Policy Analysis

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Eutrophication of U.S. Freshwaters: Analysis of Potential Economic Damages

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Human-induced eutrophication degrades freshwater systems worldwide by reducing water quality and altering ecosystem structure and function. We compared current total nitrogen (TN) and phosphorus (TP) concentrations for the U.S. Environmental Protection Agency nutrient ecoregions with estimated reference conditions. In all nutrient ecoregions, current median TN and TP values for rivers and lakes exceeded reference median values. In 12 of 14 ecoregions, over 90% of rivers currently exceed reference median values. We calculated potential annual value losses in recreational water usage, waterfront real estate, spending on recovery of threatened and endangered species, and drinking water. The combined costs were approximately $2.2 billion annually as a result of eutrophication in U.S. freshwaters. The greatest economic losses were attributed to lakefront property values ($0.3−2.8 billion per year, although this number was poorly constrained) and recreational use ($0.37−1.16 billion per year). Our evaluation likely underestimates economic losses incurred from freshwater eutrophication. We document potential costs to identify where restoring natural nutrient regimes can have the greatest economic benefits. Our research exposes gaps in current records (e.g., accounting for frequency of algal blooms and fish kills) and suggests further research is necessary to refine cost estimates.

Introduction

Human-induced eutrophication is occurring throughout the world (*1*). Eutrophication reduces water quality and alters ecological structure and function of freshwaters (*2, 3*). Biological impacts of eutrophication are well understood, however degree and costs are not. Potential economic losses can be related to social, ecological, and policy-related responses (*4*).

Assigning economic value to an ecosystem function or service has been widely debated, with investigators employing a variety of methodologies (*4*−*7*). When reliable estimates of prospective economic losses from human-caused environmental impacts can be set, they can potentially define problems for policy makers and direct focus to areas with the greatest potential societal costs.

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Lakes and rivers provide drinking water, recreation, and aesthetic benefits, all of which can be negatively influenced by eutrophication (Figure 1). Taste and odor problems increase in frequency and severity when eutrophication induces potentially toxic cyanobacterial blooms (*8*). Recreational angling and boating activities can be physically impeded by eutrophication-driven macrophyte growth and algal blooms (*8*). Water users are less likely to swim, boat, and fish during heavy algal blooms due to health risks, unfavorable appearance, or unpleasant odors (*4*). Property values can decrease with declines in water clarity (*9*). All these negative impacts can substantially influence the value of freshwater ecosystems (*3, 10, 11*).

Eutrophication management has centered on phosphorus control (*12*). Documentation of economic harm from eutrophication is limited. Characterizing costs is particularly important because the U.S. requires nutrient criteria (*13*) and other countries (e.g., the European Union Water Framework Directive) also regulate nutrients.

We investigated freshwater services where economic losses can occur from human-induced eutrophication. We first established the degree of eutrophication relative to reference nutrient conditions by U.S. nutrient ecoregion. Then we used published information to estimate potential economic losses with respect to recreational water usage, waterfront property values, threatened and endangered species recovery efforts, and spending on drinking water.

Methods

Current and Reference Nutrients. The U.S. Environmental Protection Agency (EPA) collected total nitrogen (TN) and total phosphorus (TP) data from rivers and streams (hereafter rivers) and lakes and reservoirs (henceforth lakes) for the National Nutrient Strategy aggregated level III ecoregions, (hereafter referred to as nutrient ecoregions; *14, 15*). Ambient data came from the EPA Legacy and STOrage and RETreival (STORET) data system, U.S. Geological Survey National Stream Quality Accounting Network (NASQAN), and the National Water Quality Assessment (NAWQA; (*16*)). For this data set, the EPA collected nutrient data from a representative sample of the population of water bodies in each nutrient ecoregion, and data reduction methods were used to prevent

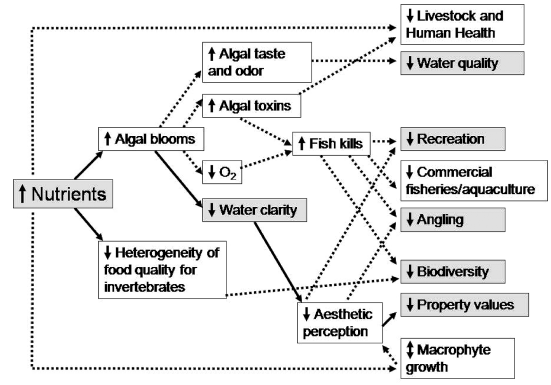

FIGURE 1. Some effects of increased nutrients that could influence the value of freshwater ecosystem goods and services. The values we could assign are in gray, the solid lines indicate the chain of influence we used to calculate the values. Some other pathways are discussed in the text as well. More indirect methods were required to calculate some other effects (see Methods for details).

