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LETTER

Cost of reactive nitrogen release from human activities to the environment in the United States

Daniel J Sobota[1,5], **Jana E Compton**[2], **Michelle L McCrackin**[3] **and Shweta Singh**[4]

1 Oak Ridge Institute for Science and Education, in residence at the Western Ecology Division, US Environmental Protection Agency, 200 SW 35th Street, Corvallis, OR-97333, USA
2 Western Ecology Division, US Environmental Protection Agency, 200 SW 35th Street, Corvallis, OR-97333, USA
3 Baltic Sea Centre, Stockholm University, Stockholm, SE-106 91, Sweden
4 University of Toronto, Toronto, Ontario, Canada
5 Current address: Environmental Solutions Division, Oregon Department of Environmental Quality, 811 SW 6th Avenue, Portland, OR 97206, USA.

E-mail: sobota.daniel@deq.state.or.us

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Abstract

Leakage of reactive nitrogen (N) from human activities to the environment can cause human health and ecological problems. Often these harmful effects are not reflected in the costs of food, fuel, and fiber that derive from N use. Spatial analyses of damage costs attributable to source at management-relevant scales could inform decisions in areas where anthropogenic N leakage causes harm. We used recently compiled data describing N inputs in the conterminous United States (US) to assess potential damage costs associated with anthropogenic N. We estimated fates of N leaked to the environment (air/deposition, surface freshwater, groundwater, and coastal zones) in the early 2000s by multiplying watershed-level N inputs (8-digit US Geologic Survey Hydrologic Unit Codes; HUC8s) with published coefficients describing nutrient uptake efficiency, leaching losses, and gaseous emissions. We scaled these N leakage estimates with mitigation, remediation, direct damage, and substitution costs associated with human health, agriculture, ecosystems, and climate (per kg of N) to calculate annual damage cost (US dollars in 2008 or as reported) of anthropogenic N per HUC8. Estimates of N leakage by HUC8 ranged from <1 to 125 kg N ha^{-1} yr^{-1}, with most N leaked to freshwater ecosystems. Estimates of potential damages (based on median estimates) ranged from \$1.94 to \$2255 ha^{-1} yr^{-1} across watersheds, with a median of \$252 ha^{-1} yr^{-1}. Eutrophication of freshwater ecosystems and respiratory effects of atmospheric N pollution were important across HUC8s. However, significant data gaps remain in our ability to fully assess N damages, such as damage costs from harmful algal blooms and drinking water contamination. Nationally, potential health and environmental damages of anthropogenic N in the early 2000s totaled \$210 billion yr^{-1} USD (range: \$81–\$441 billion yr^{-1}). While a number of gaps and uncertainties remain in these estimates, overall this work represents a starting point to inform decisions and engage stakeholders on the costs of N pollution.

Introduction

Human modification of biogeochemical cycles is essential to sustain food production and advance technology; but release of chemicals beyond these intended uses can harm human health, ecosystem function, and the global climate system (Bennett *et al* 2001, Galloway *et al* 2003, Davidson *et al* 2012, Leach *et al* 2012). Finding common measures to assess the damages of human-altered biogeochemical cycles has proven complex because of the diversity of effects, multiple spatial and temporal scales on which they are felt, and ambiguity over how alterations are caused by and affect stakeholders (Galloway *et al* 2003, Banerjee *et al* 2013, Ringold *et al* 2013). Additionally, many ecosystem service-related costs are not well understood, are not transferable to dollar values, or are unknown (Bockstael *et al* 2000). Nevertheless, cost-benefit analyses inform the development of effective management policies (Fisher *et al* 2009, Birch *et al* 2011, van Grinsven *et al* 2013). Frameworks developed to analyze the social cost of carbon (Pearce 2003) and ecosystem services (Boyd and Banzhaf 2007) provide ways to conduct such analyses and have been used to guide policy decisions (Rose 2012). However, analyses of the damages from anthropogenic nutrient use at management-relevant scales remain largely absent.

In this paper, we examined potential damage costs associated with human-moderated inputs of reactive nitrogen (N) across the conterminous United States (US). Application of synthetic N fertilizers and cultivation of N-fixing crops are essential components of the US and global agricultural economy (Smil 2002, Houlton *et al* 2013). Anthropogenic N-fixation also creates important industrial products such as explosives, nylon, and plastics (Domene and Ayers 2001). However, numerous human health and environmental problems result from use and unintentional leakage (e.g., during fossil fuel combustion) of N. These problems include increased mortality and morbidity due to air pollution, contamination of drinking water supplies by NO_3^- (a form of N that can cause blue baby syndrome or other health problems in excess amounts), increased frequency and severity of toxic algal blooms and hypoxia in freshwater and coastal marine ecosystems, and global climate change via emission of the potent greenhouse and ozone-depleting gas N_2O (Davidson *et al* 2012). The intensity of N leakage to ecosystems across the US is nearly twice that of the global average and expected to rise in the future (Galloway *et al* 2004, Sobota *et al* 2013). This makes comparisons of damages to benefits associated with N loading particularly important at regional scales across the country.

Damages of reactive N can be attributed to a given source according to economic values (Birch *et al* 2011, Compton *et al* 2011, van Grinsven *et al* 2013). In this approach, the change in damage cost (mitigation, remediation, direct damage, or substitution) according to change in N loading was calculated for specific N sources (e.g., synthetic fertilizer) and specific human health or environmental impacts (e.g., respiratory effects of air pollution or damage to fisheries production). We used this approach to produce the first estimates of damages of external N release for the entire US and to scale damage costs across watersheds.

Our objective was to assess the magnitude and spatial distribution of damages associated with N loading and leakages across the conterminous US. We connected spatial data describing current N loading and leakages by source across the conterminous US with new information on economic damages of N on agricultural production, human health, ecosystems, and climate (Birch *et al* 2011, Compton *et al* 2011, van Grinsven *et al* 2013). Damages to human health were expected to exceed costs associated with altered ecosystem functions, based on high values placed on human health (Chestnut and Mills 2005, Birch *et al* 2011).

Methods

1. Spatial distribution of N inputs

We compiled spatial data describing new (fixed directly from the atmosphere) and recycled (waste disposal and airborne ammonia) N inputs from human-mediated sources in the early 2000s for the conterminous US. We chose spatial datasets that offered complete coverage of the US land area, the highest spatial resolution, and complete metadata describing data acquisition and representation (Sobota *et al* 2013). We chose the range of the early 2000s because selected datasets did not always have common years. Although N loading rates from specific sources can vary annually, we assume that the individual years captured here approximately represent N loading for this period because year-to-year variation for most inputs is small relative to the amount of the inputs (Sobota *et al* 2013). Also, by choosing this window for comparison, we minimize effects of long-term trends in N inputs, such as decadal trends in declining NO_x emissions and increasing N fertilizer use (Sobota *et al* 2013). We summarized inputs at the spatial resolution of USGS 8-digit Hydrologic Unit Codes (HUC8s; http://water.usgs.gov/GIS/metadata/usgswrd/XML/huc250k.xml) using the Zonal Statistics tool in the Spatial Analyst feature of ArcMap 10.0 (ESRI Inc., Redlands, CA).

