Nitrate in United Kingdom Rivers: Policy and Its Outcomes Since 1970[†]

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Received April 29, 2010. Revised manuscript received July 20, 2010. Accepted July 23, 2010.

Modern conventional farming provides Western Europe and North America with reliable, high quality, and relatively cheap supplies of food and fiber, increasingly viewed as a potential source of fuel. One of the costs is continued widespread pollution of rivers and groundwater—predominantly by nutrients. In 1970, in both the United States and UK, farming was focused on maximizing yield and management practices were rapidly modernizing. Little attention was paid to the external impacts of farming. In 2010, diffuse pollution from agriculture is being seriously addressed by both voluntary and statutory means in an attempt to balance environmental costs with the continued benefits of agricultural production. In this paper we consider long-term changes in the concentration and flux of nitrate in five rural UK rivers to demonstrate the impact of agricultural intensification and subsequent policies to reduce diffuse pollution on river water quality between 1970 and 2010.

High Nitrate Concentrations in UK Water: Concerns and Legislative Response

Western Europe has, since the 1970s, been identified as a global "hot spot" of fluvial nitrogen flux (*1*−*3*). Some of the highest concentrations are found in UK rivers, where the "nitrate issue" has caused concern for several decades (*4*−*6*). The initial worry focused on the quality of drinking water (Drinking Water Directive EU 80/778) but, more recently, nitrate's role in eutrophication has become the main cause for concern, especially coastal waters where primary productivity is likely to be nitrogen-limited (*7, 8*). The 1991 Nitrates Directive (91/676/EEC) means that all water bodies are subject to a maximum nitrate (NO_3) concentration of 50 mg L^{-1} (equivalent to 11.3 mg NO_3−N L^{-1}) regardless of whether or not they are used for drinking water supply; the Directive explicitly mentioned eutrophication for the first time. The subsequent Water Framework Directive (60/2000/EC) aims to achieve "good ecological status" for fresh and marine waters by 2015 relying, in large part, on existing legislation such as the Nitrates Directive and its designation of Nitrate Vulnerable Zones to help achieve this. These pieces of European legislation have direct parallels in the United States: the Safe Drinking Water Act is the main federal law that ensures the quality of Americans' drinking water, while the Clean Water Act (CWA) is the primary federal law governing water pollution. Despite a major amendment to the CWA, the Water Quality Act of 1987, exemption for agricultural discharges continued, but by the late 1990s, the Environmental Protection Agency (EPA) had changed its focus to eliminating nonpoint source pollution, such as chemicals and nutrients in agricultural runoff. Note that in the U.S. the EPA has established a maximum contaminant level of 10 mg L^{-1} NO_3−N.

In England, limited areas of agricultural land were designated as Nitrate Vulnerable Zones (NVZs) following introduction of the Nitrates Directive. Within NVZs, there is a limit on application rates of manure, a closed season for manure and inorganic fertilizer applications, a minimum manure storage capacity for every farm, and a requirement that inorganic N application be adjusted to crop requirements (avoiding application in areas which are particularly vulnerable to diffuse pollution transfers from land to water, e.g., riparian zones). Restrictions are within the guidelines of Good Agricultural Practice, so farmers must adhere without receiving any subsidies. The area designated as NVZ was expanded in 2002 (covering 55% of England) and again in 2009 (to 70%), with only upland areas, where there is little or no intensive agriculture, excluded (*9*). (NVZs occupy much smaller areas in Wales, Scotland, and Northern Ireland). Nevertheless, despite considerable effort and investment, nitrate concentrations in many rivers have remained stubbornly high (*10*). In groundwater-dominated watersheds, this is due in part to considerable delay in solute transfer from soil through the aquifer to the river (*11*−*15*), preventing immediate detection of any water quality improvement resulting from reduced leaching. However, even in river basins dominated by more rapid (surface and near-surface) runoff pathways, observed decreases in nitrate concentration, where present, have been much slower than hoped for (*10*). This suggests that current policies for controlling nitrate pollution are imperfect because they are predicated upon a relatively rapid response of water quality to land management practices; in fact, it may be decades before a successful reduction in nitrate concentrations can be reported.

