Ecosystem services altered by human changes in the nitrogen cycle: a new perspective for US decision making

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REVIEW AND SYNTHESIS

Ecosystem services altered by human changes in the nitrogen cycle: a new perspective for US decision making

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Abstract

Human alteration of the nitrogen (N) cycle has produced benefits for health and well-being, but excess N has altered many ecosystems and degraded air and water quality. US regulations mandate protection of the environment in terms that directly connect to ecosystem services. Here, we review the science quantifying effects of N on key ecosystem services, and compare the costs of N-related impacts or mitigation using the metric of cost per unit of N. Damage costs to the provision of clean air, reflected by impaired human respiratory health, are well characterized and fairly high (e.g. costs of ozone and particulate damages of \$28 per kg NO_x-N). Damage to services associated with productivity, biodiversity, recreation and clean water are less certain and although generally lower, these costs are quite variable (< \$2.2–56 per kg N). In the current Chesapeake Bay restoration effort, for example, the collection of available damage costs clearly exceeds the projected abatement costs to reduce N loads to the Bay (\$8–15 per kg N). Explicit consideration and accounting of effects on multiple ecosystem services provides decision-makers an integrated view of N sources, damages and abatement costs to address the significant challenges associated with reducing N pollution.

Keywords

Air quality, ecosystem services, human health, human well-being, management, nitrogen, water quality.

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INTRODUCTION

Nitrogen (N) is an essential element required for the growth and maintenance of all biological tissues, and often limits primary production in terrestrial and aquatic ecosystems (Elser *et al.* 2007; LeBauer & Treseder 2008). Human population growth and increased demands for energy, transportation and food lead to increased N fixation, which in turn has increased the size and quality of the global food supply (Townsend *et al.* 2003; Galloway *et al.* 2008). It is now widely accepted that through activities such as inorganic N fertilizer production, legume cultivation and fossil fuel combustion, humans have more than doubled global rates of pre-industrial terrestrial N fixation, with even higher rates of anthropogenic N fixation expected to occur in coming decades (Vitousek *et al.* 1997; Galloway *et al.* 2004, 2008). However, in many regions human impact on the N cycle is even more dramatic. In the continental USA, human activities have increased terrestrial N fixation rates by a factor of at least 3.5 within the past century, due largely to increases in inorganic N fertilizer application rates and fossil fuel combustion (Fig. 1). While enhanced N fixation has undeniable societal benefits, N is also a powerful environmental pollutant. This intensification of N release to the environment has resulted in important and growing effects on human and ecological health (Table 1; Vitousek *et al.* 1997; Johnson *et al.* 2010), affecting essential ecosystem services such as the provision of clean air and water, recreation, fisheries, forest products, aesthetics and biodiversity.

One reason that N is particularly vexing from a management and regulatory standpoint is the complexity of the biogeochemical N cycle and its environmental effects. Once fixed from the atmosphere, a single molecule of N is often transformed and utilized multiple times before being removed from circulation via long-term storage or denitrification, magnifying the impact of anthropogenic N fixation on natural systems (Fig. 2; Galloway *et al.* 2003). The Millennium Ecosystem Assessment Board (MA) (2005) underscored that understanding the tradeoffs inherent in controlling this class of environmental pollutant is one of the major challenges to be faced in the 21st century. Pollutants like N pose a challenge to traditional pollution regulatory systems because (1) effects are not primarily due to direct toxicity but rather to changes in ecosystem structure and function, some of which could be seen as beneficial, (2) effects cross traditional

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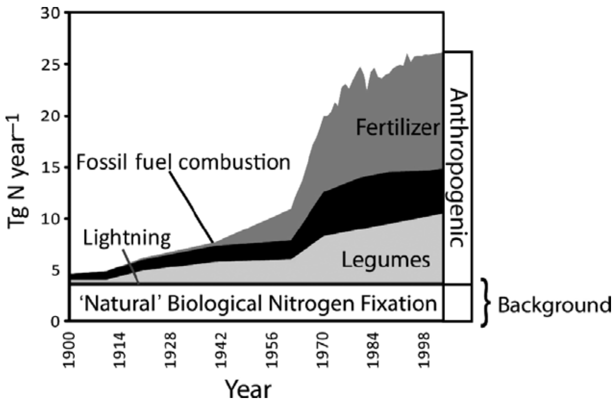

Figure 1 Natural and anthropogenic sources of 'new' N to the landscape for the continental USA. Data sources: Lightning: (Galloway *et al.* 2004; assumed constant), 'Natural' or background biological N fixation: (Bouwman *et al.* 2009), Fertilizer: (NASS for 1940, FAOSTAT for 1961–1999), Fossil fuel combustion: (US EPA 1985, 2000; R. Dennis, pers. comm.), agricultural N fixation from legumes: (calculated from FAOSTAT for 1961–1999 and NASS for 1909, and 1919). Years without data were estimated using linear interpolation.

media-specific regulatory boundaries (e.g. one atom of N can cause effects regulated by both the US Clean Air Act and Clean Water Act), (3) the pollutant can be converted from one chemical form to another, each of which has different effects and (4) sensitivity to pollutants is variable from place to place such that a fixed air or water quality standard may not apply everywhere depending upon ecosystem characteristics. For example, N, phosphorus and sometimes other nutrients can act together, sequentially or concurrently, to limit primary production. Further complicating the picture is the fact that nutrient enrichment can lead to both desirable and undesirable changes for human health and well-being. The complexity of N effects necessitates a perspective that considers the positive and negative effects of this type of pollutant. An approach that examines ecosystem services and human well-being could focus and augment more traditional approaches, which have had limited success and left us with continuing nutrient problems (US EPA 2009).

The goals of this paper are to (1) review the state of the science connecting increasing N to ecosystem services, (2) identify the research available and needed for an ecosystem services approach to management of N, and (3) compare N damage costs with mitigation, restoration and replacement costs. Many reviews have explored the effects of increasing N on terrestrial and aquatic ecosystems, and our objective is not to repeat these efforts. Rather, we investigate how to connect changes in ecosystem structure and function directly to the *services* provided by ecosystems; in particular those services that have the most direct consequences for human benefit and well-being. Table 1 illustrates qualitative effects of N on ecosystem processes and services. In addition to reviewing the science, we provide a rationale for considering ecosystem services in environmental management and policy decisions, and identify the knowledge required to construct an ecosystem services-based framework that would inform more efficient N management. Information regarding costs is drawn from across the globe (e.g. van Grinsven *et al.* 2010), but we focus our analysis on connections to US policies and actions. We present the cost data in 2008 dollars as noted; otherwise data are presented as found (not adjusted for inflation). Lastly, we build upon work in the Chesapeake Bay that has applied such a framework to examine the damage costs of excess N (Birch *et al.* 2011), in order to move closer to a better quantification of the relative magnitude of damage costs to ecosystem services and human well-being, and the costs to reduce N pollution.

