]; $n = 13$) than for TN (1.6 ± 0.1; $n = 13$) (two-tail t-value = 2.1; $p = 0.004$). This pattern in the ratio for TP suggests that 75th percentile estimates across all studies have been generally less conservative than 25th percentile estimates.

The inclusion of human-impacted streams in the reference site pool could result in 75th percentile estimates being less conservative than 25th percentile estimates. This may happen if relatively unimpacted reference sites are rare, causing managers to use sites in moderately developed watersheds (Dodds and Oakes, 2004). It also may be difficult to assess the degree of human impact (Suplee et al., 2007). Multiple-regression models may be used to estimate a reference condition in regions where

Table 1. Total nitrogen (25th and 75th percentiles) across USEPA nutrient ecoregions taken from peer-reviewed literature and compared with USEPA suggested criteria.

Reference	Aggregate USEPA nutrient ecoregion													
	I	II	III	IV	V	VI	VII	VIII	IX	X	XI	XII	XIII	XIV
	mg L^{-1}													
USEPA, 2000 (suggested criteria)	0.31	0.12	0.38	0.56	0.88	2.18	0.54	0.38	0.69	0.76	0.31	0.9	–	0.71
	25th percentiles of a general population													
Palmstrom, 2005	–	–	–	–	–	–	0.48	0.29	2.01	–	0.29	–	–	1.85
Robertson et al., 2006b	–	–	–	–	–	–	1.56‡	0.40	–	–	–	–	–	–
Suplee et al., 2007	–	0.08†	–	0.61	0.60	–	–	–	–	–	–	–	–	–
Herlihy and Sifneos, 2008	–	0.07	0.78	0.44	0.99	1.86	0.58	0.27	0.33	0.92	0.16	–	–	0.62
Longing and Haggard, 2010	–	–	–	0.61	0.86	–	–	–	0.53	–	0.21	–	–	–
	75th percentiles of a reference population													
Rohm et al., 2002	–	–	–	–	–	–	–	–	–	–	0.375	–	–	–
Smith et al., 2003	0.18	0.18	0.05	0.12	0.37	0.44	0.17	0.18	0.17	0.55	0.17	0.61	0.65	0.63
Smith et al., 2003 (N deposition incorporated)	0.21	0.21	0.11	0.21	0.51	0.62	0.33	0.28	0.28	0.67	0.29	0.71	0.79	0.76
Suplee et al., 2007	–	0.13†	–	1.30	1.12	–	–	–	–	–	–	–	–	–
Herlihy and Sifneos, 2008	–	0.15	0.29	0.93	1.19	2.5	–	0.39	0.68	–	0.29	–	–	–

† Value represents a mean of medians for the Northern Rockies, Middle Rockies, and Canadian Rockies.

‡ Value represents a mean for the following level III Omernik ecoregions: Driftless Area, Northcentral Hardwood Forests, and Southeastern Wisconsin Till Plains.

Table 2. Total phosphorus 25th and 75th percentiles across USEPA aggregate nutrient ecoregions taken from peer-reviewed literature and compared with USEPA suggested criteria.

Reference	Aggregate USEPA nutrient ecoregion													
	I	II	III	IV	V	VI	VII	VIII	IX	X	XI	XII	XIII	XIV
	μg L⁻¹													
USEPA, 2000 (suggested criteria)	47	10	21.9	23	67	76.2	33	10	36.6	128	10	40	–	3.8
	25th percentiles of a general population													
Palmstrom, 2005	–	–	–	–	–	–	20	16	42	–	12	–	–	82
Robertson et al., 2006	–	–	–	–	–	70	40	10	90	–	20	–	–	–
Robertson et al., 2006b	–	–	–	–	–	–	60‡	24	–	–	–	–	–	–
Suplee et al., 2007	–	13†	–	20	20	–	–	–	–	–	–	–	–	–
Herlihy and Sifneos, 2008	–	3	10.4	18.9	34.4	65.8	17	6.8	20.4	147	3.9	–	–	22.7
Longing and Haggard, 2010	–	–	–	20	70	–	–	–	60	–	20	–	–	–
	75th percentiles of a reference population													
Rohm et al., 2002	–	–	–	–	–	–	–	–	–	–	13	–	–	–
Smith et al., 2003	20	20	30	70	70	60	30	20	50	60	20	30	40	20
Suplee et al., 2007	–	9†	–	170	140	–	–	–	–	–	–	–	–	–
Herlihy and Sifneos, 2008	–	19	40	86.8	107	181	–	10.2	60.1	–	17.7	–	–	–

† Value represents a mean of medians for the Northern Rockies, Middle Rockies, and Canadian Rockies.

‡ Value is a mean of the following level III Omernik ecoregions: Driftless Area, Northcentral Hardwood Forests, and Southeastern Wisconsin Till Plains.

most watersheds are at least moderately developed (Dodds and Oakes, 2004). Initially, regional variation in TN and TP concentrations can be explained using region as a categorical predictor and land use as a covariate. Then, multiple linear regressions where land use classifications are independent variables and where nutrient concentrations are dependent variables can be used to calculate a y-intercept, which represents the expected nutrient concentration in the absence of human activity. This approach assumes that anthropogenic effects on water quality are confined to land-use modifications in the watershed. However, atmospheric N deposition can also alter stream chemistry in even primarily forested watersheds (Flum and Nodvin, 1995). Smith et al. (2003) attempted to correct for anthropogenic N deposition and landscape effects on reference stream TN concentrations by developing an empirical model incorporating regression models and the SPARROW transport model. The model included data collected before 1998 from 63 minimally impacted US Geological Survey (USGS) reference basins located in the 14 USEPA ANEs of the United States. When atmospheric N deposition was incorporated into the model, the 75th percentile of TN background concentrations in each region were 15 to 100% higher than those estimated when deposition was not incorporated (Table 1). Therefore, managers may want to consider atmospheric anthropogenic nutrient inputs in addition to terrestrial inputs when estimating reference concentrations depending on their management objectives.

Although the ANEs proposed by the USEPA were a useful initial way to group streams and can explain some spatial variability in TN and TP data, they may be too coarse of a scale for establishing nutrient criteria (Rohm et al., 2002) and for establishing reference water quality conditions (Robertson et al., 2006). The objective of spatial classification in the criteria development process is to minimize within-region variability and maximize across-region variability in the main environmental variables affecting water quality. Smith et al. (2003) found that as much as one order of magnitude of variation existed in background nutrient concentrations within ANEs. This variation was primarily due to variation in runoff due to variation in elevation and differences in cumulative in-stream loss at junctions of small tributaries and large rivers. Land use can also explain significant variation in TN and TP data within ecoregions (Wickham et al., 2005) and is often related to the factors used to define ecoregions (Robertson et al., 2006). The consequence of this is that human land uses, such as agricultural and urban lands, can then elevate

Table 3. Suspended chlorophyll-a and turbidity 25th and 75th percentiles across USEPA aggregate nutrient ecoregions taken from peer-reviewed literature and compared with USEPA-suggested criteria.