For agricultural N inputs (synthetic fertilizer, cultivated biological nitrogen fixation (C-BNF), and confined animal feeding operations (CAFO) manure), we used county-level data for years 2001–2002 (Ruddy *et al* 2006, USDA 2013a). All county-level estimates originate from Ruddy *et al* (2006) except C-BNF, which was estimated by applying coefficients described in Smil (1999) and Howarth *et al* (2002) to areas planted in N-fixing crops or in pasture for 2002 (USDA 2013a). County-scale data were converted to HUC8-scale data by rasterizing county-scale data to 30 arcsecond resolution (~1 km × ~1 km at the equator) and summarizing by HUC8 using the Zonal Statistics tool in ArcMap 10.0.

We estimated the spatial distributions of wastewater and inorganic N deposition to the US using the following methods. For wastewater, we applied the treatment-corrected per capita excretion rate of N (2.8 kg N person yr^{-1}; Van Drecht *et al* 2009) to a 1 km × 1 km gridded dataset of the US population in 2000 (http://lwf.ncdc.noaa.gov/oa/climate/research/population/; rounded to the nearest 10 000). We used 36 km × 36 km gridded data modeled by CMAQ for 2002 (US EPA 2013a) to estimate atmospheric N deposition (inorganic) in the US, assuming that oxidized N (NO_x) originated primarily as new N and ammonia (NH_3) originated as recycled N (Holland *et al* 2005). We summarized annual N inputs of sewage and atmospheric N deposition by HUC8 by rasterizing data (deposition data only; wastewater data were already converted to the appropriate resolution) to 30 arcsecond resolution and summarizing by HUC8 using the Zonal Statistics tool in ArcMap 10.0.

We acknowledge that fine scale variation in N deposition from agricultural activities and roadways may not be sufficiently captured at this resolution. However, our objective was to provide broad watershed and regional estimates of N inputs and ultimately damages. Thus we believe 36×36 km gridded data were sufficient for this purpose, especially when summarizing by HUC8 watershed scales. New, multiyear national scale data describing N inputs at finer scales would help improve these estimates.

2. N leakage to the environment

We estimated N leaked to the environment for individual HUC8s by multiplying the published observed and modeled data describing N inputs to land surfaces (detailed in the previous section) with published coefficients describing the transfer of N to crops, air, land, and water. We did this to calculate damage costs of N at different locations in the N cascade (Galloway *et al* 2003). For simplicity, we assumed that the loss coefficients were spatially homogeneous across the conterminous US, which is likely an oversimplification that could be improved with more unified spatially explicit modeling across systems at the national scale.

We used deposition rates of NO_x–N and NH_3–N (described in the previous section) to characterize leakage of airborne N to HUC8s. Portions of reactive N emitted to the atmosphere can be transported long distances; however, a substantial fraction, particularly ammonia, is deposited locally (Galloway *et al* 2004).

Atmospheric N_2O was estimated by multiplying published coefficients describing fractions of various land-based inputs of N converted to N_2O by the loading rates of land-based N inputs not converted to products (e.g., 60% of synthetic N fertilizer input; Houlton *et al* 2013). We used estimates that 1.1% of N inputs associated with C-BNF, 2.2% of synthetic N fertilizer and manure inputs, and 6% of anthropogenic NO_x emissions (characterized by NO_x deposition rates) were emitted as N_2O (Bouwman 1996, US EPA 2008, Davidson 2009, US EPA 2013b).

N loading to waters included proportions entering surface freshwater, groundwater, and coastal zones. We estimated the proportion of N entering surface freshwater as one-third of the sum of new and recycled anthropogenic non-point N inputs plus sewage N (SAB 2011, Houlton *et al* 2013). Of the remaining two-thirds of anthropogenic non-point N inputs, we calculated that one-third was stored in soil organic matter or denitrified, while one-third leached to groundwater (Houlton *et al* 2013). Though uncertainty behind these splits remains large, N pools calculated using this approach compares well with previous national-scale estimates (SAB 2011). Additional monitoring is needed to improve these estimates. Finally, N delivered to coastal waters from anthropogenic sources was calculated as 40% of anthropogenic N delivered to surface waters within individual HUC8s that eventually drain to coastal areas (McCrackin *et al* 2013).

3. Potential damage costs associated with N inputs

Damage costs associated with specific N inputs were compiled from Compton *et al* (2011) and van Grinsven *et al* (2013) in terms of damage cost (US dollars in the year 2008 or as reported) per kg of N input (table 1). Most of these estimates were taken from large-scale studies (national or regional in nature) to avoid the problems associated with benefit transfer where using site-specific information can produce unreasonable costs for different areas (Plummer 2009). Though we have N loading data from most HUC8s, we do not have cost data for all areas of the US. For these reasons, we consider our estimates to be potential damage costs. These values represent incremental or marginal increases in cost from a current value on a per unit of N basis and assume a linear response function. Nonlinear responses, particularly related to thresholds at low or high N loading rates, might occur but cannot be modeled currently due to limited data (Compton *et al* 2011). This could be a very important consideration, but currently there is not enough information to construct cost estimates using nonlinear effects. For more details on how damage costs associated with N were calculated and compiled, see Compton *et al* (2011) and Birch *et al* (2011).

N can cause damages multiple times along an N cascade from fixation back to N_2 gas (Galloway *et al* 2003). We therefore did not use a mass balance approach to calculate damages, because a single N input could have multiple damages. For example, oxidized N emitted during fossil fuel combustion damages human health while in the atmosphere, damages and (or) benefits to crop production when deposited, and damages water quality when leached into surface- or groundwater. We calculated the spatial distribution of damage costs by multiplying specific damage costs with corresponding N loss pathways in individual HUC8s (table 1). We summed individual damages to produce total damage costs at the scale of HUC8s and the conterminous US. For these calculations, we chose to attribute the atmospheric damages occurred where NO*x*and NH_3 were deposited. We classified individual damage costs as having effects on air/climate, land, freshwater, drinking water, or coastal zones. All

Table 1. Potential damage costs of N ($/kg N; 2008 or as reported) to air, land, and water resources in the conterminous United States in the early 2000s. Low, median, and high costs derive from the specific damage cost reference. Negative values indicate an economic benefit.