Table 1 Ecosystem services and human benefits affected by increasing N

Ecosystem Service	Impact on benefit	Mechanism of impact
Production of food and materials	+	Increased production and nutritional quality of food crops
	+	Increased production of building materials and fibre for clothing or paper
	−	Stimulation of ozone formation, which in turn can reduce agricultural and wood production
	−	Soil acidification, nutrient imbalances and altered species composition and diversity in forests and other natural ecosystems, which ultimately impact stability and resistance to disease, invasive species and fire
Fuel production	+	Increased use of fossil fuels to improve human health and well-being across the globe
	+/−	Increased N inputs required for some biofuel crops can affect other services
Clean air	−	NO_x-driven increases in ozone and particulates exacerbate respiratory and cardiac conditions
	−	Increased allergenic pollen production
Drinking water	−	Increased nitrate concentrations lead to blue-baby syndrome, certain cancers
	−	Increased acidification and mobility of heavy metals and aluminium
Swimming	−	Stimulation of harmful algal blooms that release neurotoxins (interaction with phosphorus)
	−	Increased vector-borne diseases such as West Nile virus, malaria and cholera
Fishing	+	Increased fish production and catch for some very N-limited coastal waters
	−	Increased hypoxia and harmful algal blooms in coastal zones, closing fish and shellfish harvests
	−	Reduced number and species of recreational fisheries from acidification and eutrophication
Hiking	−	Altered biodiversity, health and stability of natural ecosystems
Climate regulation	+/−	Variable and system-dependent impacts on net CO_2 exchange
	−	Stimulation of N_2O production, a powerful greenhouse gas
UV regulation	−	Increased N_2O release, which has strong ozone-depleting potential
Visibility	−	Increased NO_x in air stimulates formation of particulates, smog and regional haze
Cultural and spiritual values	−	Altered biodiversity, food webs, habitat and species composition of natural ecosystems
	−	Damage to buildings and structures from acids
	+/−	Long range trans-boundary N transport and associated effects (both negative and positive)

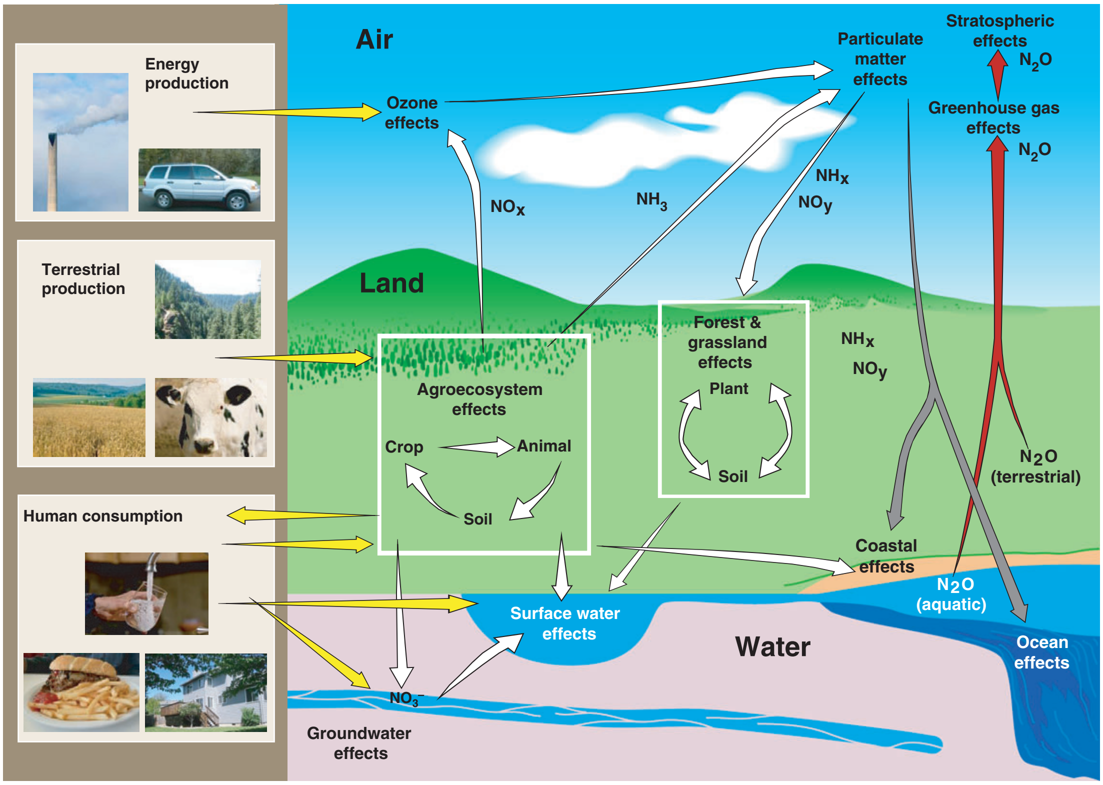

Figure 2 The nitrogen cascade. Modified from J. Galloway, pers. comm.; Photo credits J. Compton or http://intranet.epa.gov/media/phototopics.htm.

DEFINING AN ECOSYSTEM SERVICES APPROACH

Put simply, ecosystem services are the aspects of nature that benefit people. Daily (1997) defines ecosystem services as the 'conditions and processes through which natural ecosystems and species therein sustain and fulfil human life or have the potential to do so in the future.' The Millennium Ecosystem Assessment Board (MA) (2005) categorized services into provisioning services, supporting services, regulating services and cultural services. Others have refined this definition to improve the applicability of ecosystem services for decision making, as outputs of ecological functions or processes that directly ('final ecosystem services') or indirectly ('intermediate ecosystem services') relate to human well-being (Fisher *et al.* 2009). For the purposes of this paper, we define an ecosystem services approach as connecting human benefits to ecological structure and function, allowing for quantification of positive and negative impacts of decisions, being as integrative and complete as possible in quantifying the scope of impacts, and including an economic valuation component.

Figure 3 illustrates the links between N sources, N cycling, ecosystem services and benefits to people. Others have reviewed the effects of increased N in the biosphere on ecosystem structure and function, for example nutrient cycling, plant production, greenhouse gas production, pests/pathogens, habitat and biodiversity (Vitousek *et al.* 1997; Driscoll *et al.* 2003). In turn, many of the effects on structure and function alter the production of ecosystem services such as the provision of food, clean air, clean water and materials, regulation of climate and UV protection, provision of habitat and biodiversity for recreation and human well-being. Changes in ecosystem services alter the benefits for people, influencing air for breathing, visibility, aesthetics, water for drinking and a host of other services.