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TABLE 1. Reference and Current Median TP and TN Concentrations for Rivers in Each Nutrient Ecoregion during Summer Months[a]

ecoregion	ecoregion name	reference TP median mg·L^{-1}	current TP median mg·L^{-1} (*N*)	% of rivers > reference median	reference TN median mg·L^{-1}	current TN median mg·L^{-1} (*N*)	% of rivers > reference median
I	Willamette and Central Valleys	0.016	0.088 (178)	96	0.121	0.301 (16)	76
II	Western Forested mountains	0.019	0.026 (1380)	60	0.147	0.248 (239)	69
III	Xeric west	0.021	0.055 (808)	75	0.041	0.561 (153)	100
IV	Great Plains grass and shrublands	0.046	0.087 (341)	67	0.081	0.956 (65)	100
V	Central cultivated Great Plains	0.049	0.184 (489)	86	0.191	1.283 (94)	100
VI	Corn belt and Northern Great Plains	0.052	0.168 (815)	90	0.313	3.372 (77)	100
VII	Mostly glaciated dairy region	0.022	0.080 (910)	87	0.139	0.928 (125)	99
VIII	Nutrient poor glaciated upper Midwest and Northeast	0.013	0.021 (608)	65	0.156	0.566 (72)	97
IX	Southeastern temperate forested plains and hills	0.048	0.080 (2104)	68	0.141	1.457 (274)	99
X	Texas-Louisiana costal and Mississippi alluvial plains	0.048	0.176 (295)	99	0.339	1.019 (36)	92
XI	Central and Eastern Forested uplands	0.020	0.022 (1591)	53	0.148	0.712 (290)	94
XII	Southern coastal plain	0.025	0.103 (466)	85	0.521	1.216 (90)	99
XIII	Southern Florida coastal plain	0.036	0.080[b]	87	0.631	2.666[b]	100
XIV	Eastern coastal plain	0.015	0.077 (375)	95	0.540	1.141 (56)	88

[a] *N* is the number of stations providing data used for estimating current nutrient distributions. The percentage of current rivers whose TP and TN concentration is greater than the reference median was calculated as one minus the percentile of the estimated reference median in the estimated current distribution. [b] No gauging stations available. Current values estimated by regression, see Methods.

biased records from water bodies with a greater number of samples than others. In particular, each water body was represented by the median concentration of all samples taken from that river or lake. Laboratories used standard methods or U.S. EPA certified methods and were required to employ QA/QC procedures. Distribution of median water body nutrient concentrations from the EPA's nutrient data was ascertained for each water body type (i.e., rivers and lakes) in all nutrient ecoregions. All statistical analyses were conducted by nutrient ecoregion.

Reported TN and TP percentile concentrations were used to estimate the log-normal distribution of nutrient concentrations. Distributions for summer months were used because that is generally when effects of eutrophication are most extreme. Percentile values (5th, 25th, 50th, 75th, and 95th percentiles) were fitted to a cumulative log-normal distribution using the Solver add-in for Microsoft Excel by minimizing the sums of squares of differences between observed cumulative probability (EPA percentiles) and calculated cumulative probability (with mean (μ) and standard deviation (σ) of log transformed values). This solving approach allowed us to construct continuous frequency distributions from the estimate mean (μ_{river} and μ_{lake}) and standard deviation (σ_{river} and σ_{lake}) of the natural log of TN and TP concentrations for rivers and lakes.

A total of 1587 and 10360 stations were used to estimate current TN and TP concentration distributions, respectively, in rivers from all but nutrient ecoregion XIII (Table 1). Data for current TN and TP concentration distributions in lakes came from 980 and 5200 stations, respectively, in all nutrient ecoregions except I and X (Table 2). Linear regression of μ_{lake} vs μ_{river} for all other nutrient ecoregions was used to estimate current μ_{lake} for nutrient ecoregions I and X and μ_{river} for ecoregion XIII. For TP, the equation was $\mu_{river} = (0.69)\ \mu_{lake} - 1.60$ ($P = 0.003$; adjusted $R^2 = 0.61$). The equation we used to estimate mean TN in lakes of ecoregions I and X was $\mu_{river} = (0.71)\ \mu_{lake} - 0.34$ ($P = 0.006$; adjusted $R^2 = 0.53$). We assumed the ratio of μ_{lake}:σ_{lake} = μ_{river}:σ_{river} and used this relationship to estimate σ_{lake} for these ecoregions. Using log-normal frequency distributions defined by μ and σ allowed us to predict the proportional occurrence of water bodies for any value of TN or TP concentration within each ecoregion and to compare to proportional occurrence under reference conditions.

Reference means and standard deviations for TP and TN in rivers were taken from Smith et al. (*17*) who modeled background nutrient concentrations in rivers and corrected for atmospheric N deposition. Smith et al. (*17*) provided 10th, 25th, 50th, and 90th percentiles for all ecoregions. We estimated μ and σ from these data using the Excel Solver method described for EPA current nutrient distributions in the previous section.