N damage type	System	Cost ($/kg N) Low	Median	High	Reference
From atmospheric NO_x					
Increased incidence of respiratory disease	Air/Climate	12.88	23.10	38.63	Birch *et al* (2011), van Grinsven *et al* (2013)
Declining visibility—loss of aesthetics	Air/Climate	0.31	0.31	0.31	Birch *et al* (2011)
Increased effects of airborne particulates/increased carbon sequestration in forests (includes benefits)	Air/Climate	−11.59	−4.51	2.58	van Grinsven *et al* (2013)
Increased damages to buildings from acid	Land	0.09	0.09	0.09	Birch *et al* (2011)
Increased ozone exposure to crops	Land	1.29	1.51	2.58	Birch *et al* (2011), van Grinsven *et al* (2013)
Increased ozone exposure to forests	Land	0.89	0.89	0.89	Birch *et al* (2011)
Increased loss of plant biodiversity from N enrichment	Land	2.58	7.73	12.88	van Grinsven *et al* (2013)
From atmospheric NH_3					
Increased incidence of respiratory disease	Air/Climate	2.58	4.93	25.75	Birch *et al* (2011), van Grinsven *et al* (2013)
Declining visibility—loss of aesthetics	Air/Climate	0.31	0.31	0.31	Birch *et al* (2011)
Increased effects of airborne particulates/increased carbon sequestration in forests (includes benefits)	Air/Climate	−3.86	−1.93	−1.93	van Grinsven *et al* (2013)
Increased damages to buildings from particulates	Land	0.09	0.09	0.09	Birch *et al* (2011)
Increased loss of plant biodiversity	Land	2.58	7.73	12.88	van Grinsven *et al* (2013)
From N_2O					
Increased ultra-violet light exposure from ozone—humans	Air/Climate	1.29	1.33	3.86	Compton *et al* (2011), van Grinsven *et al* (2013)
Increased emission of a greenhouse gas	Air/Climate	5.15	13.52	21.89	van Grinsven *et al* (2013)
Increased ultra-violet light exposure from ozone—crops	Air/Climate	1.33	1.33	1.33	Birch *et al* (2011)
From surface freshwater N loading					
Declining waterfront property value	Freshwater	0.21	0.21	0.21	Dodds *et al* (2009)
Loss of recreational use	Freshwater	0.17	0.17	0.17	Dodds *et al* (2009)
Loss of endangered species	Freshwater	0.01	0.01	0.01	Dodds *et al* (2009)
Increased eutrophication	Freshwater	6.44	16.10	25.75	Compton *et al* (2011), van Grinsven *et al* (2013)
Undesirable odor and taste	Drinking water	0.14	0.14	0.14	Kusiima and Powers (2010)
Nitrate contamination	Drinking water	0.54	0.54	0.54	Compton *et al* (2011)
Increased colon cancer risk	Drinking water	1.76	1.76	5.15	van Grinsven *et al* (2013)
From groundwater N loading					
Undesirable odor and taste	Drinking water	0.14	0.14	0.14	Kusiima and Powers (2010)
Nitrate contamination	Drinking water	0.54	0.54	0.54	Compton *et al* (2011)
Increased colon cancer risk	Drinking water	1.76	1.76	5.15	van Grinsven *et al* (2013)
From coastal N loading					
Loss of recreational use	Coastal zone	6.38	6.38	6.38	Birch *et al* (2011)
Declines in fisheries and estuarine/marine habitat	Coastal zone	6.00	15.84[a]	26.00	Compton *et al* (2011), van Grinsven *et al* (2013)

[a] Excluding $56/kg N from submerged aquatic vegetation loss in the Gulf of Mexico from Compton *et al* (2011)

statistical analyses were conducted in R v.3.0.0 (R Development Core Team 2011).

Results

4. Anthropogenic N inputs

Median input of new human-mediated N to HUC8s in US (in the early 2000s) was 26 kg N ha^{-1} yr^{-1}, with a minimum and maximum of <1 and 130 kg N ha^{-1} yr^{-1}, respectively (figures 1(A) and 2(A)). At the national scale, we estimate that 19.4 Tg of new N entered US air, land, and waterways in the early 2000s (figure 2(B)). The average input of recycled human-mediated N to HUC8s was 9 kg N ha^{-1} yr^{-1}, with a minimum and maximum of <1 and 85 kg N ha^{-1} yr^{-1}, respectively (figures 1(B) and 2(A)). Nationally, we

