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Comment: Cultural eutrophication of natural lakes in the United States is real and widespread

In a recent paper titled "The extent that natural lakes in the United States of America have been changed by cultural eutrophication" Bachmann et al. (2013) attempted to test the idea that 75% of all natural lakes in the United States have been influenced by cultural eutrophication. Based on their analyses, they offered the broad conclusion that "The assumption of widespread cultural eutrophication for setting numeric nutrient criteria in lakes is not supported," and they also asserted that "…in the United States of America the extent that natural lakes have been changed by cultural eutrophication does not seem to be large." These extraordinary claims contradict results from more than half a century of limnological research and lake management efforts, and, as we demonstrate below, they are the result of the authors' use of a biased data set that does not broadly reflect natural lakes in the United States.

Bachmann et al. (2013) based their claims on paleolimnological estimates of pre- and post-European settlement total nitrogen (TN) and total phosphorus (TP) concentrations in 240 lakes selected from the U.S. Environmental Protection Agency (USEPA)'s National Lakes Assessment (NLA; cf. USEPA 2009, 2010). However, we demonstrate in this Comment that the conclusions of Bachmann et al. (2013) can only apply to lakes that have exhibited minimal anthropogenic influences in a relatively restricted portion of the United States because (1) the USEPA NLA database analyzed by Bachmann et al. (2013) only includes relatively short and undated cores and (2) their methodology precludes consideration both of natural lakes in disturbed watersheds and of natural lakes with high sedimentation rates. We also review evidence for pervasive increases in nutrients in natural U.S. lakes linked to human activities in their drainage basins.

The cultural eutrophication of freshwater ecosystems worldwide has been recognized as a serious environmental issue for more than half a century (Likens 1972; Smol 2008), and it remains a major water quality problem in both developed and developing regions. Humans have dramatically increased nitrogen (N) and phosphorus (P) exports from the landscape into aquatic environments across the globe (Caraco and Cole 1999; Fowler et al. 2013). In the United States alone, the U.S. Department of Agriculture (2010) has reported that 10 billion kg of N and 3.6 billion kg of phosphate were applied as fertilizer in 2010, and significant quantities of this agricultural N and P ultimately find their way into surface waters (McDowell et al. 2004). In addition, atmospheric deposition of N and P are substantial across the continental United States, with direct deposition on aquatic systems and about one third of the deposited N on drainage basins subsequently being transported to aquatic systems (Howarth et al. 2002). Not surprisingly, there has been a three- to eightfold increase in nitrate fluxes from watersheds in the northeastern United States since the early 1900s, with over 60% of this change being derived from atmospheric sources (Jaworski et al. 1997) and with measurable effects on both freshwater and marine eutrophication (Paerl 2009). Similarly, there have been large increases in the amount of P entering freshwaters worldwide (Bennett et al. 2001), and aeolian P deposition has increased fivefold in alpine regions of the western United States (Neff et al. 2008). Mitigating nutrient losses from anthropogenic nonpoint sources (NPSs) is, therefore, of particular importance for improving the water quality of many freshwater lakes (Nielsen et al. 2012), both in the United States and worldwide. In spite of the published literature, Bachmann et al.'s (2013) analysis fails to document the eutrophying effects of accelerating nutrient loading on nutrient-sensitive waters.

Bachmann et al. (2013) criticized the approach taken by the USEPA (2000), which suggests that in areas where no reference lakes or reservoirs are available, roughly 75% of lakes and reservoirs can be considered to be affected by cultural eutrophication. The USEPA also suggests that in areas where reference lakes are available, the upper 25% of these reference lakes could still potentially be affected by cultural eutrophication, but it also recommends using paleolimnological evidence as well as the best professional judgment to assess target nutrient levels that are indicative of undisturbed lakes. Even though the USEPA suggests applying the 75% method *only* in areas containing no reference-quality systems, Bachmann et al. (2013) analyzed cores from the USEPA NLA (USEPA 2009, 2010), tested these core data against the assumption of 75% eutrophication, and then concluded their paper with the remarkable assertion that "The assumption of widespread cultural eutrophication for setting numeric nutrient criteria in lakes is not supported."

