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# Managing nitrogen legacies to accelerate water quality improvement

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**Increasing incidences of eutrophication and groundwater quality impairment from agricultural nitrogen pollution are threatening humans and ecosystem health. Minimal improvements in water quality have been achieved despite billions of dollars invested in conservation measures worldwide. Such apparent failures can be attributed in part to legacy nitrogen that has accumulated over decades of agricultural intensification and that can lead to time lags in water quality improvement. Here, we identify the key knowledge gaps related to landscape nitrogen legacies and propose approaches to manage and improve water quality, given the presence of these legacies.**

The past 100 years have seen more than a threefold increase in world population, accompanied by large-scale changes in land use and an intensification of agricultural production practices to secure an adequate food supply. Human activities have greatly accelerated the nitrogen (N) cycle, with excess N leaching to surface and groundwaters, causing problems of eutrophication, aquatic toxicity and drinking-water contamination[1–4]. Protecting water quality in the face of a growing population and the corresponding demands on agriculture is critical to ensuring both water and food security for generations to come.

Incidences of eutrophication and harmful algal blooms have increased substantially in recent decades[1,2,5,6]. Task forces have been formed and policies have been set, from the local to the international levels, to address problems with water quality[7–12]. Despite these actions and widespread implementation of a range of conservation measures, water quality goals for the most part remain elusive[13–15]. Stream nitrate loads in many watersheds where conservation measures have been implemented remain high, or even continue to increase, and downstream water bodies continue to experience algal blooms, driven by excess nutrients[15,16].

Multiple examples of failures to achieve water quality goals can be found. In the United States, the Gulf of Mexico Watershed Nutrient Task Force was formed in 2008 with the goal of reducing the size of the hypoxic zone to 5,000 km$^2$ by 2015. In 2015, however, the hypoxic zone was more than three times the target size—16,000 km$^2$—and the goal has now been extended to 2035[13,14]. In Europe, collaborations for mitigating the Baltic Sea eutrophication problems were established in the 1970s with the creation of the Helsinki Commission (HELCOM)[17]. The commission's efforts contributed to establishment of the Baltic Sea Action Plan, through which specific nutrient-loading targets were set for each country in the Baltic Sea drainage basin[18,19]. For the European Union, adoption of the Nitrate Directive in 1991[20] and the EU Water Framework Directive in 2000[21] has resulted in numerous national policies designed to achieve good chemical and ecological status. Despite these efforts, hypoxic-zone extent in areas across the world, from the Gulf of Mexico and the Chesapeake Bay in the United States to the Baltic Sea in Europe, has been either increasing or not demonstrating any clear decline in the past three decades (Fig. 1)[22].

Numerous obstacles can prevent the achievement of water quality goals, including lack of knowledge regarding appropriate conservation measures, lags in implementation of new management practices, limited funding and a lack of willingness of large producers to reduce fertilizer application rates[23–25]. However, recent research suggests that one of the key drivers of the apparent lack of success in water quality improvement following implementation of watershed conservation measures is legacy stores of N[26–31]. These stores have accumulated in landscapes over decades of fertilizer application and agricultural intensification and can contribute to elevated N levels in streams, lakes and estuaries decades after inputs have ceased, leading to time lags in water quality improvement. Here, 'time lag' is defined as the time elapsed between implementation of watershed conservation measures and measurable improvements in water quality in aquatic systems. Although an understanding of N legacies and time lags has existed within the scientific community for decades, this mainly theoretical understanding has not been translated to field-based quantifications and monitoring that can adequately support the policy arena, where there still exists an expectation of short-term water quality improvement[14,16,32]. A lack of success in meeting goals generates skepticism and disillusionment by policy makers and farmers on the efficacy of conservation measures adopted[27,32].

Given the lack of water quality improvement within the targeted periods, despite billions of dollars in investment, it is critical that