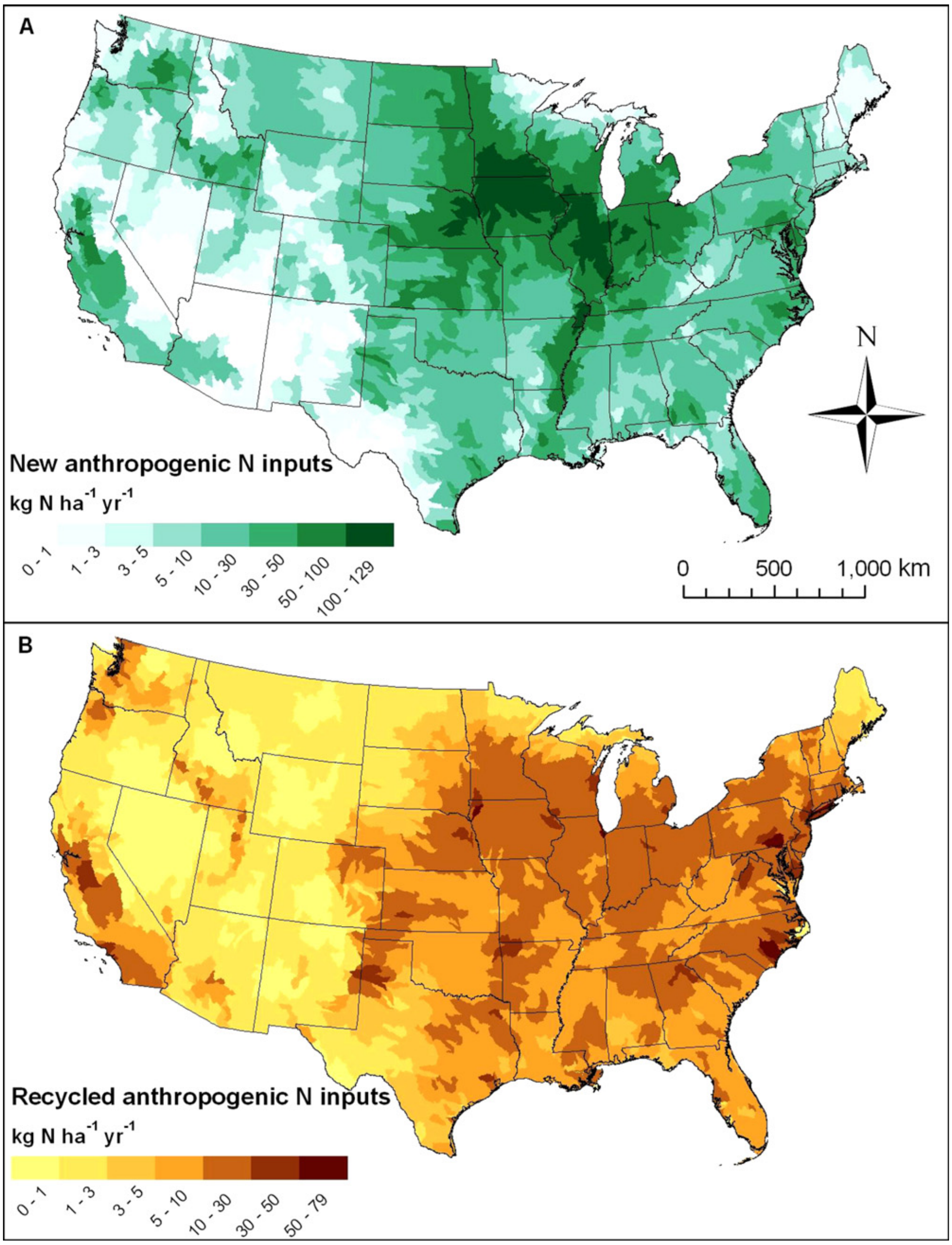

Figure 1. Distribution of new (A) and recycled (B) anthropogenic N inputs to 8-digit US Geologic Survey Hydrologic Unit Code (HUC8) watersheds in the United States in the early 2000s. New anthropogenic N inputs include synthetic fertilizer, cultivated biological N fixation (C-BNF), and the emission/deposition of oxidized forms of N associated with fossil fuel combustion. Recycled anthropogenic N inputs include emission/deposition of ammonia, spread of manure collected on confined animal feeding operations onto agricultural fields, and sewage discharge. N inputs represent average per area N input to the entire HUC8 watershed.

estimate that 6.3 Tg N of recycled N entered US air, land, and waterways in the early 2000s (figure 2(B)).

Across the conterminous US, synthetic N fertilizer and C-BNF were the largest and second-largest overall human-mediated N sources by HUC8 and at the national scale (figure 2). Oxidized N deposition was the third largest new N source by HUC8 and nationally, but dominated total inputs in many urban areas (e.g., portions of the East Coast, the Upper Great Lakes region, the Southwest, and the Pacific Northwest). Ammonia and manure N from CAFOs were the first and second largest sources of recycled N to HUC8s and nationally, and were most important in areas with high livestock populations, such as Eastern North Carolina, Northern Georgia, and Western Arkansas. Inputs of N from sewage were the smallest of either new or recycled N sources across HUC8s and nationally (figure 2), although sewage dominated overall N inputs in some HUC8s draining major urban areas such as New York, Denver, Las Vegas, and Los Angeles.

5. Anthropogenic N leaked to the environment

The amount of anthropogenic N leaked to the environment in HUC8s ranged from 0.1 to 104 kg

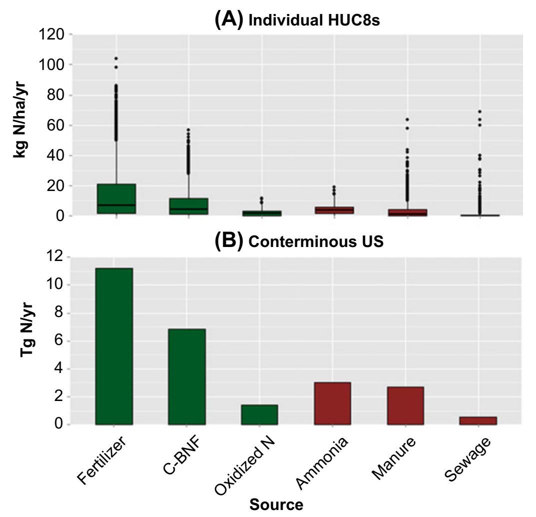

Figure 2. Boxplots of (A) per area anthropogenic N inputs by source to HUC8 watersheds in the early 2000s ($n = 2107$ per N input) and (B) total anthropogenic N input to the conterminous US in the early 2000s. For panel (A), bottom and top whiskers indicate 10th and 90th percentiles, bottom and top box edges depict 25th and 75th percentiles, and the line in the box is the median. Green indicates newly fixed N while brown indicates recycled N in both panels.

N ha^{-1} yr^{-1} with a median of 17 kg N ha^{-1} yr^{-1} (figure 3). N leakages followed a spatial pattern similar to that as new and recycled N inputs to HUC8s, with the upper Midwest and Central California losing the largest amounts of N to the environment. Based on median values of all HUC8s, the ranking of leakages was as follows: surface freshwater (4.5 kg N ha^{-1} yr^{-1}), ammonia to the atmosphere and eventually land surfaces (3.8 kg N ha^{-1} yr^{-1}), groundwater (3.7 kg N ha^{-1} yr^{-1}), oxidized N from fossil fuel combustion to the atmosphere and eventually land surfaces (3.3 kg N ha^{-1}yr^{-1}), coastal zones (1.8 kg N ha^{-1}yr^{-1}), and N_2O (0.4 kg N ha^{-1} yr^{-1}) (figure 4(A)). At the national scale, the ranking of leakages was as follows: surface freshwater (4.8 Tg N yr^{-1}), groundwater (4.2 Tg N yr^{-1}), ammonia (3.0 Tg N yr^{-1}), coastal zones (1.9 Tg N yr^{-1}), and oxidized N from fossil fuel combustion (1.4 Tg N yr^{-1}) (figure 4(B)).

6. Potential damage costs associated N inputs

Potential damage costs associated with anthropogenic N leakage ranged from \$1.94 to \$2255.00 ha^{-1} yr^{-1} across HUC8s in 2000 (figure 5). Between 73 and 77% (median = 75%) of the potential damage costs were associated with leakage of agricultural N, driven by harmful effects on aquatic habitat and eutrophication. Another 14–24% of the potential damage costs (14–\$94 billion; median = \$50 billion or 24% of the median total of \$210 billion) were associated with fossil fuel combustion. Areas with the largest damage costs corresponded to areas with the largest N inputs and leakages (figures 1 and 3), such as the upper Midwest and Central California. However, due to the differential costs of damages to human health/society, ecosystems, agriculture, and climate, several regions with smaller N inputs and leakages had damage costs comparable to areas with higher overall N loads (figures 1, 3, and 5). For example, the mid-Atlantic, Pacific Northwest, and Southern California received less N annually than intense agricultural areas such as the upper Midwest; yet damage costs associated with N leakages were similar because of the high cost of air pollution on human health.

Potential damages to aquatic ecosystems generally followed the spatial distribution of total N inputs (figure 6). In contrast, potential damages to air and climate were more evenly distributed across the conterminous US because of the high cost of air pollution on human health (figure 7). Potential damage costs of anthropogenic N to HUC8s by system ranged from median values of \$17.73 ha^{-1} yr^{-1} to drinking water to \$73.73 ha^{-1} yr^{-1} to freshwater ecosystems (figure 8(A)). At the national scale, best estimates of potential damages ranged from \$19 billion associated with drinking water impacts to \$78 billion associated with impacts on freshwater ecosystems (figure 8(B)). However, substantial ranges of total damages occur within and across systems affected based on all available damage cost estimates (error bars in figure 8(B)). Summing up HUC8 estimates across the US suggests that anthropogenic N leaked to the environment contributed \$81–\$441 billion (median estimate of \$210 billion) in potential damage costs annually to the US economy in the early 2000s. Summaries of damages to endpoint effects are detailed in appendices A and B.