How can we reconcile the sweeping statements of Bachmann et al. (2013) with the large historical literature that indicates widespread cultural eutrophication in natural lakes? There are three possible explanations: (1) natural lakes are not hydrologically connected to and do not receive nutrient flows from their watersheds, so that watershed land use has little influence on lake water quality, and also that, despite extensive anthropogenic disturbance in U.S. watersheds, prior evidence documenting the extensive cultural eutrophication of natural lakes by NPSs is wrong; (2) there are serious technical limitations with the methods and data used by Bachmann et al. (2013); and/or (3) the lakes analyzed by Bachmann et al. are not representative of the entire body of natural lakes in the conterminous United States. We critically examine each of these three possibilities below.

Explanation 1. There is a lack of linkage between natural U.S. lakes and their watersheds, such that watershed land use has little influence on lake water quality—This explanation is not supported by any published research conducted to date on lakes and their watersheds. Lakes are strongly linked to their watersheds through the transport of materials carried by surface runoff or subsurface groundwater flows (Soranno et al. 1996; Winter et al. 1998), and NPS nutrients that have entered fluvial systems from the surrounding landscape will subsequently be transported in the form of suspended and dissolved nutrients into the natural lakes that are physically connected to these flowing waters (Williamson et al. 2008). The vast majority of natural lakes in the continental United States are fed by surface- or groundwater flows, and their hydrology and their nutrient loading inevitably are very strongly influenced by these inputs. NPS inputs of N and P have caused the eutrophication of natural lakes and reservoirs throughout the United States, and they are also the dominant source of these nutrient elements to most reaches of U.S. rivers (Carpenter et al. 1998), as well as estuarine and coastal water bodies (Paerl 2009). Whereas Bachmann et al. (2013) specifically claim that their results do not apply to rivers, we take a more holistic approach; and we argue that, if watersheds were disturbed, nutrients would enter streams and rivers, these nutrients would be transported to downstream lakes, and the receiving lakes subsequently would be eutrophied.

Lakes, indeed, are typically connected to rivers, and there is abundant and widespread evidence for increases in riverine nutrients from NPSs in most watersheds in the United States. Multiple approaches have been used during the past two decades to assess changes in the nutrient content of U.S. river waters relative to non-anthropogenically affected reference conditions. For example, long-term measurements have contributed significantly to our understanding of the ongoing influences of human activities on nutrient concentrations in rivers (Jaworski et al. 1997). Smith et al. (2003) employed modeling approaches to assess reference concentrations of nutrients in the rivers and streams of the United States and then compared those estimates to current measured values. Dodds and Oakes (2004) used statistical methods to extrapolate across watersheds and to factor out human land uses to estimate reference nutrients by nutrient ecoregions. Their approach demonstrated strong influences of cropland on the nutrient content in rivers across most ecoregions in the United States. Triplett et al. (2009) and Engstrom et al. (2009) used a whole-basin, mass-balance approach to reconstruct P concentrations and loading from the sediment records of natural riverine impoundments on the upper Mississippi River.

All of the independent approaches above revealed substantial increases in the nutrient content of rivers and streams of the United States relative to predisturbance, reference conditions. In fact, in all 14 nutrient ecoregions of the United States, current median TN and TP values for rivers and lakes exceeded reference median values; and, in 12 of 14 ecoregions, more than 90% of rivers currently exceed reference median values (Dodds et al. 2009). Anthropogenic alteration of the landscape is, thus, widespread. Six major river basins in the United States have more than 40% of their total area in crop production; and agriculture, combined with urbanization and other human activities, is heavily influencing large expanses of the United States (Allan 2004). Moreover, the United States is not alone in this human-induced problem. For example, Heathwaite et al. (1996) performed a global analysis of nutrient trends and concluded that N and P concentrations had increased dramatically ($>$ 20 times background concentrations) in many watersheds in temperate North America and Europe, with the causes ranging from urbanization to changes in agricultural practices. Similarly, using export coefficient modeling, Johnes (1996) and Johnes et al. (1996) demonstrated substantial nutrient loading contributions from diffuse anthropogenic sources to the surface waters of watersheds ranging over the uplands and lowlands of England and Wales.