Despite an increasing focus on the natural capital of ecosystems related to human needs (Costanza *et al.* 1997; Boyd & Banzhaf 2007), there are few examples of scientifically defensible accounting frameworks that can link natural capital to decision making (Daily *et al.* 2009; Jordan *et al.* 2010). We believe that the ecosystem services concept could be applied effectively to decisions surrounding pollutants like N because of similarities between regulatory objectives and ecosystem services. Current regulations related to N in air and water address the effects on 'public welfare' in the case of the Clean Air Act (1970) and 'designated use' in the case of the Clean Water Act (1972). Both of these concepts identify attributes of ecosystems that should be protected for the public good. Although the statutes predate the common use and definitions of the term 'ecosystem services,' they imply a similar concept. The Clean Air Act was established to protect the environment against air pollution, including adverse effects on 'soils, water, crops, vegetation, manmade materials, animals, wildlife, weather, visibility, and climate, damage to and deterioration of property, and hazards to transportation' (section 302h, Clean Air Act 1970). The goals of the Clean Water Act are to restore and maintain the chemical, physical, and biological integrity of the nation's waters 'which provides for the protection and propagation of fish, shellfish and wildlife and provides for recreation in and on the water' – often referenced as the requirement that waters be 'fishable' and 'swimmable' (Clean Water Act 1972). These statutes describing designated use and public welfare have existed for 40 years, but the science connecting ecological research, in terms of the ecosystem service supply, and human demands for ecosystem services is relatively new.

In this review, we argue that in order to manage N optimally and efficiently, an approach is needed that allows decision-makers to

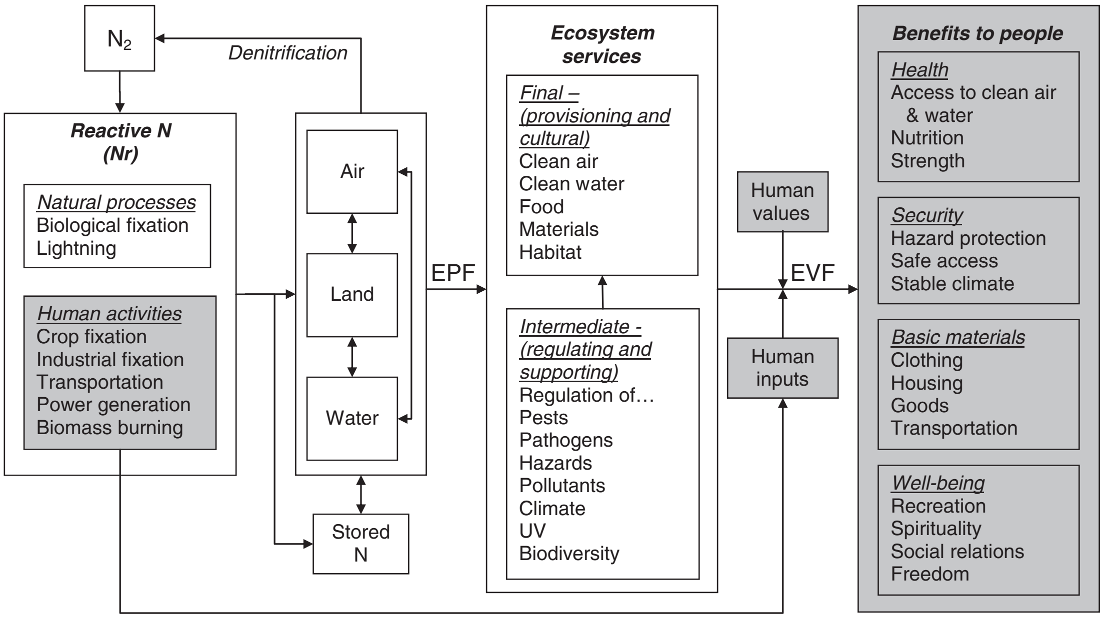

Figure 3 Diagram of the connections among reactive nitrogen inputs, ecosystem services and benefits to people. Clear boxes indicate ecological systems; grey shaded boxes represent human systems. EPF = ecological production function, EVF = ecological valuation function.

evaluate the effects of changing N management on a range of ecosystem services. We review the existing science connecting N and ecosystem services, and determine what information is available and what is still needed to undertake such an approach.

CONNECTING NITROGEN EFFECTS TO ECOSYSTEM SERVICES

Previous work has concluded that the science evaluating the connection between specific drivers and specific services is limited (Carpenter *et al.* 2009; Norgaard 2010). In this section, we connect the existing work on N-driven changes in ecosystem structure and function with ecosystem services. Overarching requirements that must be met in order for an effective accounting framework to be developed and implemented include:

(1) Ecological production functions that quantitatively connect ecological processes to a complete range of ecosystem services and human benefits (Fig. 4).
(2) Ecosystem services valuation functions that defensibly attach value to the damage costs per unit of N and the costs of abatement, restoration or replacement (Table 2).
(3) Monitoring and inventory methods that rapidly and defensibly track the status of ecosystem services in air, land and water.
(4) Knowledge about how nitrogen effects will interact with other projected changes such as land use, human populations and climate change.

An ecosystem services approach will enhance our capacity to assess the costs, benefits and tradeoffs associated with N-related management actions and policies.

In the following section, we focus on several (but by no means all) key ecosystem services that directly link to management of N in the environment: (1) food, fuel and fibre production, (2) climate regulation, (3) maintenance of human health and (4) maintenance of biodiversity and aesthetics. For each ecosystem service, we evaluate

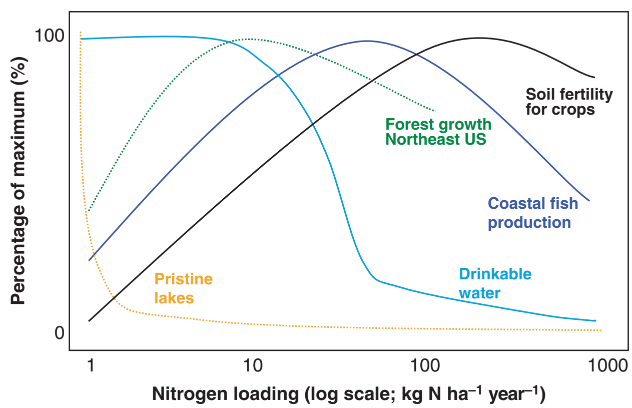

Figure 4 Ecological production functions linking nitrogen and ecosystem services (pristine lakes critical threshold for changes in algae – Baron 2006; Northeast forest growth – Thomas *et al.* 2010; Magill *et al.* 2004; fish production – Breitburg *et al.* 2009). Dotted lines represent air deposition N loads.