To our knowledge, there is no comprehensive study estimating lake reference nutrient conditions across multiple ecoregions. In the absence of this information, we assumed river nutrient concentrations were directly proportional to lake concentrations within ecoregion. The assumption is reasonable because (1) rivers are the water and nutrient source for lakes, and (2) significant relationships were identified between current concentrations in lakes and those in rivers for both total N and P (see results). We estimated reference μ_{lake} for each ecoregion assuming reference μ_{river}/ current μ_{river} = reference μ_{lake}/ current μ_{lake} within each nutrient ecoregion. We also assumed ratio of reference (μ_{river}:σ_{river}) was equal to reference ratio (μ_{lake}:σ_{lake}) to calculate σ_{lake} from known μ_{lake} within each ecoregion. Estimated μ and σ for reference and current distributions were used to estimate percentage change in TN and TP concentrations for each ecoregion. The mean of a log-normal distribution is not recommended as a measure of central tendency (*18*),

TABLE 2. Reference and Current Median TP and TN Concentrations for Lakes in Each Nutrient Ecoregion during Summer Months[a]

ecoregion	ecoregion name	reference TP median mg·L^{-1}	current TP median mg·L^{-1} (*N*)	% of lakes > reference median	reference TN median mg·L^{-1}	current TN median mg·L^{-1} (*N*)	% of lakes > reference median
I	Willamette and Central Valleys	0.007	0.038[b]	91	0.122	0.305[b]	77
II	Western Forested mountains	0.014	0.019 (296)	61	0.147	0.249 (45)	65
III	Xeric west	0.011	0.029 (170)	75	0.039	0.537 (24)	100
IV	Great Plains grass and shrublands	0.026	0.050 (127)	70	0.126	1.489 (2)	99
V	Central cultivated Great Plains	0.023	0.085 (213)	85	0.211	1.416 (2)	100
VI	Corn belt and Northern Great Plains	0.025	0.080 (393)	87	0.159	1.708 (3)	100
VII	Mostly glaciated dairy region	0.010	0.038 (787)	85	0.120	0.800 (35)	100
VIII	Nutrient poor glaciated upper Midwest and Northeast	0.007	0.012 (1238)	76	0.091	0.330 (159)	100
IX	Southeastern temperate forested plains and hills	0.024	0.040 (727)	68	0.052	0.537 (24)	100
X	Texas-Louisiana costal and Mississippi alluvial plains	0.016	0.061[b]	92	0.241	0.725[b]	100
XI	Central and Eastern Forested uplands	0.018	0.019 (267)	53	0.124	0.593 (14)	99
XII	Southern coastal plain	0.005	0.020 (692)	93	0.318	0.743 (545)	94
XIII	Southern Florida coastal plain	0.016	0.035 (10)	81	0.340	1.435 (7)	100
XIV	Eastern coastal plain	0.003	0.017 (280)	98	0.218	0.460 (120)	94

[a] *N* is the number of stations providing data used for estimating current nutrient distributions. The percentage of current lakes whose TP and TN concentration is greater than the reference median value was calculated as one minus the percentile of the estimated reference median in the estimated current distribution. [b] No data available. Current values estimated by regression, see Methods.

therefore median values were used to estimate actual change in TN and TP concentrations for each ecoregion.

The following sections describe estimated annual economic losses from eutrophication. Equations used when cost estimates were possible are summarized in the Supporting Information.

Recreation and Angling Costs. We first estimated increase in lake area closure due to eutrophication. Prolific algal and cyanobacterial blooms are most common during summer months, therefore, we assumed that all lakes classified as hypereutrophic (TP > 100 μg TP·L^{-1}; (*19*)) during this time period would be closed or not used for recreational activities for one (31 days) to three months (92 days). We used hypereutrophic status to indicate lake closures because the probability of a cyanobacterial bloom is 0.90 above 100 μg TP·L^{-1} (*20*). The proportion of reference lakes expected to be hypereutrophic under reference conditions was subtracted from the proportion of lakes currently hypereutrophic. Lake surface area for each nutrient ecoregion was calculated by using ESRI data in ArcGIS (*15*).

Value losses to recreational boating and angling were estimated by calculating loss of trip-related expenses only (e.g., travel, lodging, fuel, food, bait). Per-trip boating-related expenses were assumed to be representative of all ecoregions (*21*), and scaled to 2001 pricing (*22*). The number of day visits per water body type was assigned using the 2001 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (*23*), and the 2002 National Recreational Boating Survey Report (*24*) was used to identify total number of boating days by water body type. Number of fishing and boating days in 2001 was considered the realized use because a portion of lakes were closed to recreational use (i.e., 1−3 months) because of eutrophication. Therefore, current level of use does not represent the full potential of lakes to attract recreational users. Potential use was estimated from realized use during the summer months and scaled by proportional increase in hypereutrophic lake area (PI_l, Supporting Information). Potential number of recreational days was distributed proportionally among each ecoregion according to lake surface area because realized use was surveyed at the national level and hypereutrophic status was calculated by ecoregion. Number of closed days-per-year was calculated for recreational angling and boating separately by ecoregion.