Reference	Aggregate USEPA nutrient ecoregion													
	I	II	III	IV	V	VI	VII	VIII	IX	X	XI	XII	XIII	XIV
	Chlorophyll-a (μg L⁻¹)													
USEPA, 2000	1.80	1.08	1.78	2.40	3.00	2.70	1.50	0.63	0.93	2.10	1.61	0.40	–	3.75
Longing and Haggard, 2010	–	–	–	1.52	6.78	–	–	–	3.76	–	0.75	–	–	–
Palmstrom, 2005	–	–	–	–	–	–	–	–	3.47	–	4.35	–	–	4.00
Smith and Tran, 2011	–	–	–	–	–	–	2.30	–	–	–	–	–	–	–
	Turbidity													
USEPA 2000	4.25†	1.30‡	2.34†	4.21†	7.83†	6.36†	1.70‡	1.30†	5.70†	17.50†	2.30‡	1.90‡	–	3.04†
Palmstrom, 2005	–	–	–	–	–	–	1.70‡	1.40‡	4.00‡	–	1.60‡	–	–	4.50‡
Smith and Tran, 2011	–	–	–	–	–	–	2.70‡	–	–	–	–	–	–	–

† Turbidity measurements in FTU.

‡ Turbidity measurements in NTU.

the resulting percentile concentrations and the variation in water quality variables within ecoregions (Robertson et al., 2006).

Stream classification approaches should endeavor to minimize the importance of anthropogenic factors that cause variation in stream water quality if the intention is to identify minimally human-impacted conditions. Robertson et al. (2006) suggested using a simultaneous partial-residualization approach to remove anthropogenic land use effects on water quality data and on environmental variables used to characterize criteria regions. Streams could then be grouped spatially by using regression tree analysis (SPARTA) (Robertson and Saad, 2003) of land use–adjusted residualized water quality and environmental data. This modification of the SPARTA approach allows the dominant environmental factors corrected for anthropogenic land use and their relative weightings used in the spatial categorization to be quantified and to vary with the water quality variable of interest. Regions identified using this land use–adjusted SPARTA technique often had a smaller range of reference concentrations quantified across regions and less variation within regions when compared with the ecoregion approach (Robertson et al., 2006).

A basin approach to setting nutrient criteria may be more appropriate than an ecoregion approach in some lotic ecosystems. Management at the basin level may be needed to meet criteria goals in higher-order lotic ecosystems and in large basins that can include multiple ecoregions and political boundaries (e.g., states, tribes), as is the case in the Red River Basin (Longing and Haggard, 2010). Longing and Haggard (2010) examined extensive water quality data from 589 streams and river stations from 1996 to 2006 in the Red River Basin, which includes the Central and Forested Uplands, Great Plains Grass and Shrublands, South Central Cultivated Great Plains, and the Southeastern Temperate Forested Plains. The 25th percentiles for TN were similar to or below USEPA suggested criteria in each ANE (Table 1), but TP and sestonic chl-a 25th percentiles were most often greater than those recommended by the USEPA (Table 3), which the authors suggested might be due to the positioning of the Red River Basin at the geographic edge of four ANEs, land use variations in the basin, or variations in data acquisition. In contrast to some other studies, Longing and Haggard (2010) found no statistically significant difference among TN, TP, or chl-a criteria computed within level III ecoregions of the Southeastern Temperate Forested Plains and Hills ANEs. Samples sizes were not large enough to test level III ecoregion differences in the other ANEs sampled. Therefore, nutrient criteria development in the Red River basin could focus more on ANEs than on level III ecoregions.

We quantified the ratio of the literature-estimated 25th and 75th criteria from Tables 1 and 2 and the USEPA-suggested criteria for TN and TP in each ANE (Fig. 1 and 2) to examine how similar USEPA and regionally specific study percentile estimates were and to determine whether variation in percentile estimates was dependent on ANEs. We did not include Smith et al. (2003) estimates in this analysis because of the difficulty in comparing volumetrically weighted modeled nutrient concentrations to median or mean measured nutrient concentrations. In general, most ratio values were greater than 1 across all ANEs for TN and TP, suggesting that the USEPA-suggested criteria were more conservative than criteria using more recent or regionally focused data. Literature TN criteria for the Great Plains Grass and Shrublands (IV), the Mostly Glaciated Dairy Region (VII), and the Southeastern Temperate Forested Plains and Hills (IX) were most variable, and 95th percentiles were up to approximately 2.5 times higher than USEPA-suggested criteria (Fig. 1). Literature TP criteria were most variable for the Great Plains Grass and Shrublands (IV), and the median value of the ratio for the Southeastern Temperate Forested Plains and Hills (IX) was approximately 14 (Fig. 2). Large variation or

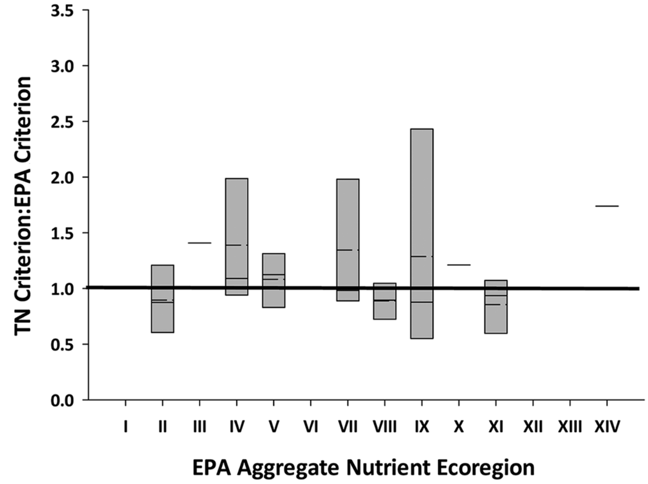

Fig. 1. Boxplots of the ratio of the literature-estimated 25th and 75th percentile total nitrogen (TN) criterions to the USEPA-suggested TN criterion for each aggregate nutrient ecoregion using data from Table 1. Smith et al. (2003) was not included in the analysis because volumetrically weighted estimated concentrations from the SPARROW model were not comparable to median concentrations computed in other studies. The boundary of the box closest to zero represents the 25th percentile. The boundary farthest from zero represents the 75th percentile. The mean and median are represented by the broken and solid lines within bars, respectively. The bold solid line represents a ratio of 1 where the USEPA criterion would be equal to the study criterion.