whether there is enough information to construct an appropriate ecological production function (the biophysical relationship between ecosystems and services; Daily *et al.* 2009). We also review attempts to examine benefits and costs of N increases on each service. Types of economic costs that N enrichment can incur include mitigation, damage, remediation and substitution costs (Moomaw & Birch 2005). Others have argued that monetary value should not be the only metric of ecosystem services within a defensible framework, in part because we do not yet have approaches to give monetary value to all relevant services (Toman 1998), and thus such a framework would be incomplete (Norgaard 2010). We maintain that economic valuation is useful because it is easily understandable by society and is a common unit that allows for simple stacking of services when comparing management options (Dodds *et al.* 2009; Birch *et al.* 2011). We identify available data that could support an ecosystem services approach to

Table 2 Damage costs of the N cascade per unit N. See Fig. 2 for link to mechanisms in the N cascade

Mechanism in cascade	Effect on services	Cost $ kg^{-1} N	Monetary values
PM and tropospheric ozone effects	Reduced visibility	$0.31	Birch *et al.* (2011)
	Human health costs of NO_x	$23.07	Birch *et al.* (2011)
	Human health costs of NH_y	$1.30–8.56	Birch *et al.* (2011)
	Crop declines from ozone	$1.51	Birch *et al.* (2011)
	Forest declines from ozone	$0.89	Birch *et al.* (2011)
Stratospheric ozone effects (N_2O)	UV damage – skin cancer and cataracts, crop production, polymers, water, corals	$1.33	This paper
Greenhouse gas effects (N_2O)	Anticipated damages of climate change	$1.24–3.10	Kusiima & Powers (2010)
Terrestrial acidification	Damage to buildings	$0.09	Birch *et al.* (2011)
	Damage to forest products	NE	
Freshwater acidification	Decline in recreational fishing	NE	
	Decline in aesthetics and value of lakes, streams	NE	
Freshwater eutrophication	Reduced lake waterfront property values	< $0.01	Dodds *et al.* (2009)
	Costs to recreational freshwater use	< $0.01	Dodds *et al.* (2009)
	Costs related to freshwater endangered species	< $0.01	Dodds *et al.* (2009)
	Cost of HABs (swimming and drinking)	NE	
Drinking water contamination	Purchases of bottled water because of eutrophication (odour and taste issues)	< $0.01	Dodds *et al.* (2009)
	Treatment for nitrate in drinking water wells	$0.16	This paper
	Health costs of nitrate in drinking water – colon cancer	$0.14–3.38	van Grinsven *et al.* (2010)
	Other health costs of nitrate in drinking water	NE	
Coastal eutrophication	Recreational use of estuary	$6.38	Birch *et al.* 2011
	Fisheries decline in Gulf of Mexico related to SAV loss from N loading and eutrophication	$56.00	S. Jordan, pers. comm.
	Beach closures due to HABs or fish kills (swimming)	NE	

NE, no estimate tied to N loading; PM, particulate matter; HABs, harmful algal blooms; SAV, submerged aquatic vegetation.

N management as well as critical knowledge gaps that could prevent making useful connections between ecosystem processes, ecosystem services, and valuation of such services. We assemble cost information where available as the metric cost per unit N, which is increasingly available from a number of recent studies (Kusiima & Powers 2010; Birch *et al.* 2011). If costs per unit N were not available, but we had total damage costs, we calculated this metric based on total damage costs (from Appendix S1) divided by N fluxes to the affected parts of the ecosystem. Finally we apply and compare these cost estimates within an example accounting framework to illustrate how it can inform decisions.

N and food, fuel and fibre production

One suite of ecosystem services that has been greatly enhanced by N addition to the environment is food, fuel and fibre production. Because ecosystems are often limited by N availability, N additions to soils and surface waters can markedly boost biological production in these systems. Within the past century intensive agricultural production has yielded tremendous increases in human nutrition and well-being, largely as a result of the invention and large scale implementation of the Haber-Bosch process for N fixation (Galloway *et al.* 2008). The development and accelerated use of nitrogen fertilizers has driven large increases in food production for both humans and animals in affluent nations, and has shifted the balance between malnutrition and an adequate diet for a huge number of people in developing nations (Smil 2002). Increases in N-based fertilizers and modern agricultural practices have more than doubled the number of people who were fed from a hectare of agricultural land managed with organic residues and N_2-fixers in the early 1900s (Evans 1980; Smil 2002).

The broad benefits of N fertilization on food and material production are well known, particularly for agriculture, but the damages to these services caused by increasing N in the environment are not as well understood. Several studies have quantified damage costs of N on food and fibre production. In Table 2, we focus on valuations of damages or benefits associated with mitigation, since remediation and substitution costs are only now becoming available for many systems (e.g. Jenkins *et al.* 2010; Birch *et al.* 2011). One of the most complete national analyses of N effects examined the consequences of US air pollution control policies (Chestnut & Mills 2005). Emissions of N and S oxides led to acidification and damage to materials that cost *c.* $133 million annually prior to the US Acid Rain Program, 1990 Clean Air Act Amendments (Chestnut & Mills 2005). Nitrogen oxides also contribute to ozone formation in the troposphere, which can reduce crop and forest production in ways that could offset any fertilization effects, particularly in areas where N loading is already high. Ozone reductions projected to result from the 1990 Clean Air Act Amendments were estimated to provide a total annual benefit to the US commercial timber industry of about $800 million, and improved yields were estimated to benefit grain crop producers by $700 million in 2010 (Chestnut & Mills 2005). Increases in N also fuel UV damages to crop production, fisheries and corals, since N_2O is currently the most important contributor to the breakdown of stratospheric ozone (Ravishankara *et al.* 2009). We discuss UV damages further in the section on human health.

In aquatic ecosystems, increasing N loads can stimulate production, particularly in estuaries and near-coastal waters, with mixed effects. At low N loading, fisheries may be limited by N, whereas increasing N loads can lead to eutrophication, hypoxia, and anoxia with the potential to reduce fish production (Fig. 2; Breitburg *et al.* 2009). Also, the desirability of enhanced production of any given species is somewhat variable: for example, greater algal production could ultimately lead to fish kills; atmospheric N loading could stimulate the production of undesirable or exotic species (e.g. Suding *et al.* 2004) leading to questions about how various increases in production should be valued. Despite these complexities, greater understanding of how to value the net benefits or detriments of N loading to the environment would contribute significantly to our understanding and ability to implement an ecosystem services approach to management of the environment and natural resources.