Lake Property Values. Secchi depth strongly correlates with property values (*4*). We used data from 37 lakes in the Mississippi River headwaters region (*9*) to calculate percent gain or loss in property values per 1 m change in Secchi depth. Median TP and the equation from Nürnberg (*29*) were used to estimate reference and current Secchi depths in each ecoregion. Increase in nutrient concentrations above reference conditions over the last 50 years was assumed to be a result of eutrophication (*25*). Therefore, to calculate annual property value loss (PVL), total change in property value was divided by 50 years (Supporting Information).

Total lake frontage (km) in each ecoregion was calculated using ESRI data in ArcGIS (*15*). Of the total lake perimeter in the U.S., we could not determine proportion of private ownership. Therefore we calculated costs for three levels of private ownership around lakes (5, 25 and 50%). Value changes along rivers as a result of water quality changes are unknown and were not calculated.

Loss of Biodiversity. Eutrophication decreases diversity and richness of aquatic macroinvertebrates (*26, 27*), fish (*27*), and aquatic primary producers (*28*). Value of total diversity is difficult to quantify so we used threatened and endangered species recovery plan costs. Richter et al. (*29*) reviewed 135 imperiled freshwater species and found nutrient regimes as a major stressor in 30% of historical and 25% of currently imperiled species cases. We assumed that 25% of all imperiled aquatic species are threatened in part by human-induced

eutrophication and therefore 25% of all recovery costs from U.S. Federal Endangered Species Act plans can be attributed to impacts of human-induced eutrophication, scaled to 2001 values (*22*).

Drinking Water Treatment Costs. Algal and cyanobacterial blooms cause taste and odor problems in drinking water (*8*). Drinking water costs attributable to eutrophication were estimated using the amount of money spent on bottled water that could potentially be attributed to avoidance of taste and odor problems in surface-water-derived tap water (Supporting Information). Data were not available to calculate total costs of drinking water treatment. We assumed that groundwater does not have taste and odor issues related to eutrophication and that most taste and odor problems in surface waters were related to algal metabolites (*30*). A survey of 241 water facilities (*31*) found that 82% of those surveyed had taste and odor issues related to algae.

Survey. We surveyed appropriate agencies from 14 states representing 13 nutrient ecoregions to better understand the perceived degree to which rivers and lakes have become eutrophic and how these problems are addressed. Questions were asked about the number of days water bodies were closed for contact and noncontact use, number of fish kills, human and livestock deaths and sicknesses, money spent on watershed restoration and developing nutrient criteria, money spent on macrophyte removal, and water treatments added by municipalities as a result of eutrophication (Supporting Information). While the survey results were not reliable enough to use in our possible economic cost analyses, we characterized variability in eutrophication management across states.

Results

Current TN and TP means for rivers significantly correlated with those for lakes justifying their use to predict lake TN and TP for the ecoregions where current lake data were missing (Supporting Information). This positive relationship also indicates that using reference river concentrations to predict reference lake concentrations was reasonable.

All nutrient ecoregions had current median TP concentrations greater than medians under reference conditions. Current TP medians exceeded reference medians across ecoregions by 0.002−0.134 $mg \cdot L^{-1}$ (1.1−5.6 with a mean of 3-fold higher) for rivers and 0.002−0.072 $mg \cdot L^{-1}$ for lakes (Tables 1 and 2). In 9 of 14 ecoregions, over 80% of rivers currently exceed median reference values. In ecoregions I, VII, and X, current and reference TP distributions in rivers were similar in shape but shifted to greater concentrations (Figure 2). All other ecoregions had a wide range of TP concentrations. This, along with a greater proportion of rivers with higher concentrations, produced broader cumulative distributions. In general, estimated reference distributions were narrow (i.e., having a narrower range of concentrations) compared to current distributions.

All nutrient ecoregions had current median TN concentrations greater than reference conditions. Current exceeded reference median values across ecoregions by 0.04−3.06 $mg \cdot L^{-1}$ (1.2−13 fold, mean of 5.5 times greater) for rivers and 0.04−1.55 $mg \cdot L^{-1}$ for lakes (Tables 1 and 2). In 12 of 14 ecoregions, over 90% of rivers currently exceed median reference values. Nutrient ecoregions III, IV, VI, IX, and X had current river nutrient distributions with shape similar to that of the reference distributions, but shifted to higher TN concentrations (Figure 3). All other ecoregions had rivers with current TN cumulative distributions broader than reference distributions and contained a greater percentage of rivers with higher TN concentrations.

The closure of lakes to recreational angling and boating because of hypereutrophic conditions resulted in substantial losses of trip-related expenditures. Lakes with hypereutrophic conditions increased in each ecoregion, up to 43% over reference conditions (Figure 4). Numbers of realized fishing and boating day visits were approximately 300 and 443 million per year, respectively. Total recreational use days (i.e., realized + potential use) were estimated at 450−465 and 305−315 million fishing and boating days respectively, of which 7.1−22.2 and 4.8−15.0 million days were potentially lost to eutrophication each year, depending on length of lake closure. Trip-related expenditures per day were estimated at $26.60 for fishing and $37.83 for boating, resulting in annual losses of $189−589 and $182−567 million, respectively. Annual value loss from eutrophication combined for recreational angling and boating could reach $1.16 billion. Ecoregion VI contributed half the total value loss.