Humans have a long history of altering habitats and causing eutrophication on a watershed scale (Dodds and Whiles 2010), and linkages between land use and surface-water quality have been broadly demonstrated both in the United States and worldwide (Bennett et al. 2001; Dodds et al. 2006). For example, the paleolimnological studies of Hutchinson et al. (1970) revealed strong effects of Roman watershed-based activities $>$ 2000 yr ago on the trophic state of Lake Ianula, Italy. Classic early limnological studies by Thienemann and by Naumann clearly demonstrated that differences in the trophic state of lakes in Europe and Scandinavia were closely associated with the characteristics of their watersheds (Naumann 1929). Similarly, Deevey (1940) demonstrated strong regional differences in the trophic state of Connecticut (United States) lakes, and his data were critical to the development of linkages between epilimnetic TP concentrations and the development of phytoplankton biomass in lakes. Since the 1970s, the linkages between anthropogenic nutrients and lake trophic state have been integrated into a strong and objective framework that has been globally adopted for the management of eutrophication by controlling point-source and NPS nutrients to lakes and other surface waters (Cooke et al. 1993).

The conclusions of Bachmann et al. (2013), therefore, are logically inconsistent with the vast body of evidence on NPS eutrophication that has accumulated globally for more than half a century. Moreover, case studies from Europe and North America provide clear evidence for strong links between NPS nutrient inputs and the trophic state of lakes. For example, Anderson (1997) used careful paleolimnological analyses of sediment cores to evaluate historical trends in epilimnetic TP in six rural lakes in Northern Ireland. Using ^{210}Pb dating methods, the cores from these lakes were found to cover periods ranging from ca. 100 yr to $>$ 150 yr BP. None of these six lakes had either sewage or industrial point-source inputs flowing into them, and each had primarily agricultural catchments. Nonetheless, diatom stratigraphies indicated significant temporal changes in algal community composition in all of these lakes, regardless of their current nutrient status. Most importantly, increasing trends in diatom-inferred TP were observed in several of these lakes between 1850 and 1900, and TP concentrations increased by three- to fivefold in all six of these lakes after 1950. Water quality in a representative set of natural, rural lakes in Northern Ireland

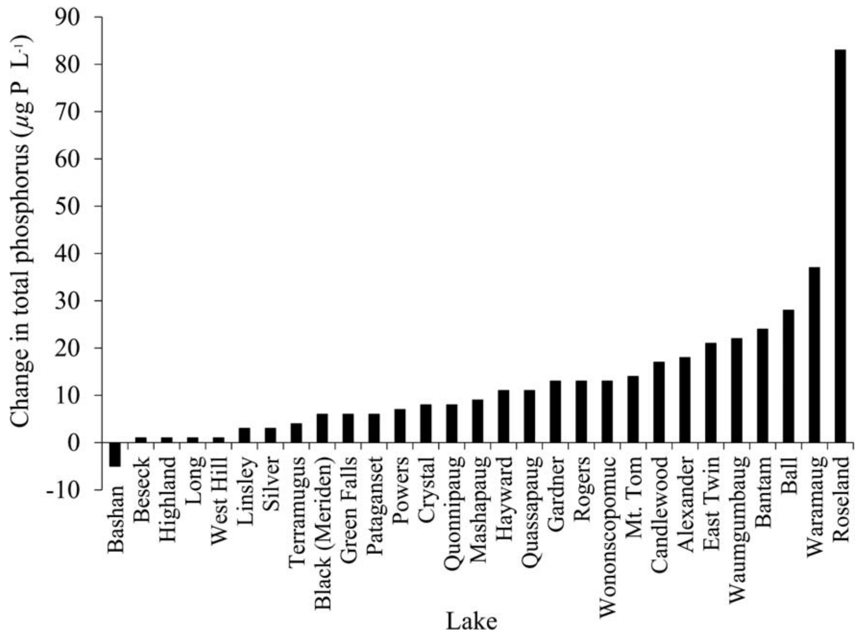

Fig. 1. Changes in lake trophic state in 29 lakes in Connecticut (United States) between the 1930s and the 1990s (data from table 2 in Siver et al. 1996).

has, thus, been demonstrably affected by NPS nutrient pollution over the last two centuries.