Harmful algal blooms (HABs) and fish kills linked to N or other nutrients have caused substantial losses to the seafood industry. Whitehead *et al.* (2003) estimated that the lost consumer surplus due to a dinoflagellate (*Pfiesteria* sp.) related fish kill is between \$37 million and \$72 million in the month following a fish kill. Jordan *et al.* (unpubl. data) provide a more comprehensive estimate of the damage costs of eutrophication on fisheries production by estimating the damage to fisheries via reductions in the area of submerged aquatic vegetation (SAV) along Mobile Bay (Gulf Coast of USA). They estimate that a 20% loss of SAV damage cost in 2008 dollars to combined shrimp and crab fisheries is \$764 ha^{-1} $year^{-1}$ per unit SAV habitat. Using an empirical response function of the impacts of N loading on SAV extent (Latimer & Rego 2010), a 20% loss in SAV due to N would have an impact on crab and shellfish production of *c.* \$56 per kg N (S. Jordan, pers. comm.). Production of shrimp and crabs in Gulf estuaries is large and sensitive to habitat loss (Jordan *et al.* 2009), and damage to this valuable fishery is one of the highest per kg N damages we identified (Table 2).

N and climate regulation

Nitrogen plays a key role in the maintenance of a stable climate, a crucial regulating ecosystem service, by influencing the production of several greenhouse gases (N_2O, CO_2 and CH_4) and through its role as a mediator of aerosol production. Human alteration of the N cycle affects Earth's climate system via direct and indirect pathways. Nitrogen availability provides a fundamental constraint on plant growth and net CO_2 uptake across much of the world, now, and in response to rising atmosphere CO_2 concentrations in the future (Hungate *et al.* 2003). As discussed above, N inputs from atmospheric deposition can enhance plant growth rates and may account for a significant fraction of current terrestrial C uptake in some systems (Liu & Greaver 2009; Thomas *et al.* 2010). Furthermore, additions of N to some soils can inhibit decomposition, slowing release of CO_2 to the atmosphere and leading to an increase in soil C stocks (e.g. Janssens & Luyssaert 2009).

However, net greenhouse benefits of C storage by some ecosystems may be somewhat dampened by the production of other greenhouse gases. In a meta-analysis, nitrogen additions were found to stimulate CH_4 production, decrease CH_4 uptake and increase N_2O production (Liu & Greaver 2009). Atmospheric N_2O concentrations are increasing rapidly in response to N enrichment of terrestrial and aquatic systems, and are presently 16% greater than during pre-industrial times (Forster *et al.* 2007). Due to high per-molecule warming potential, small changes in N_2O concentrations have a disproportionately large effect on the climate system. N enrichment directly increases N_2O production by stimulating nitrification, the oxidation of ammonium to nitrate (Robertson & Tiedje 1987), and denitrification (Seitzinger *et al.* 2006). N_2O is a byproduct of both of these microbially mediated transformations. N availability also affects the rate of N_2O production, both by increasing the overall rate of each N transformation process and by affecting the fraction of nitrification or denitrification that produces N_2O rather than nitrate or N_2 (Beauchamp 1997). The net effect of N enrichment on CH_4 emissions is a function of competing processes. Atmospheric NO_x and resulting ozone maintain high concentrations of hydroxyl in the atmosphere, which serves to remove atmospheric CH_4 (Isaksen *et al.* 2009). And in anaerobic soils, an abundance of nitrate can decrease rates of CH_4 production by increasing soil and sediment redox potential (Reay & Nedwell 2004).

Nitrogen also influences the climate system through its link to ozone. In the lower atmosphere, N plays a key role in tropospheric ozone production (Skalska *et al.* 2010). In turn, ozone affects the climate system directly by acting as a greenhouse gas with roughly double the climate effect of N_2O (Forster *et al.* 2007), and indirectly through effects on photosynthesis and plant uptake of atmospheric CO_2. Ozone damage to plants, as discussed earlier in the section on production, also may decrease plant uptake of atmospheric CO_2 by as much as 14–23% (Sitch *et al.* 2007), leading to more CO_2-driven warming.

In addition to affecting the balance of greenhouse gases in the atmosphere, production of NO_x and NH_y increases the concentrations of atmospheric aerosols, which aside from their negative health effects can provide substantial cooling, both directly (due to high reflectivity) and indirectly (by mediating cloud formation). Sulphate aerosols and nitrate aerosols act similarly in these processes, with the role of nitrate aerosols expected to increase in the future (Adams *et al.* 2001).

The influence of reactive N continues into the upper atmosphere, where ozone acts to provide a small amount of cooling. In this portion of the atmosphere, N_2O currently is the most important contributor to the breakdown of stratospheric ozone, both now and in future projections (Ravishankara *et al.* 2009). Regulatory actions stemmed the production of CFCs that were formerly the dominant driver of depletion of the protective stratospheric ozone layer, but N_2O production has continued to increase. Thus, N_2O is currently the dominant and largely unregulated driver of UV-related damages to ecosystems and human health. The global benefits of the Montreal Protocol in reducing the use of ozone-depleting chemicals were estimated to be \$300 billion (2008 dollars) for the period 1987–2060, and this did not include the human health benefits, such as 333,500 avoided skin cancer deaths (Smith *et al.* 1997a,b). We were not able to obtain damage costs to individual services, but collective UV damages associated with CFCs are estimated to be \$49,669 per metric ton (Talberth *et al.* 2006). The ozone-depleting potential of N_2O is *c.* 0.017 relative to CFCs (Ravishankara *et al.* 2009) so damages would be \$844 per metric ton of N_2O. Based on these values, potential UV-related damages related to N_2O production in the USA are *c.* \$1.33 kg^{-1} N_2O-N.

Clearly N has the potential to modulate the ecosystem service of climate regulation. However, the relative importance of various N effects on climate is poorly understood, as are interactions between effects. Birch *et al.* (2011) were not able to find economic valuation functions to monetize the effects of N on greenhouse gases and climate regulation in their analysis of the effects of decision about N management in the Chesapeake Bay watershed. Recently, Kusiima & Powers (2010) identified several efforts to provide preliminary values of the anticipated impacts of greenhouse gases of *c.* \$4–10 per ton of CO_2, equivalent to \$1.2–3.1 per kg N.

More research is clearly needed to elucidate interactions between N enrichment and climate at multiple scales and in multiple systems. In order to implement an ecosystem services approach to managing N with respect to climate influences, one would need to understand the relative magnitude of different N effects on the climate system, as well as gain an understanding of interactions between various N effects, dominant feedback mechanisms and thresholds. In addition, one would need a way to value the climate regulating properties of N in a manner that made it possible to compare the worth of such services to the value of other N-related ecosystem services. Consider the net greenhouse gas implications of N reduction efforts. Wetland and riparian restoration may be conducted in order to reduce nutrient loading and eutrophication of surface waters, but these activities have the added benefit of substantial carbon sequestration and the cost of additional greenhouse gas production (CH_4 and N_2O). Jenkins *et al.* (2010) determined that existing markets yield an estimate of \$70 ha^{-1} for wetlands in the Mississippi River alluvial valley (USA), but when accounting for additional benefits such as nitrogen mitigation, waterfowl recreation and other valued services, the wetland value estimate rose to \$1035 ha^{-1}. A framework that included a full accounting of different N reduction strategies and net benefits would allow for more optimal and efficient N management.