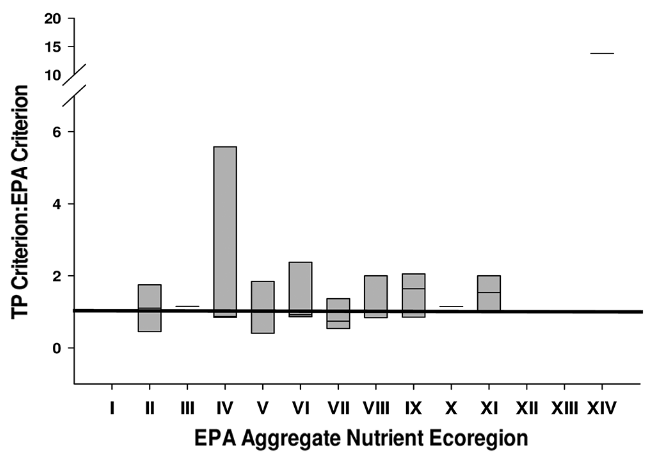

Fig. 2. Boxplots of the ratio of the literature-estimated 25th and 75th percentile total phosphorus (TP) criterions to the USEPA-suggested TP criterion for each aggregate nutrient ecoregion using data from Table 2. Smith et al. (2003) was not included in the analysis because volumetrically weighted estimated concentrations from the SPARROW model were not comparable to median concentrations computed in other studies. The boundary of the box closest to zero represents the 25th percentile. The boundary farthest from zero represents the 75th percentile. The mean and median are represented by the broken and solid lines within bars, respectively. The bold solid line represents a ratio of 1 where the USEPA criterion would be equal to the study criterion.

deviation in percentile estimates may be due to several factors, including variation in sampling procedures across studies, basin positioning on boundaries of multiple ecoregions, or the need for a more appropriate stream classification system.

A percentile analysis of chl-a data has been used as the basis for establishing stream trophic state (Dodds et al., 1998). An analysis of published data for temperate stream sites proposed using the lower and upper third of sestonic or benthic chl-a distributions to establish the boundary between oligotrophic-mesotrophic systems and between mesotrophic-eutrophic systems (Dodds et al., 1998). Oligotrophic–mesotrophic boundaries fell at 20 mg m^{-2}, 60 mg m^{-2}, and 10 μg L^{-1} for mean benthic chl-a, maximum benthic chl-a, and suspended chl-a, respectively. Mesotrophic–eutrophic boundaries fell at 70 mg m^{-2}, 200 mg m^{-2}, and 30 μg L^{-1} for mean benthic chl-a, maximum benthic chl-a, and suspended chl-a, respectively. Less percentile data on suspended chl-a were available compared with TN and TP data in the peer-reviewed literature. Of the studies examining suspended chl-a, most have estimated criteria at a higher concentration of chl-a than the USEPA criteria (Table 3). Criteria estimates from 25th percentile of general stream populations for the South Central Cultivated Great Plains and the Southeastern Temperate Forested Plains and Hills were 2 to 4 times higher than USEPA estimates. These percentile estimates for benthic and sestonic chl-a assume that there is a clear link between nutrient concentrations and algal biomass. However, at the spatial scales relevant to nutrient criteria, many factors, including nutrients, can affect algal biomass (see Predictive Relationships section).

Upper and lower quantile distributions of turbidity data were also rare (Table 3). Most studies have estimated greater turbidity criteria than USEPA-suggested criteria. One estimate existed for the Southeastern Temperate Forested Plains and Hills, and it was lower than the USEPA-suggested criteria (Palmstrom, 2005). The USEPA-suggested criteria for all water quality parameters were more conservative than estimates from individual studies, suggesting that more regional analyses are needed and that lessons learned from studies examining TN and TP likely will apply to suspended algae and turbidity as well.

Predictive Relationships between Nutrients and Ecological Conditions

Threshold Analyses

Predictive relationships between criteria variables and response variables can be linear or nonlinear. Linear analyses are more prevalent than nonlinear responses even though many responses to eutrophication are nonlinear (Dodds et al., 2010). A system can often respond rapidly with a relatively small change in a criteria variable (e.g., TN, TP, chl-a, turbidity), and the challenge becomes identifying the point or threshold where that rapid change occurs in an objective manner so that a criteria concentration can be estimated. Statistical threshold analyses have received considerable attention in water quality criteria literature since the USEPA first requested criteria development. Much of the current literature cited in this review used these threshold approaches, but we will not do an exhaustive review of the benefits and limitations of each approach because Dodds et al. (2010) have already reviewed many of these methods. Methods included in the Dodds et al. (2010) review were breakpoint or piecewise regression, cumulative frequency distributions (Paul and McDonald, 2005; Utz et al., 2009; Hilderbrand et al., 2010), nonlinear curve fitting, nonparametric changepoint analysis (nCPA) (King and Richardson, 2003; Qian et al., 2003), quantile regression (Chaudhuri and Loh, 2002; Cade and Noon, 2003), recursive partitioning or regression tree (Breiman et al., 1984; De'ath and Fabricius, 2000; De'Ath, 2002), regime shift detection (Sonderegger et al., 2009; Gal and Anderson, 2010), significant zero crossings (Sonderegger et al., 2009), threshold indicator taxa analysis (Baker and King, 2010), and two-dimensional Kolmogorov-Smirnov test (2DKS) (Garvey et al., 1998). Dodds et al. (2010) also examined thresholds in macroinvertebrate richness across stream TP concentrations using breakpoint regression, cumulative frequency, quantile regression tree, nCPA, 2DKS test, regime shift, and significant zero crossings to assess variability associated with statistical methodology. They found that threshold estimates varied 3-fold depending on the type of analyses. Breakpoint regression, 2DKS, and significant zero crossings yielded the greatest (i.e., least conservative) threshold concentrations. The authors suggested that a more conservative approach to estimating thresholds would be most prudent because it is not known that a stream can be restored to a prior condition once it has crossed a threshold point (Dodds et al., 2010).

Algal Community Responses

Suspended Algae

Suspended algae are generally positively related with nutrient concentrations (Haggard et al., 2013). A compilation of literature data that included temperate streams primarily from the United States and Europe found a positive curvilinear relationship between river-suspended chl-a and TP ($R^2 = 0.67$; $n = 292$) (Van Nieuwenhuyse and Jones, 1996). Relationships between suspended chl-a and nutrients have also been found at a smaller spatial scale. Lohman and Jones (1999) examined factors explaining suspended chl-a at 23 sites on 13 Missouri Ozark streams and found a curvilinear relationship between TN (range, 0.220–8.435 mg L^{-1}; $R^2 = 0.70$) and TP (range, 0.006–3.093 mg L^{-1}; $R^2 = 0.78$). When the relationship between nutrients and suspended chl-a was limited to 17 sites without known point-source pollution, relationships became linear, likely because the range of nutrient concentrations observed at these sites occurred in the linear concentration range of algal growth (TN range, 0.172–0.765 mg L^{-1}; TP range, 0.006–0.119 mg L^{-1}). Percent row crop and forested land use also were good predictors of suspended chl-a at sites without known point-source pollution, providing some support for a causal link between nonpoint sources and suspended chl-a (Lohman and Jones, 1999). The amount of variation in suspended chl-a described by nutrient concentrations was significant in these studies. However, the turbulent and unidirectional flow of streams can inhibit naturally reproducing suspended algal populations (Reynolds, 2000), and relationships with nutrients should be considered within the framework of hydrology.