Across a much larger geographic scale, Bennion and Simpson (2011) studied 106 European lakes, covering a range of typologies and ecoregions. Paleolimnological reconstructions showed that more than half of these study lakes exhibited significant nutrient-driven changes in diatom assemblages over the past 150 yr, with eutrophication affecting all lake types. Similarly, Nielsen et al. (2012) examined a comprehensive data set comprising land-use data, nutrient-source information, and in-lake water quality for 414 Danish lakes. After removing 210 lakes that receive point-source nutrient inputs, Nielsen et al. (2012) still found strong relationships between in-lake concentrations of TN and TP and the proportion of agricultural land use in the watershed.

An additional and unambiguous U.S. example of large-scale eutrophication stemming from human-associated, NPS pollution comes from Connecticut (United States) lakes that were originally part of the survey of Deevey (1940), none of which were significantly affected by point-source P loading. Using data from real-time lake-water samples that were carefully collected and analyzed using comparable methods, Siver et al. (1996) quantified changes in the conditions of 42 lakes surveyed during three different periods: the late 1930s (Deevey et al. 1940); the mid- to late 1970s; and the early 1990s. All 42 lakes were natural in origin except Candlewood, which was originally dammed in the early 1930s as a pump-storage facility. Siver et al. (1996) concluded that the average concentrations of TP for these lakes included in each successive limnological survey increased from 12 μg P L^{-1} (n = 29) in the 1930s, to 15 μg P L^{-1} (n = 41) in the 1970s, to 24 μg P L^{-1} in the early 1990s (n = 42). Lakes with the smallest increase in TP either were primarily located in watersheds that remained in mostly forested vegetation cover, or were located in watersheds that experienced minimal change in land use.

We have plotted in Fig. 1 the data that were available for the lakes that were first sampled by Deevey in the 1930s and then were resampled in the 1990s (*see* table 2 in Siver et al. 1996). Increases in epilimnetic TP were observed for 28 of 29 of these lakes, and an analysis of these data using a nonparametric Sign Test (Snedecor and Cochran 1978) revealed that there is only a chance of less than one in a million ($p < 0.000001$) that the observed time trends of increasing lake trophic state in mostly natural and largely rural Connecticut lakes could be due to chance alone. We thus conclude that water quality and eutrophication in a representative set of lakes located throughout the entire state of Connecticut have been significantly affected over 60 yr.

We also note that a synoptic paleolimnological survey of 55 natural Minnesota (United States) lakes spanning three major ecoregions and a wide range of catchment land-use conditions found that 30% of lakes in human-disturbed urban and agricultural settings showed a statistically significant increase in diatom-inferred TP (DI-TP, μg P L^{-1}) from presettlement times (ca. 1800) to the present (Ramstack et al. 2004). In contrast, the subset of 20 Minnesota lakes in relatively *undisturbed* boreal landscapes showed no significant change in DI-TP. Although lake selection by Ramstack et al. (2004) was not probabilistic, the lakes from each ecoregion were explicitly chosen to span a large gradient in present-day trophic condition and, therefore, can be considered to be representative of the larger population of natural Minnesota lakes. In stark contrast, the 240 NLA lakes reported by Bachmann et al. (2013) were effectively screened to *remove* most lakes with highly disturbed watersheds, thereby creating strong selection bias as is detailed in discussions of Explanations 3 and 4 below.

Table 1. Depth of Euro-American settlement horizon in sediment cores from 112 Minnesota lakes, summarized by ecoregion and local land use (data from Engstrom et al. 2007; Anderson et al. 2013). Minnesota ecoregions key: NCHF: North Central Hardwood Forest; WCBP: Western Corn-Belt Plains; NGP: Northern Glaciated Plains; NLF: Northern Lakes and Forest.