N and maintenance of human health

Tremendous benefits to human health and well-being have resulted directly or indirectly from human alteration of the N cycle, particularly in terms of nutrition, materials (e.g. wood, paper, fabric), and provision of heat, light and transportation. Many of these positive impacts are quite evident, and can be tracked through economic indicators. However, when N is transported downwind and downstream from sites where its use is primarily beneficial to humans, it can become a hazard to human health (Townsend *et al.* 2003). These detrimental impacts are more challenging to track and do not correlate with the benefits (Raudsepp-Hearne *et al.* 2010). In the atmosphere, NO_x is an important precursor of tropospheric ozone and particulate matter, which can increase rates of asthma and other respiratory issues, particularly in children and other vulnerable populations (Delucchi 2000).

The provision of clean water for drinking and other domestic uses is a key ecosystem service, and unfortunately nitrate contamination in drinking water is a growing issue in the USA. The number of drinking water violations of the nitrate standard in community drinking water wells increased from *c.* 650 to 1200 between 1998 and 2008 (US EPA 2009). Excess nitrate in drinking water has been associated with a number of illnesses, including blue-baby syndrome and several types of cancers (Townsend *et al.* 2003; Ward *et al.* 2005), although there is disagreement in the literature on these points (Powlson *et al.* 2008). Communities across the USA are dealing with nitrate contamination in drinking water, and making choices between replacement, treatment and prevention. Many of these choices will be based on costs and tradeoffs between ecosystem services.

In addition to direct effects from N enrichment of air and drinking water, excess N in surface waters can also have indirect effects on human health through, for example, stimulation of HABs that produce toxins (Camargo & Alonso 2006), outbreaks of dangerous pathogens like *Cryptosporidium*, or simply unpleasant odours and tastes that are costly to remove. There is also some suggestion that N enrichment can exacerbate pathogens such as West Nile virus, pollen allergens, swimmer's itch, malaria, and cholera (Townsend *et al.* 2003; Johnson *et al.* 2010). Even where nitrate concentrations are below the US EPA drinking water standard (10 mg nitrate-N L^{-1}), nitrate and eutrophication can increase treatment costs of safe drinking water. Some treatment processes designed to remove the products of eutrophication can introduce harmful byproducts into drinking water (Cooke & Kennedy 2001).

The costs of human health problems related to N have been evaluated in a number of studies. In a detailed review of the valuation of air quality regulations on humans and ecosystems, the mortality and illness associated with reactive N forms as precursors to PM and ozone were the most substantial of the measured effects (Table 3; Chestnut & Mills 2005). A number of US and EU studies have also examined the cost of NO_x and NH_y effects on respiratory health; most recently the ExternE project determined the health impacts of reactive N in air to be \$28 per kg of NO_x-N and \$16 per kg NH_3-N

Table 3 Abatement costs of reducing nitrogen from various individual sources and from integrated projects. For comparison, the price of N fertilizer was *c.* \$0.44 per kg N (1980–2000) and in 2008 was *c.* \$1.21 per kg N (Bruulsema & Murrell 2008)

Cost	\$ kg^{-1} N	Location	Reference
By source			
Electric utilities/NO_x	\$4.80	Chesapeake Bay, USA	Birch *et al.* (2011)
Industrial/NO_x	\$22.00	Chesapeake Bay, USA	Birch *et al.* (2011)
Mobile sources	\$14.00	Chesapeake Bay, USA	Birch *et al.* (2011)
Non-agricultural/NH_3	NE	Chesapeake Bay, USA	Birch *et al.* (2011)
Agriculture/NO_3	\$10.00	Chesapeake Bay, USA	Birch *et al.* (2011)
Urban and mixed land use/NO_3	\$96.00	Chesapeake Bay, USA	Birch *et al.* (2011)
Point Sources	\$18.00	Chesapeake Bay, USA	Birch *et al.* (2011)
Agricultural drainage water/NO_3	\$2.71	Mississippi Basin, USA	Jaynes *et al.* (2010)
Integrated plans			
Current expenditures towards meeting Chesapeake TMDLs – (1985–2009)	\$8.76	Chesapeake Bay, USA	US EPA (2009); Blankenship (2011)
Projected costs to meet Chesapeake TMDLs – (2010–2025)	\$14.27	Chesapeake Bay, USA	US EPA (2009); Blankenship (2011)
Projected costs of using wetlands to control nutrient damages	\$4.40–5.62	Mississippi Basin, USA	Kusiima & Powers (2010)
Estimated cost for achieving a 45% reduction in nitrate-N load	\$2.50	Cedar River Watershed, Iowa, USA	Helmers & Baker (2010)

NE = no estimate.

(Bickel & Friedrich 2005). The health impacts for NH_3-compounds are uncertain, and costs could be lower (van Grinsven *et al.* 2010). These values are similar to estimates used in a Chesapeake Bay N assessment (Table 2; Birch *et al.* 2011).

Few studies comprehensively address all impacts of N on drinking water and human health, but a number of pieces of the puzzle are available. A study using limited data on the link between colon cancer and nitrate determined a health cost of \$0.1–3.4 per kg N leaching to groundwater in the EU (van Grinsven *et al.* 2010). Several studies have examined the impacts of HABs in coastal areas, which may be associated with N (Table 2). Hoagland *et al.* (2002) found that impacts of illnesses resulting directly from shellfish poisoning in the USA were difficult to estimate but used mortality, hospital visits and lost worker hours to estimate that one paralytic shellfish poisoning event cost *c.* \$6 million. Corso *et al.* (2003) found that the total cost of a single *Cryptosporidium* outbreak during 1993 in Milwaukee, Wisconsin (USA) was \$96.2 million: \$31.7 in health costs and \$64.6 million in lost worker productivity. We could not obtain or develop damage cost per kg N estimates for health effects of harmful algae or pathogens, in part because the causes are sometimes unclear. More research is needed to better test exposure-response relationships and the transferability of the limited number of these relationships (van Grinsven *et al.* 2010).