Discussion

7. Fates and damages of N leaked to the environment

This work represents a first attempt to assess damage costs associated with N leakage to the environment from all human activities in the US. Nearly 75% of the damage costs were associated with agricultural N leakage and effects on aquatic systems. Although fossil fuel combustion represents less than 17% of the release to the environmental, 24% of the damages were associated with fossil fuel combustion. Fossil fuel sources cause disproportionally higher relative costs due valuation of human health impacts resulting in comparatively larger unit damage costs (through respiratory and cardio-vascular effects of particulate matter and ground level ozone) than is the case for ecosystem and crop impacts (Muller and Mendelsohn 2007, Birch et al 2011). The damage costs represent the sum of all available costs associated with N leakage; because damage cost estimates are linearly proportional to leakage, marginal reductions in a source (e.g., a 25% reduction in release of N from agriculture or sewage) would be expected to result in a concomitant reduction in damages. This assumption

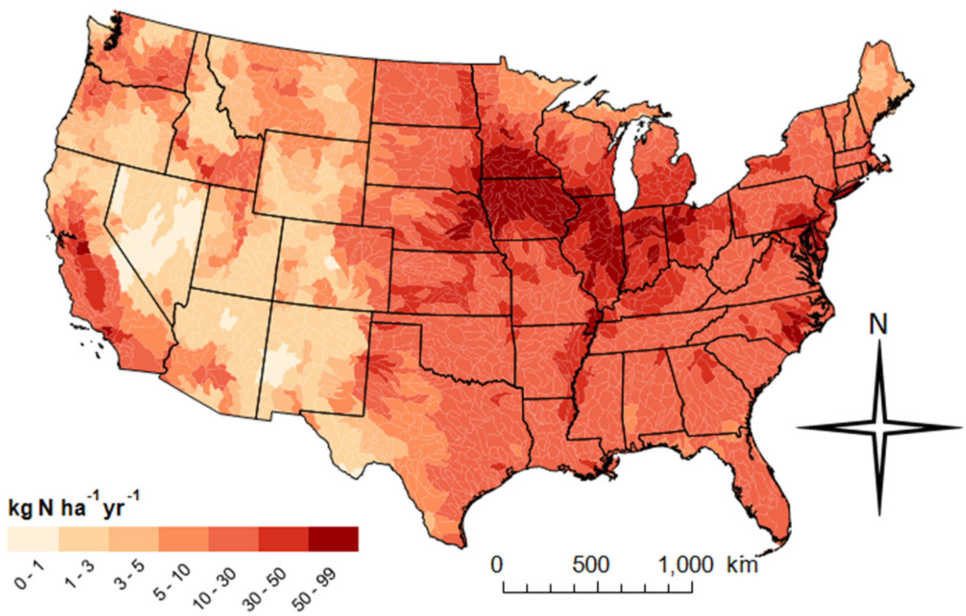

Figure 3. Distribution of total anthropogenic N leaked to the environment in HUC8 watersheds across the conterminous US in the early 2000s. Leaked N consists of the fractions of new and recycled anthropogenic N inputs that are not utilized directly in human products and are lost to air, land, or water resources (see text for details). N leakages represent average per area N leakages for the entire HUC8 watershed.

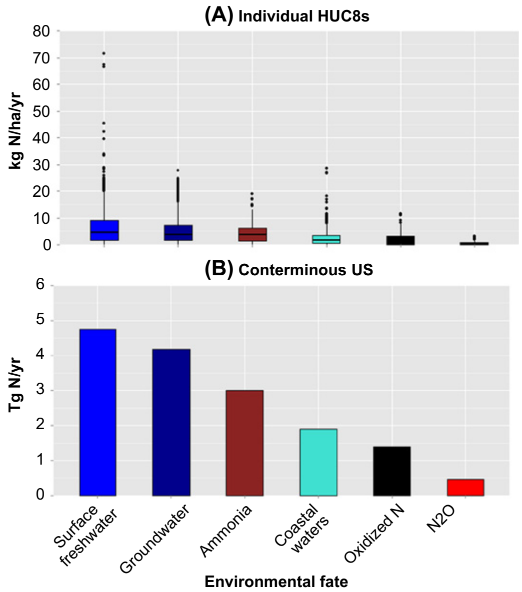

Figure 4. Boxplots of (A) per area anthropogenic N leakage to the environment to HUC8 watersheds of the conterminous United States in the early 2000s ($n = 2107$ per N input) and (B) total anthropogenic N leakage to the environment the conterminous US in the early 2000s. For panel (A), bottom and top whiskers indicate 10th and 90th percentiles, bottom and top box edges depict 25th and 75th percentiles, and the line in the box is the median.

of linearity is an important topic for further research. Nearly 71% of anthropogenic N leaked to the environment ended up in water resources, which is consistent with previous N cycling studies in the US (Jordan and Weller 1996, Howarth *et al* 2002, Alexander *et al* 2008, SAB 2011, Davidson *et al* 2012).

Areas with substantial agricultural N inputs tended to have greater damage costs due to high N loading rates compared to urban and non-cultivated lands. Within agricultural regions, application of synthetic N fertilizers, C-BNF by crops such as soybeans and alfalfa, and land application of manure generated on CAFOs largely drove N loading and leakages. Improvements to fertilizer application practices and the development of crop strains with high nutrient uptake efficiency over the past 40 years have prevented much larger N leakages (Cassman *et al* 2002). In spite of these improvements in efficiency, cultivation of major grain and fodder crops still contribute the largest share of N leaked to the environment, and economic damages, in many US watersheds (Jordan and Weller 1996, Alexander *et al* 2008).