	Undisturbed, NCHF	Urban, NCHF	Agricultural, NCHF + WCBP + NGP	Undisturbed, NLF
n	12	20	26	54
Minimum depth (cm)	24	40	26	11
Maximum depth (cm)	124	182	190	80
Median depth (cm)	56	68	70	30
10th percentile	29	49	34	14
90th percentile	99	120	129	56

There is overwhelming empirical evidence documenting significant anthropogenic nutrient enrichment of river and groundwaters, even in many, if not most, small watersheds of the United States. There is also strong and demonstrable connectivity among rivers, groundwaters, and almost all natural lakes that exist within the conterminous United States and the world. Moreover, more than a half-century of carefully designed and judiciously interpreted limnological studies such as those summarized above have incontrovertibly documented geographically extensive and highly significant cultural eutrophication of lakes of natural origin by nonpoint sources. We thus reject Explanation 1, which we can find as the only logical argument to support the contention of Bachmann et al. (2013) that there is no evidence for widespread eutrophication of natural lakes. Put simply, it is not scientifically correct to consider lakes as being separated from their watersheds. Explanations 2 and 3 further address why these incorrect conclusions were reached by Bachmann et al. (2013).

Explanation 2. Technical limitations with NLA methods and data selection are responsible for the erroneous conclusions of Bachmann et al. (2013)—Paleolimnological methods are typically used to reconstruct historical trends in lake water quality because there are few records that exceed half a century of chemical and physical data for lakes (Siver et al. 1996; Smol 2008). Many paleolimnological studies have been conducted over the last several decades and show overwhelmingly that 20th-century, anthropogenic eutrophication is pervasive in the United States and worldwide. The most common approach to trophic reconstruction involves the application of diatom-inference models (numerical transfer functions), such as those employed in the USEPA (2009) NLA study and then later repurposed by Bachmann et al. (2013).

More specifically, the NLA study employed a "top–bottom" approach, in which the core-top sample was used to represent modern conditions and the core-bottom sample, limnological conditions prior to widespread human disturbance—meaning prior to Euro-American settlement of the United States. Because the NLA study involved such a large number of lakes, cores were not objectively dated using ^{210}Pb, which is normally done for paleolimnological studies of this type. Instead, the USEPA investigators used a set of decision rules to classify each core as to whether or not it penetrated to presettlement sediments. Criteria included comparison with dated cores from previous studies and whether or not there was substantial watershed disturbance that would otherwise cause high sedimentation rates (USEPA 2009).

The USEPA's documentation specifically states that it cannot be certain that the bottom sections of all cores indeed represent pre-European conditions: *"Unfortunately, EPA was unable to date the sections of the core to confirm their age. Instead, NLA analysts used independent techniques, their own expertise, and the knowledge of regional experts to determine whether the cores were sufficiently deep for NLA purposes. The Agency acknowledges that this approach is a less reliable means of estimating the age of the cores"* (USEPA 2009). In at least three of the nutrient ecoregions examined by Bachmann et al. (2013), no cores were accepted from currently disturbed watersheds. Specific natural lake types (e.g., oxbows) were excluded a priori, and no lake smaller than 0.4 km^2 was included. Of the 293 high-confidence cores in the NLA study, 241 were classified in the USEPA data set as likely to have core-bottom samples of presettlement age. Bachmann et al. (2013) analyzed 240 of these. The remaining 52 high-confidence cores were taken from lakes specifically chosen to serve as reference lakes in the NLA's probabilistic design.

The technical failings of the analysis by Bachmann et al. (2013) thus become clear: lakes with substantially disturbed watersheds were largely *excluded* from their subset of 240 NLA lakes, and the core-bottom samples of those lakes that *were* included are in fact of uncertain age. Furthermore, some of the cores that were originally categorized by the NLA researchers as reaching presettlement times may in fact have been too short, such that their bottom samples instead represent anthropogenically altered lake conditions. A top–bottom analysis of such cores could lead to the conclusion that the lakes were naturally more productive during presettlement times than they in fact were, thus giving misleading information about time trends in eutrophication.