Another way to look at the problem is to consider the cost to treat contaminated water. Approximately 15% of the US populations or 45.4 million people use water from private wells for domestic use (Hutson *et al.* 2004), and a recent survey indicated that 4.4% of private drinking water wells in the USA were higher than the US EPA nitrate standard for human consumption (DeSimone 2009). Based on these estimates, *c.* 2 million people are on well water containing nitrate above the US EPA standard for human consumption. If it costs \$560 per person to treat well water (US EPA 2009), then we estimate that the cost to treat nitrate contaminated well water is *c.* \$1.12 billion dollars. We multiply this number by the *c.* 7 Tg of nitrogen that moves from land to water in the USA and [value of 23 Tg inputs to land from Fig. 1 and assuming that transport to groundwater is equivalent to the transport factors of 0.3 from Smith *et al.* (1997a), yielding an estimate of *c.* \$0.16 per kg N for groundwater contamination of drinking water. This value is greater than the estimate of Dodds *et al.* (2009), indicating a need for further review of these damage costs. Our review here indicates that there are tremendous health impacts and consequences of nitrogen pollution, and including the full range of these impacts, not the just the well-studied impacts in air, will better inform decisions related to the management of N.

N and maintenance of biodiversity and aesthetics

Excess N can affect the integrity, resilience and beauty of the natural world by reducing biodiversity. This loss of biodiversity can occur through a number of different mechanisms. N additions can cause shifts in primary producer communities in both terrestrial and aquatic systems, leading to decreased biodiversity (e.g. Deegan *et al.* 2002; Dupré *et al.* 2010). A recent global analysis further supports the notion that N deposition is the main driver of altered species composition in a range of ecosystem types and in some cases this includes an increase in invasive species (Bobbink *et al.* 2010). Species that tend to show increases in abundance are often non-native invasives with high vegetative and population growth rates, which have the potential to drive local populations of rare native species to extinction (Bobbink *et al.* 2010). In some, but not all types of wetlands, increased productivity is associated with decreased plant diversity (Bedford *et al.* 1999); moreover, rare or more ecologically valuable species may be replaced by generalists and invasive species (Morris 1991). Furthermore, the loss of plant species due to N deposition can be detrimental to insect herbivores that depend on them, as exemplified by checker spot butterflies in serpentine grasslands of California (Weiss 1999). The provision of habitat for organisms which influence the integrity, resilience, spiritual value and beauty of the natural world is an important service (e.g. Losey & Vaughan 2006).

Ecosystem acidification via atmospheric N deposition is another driver of species changes. Following deposition, nitrate can leach out of soils, carrying with it a loss of base cations (K, Ca, and Mg). Soil acidification can also lead to mobilization of inorganic Al (Reuss 1983) with detrimental effects on tree health, including aluminium interference with calcium uptake, cold tolerance, and aluminium toxicity to roots (Parker *et al.* 1989; Cronan & Grigal 1995). These leaching processes usually result in lower pH in soil solution and streamwater, and higher concentrations of inorganic monomeric Al. Low pH and inorganic monomeric Al are directly toxic to fish (Baker & Schofield 1982), and fishless lakes in the Adirondacks have significantly lower pH and acid neutralising capacity than lakes with fish (Gallagher & Baker 1990). Leaching of Al from soils into sensitive aquatic systems also has been shown to reduce fish diversity (Nierzwicki-Bauer *et al.* 2010). These shifts in fish abundance and diversity have implications for sport fishing and recreation, as well as cultural and existence values (Banzhaf *et al.* 2006).

High rates of N loading to surface waters can contribute to excessive productivity, or eutrophication, characterized by algal blooms that prevent swimming, fish consumption and/or other human use (Van Dolah 2000), hypoxia (Breitburg *et al.* 2009), shifts in species composition (Vaas & Jordan 1990) and food webs, and water with unpleasant tastes and odours (Pretty *et al.* 2003). These factors negatively affect fish production, biodiversity, water quality, recreation potential, aesthetics and human health. Dodds *et al.* (2009) conservatively estimate that the costs of freshwater eutrophication, including costs to recreation, waterfront real estate, and spending on recovery of threatened and endangered species in the USA are *c.* \$2.2 billion per year.

Nitrogen can also influence how humans experience nature. Nitrogen is a component of regional haze, which can affect visibility and decrease aesthetic enjoyment of places where people live, work and recreate, including parks and other rural areas (Malm 1989). Visibility damages associated with reactive N in the Chesapeake Bay watershed were \$120 million (Birch *et al.* 2011). Damage costs of HABs to recreation and tourism range from < \$1–28 million (Hoagland *et al.* 2002). Damages by reactive N to recreational use within the Chesapeake Bay estuary were estimated to be \$730 million per year (Birch *et al.* 2011).

Some studies have estimated the value of improving the quality of natural resources by asking people what they are willing to pay. Banzhaf *et al.* (2006) estimated that New York state residents are willing to pay \$45–100 each year to reduce the number of acidified lakes and improve forest health in the Adirondacks, which translates to \$300–700 million for all state residents. A key challenge to this approach was that the effects needed to be explained to and understood by the respondents. Thus, in addition to accounting for human well-being in such a decision framework, an effort must be made to reach out to and educate the public to ensure that they are aware and have sufficient knowledge about the connections between N reductions and benefits.

FROM THEORY TO PRACTICE: A DEFENSIBLE ECOSYSTEM SERVICE ACCOUNTING FRAMEWORK FOR DECISION MAKING RELATED TO N

Many challenges confront social and natural scientists in creating ecosystem service accounting systems that can be used to inform decisions. Our understanding of the connections between ecological processes, social needs and ecosystem services is improving rapidly, but we need accounting measures and databases that can be used to estimate service production in relation to a range of biophysical drivers (Daily & Matson 2008). Ecological production functions describing the linkages between human actions, biophysical factors and ecosystem services must be a component of such an accounting system but such functions are largely missing at present. These ecological production functions (e.g. Fig. 4) can be used to predict changes in the amount, quality and supply of ecosystem goods and services based on the ecosystem features and biophysical inputs driven by natural and human events (Wainger & Boyd 2009).

A defensible accounting framework for ecosystem services could inform decision making concerning the effects of a decision on a range of ecosystem services and human benefits (Daily *et al.* 2009; Sutton *et al.* 2011). An important goal of this framework should be to include a wide range of effects on ecosystem services and human benefits in order to avoid unintended consequences associated with focusing on a limited set of services or factors. Indicators and measures of ecosystem services that can be scaled and applied across a management area or ecosystem service provisioning region are integral to the utility of this approach, and must be constructed with care.

We propose that cost per unit of nitrogen (Table 2) is a good metric for comparing the relative importance of damage costs, as well as mitigation or restoration costs associated with a particular N source. A number of recent studies present costs using this metric, allowing us to compare values obtained in the different studies and test these metrics. The European Nitrogen Assessment recently estimated that excess nitrogen costs the people of Europe between \$100 and \$500 billion (Sutton *et al.* 2011). Birch *et al.* (2011) conducted an assessment of the costs associated with N in the Chesapeake Bay. Below we describe this example, to illustrate many of the components of an ecosystem services approach using the metric of a cost per unit N.