Here, we discuss the historical and future nitrate issue in the context of long nitrate concentration time series for a small selection of typical rural UK rivers, allowing comparison to be made between 1970, when postwar agricultural intensification was at its height and 2010, when nitrate source controls had been in place for some years, due to both legislation and voluntary changes in farming practices. These examples complement studies of the River Thames (*16, 17*). The link between agricultural activity and nitrate transfer from land to surface water and groundwater is now well established (*4, 5, 18*). Of the nitrogen which enters inland surface waters in England and Wales, 60% originates from agricultural land while only 32% is contributed from sewage effluent (*19*). These proportions vary in different river basins: nitrate losses from farmland predominate (>90%) in rural watersheds (*18, 20, 21*), which are the focus here.

[†] This manuscript is part of the Environmental Policy: Past, Present, and Future Special Issue.

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10.1021/es101395s

Published on Web 08/03/2010

FIGURE 1. Locations of the five monitoring stations used in this study.

Long-Term Nitrate Responses for Selected UK Watersheds

Long nitrate records for five rural UK rivers are included here (Figures 1 and 2). The basins selected are representative of rural UK river basins with respect to land use history; none contain large centers of population and it can therefore be assumed that changes in nitrate will be predominantly related to changes in land use and land management (*21*). Atmospheric N deposition is relatively low across the UK, well below average fertilizer application rates. All watersheds are intensively farmed lowland basins except for the Tees. Unfortunately, data for fertilizer application rates are only available at the national level but there are only small regional differences across a country as small as the UK (*6*). Annual UK fertilizer input is shown in Figure 3, together with details of land use, and estimates of sewage inputs and atmospheric deposition. These data confirm that fertilizer is the main N input in all cases.

Groundwater is most important in the Stour and Frome, and to an extent at Slapton; the Tees and Leet water are dominated by surface water runoff. In all cases, annual means have been calculated for water years beginning October 1; water years better reflect the annual cycle of river discharge than calendar years. Figure 2 includes annual runoff (% mean flow) for each watershed.

Nitrate regulation in the UK is primarily enforced through spot sampling of nitrate concentration; diffuse pollution problems are then identified by examination of concentration trends. Regulators do not take account of variations in discharge and therefore consider observed rather than adjusted (flow-weighted) concentrations. For this reason, our discussion here is based on annual mean nitrate-nitrogen concentrations (mg NO_3−N L^{-1}) rather than flow-weighted concentrations. Nitrate flux is estimated using total annual flow and annual mean concentration. Mean concentrations are, in most cases, based on daily or weekly samples which ensures that the entire range of discharge is covered.

1. River Stour. The River Stour (Essex) has one of the longest UK nitrate records available: weekly data for 1937−2001; thereafter monthly (*10*). The watershed (578 km^2) is low-lying (maximum altitude: 128 m) and dominated by arable land (76% arable, 13% grassland, 4% urban). The series starts just before World War II with annual mean concentrations around 2 mg NO_3−N L^{-1} (Figure 2). There is a peak in 1941 associated with extensive plowing of grassland to grow food (*22*). The peak in the early 1950s may also be related to WWII plowing with delay relating to longer solute travel times in groundwater. Then, a long upward trend begins in the late 1950s (average rate of increase: 0.22 mg NO_3−N L^{-1} yr^{-1}) peaking in the early 1980s, after which concentrations gradually decline (average rate of decrease: 0.05 mg NO_3−N L^{-1} yr^{-1}). There is a very close association between flow and flux, as would be expected, but concentration also tends to vary with flow; lower concentrations are associated with drier years; higher concentrations in wetter years indicate flushing of excess soil nitrate (*23*).

2. Slapton Wood Stream. Slapton Ley is a eutrophic freshwater coastal lagoon; the lake and its contributing watershed have been monitored continuously since the 1960s; one of the most intensively studied areas is the 1 km^2 Slapton Wood watershed (*24*−*26*). It has land use typical of the area: a mixture of intensive arable and livestock farming. Very few (<10) people live within the watershed. Despite a very different hydrology and a much wetter climate than the Stour, the temporal pattern of nitrate concentrations is very similar, with the same upward trend in the 1970s after which concentrations leveled off. Recently, annual mean concentrations have fallen as arable land has reverted to grassland.

3. River Frome. The River Frome has been the focus of much research over the last 50 years (*15, 20, 27*−*29*). Land use is largely agricultural, with intensive arable farming having replaced traditional sheep farming since World War II. The watershed (414 km^2) is dominated by discharge from the Chalk aquifer, the most significant source of groundwater for public water supply in the UK (*29*). The Frome has seen a significant increase in nitrate concentrations; unlike the other rivers, the trend has been consistently upward since the start of the record. Because of slow percolation through tens of meters of unsaturated Chalk aquifer (1 to 2 m per year), there is a long delay, of the order of several decades, between nitrate loss from the soil and groundwater discharge into the river. Although the trend appears linear, the data are better approximated by sigmoidal breakthrough curves: this suggests concentrations are beginning to stabilize, although it may be after 2020 before a new equilibrium is reached (*28*).

