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Evidence Library for Nutrient Management Practices on Canadian Croplands

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http://bit.ly/NI-ES

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INTRODUCTION

Nutrients originating from agricultural land contribute to a variety of environmental issues, such as algal blooms in lakes, hypoxic zones in estuaries, poor air quality, and greenhouse gas emissions. Improved nutrient management practices can improve retention of fertilizers where they are needed and reduce runoff from agricultural fields into bodies of water and other ecosystems.

To summarize the environmental, social, and economic impacts of improved nutrient management practices, we developed ecosystem services conceptual models that illustrate the relationships between changes to management practices and the ecological and socioeconomic systems (Olander et al. 2018). We also compiled evidence from the academic literature for each relationship shown in the conceptual models, summarized here in an evidence library.

The conceptual models and evidence library are designed to serve as a resource for managers, decision-makers, and others working to improve agricultural sustainability. While the conceptual models were designed for the Canadian agricultural context, many of the relationships and effects they illustrate are applicable for agricultural systems in other areas, in particular large-scale agriculture in the United States. Similarly, the evidence library uses literature conducted in Canada where available, but in many cases literature from the United States is included as well.

Photo Credt: Paul Hamilton

Ecosystem service conceptual models for nutrient management

Two ecosystem service conceptual models (ESCMs) for nutrient management practices – one focused on nitrogen management and one on phosphorus management – were developed based on a literature review and with feedback from experts. The nutrient management practices included in the conceptual models are from the 4R method: changes in the type, rate, timing, and placement of fertilizer.

The ESCMs on pages 3 and 4 show the cascade of changes that these nutrient management practices (dark blue box) cause in the biophysical and social systems (gray boxes), leading to effects on ecological outcomes (green boxes), human activity outcomes (light blue boxes) and socioeconomic outcomes (yellow boxes). Ecological outcomes are components of the ecological system that are commonly measured, such as dissolved oxygen. The human activity outcomes shown in the ESCMs can influence socioeconomic outcomes as well; these links are not included in the ESCMs due to the nature of the project for which these models were created. For example, the recreational activity outcomes (boating, fishing, and swimming) can lead to jobs and local revenue, which are socioeconomic outcomes not shown in the ESCMs.

Many of the relationships in the nitrogen management ESCM (page 3) also appear in the phosphorous management ESCM (page 4). These common relationships are shown as teal arrows in the phosphorous management ESCM.

There are many other approaches to the management of nutrients from agricultural sources that are not within the scope of these ESCMs, including vegetated buffer strips, protected riparian zones, constructed wetlands, conservation tillage, and crop residue management (Hart, Quin, and Nguyen 2004). In addition, because phosphorus adsorbs to soil particles, it can build up in agricultural soils and remain for long periods of time, slowly moving into the environment via soil erosion, runoff, and subsurface flow. This “legacy phosphorus” is an important source of excess phosphorus in ecosystems and can dampen the effects of phosphorus management on downstream water quality (Sharpley et al. 2013). However, the phosphorous management ESCM and associated evidence library focus on the effects that improved phosphorus fertilizer management can have on ecological and agricultural systems via reductions in incidental losses of phosphorus soon after fertilizer is applied. Legacy effects are not directly addressed.

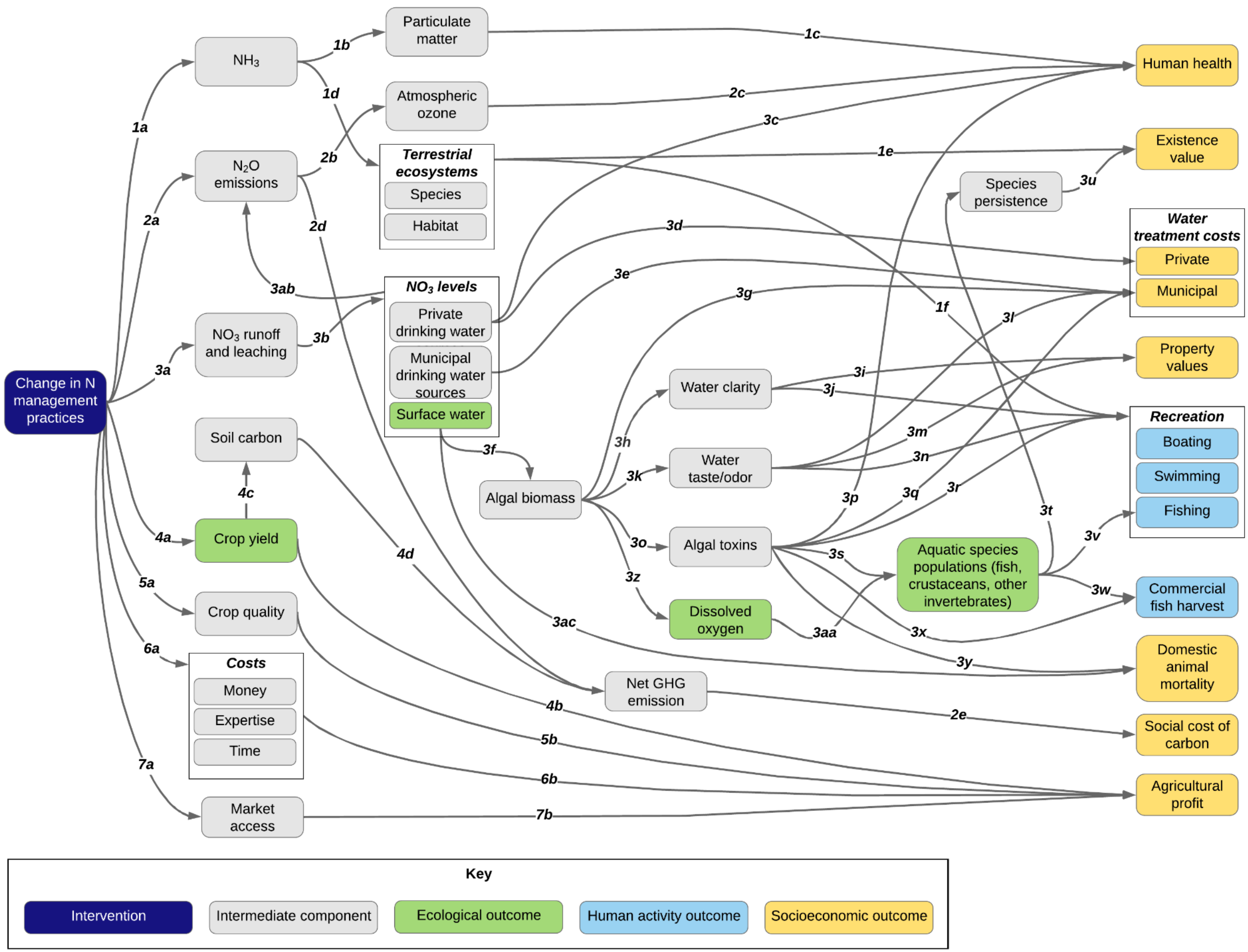
Sources

Hart, M. R., B. F. Quin, and M. L. Nguyen. 2004. Phosphorus runoff from agricultureal land and direct fertilizer effects: a review. *Journal of Environmental Quality* 33 (6):1954-1972.

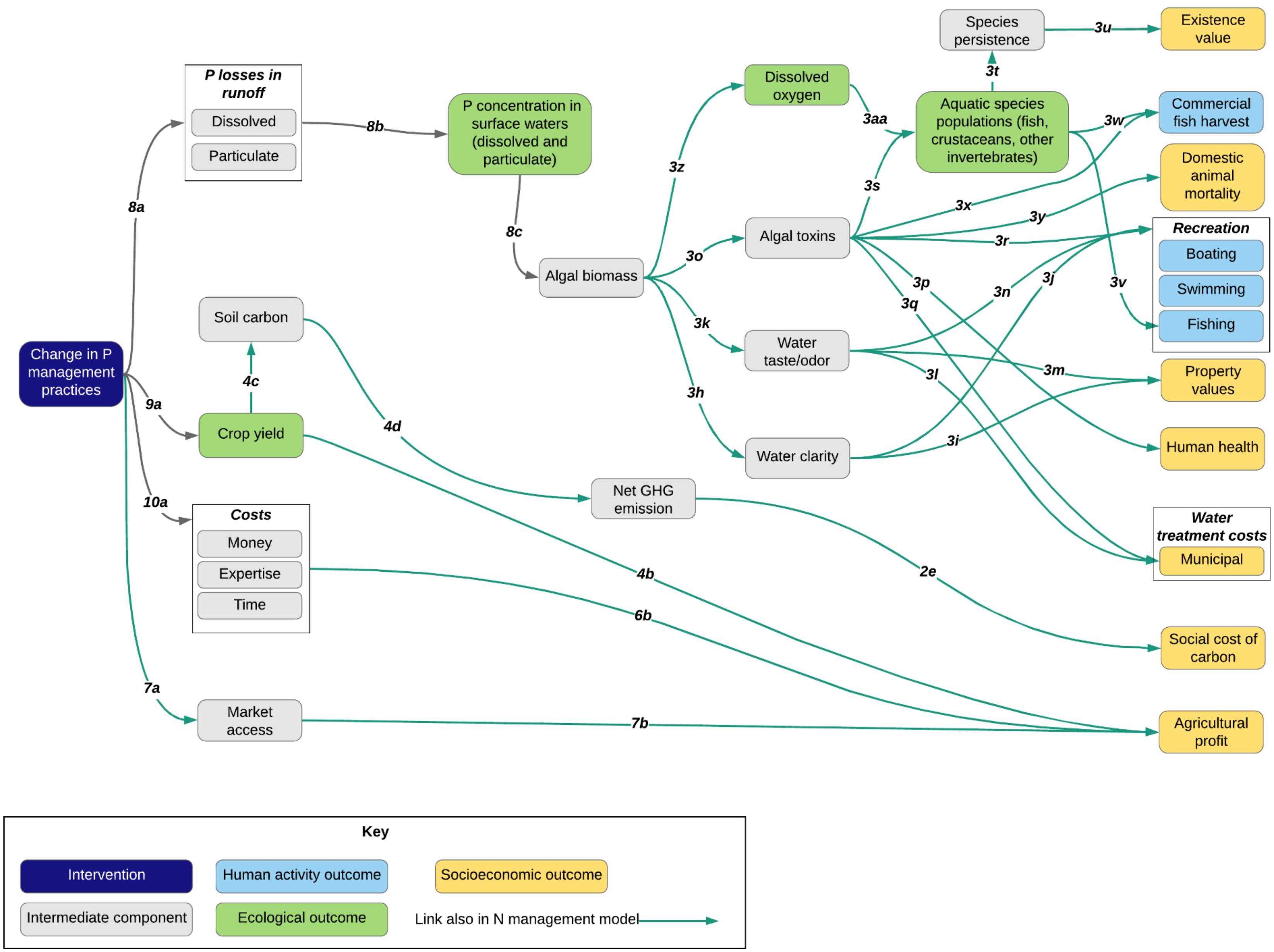
Olander, L., S. Mason, K. Warnell, and H. Tallis. 2018. Building Ecosystem Services Conceptual Models. In *National Ecosystem Services Partnership Conceptual Model Series*. Durham, NC: Duke University.

Sharpley, A., H. P. Jarvie, A. Buda, L. May, B. Spears, and P. Kleinman. 2013. Phosphorus legacy: overcoming the effects of past management practices to mitigate future water quality impairment. *Journal of Environmental Quality* 42:1308-1326.

Ecosystem service conceptual model for nitrogen management practices



Ecosystem service conceptual model for phosphorus management practices



Evidence library for oyster reef restoration ESLM

The evidence library starting on page 5 contains summaries of the evidence for each of the links in the ESCMs. Each link in the ESCMs has an identification number. To find the evidence library entry for a particular link, use the search function (keyboard shortcut Control + F) and search for "Link #" (e.g. "Link 3a"). The literature review for the evidence library was primarily conducted in 2017 and 2018; many of the topics covered are active areas of scientific investigation, so it is likely that new evidence is available for some of the relationships.

Each evidence library entry has the following components:

Description of relationship

Short description of the relationship between the starting and ending nodes (boxes), based on the evidence found. For links starting from the intervention node, there is a separate section for each of the four components of nutrient management: fertilizer source, fertilizer application rate, fertilizer application timing, and fertilizer placement.

Summary of evidence

Overview of the evidence found to support the relationship, including the types of methods used, geographic location, applicability to the Canadian context, and major conclusions.

Strength of evidence

Rating of the overall strength of evidence for the relationship, based on the following criteria, with a short explanation. For links starting from the intervention node, there is a separate section for each of the four components of nutrient management. The strength of evidence ratings for each model are summarized in the strength of evidence maps on pages 7 and 8.

Where applicable, models and tools that can be used to predict the relationship are noted in this section under a "Predictability" heading.

Confidence level	Criteria			
	Types of evidence	**Consistency of results**	**Methods**	**Applicability**
High	Multiple	Direction and magnitude of effects are consistent across sources, types of evidence, and contexts	Well documented and accepted	High
Moderate	Several	Some consistency	Some documentation, not fully accepted	Some
Fair	A few	Limited consistency	Limited documentation, emerging methods	Limited
Low	Limited, extrapolations	Inconsistent	Poor documentation or untested	Limited to none
None	None	N/A	N/A	N/A

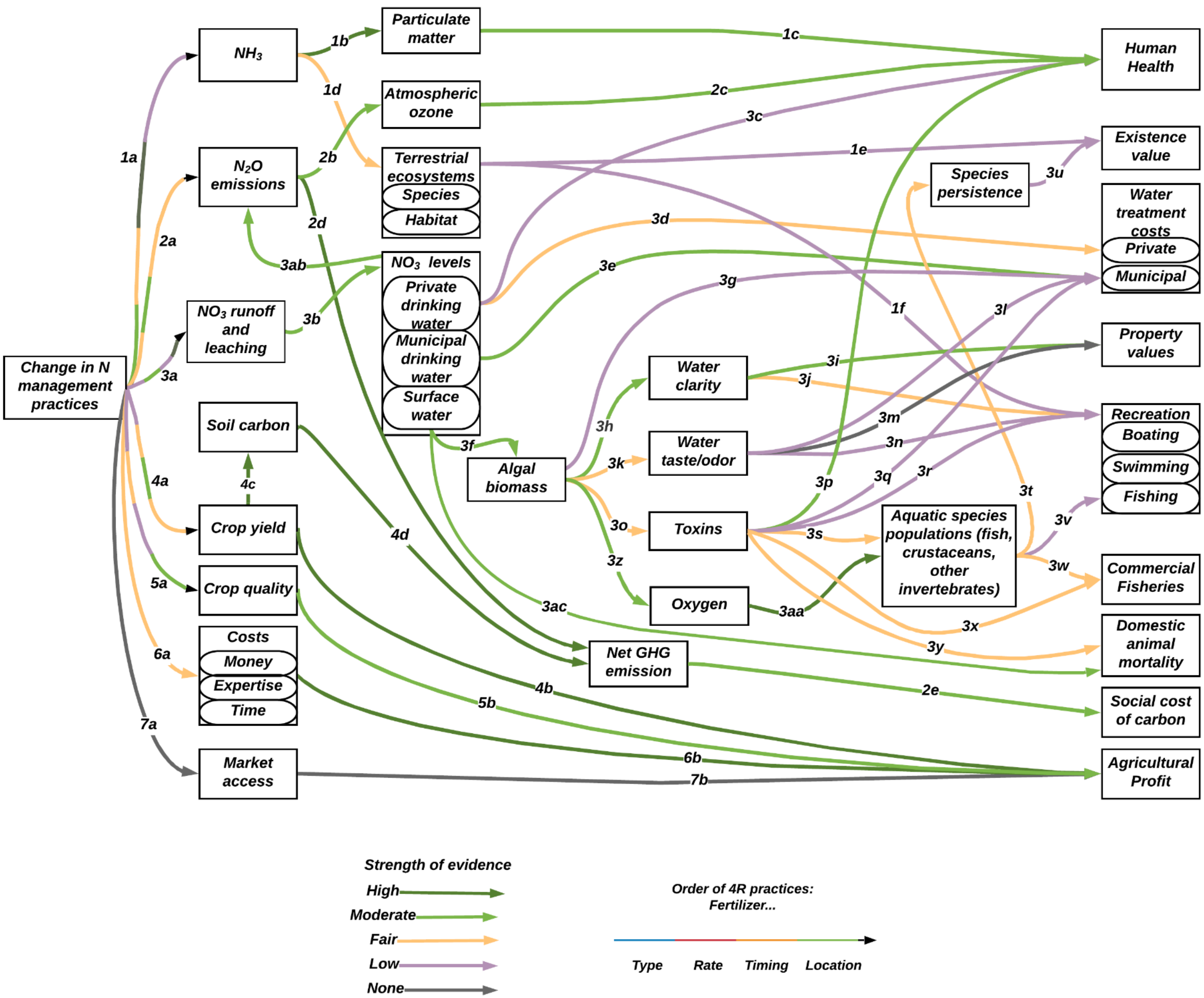
Other factors

List of external factors (including biophysical, ecological, and social factors) that influence the relationship between the starting and ending nodes (boxes), how each factor affects the relationship, and the magnitude of the effect, if known.

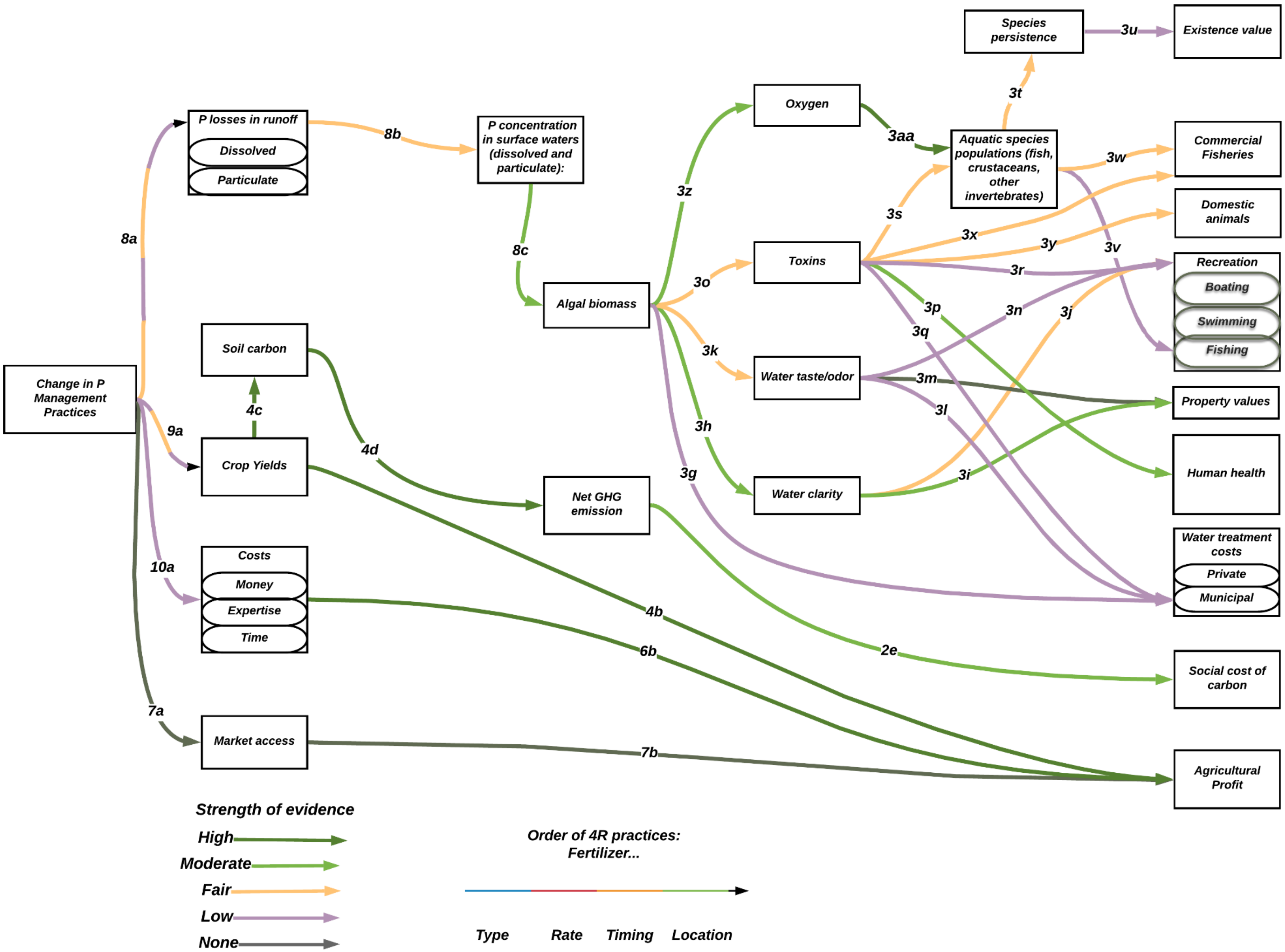
Sources

List of sources for the evidence for the relationship.

Strength of evidence map: Nitrogen management practices



Strength of evidence map: Phosphorous management practices



Nutrient Management Model Evidence Library

1a: Change in N management → Ammonia (NH_3)

Description of relationships

Fertilizer source

Inventories that estimate ammonia (NH_3) emissions on large spatial scales often use generalized emissions factors for different fertilizer nitrogen (N) sources or types, representing the percentage of N applied as fertilizer that volatilizes as NH_3-N. As shown in Table 1, manure and urea N sources have some of the highest ammonia emissions factors across a range of fertilizer types.

Table 1. Ammonia emissions factors for a range of fertilizer nitrogen sources

Fertilizer type	**Emissions factor (% applied N that volatilizes as NH_3-N)**		
Reference	*Goebes et al. (2003)*	Balasubramanian et al. (2015)	Bouwman et al. (2002)[b]
Ammonium bicarbonate	--	--	5.6
Ammonium nitrate	2	--	2.7
Ammonium phosphates	--	--	4.1
Ammonium polyphosphate (liquid)	4	--	--
Ammonium sulfate	8	--	5.9
Ammonium thiosulfate	2.5	--	--
Anhydrous ammonia	1	0.7 (0.6, 0.8)[a]	1.2
Animal manure	--	--	10.4
Aqueous ammonia	1	--	--
Calcium ammonium nitrate	2	--	1.3
Compound NK	--	--	0.8
Compound NPK	--	--	3.9
Diammonium phosphate	4	--	--
Monoammonium phosphate	4	--	--
Nitrogen solutions	8	3.2 (3.0, 3.3)	1.8
Other compound NP	--	--	3.9
Other straight N	--	--	2.3
Urea	15	14.2 (13.9, 14.5)	7.4
Potassium nitrate	2	--	--
Multi-nutrient fertilizer	4	--	--
Miscellaneous fertilizer	5.7	--	--

[a] 95% confidence intervals in parentheses

[b] Values adapted from Bouwman et al. (2002) are calculated using the specified regression model, for upland crops, with fertilizer incorporated, on soil with pH > 5.5 and ≤ 7.3, and CEC > 16 and ≤ 24, in a temperate climate. The model equation is: NH_3 volatilization rate (% of applied N lost as NH_3-N) = exp(factor values for crop type + fertilizer type + application mode + soil pH + soil CEC + climate).

Fertilizer application rate

While the amount of N volatilized as NH_3 certainly increases with higher application rates (Pan et al. 2016), the *proportion* lost may not be affected. A meta-analysis of global studies found no consistent relationship between fertilization rate and the percentage of applied nitrogen lost as NH_3 (Bouwman, Boumans, and Batjes 2002).

Fertilizer application timing

A meta-analysis of 15 global studies found that NH_3 volatilization was not affected by split application (Pan et al. 2016) .

Fertilizer placement

Incorporation of fertilizer reduces volatilization rates, as determined by Bouwman et al. (2002). Using the same regression model as that from which the N source values are adapted in Table 1, the factors for fertilizer placement are shown in Table 2.

Table 2. Impact of fertilizer N placement on ammonia volatilization

Placement	**Factor value**[a]	**Example % volatilized**[b]
Broadcast	-1.305	13.4
Incorporated	-1.895	7.4
Solution	-1.292	13.6

[a] These are the factor values given for the regression equation in Bouwman et al. (2002)

[b] These examples are for urea applied to upland crops on soil with pH > 5.5 and ≤ 7.3, and CEC > 16 and ≤ 24, in a temperate climate.

Factor values for the other soil, climate, and crop characteristics are presented in the relevant sections below (description of relationship or other factors).

Summary of evidence

Ammonia is emitted to the atmosphere from a variety of sources, including vehicles, fires, animal agriculture, and fertilizers. In Canada, agricultural sources (animals and fertilizer) make up about 85% of all anthropogenic ammonia emissions. The proportion of agricultural ammonia emissions from fertilizer use was about 35% in 2011, representing a significant increase from 22% in 2006 (Clearwater, Martin, and Hoppe 2016).

Ammonia (NH_3) is formed in soils from ammonium ($NH_4^+{}_+$); this reaction is controlled by soil and environmental conditions. Once formed, environmental factors also influence NH_3's movement into the atmosphere from the soil or water (see below).

The emissions factors for various fertilizer types from Goebes et al. (2003) are averaged by type and do not take environmental conditions or fertilizer placement into account; these factors have been widely used for NH_3 emissions inventories and do not require additional site-specific data. The emissions factors from Balasubramanian et al. (2015) were developed as part of an NH_3 emissions inventory for the Midwestern United States, based on a biogeochemical model, and are likely more accurate for that region, since local environmental factors were incorporated into the model, but still represent averages for the study area and do not consider fertilizer placement. Differences in emissions factors from these two sources, especially for anhydrous ammonia, highlight the need for further study (Balasubramanian et al. 2015).

The regression equation that provides evidence for effects of fertilizer type and application location was developed from a meta-analysis of NH_3 volatilization field measurements and results in more accurate values, but is only available for three types of fertilizer and requires more data (Bouwman, Boumans, and Batjes 2002). In addition, this analysis excluded variables related to weather, which can greatly influence NH3 volatilization rates (see other factors). In a global meta-analysis of 196 experimental observations, Pan et al. (2016) found that urease inhibitors and controlled release fertilizers reduced NH_3 emissions by an average of 54% and 68%, respectively, while nitrification inhibitors alone increased emissions by 38%. Field and laboratory studies have shown that the addition of NBPT (Agrotain) to urea reduces NH_3 volatilization by 44-65% compared to urea alone (Walker et al. 2013) . While most studies have found that incorporating or banding of urea into soil decreases NH_3 volatilization relative to surface broadcasting (as in the regression presented above), a large-scale field study in Canada found that in dry, acidic soils, the opposite may be true due to localized pH increases around the bands of concentrated urea (Rochette et al. 2009).

The meta-analysis used to create the regression equation described above found no consistent relationship between the rate of fertilizer application and the percentage of applied nitrogen lost as NH_3 (Bouwman, Boumans, and Batjes 2002).

No high-quality evidence was found for the effect of fertilizer application timing on NH_3 emissions; one experiment did include timing as a factor, but simultaneously changed the type of fertilizer applied and application method, making it impossible to distinguish between the effects (Ma et al. 2010). The evidence for higher NH_3 emissions in warmer climates suggests that applying fertilizer during cooler parts of the year may decrease the potential for NH_3 emissions, but no evidence specifically addressing this was found.

Strength of evidence

Fertilizer source

Moderate: The regression model that includes fertilizer source is based on a meta-analysis of 1900 NH_3 measurements; while it excludes weather-related factors, it provides a good basis for the comparison of likely NH_3 emissions from different fertilizer types.

Predictability: The regression model discussed above can predict the amount of fertilizer-applied N lost as NH_3-N based on fertilizer type, fertilizer application mode, and local soil and climate factors, but it does not account for weather-related factors and only includes a few types of nitrogen fertilizer, so it may not be adequate for prediction if other types of fertilizer or additives are used.

Fertilizer application rate

Fair: The regression model based on a meta-analysis provides strong evidence for the lack of a relationship between fertilization rate and the proportion of N applied lost as NH_3, but more research would be useful to assess whether there are any conditions under which a relationship does exist. However, even with no change in proportional loss, lower rates are most certainly associated with lower overall NH_3 emissions.

Fertilizer application timing

None

Fertilizer placement

Low: The meta-analysis and regression model are high-quality evidence for one particular relationship between fertilizer placement and NH_3 emissions, but a Canadian field study indicating an opposite relationship in certain circumstances highlights the need for more research in this area.