Moving from theory to practice: the economic nitrogen cascade for Chesapeake Bay

Few efforts comprehensively track the interactions between N and human benefits. Birch *et al.* (2011) attempt such a comprehensive examination of the effects of N on health and environmental endpoints for the Chesapeake Bay watershed, using economic valuation in terms of damage cost per ton of N as the common metric (Table 2). This effort to characterize an economic N cascade was able to place values on many endpoints, for example reduced recreational and residential visibility, mortality, hospitalization and work loss caused by particulate and ozone exposure, materials damage via corrosion, loss in agricultural productivity due to ozone exposure, reduced crab fisheries and impaired recreational use (Birch *et al.* 2011). Different sources of N do indeed have different impacts per unit N (Table 2), and the costs to reduce N coming from these different sources are not equal (Table 3). Birch *et al.* (2011) argue that the magnitude of N flux is not necessarily equivalent to its impact to society. Understanding the effects of different N sources to Chesapeake Bay watershed can inform the public and decision-makers about the trade-offs and integrated benefits that are more closely tied to their priorities, thereby supporting better, more cost-effective, and ultimately more sustainable policies that both reduce N and optimize N-related services. Almost as valuable as the information about what could be valued is what Birch *et al.* (2011) could not value. These effects included greenhouse gas increases, fertilization benefits, freshwater recreational fishing and other ecosystem services throughout the cascade. They explicitly illustrate where they could not find information on the damage costs, leaving room for improvement and future work.

The analysis by Birch *et al.* (2011) serves as a model approach because it ties the approaches for reductions to the benefits. They illustrate that the choice of intervention used to achieve N reduction has distinct consequences for ecosystem services and benefits to people. When considering N loading to the Chesapeake Bay watershed, N deposition is not the largest source. Yet the currently available damage costs associated with atmospheric N emissions are much greater than the other measured costs, due primarily to the high value placed on damages to human health associated with particulate matter and ozone, that is, mortality and hospital visits due to respiratory illness. Air related effects were greatest in this analysis, in part, because the cost data are available. Future improvements should attempt to include other costs, for example those associated with ozone-depletion, climate change and freshwater costs, as we have done in this paper (Table 2). Quantification of ecosystem services can help decision-makers evaluate where we can best spend our limited restoration and abatement dollars.

Moving from theory to practice: Science needs

Carpenter *et al.* (2009) identified a number of data gaps in the science related to ecosystem services and sustainability, in particular related to biodiversity. They call for improved monitoring of ecosystem services, which requires '(1) time series information on land cover and land use, (2) locations and rates of desertification, (3) spatial patterns and changes in freshwater quality, (4) stocks, flows and economic values of ecosystem services, (5) trends in human use of ecosystem services and (6) trends in components of human well-being (particularly those not traditionally measured)'. These monitoring needs also apply to nitrogen effects.

In order to understand and manage N and N-associated ecosystem services, it is first necessary to understand both natural patterns of N delivery to ecosystems and how humans have altered this delivery. A number of tools and approaches have been developed to accomplish this goal. National and regional datasets of N fertilizer consumption and application rates (Ruddy *et al.* 2006), a network of N deposition sites (National Atmospheric Deposition Program 2009), and estimates of livestock manure production have all been used to estimate spatial distribution of N inputs in the USA. In addition, a number of models have been developed and applied to estimate fertilizer N loading (EPIC), atmospheric N deposition (CMAQ; Schwede *et al.* 2009), N from sewage discharge (Van Drecht *et al.* 2009), N fixation in both crops and natural ecosystems, and N loading to surface freshwaters and the coastal zone (e.g. Smith *et al.* 1997a). This flux information, in combination with information about ecosystem service production and valuation associated with N could be used to support an ecosystem services-based approach to N management.

There are also science needs related to damage and abatement costs. Birch *et al.* (2011) illustrate the data needs for damage costs in the Chesapeake Bay. Information about restoration and abatement costs is also needed. Recent efforts indicate that the abatement costs of reducing N in agricultural drainage waters is one of the lower cost options (Table 3), and costs are much lower than damage costs (Table 2). Both damage and abatement costs are presented as static costs, and presumably, costs would increase incrementally with N load or with the fraction of the N load actions are designed to remove. There may be important ecological thresholds affected by Nr loading which could relate to rapid and persistent losses of valued ecosystem services, for example some organic rich coastal salt marshes are slowly degrading as a result of high nitrogen loading (Wigand 2003), making these systems more susceptible to sea level rise, erosion, and the loss of the service of coastal storm protection. Better models coupling N fluxes to ecosystem services are needed, highlighting a pressing need for the development of simple, yet still realistic, modelling tools that can bridge the interface between N cycle components, ecosystem services and valuation.

SUMMARY

An ecosystem services approach to evaluating costs and benefits associated with N mitigation can better inform integrated policy and management of N in air and water in the USA because it allows for a more complete presentation and analysis of the effects of particular N sources and forms on public benefits than is currently used, in a manner consistent with existing clean air and water regulation. Economic valuation is easily understandable by society and the metric would be equivalent across services, allowing for simple stacking of services when comparing management options. We propose that cost per kg N (Table 2) is a good metric for comparing the relative importance of damage costs as well as mitigation or restoration costs associated with a particular N source. One limitation of the cost per kg metric as currently conceived is that it is a static value, but this could change if N loading or proximity to a threshold were incorporated into calculations of damage and mitigation costs. Improved development of production functions describing the linkages between human actions, biophysical factors, ecosystem services and economic values must be a component of a nitrogen-related ecosystem services accounting system and constitutes an important and exciting area for future research.

Our synthesis of N-related ecosystem services and their associated monetary value reveals that there is still scant information on many N related services. Even though we have not been able to quantify all the impacts of N, the available estimates indicate that damage costs outweigh the costs associated with reducing N loading. This provides a strong rationale for mitigation of N pollution and the associated effects on ecosystem services. The fact that these initial estimates (Table 2) are incomplete means that our analysis almost certainly underestimates the societal benefits to mitigating the negative effects of nitrogen pollution.

We anticipate that additional insights and refinements will enhance the utility of an ecosystem services approach to N management, and thus efforts to develop this approach should continue to move ahead with cautious optimism, while ensuring opportunities for adaptation as new and better information is made available. In addition, because value is directly influenced by society's perception and preferences, and because the success and sustainability of a policy is dependent upon the adoption by decision-makers, managers, policy-makers and the public should be engaged in the definition and valuation of important ecosystem services for a service-providing area. Finally, our synthesis indicates that there is a growing body of information to provide monetary valuation of ecosystem services, and that this information has great potential to help decision-makers evaluate where to best spend our limited restoration and abatement dollars for better N management.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Appendix S1 Valuation of the nitrogen cascade – includes damage costs, benefits and avoided damage costs of mitigation. 'NA' indicates that the study noted information was not available to estimate the monetary the value of this services.

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