4. Leet Water. The Leet Water (114 km^2) is a lowland tributary of the River Tweed, one of Scotland's largest rivers. Given its northern location, agricultural intensification started later than in southern England, and nitrate concentrations were still low (<2 mg NO_3−N L^{-1}) in the late 1950s. Concentrations started to increase from the early 1960s with a steady upward trend, although peak concentrations were not reached until the mid 1990s. Since then, there has been a clear decline in concentrations. The Leet is today intensively farmed, a mixture of arable and livestock. Intensive agriculture is enabled by a dense network of under-drainage installed in clay soils during the 1960s and this may account for the more immediate impact of nitrate source controls compared to other lowland rivers where deeper groundwater flows dominate (*30*).

5. River Tees. The hydrology of the River Tees (catchment area 818 km^2) is dominated by runoff from its headwaters in the uplands of the Pennine hills where deep peat soils and a cold, wet climate allow only extensive sheep grazing to be undertaken. Only 11% of the watershed supports intensive agriculture, in the lowlands. Nitrate concentrations remain

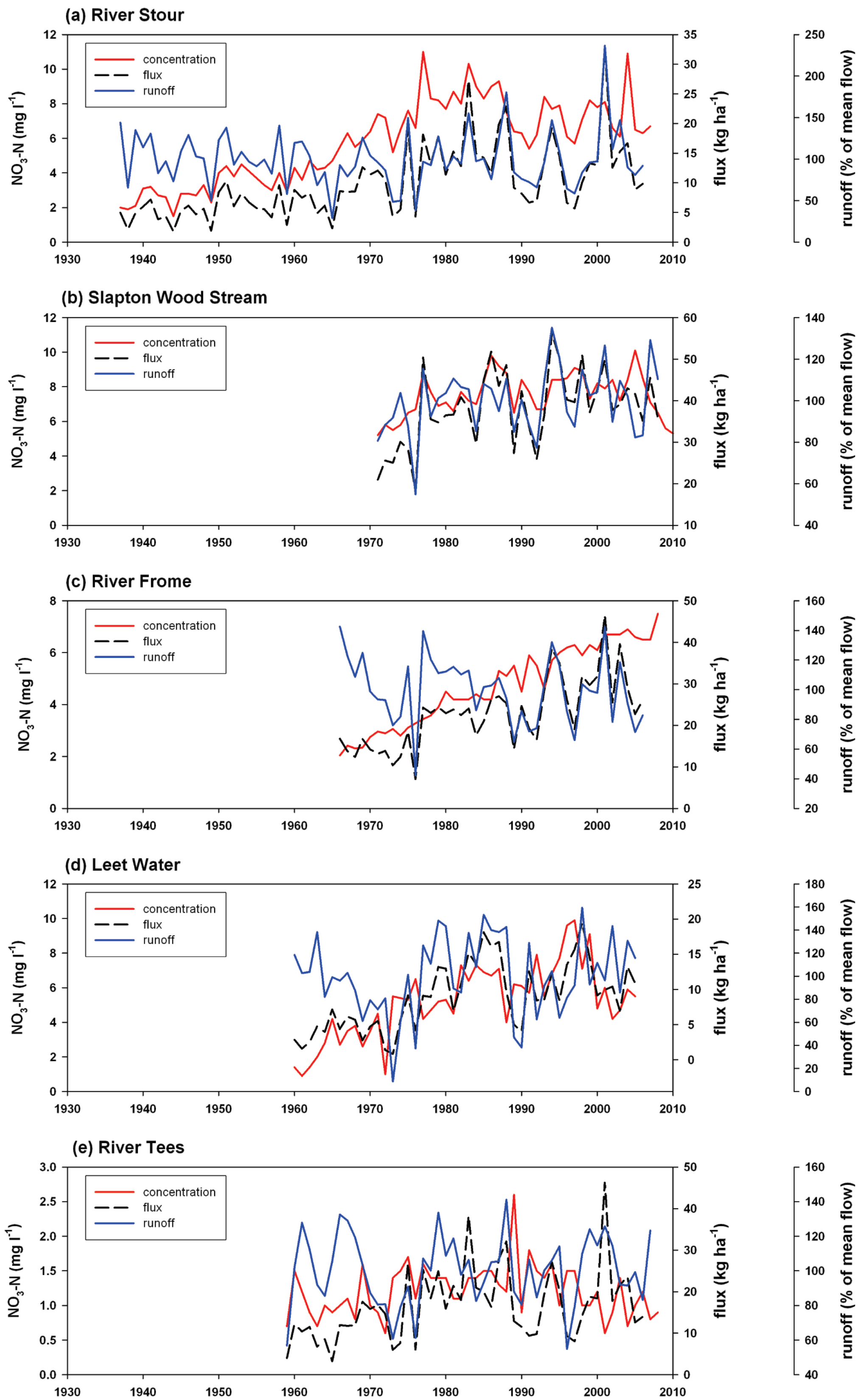

FIGURE 2. Annual mean NO_3−N concentrations, fluxes, and runoff for the five study watersheds.

very low, in stark contrast to the other rivers described here, which are entirely "lowland" in character. However, even the Tees nitrate record does show some increase in concentration during the first half of the time series: concentrations peaked at around 1.5 mg NO_3−N L^{-1} in the mid 1980s; at the beginning and end of the record they are below 1 mg NO_3−N L^{-1}. Although these concentrations are low, their doubling between the 1950s and the 1980s may still have had important ecological impact on the river, which is an essentially oligotrophic system. It is possible that recent declines in nitrate concentration reflect changing agricultural practices but most of the Tees watershed is not designated