All ecoregions had a calculated decrease in Secchi depth from reference values and had lower property values. At the low (5% private), intermediate (25% private), and high (50% private) assumed land availability, eutrophication losses were $14.1, $70.6, and $141.1 billion, respectively. When scaled by 50 years, average rates were $0.3, $1.4, and $2.8 billion in cost per year, respectively (Figure 5).

According to the 2007 U.S. Fish and Wildlife Endangered Species database, 139 fish, 70 mussels, 4 crayfish, 23 amphibians, and one dragonfly had endangered or threatened status (www.fws.gov). Of these, 156 species are covered in 112 recovery plans initiated between 1981 and 2007. We estimate 60 currently listed species are at least partially imperiled due to eutrophication (*29*). The average annual cost of 60 plans was $732,800 suggesting $44 million per year is spent to prevent eutrophication-linked losses of aquatic biodiversity.

Five billion gallons of bottled water were sold during 2000, and 27.7% of people polled purchased bottled water as a result of tap water taste issues (*32*). Bottled water cost $0.89 per gallon in 2003 and surface water sources supply 66% of U.S. domestic water. We estimate $813 million is spent annually on bottled water because of taste and odor problems potentially linked to eutrophication. This estimate is based purely on bottled drinking water costs and does not take into account additional costs related to alternative drinking water treatments such as wells or hauling drinkable water from another area. We were unable to obtain an accurate estimate of the amount of money spent on treating drinking water because not all water treatment facilities separate these costs from treating drinking water for reasons unrelated to eutrophication. Therefore, $813 million is probably an underestimate of the total cost of treating drinking water due to eutrophication.

Based on our informal survey, we found that many water quality parameters are not tracked in a comparable manner across the U.S. Only 3 of 13 states surveyed tracked the number of algal-bloom-related lake closures. Some of the states we surveyed kept more quantitative records than others which resulted in most of our answers coming from only a few states. The small number of states with detailed records made it difficult to extrapolate survey results to the rest of the U.S.

Discussion

Human-induced eutrophication has substantially increased TP and TN concentrations in U.S. rivers and lakes. All nutrient ecoregions now have median TP and TN concentrations greater than reference values for rivers and lakes. Even ecoregions with similar medians have a greater proportion of rivers and lakes with higher nutrient concentrations than reference conditions. Reference criteria for rivers were studied by both Smith et al. (*17*) and Dodds and Oakes (*33*). Both studies produced similar results using different methods.