8. Opportunities to reduce damages

Although we did not specifically examine reduction strategies, others have suggested actions to improve nutrient management and slow the release of N to air and water that in turn could reduce damages in many watersheds. Many of these efforts, such as crop breeding and improvements to N application methods, are currently underway (Cassman *et al* 2002, Robertson and Vitousek 2009). For example, N use efficiency by corn has nearly doubled since the 1970s (Cassman *et al* 2002). Improvements to N use efficiency are still possible because the complete set of recommended practices has a low adoption rate in up

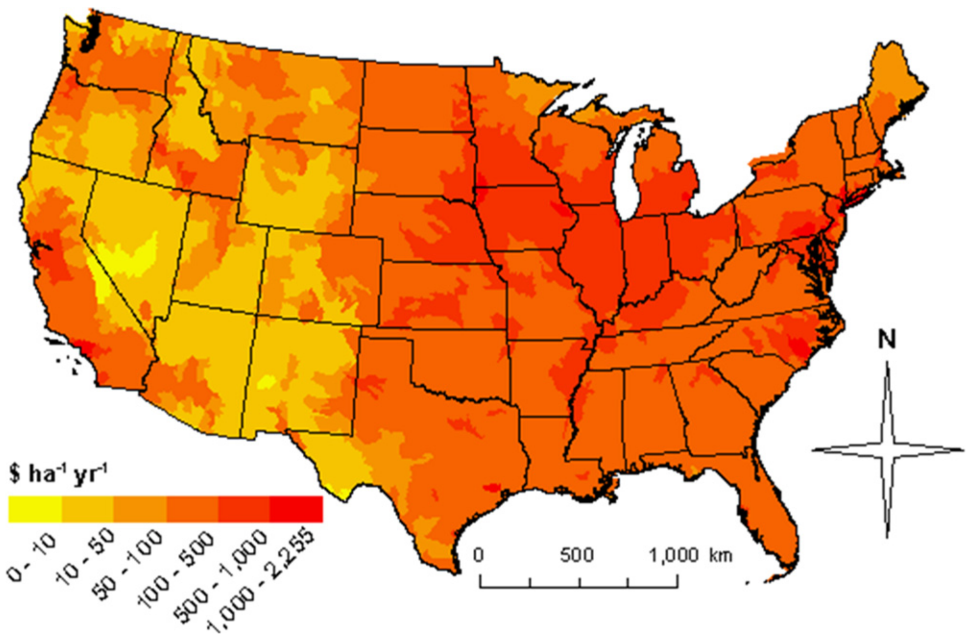

Figure 5. Distribution of total potential damage costs caused by anthropogenic N leaked to the environment by HUC8 watersheds of the conterminous United States in the early 2000s. Potential damage costs were calculated by multiplying specific new and recycled anthropogenic N inputs by source with the central damage cost estimate of US dollars (2008 or as reported) per kg of N leaked to the environment.

to 70% of croplands across the US (Ribaudo *et al* 2011). Increased N use efficiency could also be achieved in livestock production since of the nearly 7 Tg N yr^{-1} fed to livestock (SAB 2011, Foley *et al* 2011), ~70% leaks to air, land, and water via ammonia emissions and manure spreading (Sobota *et al* 2013). From a human health perspective, ammonia emissions are particularly damaging, causing significant respiratory illness with damage costs of over \$100/kg N in some locations (Paulot and Jacob 2014). Nitrate derived from manure also impacts drinking water supplies in areas where CAFOs are clustered (Rosenstock *et al* 2014). Due to social and economic realities, better agricultural N management will require increased efforts in watershed education, technical support and funding focused on nutrient management (Osmond *et al* 2014).

Social changes at the scale of individual choices could improve N use efficiency in products and reduce demand for N. Three such changes include reducing food waste (USDA 2013b), promoting diets with more plant-based protein (Howarth *et al* 2002), and increasing the use of mass transit systems (Leach *et al* 2012). Additional reductions can be achieved through continued improvements in sewage treatment and maintaining the N reductions associated with Clean Air Act regulations (SAB 2011). The damage cost information here could represent an opportunity for decision-makers to identify places and sources of N where the tradeoffs are worth these investments in improved N management.

9. Context of damage costs associated with N use

Addressing the benefits of N use within the US was beyond the scope of this study, and more work is needed to fully assess the overall costs and benefits of N use. Our national estimate of potential damages (\$210 billion yr^{-1}; range \$81–\$441 billion yr^{-1}) was equivalent to 1–3% of the national gross domestic product in 2000 (IMF 2013). This range of damages is similar in magnitude to a recent continental scale assessment for the European Union (\$97–625 billion USD, van Grinsven *et al* 2013). Our estimated potential damages associated with NO_x and NH_3 were approximately \$43 billion yr^{-1}; quite similar to \$29.5 billion gross annual damages associated with NO_x and NH_3 from Muller and Mendelsohn (2007). Potential damages from agricultural N use (\$59–\$340 billion yr^{-1}; median of \$157 billion yr^{-1}) were a large portion of the total damages. In the European Union, van Grinsven *et al* (2013) estimated that damages of agricultural N pollution exceeded economic benefits of increased agricultural production by up to fourfold.

Anthropogenic N fixation is essential to modern society and technology. In particular, at least one-third of the world's population would not be alive without synthetic N fertilizers (Smil 1997). The nutritional value of food is also greatly enhanced through use of synthetic N fertilizers or legume-based N (Smil 2002). Additionally, a number of indirect economic benefits result from anthropogenic N fixation in agriculture and industry that