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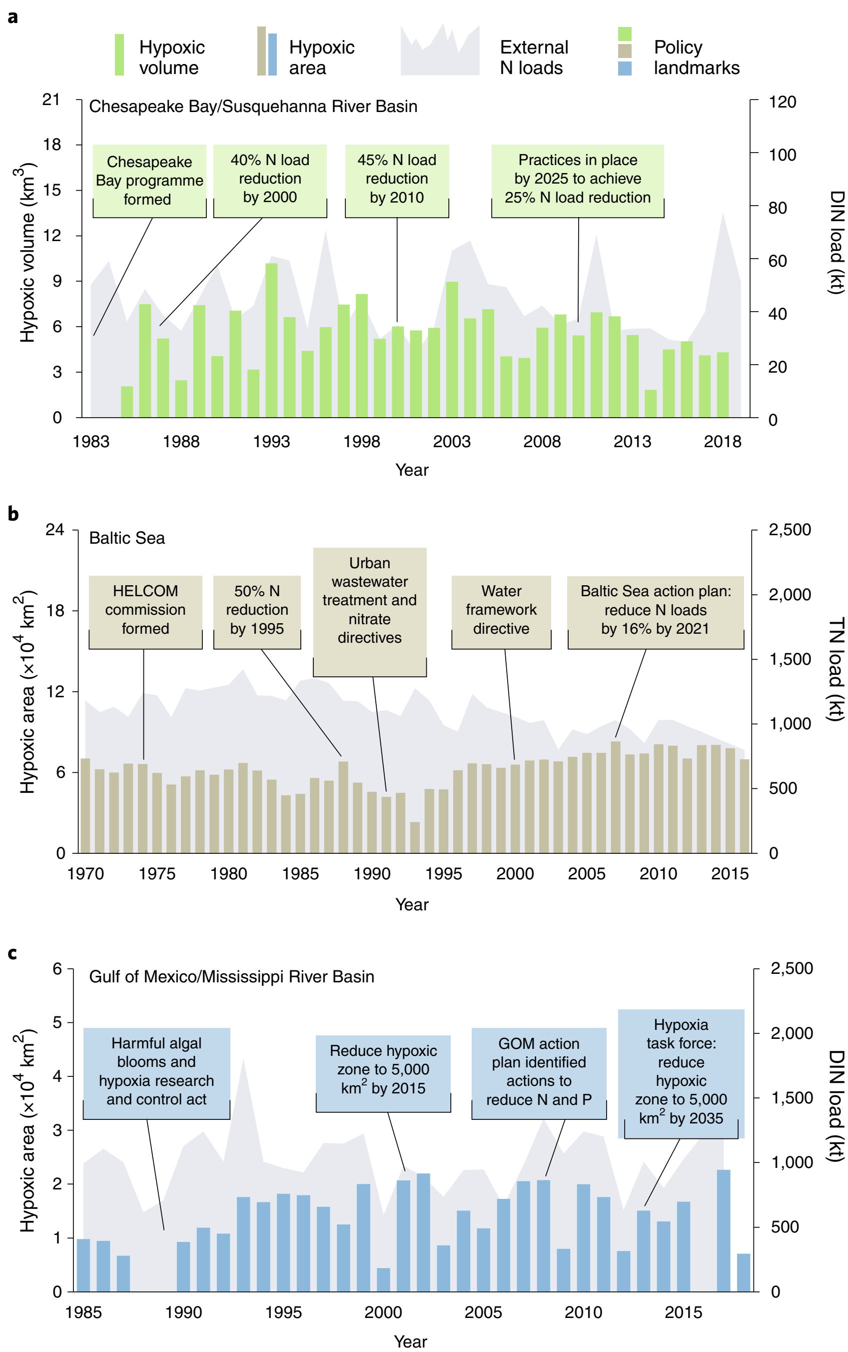



**Fig. 1 | Time lines of policy measures and hypoxic-zone size reveal lack of response to policy interventions across Europe and North America. a–c**, Time series for the hypoxic area or volume, the riverine N load and selected policy landmarks and goals for the Chesapeake Bay/Susquehanna River basin[80] (**a**), Baltic Sea (**b**) and Gulf of Mexico (GOM)/Mississippi River basin[23,26,81–83] (**c**). External loads in **a** represent dissolved inorganic N (DIN) ($NO_3^- + NO_2^-$) for the Susquehanna River, which accounts for more than half of the annual N load to the Chesapeake Bay[26]. Hypoxic volume data represent the early June volume in the Maryland mainstem of the Chesapeake Bay with dissolved oxygen (DO) concentrations <2 mg l$^{-1}$ (ref. [80]). External loads in **b** represent an estimate of all total N inputs (TN) to the Baltic Sea, including riverine inputs, atmospheric deposition, direct-point sources and other diffuse inputs[84]. Hypoxic area data represent the areal extent of the hypoxic area (DO < 2 mg l$^{-1}$) for the Baltic Sea's Bornholm basin[85]. External loads in **c** represent DIN ($NO_3^- + NO_2^-$) for the Mississippi River[13], and the hypoxic area represents the mid-summer, bottom-water hypoxic area (DO < 2 mg l$^{-1}$) (ref. [86]).

we develop methodologies to quantify N legacies and lag times. Such estimates are critical not only for managing expectations, and setting appropriate policy goals, but also for designing conservation measures that can contribute to the minimization of lag times. Despite the overall understanding of the importance of legacy N in delaying water quality improvement, we currently lack (1) a comprehensive characterization of the nature, size and reactivity of N legacies across spatial and temporal scales, (2) quantitative understanding of the relationship between the magnitude and forms of legacy accumulated and time lags to water quality improvement as a function of landscape and management drivers and (3) policy instruments and economic incentives that acknowledge time lags and balance trade-offs between short- and long-term costs, benefits and risks. Here, we synthesize the various challenges and knowledge