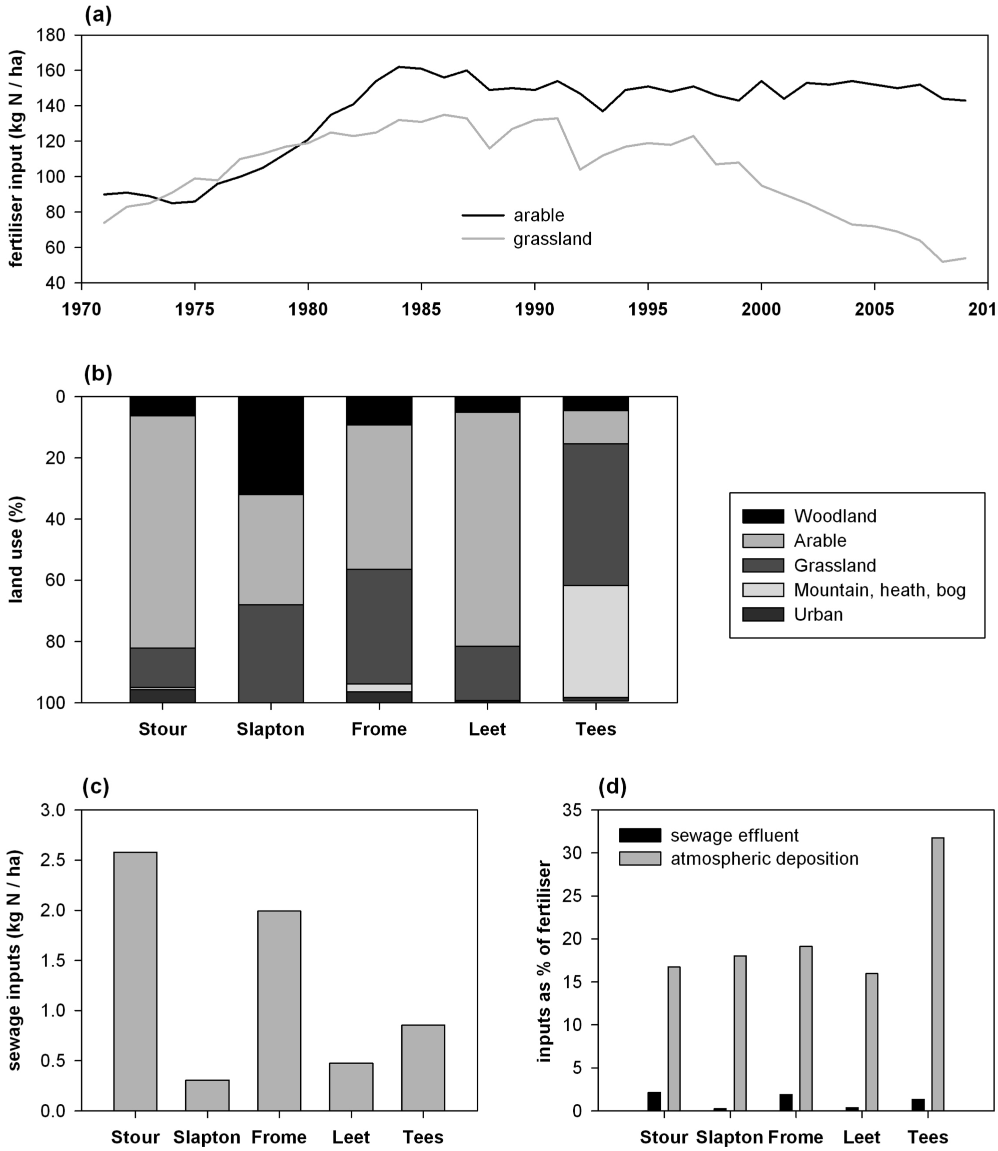

FIGURE 3. (a) Total overall nitrogen fertilizer application rates for England and Wales. Data for Scotland only start in 1983 and so were excluded here. Source: British Survey of Fertiliser Practice (http://www.defra.gov.uk/evidence/statistics/foodfarm/enviro/fertiliserpractice/index.html). (b) Land use as a percentage of watershed area, data taken from the National River Flow Archive (http://www.ceh.ac.uk/data/nrfa/catchment_spatial_information.html). (c) Sewage inputs estimated from watershed population assuming 3.3 kg N per capita per year (*21*). (d) N inputs from sewage and atmospheric deposition (20 kg ha^{-1} yr^{-1}; *21*) as a percentage of fertilizer input.

as a NVZ. Atmospheric deposition can be an important source of nitrogen in upland watersheds (*31*) but total nitrogen deposition in the upland majority of the Tees watershed is less than 20 kg N ha^{-1} yr^{-1} (*32*).

Interpretation: 1970 and 2010 in Context

There are two principal sources of nitrate in surface waters: point inputs (e.g., sewage effluent) and diffuse (nonpoint) source transfers predominantly from agricultural land. Nutrients in sewage effluent can clearly make very important contributions to fluvial loads in heavily urbanized watersheds and have been the primary targets of pollution-control legislation, both in the U.S. (*33*) and in Europe (*6*). However, as nutrient contamination from point sources has been curtailed, there has been increasing emphasis on tackling nonpoint sources from agricultural land.

The well-established link between changes in nitrate concentration in ground and surface waters and changes in land use and land management (*34*) has been used to justify source control measures. However, the processes controlling both the availability of nitrate in the soil and its transport to rivers are complex, so identifying effective management strategies is often not straightforward. Implementation of land use controls can be hampered by the multiowner nature of land and by political resistance to legislative restrictions in farming practice from powerful farming lobbyists, fearful of negative economic impacts. Diffuse pollution is difficult to control and, as noted above, these sources have only recently been the subject of legislation.