Predictability: The regression model discussed above can predict the amount of fertilizer-applied N lost as NH_3-N based on fertilizer type, fertilizer application mode, and local soil and climate factors, but it may not adequately capture all of the factors influencing this relationship in the Canadian context; as noted above, a field study in Canada had results inconsistent with the meta-analysis upon which the regression model is based.

Other factors

Soil characteristics

pH: Conversion of NH_4^+ to NH_3 increases with soil pH; a pH change from 6 to 9 is associated with an increase in the relative concentration of NH_3 to NH_4^+ from 0.1% to 50% (Bouwman, Boumans, and Batjes 2002; Rao and Batra 1983). The type of fertilizer may also play a role here; a laboratory experiment found that different fertilizer types caused different changes in soil pH (Whitehead and Raistrick 1990). The regression model from Bouwman et al. (2002) includes soil pH as a factor (Table 3).

Table 3. Impact of soil pH on ammonia volatilization of fertilizer N

Soil pH	Factor value	Example % volatilized[a]
pH ≤ 5.5	-1.072	6.5
5.5 < pH ≤ 7.3	-0.933	7.4
7.3 < pH ≤ 8.5	-0.608	10.3
pH > 8.5	0	18.9

[a] These examples are for urea applied to upland crops and incorporated, on soil with CEC > 16 and ≤ 24, in a temperate climate.

Cation exchange capacity: Soils with low CEC have higher NH_3 volatilization rates; the regression model from Bouwman et al. (2002) also includes soil CEC as a factor.[1] Researchers have proposed critical CEC values of between 25 cmol/kg to 32 cmol/kg, above which ammonia loss via volatilization is significantly reduced (Bouwman, Boumans, and Batjes 2002; O'Toole, McGarry, and Morgan 1985).

Weather

Rainfall just after fertilizer application reduces NH_3 volatilization (Bouwman, Boumans, and Batjes 2002). Increased wind speeds promote atmospheric mixing and increase NH_3 volatilization rates (Flechard et al. 2013).

[1] Factor values for CEC are not shown here, as the values given in Bouwman et al. (2002) seem to have an error – as they do not follow a consistent pattern in relation to CEC.

Climate

NH_3 emissions tend to be greater in warmer climates; Bouwman et al. (2002) found higher volatilization rates in tropical climates than in temperate ones (Table 4).

Table 4. Impact of climate on ammonia volatilization of fertilizer N

Climate	Factor value	Example % volatilized[a]
Temperate	-0.402	7.4
Tropical	0	11.1

[a] These examples are for urea applied to upland crops and incorporated, on soil with pH > 5.5 and ≤ 7.3, and CEC > 16 and ≤ 24.

Sources

Balasubramanian, S., S. Koloutsou-Vakakis, D.M. McFarland, and M.J. Rood. 2015. Reconsidering emissions of ammonia from chemical fertilizer usage in Midwest USA. *Journal of Geophysical Research-Atmospheres* 120:6232-6246.

Bouwman, A.F., L.J.M. Boumans, and N.H. Batjes. 2002. Estimation of global NH_3 volatilization loss from synthetic fertilizers and animal manure applied to arable lands and grasslands. *Global Biogeochemical Cycles* 16 (2):no. 1024.

Clearwater, R. L., T. Martin, and T. Hoppe. 2016. Environmental sustainability of Canadian agriculture. Ottawa, ON: Agriculture and Agri-Food Canada.

Flechard, C.R., R.-S. Massad, B. Loubet, E. Personne, D. Simpson, J.O. Bash, E.J. Cooter, E. Nemitz, and M.A. Sutton. 2013. Advances in understanding, models, and parameterizations of biosphere-atmosphere ammonia exchange. *Biogeosciences* 10:5183-5225.

Goebes, M. D., R. Strader, and C. Davidson. 2003. An ammonia emission inventory for fertilizer application in the United States. *Atmospheric Environment* 37 (18):2539-2550.

Ma, B.L., T.Y. Wu, N. Tremblay, W. Deen, N.B. McLaughlin, M.J. Morrison, and G. Stewart. 2010. On-farm assessment of the amount and timing of nitrogen fertilizer on ammonia volatilization. *Agronomy Journal* 102:134-144.

O'Toole, P., S. J. McGarry, and M. A. Morgan. 1985. Ammonia volatilization from urea-treated pasture and tillage soils: effects of soil properties. *Journal of Soil Science* 36 (4):613-620.

Pan, Baobao, Shu Kee Lam, Arvin Mosier, Yiqi Luo, and Deli Chen. 2016. Ammonia volatilization from synthetic fertilizers and its mitigation strategies: A global synthesis. *Agriculture, Ecosystems & Environment* 232:283-289.

Rao, D.L.N., and L. Batra. 1983. Ammonia volatilization from applied nitrogen in alkali soils. *Plant and Soil* 70:219-228.

Rochette, P., J.D. MacDonald, D.A. Angers, M.H. Chantigny, M.-O. Gasser, and N. Bertrand. 2009. Banding of urea increased ammonia volatilization in a dry acidic soil. *Journal of Environmental Quality* 38:1383-1390.

Walker, J.T., M.R. Jones, J.O. Bash, L. Myles, T. Meyers, D. Schwede, J. Herrick, E. Nemitz, and W. Robarge. 2013. Processes of ammonia air-surface exchange in a fertilized Zea mays canopy. *Biogeosciences* 10:981-998.

Whitehead, D.C., and N. Raistrick. 1990. Ammonia volatilization from five nitrogen compounds used as fertilizers following surface application to soils. *European Journal of Soil Science* 41 (3):387-394.

1b: NH_3 → Particulate matter

Description of relationship

NH_3 reacts with H_2SO_4 and HNO_3 in the atmosphere to form particulate matter ($PM_{2.5}$). The relationship between NH_3 and $PM_{2.5}$ is strongly influenced by other atmospheric components, but models are available to predict $PM_{2.5}$ concentrations.

Summary of evidence

The Clean Air for Europe (CAFE) project found that ammonia contributes to about 20% of secondary particulate matter in the atmosphere, by mass (Bittman et al. 2014). The particulate matter formed from ammonia can be transported long distances and usually affects air quality 1-1000 kilometers downwind of the ammonia source (Bittman et al. 2014).

Ammonia in the atmosphere reacts with other precursor gases to form particulate matter (Makar et al. 2009). The relevant reactions are:

$$NH_3 + H_2SO_4 \rightarrow NH_4SO_4 + NH_3 \rightarrow (NH_4)_2SO_4$$

$$NH_3 + HNO_3 \leftrightarrow NH_4NO_3$$

Since ammonia reacts preferentially with H_2SO_4 or NH_4SO_4 over HNO_3, most aerosol ammonium is associated with the sulfate ion (CENR 2000).

Chemical transport models can simulate the atmospheric chemical reactions that transform NH_3 into $PM_{2.5}$. GEOS-CHEM is a model that has been used to predict $PM_{2.5}$ concentrations based on levels of NH_3, SO_2, and NO_x from global or regional emissions inventories (Holt, Selin, and Solomon 2015) or predicted from other models (Paulot and Jacob 2014). Model results were correlated with air quality measurements ($r^2 > 0.3$), but do not perfectly predict concentrations of the relevant atmospheric components (Holt, Selin, and Solomon 2015).

Strength of evidence

High: Atmospheric ammonia forms particulate matter in the atmosphere; the chemical reactions behind this relationship are well understood. Ammonia is recognized as a significant contributor of atmospheric particulate matter by national and international clean air organizations, and is included in models of particulate matter.

Predictability: Complex atmospheric dynamics make it difficult to assess the exact effect that a change in ammonia concentration has on particulate matter concentrations. The models discussed above can predict particulate matter concentrations, but they require information about other atmospheric components that adds uncertainty to the model projections. Comparisons of model predictions to air quality measurements show that there is a fairly high degree of error in the model outputs.

Other factors

Atmospheric components

The formation of $PM_{2.5}$ from NH_3 is dependent on the concentration of other atmospheric components, especially NO_x and SO_2. A study that simulated the formation of $PM_{2.5}$ under various emissions scenarios for NO_x and SO_2 found that when emissions of those components are reduced, $PM_{2.5}$ concentrations are much less sensitive to changes in NH_3 emissions, because the formation of $PM_{2.5}$ is limited by NO_x and

SO_2, not NH_3 (Holt, Selin, and Solomon 2015). Large reductions in both NO_x and SO_2 emissions between 2005 and 2012 may mean that NH_3 plays a smaller role in $PM_{2.5}$ formation than it has previously (Holt, Selin, and Solomon 2015).

Location

The concentration of atmospheric components varies by location; in southern California and the mountain west, there are high ammonia and NO_x emissions, but low sulfate emissions, so the formation of ammonium particulate matter is primarily controlled by NO_x instead of sulfate (CENR 2000).

Sources

Bittman, S., J.R. Brook, A. Bleeker, and T.W. Bruulsema. 2014. Air quality, health effects and management of ammonia emissions from fertilizers. In *Air Quality Management*, edited by E. Taylor and A. McMillan. Dordrecht: Springer Science+Business Media.

CENR. 2000. Atmospheric Ammonia: Sources and Fate, a review of ongoing federal research and future needs. Committee on the Environment and Natural Resources (CENR), Air Quality Research Subcommittee.

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Makar, P.A., M.D. Moran, Q. Zheng, S. Cousineau, M. Sassi, A. Duhamel, M. Besner, D. Davignon, L.-P. Crevier, and V.S. Bouchet. 2009. Modelling the impacts of ammonia emissions reductions on North American air quality. *Atmospheric Chemistry and Physics* 9:7183-7212.

Paulot, F., and D.J. Jacob. 2014. Hidden cost of U.S. agricultural exports: Particulate matter from ammonia emissions. *Environmental science & technology* 48:903-908.

1c: Particulate matter → Human health[2]

Description of relationship

A long-term increase in $PM_{2.5}$ of 10 $\mu g/m^3$ has been associated with an increase in respiratory mortality of 1.7-8% and an increase in cardiovascular mortality of 9-76%.

Summary of evidence

Particles of less than 5 µm diameter can penetrate human lungs and eventually move into the bloodstream, causing asthma, reduced lung function, lung cancer, myocardial infarction, and congestive heart failure (Kim, Kabir, and Kabir 2015). A review of the effects of particulate matter air pollution on human health found many individual studies demonstrating a link between long-term exposure to particulate matter and increased risks of mortality from lung cancer, congestive heart failure, and cardiopulmonary disease (Anderson, Thundiyil, and Stolbach 2012).

Strength of evidence

Moderate: Many individual studies using accepted epidemiological methods show a link between long-term exposure to particulate matter and adverse human health outcomes, but no meta-analyses assessing dose-response relationships between particulate matters and health outcomes were found.

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[2] This section was adapted from (Warnell, Olander, and Mason 2018 301 301)

Other factors

Certain sub-populations (children, the elderly, and people with compromised lung function) are more vulnerable to health effects from particulate matter (Anderson, Thundiyil, and Stolbach 2012).

Sources

Anderson, J.O., J.G. Thundiyil, and A. Stolbach. 2012. Clearing the air: a review of the effects of particulate matter air pollution on human health. *Journal of Medical Toxicology* 8:166-175.

Kim, K.-H., E. Kabir, and S. Kabir. 2015. A review on the human health impact of airborne particulate matter. *Environment International* 74:136-143.

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1d: NH_3 → Terrestrial ecosystems

Description of relationship

Deposition of NH_3 in terrestrial ecosystems can increase plant biomass and shift plant communities.

Summary of evidence

Nitrogen is the limiting nutrient for plant growth in many terrestrial ecosystems. Deposition of NH_3 in these ecosystems promotes plant growth overall, but can shift the competitive balance from plants adapted to nutrient-poor soils to more generalist, faster-growing species (Bobbink, Hornung, and Roelofs 1998; Bobbink et al. 2010). In many cases, this leads to a reduction in species richness; in extremely nitrogen-deficient soils, NH_3 deposition can increase overall plant species richness, but the original plant species are often excluded from the community in this case (Bobbink, Hornung, and Roelofs 1998). High NH_3 concentrations in soils can also lower soil pH, promoting acid-tolerant plant species, and high NH_3 concentrations in plants can increase insect herbivory and make plants more susceptible to stressors (e.g., pathogens, frost, drought) (Bobbink, Hornung, and Roelofs 1998).

Forests across North America and Europe are growing faster now than they were a century ago; this increase is partially attributed to greater inputs of atmospheric nitrogen (Matson, Lohse, and Hall 2002). Long-term studies in western Europe have shown an increase in nitrophilic species over the past several decades thought to be promoted by nitrogen deposition over that time period (Bobbink, Hornung, and Roelofs 1998). Few such studies of biodiversity changes associated with nitrogen deposition have been conducted outside of Europe, but nitrogen fertilization experiments in North America confirm that increased nitrogen availability enhances the growth of fast-growing, nutrient-rich species, resulting in a decline in species richness (Matson, Lohse, and Hall 2002; Suding et al. 2005).

A meta-analysis of 34 nitrogen fertilization experiments in North America found that fertilization caused an increase in net primary productivity and a decrease in plant species richness in all included terrestrial ecosystems (tundra, grasslands, and abandoned agricultural fields); rare species were more likely than common species to be lost due to fertilization (Suding et al. 2005). Another meta-analysis of nitrogen addition experiments found that species richness declined more quickly at low cumulative nitrogen doses (total nitrogen additions over time) than at higher doses, and cumulative nitrogen dose was a better predictor of species loss than the amount of nitrogen added annually (Schrijver et al. 2011).

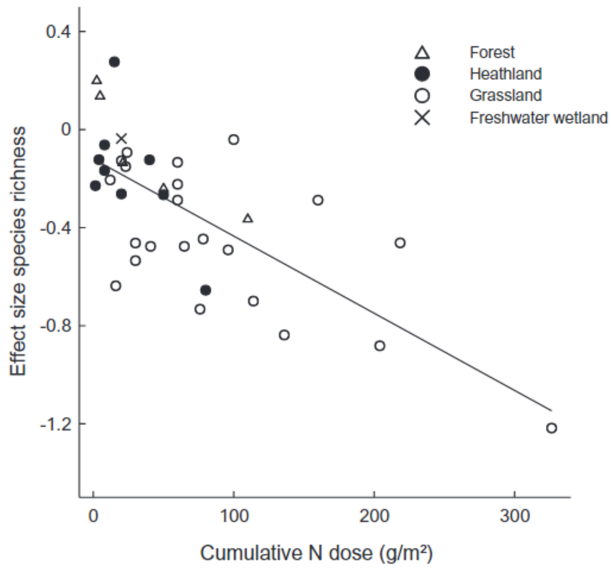

Figure 1. Effect size of species richness versus cumulative nitrogen dose. Effect size is logarithmic, so the relationship shown here indicates that species richness loss occurs faster at low cumulative nitrogen inputs. Source: De Schrijver et al. (2011).

Certain species are very susceptible to nitrogen enrichment; a cyanobacterial photobiont found in some lichens is sensitive to both nitrogen and acidity; a study in Scandinavia found that these lichens declined when nitrogen deposition exceeded 5-10 kg/ha/year (Bobbink, Hornung, and Roelofs 1998).

NH_3 can have direct toxic effects on plants, but concentrations from deposition do not reach toxic levels except very close to point sources of NH_3 (Bobbink, Hornung, and Roelofs 1998).

If the plant community changes significantly, certain wildlife species that depend on the plants for food or cover may also be affected. No evidence was found linking NH_3 deposition to a change in the wildlife community.

Strength of evidence

Fair: Long-term studies in Europe and ecosystem-level nitrogen addition experiments in North America confirm that increases in nitrogen availability can cause net primary productivity to grow and species communities to shift. However, meta-analyses of North American experiments show substantial variability in the occurrence and magnitude of these changes, and nitrogen addition experiments do not perfectly simulate nitrogen deposition (the form of added nitrogen, amount of nitrogen added, and period of time over which nitrogen is added all potentially vary from deposition), which may influence the results.

Other factors

Soils

A meta-analysis of nitrogen addition experiments in North America found that the number of species lost due to fertilization was negatively correlated with soil cation exchange capacity – reflecting higher potential for soil acidification (Clark et al. 2007).

Climate

A meta-analysis of nitrogen addition experiments in North America found that the number of species lost due to fertilization was negatively correlated with temperature. This is likely because of higher potential for frost damage associated with low background nitrogen availability (Clark et al. 2007).

Ecosystem type

Specific effects of NH_3 deposition are influenced by many ecosystem attributes, including plant species sensitivity and baseline nutrient availability (Bobbink et al. 2010). A global meta-analysis of nitrogen addition experiments found a positive effect of nitrogen addition on plant biomass in grasslands, but no effect in forest understory or heathland (Schrijver et al. 2011). Species richness declined with nitrogen fertilization in grasslands and heathland, but no effect was seen in forests.

Sufficient studies have been conducted in Europe to establish critical loads for nitrogen deposition (an estimate of the minimum level of deposition at which harmful effects on sensitive parts of the ecosystem are observed) by ecosystem type, but critical loads have not been estimated for North American ecosystems so far (Bobbink et al. 2010). On the other hand, the usefulness of the critical loads concept for nitrogen deposition has been questioned, as even very low nitrogen addition rates have been shown to cause species losses over a long period of time (Schrijver et al. 2011).

Sources

1e: Terrestrial ecosystems → Existence value[3]

Description of relationship

Many people appreciate nature, and some are willing to pay for the continued existence of an intact terrestrial ecosystem, even if they do not personally expect to visit or live near it. This is called "existence value" and has been measured via surveys in a few locations, but is highly context-dependent.

Summary of evidence

The public and the media express alarm at reports of species extinctions and other ecosystem damages. However, there have been few studies of existence values for terrestrial ecosystems, and those studies that do exist are several decades old. A recent review of economic values of wilderness found six studies that included non-use values; for those studies, annual household willingness to pay ranged from $0.01-$0.61/1000 acres (Holmes et al. 2016). With so few studies available, it is not possible to predict existence values for different types of terrestrial ecosystems in various locations.

Contingent valuation methods provide a way to estimate the existence value of a particular ecosystem. A study assessing the existence value of a wilderness area in Vermont provides an example of this method (Gilbert, Glass, and More 1992).

Strength of evidence

Low: No general relationship between intact terrestrial ecosystems and existence values can be assessed due to the scarcity of relevant studies.

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[3] This section was adapted from (Warnell, Olander, and Mason 2018 301 301).

Other factors

Location

The distance of a particular ecosystem from people who value it may influence its overall existence value; no evidence was found to assess this hypothesis.

Other

The particular type of ecosystem in question may influence its existence value; no studies that compared existence values across ecosystem types were found.

Sources

1f: Terrestrial ecosystems (species) → Recreation value[4]

Description of relationship

Some people are willing to pay more to visit a recreational area with a given species present.

Summary of evidence

The presence of terrestrial species is an important component in many recreational activities, including hunting and wildlife watching. Multiple methods exist for estimating recreational value, including the amount spent directly on a recreational activity or travel to a particular area and stated preference surveys. Studies of birdwatchers have shown that birders are willing to pay more to visit a site that has been known to host endangered species, and that there is a positive relationship between the rarity of a bird species and the number of people who go see it (Booth et al. 2011; Kolstoe and Cameron 2017). No studies were found that linked the presence of certain plant species or community to recreational value, but it is possible that certain plant species are important to a subset of recreationists.

The USGS Benefit Transfer Toolkit has regression functions created from many individual studies that can estimate the value of hunting and wildlife watching opportunities based on the type of species involved (USGS n.d.). The wildlife viewing function gives separate values for birds, charismatic megafauna, and general wildlife. The hunting function gives separate values for deer, elk, moose, mountain goat, pheasant, waterfowl, large game, and small game. These functions can give a general estimate of the recreational value of a particular species' presence. The Toolkit also includes a large database of recreational value studies for other types of recreation (hiking, camping, backpacking, etc.). If the location and context of a study in the database closely matches the location of interest, these values can be used as a rough point estimate for recreational value in that location.

Studies specifically conducted for the activity and context of interest provide strong evidence for the recreational value of a particular ecosystem or species. Several methods are available; a study of elk-viewing in Oregon provides an example of the travel cost method (Donovan and Champ 2009), and a study of bear-viewing in Yellowstone National Park provides an example of the contingent valuation method (Richardson et al. 2014).

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[4] This section was adapted from (Warnell, Olander, and Mason 2018 301 301).

Strength of evidence

Low: Many studies have estimated the value of recreation in natural ecosystems, but the great variety in geographical location, ecosystem type, and recreational activities makes it difficult to state a general relationship, and most studies relate to the presence of certain wildlife species, not plant communities.

Predictability: The USGS Benefit Transfer Toolkit regression functions provide estimates for certain recreational activities, but do not account for all of these factors. The strength of a point estimate benefit transfer from a study in the USGS database depends on how well a study in the database matches the context of the recreational activity of interest. In addition, all of the studies in the database were conducted in the United States, so they may not be fully applicable to the Canadian context.

Other factors

Location

The value of a recreational experience varies by location, likely due to differences in the quality of recreational experiences and in cost of living. The USGS Benefit Transfer Toolkit regression functions include region as a variable, but does not include options for locations outside of the United States. There may be regional differences for recreational values within Canada as well.

Species

Recreational values associated with wildlife vary by species. The USGS Benefit Transfer Toolkit regression functions include species as a variable, but not all relevant species are incorporated.

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2a: Change in N management → N_2O emissions

Description of relationships

Fertilizer source

Using fertilizer with nitrification inhibitors (e.g., SUPERU, AGROTAIN PLUS) reduces N_2O losses by 31% (Eagle et al. 2017).

Fertilizer application rate

Several meta-analyses have shown an exponential relationship between fertilization rate and N_2O emissions (Shcherbak, Millar, and Robertson 2014; Eagle et al. 2017). Each additional 10 kg fertilizer N/ha/year increases N_2O emissions by 2.7% (for fertilization rates between 110 and 270 kg N/ha) (Eagle et al. 2017). However, a focus on reducing rate can risk jeopardizing yield and food security. This issue can be addressed by instead reducing N balance (e.g., N inputs as fertilizer minus N outputs as crop), for which a reduction by 10 kg N/ha decrease N_2O emissions by 6.6% (McLellan et al. 2020). Farmers can reduce a high N balance by lowering fertilizer N rate without changing yield, or by increasing yield at the same fertilizer rate.

Fertilizer application timing

Switching from applying all fertilizer before planting to applying a portion of fertilizer as side-dress reduces N_2O emissions by 26% (Eagle et al. 2017).

Fertilizer placement

Broadcasting fertilizer instead of injecting or banding reduces N_2O emissions by 33% (Eagle et al. 2017).

Summary of evidence

Most of the relationships described above are from a recent meta-analysis of N_2O losses from corn production in North America, which used hierarchical multi-level regression models to evaluate the effect of a variety of environmental factors and management techniques on N_2O emissions (Eagle et al. 2017). The changes in N_2O emissions described in these relationships are on a per-area basis; the study also calculated changes in emissions on a per-yield basis, with the results consistent with the per-area results. Except for the fertilizer type relationship, the relationships reported here are from models limited to "standard" nitrogen fertilizer application rates (110-270 kg N/ha); changes in management practices for fertilization outside of these rates have slightly different results. Limitations of this analysis are that it only includes corn and may not be fully applicable to other crops and that existing studies do not represent the full geographical range (and thus environmental and climate factor ranges) of corn production in North America. Evidence for these relationships from other meta-analyses and individual research studies generally aligns with these results in terms of the direction of the effects of nutrient management practices on N_2O emissions, but the magnitude of effects vary and appear to be influenced by a variety of climate and environmental factors.

Strength of evidence

Fertilizer source

Fair: The relationship described above is from the meta-analysis of North American corn systems; it is unclear how relevant this is to other systems.

Fertilizer application rate

Moderate: The best available information is from a recent meta-analysis limited by the availability of studies examining N_2O emissions in certain geographic regions of North America (Eagle et al. 2017), although the N balance–N_2O model is more broadly applicable to temperate-region, rainfed crops (McLellan et al. 2020). Several other meta-analyses exist that examine some of these relationships, but these are limited in the types of nutrient management practices they include or include studies from other parts of the world and may be less relevant to North America (Helgason et al. 2005; Shcherbak, Millar, and Robertson 2014).

Predictability: The results of the meta-analysis described above can be used to predict N_2O emissions from corn when the context (geography, climate, and fertilization rate) is within the range of studies included in the meta-analysis. If using N balance instead of rate, the model outlined by McLellan et al. (2020) is applicable to major rain-fed crops across a larger suite of temperate regions.

Fertilizer application timing

Fair: The relationship described above is from the meta-analysis of North American corn systems; it is unclear how relevant this is to other systems.

Fertilizer placement

Fair: The relationship described above is from the meta-analysis of North American corn systems; it is unclear how relevant this is to other systems.

Other factors

Climate

Temperature: N_2O emissions increase by 17% (full model) and 28% (model restricted to "standard" fertilizer application rates) for each 1°C rise in average July temperature (Eagle et al. 2017).

Environmental factors

Soil carbon content: The microbes that carry out denitrification are heterotrophic and require a carbon source; if soil carbon contents are low, the denitrification process may become carbon-limited, not nitrate-limited (Zebarth et al. 2008). There is some support for this in the two meta-analyses that included soil carbon content: soil carbon had a marginally-significant, weak positive correlation with N_2O emissions in one analysis (Eagle et al. 2017), and soil carbon contents >1.5% had N_2O emissions factors than those <1.5% in another (Shcherbak, Millar, and Robertson 2014).

Soil moisture content: Denitrification, the process by which N2O is formed, requires anaerobic conditions and is therefore dependent on soil moisture content. Even if other conditions (e.g., soil NO_3 concentration) are right, N_2O emissions do not peak until a rainfall event occurs (Burton et al. 2008).

Soil pH: N_2O emissions factors have been shown to be higher in acidic or neutral soils than basic soils (Shcherbak, Millar, and Robertson 2014).

Crop type

A global meta-analysis found that nitrogen-fixing crops had greater N_2O emissions factors than non-nitrogen-fixing crops as a function of fertilizer N applied, but in practice nitrogen-fixing crops generally receive less fertilizer than non-nitrogen-fixing crops, so this is unlikely to have a practical effect (Shcherbak, Millar, and Robertson 2014).

Agricultural management techniques

Tillage: Tillage practices can influence N_2O emissions via effects on many mediating factors, which is why studies of tillage effects on N2O emissions often have inconsistent results. A meta-analysis of 239 studies found interactions between tillage practices, climate, length of time that practices were implemented, and nitrogen fertilizer placement (van Kessel et al. 2012). Long-term use (10 years or longer) of no-tillage and reduced-tillage systems in dry climates (Global-Aridity values <0.65) reduced N_2O emissions by 34% (relative to conventional tillage), while short-term (less than 10 years) use of no-tillage and reduced-tillage in dry climates increased N_2O emissions by 36%. When nitrogen fertilizer was placed >5cm underground, no-tillage and reduced-tillage systems in humid climates (Global-Aridity values >0.65) reduced N_2O emissions by 27% (van Kessel et al. 2012). Other meta-analyses show no effect (Eagle et al. 2017) or a weak and inconsistent effect (Helgason et al. 2005) of tillage on N_2O emissions.

Cover crops: Cover crops can affect N_2O emissions in different ways depending on specific attributes of the cover crop management; the overall effects of cover crops are not well understood. A meta-analysis found that legume cover crops (at low nitrogen fertilization rates) and incorporation of cover crop residues into the soil increase N_2O emissions relative to no cover crops (Basche et al. 2014). Cover cropping can also interact with climate; cover crops in areas with higher precipitation and higher variability in precipitation had greater increases in N_2O emissions relative to no cover crops (Basche et al. 2014). While cover crops seem to affect N_2O emissions differently at different points in their life cycle, insufficient full-year measurement data for these systems limited conclusions about overall N_2O implications (Basche et al. 2014).

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2b: N_2O → Atmospheric ozone

Description of relationship

An increase in the concentration of N_2O in the stratosphere results in a decrease in global mean ozone density. The effect of a specific change in N_2O concentration on ozone varies by altitude and the concentration of other atmospheric components, and can be predicted by atmospheric models.