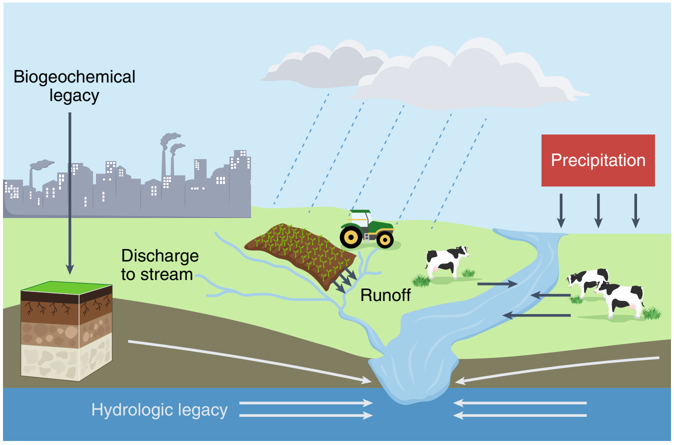



**Fig. 2 | Transport and retention of N across human-impacted landscapes.** Agricultural areas are N sources due to over-application of commercial fertilizers and livestock manure. N can be retained as biogeochemical legacy in agricultural soils in the form of organic N. N can also be retained in a dissolved form in groundwater as hydrologic legacy. N from current application, as well as legacy stores, is transported to lakes and coastal zones via overland flow as well as through riverine and subsurface transport pathways.

gaps that exist regarding N legacies and propose strategies for managing water quality, given the presence of these legacies.

## Legacy stores build up over decades of agricultural intensification

To more effectively manage legacy N stores and their effects on water quality, it is important to distinguish between various types of N accumulation. Legacy N can accumulate in numerous landscape elements, including soils, groundwater, reservoirs, lake and stream sediments, riparian areas and landfills (Fig. 2). This legacy N can exist in various forms, including as dissolved nitrate in groundwater and soil water[33,34] and as organic N within the soil profile[35,36].

Mass-balance studies can theoretically be used to estimate the legacy mass as a function of N inputs (for example, fertilizer N, manure N, atmospheric N deposition, biological N fixation, wastewater N) and outputs (crop N uptake, denitrification, riverine N export)[37]. The magnitude of denitrification, however, is extremely difficult to quantify at the landscape scale, such that until recently it has been commonly used to close the N mass budget on the basis of an underlying assumption that there is no legacy accumulation in the landscape[38–40]. This assumption has been challenged by recent studies that have demonstrated evidence of organic N build-up in the soil profile and dissolved N build-up in the vadose zone and groundwater[31,35,41]. Currently, however, there is a lack of quantitative understanding of the magnitudes of legacy N accumulated in the various environmental compartments. Also uncertain are the timescales over which legacy N continues to leach into water bodies, which is a function of not only the total mass of legacy accumulated, but also the timescales of depletion. These timescales depend on the mechanisms of release from legacy stores and the degree of connectivity in the system, which is controlled by a range of factors, from the existence of tile drains, which provide fast transport of water and nutrients from agricultural fields[42–44], to extreme precipitation events that mobilize stored legacies.

## Relationships between watershed N inputs and stream N loads

To set appropriate policy goals and to design effective management strategies, it is critical to better our understanding of the relationship between N inputs to a watershed and N loads in streams exiting the watershed. Traditionally, stream N loads have been conceptualized as being linearly correlated with net N inputs[45–47], with net N inputs being quantified as the difference between inputs (fertilizer N, manure N, biological N fixation, atmospheric N deposition, N in human waste) and outputs (crop N uptake). These positive correlations have been established on the basis of a snapshot quantification of net N inputs and N loads for a range of watersheds (Fig. 3a), with stream N loads constituting approximately 25% of net inputs[47]. Although this relationship between inputs and outputs appears to hold true across watersheds, especially in watersheds with relatively higher N inputs, the relationship often begins to break down when considering how individual watersheds respond to changes in N inputs over time. More specifically, while some watersheds will demonstrate proportional decreases in annual N loads in response to unit decreases in N inputs, this linear behaviour will generally be observed only in watersheds with minimal lag times (Fig. 3b,c). By contrast, analysis of multi-decadal trajectories of N inputs and N loading demonstrates that watersheds commonly exhibit nonlinear responses to decreases in N inputs (Fig. 3d–k)[26,32,48]. In other words, decreases in watershed N inputs do not result in proportional decreases in N loading.

This nonlinearity in response to changes in N inputs can be described as a hysteresis effect, meaning that watershed N loads at any point in time are not just a function of current N inputs, but of the history of N inputs to that watershed over time. If net N inputs to a watershed have been high for decades, leading to an accumulation of N in soils, sediments and groundwater, N loading may remain elevated, even after decreases in inputs, due to a slow depletion of legacy N from the system. As an example, net N inputs to the Wisconsin River watershed in 1970 were approximately 25 $kg\,ha^{-1}\,y^{-1}$ (Fig. 3d). Over time, N inputs increased by approximately 40% and then, in the early 1980s, began to decrease again. By 2015, N inputs had decreased back to 1970 levels, but N loads remained nearly double what they were in the earlier period. This phenomenon of higher present-day N loading at 1970-level net N inputs is a clear example of a nonlinear, hysteretic watershed response and demonstrates the effects of legacy N accumulation and depletion trajectories over periods of substantial changes in N inputs. As shown in Fig. 3, these trajectories for accumulation and depletion of legacy N can vary across watersheds, leading to variations in the size and shape of the anticlockwise hysteresis loop that are governed by numerous natural and anthropogenic controls, including (1) nutrient source and distribution, (2) watershed topography, (3) soil type, (4) climate, (5) tile-drainage densities and (6) groundwater travel times[48]. As an example, in our modelled results (see Supplementary Fig. 1 for model details), we show that as groundwater travel times increase, watershed response times also increase, leading to a wider hysteresis loop and a slower path to watershed recovery (Fig. 3b,c). The watershed trajectories shown in Fig. 3 are consistent with the results of both field-scale and watershed-scale studies indicating that lag times to improvements in water quality are ubiquitous and often multi-decadal[16,32].