Such a bias is especially likely for lakes in urban or agricultural settings, where sediment accumulation rates are typically quite high. This point is readily illustrated by a data set of ^{210}Pb-dated cores from 112 Minnesota lakes (Engstrom et al. 2007; Anderson et al. 2013). These cores were all collected by piston corer, with some cores exceeding 2 m in length and all penetrating well

Table 2. Percentages of land use and land cover in basin for 241 high-confidence, nonreference lake cores from the USEPA National Lakes Assessment (http://water.epa.gov/type/lakes/NLA_data.cfm, data downloaded June 2013). The sum of the six undisturbed land-use categories (Water, Barren, Forested, Shrubland, Grassland, and Wetland) and the sum of the three human-disturbed land-use categories (Cropland, Pasture, and Developed lands) are also presented.

Land use or land cover	Mean %	Median %	Minimum %	Maximum %	Lower quartile	Upper quartile
Water	12	9	0	100	5	16
Barren	2	0	0	80	0	0
Forested	46	52	0	94	18	72
Shrubland	5	1	0	85	0	5
Grassland	6	1	0	96	0	3
Wetland	8	4	0	52	1	12
Sum undisturbed	80	93	18	100	63	99
Cropland	8	0	0	70	0	8
Pasture	5	0	0	42	0	8
Developed	7	4	0	72	1	8
Sum disturbed	20	8	0	82	1	37

into presettlement times (ca. 1860). In comparison, the USEPA's NLA cores were collected by gravity corer and were, on average, comparatively short (39 ± 9 cm) for the 240 lakes used by Bachmann et al. (2013).

The Minnesota lakes themselves span several ecoregions and local land-use categories, similar to the larger NLA data set. Almost no lakes in the urban (Minneapolis–St. Paul) or agricultural land-use categories have a settlement horizon shallower than 30 cm (10th percentile), and their median settlement horizons are close to 70 cm (Table 1). Among Minnesota ecoregions, only the Northern Lakes and Forests (NLF) have a substantial number of cores (50%) with a depth of settlement of less than 30 cm. These latter lakes are located in the northeastern part of Minnesota, with minimal or no human disturbance in their watersheds—almost all are in federal wilderness areas, national parks, and national forests.

Given the methods used in the USEPA dating assessments and the qualifications the USEPA provided on such assessments, it is clear the subset of lakes used by Bachmann et al. (2013) were highly filtered to *exclude* disturbed sites located in agricultural or urban settings. This strong selection bias is evident in the disproportionate number of lakes classified as oligotrophic and mesotrophic in Bachmann et al. (2013, their fig. 2). Bachmann et al. (2013) thus chose to examine a filtered subset of a probabilistic study (the NLA), thereby nullifying the probabilistic sampling design and precluding any useful conclusions regarding trophic change in the larger population of natural U.S. lakes. The nature of this data "filtering" is outlined in Explanation 3 below.

Because of our major concerns about technological limitations in the original NLA program, and because Bachmann et al. (2013) either did not recognize or ignored those limitations, we accept Explanation 2 as a highly likely and objectively supportable reason for strong inconsistencies between the conclusions of Bachmann et al. (2013) and more than half a century of eutrophication research.

Explanation 3. The NLA lakes are not representative of natural U.S. lakes as a whole—As noted above, Bachmann et al. (2013) opted to use a filtered data set to arrive at their final subset of 240 lakes; but they, nonetheless, did not critically consider whether those lakes were truly represen-

Table 3. Mean values of total phosphorus concentrations (TP, μg L^{-1}) for 12 U.S. nutrient ecoregions obtained from 241 high-confidence (HC), nonreference lake cores in the USEPA National Lakes Assessment (http://water.epa.gov/type/lakes/NLA_data.cfm, data downloaded June 2013). Reference TP concentrations for streams in each region are from Dodds et al. (2009). The total estimated numbers of natural lakes in each respective ecoregion are from Herlihy et al. (2013). ND = not determined.