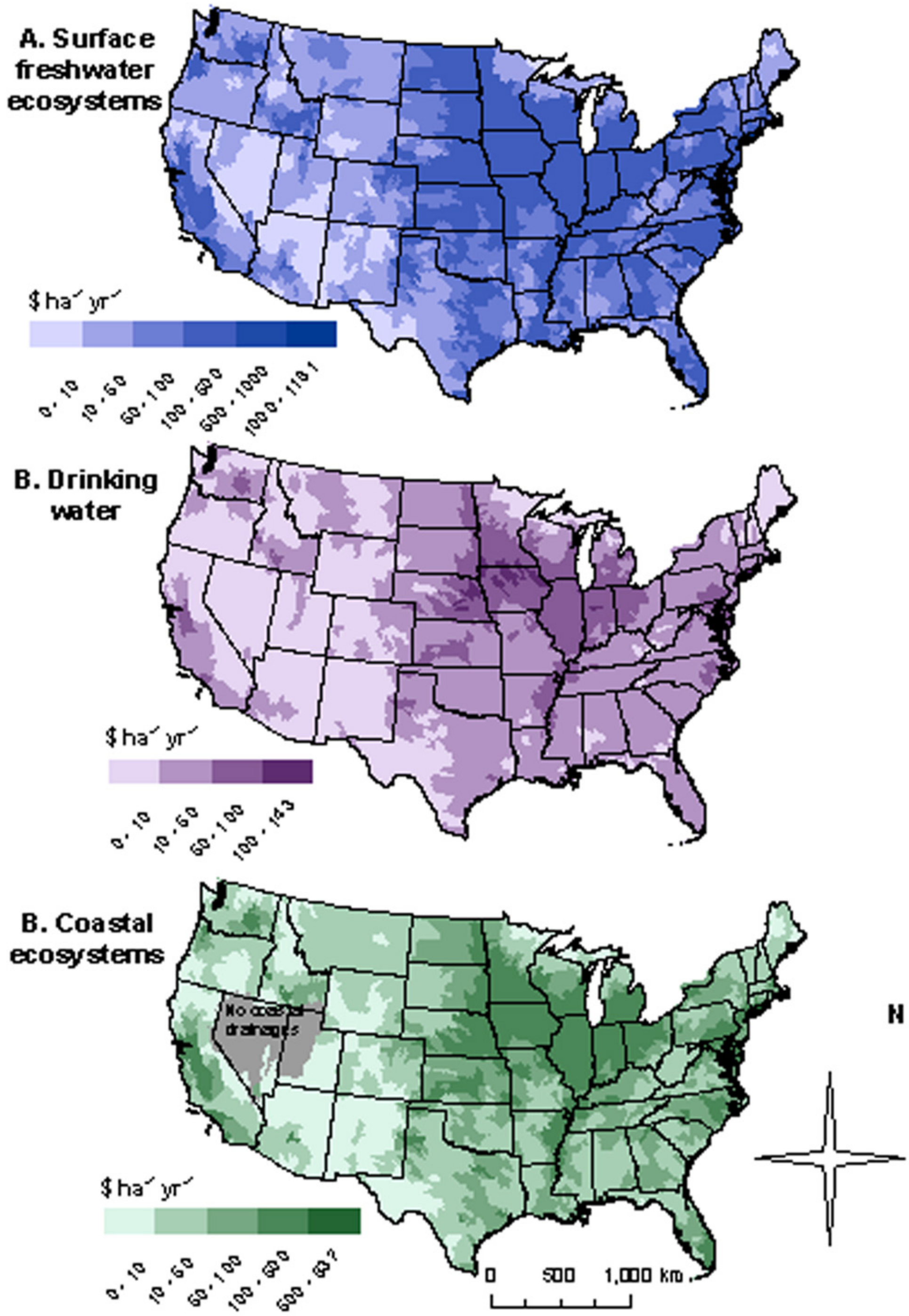

Figure 6. Distribution of potential damage costs to (A) freshwater ecosystems, (B) drinking water, and (C) coastal ecosystems caused by anthropogenic N leaked to the environment by HUC8 watersheds of the conterminous United States in the early 2000s. Potential damage costs were calculated by multiplying specific new and recycled anthropogenic N inputs by source with the median damage cost per kg of N leaked to the environment.

are not directly quantified by on-site farm profit margins (Singh and Bakshi 2013). These include retail sales, transportation, and international transport of agricultural goods and industrial products reliant on anthropogenic N fixation. Our estimate of the damages of reactive N leakage to the environment thus could serve as a starting point for the costs component of needed research to assess the tradeoffs associated with N use and release.

10. Limitations and research needs

Our estimates highlight the need for improved spatial estimates of N leakages throughout the US and more data describing the link between N overabundance and damages to health and the environment. Key research needs include:

- *Response curves of damage costs.* Because of the lack of data describing marginal response curves of

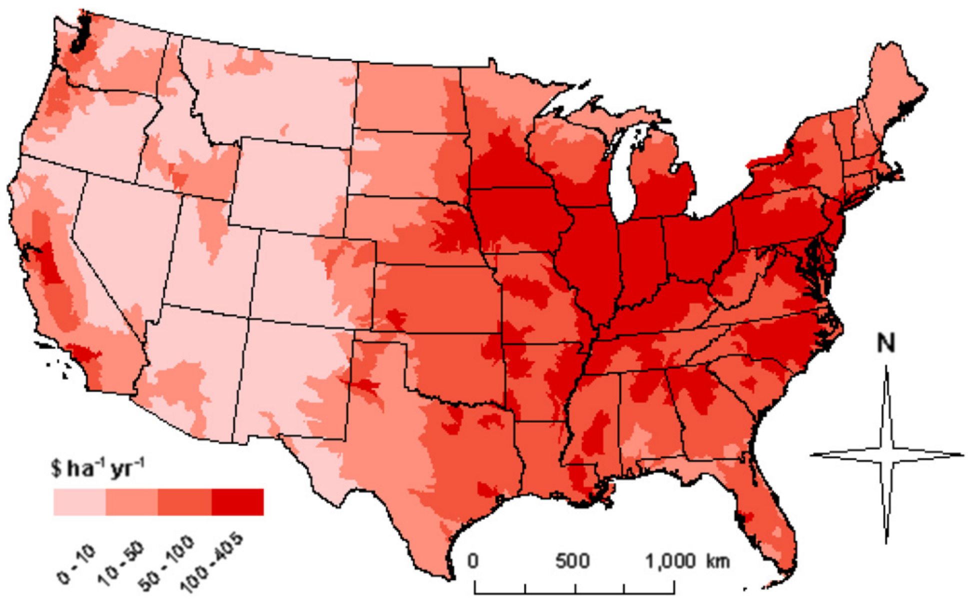

Figure 7. Distribution of potential damage costs (2008 or as reported) to air and climate caused by anthropogenic N leaked to the environment by HUC8 watersheds of the conterminous United States in the early 2000s. Potential damage costs were calculated by multiplying specific new and recycled anthropogenic N inputs by source with the median damage costs in US dollars (2008 or as reported) per kg of N leaked to the environment.

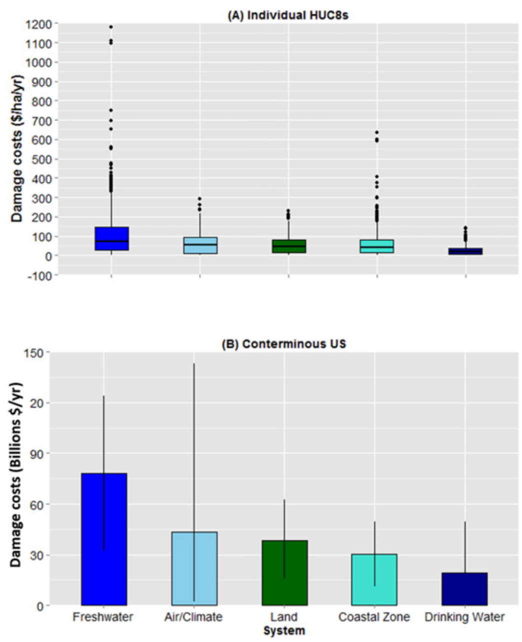

Figure 8. Boxplots of (A) per area potential damage costs (2008 or as reported) from anthropogenic N by system affected to HUC8 watersheds in the early 2000s ($n = 2107$ per system) and (B) total potential damage costs from anthropogenic N input to the conterminous US in the early 2000s. For panel (A), bottom and top whiskers indicate 10th and 90th percentiles, bottom and top box edges depict 25th and 75th percentiles, and the line in the box is the median of the median potential damage costs listed in table 1. For panel (B), bars represent potential damage costs from median damage costs in table 1 while the whiskers represent total damage costs calculated from low and high damage cost estimates.

economic damages with incremental increases in N loading by source, our estimates are constrained by an assumption of linear scaling of damages with loading rates. Many marginal costs respond nonlinearly to incremental changes in stressors (Boyd and Banzhaf 2007); undoubtedly this is the case with N loading. Improved marginal cost response curves of N loading to freshwater ecosystems, coastal zones, and air are particularly important because they have the highest per area damage costs identified in our analysis. One way to address this in the future would be to incorporate a critical loads approach, where there is a threshold below which damages are minimal and above which costs are asymptotic (Pardo *et al* 2011, Clark *et al* 2013). For economic damages we are not yet able to define such a threshold.