## Modelling N legacies and predicting lag times

Quantification of watershed lag times, and the spatial variability of lag times within and across watersheds, is critically important for setting realistic policy goals and expectations and for choosing appropriate mitigation strategies. Existing nutrient fate and transport models, including statistical models such as SPARROW, GlobalNEWS and GWLF and process-based models such as INCA, HBV and SWAT, rarely account for N legacies and lag times[16,49]. Statistical models generally assume the N cycle to be at a steady state and thus cannot account for legacy effects[47,50,51]. Process-based models theoretically have the ability to capture legacy effects; however, they usually ignore historical inputs and thus legacy

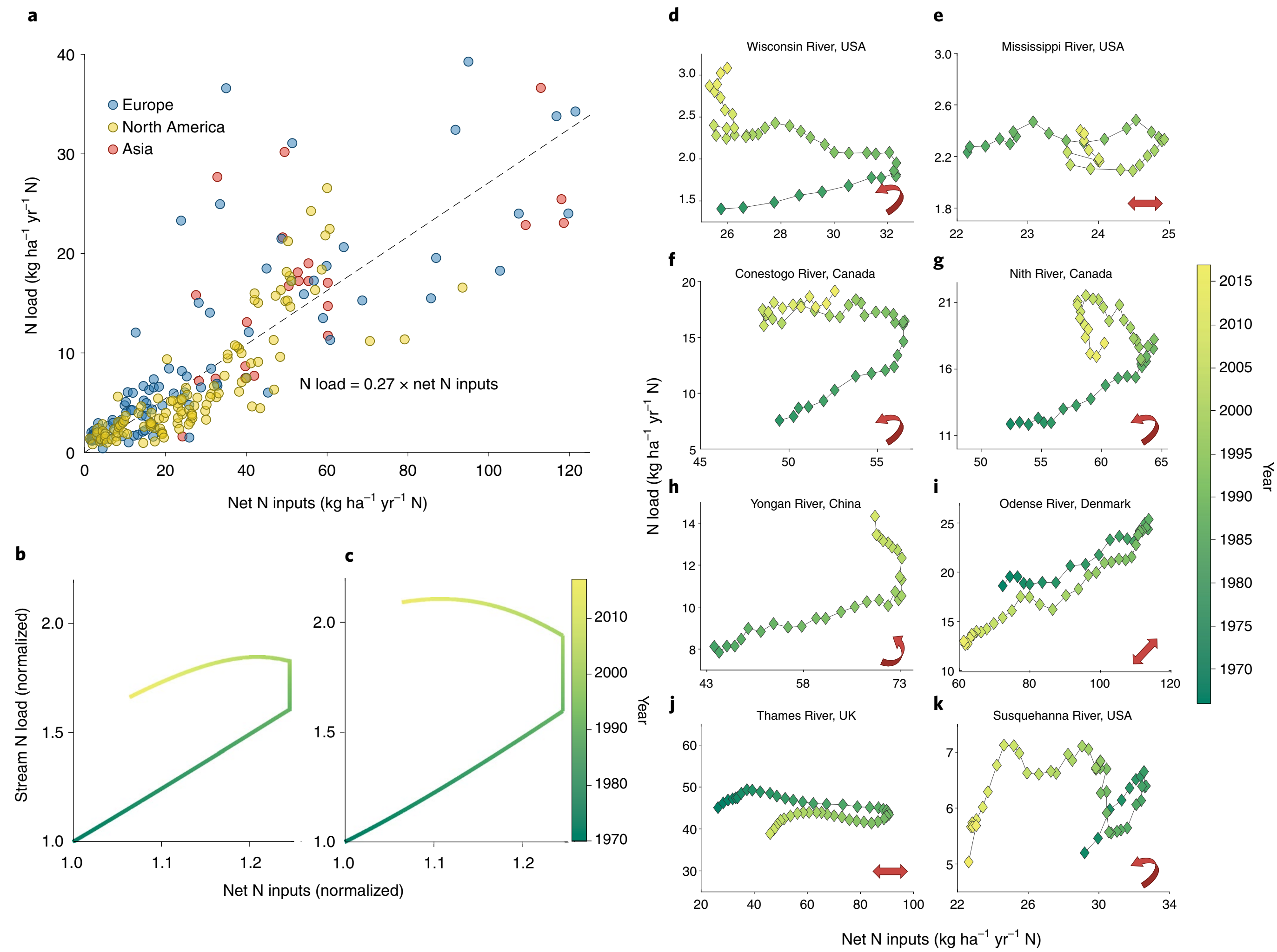



**Fig. 3 | Relationships between stream N loads and watershed net N inputs. a**, Markers, which represent individual watersheds across Europe, North America, and Asia[48,87–89], are positioned within the plot on the basis of net anthropogenic N inputs to the watershed and stream N loads. **b**,**c**, The ELEMeNT model[26] was used to simulate changing relationships between N inputs and stream N loads over time. Trajectories for net N inputs to the watershed are based on a typical US N input trajectory, with inputs increasing linearly between 1945 and 1990 and then decreasing linearly between 1990 and 2021, assuming a 33% decrease in N inputs between the 1990 peak and current-day values. In **b**, we assumed a mean groundwater travel time of 3 years and in **c** a travel time of 30 years. Lag times increase in **c** with the increase in groundwater travel times. **d**–**k**, These plots represent changing relationships between N inputs and stream N loads over time for watersheds around the world[41,48,55,88,90–96]: Wisconsin River, USA (**d**), Mississippi River, USA (**e**), Conestogo River, Canada (**f**), Nith River, Canada (**g**), Yongan River, China (**h**), Odense River, Denmark (**i**), Thames River, UK (**j**) and Susquehanna River, USA (**k**). Arrows in the plot represent the direction of the hysteresis loop, with a wider anticlockwise loop indicating stronger legacy effects.

accumulation and depletion. For example, ref. [52] developed a process-based modelling framework to estimate the effects of changes in agricultural management on the delivery of N to the Gulf of Mexico, leading them to estimate that it would cost US$2.7 billion annually to reduce the size of the Gulf hypoxic zone to the action-plan goal of 5,000 km$^2$. However, they acknowledge that their modelled conservation scenarios do not consider lag times, and thus even an immediate adoption of these measures would most likely not have an immediate effect on hypoxia.