Ecoregion	HC cores, mean TP	n	All NLA lakes, mean TP	Natural lakes, mean TP	Reference streams, mean TP	Total no. of natural lakes
I	32	1	170	68	19	ND
II	13	57	70	67	19	2738
III	151	2	42	46	21	72
IV	366	8	99	120	59	126
V	3	1	9	108	22	1926
VI	30	3	209	132	23	4325
VII	25	64	91	94	23	7628
VIII	15	85	154	193	13	10,063
IX	ND	0	107	146	30	567
X	112	1	8	12	48	574
XII	50	6	74	162	25	592
XIV	13	13	130	70	15	493

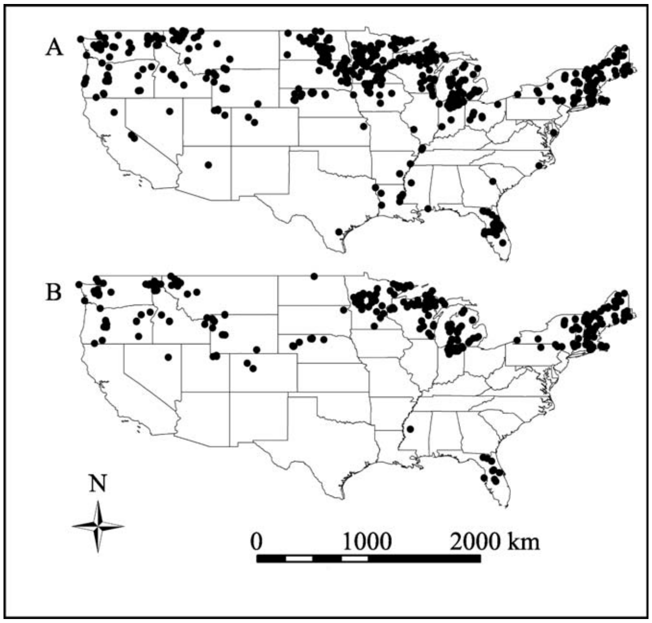

Fig. 2. (A) Locations of all natural lakes sampled in the NLA assessment. (B) Locations of the 293 natural lakes in the NLA assessment with high-confidence (HC) cores.

tative of the larger data set from which they were extracted. The lakes associated with those "high-confidence" cores are only modestly influenced by human activities (*see* Table 2 and associated legend), and most of them are located in areas in which the dominant land cover category is undisturbed land. Cropland, which is the land use expected to have the greatest influence on the magnitude of nonpoint nutrient exports, only covers a median of $< 1\%$ of watershed area. It is, thus, expected that such undisturbed lakes should show little change in nutrient concentration over time; and our independent analyses, indeed, verify a lack of significant increases (Table 3).

The highly selective nature of the data set used by Bachmann et al. (2013) is also clearly evident both in Fig. 2 and in Tables 3 and 4. Within the 14 nutrient ecoregions in the United States established by the USEPA, only four ecoregions had more than 10 natural lakes sampled, three ecoregions had one lake sampled, and three ecoregions had no lakes sampled. Considering the ecoregions that had 5 or fewer lakes sampled, a total of only 8 lakes were sampled in six ecoregions to represent more than 7000 natural lakes! These 8 lakes therefore are not necessarily "representative" of the lakes in this large geographical area. Whereas the USEPA attempted to find representative lakes for their NLA, this substantially smaller subset of lakes is unlikely to adequately reflect the heterogeneity of soils, hydrology, and lake types that naturally exist within these six ecoregions. Applying the results of Bachmann et al. (2013) to all natural lakes in the United States would require an analysis of how well the lakes with cores represented conditions within every ecoregion of the entire United States. This has not been done, however; and this failure precludes making broad assertions about U.S. lakes, as Bachmann et al. (2013) have done.