Relationships between nutrient concentrations and suspended chl-a often include catchment area, which has been positively related to and can explain from around 10 to 20% (Van Nieuwenhuyse and Jones, 1996; Lohman and Jones, 1999) to as much as 50% (Royer et al., 2008) of the variation in suspended chl-a. These relationships may be due to physical factors relating to catchment area, such as hydraulic flushing rate (Kilkus et al., 1975; Soballe and Kimmel, 1987; Honti et al., 2010). Reservoirs located in river catchments, which increase water residence time and reduce hydraulic flushing, can be sources of suspended algae. The Kalamazoo River in Michigan has seven impoundments whose reservoir phytoplankton populations increased and became greater suspended algal sources to rivers with increasing water residence time (Reid and Hamilton, 2007). Free-flowing sections generally had longitudinally declining concentrations of suspended algae even though N and P concentrations were high throughout the river length, suggesting that these sections were sinks for algal cells.

Benthic algal communities may also be sources of suspended algal cells, and their contributions may depend on hydrology. Sloughing may contribute to suspended chl-a at low flow, and scour may contribute at high flow. Lohman and Jones (1999) examined whether time after a catastrophic flooding event altered the relationship between explanatory factors and suspended chl-a. They found that TP, TN, and catchment area remained the main factors explaining variation in suspended chl-a 0, 14, 28, and 42 d after a catastrophic flood (55–74% of variance explained), but models explained less variance than those based on long-term averages. Models based on long-term averages also overestimated suspended chl-a right after flood events, which is not surprising given that much of the algal washout likely left during the storm event and the benthic algal populations remaining were lower.

Relationships between suspended chl-a and nutrients may depend on light availability in addition to catchment area and hydrology. Royer et al. (2008) examined suspended chl-a collected in statewide surveys of >100 Illinois stream and river sites, with 75% of the sites having a TP concentration ≥0.112 mg L^{-1} (range, 0.007–2.8 mg L^{-1}) and TN ≥1.0 mg L^{-1} (range, 0.21–18.7 mg L^{-1}) at base-flow discharge. Watershed area was the best predictor of suspended chl-a, as has been found in other studies (R^2 range, <0.20–0.51) (Van Nieuwenhuyse and Jones, 1996; Lohman and Jones, 1999). However, no relationships were found between nutrient concentrations and seston chl-a at high flow or at base flow. The lack of relationship between water chemistry and algal variables might be due to variable light levels across streams. A statistically significant correlation was found between TP and suspended chl-a in the low flow data when sites with canopy cover ≤25% and TP ≤0.2 mg L^{-1} were examined ($r = 0.62$), and a threshold was visually identified at approximately 0.07 mg L^{-1}.

Benthic Algal Biomass Metrics

Nutrients can often limit benthic algal growth in streams (Tank and Dodds, 2003). Therefore, it is not surprising that many positive relationships exist between benthic algal biomass measures and nutrient concentrations. Dodds et al. (2002) and Dodds and Oakes (2006) compiled temperate stream periphyton biomass and nutrient concentration data from the literature ($n = 300$) and from a subset of 620 National Water Quality Monitoring Network (NAWQA) sites to determine if water column nutrients and non-nutrient factors (e.g., temperature, latitude, land use, and substrate type) were linked to periphyton biomass. The greatest amount of variation in mean and maximum periphyton chl-a was explained by TN or TP (~40%; positive relationships). Further variation in the benthic chl-a data could result from hydrological disturbances (Lohman et al., 1992; Biggs, 1995; Biggs, 2000), differences in grazing pressure (Stevenson et al., 2006), or variation in light availability and current velocity (Hill, 1996; Stevenson, 1996). Therefore, relationships describing a greater amount of variation in algal biomass may be more attainable at a smaller spatial scale where these variables can be carefully measured.

Regional-scale studies that attempted to account for local variations in resources and conditions have also reported positive relationships between nutrient concentrations and benthic algal measures. Busse et al. (2006) measured algal cover, algal biomass as chl-a, and physical and chemical variables at 14 sites with a range of land use types in a southern California watershed. They found that benthic chl-a was positively correlated with TN (range, 0.395–4.490 mg L^{-1}) and TP (range, 0.037–0.398 mg L^{-1}). Total P explained more variation ($r = 0.86$–0.88) in algal measures than did TN ($r = 0.75$–0.82). A quadratic relationship between TP and benthic chl-a explained more variation than a linear one, suggesting that saturating conditions were reached between 100 to 200 μg TP L^{-1}. Percent full sun and current velocity were also positively related to benthic chl-a in some seasons.

The effects of grazing on benthic algae can be as significant as the effect of nutrients (Hillebrand, 2002). Therefore, studies should endeavor to control for grazing impacts on benthic algal–nutrient relationships. Stevenson et al. (2006) examined benthic algal biomass in 104 streams located in Kentucky, where hydrology constrains invertebrate grazer biomass, and in Michigan streams that had a more stable hydrology and greater macroinvertebrate grazer densities. Positive correlations were found between nutrients and benthic chl-a and percent area of substratum covered by *Cladophora* in both regions, but *Cladophora* responded more positively to nutrients in Michigan than in Kentucky. This greater slope associated with Michigan streams is partly driven by much lower *Cladophora* cover in low-nutrient streams in Michigan relative to Kentucky. This result may have been due to higher grazing pressure in Michigan streams. Therefore, although TN and TP explained similar amounts of variance in benthic chl-a ($r = 0.30$–0.43) and *Cladophora* cover ($r = 0.30$–0.67) within each region, the magnitudes of slopes differed regionally and may have been driven by regional differences in flow regime and grazing pressure.

Another way to examine the link between nutrients and algae is to determine nutrients limiting algal growth. Stevenson et al. (2008) measured the potential for P limitation across a gradient of TP in mid-Atlantic Highland streams ($n = 607$). Acid and alkaline phosphatase production was measured along with periphyton chl-a, ash-free dry mass (AFDM), and diatom taxonomic composition. Acid and alkaline phosphatase activity was negatively related to TP, suggesting that the increasing concentrations of TP relieved P limitation. Chlorophyll *a* and AFDM were positively related to TP concentrations. Thresholds calculated by nCPA occurred between 0.01 and 0.02 mg L^{-1} (Table 4) and corresponded with the 75th percentile concentration of reference sites, which was 0.012 mg L^{-1} in this region.