Summary of evidence

N_2O causes ozone loss through a series of reactions (Portmann, Daniel, and Ravishankara 2012). In the stratosphere, about 10% of N_2O is converted to reactive NO:

$$N_2O + O \rightarrow 2NO.$$

O_3 and O rapidly interconvert via photolysis and reaction with O_2:

$$O_2 + h\nu \rightarrow O + O;$$

$$O + O_2 + M^{5} \rightarrow O_3 + M;$$

$$O_3 + h\nu \rightarrow O_2 + O.$$

[5] M is a third body required for this reaction, which can be any stable molecule that removes excess energy from the excited intermediate product. http://acmg.seas.harvard.edu/people/faculty/djj/book/bookchap9.html#43510

NO destroys O_3 by converting it to O_2; the resulting NO_2 molecule reacts with O to form NO:

$$NO + O_3 \rightarrow NO_2 + O_2;$$

$$NO_2 + O \rightarrow NO + O_2.$$

Since NO is both an input to and an output from this process, a molecule of NO can destroy many (10^3 to 10^5) O_3 molecules before it is converted to a nonreactive form (Portmann, Daniel, and Ravishankara 2012). While other compounds (HO and Cl) can also destroy O_3 via a similar process, the NO cycle is dominant in the middle stratosphere, near the area of maximum O_3 density. However, since only 10% of N_2O emissions are converted to NO in the stratosphere, the ozone-depleting potential (ODP), a measure of a gas's ability to destroy ozone relative to CFC-11, of N_2O is lower than that of most halocarbons that were the initial focus of ozone depletion research and are regulated under the Montreal Protocol (Ravishankara, Daniel, and Portmann 2009). Due to the success of these regulations in reducing halocarbon emissions, atmospheric chemists expect that N_2O will become the dominant ozone-depleting substance in the atmosphere by the end of the 21st century (Portmann, Daniel, and Ravishankara 2012; Randeniya, Vohralik, and Plumb 2002; Daniel et al. 2010). This prediction is based on computer models of the atmosphere, including the NOCAR model, the Goddard Space Flight Center model and the Goddard Earth Observing System chemistry-climate model, which are widely used and show good agreement in their predictions, though they are simplifications of actual atmospheric processes (Daniel et al. 2010; Portmann, Daniel, and Ravishankara 2012; Randeniya, Vohralik, and Plumb 2002; Ravishankara, Daniel, and Portmann 2009; Portmann and Solomon 2007).

Strength of evidence

Moderate: The destruction of O_3 by N_2O is caused by well-understood chemical processes. The effects of N_2O on O_3 in the stratosphere are consistent in direction and moderately consistent in magnitude, but can be influenced other atmospheric components (see other factors).

Predictability: Computer models of atmospheric processes based on chemical principles can predict the effect of N_2O on atmospheric O_3. These models are well-documented and widely accepted by atmospheric and climate researchers as reasonable approximations of atmospheric processes.

Other factors

Temperature: Changes in stratospheric temperature can affect the rate of relevant reactions and the concentration of more stable N compounds relative to reactive N compounds, which drives the amount of ozone destruction caused by a molecule of N_2O. Lower stratospheric temperatures result in less ozone loss from N_2O (Portmann, Daniel, and Ravishankara 2012).

Other atmospheric components: CO_2 and ClO_x in the stratosphere influence the effect of N_2O emissions on O_3. The largest effect is due to CO_2 decreasing the temperature in the stratosphere, which decreases N_2O-mediated destruction of O_3 as described above, so the ozone-depleting effect of N_2O is expected to decrease as CO_2 levels continue to rise (Portmann, Daniel, and Ravishankara 2012). N_2O emissions partially mitigate the ozone-depleting effect of chlorinated hydrocarbons by forming a longer-lived, less-reactive chlorinated molecule, so as atmospheric hydrocarbon concentrations continue to decline, N_2O's ozone-depleting effect is expected to increase (Portmann, Daniel, and Ravishankara 2012). Overall, the ozone-depleting potential of N_2O could change by up to 20% depending on changes in other atmospheric components; however, this uncertainty is not expected to change the direction of N_2O's effect on O_3 or researchers' conclusion that N_2O will be the dominant O_3-depleting substance by the end of the 21st century (Portmann, Daniel, and Ravishankara 2012).

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2c: Atmospheric ozone → Human health

Description of relationship

A decrease in stratospheric ozone density results in more solar ultraviolet radiation (UVR) reaching Earth's surface. Exposure to solar ultraviolet radiation has a variety of adverse health effects in humans, primarily related to skin cancers, eye damage, and immunosuppression. It also has positive human health effects by promoting vitamin D synthesis.

Summary of evidence

Effects of ozone on ultraviolet radiation

Ozone does not absorb all wavelengths of UV radiation equally, and different UV wavelengths are thought to have different health effects. The radiation amplification factor (RAF) is the percent increase in biologically active UV reaching the Earth's surface due to a 1% decrease in the ozone layer at that location; RAFs have been estimated at 1.1-1.6 for the wavelengths linked to skin cancers and 1.5-2.3 for the wavelengths linked to DNA damage (Madronich and de Gruijl 1994).

Effects of ultraviolet radiation on human health

The influence of solar ultraviolet radiation on several adverse human health effects has been extensively studied. It is difficult to accurately assess UVR exposure (generally, exposure classes are based on self-reported number of sunburns or lifetime sun exposure hours), which makes it hard to estimate relative risk or determine dose-response relationships between UVR exposure and health effects.

Skin: Solar UVR is estimated to cause more than 90% of melanomas in North America (Gallagher and Lee 2006). Melanomas have been most directly linked to intermittent sun exposure (e.g., recreation, sunbathing) than chronic sun exposure (e.g., occupational exposure). A meta-analysis of melanoma risk found an odds ratio of 1.61 for people in the highest category of intermittent sun exposure compared to those in the lowest category (Gallagher and Lee 2006). One study calculated an "amplification factor", or the percent increase in nonmelanoma skin cancer resulting from a 1% decrease in ozone, of 1.7 for basal cell carcinoma and 3.0 for squamous cell carcinoma (Lim and Cooper 1999). Similar to melanoma, the risk of basal cell carcinoma has shown to increase with recreational sun exposure; conversely, the risk of squamous cell carcinoma increases with chronic sun exposure (Gallagher and Lee 2006).

Eyes: Several eye-related diseases and conditions have been shown to be associated with exposure to UVR. A model showed that 43.6% of the risk of pterygium, a condition in which excess tissue growth blocks vision, is due to the cumulative dose of ocular UV-B exposure (Norval et al. 2007). No meta-analyses for the effect of UVR exposure on cataracts have been conducted due to the variety in research study design, but a systematic review conducted in 2002 found that 15 of 22 epidemiologic studies showed a significant association between UV exposure and cortical cataracts (Gallagher and Lee 2006). One study that modeled US cataract risks due to ozone depletion estimated that the number of cortical cataract cases would increase by 1.3-6.9% by 2050 with a 5-20% decrease in ozone (West et al. 2005). One meta-analysis linked sun exposure to the risk of age-related macular degeneration (AMD) (Lucas et al. 2015), but other individual studies have shown mixed results (Gallagher and Lee 2006).

Immune system: Exposure to UVR can influence both the human immune response to viruses and the viruses themselves; UVR has been shown to exacerbate the effects of herpes simplex type 1 and human papilloma virus (Norval et al. 2007).

Vitamin D: Exposure to UVR is necessary for vitamin D synthesis, and most people get more than 90% of their vitamin D requirement from exposure to UVR (Norval et al. 2007). Adequate vitamin D levels lower the risk of skeletal disorders such as osteoporosis and rickets, as well as fractures (Norval et al. 2007; Autier et al. 2013). Low levels of vitamin D have been linked to a variety of other adverse health effects, including some cancers, autoimmune diseases, and hypertension; however, two recent meta-analyses concluded that low vitamin D levels are likely to be a result of inflammation caused by these illnesses, rather than a cause of the illness (Theodoratou et al. 2014; Autier et al. 2013).

Strength of evidence

Moderate: The relationship between stratospheric ozone depletion and increased levels of ultraviolet radiation reaching the Earth's surface is straightforward and well-understood, but connecting UVR exposure to health effects is more difficult. The risk of a variety of adverse health outcomes (melanoma, non-melanoma skin cancers, cortical cataracts, and pterygium) has been linked to UVR exposure through large-scale epidemiologic studies and/or meta-analyses, so it is reasonable to conclude that higher UVR levels would lead to a greater risk of these outcomes.

Predictability: Determining specific dose-response relationships is complicated by the lack of a reliable way to measure individuals' past UVR exposures; therefore, it is difficult to assess to what extent an increase in UVR radiation (due to ozone depletion) increases the risk of these health outcomes.

Sources

2d: N_2O → Net GHG emissions

Description of relationship

The emission of one metric ton of N_2O is equivalent to the emission of 298 metric tons of CO_2, in terms of global warming effects assessed on a 100-year timescale.

Summary of evidence

Many different greenhouse gases, including N_2O, contribute to global warming. To allow for comparison of the effects of various greenhouse gases, and to assess the total global warming effects of the mixture of greenhouse gases in the atmosphere, the global warming potential (GWP) of various gases has been determined. The GWP of any gas is relative to the warming potential of CO_2, which by definition has a GWP of 1. The GWP of a certain gas is determined by its radiative efficiency and its atmospheric lifetime (IPCC 2007). GWPs vary depending on the timescale of the assessment; for example, if a particular greenhouse gas has a very high radiative efficiency compared to CO_2, but has a very short atmospheric lifetime, it may have a very large GWP over a 10-year period but a much smaller GWP over a longer period. The UN Framework Convention on Climate Change requires the use of 100-year GWPs for inventory reporting, and the Canadian Environmental Protection Act uses the same GWPs for emissions reporting (Government of Canada 2017). The 100-year GWP for N_2O is 298; this can be multiplied by the amount of N_2O emitted to estimate the equivalent amount of CO_2 emissions (Government of Canada 2017; IPCC 2007).

The use of GWP as a metric to compare global warming effects of different gases has been criticized because it does not measure effects in terms of a clear impact parameter (e.g., temperature change, economic impact) that would make it more relevant to emissions policymaking. However, GWP is widely used (especially since it was adopted in the Kyoto Protocol), and its calculation is transparent and includes fewer assumptions (about atmospheric dynamics, economic costs of impacts, etc.) than other possible metrics (Shine 2009).

Strength of evidence

High: Nitrous oxide's identity as a greenhouse gas is an established fact, and the global warming potential reported here, while possibly not an ideal metric, is widely accepted and based on physical science.

Predictability: The use of global warming potential allows the effect of N_2O emissions to be directly predicted and compared with other greenhouse gases in terms of their contribution to global climate change.

Sources

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2e: Net GHG emissions → Social cost of carbon[6]

Description of relationship

The emission of one metric ton of CO_2 in 2020 has a social cost of $45.1 (2012 Canadian dollars, at a 3% discount rate).

Summary of evidence

Environment and Climate Change Canada has developed social cost of carbon estimations through 2050 for use in government agencies' Regulatory Impact Analysis Statements (Government of Canada 2016). The social cost of carbon represents the global damages from climate changed by an additional tonne of carbon dioxide released into the atmosphere in a given year. It can also be used to estimate the avoided damages from decreased carbon dioxide emissions. Environment Canada's social cost of carbon values are modified from the United States Interagency Working Group values, based on three integrated economics and climate science models (Interagency Working Group 2016). The Canadian values were last updated in 2016 (Table 5).

Table 5. Canadian social cost of carbon central estimate and 95th percentile for carbon dioxide emissions from 2010-2050 ($C 2012). Source: Government of Canada (2016).

Year	Central estimate	95th percentile
2010	34.1	131.5
2013	37.4	149.3
2015	39.6	161.1
2016	40.7	167.0
2020	45.1	190.7
2025	49.8	213.3
2030	54.5	235.8
2025	59.6	258.9
2040	64.7	281.9
2045	69.7	300.9

[6] This section was adapted from (Warnell, Olander, and Mason 2018 301 301).

2050	74.8	319.8

Strength of evidence

Moderate: The adverse social effects of climate change are widely recognized, and substantial research has been done to attempt to quantify and predict those effects. However, effects will vary over space and time.

Predictability: The social cost of carbon estimates are based on models that are widely used, but the specific cost estimates vary among the models, and the uncertainties of projecting climate and economic outcomes are unavoidable (National Academies of Sciences 2017).

Sources

3a: Change in N management → NO_3 runoff and leaching

Description of relationships

Fertilizer source

In irrigated systems, the use of fertilizers with nitrate inhibitors and controlled release fertilizer reduced nitrate leaching by 27%; the effects of these improved fertilizer types were not assessed separately (Quemada et al. 2013). No evidence was found for effects of fertilizer type on nitrate leaching in non-irrigated systems.

Fertilizer application rate

Two meta-analyses found a linear relationship between fertilizer N application rates and nitrate leaching losses (Eagle et al. 2017; Zhou and Butterbach-Bahl 2014). Other studies suggest a nonlinear effect in which nitrate leaching losses increase exponentially at high fertilization rates (Di and Cameron 2002).

Fertilizer application timing

No effect of fertilizer application timing on nitrate leaching was detected in irrigated or non-irrigated corn systems (Eagle et al. 2017). In irrigated systems, optimized fertilizer application timing reduced nitrate leaching by 22% (Quemada et al. 2013).

Fertilizer placement

No evidence found.

Summary of evidence

Nitrate leaching from soil occurs in two steps: nitrate formation by the oxidation of ammonium and nitrate transport from the soil in water (Cameron, Di, and Moir 2013). Nitrate formation is controlled by heterotrophic bacteria and is influenced by soil moisture, pH, temperature, and the amount of nitrogen applied to the soil; nitrate transport is influenced by the rate of water moving through the soil (Cameron, Di, and Moir 2013).

Many studies and several meta-analyses following standard methods have been conducted for nitrate leaching losses in agricultural systems. A meta-analysis of North American corn systems found evidence of a linear relationship between nitrogen fertilizer application rates and nitrate leaching losses, with about 7% of each additional kilogram of nitrogen applied as fertilizer being lost through nitrate leaching

(Eagle et al. 2017). The same study found no effect of fertilizer application timing on nitrate leaching, but noted effects from climate, soil characteristics, and irrigation (Eagle et al. 2017). Another meta-analysis of nitrate losses in wheat and corn systems also found a linear relationship between nitrogen fertilizer application rates and nitrate leaching losses, with some differences found among different crop types (Zhou and Butterbach-Bahl 2014). A meta-analysis that only included irrigated systems found an effect of fertilizer type and application timing on nitrate losses (Quemada et al. 2013). Other studies have assessed the effects of additional agricultural management techniques, especially the use of cover or catch crops, on nitrate losses (Di and Cameron 2002; Valkama et al. 2015; Quemada et al. 2013).

Strength of evidence

Fertilizer source

Low: One meta-analysis was found that examined the effects of fertilizer type on nitrate leaching in irrigated systems, but it did not differentiate between different types of "improved" fertilizers (Quemada et al. 2013). No evidence was found for the effect of fertilizer type on nitrate leaching in non-irrigated systems.

Fertilizer application rate

Moderate: Consistent evidence for a positive linear relationship between fertilization rate and nitrate leaching was found by two meta-analyses. Other studies suggest that the effects may be nonlinear in certain cases (Eagle et al. 2017; Zhou and Butterbach-Bahl 2014; Di and Cameron 2002).

Fertilizer application timing

Low: Two meta-analyses had inconsistent results; an analysis of only corn systems found no effect, while an analysis of only irrigated systems found that optimized fertilizer timing reduced nitrate leaching (Eagle et al. 2017; Zhou and Butterbach-Bahl 2014). More research is needed to clarify the relationship and influential factors.

Fertilizer placement

None.

Other factors

Climate

In non-irrigated agroecosystems, areas receiving >800 mm annual precipitation lost twice as much NO_3 as sites with <800 mm annual precipitation (Eagle et al. 2017).

Environmental factors

Soil carbon: An increase in soil carbon by 10 g/kg reduced nitrate leaching loss by 8-13 kg N/ha/year (Eagle et al. 2017).

Soil texture: Coarse-textured soils have greater nitrate leaching losses than fine-textured soils due to the higher drainage rates and denitrification potential in coarse-textured soils (Di and Cameron 2002).

Crop type

On an area-scaled basis, a higher percentage of N applied to corn fields was lost as nitrate than N applied to wheat fields, partially due to higher rates of fertilizer applications to corn systems; yield-scaled nitrate leaching losses were similar for corn and wheat (Zhou and Butterbach-Bahl 2014).

A meta-analysis in irrigated systems found that strategies to reduce nitrate leaching were more effective in cereal crops than in vegetables (Quemada et al. 2013).

Agricultural management techniques

Irrigation: Compared to non-irrigated systems, irrigated systems lost an additional 11 kg N/ha/year through nitrate leaching (Eagle et al. 2017). A meta-analysis that compared fertilizer management strategies to water management strategies in irrigated systems found that improved water management has a greater effect on nitrate leaching rates than improved fertilizer management does (Quemada et al. 2013).

Drainage systems: The use of drainage systems increases nitrate leaching losses, likely due to increased N mineralization (Di and Cameron 2002).

Cover crops: The use of cover crops decreases nitrate leaching losses by up to 50% (Di and Cameron 2002; Quemada et al. 2013). Two meta-analyses showed that planting non-legume catch crops reduced soil nitrate concentrations by 35% and nitrate leaching losses by 50% compared to controls with no catch crops (Valkama et al. 2015).

Sources

3b: NO_3 runoff and leaching → NO_3 levels (surface and drinking water)[7]

Description of relationship

Nitrate leached from agricultural soils moves through groundwater and into surface water. Computer models can be used to predict nitrate concentrations at particular locations.

Summary of evidence

Nitrate leached into groundwater can be transformed into other forms of nitrogen or transported throughout the groundwater and surface water systems. Most nitrate in surface water is from groundwater discharged into the surface water (Almasri and Kaluarachchi 2007). Due to the localized nature of groundwater flows and groundwater-surface water interactions, computer models must be used to assess the impacts of nitrate leaching at a particular site on nitrate levels in groundwater and surface water. Models developed by the USGS (MODFLOW and MT3D) are widely used for modeling groundwater flow, the transformation and transport of nitrate within groundwater, and nitrate concentrations in groundwater and surface water on the watershed scale (Bedekar et al. 2016). The Soil and Water Assessment Tool (SWAT) can also simulate nitrate concentrations in waterways based on land use, management, and soil and has been used at broad spatial scales (Rabotyagov et al. 2010). Model verification (by comparing simulated nitrate concentrations with measured nitrate concentrations) has shown good results when the model is calibrated to the study site using measured nitrate concentrations (Almasri and Kaluarachchi 2007).

[7] The effect of fertilizer management on NO_3 levels in bodies of water has been separated into two links (3a and 3b) because the evidence for effects of fertilizer management on NO_3 leaching (captured in link 3a) is primarily at the field scale. An additional link using hydrologic modeling to represent the movement of NO_3 after it is leached from the soil is necessary to estimate the changes to NO_3 concentrations in bodies of water.

Strength of evidence

Moderate: Nitrate that moves from agricultural soils to groundwater and surface water will increase the nitrate concentrations in the body of water, but processes occurring during the movement of leached nitrate determine how much of the nitrate reaches a water body. Therefore, any reduction in the amount of nitrate leaching is expected to reduce the nitrate concentration of a water body to which the leachate moves, but modeling is required to determine which water bodies will be affected and the extent to which their nitrate concentrations will be influenced.

Predictability: The MODFLOW and MT3D models are widely accepted and used to simulate nitrate movement through groundwater and surface water. However, they do not perfectly predict nitrate concentrations in water sources, and site-specific factors can decrease their accuracy.

Sources

3c: NO_3 levels (drinking water, private) → Human health

Description of relationship

The ingestion of nitrate in drinking water may increase the risk of methaemoglobinaemia and certain cancers.

Summary of evidence

The two main human health effects associated with nitrate ingestion are methemoglobinemia and cancers of the digestive tract (Powlson 2008). Methemoglobinemia, also known as "blue baby syndrome" as it primarily occurs in infants, occurs when nitrate is converted to nitrite in the intestinal tract, which then binds to hemoglobin to form methemoglobin, which interferes with blood's capacity to carry oxygen (Ward et al. 2005). The current drinking water standards for nitrate are primarily based on several studies from the 1940s and 1950s that found a link between infant nitrate ingestion (via water used in formula) and methemoglobinemia cases (Powlson 2008). However, this link has been recently questioned by researchers who looked back at the data from those studies and noticed that all of the wells associated with infant methemoglobinemia cases not only had high nitrate concentrations, but were also contaminated with human or animal waste (Powlson 2008). Other studies have shown that methemoglobinaemia can be caused by gastroenteritis, so it is possible that nitrates were not the primary cause of those cases. There have been few recent studies of nitrate-caused methemoglobinemia; several showed no relationship between nitrate ingestion and methemoglobin levels, one showed a positive correlation between nitrate ingestion and methemoglobin but no clinical signs of methemoglobinemia, and one showed a positive correlation between nitrate ingestion and clinical methemoglobinemia, but patients also showed signs of gastroenteritis (Ward et al. 2005). More research is needed to determine whether ingestion of nitrate in drinking water contributes to methemoglobinemia, and the dose-response relationship between nitrates and methemoglobinemia cases.

There are some indications that nitrate ingestion may contribute to other human health effects, including cancers and reproductive health, but more research is needed (van Grinsven 2006). In particular, nitrate can be converted to N-nitroso compounds (NOC) in the stomach, which are likely carcinogens, but there is inconsistent evidence on whether cancers or other health problems actually result from NOCs (van Grinsven 2006; Powlson 2008; Villanueva et al. 2014; Ward et al. 2005).

Strength of evidence

Low: The evidence for this relationship is dated and has been called into question by more recent studies that suggest nitrate ingestion is not a primary cause of methemoglobinaemia in infants.

Sources

3d: NO_3 levels (drinking water, private) → Water treatment costs (private)

Description of relationship

When nitrate concentrations in private wells exceed 45 mg/L (the current Canadian drinking water standard), and the owner detects the high concentration and decides to treat the water, the cost to install a treatment system to lower nitrate concentrations is $150-3,000, plus maintenance costs.

Summary of evidence

About 1 in 8 Canadians use private water supplies, usually from rural groundwater wells (Charrois 2010). As described in link 3e, treatment is recommended when nitrate concentrations in drinking water exceed the 45 mg/L standard. While people who get water from private wells are legally responsible for the quality of their water supplies, there is limited oversight of private water supplies, with local health authorities generally providing guidance related to water testing and, in some cases, subsidized testing.

Studies show that many owners of private water supplies test for contaminants less frequently than recommended (Charrois 2010; Hexemer et al. 2008). Two surveys of private water users in Canada found that 45% of households had their water tested in the previous year, and that only 8% of private water users in Ontario met the local recommendation for testing frequency (Charrois 2010). A study designed to remove barriers to testing by sending well owners testing kits and processing samples free of charge had a response rate of just under 50%, suggesting that cost and inconvenience both play a role in discouraging private water supply testing (Hexemer et al. 2008).

Several options are available for private water users who find high nitrate concentrations in their well water to reduce those concentrations. Residential-scale reverse osmosis and ion exchange systems are very effective in removing nitrates. Point-of-use reverse osmosis systems can be installed underneath a single faucet that is used for drinking and cooking water and cost $150-500; filters must be replaced periodically (usually every 1-2 years) at a cost of $50-100 (prices from August 2017 Amazon listings for the recommended RO systems on ro-system.org). Whole-house ion-exchange systems cost about $1500-3000, depending on household water use (raindancewatersystems.com/nitrate.html).

Strength of evidence

Fair: To assess the change in water treatment costs to private water supply owners from increased nitrate levels, two components are required: the number of affected people who will detect the increased nitrate levels and treat for them, and the cost of a household water treatment system that is capable of removing nitrates. Few studies of the proportion of private water supply owners that regularly test their water quality have been conducted, and no studies were found that evaluated the proportion that follow through on test results by installing water treatment systems were found. The costs of particular water treatment systems are available, but many options are available that result in a wide range of costs.

Other factors

Location

Local recommendations for private water supply testing and programs to offset testing costs may influence testing frequencies (Charrois 2010).

Water quality

The reverse osmosis and ion exchange systems described here have specific requirements for the quality of incoming water; if other contaminants are also present, pre-treatment may be required, increasing the total cost of the system.

Sources

3e: NO_3 levels (drinking water, municipal) → Water treatment costs (municipal)

Description of relationship

When the nitrate concentration in public water supplies exceeds 45 mg/L (the current Canadian drinking water standard), a water supplier must spend more money to treat the water so it meets drinking water standards.

Summary of evidence

Canadian drinking water standards set the maximum acceptable concentration (MAC) of nitrate at 45 mg nitrate/liter of water, which is equivalent to 10 mg nitrate-nitrogen/liter of water (Health Canada 2013). When drinking water supplies exceed the MAC for nitrate, water suppliers must take action to ensure that water delivered to customers meets drinking water standards. There are several ways to accomplish this: switching to a new source of water (if available), blending high-nitrate water with lower-nitrate water so that the final product meets drinking water standards (this can also require a new water source, depending on the existing water sources' nitrate content), and nitrate removal. Conventional municipal water treatment processes (coagulation, sedimentation, filtration, and chlorination) do not remove nitrate, so nitrate removal usually requires the water supplier to invest additional resources. Several options are available for nitrate removal, including ion exchange, reverse osmosis, electrodialysis, and biological denitrification. Costs to install and run these systems are dependent on many factors, but a 2012 overview of water treatment for nitrate in California provided cost estimates for these technologies based on existing installations. These estimates are based on small sample sizes in many cases and do not take into account differences in water quality, system configuration, and waste management (Jensen et al. 2012). Table 6 gives the total annualized cost of various nitrate removal technologies for water suppliers based on the volume of water processed each day (millions of gallons per day, MGD).

Table 6. Annualized cost of different nitrate removal technologies for municipal water suppliers

Nitrate removal technology	**Total annualized cost ($USD/1000 gallons)**		
	<0.5 MGD	**0.5-5 MGD**	**>5 MGD**
Ion exchange	0.97-5.71	0.74-2.19	0.65-1.44

Reverse osmosis	5.73-19.70	2.52-3.21	2.58
Biological denitrification	1.13	1.03-1.13	1.25-1.56

Very little information is available for the cost of electrodialysis systems; a manufacturer estimated the capital investment for a 0.5 MGD system in 2005 was about $475,000 (USD), and the operation and maintenance costs for one installed electrodialysis system was $1.41/1000 gallons (Jensen et al. 2012).

While blending can be more cost-efficient than the treatment options described above, it requires more complex management (especially when the amount of water from different sources and nitrate levels in those sources varies over the year) and is less reliable long-term, since it depends on consistently low nitrate levels in the alternate water sources with which high-nitrate water is blended (Jensen et al. 2012).

Strength of evidence

Moderate: Though limited evidence for nitrate removal system installation and operations costs was found, since conventional municipal water treatment processes do not remove nitrate, we can be confident that in the vast majority of cases, an exceedance of the maximum acceptable concentration for nitrate in drinking water supplies will result in additional water treatment costs via implementation one of the management strategies described above.

Other factors

Water quality: Water sources with higher nitrate concentrations will likely cost more to treat, but there is little data available to assess the magnitude of this effect. Jensen et al. (2012) used vendor cost estimates to calculate approximate ion exchange nitrate removal treatment costs for water with initial nitrate levels of one, two, or three times the maximum contaminant level (MCL) (Table 7).

Table 7. Annualized cost estimates for ion exchange nitrate removal as affected by system size and input nitrate concentration. Values shown are the range in cost ($/1000 gallons), with average in parentheses.

Raw nitrate level	System size			
	Very small	Small	Medium	Large
1X MCL	0.62-4.60 (1.22)	0.34-2.73 (1.05)	0.36-2.04 (1.06)	0.22-1.81 (0.97)
2X MCL	0.69-11.27 (2.88)	0.38-7.33 (1.70)	0.39-5.00 (1.60)	0.25-4.24 (1.46)
3X MCL	0.76-17.94 (3.80)	0.42-11.94 (2.36)	0.42-7.96 (2.32)	0.29-6.68 (1.96)

Source: Jensen et al. (2012)

Water sources: The availability of other sources of water with lower nitrate concentrations determines a water supplier's ability to use blending instead of a nitrate removal system.

System size: It is generally more cost-efficient for large water suppliers to install and run nitrate removal systems than it is for smaller water suppliers, as evident in the cost estimates for various system sizes presented above.

Sources

3f: NO_3 levels (surface water) → Algal biomass

Description of relationship

NO_3 concentration in surface waters is positively correlated with algal biomass in aquatic systems.

Summary of evidence

An increase in algal biomass due to higher NO_3 concentrations depends on at least some of the algae species present being limited by nitrogen availability. While this is determined by many factors specific to the individual water body or waterway, there has been substantial controversy about whether nitrogen or phosphorus is the primary limiting nutrient in freshwater and marine systems, and therefore which is more important to regulate (Lewis, Wurtsbaugh, and Paerl 2011). Historically, phosphorus has been considered the more limiting nutrient in freshwater systems, and nitrogen the more limiting nutrient in marine systems (Dodds and Smith 2016). However, more recent studies and syntheses have shown that both nutrients can be important limiting factors across ecosystems.