The past few years have seen some progress in the modelling of watershed legacy N dynamics and lag times. Statistical approaches, involving prediction of current-year N loads as a function of N inputs over the past few decades, have been developed to capture lag times in the Mississippi River basin[53] and the Yongan watershed in eastern China[49,54]. The number of process-based models is limited, with existing models including the Exploration of Long-tErM Nutrient Trajectories (ELEMeNT) model[14,26,27,55,56], the nitrate time bomb model to estimate groundwater nitrate concentrations[57], the watershed-scale LM3-TAN model[58], SWAT-LAG, a modification of the SWAT model with addition of a groundwater travel time distribution[59], and a hillslope-scale aquifer model, also employing groundwater travel times[60]. Use of the ELEMeNT model within the Mississippi River basin has led to estimates of multi-decadal lag times to achieve policy goals for the Gulf of Mexico, even with the most aggressive scenarios to reduce surplus N within the watershed[13,14,59]. Future research is needed to more explicitly consider legacy processes in predicting lag times to watershed response.

## Lag times confound economic analysis and development of policy measures

Development of economically efficient water quality improvement strategies requires rigorous analysis of costs, benefits and effectiveness that explicitly take legacy effects into account[61]. One of the primary challenges in carrying out such analyses, given legacies, is that most of these approaches weigh short-term costs and benefits higher than long-term improvements in water quality. Two general categories of tools are used for carrying out cost–benefit analyses of water quality policies: integrated assessment models (IAMs) and econometric approaches. IAMs couple economic and ecological models to describe emission of pollutants, their transport through the landscape, outcomes of the pollutants in the environment and the valuation of these outcomes. The flexibility of IAMs makes them ideal for use in analysing potential regulatory policies. However, pollutant transport modules in most IAMs, including the new Hydrologic and Water Quality System used by the Environmental Protection Agency, do not consider legacy effects[61]. A recent modification of the Soil and Water Assessment Tool, the pollutant transport model in the Hydrologic and Water Quality System, clearly shows how water quality benefits can be overestimated by not accounting for time lags[59].

Legacy effects are also not considered in ad hoc econometric approaches, which are based on empirical water quality data and involve the use of regression models to evaluate the effects of past policies on water quality. In a study of water quality data from 240,000 monitoring sites across the United States, ref. [62] found declines in dissolved oxygen and pH and noted that the biggest exception was for nitrates, for which there was a slightly positive trend. Legacy effects would explain this discrepancy as nitrate is known to accumulate in groundwater and have slow subsurface travel pathways to streams. In another study of water quality trends across the United States, ref. [63] found a counter-intuitive positive correlation between the area of land enrolled in the US Conservation Reserve Program and poor water quality. As argued by the authors, this finding may be due to their analysis not accounting for potentially important lags between land conservation and water quality benefits. In other words, the long-term influence of legacy N was not taken into account.