Benthic Algal Community Composition

Many studies have found that algal community composition can provide a strong relationship with nutrient concentrations (Stevenson et al., 2006; Porter et al., 2008; Stevenson et al., 2008; Justus et al., 2010; Black et al., 2011). Stevenson et al. (2008)

found that the number of diatom taxa, evenness, proportion of expected native taxa, and the number of high P taxa were positively related to TP. Total P generally explained more variation (R^2 = 0.04–0.33) for these community composition variables than for algal standing stock measures such as chl-a (R^2 = 0.07) and AFDM (R^2 = 0.07). Porter et al. (2008) analyzed benthic algal community metrics from 976 streams and rivers collected from 1993 to 2001 through the NAWQA Program and found that algal species indicators for taxa richness, salinity, organic enrichment, motility, and trophic condition had positive correlations with TN and TP concentrations (range, r = 0.31–0.57). In addition, diatom species associated with high dissolved oxygen were often negatively correlated with TN and TP (range, r = 0.34–0.40). Justus et al. (2010) also found positive correlations between a nutrient index that combined TN and TP and algal indices, including the relative abundance of most tolerant diatoms (r = 0.80), the combined relative abundance of three *Cymbella* spp. (r = −0.71), mesosaprobic algae percent taxonomic richness (r = 0.65), and the relative abundance of obligate N heterotrophic diatoms (r = 0.57).

The substrate type sampled may alter the algal community composition relationships with nutrient concentrations. Black et al. (2011) examined algal communities on coarse-grained (i.e., rock and wood >64 mm) and fine-grained substrate from

Table 4. Benthic algal total nitrogen and total phosphorus thresholds determined by various statistical analyses.

Dependent variable†	Criteria estimation method‡	TN§ estimated criteria	TP¶ estimated criteria	Citation
		mg L^{-1}		
Mean chl-a	regression	0.537	0.043	Dodds et al., 2002, 2006
Maximum chl-a	regression	0.602	0.062	Dodds et al., 2002, 2006
Mean chl-a	2DKS	0.515	0.027	Dodds et al., 2002, 2006
Maximum chl-a	2DKS	0.367	0.027	Dodds et al., 2002, 2006
chl-a	regression tree	0.918	0.039	Robertson et al., 2006b
Diatom nutrient index	regression tree	1.169	0.072	Robertson et al., 2006b
Diatom siltation index	regression tree	0.872	0.074	Robertson et al., 2006b
Diatom biotic index	regression tree	1.169	0.072	Robertson et al., 2006b
Mean chl-a	nCPA	NA	0.013	Stevenson et al., 2008
Mean AFDM	nCPA	NA	0.008	Stevenson et al., 2008
Acid phosphatase activity	nCPA	NA	0.006	Stevenson et al., 2008
Alkaline phosphatase activity	nCPA	NA	0.006	Stevenson et al., 2008
Number of diatom taxa	nCPA	NA	0.011	Stevenson et al., 2008
Diatom evenness	nCPA	NA	0.019	Stevenson et al., 2008
Proportion of native diatom taxa	nCPA	NA	0.011	Stevenson et al., 2008
Proportion of low-P native taxa	nCPA	NA	0.018	Stevenson et al., 2008
Diatom species similarity to reference	nCPA	NA	0.026	Stevenson et al., 2008
Low-P diatom individuals, %	nCPA	NA	0.018	Stevenson et al., 2008
High-P diatom individuals, %	nCPA	NA	0.011	Stevenson et al., 2008
Mean chl-a	nCPA	0.435#	0.038	Miltner, 2010
		Fine-grained depositional substrate		
Abundance of pollution tolerant diatoms, %	regression	0.86	0.28	Black et al., 2011
Alkalophilus diatom richness	regression	NS††	0.05	Black et al., 2011
Abundance of pollution-sensitive diatoms, %	regression	NS	0.09	Black et al., 2011
Abundance of high-TN diatoms, %	regression	0.61	0.06	Black et al., 2011
Abundance of high-TP diatoms, %	regression	0.71	0.06	Black et al., 2011
Abundance of N heterotrophs, %	regression	1.5	0.10	Black et al., 2011
Abundance of motile algae, %	regression	0.27	0.06	Black et al., 2011
Richness of motile algae, %	regression	1.49	0.09	Black et al., 2011
		Coarse-grained substrate (rock or wood)		
Alkalophilus diatom richness	regression	1.25	0.03	Black et al., 2011
Abundance of high TN diatoms, %	regression	1.45	0.07	Black et al., 2011
Abundance of high-TP diatoms, %	regression	1.3	0.08	Black et al., 2011
Abundance of N heterotrophs, %	regression	0.59	0.13	Black et al., 2011
Abundance of motile algae, %	regression	NS	0.20	Black et al., 2011
Richness motile algae, %	regression	1.79	0.07	Black et al., 2011

† AFDM, ash-free dry mass; Chl-a, chlorophyll a.

‡ 2DKS, two-dimensional Kolmogrov Smirnov test; nCPA, nonparametric changepoint analysis. Regression is breakpoint or piecewise.

§ Total nitrogen.

¶ Total phosphorus.

Dissolved inorganic nitrogen.

†† Not significant.

73 stream sites sampled as part of USGS NAWQA. Sites from two agricultural regions (Washington [n = 23] and Nebraska [n = 23]) and sites from across the western United States that were characterized by <10% agricultural or urban land use were used in the analysis. Piecewise regression identified thresholds in several diatom indices (Table 4). Thresholds for fine and coarse-grained substrate were reported, but they were not statistically different, suggesting that taxonomic information from both substrate types may not be needed to establish algal thresholds (Black et al., 2011).

Diatom communities growing on artificial substrates can be used if there is a concern about sampling from natural substrata. Artificial substrata (porcelain crucible covers) were used to examine relationships between diatom community composition and nutrient concentrations from 12 large rivers (>fifth order) in Idaho across a dissolved inorganic N (range, 0.002–1.36 mg L^{-1}) and TP (range, 0.007–0.100 mg L^{-1}) gradient (Snyder et al., 2002). They found no community level response to increasing nutrients. Instead, principle component analysis groups were mainly determined by drainage basin. Ponader et al. (2008) did find relationships between artificial substrate (Plexiglas Catherwood Diatometers) diatom community composition and nutrients in coastal plain clay- and sandy-bottom streams. They found that TP explained a significant portion of the variation, but no relationships were found between diatom community composition and TN.

Large-scale gradients in environmental and geographic factors can also be important determinants of benthic diatom community composition and could be important covariates in analyses. Analysis of NAWQA data indicated that three major factors correlated with diatom communities (Potapova and Charles, 2002). There was a downstream (mountainous headwater to lowland rivers) gradient in diatom community structure that could be the result of a complex mixture of variables, such as decreasing slope and elevation, increasing concentrations of nutrients and temperature, and changes in land use along the continuum. The second gradient observed integrated ionic strength (conductivity and ion concentrations) and pH and resulted in community distinctions between the soft and more acidic waters of the eastern United States to the more alkaline waters of the arid west. The final community gradient was related to variation in mean annual air temperature associated with latitude and altitude gradients. At a continental scale, geographic factors explained up to a third of the variation in diatom community composition data (Potapova and Charles, 2002). These factors may confound relationships with nutrients at the large continental scale and even at smaller regional scales (Ponader et al., 2008) and should be considered during analyses. Diatom species optimal values for conductivity, major ions, and proportions of these ions have been provided based on analysis of the NAWQA data set (Potapova and Charles, 2002) and could be beneficial in interpreting and assessing relationships with nutrients. Diatom nutrient metrics for assessing nutrient enrichment have been developed for the United States (Potapova and Charles, 2007).