Compilations of field studies that measured chlorophyll concentrations (a proxy for algal biomass) as well as nitrogen and phosphorus concentrations have found that the increase in chlorophyll per additional unit of phosphorus is greater when nitrogen concentrations are higher, and that the increase in chlorophyll per additional unit of nitrogen is greater when phosphorus concentrations are higher (Dodds, Smith, and Lohman 2002; Smith 1982). Regression analyses modeling summer mean chlorophyll concentrations as a function of summer mean total P and total N concentrations gave concordant results (Figure 2). Similarly, a meta-analysis of the effects of nitrogen and phosphorus availability on primary producer biomass across freshwater and marine systems showed substantially greater responses to the simultaneous addition of both nitrogen and phosphorus than to either nutrient added individually (Figure 3).

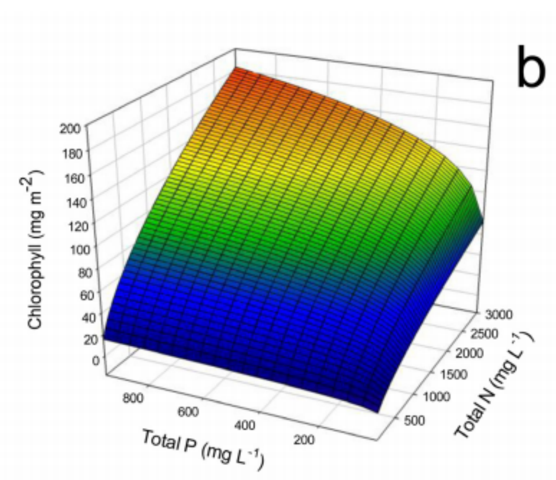

Figure 2. Modeled summer mean chlorophyll concentrations as affected by mean total P and total N concentrations. Source: Dodds et al. (2016).

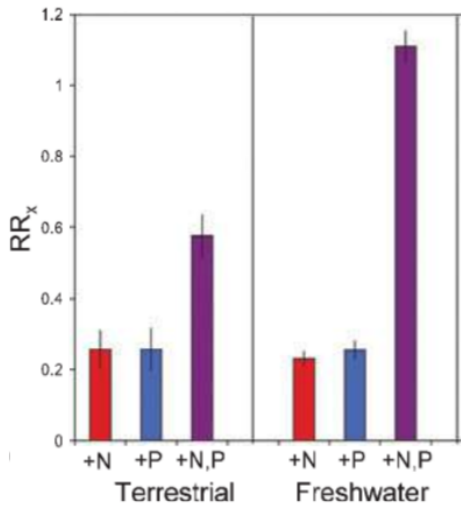

Figure 3. Response ratio of autotroph biomass to nitrogen, phosphorus, or combined nitrogen and phosphorus addition in terrestrial and marine ecosystems. Response ratios were calculated by dividing the autotroph biomass in the enriched treatment by the autotroph biomass in the control and taking the natural log of the result. Source: Elser et al. (2007).

These results suggest that the levels of nitrogen and phosphorus in aquatic systems are usually balanced, so that addition of one nutrient without the other causes the other to become limiting (Elser et al. 2007). The "balanced" N:P molar ratio for algal growth is 16:1, so the response of aquatic autotroph biomass to nitrogen additions would be expected to decrease as the N:P ratio in the water body exceeds this (Dodds and Smith 2016).

Quantitative models such as the Environmental Fluid Dynamics Code (EFDC) and the Water Quality Analysis Simulation Program (WASP) can predict chlorophyll a concentrations in water bodies based on many local input variables including nitrogen concentrations. These models are data-intensive and require field sampling for model calibration, so while they have been shown to be fairly accurate for individual water bodies over short time periods, they are unlikely to be useful for predicting this relationship at a broader scale (Wu and Xu 2011; Wool, Davie, and Rodriguez 2003).

Strength of evidence

Moderate: The importance of nitrogen as a limiting nutrient for algae in aquatic ecosystems has been demonstrated through field studies and syntheses in a variety of locations, but there is still controversy among some scientists about the relative importance of nitrogen compared to phosphorus. In addition, it is difficult to state a general relationship between nitrogen levels and algal biomass due to the many factors (including phosphorus levels) that can influence the relationship.

Predictability: Models allow the prediction of chlorophyll a concentrations in individual water bodies, but are data-intensive and often require field sampling.

Other factors

Water body type

Eutrophication in streams has been less well studied than in lakes, but there is some evidence that the nutrient thresholds that result in increased primary production are higher in streams than in lakes (Lewis, Wurtsbaugh, and Paerl 2011).

Autotroph species
Algal species differ in their ability to take up nutrients and optimal N:P ratios, so different nutrients may be limiting to different species under given conditions, and individual species will not respond identically to nutrient additions (Lewis, Wurtsbaugh, and Paerl 2011).

Climate
There is a positive correlation between the additional chlorophyll yield per unit of total nitrogen and the water temperature (Dodds, Smith, and Lohman 2002). This may partially explain the negative correlation between chlorophyll yield per unit of total nitrogen and latitude, but other studies have found that latitude has no effect on nutrient limitation (Dodds, Smith, and Lohman 2002; Elser et al. 2007).

Sources

3g: Algal biomass → Water treatment costs (municipal)

Description of relationship
Increased algal and cyanobacterial growth increases water treatment costs by clogging filters and slowing the treatment process.

Summary of evidence
High levels of algal and cyanobacterial biomass in municipal water supplies can cause clogs in filters, increase the filtration time, and increase the use of water and energy to produce drinking water (Walker 1983; Shen et al. 2011). A survey of 172 US water treatment plants published in 2004 found that 73% of respondents had algae-related problems (Knappe et al. 2004). Of those, 48% of those had issues with filter clogging, 36% with increased coagulant demand, and 50% with increased chlorine demand. Common changes to water treatment processes to address algae-related issues included treatment with copper sulfate, powdered activated carbon, chlorine, and potassium permanganate. The reason for each change (whether to address filter clogging or another algae-related issue) was not specified, but additional chlorine is noted as helpful for preventing filter clogging (Knappe et al. 2004).

In addition, some utilities implemented source water management strategies including aeration, changing or blending water sources, varying the elevation of water intakes, and pumping water to prevent stratification and algal growth ((Knappe et al. 2004). While the costs to water treatment plants of these treatment and source water strategies were not examined, any change to normal treatment processes or additional use of chemicals is expected to increase total costs. Additional chemical treatments are generally less expensive than changes to the normal water treatment process (Shen et al. 2011).

This link only considers water treatment issues arising from the physical presence of algal biomass in source water. Additional algae-related drinking water concerns include taste, odor, and algal toxins; these are addressed in links 3l and 3q.

Strength of evidence
Low: While it is clear that the presence of algal biomass in drinking water sources can cause issues with drinking water treatment, therefore increasing treatment costs, no evidence was found related to the concentration of algal biomass that causes filtration problems and requires additional treatment, or

about the additional costs associated with high algal biomass. It is likely that factors specific to individual treatment plants determine these outcomes, so it is not possible to make general predictions about the magnitude of effects on water treatment costs.

Sources

3h: Algal biomass → Water clarity

Description of relationship

Increased algal biomass in a body of water decreases water clarity.

Summary of evidence

Several studies have examined the relationship between chlorophyll *a* concentrations (a proxy for algal biomass) and Secchi disc depth (a measure of water transparency) for North American Lakes. Data from 193 natural lakes and 433 artificial lakes in the United State showed a strong negative relationship between chlorophyll *a* and Secchi disc depth (Figure 4). Thus, water clarity declines significantly with increased chlorophyll a concentrations (and increased algal biomass).

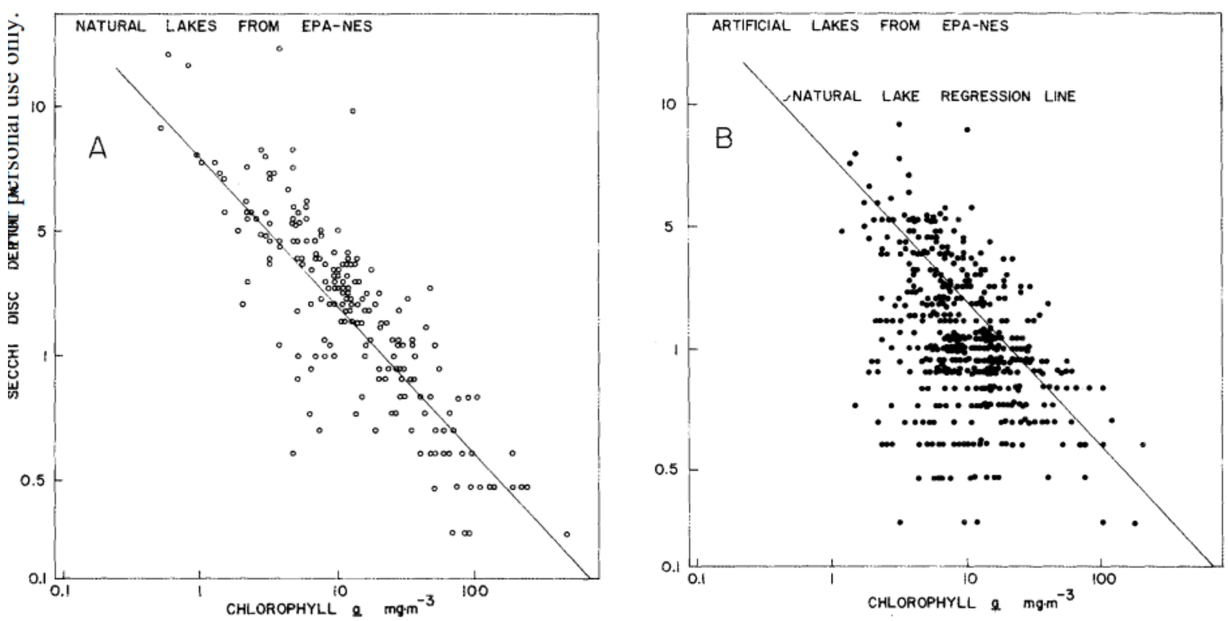

Figure 4. Relationship between chlorophyll a and Secchi disc depth in natural (A) and artificial lakes (B). Source: Canfield and Bachmann (1981).

A study combining two datasets, one with 125 lake-years for temperate lakes in North America and Europe, and another with 59 subtropical lakes in Florida, found a similar negative relationship (Mazumder and Havens 1998). A sample of 205 lakes in Florida also showed a sharp decrease in water clarity with chlorophyll *a* concentrations, leveling off at low clarity (< 1 meter) when chlorophyll concentrations exceeded approximately 10 mg/m^3 (Canfield and Hodgson 1983).

Strength of evidence

Moderate: While there are a few site-dependent factors (see below) that will influence the relationship between chlorophyll *a* and water clarity in a specific body of water, the three studies described above found generally congruent results with data from a wide variety of lakes, which establishes that there is a negative relationship between chlorophyll *a* and water clarity.

Predictability: The large-scale studies described above enable the estimation of the magnitude of expected water clarity effects from a given change in chlorophyll *a* concentrations.

Other factors

Lake type: A study using data from the EPA's National Eutrophication Survey found that the relationship between chlorophyll *a* and Secchi disc depth was much weaker in artificial lakes than in natural ones (see the difference between natural and artificial lakes in Figure 4) (Canfield and Bachmann 1981). This suggests that particulates not associated with algae play a greater role in water clarity in artificial lakes than in natural ones.

Climate: One of the studies described above classified lakes as temperate or subtropical (Florida). For a given chlorophyll *a* concentration, the temperate lakes had clearer water (greater Secchi disc depth) than the subtropical lakes (Mazumder and Havens 1998).

Grazer community: One study also classified lakes as containing large herbivores (*Daphnia*, which are thought to regulate phytoplankton growth) or small herbivores (no *Daphnia*). For a given chlorophyll *a* concentration, the temperate lakes with large herbivores had clearer water (greater Secchi disc depth) than the temperate lakes with small herbivores (Mazumder and Havens 1998). All subtropical lakes were classified as small herbivore because the only native species of *Dapnhia* is small and not usually present during the summer.

Sources

3i: Water clarity → Property values

Description of relationship

Increased clarity of a body of water can enhance the value of nearby properties, and decreased clarity can degrade the value of nearby properties.

Summary of evidence

The value that people hold for clear bodies of water can be seen in their willingness to pay more for properties on clear bodies of water than for properties on low-clarity bodies of water, all other factors being equal. Several researchers have assessed this relationship by using data on property characteristics and sale prices with water clarity data to construct hedonic models, which model the price of a property as a function of many characteristics, including water clarity.

A study of lakefront property sales on 25 Maine lakes from 1990-1995 found a positive effect of lake water clarity on property values (Boyle, Poor, and Taylor 1999). The change in value per meter of water clarity gained or lost varied by market group, but in general, the loss of value from a reduction in water clarity is greater than the gain in value from an equivalent improvement in water clarity. This suggests that protecting existing water clarity is more valuable than improving water clarity (Boyle, Poor, and Taylor 1999).

A study of lakefront property sales on 37 Minnesota lakes (1996-2001) used similar methods as the Maine study and also found that water clarity was a significant explanatory variable for lakeshore property prices (Krysel et al. 2003). The specific change in property value for a 1-meter increase or decrease in water clarity varied by the lake in question. Again, the absolute change in value for an improvement in water clarity was always less than that for a reduction in water clarity.

Both of the studies described above only examined changes in value for lakefront properties. However, if lake water clarity also influences the values of non-lakefront properties located near a lake, those could have a greater effect on the total valuation of water clarity than the lakefront properties, which tend to be more expensive than non-lakefront properties but make up a small proportion of all of the properties surrounding a lake. A study of 54,000 property sales within one kilometer of a lake in Orange County, Florida, examined the effect of water clarity on the values of both lakefront and non-lakefront properties (Walsh, Milon, and Scrogin 2011). Results showed a positive effect of lake water clarity on property values for both lakefront properties and non-lakefront properties within one kilometer of a lake.

Strength of evidence

Moderate: Multiple studies using established analytical techniques and large databases of property sales have found evidence of a positive relationship between water clarity and property values, but large differences in property values across real estate markets and specific lakes make it difficult to extrapolate these findings beyond the specific contexts in which the studies were conducted. Therefore, the evidence summarized here provides a strong basis for stating that a positive relationship between water clarity and property values exists, but not for assessing the magnitude of that relationship in general.

Predictability: The hedonic models described above can be used to predict the effect that a given change in water clarity will have on property values in the contexts in which they were conducted, but not for predicting the effect that a given change in water clarity will have on property values on a broad scale or in other contexts (different real estate markets, lake attributes, etc.).

Other factors

Location: Local real estate markets and overall property values influence the specific effect of water clarity changes on property values. The Maine and Minnesota studies described above both found large differences in property value changes for a standard change in water clarity across real estate markets and lake groups (Boyle, Poor, and Taylor 1999; Krysel et al. 2003).

Property distance from lake: The Florida study found that enhanced water clarity has a larger positive impact on lakefront property values than on non-lakefront property values, and that the effect of water clarity on non-lakefront property values declines with the property's distance to the nearest lake (Walsh, Milon, and Scrogin 2011).

Lake characteristics: Increased water clarity has a larger positive impact on property values for properties near a larger lake than near a smaller lake (Walsh, Milon, and Scrogin 2011).

Sources

3j: Water clarity → Recreation

Description of relationship

Low water clarity can make water-related recreation unsafe and unappealing, therefore reducing the number of recreational visits to a water body.

Summary of evidence

Clear water in bodies of water used for recreation is more visually attractive to recreationists and makes recreation safer by allowing people to estimate the depth of the water, see underwater hazards, and locate submerged people. Several groups of researchers have taken various approaches to measuring the effect of water clarity on recreational activity in water bodies.

A study in Finland combined a national survey on outdoor recreation activities, which included the total number of trips taken for various water-related activities, with field data on surface water quality, which included water clarity as measured by a Secchi disc (Vesterinen et al. 2010). While there was no relationship between the mean water clarity in water bodies near a person's residence and whether or not that person had engaged in water-related recreation, increased water clarity near a person's residence was positively correlated with the number of fishing and swimming trips taken by that person. Water clarity was not related to the frequency of boating trips, suggesting that boaters are not as concerned about water clarity as anglers or swimmers (Vesterinen et al. 2010). In another study, researchers modeled the recreational use of 129 lakes in Iowa based on a survey of 1,286 households and field measurements of water quality parameters including clarity (Secchi depth) (Egan et al. 2009). Lake clarity was positively correlated to the number of visits to the lake. Finally, a survey of water monitoring field crews in New York state found that low water clarity was a significant predictor of impaired recreational use (as judged by the field crews) for both primary and secondary contact uses at 203 wadeable streams (Kooyoomjian and Clesceri 1974).

Two studies by the same research group in New Zealand interviewed people at rivers and lakes about their perceptions of the water's suitability for bathing and clarity and conducted field measurements of water clarity (Smith, Croker, and McFarlane 1995; Smith, Cragg, and Croker 1991). Increased water clarity was correlated with increasing suitability for bathing; overall, water was perceived as suitable for bathing when clarity was at least 1.2 m (as measured by horizontal black disc visibility; equivalent Secchi disc visibility of 1.5 m). At this clarity, about 75% (Smith, Cragg, and Croker 1991) to 80% (Smith, Croker, and McFarlane 1995) of respondents thought that the water was suitable for bathing. A follow-up study that surveyed field staff who carry out water quality monitoring about their perception of suitability for bathing based on water clarity found similar results, with water considered marginally suitable when clarity exceeded 1.1 m and suitable when clarity exceeded 1.6 m (Smith and Davies-Colley 1992).

Strength of evidence

Fair: Multiple studies conducted in a variety of locations (Iowa, New York, Finland, New Zealand) confirm that people are aware of water clarity and prefer to recreate in clear bodies of water. The only studies found that specified a threshold water clarity for recreational use employed various methods and had highly consistent results, but were all conducted in New Zealand, which may not be applicable to preferences in Canada and the US (Smith and Davies-Colley 1992; Smith, Croker, and McFarlane 1995; Smith, Cragg, and Croker 1991). Local factors, particularly issues of substitutability, are likely to play a large role in the effect of water clarity on recreational use across broad scales (see other factors)

Other factors

Substitutability: The availability of alternate opportunities for outdoor recreation (water-related or not) can influence the effect that water clarity has on water-related recreation. In areas with many lakes and rivers that can be used for recreation, a reduction in water clarity of a few bodies of water may encourage people to shift their activities to other bodies of water rather than decrease total water-related recreation. In areas with a variety of other (non-water-related) outdoor opportunities, some people may respond to decreased water clarity by engaging in other types of recreation, therefore reducing the total recreational benefits lost due to decreased water clarity relative to areas without other types of recreational opportunities.

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3k: Algal biomass → Water taste/odor

Description of relationship

Algae produce chemical compounds that can affect water taste and odor.

Summary of evidence

Some species of algae produce metabolites that can affect the taste and smell of water. The specific chemical compounds produced depend on the type of algae present (Table 8).

Table 8. Common compounds produced by algae that affect water taste and odor. Source: Knappe et al. (2004)

Compound	Algae genera	Odor description

Geosmin	Anabaena, Aphanizomenon, Fischerella, Lyngbya, Oscillatoria, Phormidium, Schizothrix, Symploca	Earthy, corn, musty
2-methylisoborneol (MIB)	Oscillatoria, Phormidium, Pseudanabaena, Synechococcus	Earthy, musty
2t,4c,7c-decatrienal	Synura, Dinobryon	Fishy
2t,6c-nonadienal	Synura	Cucumber
Linolenic acid	Microcystis, Oscillatoria, Chlamydomonas	Sweet, melon, watermelon
B-cyclocitral	Microcystis, Oscillatoria	Sweet, fruity, chocolate, pipe tobacco
Isovaleric acid	Chlamydomonas	Rancid, sour

A 14-month study of algal growth, water quality parameters, and taste/odor complaints from water customers for a drinking water reservoir for the City of Wichita provides some support for the hypothesized link between algal biomass and taste and odor production (Smith et al. 2002). It found a positive relationship between the combined biomass of *Anabaena* and *Aphanizomenon* (odor-producing algae genera) and the number of taste/odor complaints. There was also a positive relationship between chlorophyll *a* concentration and the concentration of geosmin, the likely odor-causing compound (Smith et al. 2002). A model predicting odor production in Buffalo Pound Lake, a drinking water supply in Saskatchewan, found that the amount of algae from specific genera was an important predictor (more so than chlorophyll *a*, which is a proxy for the total algal biomass) (Kehoe, Chun, and Baulch 2015).

Strength of evidence

Fair: The ability of certain types of algae to produce compounds that alter the taste and odor of water is well understood and accepted, and the logic of a positive relationship between algal biomass and malodorous compounds is strong. However, there is little evidence to support that relationship, and the specificity of chemical compounds to particular algal species means that no general relationship can be stated between algal biomass and taste or odor. A model to predict odor production at a certain lake was fairly accurate, but requires a large amount of input data that is not available for most bodies of water, and has not been tested in other locations (Kehoe, Chun, and Baulch 2015).

Sources

3l: Water taste/odor → Water treatment costs (municipal)

Description of relationship

Actions taken by water suppliers to remove unpleasant tastes or odors from drinking water increase the cost of water treatment.

Summary of evidence

When people can taste or smell odors associated with algal growth in their drinking water, they often complain about it; as described in link 3k, the City of Wichita received complaints about water taste and odor corresponding with the amount of odor-producing algal biomass in the water supply (Smith et al. 2002). A 2004 survey of 172 US water treatment plants that draw water from rivers, lakes, and reservoirs found that 73% of respondents had algae-related water quality problems; of those, 91% had problems with water taste or odor (Knappe et al. 2004).

Link 3g describes a variety of actions taken by water treatment plants in response to algae-related problems, including taste and odor issues. The reason for each action (to deal with taste/odor, filter clogging, or other issues) is not specified, but the use of powdered activated charcoal is noted to be useful for controlling taste and odor problems in some cases (Knappe et al. 2004). In addition, as described in link 3g, water treatment plants may implement source water management strategies to prevent algae-related issues, including taste and odor problems, from occurring. While costs for these actions are not reported, any change from normal operation or the use of additional materials is assumed to increase the cost of water treatment.

This link only considers water treatment issues arising from the presence of algae-related taste and odors in drinking water. Additional algae-related drinking water concerns include physical clogging and algal toxins; these are addressed in links 3g and 3q.

Strength of evidence

Low: While it is clear that algae-related tastes and odors in drinking water can require additional treatment of drinking water, therefore increasing treatment costs, no evidence was found about the additional costs associated with algal tastes and odors or the frequency with which water suppliers experience taste and odor issues severe enough to require additional treatment. It is likely that factors specific to individual treatment plants determine these outcomes.

Sources

3m: Water taste/odor → Property values

Description of relationship

Unpleasant water taste or odor may reduce the market value of properties near a body of water.

Summary of evidence

This link seems logical, but no evidence was found to support it. Most studies that examine the effect of eutrophication-related water quality issues on property values used other characteristics (often water clarity, sometimes nutrient concentrations) to indicate for eutrophication (Krysel et al. 2003; Walsh, Milon, and Scrogin 2011). This may be because odor is less straightforward to measure in a standardized way than other water quality parameters (Kehoe, Chun, and Baulch 2015). In a 1970 survey of homeowners on four lakes in New York found that some property owners did report strange odors as an objectionable quality of the lake, but did not connect those results to a measure of odor or to any data on property values (Kooyoomjian and Clesceri 1974). The effect of another aspect of eutrophication on property values is captured in link 3i (water clarity → property values).

Strength of evidence

None: No evidence was found that directly assessed this relationship.

Sources

3n: Water taste/odor → Recreation

Description of relationship

Unpleasant water taste or odor may discourage people from using a body of water for recreation.

Summary of evidence

While this link seems logical, there is little evidence available to support it, and what evidence does exist suggests that water odor is less important to recreationists than water appearance. A survey of 240 people was conducted at three state park lakes in Minnesota, one of which had an algal bloom. While the researchers noted a "strong, offensive odor" at that lake, only 6% of respondents at that lake detected an odor, and 91% reported that there was no odor (Nicolson and Mace 1975). Many more respondents noted the lake's color as a potential indicator of pollution. A survey of water monitoring field crews in New York state found that odor was a significant predictor of impaired recreational use (as judged by the field crews), but not as strong a predictor as water clarity, at 203 wadeable streams (Smith, Duffy, and Novak 2015). Both of these studies suggest that people are more aware of visual indicators of water pollution than of non-visual characteristics.

A survey carried out at four lakes in New York in 1970 and 1971 interviewed homeowners, recreationists, and anglers about the aspects of the lakes they found objectionable (Kooyoomjian and Clesceri 1974). Recreationists selected taste and odor as objectionable characteristics more often than anglers did, likely because they have more direct contact with the water than the anglers.

Strength of evidence

Low: Very little evidence for this link was found, and two of the three sources cited here are more than 40 years old. Based on the available evidence, it is not possible to assess the magnitude of the effect of taste and odor on recreational activities, or to understand additional factors that may influence this relationship.

Sources

3o: Algal biomass → Toxins

Description of relationship

Many types of algae produce toxins; the species and amount of algae will determine the type and concentration of toxins in a body of water.

Summary of evidence

Many freshwater algae release secondary metabolites, some of which can adversely affect other organisms and are considered toxins (Leflaive and Ten-Hage 2007). There are potentially harmful species in most major freshwater algal groups; cyanobacteria, or blue-green algae, are generally considered the most hazardous (Paerl et al. 2001). Specific genera of toxin-producing cyanobacteria that are often found in freshwater are *Microcystis* and *Planktothrix*, which both produce microcystin (a hepatotoxin), and *Nodularia*, which produces nodularin (also a hepatotoxin) (Ibelings and Chorus 2007).

There are currently no laws requiring testing for algal toxins in recreational water bodies or drinking water in the United States and Canada, so there is limited data about the prevalence and concentration of algal toxins (Weirich and Miller 2014; Orihel et al. 2012). The types and amounts of algal toxins commonly found in North American water bodies are unknown (Hudnell 2010). The best available dataset on algal toxins in Canadian freshwater bodies is a 2012 compilation of microcystin measurements in Canadian freshwater dating from 2001 to 2011 (Orihel et al. 2012). These data were collected from academic studies, government agencies, and private companies and includes microcystin measurements from 246 water bodies across Canada, with measurements from every province. In total, 41% of lakes had microcystin concentrations > 1 µg/L, and 9% had concentrations > 20 µg/L (Orihel et al. 2012). The most comprehensive study found of microcystin prevalence in US freshwaters sampled algae from 224 lakes in Missouri, Iowa, Kansas, and Minnesota and found known toxin-producing cyanobacteria in 91% of lake visits. However, microcystin concentrations were generally low; only 2% of samples had microcystin concentrations greater than 1000 ng/L (Graham et al. 2004).

This link only considers water treatment issues arising from the presence of algal toxins in source water. Additional algae-related drinking water concerns include taste, odor, and physical clogging; these are addressed in links 3g and 3l.

Strength of evidence

Fair: While it is true that increasing amounts of toxin-producing algae in a body of water correspond with increased concentrations of the toxin, it is not possible to assess the relationship between total algal biomass and the type or concentration of algal toxins. The specific type of algae must be identified to determine whether it is a toxin-producing species. However, the presence of certain environmental conditions (see “other factors”) in conjunction with a large amount of algal biomass can suggest that algal toxins are likely to be present.

Other factors

Both of the studies described above found that the N:P ratio explained a significant proportion of microcystin concentrations, with more microcystin present when N:P ratios were low (Graham et al. 2004; Orihel et al. 2012). A regression tree analysis done for the Canadian study found that both high nutrient levels and a low N:P ratio were necessary for high microcystin concentrations, with microcystin concentrations only exceeding 0.39 µg/L (+/- 0.99 µg/L) when N concentrations exceeded 2600 µg/L and the N:P ratio was less than 23 (Orihel et al. 2012). This may be due to a competitive advantage for cyanobacteria (some of which can fix nitrogen) under low-nitrogen conditions, but this hypothesis has been debated. It is also possible that large amounts of cyanobacteria lower the N:P ratio by taking up P from sediments and increasing total P in the water column (Orihel et al. 2012).