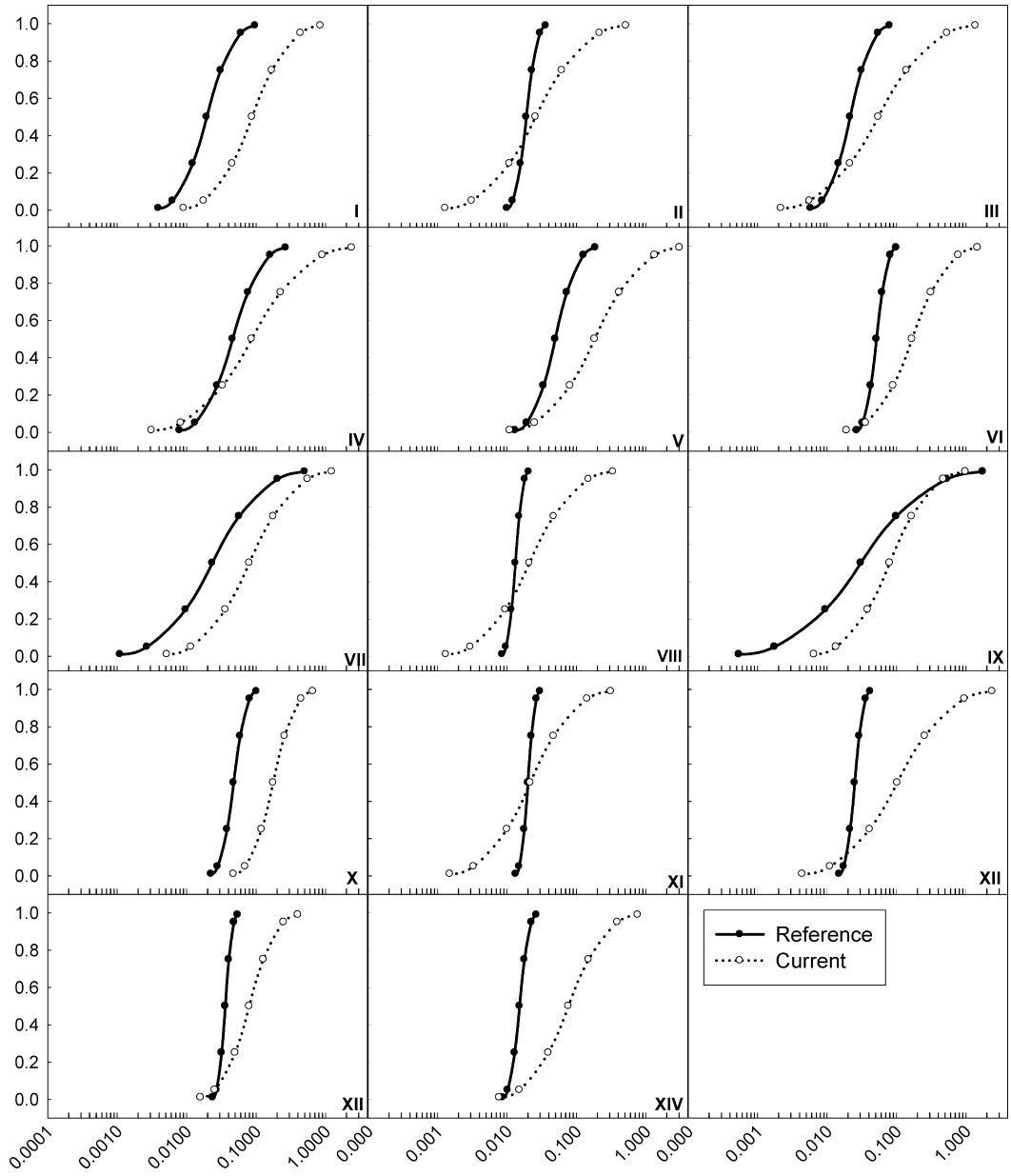

FIGURE 2. Cumulative frequency of total phosphorus for reference (filled circles) and current (open circles) conditions in rivers by nutrient ecoregion (as indicated by roman numerals).

Their similar results support our findings of increased nutrient concentrations across all ecoregions. Without an existing comprehensive study on reference conditions for lakes and reservoirs, we had greater confidence in river distributions than those for lakes derived from previous estimates. Still, rivers in an ecoregion form the major source of nutrients for lakes in that ecoregion. Dodds et al. (*34*) estimated reference values for reservoirs in Ecoregions IV, V, and VI of 23, 27−62, and 15 $\mu g \cdot L^{-1}$ TP, respectively, and our estimates were 23, 13, and 15 $\mu g \cdot L^{-1}$ respectively, showing moderate agreement.

We expect that the degree of eutrophication documented in the U.S. represents a global phenomenon. A substantial portion of nutrients from human-induced eutrophication are ultimately derived from fertilizers, and fertilizer use patterns can be used to indicate global trends of eutrophication. Bumb and Baanante (*35*) predict continued increase in fertilizer use over the next 20 years, with greater increases in developing countries.

Over $1 billion in recreation expenditures were estimated lost annually, yet our methods could not account for all recreation losses and required several assumptions. We assumed that users do not substitute a nearby lake with lower nutrient loadings for a "closed" hypereutrophic lake. The scale of our data required that we assume water body use was evenly distributed throughout the year, when in fact most use and most algal blooms occur during the summer months (*24*). Further, we had to assume that values for angling and boating daily trip expenditures were representative across all ecoregions, and could be extrapolated to the entire U.S. We were unable to account for losses from the Great Lakes region (54 million fishing/boating days) or rivers (260 million fishing/boating days), because of difficulty estimating frequency of algal blooms in these water bodies. Our calculations did not include equipment purchases that would decline in areas where recreational opportunities decrease because of eutrophication.

Clear water is aesthetically pleasing and lakefront property has significantly greater value with increased clarity (*9, 36*); a decrease in property value of 15.6% occurs with every 1-m loss in Secchi depth (*9*). Thus, we predict substantial losses in the value of U.S. lakefront property. These losses are probably underestimates because nationally reported me-