Macroinvertebrate and Fish Community Responses

The link between macroinvertebrate and fish communities or traits and water quality was highlighted many years ago (Karr, 1981; Washington, 1984; Hilsenhoff, 1987), and fish and macroinvertebrate bioassessment techniques remain important tools to assess water quality (Waite and Carpenter, 2000; Smith et al., 2007; Justus et al., 2010). The current focus in bioassessment has shifted from not only determining biological traits and species that are associated with changes in water quality but also to developing methods that can attribute causation in large spatial scale observational studies (King and Richardson, 2003; De Zwart et al., 2006; Wang et al., 2007; Yuan, 2010) and to defining particular nutrient concentrations or thresholds where traits or species shift (reviewed by Dodds et al. [2010]). This shift has coincided with the USEPA's Causal Analysis Diagnosis Decision Information System effort (USEPA, 2010). One of the earliest studies endeavoring to attribute causation used observational and experimental data to examine macroinvertebrate responses to changes in TP concentrations in wetlands (King and Richardson, 2003). A nCPA was used to examine threshold shifts in experimental and in observational data, allowing more effective estimates of impairment or risk-associated TP than by examining observational data alone. This analysis was also important because it described a way to attach confidence levels to thresholds (King and Richardson, 2003; Smith et al., 2005).

Several studies have used some type of threshold analysis to examine benthic macroinvertebrate communities (Table 5). Wang et al. (2007) used regression tree analysis, which is similar to nCPA, and a 2DKS analysis to define thresholds in Wisconsin wadeable streams (n = 240) with seasonal median TP and TN concentrations ranging from 0.012 to 1.641 mg L^{-1} and from 0.131 to 21.260 mg L^{-1}, respectively. Percentages and individuals of Ephemeroptera, Trichoptera, and Plecoptera (r = −0.34 to −0.50), the Hilsenhoff biotic index (r = 0.31–0.55), and richness (r = −0.27 to −0.45) had the greatest correlation coefficients, with nutrient measures that included several forms of dissolved N and P. Threshold estimates ranged from 0.61 to 1.68 mg L^{-1} TN and from 0.04 to 0.09 mg L^{-1} TP (Table 5). Weigel and Robertson (2007) also used a regression tree analysis to examine possible threshold in macroinverebrate assemblages sampled from 41 sites on 34 nonwadeable Wisconsin rivers with TN ranging from 0.415 to 5.485 mg L^{-1} and TP ranging from 0.023 to 0.497 mg L^{-1}. The two biological metrics most strongly and consistently correlated with nitrogen and phosphorus variables were taxa richness (r = −0.33 to −0.60) and the mean pollution tolerance value (r = 0.42–0.78). Threshold concentrations associated with reductions in richness were greater than those associated with increases in mean pollution tolerance values (Table 5). Evans-White et al. (2009) also found threshold reductions in richness with increasing TN (r = −0.17 to −0.23) and TP (r = −0.19 to −0.43) (Table 5) in a study examining wadeable streams from Nebraska, Kansas, and Missouri. Total macroinvertebrate taxa richness TN and TP thresholds were similar between the Wang et al. (2007) and the Evans-White et al. (2009) studies, suggesting some agreement in biological responses to nutrients in wadeable streams in the mid-continental United States.

Peer-reviewed threshold analyses of fish community metrics were confined to Wisconsin wadeable and nonwadeable stream assessments (Wang et al., 2007; Weigel and Robertson, 2007). Fish indices, including percentage carnivores (r = −0.27 to −0.49), percentage intolerant (r = −0.27 to −0.40), percentage omnivores (r = 0.28–0.42), index of biotic integrity (r = −0.29

Table 5. Benthic macroinvertebrate total nitrogen and total phosphorus criteria determined by various statistical analyses.

Dependent variable	Criteria estimation method†	TN‡ estimated criteria	TP§ estimated criteria	Citation
		—— mg L^{-1} ——		
Percentage of EPT individuals	regression tree	1.68	0.08	Wang et al., 2007
Percentage of EPT taxa	regression tree	1.3	0.09	Wang et al., 2007
Hilsenhoff Biotic Index	regression tree	1.14	0.09	Wang et al., 2007
Taxa richness	regression tree	0.87	0.04	Wang et al., 2007
Percentage of EPT¶ individuals	2DKS	0.98	0.09	Wang et al., 2007
Percentage of EPT taxa	2DKS	1.11	0.09	Wang et al., 2007
Hilsenhoff Biotic Index	2DKS	0.61	0.09	Wang et al., 2007
Taxa richness	2DKS	0.85	0.04	Wang et al., 2007
Taxa richness	regression tree	1.92	0.15	Weigel and Robertson, 2007
Mean pollution tolerance value	regression tree	0.63	0.06	Weigel and Robertson, 2007
Taxa richness	nCPA	1.04	0.05	Evans-White et al., 2009
Primary consumer richness	nCPA	1.14	0.05	Evans-White et al., 2009
Gathering consumer richness	nCPA	0.93	0.06	Evans-White et al., 2009
Scraping consumer richness	nCPA	NS	0.05	Evans-White et al., 2009
Shredding consumer richness	nCPA	NS	0.05	Evans-White et al., 2009

† 2DKS, two-dimensional Kolmogrov Smirnov test; nCPA, nonparametric changepoint analysis.

‡ Total nitrogen.

§ Total phosphorus.

¶ Ephemeroptera, Trichoptera, and Plecoptera.

to −0.39), and salmonid abundance (r = −0.37 to −0.63) had the greatest correlation coefficients with nutrient measures in wadeable streams. For nonwadeable streams, the index of biotic integrity (r = −0.43 to −0.51) and the percentage of fish biomass composed of round suckers (*Cycleptus* spp., *Hypentelium* spp., *Minytrema* spp., and *Moxostoma* spp.) (r = −0.42 to −0.51) were the most highly correlated with nutrient variables. Mean thresholds of all fish metrics ranged from 0.540 to 1.830 mg L^{-1} of TN and from 0.060 to 0.139 mg L^{-1} TP (Table 6). These ranges in nutrient concentrations where fish community shifts occur were similar to those found for benthic macroinvertebrates (Table 5), suggesting an overall decrease in biological condition within this range.

Threshold-type analyses have also been used to make trophic level categories. Smith et al. (2007) examined nutrients and macroinvertebrates in 129 locations from 116 streams sampled across New York State. They established a nutrient biotic index estimated from species TP and nitrate (NO_3^-) optima that was linearly related to TP (r = 0.65) and to NO_3^- concentrations (r = 0.57). The trophic state of sample sites was then estimated by using additive tree clusters based on mean pairwise Bray-Curtis similarities yielding oligotrophic and eutrophic boundaries at ≤0.0175 and ≥0.065 mg L^{-1} and ≤0.24 and ≥0.98 mg L^{-1} of TN and TP, respectively. They suggested using these trophic boundaries as thresholds to establish nutrient criteria.