- *Costs associated with aquatic eutrophication.* Data describing damage costs associated with eutrophication in freshwater and coastal ecosystems are sparse and may not capture the full range of important effects (Dodds *et al* 2009). Future studies that link N loading to aquatic ecosystems with short and long-term health impacts (e.g., hospital visits and chronic diseases) as well as with loss of economic development (e.g., loss of recreational activities) would advance our understanding of the widespread impacts of N leakages to freshwater systems.

- *Health and treatment costs of N contamination of drinking water.* As the number of community water supplies with NO_3^- violations have increased over the past two decades (US EPA 2013c), more information is needed concerning the long-range health consequences with N pollution of drinking water supplies (Davidson *et al* 2012, Brender *et al* 2013). Research is needed that examines spatial variability of these costs due to differences in treatment technologies or the magnitude, frequency, and duration of exposure to harmful N levels.

- *Economic impacts of atmospheric N emissions on global climate.* Atmospheric levels of N_2O have increased significantly over the past century (Davidson 2008). At the same time, particulates formed from oxidized N and ammonia have had a cooling effect on the global climate (Pinder *et al* 2013). Additionally, broad-scale N fertilization of terrestrial ecosystems from N deposition may be enhancing carbon sequestration (Pinder *et al* 2013), offsetting effects of increased carbon emissions. Uncertainty about these interactions makes research linking N leakages with climate critical.

Conclusions

Here we provide initial estimates of damage costs associated with leakages of anthropogenic N to the environment across the conterminous US. Most N (71% of leakage) ended up in water resources (surface freshwater, groundwater, and coastal zones), where it led to several costly effects. Health impacts of air pollution were also costly across the nation, disproportionately more expensive relative to the amount of N leaked to air versus water because of the high cost of respiratory illnesses associated with ozone and particulate matter precursors. Cooling associated with particulates had a slight climate benefit based on current data. Improving N use efficiency, particularly in agricultural ecosystems, and modifying social behavior to demand less N will be critical to reduce damages to human health and aquatic ecosystems.

Currently, damages of N leakages from agriculture and other non-point sources are considered externalities not captured in the cost of doing business. Our current analysis could provide a starting point to aid N management at watershed, regional, and national scales in the US. It could also allow stakeholders to illustrate benefits associated with targeted N reductions by agricultural or industrial sector. This information could provide insight on N use choices in individual HUC8s, and illustrate to decision-makers and key stakeholders the ecosystem and human health benefits of improved N management. Although there are a number of gaps and uncertainties in these estimates, overall this work represents a starting point to inform decisions and engage stakeholders on the costs of nitrogen pollution.

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Appendix A

Table A1 Table of codes used for boxplots of potential damage costs to the conterminous US in appendix B.

N damage type	Code
From atmospheric NO_x	
Human health-respiratory	HH-NOx
Visibility	VIS
Climate change	COOL
Infrastructure damage	ID
Ozone effects on crops	O_3AG
Ozone effects on forests	O_3FOR
Plant biodiversity loss	BIOD
From atmospheric NH_3	
Human health-respiratory	HH-NH_3
Visibility	VIS
Climate change	COOL
Infrastructure damage	ID
Plant biodiversity loss	BIOD
From N_2O	
Ozone–UV light exposure	HH-UV
Greenhouse gases	GHG
Ozone–UV damage	AG-UV
From surface freshwater N loading	
Waterfront property value	WFP
Recreational use	FWREC
Endangered species	END
Eutrophication	EUT
Odor and taste	ODOR
Nitrate level	NIT
Colon cancer risk	CANC
From groundwater N loading	
Odor and taste	ODOR
Nitrate level	NIT
Colon cancer risk	CANC
From coastal N loading	
Recreational use	CZREC
Fisheries	FISH

Appendix B

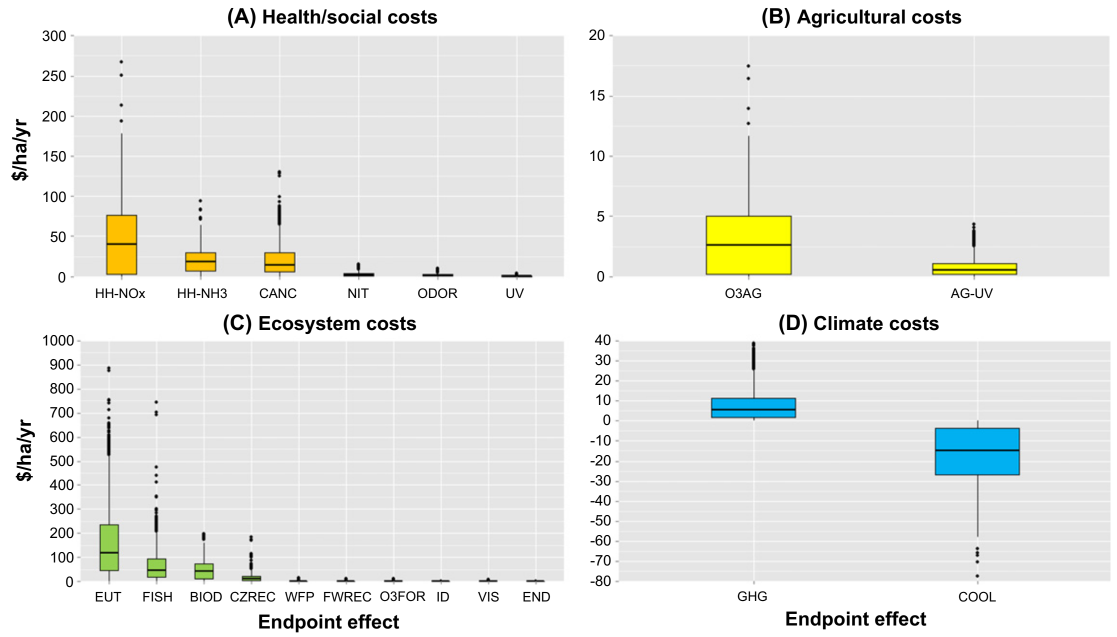

Figure B1. Boxplots of potential damage costs (2008 or as reported) associated with anthropogenic N leaked to the environment by HUC8 watershed in the early 2000s for different endpoint effects ($n = 2107$ each) by (A) health/social costs, (B) agricultural costs, (C) ecosystem costs, and (D) climate costs. Letter code identifications are listed in appendix A. For each boxplot, bottom and top whiskers indicate 10th and 90th percentiles, bottom and top box edges depict 25th and 75th percentiles, and the line in the box is the median. Damage cost calculations represent median damage costs presented in table 1.

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