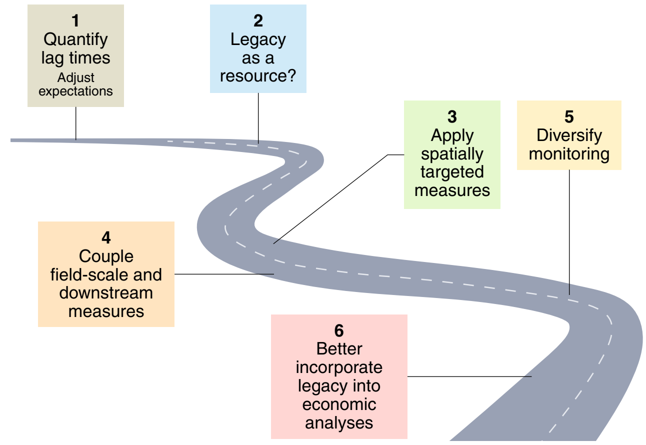



**Fig. 4 | Strategies for solving the legacy N problem.** We identify five key focus areas for developing solutions to problems associated with landscape-scale N legacies. (1) Magnitudes of N accumulation must be quantified, and realistic lag times associated with the depletion of these legacies should be estimated. (2) Nutrient management scenarios should be developed that allow for a drawdown of N legacies in upland soils. (3) Spatially explicit estimates of legacy N accumulation will allow us to target implementation of conservation measures. (4) Watershed conservation measures should rely on both field-scale (nutrient management, cover crops) and downstream (wetlands, buffers) measures to minimize lag times. (5) Diversification of monitoring approaches (for example, measure nitrous oxide emissions or soil nitrate concentrations instead of focusing on watershed outlet) to find evidence of success. (6) Ensure that hydro-economic modelling approaches account for legacy effects and that time windows for evaluating the success of implemented policy are sufficiently long.

## Policy instruments for water quality protection

The scientific understanding of time lags has not been adequately translated to the policy arena, where there still exists an expectation of short-term improvements in water quality. These expectations stem from the dramatic improvements in water quality that have occurred in many regions after implementation of point-source control measures, such as upgrades to wastewater treatment. Such improvements have occurred after passage of the 1972 Clean Water Act in the United States, the 1970 Canada Water Act and the European Urban Waste Water Treatment Directive of 1991, as well as widespread implementation of eutrophication control programmes in China[62,64–66]. The very different timescales of response for point-source versus non-point-source pollutants is not often appreciated, and time lags are sometimes argued to be a generic excuse to not meet water quality targets[32]. The problem is further confounded by two key challenges: (1) a lack of measured water quality data at appropriate scales often makes it challenging to evaluate where goals have been met and where they have failed;[15] (2) soft or voluntary measures adopted for water quality improvement (for example, Germany, United Kingdom, Canada, United States) often have low adoption rates while more prescriptive measures (for example, Denmark, Netherlands) can lead to dissatisfaction[24,67]. It is often difficult, if not impossible, to disentangle these effects from delay in water quality improvement due to legacies.

## Call for action to accelerate water quality improvement

Given the critical role of legacy N in delaying water quality improvement, it is important to integrate legacy considerations into watershed management and planning. While others have emphasized the importance of taking legacy into account in the policy arena, recommendations have focused narrowly on quantifying legacy and raising awareness about time lags[68]. In the following, we build on these recommendations to develop a more integrated, systemic approach to improving water quality, given the presence of legacies. Specifically, we outline six key strategies towards reaching this goal (Fig. 4).

**Strategy 1—quantify lag times and adjust expectations.** While the existence of lag times is well established, there is still considerable uncertainty regarding how lag times vary across the landscape. This uncertainty makes it difficult, if not impossible, for policy makers to make realistic estimates regarding time frames for achieving water quality goals. To address this issue, it is important to (1) provide estimates of lag times in various landscapes using a combination of field data and modelling and (2) develop methodologies to communicate lag times to stakeholders[68]. The former helps in the setting of appropriate goals, while the latter helps in managing expectations when the goals are not met. Given the complexity of describing watershed biogeochemical processes over long timescales, legacy modelling is still in its infancy, and data needed to validate existing models are often limited. Model development needs to be accompanied by targeted measurement of multiple N stores (for example, legacy N in soil and groundwater) and fluxes (for example, $N_2O$ fluxes to constrain denitrification, hydrogeologic datasets to estimate groundwater travel times and fluxes) instead of sole reliance on water quality monitoring at the watershed outlet. Multiple datasets will help reduce model equifinality and contribute to more robust predictions over long timescales.

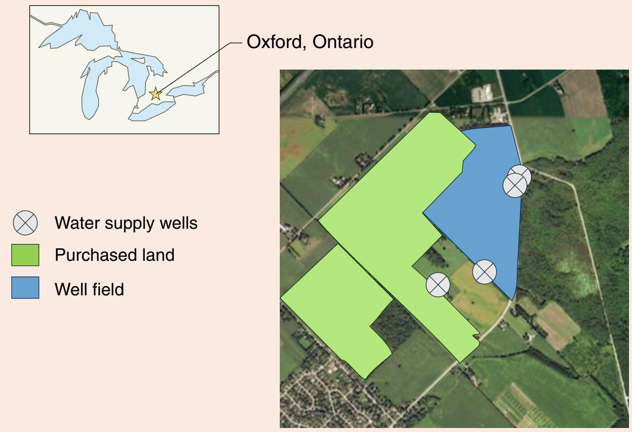
**Strategy 2—legacy as a resource.** The existence of soil N legacies implies that, where soil N availability is high, lower fertilizer application rates may not lead to notable declines in crop yields. Indeed, a global meta-analysis of N sources to cereal crops used $^{15}$N-labelled fertilizer to show that only a fraction of N in crops (41% for maize, 32% for rice and 37% for small grains) is from current-year