Sources

3p: Toxins → Human health

Description of relationship

Human exposure to algal toxins can occur through drinking water, recreational water bodies, and food from aquatic ecosystems, and can cause a variety of symptoms depending on the specific toxin.

Summary of evidence

People can be exposed to algal toxins in several ways. Toxins in drinking water can be directly ingested, and swimming in bodies of water containing toxins can lead to accidental ingestion and skin exposure. Cyanotoxins can also be inhaled when activities such as boating and waterskiing result in the aerosolization of toxins. Symptoms that have been associated with cyanotoxin exposure include skin rashes, gastrointestinal symptoms, upper respiratory symptoms, headache, pneumonia, and fever (Stewart et al. 2006).

Despite the widespread occurrence of cyanotoxins in North American freshwaters (see link 3o), there are few verified reports of illness or fatalities resulting from exposure to toxins. There are several reasons for this: recreational exposures to toxins are usually isolated events, symptoms are general and vary among individuals, and many of those affected do not recognize a connection to recreational water exposure. In addition, there are many other causes of illness associated with freshwater systems (bacteria, viruses, etc.) that cause similar symptoms, and toxins break down or transform quickly within the body, making them difficult to identify through toxicological tests (Weirich and Miller 2014).

There have been a few reports of illnesses after recreational exposure to freshwater in the United States. In 2004, a lake in Nebraska had microcystin levels > 15 ppb; while some warning signs were posted, these were not sufficient to prevent people from using the lake for recreation, and there were reports of flu-like illnesses following activities in the lake (Hudnell 2010). In 2002, several teenage boys developed gastrointestinal symptoms after swimming in a golf course pond in Wisconsin; those who had been completely submersed in the water or swallowed water had more severe symptoms. One boy died about 48 hours after exposure, and the cyanotoxin anatoxin-a was eventually implicated in his death, although this took a year to determine and has been questioned since (Weirich and Miller 2014; Stewart et al. 2006). Health Canada's guideline for total microcystins in recreational waters is 20 µg/L (Health Canada 2012).

There are also few reports of illness after toxin exposure through drinking water, but one survey of finished drinking water in the US found that 75% of the samples had detectable levels of microcystin (> 2 ng/L), and one sample had a microcystin concentration too high for human consumption (Weirich and Miller 2014). There is no regular testing of finished drinking water for algal toxins in the United States; it is thought that standard drinking water treatment removes some, but not all, toxins, and that the efficacy depends on a variety of factors (specific treatment process used, toxin type and concentration, and water pH, organic matter content, and temperature) (Weirich and Miller 2014). In 1979, 148 residents of Palm Island, Australia, became sick after their water supply (which was undergoing an algal bloom) was treated with copper sulfate to improve taste and odor. The copper sulfate caused algal cells to lyse and release their toxins en masse (Hawkins et al. 1985). A study in China suggested that long-term consumption of low levels of microcystin in drinking water is linked with an increased prevalence of primary liver cancer (Ueno et al. 1996). Health Canada has proposed a seasonal maximum acceptable concentration of 1.5 µg/L for total microcystins in drinking water (Health Canada 2016).

There is some evidence that algal toxins may accumulate in aquatic organisms that are later eaten by people, which would create another route of human exposure to toxins (Ibelings and Chorus 2007; Weirich and Miller 2014). No evidence of human health impacts from consumption of toxin-containing aquatic organisms was found.

Strength of evidence

Moderate: While there are not many reported cases of human illness or fatality resulting from exposure to algal toxins, it is clear from the literature and the cases that have been reported that algal toxins do pose a threat to human health when people are exposed to them in sufficient concentrations. Not enough evidence is available to assess the total effect on human health from algal toxins. Canadian standards for microcystin in drinking water and recreational water bodies can be used to determine whether toxins present in a particular body of water are likely to impact human health.

Other factors

Some subpopulations may be more sensitive to algal toxins than others; there have been some reports of allergy-like symptoms affecting some members of groups of people recreating together in a water body, while others were unaffected.

Sources

3q: Toxins → Water treatment costs (municipal)

Description of relationship

The presence of algal toxins in drinking water can require additional water treatment and disrupt drinking water supplies.

Summary of evidence

When the concentration of algal toxins in finished drinking water exceeds standards, water supply must be stopped until toxin concentrations can be reduced to acceptable levels (e.g., by additional processing, waiting for toxin concentrations in source water to decrease, or changing water sources). Studies on the efficacy of water treatment in removing algal toxins have focused on microcystins; activated carbon filtration appears to be the most effective treatment method, but this has not been established for other types of algal toxins (Weirich and Miller 2014).

A recent example of drinking water contamination with algal toxins occurred in Toledo in August 2014 (Carmichael and Boyer 2016). Toledo's drinking water comes from western Lake Erie, which was experiencing a *Microcystis* bloom. When microcystin levels above 1 µg/L (the World Health Organization drinking water guideline) were detected, the drinking water plant was shut down for three days, affecting 500,000 water customers as well as restaurants, breweries, and pools. The city responded by increasing activated carbon treatment to reduce microcystin concentrations below the guideline (Carmichael and Boyer 2016).

Strength of evidence

Low: While toxin concentrations exceeding drinking water standards in finished drinking water require action to be taken that is very likely to increase the cost of water treatment, the relationship between toxin concentrations in water sources, toxin concentrations in finished drinking water, and water treatment cost is not straightforward. In the Toledo case, the algal bloom in the water source was not especially large or severe, but winds pushed large amounts of algae into the water treatment system (Carmichael and Boyer 2016). The level of disruption to water provision and the cost to reduce toxin concentrations in drinking water depend on a variety of location-specific variables and would be difficult to predict in advance.

Sources

3r: Toxins → Recreation (boating, swimming, fishing)[8]

Description of relationship

Concentrations of algal toxins in waters used for recreation exceeding local regulations can cause those waters to be closed to recreational activities.

Summary of evidence

Regulations can require the closure of recreational water bodies and beaches when certain algal toxins are detected. In the summer of 2008, lakes, rivers, and/or beaches were closed or advisories posted in at least 13 states due to the presence of cyanobacteria or cyanotoxins (Graham, Loftin, and Kamman 2009).

Moratoria on fish and shellfish harvesting in response to certain toxins can also prevent or discourage recreational fishing and harvesting, depending on the specific regulation. Table 9 summarizes the type of toxins that can cause fishery closures, the reason for the closure, and areas that have been impacted.

Table 9. Algal toxins known to cause fishery closures in marine systems. Source: Interagency Working Group (2016).

HAB Taxa	Freshwater or marine	Toxin/ Bioactive Compound	Recreational impact	Impacted Areas in U.S.
Cyanobacteria	Freshwater	Microcystins, cylindrospermopsin, anatoxin-a, saxitoxins, geosmins, methylisoborneol	Fish kills, can make fish caught inedible (including bad taste)	Great Lakes and many inland water bodies
Haptophytes	Freshwater	Prymnesins, ichthyotoxins	Fish kills	States throughout US
Euglenophytes	Freshwater	Euglenophycin	Fish kills	Great Lakes, Texas, North Carolina
Dinoflagellates	Freshwater		Fish kills	

[8] This section was adapted from (Mason, Olander, and Warnell 2018 300 300).

Pseudo-nitzschia	Marine	Domoic Acid	Amnesic Shellfish Poisoning, Shellfish harvesting closure	West Coast, Florida, Maine
Dinophysis; Prorocentrum	Marine	Okadaic acid, Dinophysotoxins	Shellfish Poisoning, Shellfish fishery closure	Oregon, Texas, Washington
Karenia	Marine	Brevetoxins	Neurotoxic Shellfish Poisoning, Shellfish fishery closure, fish kills	Gulf of Mexico, Atlantic coast up to NC
Alexandrium; Gymnodinium; Pyrodinium bahamense	Marine	Saxitoxins	Paralytic Shellfish Poisoning, Shellfish fishery closure, some recreational fishery closures (puffer fish in FL)	Pacific coast (incl. AK), NE Atlantic coast, Florida
Karlodinium	Marine	Karlotoxins	Fish kills	Atlantic and Gulf coasts
Aureococcus anophagefferens	Marine	Not characterized	Shellfish die-off	Mid-Atlantic coast
Heterosigma akashiwo	Marine	Ichthyotoxin	Fish kills	Washington, mid-Atlantic coast
Alexandrium monilatum	Marine	Goniodomin	Fish and shellfish mortality	Gulf of Mexico and Atlantic Coast up to New Jersey
Cochlodinium	Marine	Not characterized	Fish kills	West Coast, Mid-Atlantic

Strength of evidence

Low: When algal toxins are detected in recreational waters where regulations exist, recreational activities are likely to be impacted. However, lack of consistency in monitoring and regulations makes it difficult to determine whether toxins will be detected and whether any applicable regulations are in force (see other factors). Therefore, the effects of algal toxins in recreational waters on recreational opportunities is very difficult to assess at a broad scale. At a very local scale, it may be possible to predict recreational outcomes of toxin occurrence.

Other factors

Regulations: The specific regulations in effect in a certain location will determine the impacts on recreational activities due to the presence of algal toxins. In the U.S., there are no federal regulations on algal toxins in recreational waters, so regulations vary by state (Graham, Loftin, and Kamman 2009). Canada has guidelines for cyanobacteria and toxin concentrations in recreational waters, but advisories and closures are left to local authorities (Health Canada 2012).

Monitoring: The type and frequency of monitoring for algal toxins in certain locations will determine whether the presence of certain toxins is detected, and therefore whether regulations that would impact recreational activities will be enforced. Monitoring of recreational waters in the U.S. varies by state (Graham, Loftin, and Kamman 2009). Health Canada recommends monitoring recreational waters known to be susceptible to algal blooms, but no systematic monitoring program exists (Health Canada 2012).

Substitutability: The presence of nearby alternate locations for recreational activities that are not affected by algal toxins can mitigate the overall effect of toxins on recreational opportunities.

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3s: Toxins → Aquatic species populations[9]

Description of relationship

Algal toxins can harm aquatic species, including fish, shellfish, marine mammals, reptiles, and sea birds.

Summary of evidence

Marine ecosystems

A report on harmful algal blooms by the National Science and Technology Subcommittee on Ocean Science and Technology provides a summary of the types of toxins produced by various algal taxa, the effects of those toxins on exposed aquatic organisms, and locations where impacts from the toxin have been observed (Table 10).

Table 10. Algal toxins and their effects on aquatic animals. Source: Interagency Working Group (2016).

Algal taxa	**Toxin/ Bioactive Compound**	**Animal Effects**	**Impacted Areas in U.S.**
Pseudo-nitzschia	Domoic Acid	Sea bird and marine mammal mortality	West Coast, Florida, Maine
Gambierdiscus; Prorocentrum; Ostreopsis	Ciguatoxins	Possible marine mammal illness	Florida, Gulf Coast, Hawaii, Pacific, Caribbean
Karenia	Brevetoxins	Fish kills, manatee, dolphin, marine turtle, and bird deaths	Gulf of Mexico, Atlantic coast up to NC
Alexandrium; Gymnodinium; Pyrodinium bahamense	Saxitoxins	Marine mammal deaths	Pacific coast (incl. AK), NE Atlantic coast, Florida

[9] This section was adapted from (Mason, Olander, and Warnell 2018 300 300).

Karlodinium	Karlotoxins	Fish kills	Atlantic and Gulf coasts
Aureococcus anophagefferens—Long Island Brown Tide	Not characterized	Shellfish die-offs	Mid-Atlantic coast
Akashiwo sanguineum	Surfactants	Migratory bird deaths	Pacific coast
Heterosigma akashiwo	Ichthyotoxins	Fish kills	Washington, Mid-Atlantic coast
Other Raphidophytes: Chattonella, Fibrocapsa	Brevetoxins; Ichthyotoxins	Fish kills	Mid-Atlantic coast
Alexandrium monilaturm	Goniodomin	Fish and shellfish mortality	Gulf of Mexico and Atlantic coast up to NJ
Cochlodinium	Not characterized	Fish kills	West Coast, Mid-Atlantic
Macroalgae	H2S, dopamine	Impair nesting protected species	All coasts

Source: Mason et al. 2018. Forthcoming.

Freshwater ecosystems

There are few reports of adverse impacts to freshwater aquatic species from exposure to algal toxins; most fish kills associated with algal blooms have been attributed to hypoxia (see links 3z and 3aa). Laboratory studies have shown that a variety of fish species, including rainbow trout, brown trout, common carp, and roach, have adverse responses to algal toxins; specific responses vary by the type and concentration of toxin, fish species, and fish life stage, but included reduced growth rates, developmental abnormalities, and mortality (Zanchett and Oliveira-Filho 2013; Malbrouck and Kestemont 2006).

Strength of evidence

Fair: While there is clear evidence for algal toxins' effects on aquatic species, organisms must be exposed to a specific toxin in order for effects to occur. The extent to which aquatic wildlife in a given location will be exposed to algal toxins depends on many factors (presence of susceptible wildlife species; type, extent, and concentration of algal toxins; water movement that disperses or concentrates toxins).

Sources

3t: Aquatic species populations → Species persistence[10]

Description of relationship

Decreasing the population size reduces the population's long-term viability (probability of persistence).

Summary of evidence

A reduction in the size of a wildlife population can influence the population's long-term viability in several ways. Population size thresholds represent a minimum viable size for a population of a given species to persist; if a population falls below that threshold, it will go extinct (Traill, Bradshaw, and Brook 2007). There are multiple reasons for the existence of population size thresholds. Demographic stochasticity (the probabilistic nature of reproduction and death) causes population size fluctuations that average out in large populations but can cause extinction in small populations. Allee effects refer to the positive effects of higher population density on processes that lead to individual fitness (e.g., finding

[10] Adapted from (Mason, Olander, and Warnell 2018 301 301)

mates, social dynamics, predator-prey interactions); at low population densities, these processes can break down (Kramer et al. 2009). A decline in the population size brings the population closer to its minimum viable size and lowers the probability of long-term persistence (Traill, Bradshaw, and Brook 2007).

Smaller populations also have reduced genetic diversity and inbreeding depression, which can decrease their probability of persistence (Frankham 2005). Loss of genetic diversity and inbreeding depression both depend on the effective population size (the number of adults that are actually breeding in the population) (Frankham 2005).

In laboratory studies, inbreeding depression has been shown to affect many aspects of reproduction and survival, decreasing overall fitness rates; subsequent research in captive and wild populations of wildlife species has shown that wildlife in natural habitats experience inbreeding depression (Frankham 2005). Few field studies have examined the effect of inbreeding depression on extinction risk for wild populations, but those that do exist have found a significant effect of inbreeding depression on extinction risk, and computer simulations of inbred populations showed that the median time to extinction was reduced by 25-31% relative to populations with no inbreeding depression (Brook et al. 2002). A later study that estimated the levels of inbreeding depression in wild populations using a meta-analysis found much higher inbreeding depression levels than was assumed in the Brook study. When population persistence was simulated using these results, it was found that the mean overall inbreeding effect seen in wild populations decreased the median time to extinction by 37% on average (O'Grady et al. 2006).

Lower genetic diversity limits the ability of the population to adapt to environmental change in the future through evolution. This effect takes place over a much longer time period than effects from inbreeding depression, and some studies have shown that inbreeding depression is likely a much stronger determinant of extinction risk than reduced genetic diversity (Frankham 2005).

Population viability analysis models are available to model the potential effects of population size decrease on long-term population viability; a long-term retrospective analysis of the predictive accuracy of these models found that they accurately predicted population sizes and growth rates (Brook et al. 2000). Population viability studies have been completed for multiple species of salmon (Ratner, Lande, and Roper 1997; Legault 2005), marine mammals (Burkhart and Slooten 2003; Heinsohn et al. 2006; Winship and Trites 2006) and commercially fished species (Curtis and Vincent 2008). A comparison of six population viability analysis models for the whooping crane found that the projected mean population size and extinction risk (after 50 years) varied among PVA packages, mostly due to differences in package features (Brook et al. 1999). When the models were standardized to remove these differences (essentially, the more complex features were simplified to match features available in the simplest models), results across packages were much more similar. Since researchers generally want to be conservative in modeling rare and threatened species, it is generally better to use the full models that include more potential threats. It is not usually known which of the models will provide the most accurate prediction for the species in question, so there is a moderate degree of uncertainty associated with population viability analysis.

Strength of evidence

Fair: As discussed above, there is evidence that small population size decreases the long-term persistence of a wildlife population in several ways. A meta-analysis of minimum viable population studies estimated a mean minimum viable population for various taxa, but found that minimum viable population is very specific to each individual population. Without knowing the minimum viable population size and history of a particular population, it is difficult to determine whether a given decrease in population size represents a minor fluctuation or a significant drop toward the minimum viable population.

Predictability: Population viability analysis models can predict the long-term effects of a population size decrease on viability and appear to be fairly accurate when adequate data on the focal population is available.

Other factors

Demographic factors

A population's demographic structure, including age distribution and background population level, influence its likelihood of persistence in the face of threats. Population viability analysis models take these factors into account.

Other threats

Populations affected by multiple threats at once are less likely to persist; some population viability analysis models include the effects of multiple threats to the population.

Sources

3u: Species persistence → Existence value[11]

Description of relationship

Some people hold non-use values for particular species and populations, and are willing to pay to increase the probability of the species' continued existence even if they do not personally expect to visit or live near the species. This is called "existence value" and has been measured via surveys in a few locations for certain species, but is highly context-dependent.

Summary of evidence

A variety of studies have examined willingness to pay for conservation efforts targeting certain species, with various outcomes including changes to population size, listing status, and probability of extinction in a given timeframe. However, none of these perfectly capture the non-use values of aquatic species; studies either fail to differentiate between use and non-use values or do not focus on relevant species. Nevertheless, they can be used to understand some of the general trends in willingness to pay for non-use values and as examples for how more targeted studies could be conducted.

A meta-analysis of willingness to pay studies for endangered and threatened species, including several aquatic species, in the US found that people were willing to pay $0.101 more for each 1% increase in population size that a particular program created; this reflects the total value of those species (not just

[11] This section was adapted from (Warnell, Olander, and Mason 2018 301 301).

existence value) and only applies to threatened/endangered species (Richardson and Loomis 2009). Whether a species had only nonuse or both use and nonuse value was included as a factor in this analysis; species with only nonuse values were valued at about $39 lower than species with both use and nonuse values, when all other factors were equal (Richardson and Loomis 2009). The other factors found to influence people's valuation of a species may also be relevant to nonuse valuation (see "other factors").

A stated preference choice survey estimated the willingness to pay for a dam removal and restoration project in the Klamath River Basin, based on changes to the extinction risk for two sucker species and the Coho salmon. Both use and non-use values for the target fish species were included in the survey and could not be separated. It found that the 20-year annual willingness to pay to reduce the extinction risk of the coho salmon was $21.28/household nationally, and that the 20-year annual willingness to pay to reduce the extinction risk of both the coho salmon and the suckers was $78.77/household nationally (Mansfield et al. 2012).

A choice experiment of the Canadian public's willingness to pay for the protection of four coastal fish species found in Canada (Atlantic salmon, Atlantic whitefish, porbeagle shark, and white sturgeon) showed that different segments of the population responded differently (different preferred species and importance of program attributes like change in population abundance, change in listing status, funding mechanism, and probability of program success) (Rudd 2009). This study did not separate use and non-use values for some species.

The USGS Benefit Transfer Toolkit provides estimates of total economic value for a variety of threatened, endangered, and rare species within the United States, based on a database of individual nonmarket valuation estimates (USGS n.d.). These estimates may include values other than existence value (e.g., recreational value), but they can provide a starting point for valuation of a particular species' persistence.

The stated preference choice survey conducted to assess the benefits from dam removal and restoration in the Klamath River Basin was well-designed and potential biases are clearly described; while it does not separate out non-use value, it is a good example of how a valuation study can be conducted for a specific species and location (Mansfield et al. 2012). New studies conducted according to established methodology would be expected to provide strong evidence for the non-use value of the studied population.

Strength of evidence

Low: Several studies have examined willingness to pay for conservation programs or for particular species, but many do not explicitly separate existence values from other types of value that wildlife can provide to people or associate values with changes in population persistence. Existence values from previous studies can be used as point estimates for existence values of non-studied populations, but contextual differences limit the accuracy of this method.

Other factors

Location

The distance between people and the population in question may influence the value they place on its continued existence (Loomis 2000). For example, the Klamath River Basin study found differences in responses among people within the Klamath area, outside of that area but within Oregon or California, and in the rest of the United States (Mansfield et al. 2012).

Sources

3v: Aquatic species populations → Recreation (fishing)

Description of relationship

An increase in the population of recreationally-fished aquatic species will result in an increase in angling-related recreation.

Summary of evidence

While it makes logical sense that recreational fishing would increase with fish populations, this relationship is difficult to quantify because of large uncertainties in data on both fish populations and fishing effort (Post 2013), and because the relationship goes both ways (fish populations may influence fishing activity, and fishing activity influences fish populations). Reports on changes in recreational fishing quality are rare, likely because recreational fisheries (especially freshwater fisheries) are small and distributed, and therefore each one only affects a few people; there are indications of declines in recreational fish stocks across Canada (Post et al. 2002). One study measured the relationship between fish density and angler density using data from 76 lakes in British Columbia stocked with rainbow trout and found that rainbow trout density (along with other factors, below) did influence recreational fishing activity (Post et al. 2008).

The harvest moratoriums in response to significant declines in fish stocks described in link 3w can also apply to recreational fisheries.

Strength of evidence

Low: This relationship makes logical sense, but little evidence exists to support it due to a lack of accurate data on fish populations and recreational fishing activity.

Other factors

Angler characteristics

A meta-analysis of anglers' willingness to pay to catch one additional fish showed that anglers who travel farther distances and those who take fewer total fishing trips are willing to pay more per fish than those who travel shorter distances to fish and take more total fishing trips (Johnston et al. 2006).

Fish species

Anadromous fish and big game fish are more highly valued by recreational anglers than other types of fish (Johnston et al. 2006).

Location characteristics

The study of stocked lakes in British Columbia described above found that the best model of fishing activity included not only rainbow trout density, but also the distance from the lake to a population center (Vancouver) and the presence or absence of a fishing lodge on the lake (Post et al. 2008).

Sources

3w: Aquatic species populations → Commercial fisheries

Description of relationship

An increase in the population of commercially-fished aquatic species will result in an increase in the commercial harvest of those species.

Summary of evidence

There is some evidence that when the abundance of a target fish species declines, so does the catch per unit effort (CPUE, the number of fish caught per a defined period of effort, such as trap-net nights). This means that as fish abundance decreases, fewer fish will be caught if the same fishing effort is expended; conversely, as fish abundance increases, more fish will be caught if the same effort is expended. While CPUE is often assumed to decline linearly with fish abundance, a meta-analysis comparing relative CPUE to relative abundance estimates found evidence for "hyperstable" relationships, or CPUE declining more slowly than fish abundance (Figure 5), in 70% of included datasets (Harley, Myers, and Dunn 2001). Therefore, CPUE cannot be used to predict fish populations; fish abundance may be quite low before a substantial decrease in CPUE is evident.

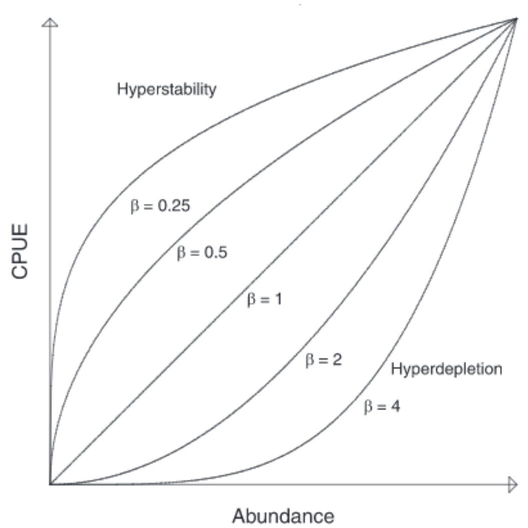

Figure 5. Possible relationships between fish abundance and catch per unit effort (CPUE).

A study of global fish harvest and fishing effort also supports this relationship, showing that total catches have remained constant despite increasing fishing effort, likely due to decreased fish abundance (Watson et al. 2013).

In addition, if populations of commercially fished aquatic species drop too low, commercial fishing may be prohibited until stocks recover. For example, a sharp decline in striped bass populations in the Chesapeake Bay during the 1980s resulted in moratoriums on harvest in adjacent states and decreases in harvest in other states, which were not lifted until populations had rebounded (Pendleton 2010). Any increase in the population of a target fish species can provide a buffer against a population decline that could prompt harvest prohibitions.

Strength of evidence

Fair: The general positive relationship between fish abundance and CPUE is supported by evidence from many individual studies, and prohibitions on harvest of certain species have been implemented in the past when populations crashed. These effects can result in increased harvesting costs or decreased harvest of commercial species.

Predictability: The specific shape of the relationship between fish abundance and CPUE is unclear, making it difficult to predict what the effect on CPUE will be from a certain population increase.

Other factors

The shape of the relationship between fish abundance and CPUE appears to vary by species (Harley, Myers, and Dunn 2001).

Sources

3x: Toxins → Commercial fisheries[12]

Description of relationship

Algal toxins can cause commercial fishery closures and loss of farmed fish.

Summary of evidence

Due to potential human health effects, certain commercial fisheries are often closed in response to the presence of algal toxins. A report from the National Science and Technology Council summarizes toxins produced by algae that have caused fishery closures, the reason for fishery closure, and areas that have been impacted in the past (Interagency Working Group 2016) (Table XX).

HAB Taxa	Freshwater or Marine	Toxin/ Bioactive Compound	Impact on fishery or fish farming	Impacted Areas in U.S.
Cyanobacteria	Freshwater	Microcystins, cylindrospermopsin, anatoxin-a, saxitoxins, geosmins, methylisoborneol	Makes farmed freshwater fish inedible (including bad taste)	Great Lakes, many inland water bodies
Haptophytes	Freshwater	Prymnesins, ichthyotoxins	Fish kills	States throughout US
Euglenophytes	Freshwater	Euglenophycin	Fish kills, loss of aquaculture operations	Great Lakes, Texas, North Carolina
Pseudo-nitzschia	Marine	Domoic Acid	Amnesic Shellfish Poisoning, Shellfish harvesting closure	West Coast, Florida, Maine
Dinophysis; Prorocentrum	Marine	Okadaic acid, Dinophysotoxins	Diarrhetic Shellfish Poisoning,Shellfish fishery closure	Oregon, Texas, Washington
Gambierdiscus; Fukuyoa	Marine	Ciguatoxins	Ciguatera Fish Poisoning, Bans on fish sales from affected areas	Florida, Gulf Coast, Hawaii, Pacific, Caribbean

[12] This section was adapted from (Mason, Olander, and Warnell 2018 300 300).

Karenia	Marine	Brevetoxins	Neurotoxic Shellfish Poisoning, Shellfish fishery closure	Gulf of Mexico, Atlantic coast up to NC
Alexandrium; Gymnodinium; Pyrodinium bahamense	Marine	Saxitoxins	Paralytic Shellfish Poisoning, Shellfish fishery closure	Pacific coast (incl. AK), NE Atlantic coast, Florida
Prorocentrum minimum—Mahogany Tides	Marine	Not characterized	Mortality of spat in shellfish hatcheries, lost shellfish	Chesapeake Bay

There have been reports of large-scale fish kills at aquaculture facilities due to algal toxins, including striped bass at a facility on the Chesapeake Bay and catfish in southeastern ponds (Deeds et al. 2002; Zimba et al. 2001).

A review of studies examining harmful algal bloom effects on commercial fisheries from 1987-1992 found annual costs in the US from $7-19 million (2000 USD) (Zimba et al. 2001). These effects include harvest losses, reduced sales, and farmed fish kills; some effects may be from other components of harmful algal blooms, such as hypoxia, so losses directly related to toxins are likely lower.

Strength of evidence

Fair: The closure of commercial fisheries due to algal toxins is a direct relationship, although no information was found about particular levels of algal toxins that cause fisheries to be closed. Algal toxins have also been shown to affect commercial fisheries via fish kills and harvest losses in various contexts, but the concentration of algal toxins required to cause an effect is unclear, and some evidence cannot differentiate between effects from algal toxins and effects from other algal bloom-related issues such as hypoxia.

Sources

3y: Toxins → Domestic animal mortality

Description of relationship

The exposure of domestic animals to algal toxins through drinking water can cause mortality.

Summary of evidence

Similar to human health effects from algal toxins (see link 3p), there is a lack of data about domestic animal mortalities from cyanotoxins, but case studies and reports show that domestic animals can be killed by drinking from bodies of water with toxic cyanobacteria.