Macroinvertebrate and fish metrics are often related to nutrient concentrations (Wang et al., 2007; Weigel and Robertson, 2007), but the role that nutrients play in these relationships relative to other confounding variables (e.g., instream habitat) is important to establish. The Wisconsin wadeable and nonwadeable stream studies used redundancy analysis to attribute the relative effects of different physicochemical variables. In wadeable streams, redundancy analysis indicated that nutrients directly explained 22 and 15% of the variance in

Table 6. Stream fish total nitrogen and total phosphorus thresholds determined by various threshold analyses.

Dependent variable	Criteria estimation method†	TN† estimated criteria	TP‡ estimated criteria	Citation
		—— mg L^{-1} ——		
Percentage of carnivorous individuals	regression tree	1.22	0.09	Wang et al., 2007
Index of biotic integrity	regression tree	1.36	0.07	Wang et al., 2007
Salmonid individuals	regression tree	0.63	0.06	Wang et al., 2007
Percentage of intolerant individuals	regression tree	1.83	0.09	Wang et al., 2007
Percentage of carnivorous individuals	2DKS§	0.54	0.06	Wang et al., 2007
Index of biotic integrity	2DKS	0.54	0.06	Wang et al., 2007
Salmonid individuals	2DKS	0.61	0.06	Wang et al., 2007
Percentage of intolerant individuals	2DKS	0.54	0.07	Wang et al., 2007
Index of biotic integrity	regression tree	0.634	0.139	Weigel and Robertson, 2007
Percent biomass of round suckers	regression tree	0.634	0.091	Weigel and Robertson, 2007

† Total nitrogen.

‡ Total phosphorus.

§ Two-dimensional Kolmogrov Smirnov test.

macroinvertebrate and fish communities, respectively (Wang et al., 2007). However, interactions between nutrients and other environmental variables explained >50% of the variation in both communities. In nonwadeable streams, nutrients, suspended chl-a, water clarity, and land cover (forest or row-crop agriculture) explained 61% of the variation in macroinvertebrate variables, but they were correlated with each other to such an extent that redundancy analysis could not attribute variation to individual factors (Weigel and Robertson, 2007). The same variables explained 44% of the variation in fish community data, with nutrients and other water chemistry variables explaining 25 and 13% of the variation, respectively. Therefore, these analyses suggest that nutrients were among the key factors determining macroinvertebrate and fish metrics.

Another approach to examining the mechanistic effect of nutrient enrichment on biota is to link experimental data with observational data (King and Richardson, 2003; Evans-White et al., 2009). Macroinvertebrate richness can often have negative threshold relationships with water-quality variables at large spatial scales (Wang et al., 2007; Weigel and Robertson, 2007), but the specific mechanisms driving these threshold relationships are not well established, particularly for forested, heterotrophic-based systems. Evans-White et al. (2009) hypothesized that resource quality (i.e., carbon:phosphorus ratios) might partly drive macroinvertebrate primary consumer (grazer, collector, and detritivore) richness thresholds by altering growth or competitive interactions among species with differing resource demands, as has been found in some manipulative P dosing studies (Cross et al., 2006; Cross et al., 2007; Singer and Battin, 2007). Their objective was to determine if mean taxon body C:P ratios of feeding groups with diversity losses were negatively related to TP, which would suggest that communities from enriched streams had relatively more taxa with high dietary P requirements. Primary consumers were more sensitive to TN and TP (threshold mean, 1.0 and 0.06 mg L^{-1}, respectively) than secondary consumers (TP threshold mean, 0.09 mg L^{-1}), a result supporting the resource quality hypothesis because there has been little evidence for N or P limitation in predators. Turbidity reduced richness of all functional feeding groups (threshold mean, 4.7 NTU; range, 1.79 [scraper richness] to 10.75 [collector-filterer richness]), a result suggesting that turbidity and nutrient macroinvertebrate thresholds were caused by different factors. The mean body C:P ratio of detritivorous functional feeding groups (shredding and collector-gathering taxa) declined as TP increased (threshold mean, 0.07 and 0.75 mg L^{-1}, respectively), suggesting that food quality or quantity across this nutrient enrichment gradient might be the cause of detritivore richness declines. Mean scraper C:P was not related to TP, suggesting that factors besides periphyton quality may be causing declines in scraper richness. Results from large, spatial scale assessments (Evans-White et al., 2009; Woodward et al., 2012) and several other manipulative studies (Cross et al., 2006; Cross et al., 2007; Singer and Battin, 2007) provide evidence that enrichment alters detrital resource quantity and quality, and this pattern may contribute to losses in detritivore biodiversity in nutrient-enriched, heterotrophic-based streams. Heterotrophic bases for criteria establishment should be considered in conjunction with the more traditional autotrophic bases for criteria establishment.

De Zwart et al. (2006) used another approach that linked fish, habitat, and chemistry data collected from Ohio River sampling events (n = 1552) that assessed the biological condition at each site and attributed impairment to multiple probable causes. Biological condition was estimated from the proportion of native species predicted to occur at a site that were actually observed. Predicted occurrences were estimated in a similar manner to the River Invertebrate Prediction and Classification System (Wright et al., 1998), which provides species-specific probabilities of capture based on the geographic location and habitat characteristics of each site. Toxic exposure effects were estimated using species sensitivity distributions and toxicity mixture principles, and generalized linear regression models described species abundance and habitat relationships. They found that average losses of fish species in Ohio rivers were 40%, and water chemistry, toxicity, effluent, and habitat loss were estimated to explain 28, 3, 3, and 16% of losses. Site-specific causes were also shown in pie charts mapped onto river segments that allowed easier communication of the causes of impairment.

Yuan (2010) proposed another method that may be used to identify with more confidence causal effects of particular factors in large spatial scale observational data. He used propensity scores that have frequently been used in epidemiological, sociological, and economic studies but had not been used in ecology. Propensity functions can summarize multiple covariate contributions as one parameter. Nutrient concentrations can be related to covariate values with regression, and then the predicted concentrations in each stream becomes the propensity score. The scores can be stratified into groups with similar covariate distributions, and then the causal effects of nutrients on dependent variables can be estimated more confidently. Yuan (2010) used propensity scores to estimate the effect of increasing TN on benthic macroinvertebrate grazers in small streams of the western United States (n = 827 sampling sites). The response of grazer richness to increasing TN varied across groups with similar covariate distributions. In large, wadeable, open-canopied streams, grazer richness was negatively related to TN, but in small, closed-canopied streams, grazer richness responded positively to increasing TN. Benthic chl-a responded positively to TN in each stream type. Thus, Yuan (2010) proposed that increasing algal biomass stimulated grazer richness in small, closed-canopied streams, and shifting algal community composition may have caused declines in grazer richness in the larger streams. Although this method can increase confidence in a causal relationship, it can only account for covariates that are measured.