**Box 1 | Legacy solutions: a success story**

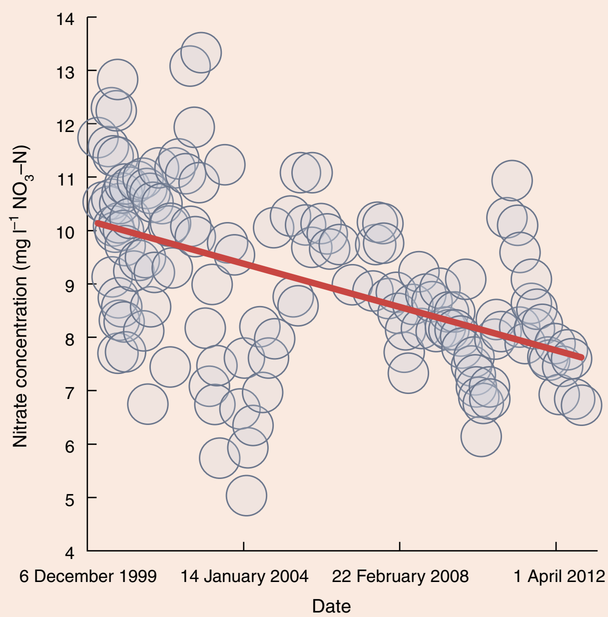
In the mid-1990s, nitrate concentrations in the three drinking-water production wells in Oxford County, Ontario, were either approaching or exceeding the nitrate drinking-water standard (10 mg l$^{-1}$ $NO_3$–N) (ref. [70]). Concentrations had been increasing over the past two decades in response to intensive application of fertilizer and livestock manure on surrounding cropland, and county water managers at this point were faced with a choice: either install a drinking-water treatment plant or directly improve the quality of groundwater being tapped by the supply wells. Acknowledging the existence of N legacies in the system (Strategy 1), water managers adopted a strategy that included both long-term solutions and short-term actions (Strategy 4).

First, in 2002, the county purchased 111 ha of land within the 2 yr capture zone of the supply wells with the intent of implementing agricultural management practices that would reduce the leaching of nitrate to these wells (upper figure). This land was then rented back to local farmers, with the stipulation that the farmers would strictly follow guidelines about how the land would be managed. These guidelines included a change in cropping system, from continuous corn to a corn–soybean–wheat rotation, the planting of winter cover crops and reductions in fertilizer use. Changes began to be implemented during the 2003 growing season, and fertilizer application rates were reduced by approximately 50%, such that the site went from an N surplus of 25 kg ha$^{-1}$ yr$^{-1}$ to an N deficit of 27 kg ha$^{-1}$ yr$^{-1}$. Interestingly, the reduction in fertilizer application led to no decrease in crop yield, indicating that the harnessing of accumulated N legacies within the soil profile can contribute to both cost savings and environmental benefits (Strategy 2)[70,97].

To ensure that the nutrient management approach was succeeding (Strategy 5), soil data from the shallow vadose zone was used to estimate changes in stored nitrate over time. After two years, these monitoring data revealed that the total stored $NO_3$–N mass beneath the root zone, to a depth of 2.5 m, had decreased by approximately 60% and that porewater nitrate concentrations had decreased by 50%, parallel to the 50% decrease in fertilizer N inputs. Nitrate concentrations in groundwater wells, however, showed a slower rate of decline, decreasing from ~10 mg l$^{-1}$ in 2002 to ~7 mg l$^{-1}$ in 2013 (lower figure)[70]. The slower reduction in the groundwater wells is indicative of longer unsaturated-zone travel times, with numerical modelling demonstrating that the total travel time to the drinking-water wells could range from 7 to 40 years (ref. [97]). By the spring of 2012, these interventions brought nitrate concentrations in Woodstock public supply wells back down to safe levels while maintaining agricultural productivity in the area of the wells.




Well locations in Oxford County, Ontario.




Declining temporal trend in nitrate concentrations in groundwater wells.

N fertilizer while the remaining comes from mineralization of soil organic N[69]. Field studies also indicate that lowering fertilizer application rates does not necessarily impact crop yields[70,71], alluding to the existence of legacy N stores in the landscape (Box 1). Effective use of these stores can contribute to lower N fluxes to streams without sacrificing crop production while also reducing greenhouse gas emissions given the strong linear relationship between fertilizer addition and $N_2O$ fluxes[72]. However, the ability of legacy N stores to sustain crop yields would vary spatially as a function of soil and crop type, topography, land use history and climate drivers. Large-scale adoption of such changes would thus require both technological and societal innovations. High-resolution airborne and spaceborne technologies such as lidar and hyperspectral sensors can be used for developing estimates of soil N demand at the field scale and target fertilizer applications[73]. Such precision farming approaches need to be coupled with agronomic research that focuses on methodologies for effective use of soil legacy N. Technological innovations need to be accompanied by societal changes that provide incentive structures to protect farmers against crop yield loss from lower fertilizer application rates as well as regulatory approaches to penalize over-application of fertilizers and livestock manure.