Descriptions of domestic animal mortalities associated with cyanotoxin exposure date back to the 19th century (Stewart, Seawright, and Shaw 2008). Early reports noted the deaths of domestic and wild animals near water bodies with active algal blooms, sometimes with follow-up investigations exposing other animals to water from the algal bloom. More recent case studies that included more extensive diagnostic tests include a dog mortality in South Africa from nodularin, cattle deaths in Australia from cylindrospermopsin, and dog mortalities in France from anatoxin-a (Stewart, Seawright, and Shaw 2008). In North America, there have been reports of cattle, dogs, pigs, and ducks dying due to cyanotoxin exposure (Briand et al. 2003). Several of the studies on human exposure to algal toxins referenced in link 3p mention animal and livestock deaths as further support for human health effects being due to cyanotoxin exposure, or as a way of identifying lakes for further study that may contain algal toxins (Kotak et al. 2000; Hudnell 2010; Weirich and Miller 2014).

Strength of evidence

Fair: Reports of domestic animal poisoning from cyanotoxins show that algal toxins do pose a threat to domestic animals, but the lack of data on animal poisoning cases makes it difficult to assess the magnitude of the threat or to predict how animal poisoning cases might be affected if algal toxin concentrations or distribution were to change.

Sources

3z: Algal biomass → Oxygen

Description of relationship

The decomposition of large amounts of algae reduces the dissolved oxygen content in the water and can lead to hypoxic conditions.

Summary of evidence

The excessive growth of algae and cyanobacteria in aquatic systems (see link 3f) is eventually followed by a widespread die-off and subsequent decomposition of a large amount of organic material. When oxygen is consumed by aerobic decomposition of organic material more quickly than it is re-aerated through photosynthesis, the dissolved oxygen concentration in the water decreases (Rabalais et al. 2010). Under certain conditions (see “other factors”), this can create areas of hypoxic (< 2 mg oxygen/liter of water) or anoxic (no oxygen) water (Rabalais et al. 2010).

A variety of studies confirms the importance of the aerobic decomposition of organic matter in determining dissolved oxygen concentrations of estuaries and large lakes. A model for the Gulf of Mexico was able to explain 81% of the variation in the size of the hypoxic zone based on May nitrate levels, nitrate N loading, and percent biogenic silica (a proxy for organic matter) in sediments (Rabalais et al. 2010). Similarly, a model of dissolved oxygen concentration near the mouth of the Mississippi River showed that chemical-biological processes (the balance between photosynthesis and aerobic decomposition) control dissolved oxygen more strongly than transport processes (Bierman et al. 1994). A model that simulated nutrients and dissolved oxygen in the Chesapeake Bay found an annual cycle consistent with a decline in dissolved oxygen in the summer, following the die-off of a spring algae bloom, and that the lowest oxygen concentrations were found in the area of highest algal productivity (Cerco and Cole 1993). In non-estuarine systems, a model constructed to assess the effect of annual climate variability on hypoxia in Lake Erie found that the variation in production of organic matter, not climate variability, is the dominant driver of hypoxia in that lake (Rucinski et al. 2010).

Hypoxia is thought to be more likely to occur when organic carbon production is high, but other factors not related to carbon production play a role in controlling whether hypoxia occurs, so no straightforward predictive relationship between algal biomass production and dissolved oxygen concentrations can be stated (Rabalais et al. 2010).

Strength of evidence

Moderate: Field studies and models from several estuaries and large lakes confirm that aerobic decomposition of organic matter is an important cause of hypoxic conditions in water bodies, and the biological processes that consume oxygen during decomposition are well understood.

Predictability: Many other factors determine whether increased oxygen consumption results in hypoxic conditions. Therefore, it is not possible to make general predictions about dissolved oxygen levels or the presence of hypoxia based on algal biomass. Models have been developed to predict dissolved oxygen in particular water bodies, but these are data-intensive and their results are not transferable to other contexts.

Other factors

In large lakes and estuaries, dissolved oxygen concentrations do not decrease enough that the entire body of water becomes hypoxic; rather, a section of the water body separated (in terms of oxygen diffusion) from more aerated water experiences a more dramatic drop in oxygen concentration (Jenny et al. 2016). Therefore, hypoxia is more likely to occur under conditions that facilitate stratification, which is driven by temperature and salinity gradients and low rates of water exchange (Rabalais et al. 2010).

Hypoxia is more likely to occur with long water residence times and limited water exchange or flushing (Rabalais et al. 2010).

Sources

3aa: Oxygen → Aquatic species populations[13]

Description of relationship

Reduced dissolved oxygen concentrations adversely affect aquatic species, potentially causing behavioral changes, low growth rates, and death.

Summary of evidence

Aquatic organisms, including fish and crustaceans, require certain levels of dissolved oxygen in the water in order to maintain normal physiological functioning. Long-term hypoxic conditions can lead to aquatic ecosystems dominated by bacteria, and it can take multiple decades for the original community to recover after oxygen levels return to normal (Steckbauer et al. 2011). While hypoxia is traditionally defined as dissolved oxygen concentrations less than 2 mg O_2/liter of water, several recent meta-analyses have shown that many aquatic organisms are adversely affected at dissolved oxygen concentrations above this threshold (Steckbauer et al. 2011).

A meta-analysis of 872 experimental assessments of oxygen thresholds for 206 species of marine benthic organisms found no evidence for a universal dissolved oxygen threshold, but a large degree of variability in oxygen thresholds among species. Approximately 10% of experiments included in the analysis had oxygen thresholds greater than 5 mg O_2/liter (Vaquer-Sunyer and Duarte 2008). Another meta-analysis of effects of hypoxia on fish food consumption and growth found negative effects up to

[13] This section was adapted from (Mason, Olander, and Warnell 2018 300 300).

4.5 mg O_2/liter (Hrycik, Almeida, and Hook 2017). A 1975 review of oxygen requirements of aquatic organisms, conducted to establish water oxygen quality criteria for Canadian aquatic ecosystems, calculated mean seasonal minimum oxygen levels at which the average individual shows signs of oxygen distress for several different groupings of fish (Table XX) (Davis 1975). However, the data these thresholds are based on are quite old and may not be consistent with more recent research.

Fish group	Seasonal minimum oxygen threshold (mg/L)
Mixed freshwater fish, including salmonids	5.26
Mixed freshwater fish, excluding salmonids	3.98
Freshwater salmonids	6.0
Salmonid larvae/eggs	8.09
Marine nonanadromous fish	6.72
Marine anadromous fish, including salmonids	6.43
Anadromous salmonids in seawater	6.94

The effects of hypoxia on freshwater species are less well studied than those on marine and coastal species (Saari, Wang, and Brooks 2017). One meta-analysis of hypoxia effects on freshwater aquatic species found that invertebrates (especially the ephemeroptera, plecoptera, and trichoptera classes often used as biological indicators of water quality) are more sensitive to oxygen levels than fish are for mortality endpoints, but fish growth responses are more sensitive than invertebrate mortality (Saari, Wang, and Brooks 2017). In Lake Erie, hypoxic conditions at the bottom of the lake cause planktivorous fishes to move both horizontally and vertically to find more oxygenated water, which can mean moving into areas of less-suitable water temperature or where they are more vulnerable to prey (Vanderploeg et al. 2009). A model of the impacts of hypoxia on habitat quality for four fish species common throughout North America and particularly important in the Great Lakes (yellow perch, rainbow smelt, round goby, and emerald shiner) found that for all except the emerald shiner, the hypoxic zone overlapped both spatially and temporally with what would have been the areas of highest habitat quality, suggesting that hypoxia is decreasing the area of highly suitable habitat available to those species (Arend et al. 2011).

Strength of evidence

High: Many research studies and several meta-analyses show that hypoxia causes mortality and sublethal effects in a wide variety of aquatic organisms, including invertebrates and multiple life stages of fish species.

Other factors

Additional stressors

The combination of low dissolved oxygen levels with other stressors can have a greater effect on aquatic organisms than the sum of the individual stressors.

Sulfide is a toxic compound that often co-exists with hypoxia; a meta-analysis of 68 experiments reporting the median survival time under hypoxia with and without sulfide found a 30% reduction in survival time, on average, when sulfide was present (Vaquer-Sunyer and Duarte 2010).

Higher water temperatures increase metabolic rates and demand for oxygen. A meta-analysis of 363 experiments examining the effects of hypoxia and temperature on marine benthic species found that the oxygen level associated with 50% mortality increased with temperature for crustaceans, but not for fish (Vaquer-Sunyer and Duarte 2011). Water temperature can also influence the sublethal effects of hypoxia; the growth rate of southern flounder was unaffected by hypoxia when water temperature was 27°C, but decreased by 50% under the same level of hypoxia when water temperature was 29°C (the optimal temperature for growth under non-hypoxic conditions) (McBryan et al. 2013).

Species

Differences among species' tolerance of low oxygen levels and ability to move to areas with higher dissolved oxygen levels influence their responses to hypoxia. A meta-analysis of marine benthic organisms found that crustaceans experienced mortality at higher oxygen levels than did fish, cnidarians, priapulids, and molluscs, but that fish show sublethal responses (e.g., avoidance of hypoxic areas) at the highest dissolved oxygen levels of species tested (Vaquer-Sunyer and Duarte 2008).

Sources

3ab: NO_3 levels → N_2O emissions

Description of relationship

A portion of nitrate in aquatic ecosystems is converted to nitrous oxide through denitrification.

Summary of evidence

Bacteria convert nitrate to nitrogen gas (N_2) through a process called denitrification, which occurs under sub-oxic or anoxic conditions (generally < 2 mg O2/liter) when a source of energy (e.g., organic carbon) is present (Fennel et al. 2009; Boyer et al. 2006). During this process, a small amount of the nitrate is converted to N_2O instead of N_2 (Boyer et al. 2006). The amount of N_2O created through denitrification of nitrate in aquatic systems is a function of the total amount of nitrate lost through denitrification and the proportion of that nitrate that is converted to N_2O (as opposed to N_2).

The nitrate concentrations in a waterway have been shown to influence denitrification rates. A meta-analysis of denitrification rates in aquatic systems found a positive linear relationship between nitrate concentration in the water and denitrification rates, over the range of 1-970 µmol NO_3 (Pina-Ochoa and Alvarez-Cobelas 2006). The highest denitrification rates found in studies included in the meta-analysis were in agricultural rivers in the Midwestern US, suggesting that anthropogenic nitrogen inputs influence denitrification rates. While models of nitrogen loading and losses exist, most do not separate specific types of nitrogen losses (e.g., denitrification vs. storage) and so are not useful for determining this relationship. More complex models that do quantify denitrification explicitly rely on many highly uncertain parameters; more experimental data on denitrification rates is needed to improve them (Boyer et al. 2006).

The proportion of nitrate lost through denitrification that is converted to N_2O is the N_2O yield. One large-scale field study measured N_2O production in 72 headwater streams across the United States using whole-stream NO_3 tracers (Beaulieu et al. 2011). It found a mean N_2O yield of 0.9%, which is similar to yields reported in previous studies, and found no relationship between NO_3 concentrations and N_2O yield. This suggests that the same proportion of denitrified NO_3 is converted to N_2O regardless of NO_3

concentrations. This study also found a positive relationship between NO_3 concentrations and N_2O produced through denitrification, which supports the positive relationship between NO_3 concentrations and denitrification rates described above (Beaulieu et al. 2011).

Rather than separately quantifying denitrification rates and N_2O yields to estimate losses of N_2O from aquatic ecosystems, an alternate approach is to use N_2O emissions factors, which are calculated from the ratio of annual N_2O emissions from a body of water to its annual nitrogen loading. However, this approach generally does not separate out N_2O emissions resulting from NO_3 concentrations from N_2O emissions from other sources (e.g., coupled nitrification-denitrification, direct inputs of N_2O from groundwater) and therefore is not appropriate for quantifying this relationship. One study of N_2O emissions factors does provide additional support for the link between aquatic NO_3 concentrations and N_2O emissions; emissions factors calculated from a meta-analysis of 169 observations found a positive correlation between NO_3 concentration and N_2O flux (Hu, Chen, and Dahlgren 2016).

Strength of evidence

Moderate: The chemical reactions underlying the conversion of nitrate in water bodies to N_2O are well understood. Field studies and a meta-analysis show positive correlations between nitrate concentrations and denitrification rates. The proportion of denitrified nitrate that is converted to N_2O has been measured in studies and a meta-analysis with generally consistent results and does not appear to be influenced by nitrate concentrations, but the denitrification rate is thought to be influenced by nitrate concentrations.

Predictability: Since the proportion of denitrified nitrate converted to N_2O does not appear to be influenced by nitrate concentrations, predicting the change in N_2O emissions resulting from a change in nitrate concentrations is driven by the ability to predict denitrification rates. Predicting denitrification rates for a particular body of water is complicated by uncertainties in the many factors that influence the outcome; models for this relationship do exist, but more experimental data is needed to increase their accuracy.

Other factors

Water body characteristics

A meta-analysis found that annual denitrification rates were highest in lakes (0.10 – 3.72 mol $N/m^2/year$), followed by rivers (1.1 – 3.54 mol $N/m^2/year$), coastal ecosystems, and estuaries (Pina-Ochoa and Alvarez-Cobelas 2006).

Nitrogen loss rates (through denitrification) are negatively correlated with stream flow, water depth, and hydraulic load (ratio of water discharge to surface area) and positively correlated with increases in the water time of travel (reciprocal velocity) (Boyer et al. 2006).

Water chemistry

High annual denitrification rates are associated with high interstitial dissolved organic carbon, low concentrations of phosphorus, and low dissolved oxygen concentrations (Pina-Ochoa and Alvarez-Cobelas 2006).

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3ac: NO_3 levels → Domestic animals

Description of relationship

Ingestion of large amounts of NO_3 causes adverse health and reproductive effects in ruminant livestock including cattle, sheep, and goats.

Summary of evidence

Nitrates ingested by ruminant livestock can be converted to nitrite in the rumen, which reacts with hemoglobin to reduce the blood's capacity to carry oxygen (Pfost, Fulhage, and Casteel 2001). This can cause difficulty breathing, muscle tremors, diarrhea, reduced weight gain, collapse, and death; developing fetuses are particularly sensitive to lack of oxygen, so high levels of nitrate ingestion often cause abortion or delays in conception (Kallenbach and Evans 2014; Rasby, Anderson, and Kononoff 2014; Adams, McCarty, and Hutchinson n.d.). Two long-term studies of nitrate ingestion in dairy cows found lower reproductive performance (lower conception rates and longer calving intervals) in cows ingesting more nitrate via water sources, even at relatively low levels (<100 ppm NO_3-N) (Beede 2008).

Because livestock can also ingest nitrates via food, which can be a significant source of nitrates under certain conditions, and the total amount of nitrate ingested determines the health effect, it is difficult to establish concrete guidelines for safe nitrate levels in livestock water sources (Table XX). This may also explain why some sources warn of potential adverse effects from moderately elevated nitrate concentrations in water, while others characterize chronic effects from moderately elevated levels of nitrate in water as rare (Beede 2008; Undersander et al. n.d.).

Table XX: Comparison of nitrate drinking water guidelines and likely symptoms in livestock from several sources. All nitrate concentrations are given as NO_3-N, ppm.

(Pfost, Fulhage, and Casteel 2001)	**(Adams, McCarty, and Hutchinson n.d.)**	**(Undersander et al. n.d.)**	**(Beede 2008)**
< 100: safe	< 23: no effect	< 10: safe	< 10: safe
		10 – 20: safe for livestock unless feed also has high levels	10-20: generally safe in balanced diet with low-nitrate feed
	23 – 114: reduced gains, increased infertility	20-40: may cause issues in livestock, feed nitrate concentration should be evaluated	20-40: potentially harmful if consumed for long periods
		40 – 100: risky, make sure feed is low in nitrates, consider vitamin A supplement	40 – 100: possible mortality

100 – 300: may contribute to health issues if food sources contain high levels of nitrate	114 – 227: nitrate poisoning symptoms: gray-brown mucous membranes, shortness of breath, rapid breathing	100 – 200: do not use. May cause general symptoms such as poor appetite.	> 100: possible mortality, should not be used
> 300: do not use, may cause nitrate poisoning	> 227: suffocation, incoordination, staggering, death	> 200: do not use. Acute toxicity and death may occur in swine. Total intake likely to be too high for ruminants.	

Strength of evidence

Moderate: While it is clear that the ingestion of nitrate in water supplies has the potential to cause adverse health effects in livestock, the complex role of diet in this relationship is not completely understood, and sources differ on whether chronic ingestion of water with moderately elevated nitrate levels is likely to cause health effects.

Other factors

Some other components of diet influence the conversion of nitrate to nitrite in the rumen. Animals eating exclusively forage and no grain may be at greater risk of nitrite accumulation because of a lack of available carbohydrates, molybdenum, copper, iron, magnesium, and manganese (Adams, McCarty, and Hutchinson n.d.; Undersander et al. n.d.).

The method and rate of feeding can also influence nitrate accumulation; animals allowed to graze may be at lower risk than those that eat large amounts of forage during several daily feeding periods because they ingest less nitrate per unit of time (Adams, McCarty, and Hutchinson n.d.).

Stress, for example caused by heat, calving, and changes in food type or amount, may make livestock more susceptible to nitrate toxicity (Adams, McCarty, and Hutchinson n.d.).

Sources

4a: Change in N management →Crop yield

Description of relationships

Fertilizer type

No consistent relationship between fertilizer type and yield has been found (Cook et al. 2015; Abalos 2014).

Fertilizer application timing

Pre-plant application and fall application resulted in statistically similar yields. Side-dress application and at-planting application may increase yields relative to fall application. Splitting nitrogen application between at-planting and side-dress may further increase yields.

Fertilization rate

Crops that do not receive nitrogen fertilizer yield 63% of the yield obtained when crops are fertilized at the recommended rate. Doubling the recommended fertilization rate increases yield by 4% (Quemada et al. 2013). Above a certain rate (generally between 100 and 200 kg N ha^{-1}), the effect of fertilizer application on yield diminishes (Von Blottnitz et al. 2006).

Fertilizer application location

The localized application of fertilizer to surface or subsurface soil near plant roots and seeds increases yield by 3.7% compared to broadcast fertilizer application (Nkebiwe et al. 2016).

Summary of evidence

Fertilizer type

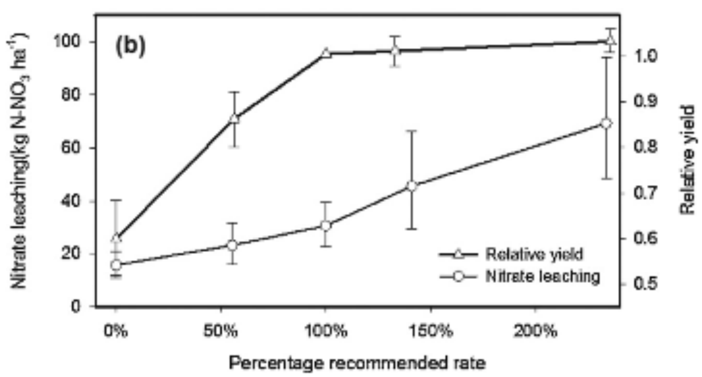
Enhanced efficiency fertilizers may delay the bacterial oxidation of ammonium or the hydrolysis of urea to reduce nitrogen losses in the soil, thus increasing yield. However, several meta-analyses examining this relationship have conflicting results. A meta-analysis of 46 studies comprising 1248 observations that controlled for management practices, soil types, and meteorological factors determined that enhanced efficiency fertilizers such as nitrification inhibitors, urease inhibitors, and polymer coated fertilizers have no statistically significant effect on yield (Cook et al. 2015). Yield is highly variable based on other conditions, and this study only investigated corn in the Midwest of the United States. This may account for another meta-analysis's conclusion that nitrification inhibitors can increase yield by 7.5% (Abalos 2014). Using 254 observations from 48 studies, this meta-analysis accounted for crop type, soil texture, climate, soil pH, and fertilizer application rate. Although the study reports that fertilizer use can be recommended to increase yield, it also acknowledges that some of these factors can have a much greater effect than fertilizer type on yield (see “other factors”). The meta-analysis used data from studies across the globe and from varied environmental conditions, for which it attempted to control. Both of these studies compared each base fertilizer type (anhydrous ammonia, UAN, and urea) with relevant enhanced efficiency fertilizers but did not compare the effects of different base fertilizers on yield. An additional meta-analysis of 44 studies found no significant effect from nitrification inhibitors and a negative effect from controlled release fertilizers on yield (Quemada et al. 2013).

Fertilizer application timing

One of the meta-analyses mentioned above intended to analyze the effects of enhanced efficiency fertilizers on yield but found that fertilizer application timing has a much more significant effect on yield than fertilizer type (Cook et al. 2015). Other variables such as fertilizer type and fertilizer application location were controlled to maximize the accuracy of the results. Fertilizer application timing was divided into fall, pre-planting (winter or spring > 5 days prior to planting), at planting (5 days before to 20 days after planting), side-dress (> 20 days after planting), pre-planting/side-dress, and at planting/side-dress. Three different types of fertilizer were included: anhydrous ammonia (148 observations from 24 studies), UAN (262 observations from 17 studies), or urea (479 observations from 38 studies); each of these fertilizer types responded similarly to changes in application timing. Side-dress, at planting, and split at planting/side-dress applications maximized yield relative to fall and pre-planting applications. The meta-analysis is limited to corn yields in the Midwest, and its initial focus on the effect of enhanced efficiency fertilizers makes its discovery of the effect of fertilizer application timing coincidental and potentially biased by the studies it includes, which were chosen based on their inclusion of enhanced efficiency fertilizers. Another meta-analysis of 44 studies found that optimized timing of fertilizer application has no effect on yield (Quemada et al. 2013). However, this analysis was also designed to investigate the effects of nutrient management on nitrate leaching control strategies, rather than the effects of fertilizer application timing on crop yields.

Fertilization rate

As shown in the figure below, the diminishing effects of fertilization rate are confirmed in a meta-analysis of 44 studies which showed that doubling the recommended fertilization rate increases yield by only 4% (Quemada et al. 2013). This study focused on nitrogen leaching as a result of different fertilization rate practices but reported information about yield as well, having 166 observations on yield in the database. The data demonstrated that treatments receiving no nitrogen fertilizer result in 63% of the yield obtained with recommended rate treatments. This meta-analysis compiled data from a variety of geographic locations, with the majority coming from the European Mediterranean Basin (35%) and the Midwest of the United States (30%). Additionally, data was compiled from both cereals (63%) and vegetables (37%), and the four climate types of the data minimized bias. Environmental effects were considered of little relevance to the results after analysis. A second meta-analysis also found a saturating response curve of yield to fertilization rate, although the specific relationship varied by site and year, suggesting strong influences of environmental factors; in some cases, saturation only occurred at very high fertilization rates (Eagle et al. 2017). Another study supported the idea of diminishing effects of increasing fertilization rate (Von Blottnitz et al. 2006). Focusing on European data, this article concluded that crops have optimal application rates, and once these rates have been surpassed, the positive effect of fertilizer on yield is no longer evident.



Fertilizer application location

Another meta-analysis of 39 field studies compiling 772 datasets reviewed the effects of fertilizer application location on yield (Nkebiwe et al. 2016). Among other requirements, data was only used from studies where the same or comparable fertilizers were used in both broadcast and placement fertilizer treatments. "Broadcast" refers to fertilizer application on the entire soil surface with or without incorporation, while "placement" refers to the variety of techniques used to locally apply fertilizer in or on the soil, an application process thought to make fertilizer nutrients more directly available to the crops. Overall, data analysis fixed for effects of the variety of crop types, individual placement techniques, and fertilizer types displayed a 3.7% increase in yield when fertilizer placement was used instead of broadcast application. Yield effects varied by the specific type of fertilizer placement, and there were very few datasets available for many of the placement techniques. Yield was found to be higher for eight out of the 11 placement techniques studied. Of the placement techniques with at least

100 datasets, subsurface deep point injection had the greatest positive effect on yield. Three placement techniques (seed, below-seed, and subsurface shallow point injection) showed no effect on yield relative to broadcast application. The meta-analysis also discusses the differences in placement effect by fertilizer type, but the effects of fertilizer type on yield cannot be separated from those of fertilizer application location in this study.

Strength of evidence

Fertilizer type

Low. Although multiple meta-analyses have examined this relationship, they have inconsistent results (Cook et al. 2015; Quemada et al. 2013; Abalos 2014). One meta-analysis is limited to corn crop in the Midwest United States (Cook et al. 2015). Another meta-analysis has limited applicability as it investigated yield as an additional component of its study on the effect of these fertilizer types on crop productivity and nitrogen use efficiency (Abalos 2014). As a result, this meta-analysis does not include studies that specifically evaluate the relationship between fertilizer type and crop yield but fertilizer type and the variable in question. No studies were found that directly compared the effects of different base fertilizer types on yield; all of the studies discussed here examined the yield effects of enhanced efficiency fertilizers compared to the same base fertilizer type without the additive.

Fertilizer application timing

Fair. Two meta-analysis studies found consistent results relating fertilizer application timing and yield. However, the applicability of these results is limited as both studies were designed to assess different effects of fertilizer management practices. For example, the first study intended to examine the effects of enhanced efficiency fertilizers on yield but found the effects of fertilizer application timing to be much more significant (Cook et al. 2015). The other meta-analysis investigated fertilizer management and N_2O and NO_3 losses, but provided additional information about yield (Quemada et al. 2013). Both studies were limited to a corn crop.

Fertilization rate

Moderate. Two meta-analyses had consistent results about the diminishing effects of increasing fertilization rate on yield. Additionally, all of the component studies in the meta-analyses were based on optimal or recommended fertilization rates which can vary by crop type, although the general trend of diminishing effects on yield as fertilizer rate increases was consistent.

Fertilizer application location

Fair. One meta-analysis was found which assessed crop yield as it changes relative to different fertilizer application locations (Nkebiwe et al. 2016). Overall, it found that fertilizer application location using the placement method can increase crop yield, but the placement methods studied themselves include 11 different techniques. Eight of these techniques maintained higher crop yield relative to broadcast fertilizer application, but three did not. Because the meta-analysis discovered different outcomes for many of the placement techniques, and some of them had very small sample sizes, more research is needed to establish whether the relationships found are robust.

Other factors

Environmental factors

Soil moisture content: Deep subsurface fertilizer placement may increase yields due to the higher moisture content present in deeper layers of soil (Nkebiwe et al. 2016). Higher moisture content makes fertilizer more readily available to plants due to the increased capacity for nutrient movement to the roots. Even under drought stress, deep fertilizer placement alone is still seen to increase crop resilience and thus increase yields.

Soil pH: Nitrification inhibitor activity is highest in acidic soils (pH ≤ 6), which maximizes the positive effect on yield from enhanced efficiency fertilizers by decreasing N losses due to reduced NH_3 volatilization (Abalos 2014).

Soil texture: The positive effect of urease and nitrification inhibitors on yield is higher in medium- and course-textured soil relative to fine-textured soil (Abalos 2014). This may be because fine-textured soils likely already have lower NO_3^- loss through leaching, which is what nitrification inhibitors aim to decrease.

Soil temperature: The inhibitory effect of NBPT decreases as soil temperature increases, thus decreasing the effect of this enhanced efficiency fertilizer on crop yield (Abalos 2014). Mulching can increase yield by 40% relative to control due to numerous agronomic effects, including increased soil temperature resulting from the use of mulched soil (Quemada et al. 2013).

Agricultural management techniques

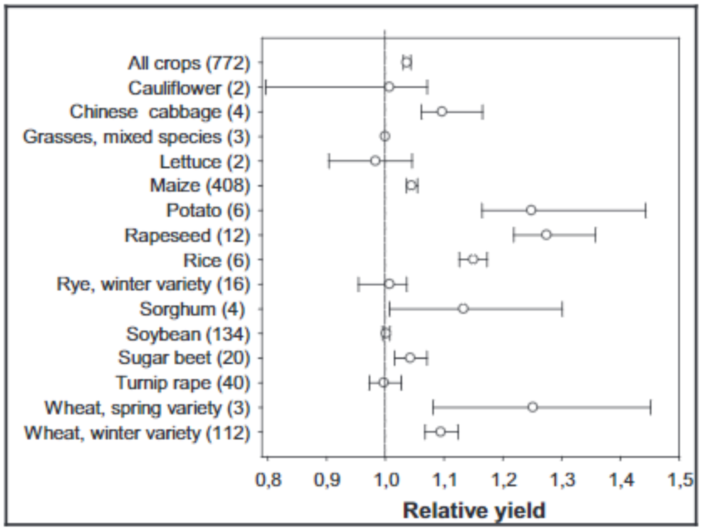
Tillage: The mean yield for tilled plots is higher than for no-tilled plots (Cook et al. 2015).