Further examination of the data reveals that lake types unsuitable for paleolimnology (e.g., riparian or oxbow lakes located along major rivers) also were not included in Bachmann's analysis. In addition, no lakes smaller than 0.1 km^2 were included, even though such small lakes are the most numerous in the United States (*see* table 2 in McDonald et al. 2012). We therefore conclude that the use of data from a selected subset of lakes having highly specific sediment attributes and extremely sparse geographic representation further precludes acceptance of any sweeping generalizations regarding NPS nutrient inputs to all natural lakes in the United States, such as were made by Bachmann et al. (2013).

Because the subset of lakes analyzed by Bachmann et al. (2013) is small, highly filtered, and unrepresentative of the entire set of natural lakes in the United States, we accept Explanation 3 as a highly likely and objectively supportable reason for strong inconsistencies between the conclusions of Bachmann et al. (2013) and more than half a century of eutrophication research.

Table 4. Mean values of total nitrogen concentrations (TN, $\mu g\ L^{-1}$) for 12 U.S. nutrient ecoregions obtained from 241 high-confidence (HC), nonreference lake cores in the USEPA National Lakes Assessment (http://water.epa.gov/type/lakes/NLA_data.cfm, data downloaded June 2013). Reference TN concentrations for streams in each region are from Dodds et al. (2009). ND = not determined.

Ecoregion	HC cores, mean TN	*n*	All NLA lakes, mean TN	Natural lakes, mean TN	Reference streams, mean TN
I	432	1	4309	907	261
II	283	57	939	1245	147
III	1290	2	730	975	41
IV	4673	8	917	1515	81
V	841	1	223	1374	566
VI	928	3	1857	1672	215
VII	716	64	884	1276	139
VIII	473	85	1395	2653	156
IX	ND	0	1275	1677	370
X	1501	1	780	437	339
XII	1112	6	906	1421	631
XIV	378	13	1927	950	359

The suggestion of Bachmann et al. (2013) that there is not widespread eutrophication of natural U.S. lakes due to NPS pollution is, therefore, not substantiated. Bachmann et al. (2013) have simply shown that a highly filtered subset of natural U.S. lakes (those occurring in watersheds with low levels of human disturbance, and those having both low sediment deposition rates and low nutrient concentrations) have not exhibited long-term changes in trophic state. A lack of temporal change in water quality in undisturbed lakes is a trivial and foregone conclusion, and it does not warrant the broad generalizations that were made by Bachmann et al. (2013). Furthermore, their analysis has little bearing on USEPA-suggested procedures for determining regional nutrient criteria for natural lakes and reservoirs.

A broad future survey of lakes in the United States could be used to perform the analysis attempted by Bachmann et al. (2013), but these efforts would require substantially deeper lake cores, explicit core dating, and a more careful selection of lakes that are truly representative of the conditions that exist within each ecoregion. The documented environmental history of each watershed would also assist in evaluating the nature, extent, and degree of human influence on the watershed. For example, although many of the lakes with high-confidence cores from the northeastern United States may have been under cultivation in 1850, their watersheds have been reverting to forest since that time; thus, these lakes would be expected to be becoming more oligotrophic, making a simple two-timepoint paleolimnological assessment invalid or completely misleading for these systems. Moreover, lakes in the upper Midwestern United States that have been under cultivation since the 1850s would require the analysis of very long cores ($>$ 1 m) to reconstruct trophic change since that time.

More importantly, given the serious flaws in their study that we have detailed in this rebuttal, their final statement that "The assumption of widespread cultural eutrophication for setting numeric nutrient criteria in lakes is not supported," is a non sequitur that has no merit and is highly misleading to managers responsible for water quality in "natural" lakes. We also note here that Bachmann et al. (2013) have chosen wording in this concluding statement that could be construed to refer to *all* U.S. lakes, and not just those of natural origin. Such an unsupportable assertion could potentially be used to provide support for efforts designed to thwart or obstruct the important environmental protection efforts that will be necessary to manage and remediate future, eutrophication-based water quality problems in the United States and elsewhere. NPS eutrophication of lakes is both omnipresent and real; it is a recognized environmental problem that warrants the attention that it is being given both by the USEPA and by all others worldwide who are concerned with the protection of surface-water quality.

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