Synthesis and Conclusions

Two main approaches for nutrient criteria development were assessed in this review. The focus of literature using the percentile approach to nutrient criteria development has been on comparing 25th and 75th percentile approaches, examining factors that cause variability within ANEs, and comparing 25th and 75th percentile results obtained with more modern data or with data collected using a more rigorous statistical design compared with the USEPA-proposed criteria. USEPA-suggested criteria for these ANEs were often more conservative than criteria estimated using more modern data based on our compiled data set (Fig. 1 and 2). A compilation of data comparing 75th and 25th percentile approaches indicated that 75th percentile estimates were generally less conservative than 25th percentile estimates and that the discrepancy between the two percentile estimates is greater for TP than for TN. Therefore, 25th percentiles may not be good surrogates for 75th percentiles and may be a more conservative approach to nutrient criteria development than 75th percentile estimates based on existing reference sites. Large variation or deviation in percentile estimates may be due to several factors, including variation in sampling procedures across studies, basin positioning on boundaries of multiple ecoregions, or the need for a more appropriate stream classification system or spatial scale. Finally, the few estimates of percentile-derived criteria for suspended chl-a and turbidity from the peer-reviewed literature were also often less conservative than USEPA-suggested criteria.

Predictive approaches have focused on establishing relationships between water quality and algae, macroinvertebrate, and fish communities; attributing causation; and determining whether threshold points exist that can aid in nutrient criteria development. Many studies have found linear and nonlinear relationships between benthic and suspended algae, macroinvertebrate communities, and fishes. Most of the predictive approaches have occurred at the state level and may not be directly comparable to USEPA ANEs. However, benthic algal criteria threshold estimates ranged from 0.007 to 0.100 mg L^{-1} for TP and from 0.270 to 1.500 mg L^{-1} for TN (Table 4); these values are within the range of criteria proposed by percentile analysis (Tables 1 and 2). Benthic macroinvertebrate and fish TN thresholds were generally higher than those predicted by benthic algae and ranged from 0.610 to 1.925 mg L^{-1} and from 0.540 to 1.830 mg L^{-1}, respectively (Tables 5 and 6). Published benthic macroinvertebrate TP thresholds ranged from 0.040 to 0.150 mg L^{-1}. Published fish TP thresholds ranged from 0.060 to 0.139 and were comparable to benthic macroinvertebrate thresholds. Although not completely comparable, criteria estimated via the percentile method had similar ranges to biological threshold criteria. Comparability between chemically and biologically derived criteria has also been found for streams in several Canadian regions (Chambers et al., 2011; Chambers et al., 2012a; Chambers et al., 2012b).

The challenge for states developing nutrient criteria is to assess the results of these multiple approaches and to come to a consensus on criteria levels. Several states have site-specific criteria for streams and rivers, but only Wisconsin and Florida have statewide criteria along with several peer-reviewed and technical papers documenting the process of criteria development (Table 7). Florida used a reference- or benchmark-based approach to develop nutrient criteria (FDEP, 2012). First, five nutrient watershed regions were identified based on regional biological, geological, and drainage basin patterns. Then, benchmarks were established using a vetting process that included analysis of a Landscape Development Intensity Index (Brown and Vivas, 2005) to minimize the effects of human impacts on benchmark stream populations. The 90th percentile of concentrations at benchmark sites for most nutrient watershed regions was

Table 7. Summary of numeric nutrient criteria for streams and rivers in water quality standards and the year they were published across the 48 conterminous states (USEPA, 2013).

State†	Local	TN‡	Local	TP§	WQS¶ year
		mg L^{-1}		mg L^{-1}	
Arizona[1]	site-specific	0.50–1.00	site-specific	0.05–0.20	2010
California[2]	site-specific	–	site-specific	–	–
Florida[3]	statewide	0.67–1.87	statewide	0.06–0.49	2012
Georgia[4]	–	–	site-specific	–	2012
Montana[5]	site-specific	0.13–1.36	site-specific	0.01–0.12	2012
Nevada[6]	site-specific	1.5–2.9 (2.4–4.0)	site-specific	0.10–0.33 (0.05–0.10)	2012
New Jersey[7]	site-specific	2.0	statewide	0.10	2011
New Mexico[8]	–	–	site-specific	0.10	2012
New York[9]	site-specific	–	–	–	–
Oklahoma[10]	–	–	site-specific	0.04	2002
Oregon[11]	–	–	site-specific	0.07	2012
Vermont[12]	statewide	0.2–5.0	site-specific	0.01	2012
Washington[13]	–	–	site-specific	0.03	2012
Wisconsin[14]	–	–	statewide	0.07–0.10	2012

† [1]Annual mean of at least 10 samples collected 10 d apart with a 12-mo period. [2]California's numeric nutrient criteria were not readily available through review of the water quality standards. Regional water quality control plans were not reviewed but are available. [3]Annual geometric mean not exceeded more than once in 3 yr. [4]Georgia defines phosphorus loading limits to specific lotic water bodies, which are not translated into a phosphorus concentration. [5]Montana numeric criteria were pulled from a white paper making such recommendations, and these numbers apply seasonally (i.e. July–September). [6]Nevada defined site-specific criteria by single values exceeding or as average seasonal value for April through November shown in parentheses. [7]New Jersey nitrogen criteria is for nitrate as nitrogen, not total nitrogen. [8]New Mexico did not define numeric criteria for total nitrogen. [9]New York's numeric criteria for total nitrogen were not found through review of the water quality standards, and phosphorus criteria have not been recommended. [10]Oklahoma has a site-specific total phosphorus criterion for its scenic rivers. [11]Oregon's phosphorus criteria apply to the monthly median during low flow between May and October. [12]Vermont's numeric nutrient criteria vary with the elevation of the streams and rivers across the state, where lesser concentrations apply at the higher elevations. [13]Washington's phosphorus criteria applies to the Spokane River in the euphotic zone from June through October. [14]Wisconsin's phosphorus criteria are statewide but may vary by receiving lake or reservoir or watershed.

‡ Total nitrogen.

§ Total phosphorus.

¶ Water quality standards.

used as the criteria level. A stream condition index based on Florida streams (e.g., Didonato et al. [2003]) was used to help confirm the health of streams greater than the 90th percentile, which minimized the probability of classifying an impaired site as acceptable. Wisconsin completed an extensive analysis of their wadeable and nonwadeable rivers, including developing their own spatial classification system based on environmental phosphorus zones (Robertson et al., 2006), correlation and regression tree threshold analyses, and redundancy analysis. Wisconsin statewide TP criteria range from 0.07 to 0.10 mg L^{-1}. This is toward the high range of results of percentile analysis (Table 2) (Robertson et al., 2006b) and biological threshold analysis (Tables 4–6) (Robertson et al., 2006b; Wang et al., 2007; Weigel and Robertson, 2007; Robertson et al., 2008). Therefore, both states have considered chemically and biologically derived data when developing criteria. In cases where chemically and biologically derived criteria do not overlap, managers may want to further consider the relative rigor of approaches that depend on data quality and quantity to determine which criteria to adopt (Chambers et al., 2012a).

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