Irrigation: Deficit irrigation reduced yield on average by 23% due to reduced nitrogen uptake by plants (Quemada et al. 2013). Improving irrigation technology can increase yield by 10%. Additionally, improving an irrigation schedule for crop needs can increase yield. However, excessive irrigation can deprive topsoil of adequate nitrogen, leading to a decrease in yield (Gabriel et al. 2012).

Cover crops: In irrigated systems, replacing a fallow with a legume cover crop can increase yield by a mean of 25% if the subsequent crop grows fast enough to exploit the additional nitrogen left behind by cover crop residues (Quemada et al. 2013). When a non-legume cover crop is used in this situation, the mean effect is not significant.

Crop type

In nine of fifteen crop species analyzed, yield from placement fertilizer application was higher than yield from broadcast. See the table below for the yields for placement fertilizer application (relative to broadcast fertilizer application) for the different species analyzed (Nkebiwe et al. 2016).



Sources

4b: Crop yield → Agricultural profit

Description of relationship

The following table details the average price (Canadian $/t) of different Canadian crop types (Lavergne and Oleson 2017).

Crop Type	Average Price (Canadian Dollars/Tonne)		
	2015-2016	2016-2017	2017-2018
Durum	290	275	270-300
Wheat (excluding durum)	225	235	240-270
Barley	209	169	185-215
Corn	179	171	165-195
Oats	193	209	220-250
Rye	221	115	120-150
Canola	509	529	510-550
Flaxseed (excluding solin)	449	458	410-450
Soybeans	440	454	400-440
Dry Peas	365	300	280-310
Lentils	965	575	720-750
Dry Beans	775	885	825-855
Chickpeas	815	1000	1000-1030
Mustard seed	985	660	730-760
Canary Seed	580	485	490-520
Sunflower Seed	550	565	580-610

Summary of evidence

For the 2015-2016 and 2016-2017 average price data, Statistic Canada's reports on Production of Principal Field crops are used. Statistics Canada collects data through agricultural censuses and surveys for this project. Estimates for the 2017-2018 year are based on Statistic Canada's first production estimate accounting for weather conditions, climate conditions, and other variable factors. Agriculture and Agri-Food Canada compiled this data to create an Outlook for Principal Field Crops (Lavergne and Oleson 2017).

Strength of evidence

High: All other things being equal, an increase in yield will result in an increase in income. The values reported above come from government-sponsored and national agencies, encompassing an accurate and comprehensive picture of Canada's agriculture and agri-food industry (Lavergne and Oleson 2017). This data has been tracked over years, and the number of years included in this report gives a perspective on the trajectory of average prices. The applicability of this data is high, because it is calculated from direct reports from a Canadian agricultural census and survey data.

Other factors

Foreign supply: Supply of crops, especially cheaper crops, of the same kind around the world may drive down Canadian prices. As total supplies tighten or foreign prices rise, Canadian prices may increase (Lavergne and Oleson 2017).

Domestic supply: An increase in Canadian supply of a certain crop may drive down prices of that crop. A decrease in Canadian supply of a certain crop may raise prices of that crop (Lavergne and Oleson 2017).

Futures market: A higher futures price may increase a crop's price (Lavergne and Oleson 2017).

Substitutes: A decline in supplies of substitute crops may increase demand for a certain crop, increasing its average price (Lavergne and Oleson 2017).

Demand: When demand for a crop rises, it sells at a higher prices. When demand for a crop falls, it sells at a lower price (Lavergne and Oleson 2017).

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4c: Crop yield → Soil carbon

Description of relationships

Soil organic carbon (SOC) stocks may be affected by changes in nutrient management that increase/decrease net primary productivity (i.e., total biomass or crop yield) or impact soil disturbance. Most nutrient management changes adopted for improved efficiency will have little to no effect on soil C compared with business as usual (Eagle and Olander 2012).

Summary of evidence

Long-term experiments where N fertilizer application rates were increased substantially in a given year (96 to 192 kg N/ha/yr more fertilizer applied to wheat) measured soil C gains amounting to between 0.1 t C/ha/yr and 0.43 t C/ha/yr (Poulton et al. 2018). Researchers suggested the most likely mechanism for this soil C gain is increased crop growth. Studies looking at management effects on soil C find that even with large changes in primary productivity or soil disturbance, gains/losses of soil C are unlikely to be detected for a long period of time, given high background levels and slow rates of change. Also, any small increase in soil C is likely to balance out to no net GHG impact, as these are balanced out by the CO_2 and CH_4 emissions associated with fertilizer manufacture and the N_2O emissions that accompany higher nitrogen fertilizer application rates. Therefore, this is not a viable GHG mitigation option.

Strength of evidence

High: Soil organic C, and changes thereof, are linked to primary productivity. The changes in crop productivity resulting from fertilizer management changes are small in relation to changes in SOC needed before they can be detected over baseline levels, with the exception of large rate increases – the latter of which are accompanied by high N_2O losses that offset any SOC gained (see link 2a).

Other factors

Soil C (in soil organic matter) is linked to soil organic N, with C:N ratios consistently around 12:1 in most soils. Therefore, any increases in soil C will require additions of N as well (van Groenigen et al. 2017). This may be satisfied by "soaking up" N otherwise lost to the environment, but any additional N to the system in the form of fertilizer comes with risks of increased N_2O emissions and NO_3 leaching. Therefore, tradeoffs (and synergies) in environmental and other outcomes must be considered.

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4d: Soil carbon → net GHG emission

Description of relationships

New carbon stored in soil organic matter removes CO_2 from the atmosphere via photosynthesis, reducing net GHG emissions (Ontl and Schulte 2012).

Summary of evidence

This is a straightforward relationship, as soil organic carbon originates from decaying plant and animal matter, for which all C originated from CO_2 in the atmosphere.

Strength of evidence

High: This relationship comes directly from known stoichiometry.

Other factors

Emissions of other GHGs (N_2O and CH_4) can be affected by management action that increases or decreases SOC. These impacts on net GHG emissions are discussed in the above section on management and soil carbon.

Sources

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5a: change in N management → crop quality

Description of relationships

Fertilizer type

Evidence found was inconclusive.

Fertilization rate

In cereal crops, higher nitrogen fertilizer application rates cause increased protein content (Jensen et al. 2011).

Fertilizer application timing

There is limited evidence that suggests a nitrogen deficiency early in the wheat growth cycle does not affect protein content (Ravier et al. 2017).

Fertilizer application location

The localized placement application of fertilizer to surface or subsurface soil near plant roots and seeds increases grain protein content 6.3% relative to broadcast application (Nkebiwe et al. 2016).

Summary of evidence

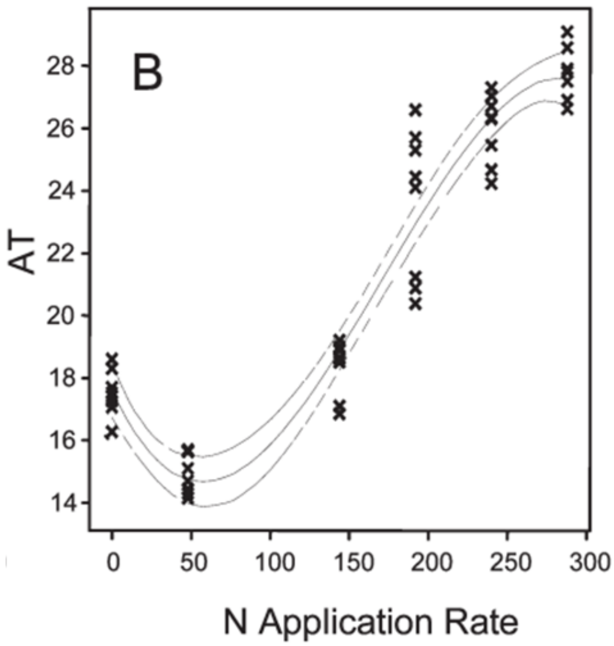
Fertilizer type

Evidence found relating fertilizer type and crop quality was inconclusive. A study of hard red spring wheat cultivation in Minnesota over six site-years concluded that at the same application rates of nitrogen, polymer-coated urea fertilizer increased protein content compared with urea fertilizer due to increased nitrogen availability during later stages of the growing season (Farmaha and Sims 2013). A two-year field experiment in central Montana found that in camelina cultivation, enhanced efficiency fertilizers do not have any relative effect on protein content (Afshar, Mohammed, and Chen 2016). Conclusions from these studies are contradictory, and no general meta-analysis of data is available, so it is impossible to derive a single, general relationship considering the specificity of the studies.

Fertilization rate

Higher nitrogen fertilizer application rates cause increased grain protein content due to greater synthesis and accumulation of storage proteins, particularly gluten proteins (Jensen et al. 2011). Gluten proteins consist of gliadins and glutenins. A study using the Broadbalk long-term experiment at Rothamsted found that in wheat, increasing applications of nitrogen fertilizer increase content of gliadin proteins (Godfrey et al. 2010). These fields lie in the United Kingdom and were established in 1843. The authors chose to compare plots that had received no nitrogen or nitrogen as farmyard manure since 1843 with plots receiving nitrogen since 1985 or before. Figure XX shows the relationship between nitrogen application rate and total protein content of flour made from the wheat from each plot studied. The total protein content drops initially, suggesting that low levels of fertilizer application stimulate crop yield before affecting protein content.

Figure XX: Fitted polynomial model between N application rate and total protein (AT) in white flour. Dashed lines are 95% confidence intervals. Source: (Godfrey et al. 2010)



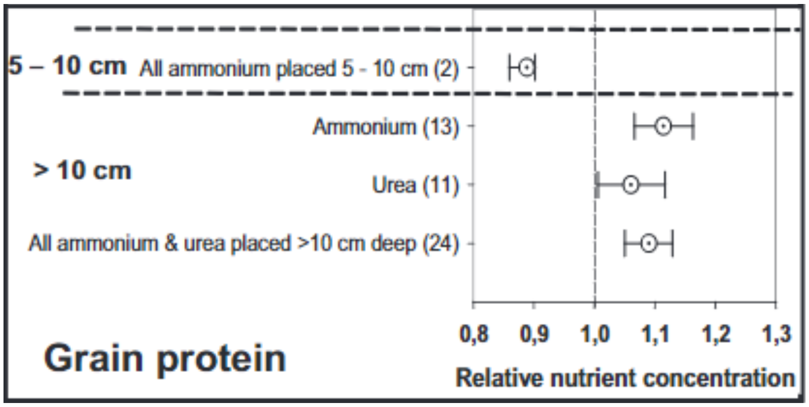
Fertilizer application timing

A study of 18 site-year experiments, each with 1-14 cultivars and 2-8 fertilization strategies, concluded that a nitrogen deficiency in the early stages of wheat growth does not affect protein content (Ravier et al. 2017). Specifically, nitrogen deficiencies can be properly tolerated with no protein loss during the vegetative growth phase. This study used data from sites in France and mostly used mineral fertilizer, with one case of organic fertilizer use. The ability of the wheat to tolerate nitrogen deficiencies may depend on that specific type of wheat cultivar. Establishing these minimum requirements for nitrogen application can allow more strategic and sustainable nitrogen management practices. While these results suggest that delaying fertilizer application to later in the growing season would not adversely affect protein content, no evidence was found to compare grain protein content across fertilizer application timings.

Fertilizer application location

A meta-analysis using 357 datasets from 11 studies, two of which were published in 1982 and nine of which were published from 2000 to 2013, found that localized placement application of fertilizer increases grain protein content 6.3% relative to broadcast application (Nkebiwe et al. 2016). In this analysis, "broadcast" refers to fertilizer application on the entire soil surface with or without incorporation, while "placement" refers to the variety of techniques used to locally apply fertilizer in or on the soil, an application process thought to make fertilizer nutrients more directly available to the crops. The authors noticed a tendency of the relative placement effect on nutrient concentration, including protein content, to increase with increasing fertilizer placement depth, likely due to reduced nitrogen losses. Figure XX demonstrates the increasing effect of placement fertilizer application with increasing depth. The strength of the increasing effect of placement fertilizer application relative to broadcast application on nutrient content depends on the crop type of focus.

Figure XX: Grain nutrient concentration by depth of fertilizer placement, relative to surface broadcasting. Source: Nkebiwe et al. (2016)



Strength of evidence

Fertilizer type

Low: Although multiple studies were found, all evidence was site-specific, limiting the applicability of the studies to the model (Farmaha and Sims 2013; Afshar, Mohammed, and Chen 2016). The studies differed on factors such as geographic location and crop studied. A target study for applicability would have focused on wheat and a climate similar to that of Canada, but none of the studies fully encompassed these characteristics. Additionally, no conclusive result could be found across the studies. Instead, each study reported contradicting results, probably due to the variety of site-specific data.

Fertilization rate

Fair: Though the main study found to support the relationship between fertilization rate and crop quality was site-specific, including a limited sample of crop types, the methods are strong and rely on an experimental site over 150 years old (Godfrey et al. 2010). Additionally, the relationship is confirmed in other studies, which stress the importance of fertilization rate in the interactions that affect grain protein content (Farmaha and Sims 2013). This extends the applicability of the relationship.

Fertilizer application timing

Low: The relationship between fertilizer application timing and crop quality is based on a study of wheat growth in France (Ravier et al. 2017). The authors discuss the variable climate and situations throughout the sites from which data was taken, such as water stress and crop disease. This limits the applicability of the data because results may be dependent on variables specific to experiment in France. Additionally, no evidence was found that demonstrates how the presence of fertilizer at certain times affects crop quality. This study researches the effects of the absence of the fertilizer, which has fewer implications in nitrogen management practices.

Fertilizer application location

Moderate: A meta-analysis study established the relationship between fertilizer application location and crop quality (Nkebiwe et al. 2016). The study did not consider the individual effects of each different types of fertilizer application location beyond than the broad categories of “placement” or “broadcast” for grain protein content.

Other factors

Climate

Temperature: Regardless of nitrogen application levels, high temperatures may lead to higher protein content due to a resulting increase in protein synthesis (Jensen et al. 2011). Additionally, as temperature increases, so does the release of nitrogen from polymer-coated urea (Gandeza, Shoji, and Yamada 1991; Fujinuma, Balster, and Norman 2009). The delayed release of nitrogen from polymer-coated urea can increase grain protein content (McKenzie et al. 2010).

Precipitation: Regardless of nitrogen application levels, low levels of precipitation may lead to higher protein content due to a resulting increase in protein synthesis (Jensen et al. 2011).

Crop type

The cultivar used may account for a crop's ability to withstand nitrogen deficiencies at certain stages in the growth cycle (Ravier et al. 2017).

Environmental factors

Soil moisture content: As soil moisture content increases, so does the release of nitrogen from polymer-coated urea (Gandeza, Shoji, and Yamada 1991). The delayed release of nitrogen from polymer-coated urea can increase grain protein content (McKenzie et al. 2010).

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5b: crop quality → agricultural profit

Description of relationship

Farmers can receive an increase in payment, known as a premium payment, for high quality wheat or a decrease in payment, known as a discount payment, for low and medium quality wheat (Farmaha and Sims 2013).

Summary of evidence

The following tables detail protein premiums (Canadian $/t) reported by the Grain Farmers of Ontario in 2017 for two wheat classifications with a basis moisture of 14.0% (Grain Farmers of Ontario 2017). Prices change over time.

Hard Red Winter (Pool B Protein Premium)	
Protein Content	**Premium (Canadian Dollars/Tonne)**
10.9% or less	Nil
11.0% to 11.9%	8
12.0% or more	15

Hard Red Spring (Pool C Protein Premium)	
Protein Content	**Premium (Canadian Dollars/Tonne)**
11.9% or less	Nil
12.0% to 12.4%	10
12.5% or more	18

There is some discrepancy about the relationship as some sources claim that the prices for baking quality wheat are not substantially higher than feed quality wheat (Jensen et al. 2011). No data has been found to support these claims.

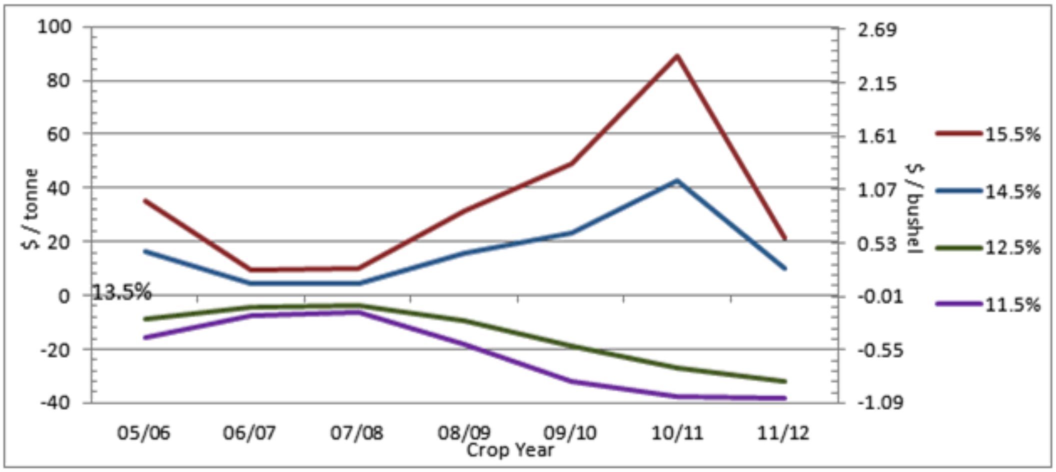
Strength of evidence

Moderate: Evidence reported comes from nationally-recognized organizations that use specific data from Canada's agricultural sector, making the relationship well-founded and applicable. Additionally, the same general trend seems to emerge with higher quality wheat yielding greater prices relative to lower quality wheat. However, specific statistics could only be found for a single crop.

Other factors

Relative abundance: The chart below displays the historical fluctuations of price ($/t and $/bu) received for Canada Western Red Spring as provided by the Canadian Wheat Board (CWB) (Blue 2012). Payments fluctuate as the abundance of certain qualities of wheat change. In the 2006/07 crop year, an abundance of high quality wheat provided a premium of only $4.56/tonne ($0.12/bu) for the 14.5% protein content Canada Western Red Spring crop, relative to the 13.5% protein content crop. Comparatively, the 2011/12 crop year produced excessive low quality wheat due to climatic conditions. This drove the premium for the 14.5% protein content up to $42.82/tonne ($1.17/bu). Thus, the premium is smaller when high quality wheat is abundant and larger when high quality wheat is lacking.

Figure 1. Historical Canadian Wheat Board Payment Spreads around #1 CWRS 13.5% protein



Source: Canadian Wheat Board (CWB)

Sources

6a: change in N management → costs

Description of relationship

Changes in N management practices increase costs for farmers and require a greater time investment and more expertise than continuing status quo N management practices.

Summary of evidence

Costs

Changes in N management may influence farmers' fertilizer costs (if more expensive fertilizer types are used or if the total amount of fertilizer use changes) and equipment costs (if fertilizer placement techniques require additional equipment). It is difficult to generalize these costs, as they depend on the farmer's current fertilizer strategy and equipment, farm-specific factors that determine equipment suitability, and regional cost differences. However, several general equations have been developed to describe the cost impact of fertilizer management changes.

A farmer's fertilizer cost under a fertilizer best management practices implementation strategy (adjusting N rate, timing, and location) can be calculated using the following equation (Kanter, Zhang, and Mauzerall 2015):

$$Cost_i = (P_{N,i} \times RE_{N,i}) + [P_{FBMP,i} \times (BAU_{N,i} - RE_{N,i})]$$

where $P_{N,i}$ is the fertilizer N price ($ kg^{-1} N), *RE* is the N application rate under a fertilizer recovery efficiency scenario in year *i* (kg N ha^{-1}), $P_{FBMP,i}$ is the price to implement a fertilizer best management practice in year *i* ($ kg^{-1} N reduced), and $BAU_{N,i}$ is the N application rate under the business as usual scenario in year *i* (kg N ha^{-1}). In practice, this equation is of limited usefulness due to the difficulty of estimating the price to implement a fertilizer best management practice and the rate of N application that can be used with best management practices. There have been studies of reductions in N fertilization rates that can be achieved with certain fertilizer management practices; one field study in Minnesota estimated that variable N fertilization at rates up to 145 kg N ha^{-1} would have avoided the use of 69-95 kg fertilizer as compared to the recommended uniform rate of 145 kg N ha^{-1} (Mamo et al. 2003).

A farmer's fertilizer cost under an enhanced efficiency fertilizer strategy (changing fertilizer type) can be calculated using the following equation:

$$Cost_i = [(P_{EEF,i} \times \alpha_{EEF,i}) + P_{N,i} \times (1 - \alpha_{EEF,i})] \times RE_{N,i}$$

where $P_{EEF,i}$ is the price of enhanced efficiency fertilizer in year i (\$ kg^{-1} N), $\alpha_{EEF,i}$ is the portion of N applied as EEF, $P_{N,i}$ is the standard fertilizer N price (\$ kg^{-1} N), and RE is the total N application rate (Kanter, Zhang, and Mauzerall 2015).

Slow and controlled-release fertilizers are more expensive than conventional fertilizers. The prices for different fertilizer types in the US in 1996 were (Trenkel 1997):

Fertilizer type	**Product**	**Market price (US$ per kg of N)**
Conventional	Granulated urea	0.66
Slow and controlled-release	Urea-formaldehyde (38% N)	1.50 - 1.58
Slow and controlled-release	Urea-isobutyraldehyde (31% N)	2.90 - 3.55

A separate study estimated the costs of improved nutrient management at 5 US\$ ha^{-1} yr^{-1} for croplands when considering the following strategies (Smith et al. 2008):

- Adjusting fertilization rate – avoiding excess N applications or eliminating them where possible
- Fertilizer application timing – avoiding time delays between N application and plant N uptake
- Fertilizer application location – placing the N more precisely to make it accessible to roots

However, this study mainly focused on avoided greenhouse gas emissions and did not describe how this cost estimate was reached.

Time

The implementation of best fertilizer management practices can have high labor requirements, especially when adjusting fertilizer timing or placement. The availability of labor and equipment strongly influence the timing and quantity of fertilizer application. Best practice calls for two applications (at planting and several weeks later), but usually producers need to minimize the number of field operation for time and cost reasons, resulting in a single application (Robertson and Vitousek 2009).

Most farmers do not have the time to adopt new precision agriculture practices. This is not specific to fertilizer use, but common to the adoption of new technologies: broad adoption of the tractor took more than 30 years (Robert 2002).

Expertise

In addition to improving management practices, farmers will need to rely on a rapidly expanding base of biological and agronomic knowledge that is often specific to certain agroecosystems, regions, soil types and slopes. Making the right decisions at the farm level in terms of input-use efficiency, human health and resource protection is becoming an increasingly knowledge-intensive task (Tilman et al. 2002).

Precision agriculture practices require new skills, and only a limited number of farmers have them or are willing to obtain them. Age, attitude and education of producers have been identified as significant barriers in the US: the majority of farmers are over 55 years old, have partial or complete high school education, and have limited interest in changing practices and using computerized systems (Robert 2002).

A study of the profile of farmers that choose to adopt conservation-compatible practices in the US, including nitrogen management, found that the greater the management skills needed to make a farming practice profitable, the greater the size (hectares, income and commodity payments received) of adopting farms. This indicates that these practices hold more appeal for large-scale farm operators concerned with maximizing farm profits, and are less likely to be adopted by operators of small farms focused primarily on non-farm activities (Lambert et al. 2007).

Strength of evidence

Fair: Multiple sources confirm that adoption of nutrient management practices for nitrogen fertilizers represents a significant investment, but it is often difficult to determine the cost of changing fertilizer management practices for a specific farm beforehand. In some cases, evidence shows that the up-front investment is later recovered due to greater yields (see link 5a) and lower fertilizer use. The factor "time invested" is often included in monetary cost estimates, but not much independent evidence exists for the additional time that farmers must invest for different nutrient management strategies. Limited evidence shows that changes in nutrient management practices require more time and labor. A few sources confirm that implementing best practices for nutrient management requires specific skills, and that only a limited number of farmers possess these skills or are willing and able to acquire them.

Sources

6b: Costs → agricultural profit

Description of relationship

Any additional costs incurred from implementing changes in nutrient management practices will decrease the total agricultural profit by that amount.

Summary of evidence

This is a straightforward link based on the basic formula for profit (Investopedia 2018):

$$\text{Profit} = \text{total revenue} - \text{total costs}$$

Therefore, any increase in total costs will decrease profit by an equivalent amount. Note that this does not take into account any change in revenue from implementing changes in nutrient management practices, which may offset some or all of the costs in certain cases and is captured in links 5b, 6b, and 8b.

Strength of evidence

High: This relationship comes directly from the formula used to calculate profit.

7a: Change in N management → market access

Description of relationship

Changes in nitrogen management practices could enhance farmers' access to markets for their produce if some consumers prefer produce from farms using improved management practices.

Summary of evidence

Consumers are becoming more interested in the environmental and social effects of products they purchase, and may be more loyal to or willing to pay more for products that they know are produced sustainably (Yue et al. 2011). Consumers cannot tell by looking at the product if it is produced in a sustainable or "green" manner, so there must be a trustworthy and recognized way to communicate this distinction to consumers in order for these effects to occur (Schumacher 2010). Ecolabeling, which is certified by an independent organization, can provide a way for consumers to differentiate between products based on sustainability criteria (Schumacher 2010). Two widespread examples of ecolabels, organic and fair trade, have grown rapidly in the United States over the past few decades (organic sales grew up to 20% annually from 1990-2008, and Fair Trade up to 100% annually after it was introduced) (Howard and Allen 2010). It is possible that an ecolabeling scheme for sustainable nitrogen management practices would make the resulting crops more attractive to certain groups of consumers; however, no such ecolabel currently exists.

A survey of consumers in Europe found that consumers that consider environmental impacts of products are more likely to buy ecolabeled goods, and that consumer education (used as a proxy for income level) is positively correlated with buying ecolabeled products (Schumacher 2010). Consumers feel that ecolabeling provides information about the product's environmental impact, and conscious consumers (who are influenced by environmental and quality considerations) buy more ecolabeled products than price-oriented consumers (Schumacher 2010). A consumer survey in the US showed that people are more interested in food-related issues connected to personal health (food safety and nutrition) than in ethical and environmental issues (treatment of animals, environmental impacts, working conditions). This suggests that the enhanced market access for sustainable nitrogen management practices may be smaller than enhanced market access for organic produce, which is associated with improved health and safety.

Strength of evidence

None: While there is some evidence that consumers prefer products with ecolabels such as organic and Fair Trade, no ecolabel for sustainable nitrogen management practices currently exists, so there is no direct evidence that such a label would provide enhanced market access for farmers carrying out these practices. Underlying differences in what each of these labels signals to a consumer make it difficult to generalize the effect of a particular ecolabel to products with a different ecolabel.

Other factors

As described above, differences in consumer education levels and drivers of purchase decisions affect the extent to which ecolabels influence consumer decisions.

Sources

7b: Market access → agricultural profit

Description of relationship

Increased market access for produce grown using sustainable nitrogen management practices could increase the prices consumers are willing to pay for that produce, thus adding to farmers' profits.

Summary of evidence

As described in link 8a, the use of ecolabeling to inform consumers about products' environmental or social attributes can increase consumer preference for ecolabeled products. This may translate into a price premium for these products.

A study of prices for frozen processed pollock products in the United Kingdom found a 14.2% price premium for products with the Marine Stewardship Council label, which certifies that the product is from a sustainable fishery (Roheim, Asche, and Santos 2011). Data on American produce purchases in 2005 showed that premiums for organic produce ranged from 20%-42% for fruits and 15%-60% for vegetables (Lin, Smith, and Huang 2008). However, a study of consumer preference for ecolabeled wood products showed that preference does not always translate into a price premium; while customers preferred ecolabeled plywood over unlabeled plywood when their prices were the same, the majority of consumers were unwilling to pay a 2% price premium for ecolabeled plywood (Anderson and Hansen 2004). Still, about 37% of plywood sales were ecolabeled wood with the 2% premium, suggesting that there is a subpopulation willing to pay more for environmentally friendly products. Even if the majority of consumers is unwilling to pay a premium for crops grown with sustainable nitrogen management practices, there may be a premium level that would be acceptable to enough consumers that farmers could recover their certification costs and increase their profits.