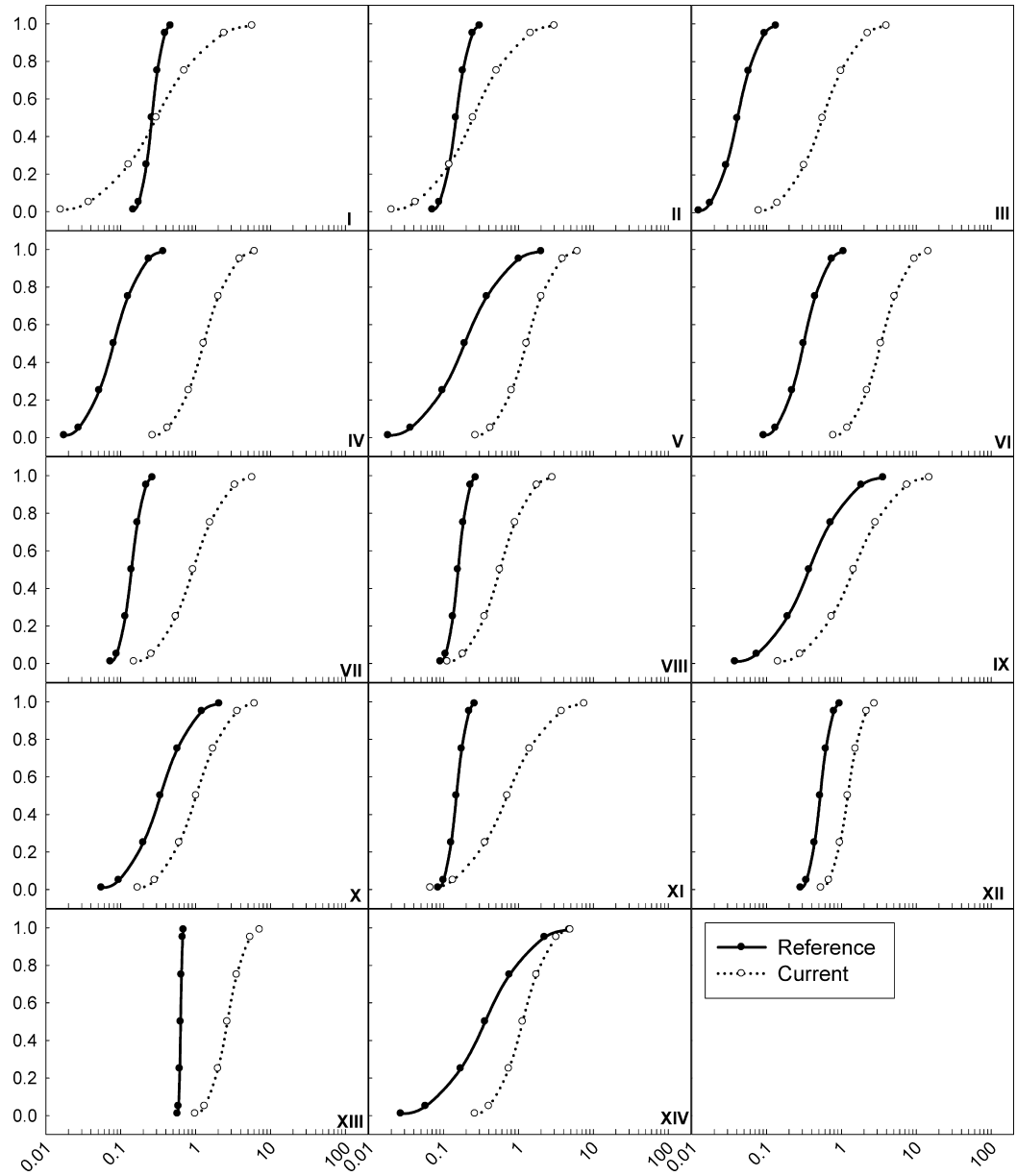

FIGURE 3. Cumulative frequency of total nitrogen for reference (filled circles) and current (open circles) conditions in lakes by nutrient ecoregion (as indicated by roman numerals).

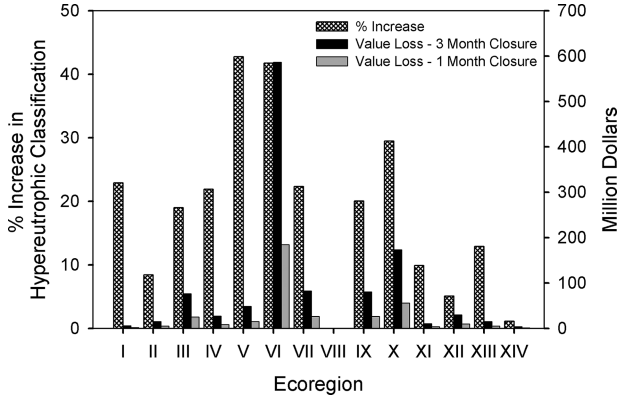

FIGURE 4. Percent increase in hypereutrophic status for lakes, reservoirs, and ponds and the combined annual fishing and boating value loss with a 3 or 1 month summer closure period for each ecoregion.

dians for each ecoregion might not accurately represent waterfront properties. Waterfront properties in the United Kingdom are worth 10−40% more than equivalent nonwaterfront properties (*36*). Also, water clarity affects property values up to 1.2 km from the shoreline (*37*) but we only considered adjacent property.

Altered nutrient regimes are one of many stressors leading to species endangerment, but we consider $44 million/year a highly conservative estimate for loss of biodiversity because it underestimates the true value of altering natural species assemblages (i.e., local extirpation of diversity does not lead to federal listing). Human-perceived aesthetic values of biodiversity, ecosystem services, and local losses of fairly common species that would not be recognized as threatened or endangered (e.g., those listed as threatened or endangered at the state level) were not included in our estimate. We also expect some species will eventually be listed as endangered as a result of current activities. For instance, many unionid mussels continue to decline even though management actions have not substantially changed over the last few decades (*38*).