**Strategy 3—spatial targeting of watershed conservation measures.** In many cases, conservation practices are advocated broadly, with little regard for differences in agricultural practices or legacy-related risks and opportunities. As a result, both public and private funds are often spent for adoption of new practices that might provide greater water quality benefits elsewhere[74,75]. Better spatial targeting can be achieved in a variety of ways, including better dissemination of information regarding geographically appropriate management practices and the use of strong economic incentives to strategically drive adoption of new practices in targeted locations. Regulatory approaches, which have been more widely adopted in some European countries, are another option for ensuring that conservation measures are implemented at key locations[76]. For example, if a region has large legacy N accumulation in soils, incentive or regulatory strategies that limit fertilizer applications can improve water quality without measurable impacts on crop yield while also reducing emissions of greenhouse gases such as $N_2O$ (ref. [72]). A given management strategy in an area with long lag times may not translate into immediate benefits, while the same strategy applied to an area with shorter groundwater travel times may yield a faster response[77]. Non-point-source mitigation strategies based on such spatially and temporally differentiated best management practices are the most cost-effective and efficient means of minimizing trade-offs between agricultural production and sustainable water resources[74,76,78].

**Strategy 4—couple field-scale and downstream measures to minimize lag times.** N legacies can accumulate in the soil or in groundwater, and different strategies are required for accessing these different legacies. Reducing fertilizer application can result in crops accessing soil N legacies, but it does not address groundwater legacies that have accumulated over decades of fertilizer application. While some denitrification does occur within the subsurface, it is limited due to a lack of organic carbon, which limits N removal. Groundwater legacies can be addressed only through implementation of more downstream controls such as wetlands[74,78,79], reservoirs and riparian buffers that intercept groundwater flow pathways and remove nitrate through plant uptake or denitrification. Such downstream measures contribute to more immediate effects but need constant maintenance to maintain functionality. Field-scale measures such as cover crops and nutrient management address the source of the problem in the soil root zone, but their benefits sometimes take longer to realize due to groundwater legacies that have accumulated over decades. A strategic combination of watershed conservation measures that address both soil and groundwater legacies can lead to the fastest watershed response times.

**Strategy 5—diversify monitoring to evaluate outcomes and inform adaptive management.** Successful adoption of watershed conservation is strongly dependent on public perception of the efficacy of watershed management practices. Given the existence of legacy, and potentially long, time lags to achieve measurable water quality benefits at larger scales, it is important to quantify the efficacy of watershed conservation practices at a multitude of scales, from the single tile-drained field to small first-order watersheds to large drainage basins. Impacts will be apparent at some scales and for some elements earlier than others, and these initial successes (or failures) can be used for adaptive watershed management. For example, adoption of measures such as cover crops and fertilizer reduction might lead to a more immediate reduction in nitrate in the soil water, as well as in $N_2O$ emissions, but it might take decades for reductions to be measurable at the watershed outlet[72]. Thus, for effective watershed management, we should monitor both aqueous and gaseous N fluxes at multiple scales, in surface water, groundwater and soil water, and focus on adaptive management that alters practices on the basis of measured responses.

**Strategy 6—better incorporate assessments of both long-term and short-term benefits into economic analyses.** Current economic assessments of water quality policy are often flawed due to assumptions of immediate water quality benefits in response to implementation of improved management practices. Economic tools such as cost-effectiveness analyses and cost–benefit analyses must therefore be modified to explicitly account for N legacies and time lags. Hydro-economic modelling approaches must explicitly include estimates of the time required to achieve water quality benefits through the coupling of economic models with process-based water quality models that take N legacies into account. In addition, econometric approaches, designed to evaluate the effectiveness of past policy measures, must account for lag times and ensure that the period over which effects are evaluated is sufficiently long.

## Conclusion

For the past century, the nitrate pollution problem has grown in tandem with growing populations, an intensification of agriculture and a warming climate. While many efforts are now focused on reversing this problem through increased controls on wastewater treatment plants and implementation of better agricultural management practices, our long history of N overuse to maximize crop yields is continuing to drive high stream nitrate concentrations and coastal eutrophication. Better understanding the role of legacy N in controlling water quality is crucial to better policy and better environmental outcomes.

## Data availability

Datasets for this research have been previously published[13,26,41,80,84–87,89,91,92,94,95]. All other data are available upon request.

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N.B.B. conceptualized the project, wrote the first draft, edited the paper and acquired the funding. K.J.V.M. conceptualized the project, edited the draft and wrote sections, contributed materials/analysis tools and analysed data. D.K.B. contributed materials/analysis tools, analysed data and edited the draft. P.V.C., R. Brouwer, G.D., J.J., R. Bhattacharya, B.H.J., M.C.C., G.D., N.N., S.B.O. and D.L.R. edited the draft. P.V.C. and N.B.B. acquired funding.

## Competing interests

The authors declare no competing interests.

## Additional information

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