Strength of evidence

None: While there is some evidence that consumers will pay a premium products with ecolabels such as organic produce and sustainably harvested seafood, no ecolabel for sustainable nitrogen management practices currently exists, so there is no direct evidence that such a label would provide enhanced profits for farmers carrying out these practices. Underlying differences in what each of these labels signals to a consumer make it difficult to generalize the effect of a particular ecolabel to products with a different ecolabel.

Sources

8a: Change in phosphorus management practices → phosphorus losses in runoff

Description of relationship

Fertilizer Type

More phosphorus is lost in runoff following fertilizer application when more highly water soluble fertilizers are applied (Shigaki, Sharpley, and Prochnow 2007). Fluid phosphorus fertilizers may be less susceptible to phosphorus losses than granular fertilizers (Chien et al. 2011).

Fertilization Rate

Higher rates of phosphorus fertilization increase the potential for incidental phosphorus losses.

Fertilizer application timing

Fertilizer phosphorus applied soon before a rainfall or irrigation event results in greater incidental phosphorus losses than when there is a longer period between fertilizer application and rainfall or irrigation (Vadas, Owens, and Sharpley 2008).

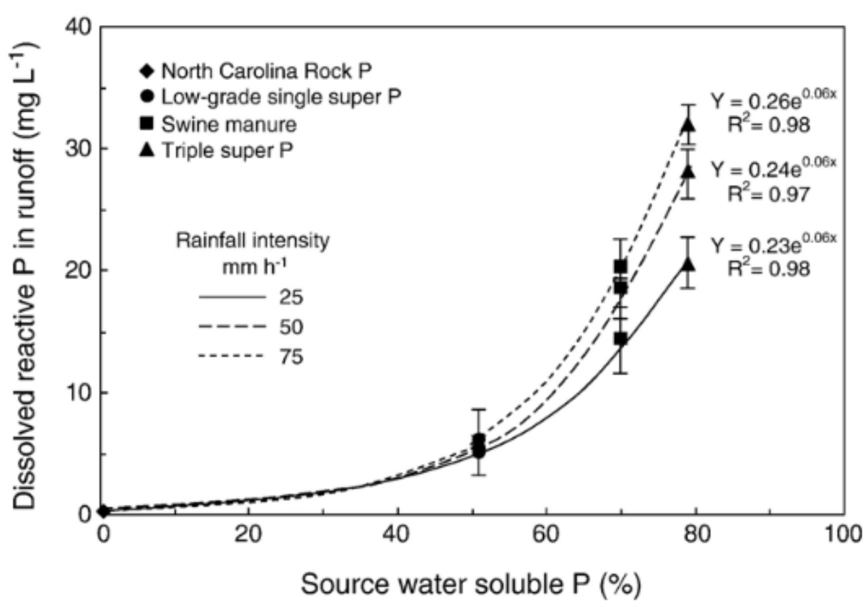
Fertilizer application location

Incorporation of phosphorus fertilizer following application results in lower total runoff phosphorus losses and lower dissolved reactive phosphorus losses than fertilizer application without incorporation (Kleinman et al. 2002).

Summary of evidence

Fertilizer Type

Phosphorus fertilizers differ in their solubility (in water or dilute acids), which are indicative of their availability to plants (highly soluble fertilizers are more immediately available) (Withers et al. 2003). Fertilizer solubility also influences how susceptible the fertilizer phosphorus is to being lost in runoff, with much greater losses from more soluble fertilizers (Withers et al. 2003). Commercial-grade SSP, TSP, MAP, and DAP are all highly soluble (>85-90% of total phosphorus is water-soluble, and the rest is citrate soluble), but some lower-quality SSP can have lower water solubility (50-60%) (Chien et al. 2011). Several laboratory runoff experiments compared dissolved reactive phosphorus and particulate phosphorus in runoff from simulated rainfall following the application of fertilizer phosphorus from three sources (North Carolina rock P, low-grade single super P, and triple super P) varying in solubility (Shigaki, Sharpley, and Prochnow 2007, 2006). Both found strong positive relationships between the water solubility of the phosphorus source and runoff dissolved reactive phosphorus and particulate P, especially following the first rainfall after fertilization (Figure 1).



Source: (Shigaki, Sharpley, and Prochnow 2007)

The form of phosphorus fertilizer (whether it is fluid or granular) may also influence incidental phosphorus losses. Studies have suggested that fluid phosphorus fertilizer can penetrate into the soil more quickly than granular fertilizer, resulting in less phosphorus being available for runoff (Chien et al. 2011). However, there are not many studies that have maintained the same phosphorus compound and fertilizer placement when testing the difference between liquid and granular fertilizers, which has resulted in conflicting findings (Chien et al. 2011).

Fertilization Rate

Several field studies have found evidence for a positive relationship between the phosphorus fertilization rate and phosphorus losses in runoff, but these were conducted in paddy fields in China and the results may not be fully applicable to Canadian agricultural systems (Zhang et al. 2003). The total amount of phosphorus applied in fertilizer puts an upper limit on the amount of phosphorus that can be lost from fertilizer sources. In some systems, farmers apply "bulk" phosphorus fertilizer every few years to save time; this increases the risk of high incidental phosphorus losses following fertilizer application (Sims and Sharpley 2005). The rate of phosphorus fertilizer application is also included as a factor in phosphorus indices, which are widely used to qualitatively assess the risk of phosphorus loss from agricultural fields, as follows:

Phosphorus loss rating	None	Low	Medium	High	Very high
P fertilizer application rate (lb/acre)	None	1-30	31-90	91-150	>150

Source: (Sims and Sharpley 2005)

Fertilizer application timing

Most of the applied fertilizer phosphorus that is lost in incidental losses is released during the first storm event after the fertilizer is applied (Vadas, Owens, and Sharpley 2008). As more time passes after application, the fertilizer phosphorus adsorbs to the soil, and less is available to run off. Therefore, most studies of phosphorus fertilizer application timing and phosphorus losses have focused on the timing of fertilizer application relative to irrigation or rain (Chien et al. 2011).

While most phosphorus transport models don't include direct losses of fertilizer phosphorus (i.e. incidental losses), researchers created and validated a model for the surface runoff of water-soluble phosphorus fertilizer based on the length of time between application and rainfall and the rainfall intensity (Vadas, Owens, and Sharpley 2008).

Fertilizer application location

The incorporation of phosphorus fertilizer results in lower phosphorus losses relative to surface application; fertilizer incorporation promotes phosphorus adsorption to soil particles, and runoff water only interacts with soil from the surface to several centimeters deep, so phosphorus fertilizer left at the surface is more susceptible to runoff losses (Sharpley 1985). A laboratory runoff experiment with DAP found that incorporation of fertilizer following application resulted in lower total runoff phosphorus losses than from surface-applied fertilizer (Kleinman et al. 2002). The amount of dissolved reactive phosphorus as a proportion of total phosphorus in runoff was also lower following fertilizer incorporation than surface-applied fertilizer. However, there was greater loss of phosphorus adsorbed to suspended sediment from fertilizer incorporation, resulting from higher erosion rates following soil disturbance (Kleinman et al. 2002). Fertilizer application location is also included in phosphorus indices (described above) with fertilizer placement >2" deep receiving the lowest risk rating, followed by incorporation, and surface application in the high risk category (Sims and Sharpley 2005).

Strength of evidence

Fertilizer Type

Fair: Laboratory studies show that the amount of phosphorus lost in runoff after fertilizer application is positively correlated with the fertilizer's water solubility, and this relationship makes sense conceptually. However, there were few laboratory studies and no field studies found that examined this effect, so its general applicability cannot be established. Well-designed studies examining the difference in phosphorus losses between granular and liquid fertilizers are lacking.

Fertilization Rate

Low: This relationship makes logical sense, but only a few studies that examined the correlation between phosphorus fertilization rates and incidental phosphorus losses were found, and these were conducted in paddy soils in China, which may respond differently than Canadian agricultural systems.

Fertilizer application timing

Fair: Laboratory runoff experiments show a strong negative relationship between the amount of phosphorus in runoff and the amount of time between fertilizer application and precipitation, and this makes sense given the importance of fertilizer solubility in predicting incidental phosphorus losses and the adsorption of phosphorus to the soil over time. However, no field experiments were found, and experiments covering a variety of fertilizer and soil types are lacking.

Predictability: A model that predicted incidental phosphorus losses based on fertilization rate and time until rainfall performed well with validation datasets (Vadas, Owens, and Sharpley 2008).

Fertilizer application location

Low: While this relationship is logical, only one laboratory study that directly examined the relationship was found, and it only considered one type of phosphorus fertilizer, so these results cannot be generalized.

Other factors

Drainage: Underdrainage can increase the hydrological connectivity between agricultural fields and waterways, which increases the potential for phosphorus dissolved in water to reach waterways (Withers et al. 2003).

Tillage: Because soil is not mixed in no-till fields, phosphorus accumulates at the surface, where most rainfall-soil interaction occurs, resulting in increased potential for phosphorus losses (Chien et al. 2011).

Background soil P: Soils with low background phosphorus concentrations may be able to adsorb phosphorus more quickly than soils with higher background phosphorus concentrations. This results in lower phosphorus losses from low-P soils after phosphorus fertilizer application (Kleinman et al. 2002).

Rainfall characteristics: Several studies have shown greater phosphorus losses in runoff with greater rainfall intensity, and a greater proportion of the phosphorus lost is particulate P, especially for less soluble fertilizer sources (Vadas, Owens, and Sharpley 2008; Shigaki, Sharpley, and Prochnow 2007).

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8b: Phosphorus losses in runoff → phosphorus concentration (dissolved and particulate) in waterways

Description of relationship

Increased phosphorus losses in runoff lead to increased phosphorus concentration in waterways.

Summary of evidence

The relationship between phosphorus losses in agricultural runoff and phosphorus concentration in waterways and water bodies is widely accepted; agricultural phosphorus inputs to waterways has been recognized as an environmental issue for decades, and since point source phosphorus pollution has been largely controlled, the contribution of nonpoint phosphorus sources, such as agriculture, drives phosphorus loading in aquatic systems (Sharpley, Foy, and Withers 2000; Sharpley et al. 1994).

The change in phosphorus concentration in aquatic systems as a result of changes to incidental phosphorus losses in runoff is difficult to predict. In some areas, incidental phosphorus losses make up a very small part of all phosphorus losses, so even a total elimination of runoff phosphorus losses will have a small impact on phosphorus loading to waterways (Withers et al. 2003). In addition, runoff interacts with soil, debris, and vegetation as it moves from an agricultural field to a body of water; phosphorus can be transformed or lost from the runoff water during this process (Sharpley et al. 1994). Once it has reached a body of water, phosphorus can adsorb to and desorb from sediment, causing fluctuations in the concentrations of dissolved and sediment phosphorus (Hart, Quin, and Nguyen 2004).

Due to these complex processes, determining the effect that a change in runoff phosphorus losses will have on phosphorus concentration in waterways is not straightforward. A variety of models have been developed to predict phosphorus movement from agricultural fields to streams, but they often suffer from accuracy issues or do not directly predict phosphorus concentrations in waterways. Currently used models may be behind scientific understanding of phosphorus transport processes; many commonly used models (including SWAT, EPIC, and APEX) are based on soil phosphorus processes from research dating back to the 1980s (Vadas, Bolster, and Good 2013). The PSYCHIC model estimates the proportion of phosphorus delivered to watercourses from each point in a watershed based on the proximity to the nearest river, but does not directly predict phosphorus concentrations in waterways (Davison et al. 2008). INCA-P is a commonly used mechanistic catchment phosphorus model; when model results were compared with water quality data for a catchment in northeast Scotland, the simulation of particulate phosphorus was poor (with modeled results sometimes moving in the opposite direction of the observed data), while the simulation of total dissolved phosphorus was better overall but showed low sensitivity to small rainfall events (Jackson-Blake et al. 2015). A study that combined two models (GLEAMS and REMM) to predict nutrient transport from swine farms in NC found that the inaccurate prediction of phosphorus transport from the field (the GLEAMS model) made it impossible for the REMM model to estimate phosphorus movement through the riparian buffer with any accuracy (Gerwig et al. 2001). The REMM model may be the most appropriate for this particular relationship, but no model validations focusing on phosphorus concentrations were found (REMM also models the movement of water, nitrogen, and carbon) (Lowrance et al. 2000).

Strength of evidence

Fair: The potential for phosphorus moving from an agricultural field in runoff to reach a surface waterway is clear. However, the relationship between phosphorus losses in runoff and phosphorus concentrations in surface water is less straightforward and depends on the other sources of phosphorus in the waterway and the land over which the runoff moves. Therefore, it is very difficult to tell whether a relationship will be apparent in any particular case.

Predictability: As described above, there are models that can predict phosphorus movement from agricultural fields to water bodies, but these often perform poorly when compared with observational data.

Other factors

Phosphorus sources: The strength of the relationship between incidental phosphorus losses in runoff and the phosphorus concentration in a body of water will be weaker where incidental phosphorus losses make up a small proportion of all phosphorus entering the body of water.

Landscape characteristics: As the models described above attempt to capture, the amount of incidental phosphorus in runoff that reaches a body of water is influenced by slope, vegetation, and soil characteristics of the land over which the runoff moves before it reaches the water.

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8c: Phosphorus concentration in surface waters → Algal biomass

Description of relationship

High total phosphorus concentrations in surface waters promote algal growth.

Summary of evidence

An increase in algal biomass due to higher total phosphorus levels depends on at least some of the algae species present being limited by phosphorus availability. While this is determined by many factors specific to the individual water body or waterway, there has been substantial controversy about whether nitrogen or phosphorus is the primary limiting nutrient in freshwater marine systems, and therefore which is more important to regulate (Lewis, Wurtsbaugh, and Paerl 2011). Historically, phosphorus has been considered the more limiting nutrient in freshwater systems, and nitrogen the more limiting nutrient in marine systems (Dodds and Smith 2016). However, more recent studies and syntheses shows that both nutrients can be important limiting factors across ecosystems.

A meta-analysis of experiments evaluating primary producer biomass response to changes in nitrogen and phosphorus concentration in freshwater and marine ecosystems found that phosphorus limitation was equally strong across ecosystem types and that the addition of either nitrogen or phosphorus separately results in statistically identical growth responses in freshwater ecosystems (Elser et al. 2007). Phosphorus had a weaker effect on primary producer growth than nitrogen in marine systems. In all cases, the addition of both phosphorus and nitrogen together resulted in larger growth responses than the addition of either nutrient alone. This suggests that the two nutrients are generally closely balanced in aquatic systems, so the addition of one of them soon causes the other to become limiting (Elser et al. 2007).

A 2012 effort by the Canadian government to define thresholds for phosphorus used long-term monitoring data from agricultural watersheds across Canada to assess the relationship between phosphorus concentrations and chlorophyll a, and to identify threshold phosphorus concentrations at which algal community metrics showed significant changes (Chambers et al. 2012). There was a highly significant relationship between the summer mean concentration of sestonic (suspended algae) chlorophyll a and summer mean total phosphorus that explained 38% of the variance in sestonic chlorophyll a. The relationship between the summer mean concentration of benthic chlorophyll a and summer mean total phosphorus was weak (benthic chlorophyll a was more strongly related to nitrogen concentration). Two metrics of the benthic algal community were correlated with total phosphorus: the Trophic Diatom Index and the Eastern Canadian Diatom Index for alkaline streams (IDEC-alkaline), with total phosphorus thresholds of 0.032 mg/L and 0.022 mg/L, respectively (Chambers et al. 2012).

While the two large-scale studies described above focused on lakes, there is also evidence for a positive relationship between phosphorus and algal biomass. A compilation of chlorophyll concentrations and nutrient levels from 300 stream sites across North America, New Zealand, Spain, and Australia found that the total phosphorus concentration was positively correlated with both mean and maximum benthic chlorophyll concentrations (Dodds, Smith, and Lohman 2002).

Some effects of increased algal biomass are only evident during periods of extremely high algae growth known as algal blooms. Therefore, the relationship between total phosphorus concentrations and the probability of an algal bloom may also be of interest. No evidence describing this relationship generally was found, but a study using long-term monitoring data at Lake Okeechobee, Florida, provides an example of how this can be assessed for an individual water body when data is available (Havens and Walker 2002).

Strength of evidence

Moderate: Several large-scale, long-term field studies including data from Canada and a meta-analysis have found a positive relationship between total phosphorus and sestonic algal chlorophyll concentrations in lakes, as well as total phosphorus and benthic algal chlorophyll in streams. However, the relationship between total phosphorus and benthic algae in lakes appears to be weaker, and the influence of other factors (see below) may weaken or eliminate the relationship in certain contexts.

Other factors

Nitrogen concentrations: Many studies found an interaction in the effect of phosphorus and nitrogen on algal biomass, suggesting that the two nutrients are colimiting to algal growth (Dodds, Smith, and Lohman 2002). This can be due to variation in algal species responses to nutrient levels (species vary in their optimal N:P ratios and their ability to take up and store nutrients) or to non-equilibrium conditions (Lewis, Wurtsbaugh, and Paerl 2011).

Stream characteristics: Streams with steeper gradients tend to have lower benthic chlorophyll concentrations than streams with shallow gradients, and benthic chlorophyll concentrations are generally higher in streams with natural substrates than artificial substrates (Dodds, Smith, and Lohman 2002).

Climate: Some studies have found a positive relationship between water temperature and chlorophyll concentrations, corresponding to a negative relationship between latitude and chlorophyll concentrations (Dodds, Smith, and Lohman 2002).

Hydrology: The hydraulic residence time can influence the relationship between phosphorus and algal biomass, especially when the water residence time is shorter than the doubling time for phytoplankton (Smith 2003).

Sources

9a: Change in phosphorus management practices → Crop yields

Description of relationships

Fertilizer type

Highly soluble phosphorus fertilizers increase crop yields more than less soluble fertilizers. Fertilizers containing ammonium can decrease crop yields when applied in high concentrations near seeds.

Fertilization rate

Crops in soils with low background phosphorus concentrations show positive yield responses with moderate fertilizer applications (13-30 kg P/ha). Crops in soils with higher background phosphorus concentrations may show no yield response to fertilizer application.

Fertilizer application timing

P fertilizer application early in the growing season has a stronger positive effect on yield than fertilizer application later in the growing season.

Fertilizer application location

P fertilizer application in a band at or near the seeds may have a positive effect on yield compared broadcast application, but evidence is inconsistent. Fertilizers containing ammonium may need to be placed in a band slightly away from the seeds to prevent toxicity.

Summary of evidence

Fertilizer type

Two aspects of fertilizer type are important to crop yields. First, fertilizer phosphorus in more highly soluble forms is more immediately available to plants than fertilizer phosphorus in less soluble forms; however, as discussed in link 1a, most commercially available inorganic phosphorus fertilizers are highly soluble. It is thought that highly soluble phosphorus fertilizers all promote crop yields equally, but no evidence directly comparing highly soluble phosphorus fertilizers was found (Chien et al. 2011). Second, phosphorus fertilizers containing ammonium (MAP and DAP) can reduce yields through direct toxicity to seeds when they are placed in high concentrations near the seeds (McKenzie et al. 2001; Grant et al. 2001; Nyborg and Hennig 1969). Evidence for this effect is discussed in the fertilizer application location section below.

Fluid phosphorus fertilizers may be more effective in increasing plant yields than granular phosphorus fertilizers because fluid fertilizers may be able to diffuse farther from the application site and remain available to plants longer than granular fertilizers; however, most studies on this topic do not hold other relevant factors constant and therefore have conflicting results, so this effect is highly uncertain (Chien et al. 2011).

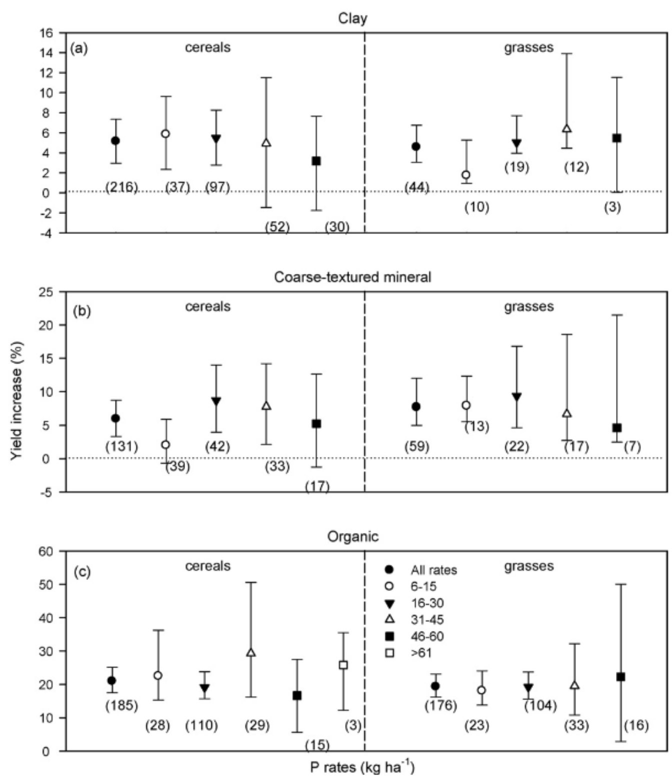
Fertilization rate

An often-mentioned concept in phosphorus fertilizer management is that there is an optimum soil test phosphorus level for each crop type, and that crop yields only respond to phosphorus fertilization if the soil test phosphorus falls below that level (Sims and Sharpley 2005). Several large-scale field trials in Alberta have provided some support for this assertion. A field study that applied TSP to pea crops found that crop yields responded positively to phosphorus fertilization when soil test phosphorus was below

30 kg/ha, but did not respond to phosphorus fertilization when soil test phosphorus was above that level. The same study found that the application of 13.1 kg P/ha optimized phosphorus yields, and that higher fertilization rates did not further increase yields (McKenzie et al. 2001). Similarly, a set of 154 field fertilizer trials across Alberta that applied MAP found that the probability of a yield response to phosphorus fertilization decreased with the fertilization rate (McKenzie et al. 2003).

No syntheses of phosphorus fertilization studies in North America were found, but a meta-analysis of 80 years of phosphorus fertilization and crop yield research in Finland gave conceptually similar results to these field trials. The meta-analysis included 444 observations representing the application of SSP at rates ranging from 6 to 100 kg/ha and found that cereal crop yields were significantly increased (by 6-23%) at low fertilization rates (6-15 kg/ha in clay and organic soils, 16-30 kg/ha in coarse mineral soils), but that higher fertilization rates did not further increase the crop yields, as shown in Figure 2 (Valkama et al. 2009).

Figure 2: Yield response of cereals and grasses to varying phosphorus fertilization rates in clay, coarse-textured mineral, and organic soils.



Source: (Valkama et al. 2009)

Fertilizer application timing

Fertilizer phosphorus application timing relative to planting is crucial for crop yields. Because phosphorus availability is crucial during early plant growth, a phosphorus deficit during the first few weeks of growth can result in low crop yields. Phosphorus deficits later in the growing season have a much weaker effect on yields. For this reason, most phosphorus fertilizers are applied with or just before seeding (Grant et al. 2001).

Fertilizer application location

P fertilizer placement at the seed or in a band near the seed is thought to allow for more effective root uptake during early growth than broadcasting, which is rarely used for phosphorus fertilizers; several individual research studies on various crop types have shown enhanced yield increases from banded or seed-placed phosphorus fertilizer compared to broadcasting (Grant et al. 2001). However, a meta-analysis of experiments comparing the yield effects of broadcasting to fertilizer placement, including 136 datasets for phosphorus fertilizers, found no difference between the two application locations for phosphorus fertilizers (Nkebiwe et al. 2016).

As mentioned above, some crops are sensitive to seed damage and low seedling emergence when high concentrations of phosphorus sources containing ammonium (MAP, DAP) are placed near the seed. In these cases, banding the fertilizer slightly away from the seeds can prevent toxicity while still providing sufficient phosphorus to the growing plant. A field study of phosphorus fertilizer placement in Canada showed that rapeseed and barley produced higher yields when MAP was placed 2.5 cm below the seed row than in the seed row, but this effect only occurred when fertilization rates were high (78 kg/ha for barley, 39 kg/ha and 78 kg/ha for rapeseed) (Nyborg and Hennig 1969).

Strength of evidence

Fertilizer type

Low: No studies were found that directly compared yield effects from different types of soluble phosphorus fertilizers. The potential negative effect of ammonium-containing fertilizers on yield is well documented in several field studies.

Fertilization rate

Fair: Consistent evidence across several large-scale field trials in Canada and a meta-analysis of studies in Finland shows that crop yield responses to phosphorus fertilizer increases with fertilization rate when soil test phosphorus is low, but only for relatively low phosphorus fertilization rates. However, these studies do not provide enough information to assess the critical soil test phosphorus level that determines this relationship, especially for different types of crops and in different locations.

Fertilizer application timing

Fair: Evidence that phosphorus availability during the first few weeks of growth is crucial to yield responses is consistent, but there is little information available on the length of the critical window for various crop types, which would allow farmers to make informed decisions about when to apply phosphorus fertilizer.

Fertilizer application location

Low: Studies comparing various locations for fertilizer phosphorus banding and a meta-analysis that compared phosphorus fertilizer broadcasting to placement had inconsistent results. The importance of banding ammonium-containing fertilizers away from seeds seems logical, but only one study provided evidence for that effect.

Other factors

Crop type: Yield responses to phosphorus fertilizer were similar across crop types, but wheat yield response was slightly lower than barley and canola yield responses (McKenzie et al. 2003).

Soil test P: Crop yield responses to phosphorus fertilization tend to be larger when soil test phosphorus is low (Valkama et al. 2009; McKenzie et al. 2003).

Soil type: The meta-analysis of Finnish studies found a stronger effect of phosphorus fertilization on organic soils (mean yield increase, 15%) than on coarse-textured mineral soils (10%) or clay soils (5%). These differences were greater when soil test phosphorus levels were low (Valkama et al. 2009).

Temperature: The positive effects of fertilizer phosphorus on crop yield when applied at or before planting are stronger when soil temperatures are lower, because soil phosphorus reserves are less available to plants under these conditions (McKenzie et al. 2003).

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10a: Change in phosphorus management practices → Costs

Description of relationship

Changing phosphorus management practices increases costs for farmers and requires a greater time investment and more expertise than continuing status quo phosphorus management practices.

Summary of evidence

Changes in phosphorus management may influence farmers' fertilizer costs (if more expensive fertilizer types are used or the total amount of fertilizer use changes) and equipment costs (if fertilizer placement techniques require additional equipment). It is difficult to generalize these costs, as they depend on the farmer's current fertilizer strategy and equipment, farm-specific factors that determine equipment suitability, and regional cost differences. Since enhanced-efficiency phosphorus fertilizers have not been widely tested or adopted, changing fertilizer types is not likely to be a driver of cost changes. Farmers who are able to apply less phosphorus fertilizer to certain fields with sufficient phosphorus reserves may save money on fertilizer, but any additional costs from soil testing must be accounted for as well.

Adjusting the timing and placement of phosphorus fertilizer can increase labor requirements. Farmers often minimize the time spent applying phosphorus fertilizer by adding it in one large application every few years (Sims and Sharpley 2005). Switching to smaller annual applications could double or triple the time spent on fertilizer applications, which can affect labor costs and time available for other tasks.

In addition to improving management practices, farmers will need to rely on a rapidly expanding base of biological and agronomic knowledge that is often specific to certain agroecosystems, regions, soil types and slopes. Making the right decisions at the farm level in terms of input-use efficiency, human health and resource protection is becoming an increasingly knowledge-intensive task (Tilman et al. 2002).

Precision agriculture practices require new skills, and only a limited number of farmers have them or are willing to obtain them. Age, attitude and education of producers have been identified as significant barriers in the US: the majority of farmers are over 55 years old, have partial or complete high school education, and have limited interest in changing practices and using computerized systems (Robert 2002).

A study of the profile of farmers that choose to adopt conservation-compatible practices in the US found that the greater the management skills needed to make a farming practice profitable, the greater the size (hectares, income and commodity payments received) of adopting farms. This indicates that these practices hold more appeal for large-scale farm operators concerned with maximizing farm profits, and are less likely to be adopted by operators of small farms focused primarily on non-farm activities (Lambert et al. 2007).

Strength of evidence

Low: Changing phosphorus management practices has the potential require additional resources (money, time, expertise) or to decrease expenditures, depending on the farmer's current practices. No studies were found that specifically examined the cost of changing phosphorus management practices; the evidence discussed here is for more general assessments of barriers to precision agriculture and conservation practices.

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