Eutrophic systems have more taste and odor problems from eutrophication. Arruda and Fromm (*39*) reported a

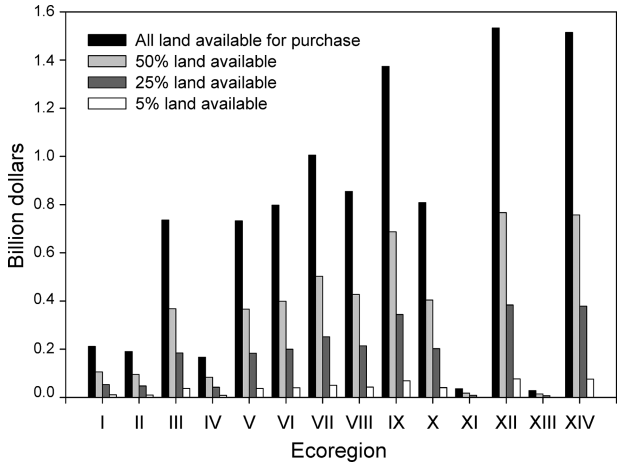

FIGURE 5. Property value lost per year by ecoregion due to changes in Secchi depth from increased phosphorus loadings into lakes. Value lost was calculated assuming all land around lakes, reservoirs and ponds is available, 50% is available, 25% is available, and 5% is available for private land ownership.

strong positive linear relationship between odor rank, trophic state, and mean chlorophyll concentration in Kansas lakes. Our estimate of drinking water costs does not account for additional drinking water treatment costs related to taste and odor problems. In 1976 the US EPA set aside $7.5 million, and in 2004 set aside $102 million, for the Public Water System Supervision program (*40*). The Drinking Water Infrastructure Needs Survey concluded that $150.9 billion will need to be invested in drinking water systems to provide safe treatment, storage, and distribution (*40*). Eutrophication is a major component of taste and odor problems that this funding partially addresses through drinking water supervision and infrastructure development.

Macrophyte abundance increases with greater nutrients until conditions become hypereutrophic and algal blooms dominate the water column (*12*). Management of overabundant macrophytes is necessary because they restrict navigation, recreation, and lake processing (*41*), and detract from aesthetic appeal. Treatment costs for a variety of aquatic weeds range from $1,247 to $19,227 per hectare for mechanical harvest and $246 to $1,190 per hectare for chemical treatment (*41*−*43*). If just 1% of major surface waters (approximately 162,384 ha; (*15*)) in the U.S. needed macrophyte control because of eutrophication-induced excessive growth, it would cost an average of $1.2 billion annually for mechanical harvest or $105 million dollars annually for herbicide treatment.

Eutrophication can be beneficial to fisheries as fish biomass increases with primary production (*44*). However, under hypereutrophic conditions, more valuable fishes are often replaced by undesirable "rough" fishes (*44*). Commercial aquaculture (*45*) was assumed to not be negatively impacted by eutrophication. The majority of freshwater commercial fishing occurs on the Laurentian Great Lakes (*46*). Despite problems with eutrophication (*47*), the overall economic value of commercial fishing in the Great Lakes has remained relatively stable, independent of nutrient fluctuations.

Algal blooms as a result of eutrophication have caused harmful health effects to humans and livestock (*48*). Mass mortalities of wildlife have been attributed to cyanobacterial blooms (*49*). For humans, algal blooms cause sicknesses and rarely result in death (*50*). We did not include human health costs because they appear to be minor compared to other factors we investigated. Still, people might be more likely to spend considerable amounts to avoid toxic blooms.

Trends in nutrient concentrations across the continental U.S. have been well documented (*51*−*53*), but not compared to reference conditions. We estimated current nutrient concentrations related to reference conditions, and tied these to economic costs where possible. We provide broad annual estimates of economic losses in recreational water usage ($1 billion), waterfront property ($0.3−$2.8 billion), recovery of threatened and endangered species ($44 million), and drinking water ($813 million), resulting from human-induced eutrophication. These potential losses total over $2.2 billion annually and our estimates are probably conservative.

Our study shows some areas where research should be focused and where better records should be kept. Accounting for eutrophication-related drinking water costs, as well as macrophyte-removal costs resulting from increased nutrient loading, is needed. Accounts of fish kills associated with eutrophication and the negative impact of enriched waters on biodiversity as a whole are lacking. Agricultural production and fertilizer use will likely increase (*54*), resulting in intensified eutrophication-related losses. Trajectories of reactive nitrogen loadings worldwide show increases in freshwater transport of 21 million tons in preindustrial times to 40 million tons per year today, while riverine transport of dissolved inorganic nitrogen has increased from 2−3 million to 15 million tons (*55*). Our estimates could help society quantify potential costs associated with increased nutrients entering freshwater ecosystems, but more importantly highlight the value of clean water to society.

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Supporting Information Available

(1) Table of all equations used to calculate values, (2) a table with costs for macrophyte removal, (3) a table summarizing the results of surveying state agency personnel about eutrophication and costs, and (4) figures documenting the positive relationships for current total phosphorus in rivers and lakes and current total nitrogen in rivers and lakes. This information is available free of charge via the Internet at http://pubs.acs.org.

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