There are four major influences on nutrient concentrations in rivers and groundwater: land use (including land management), soil drainage, underlying geology, and depth to groundwater (*33*). The long-term changes revealed in the record for the River Stour show the predominant influence of land use change, moderated via geological controls. As noted above, the initial peaks (1940s and 1950s) were triggered by the widespread plowing during WWII. After the war, the use of nitrogen fertilizer increased dramatically (Figure 3):

in the U.S., its use increased 20-fold between 1945 and 1993 (*31, 35*) while in the UK, there was an order-of-magnitude increase over the same period (*36, 37*). In addition, plowing became more frequent, given the shift to arable cropping, augmenting the availability of soil inorganic nitrogen via enhanced mineralization of soil organic matter accumulated under permanent pasture (*38*). There was also a major program of land drainage in the UK post-WWII (*39*) that enabled the move to arable farming on poorly drained land (e.g., clay vales, floodplains). As well as lowering water tables and increasing soil aeration (which enhances mineralization and nitrification), artificial drainage reduces the residence time of soil water, delivering leached nitrate more quickly to streams. Also, land drainage means that water moving toward local streams and ditches "short-circuits" floodplain soils, reducing the opportunity for landscape mitigation of nitrate fluxes. Floodplain soils have high denitrification potential, being anaerobic with high stocks of organic carbon (*40*).

In groundwater-dominated catchments, the appearance of high nitrate concentrations in river water following land use change is considerably delayed by long travel times in both the unsaturated and saturated zones of the aquifer. Rising trends post-WWII were caused, in part, by enhanced soil nitrogen mineralization after plowing of grassland in WWII (*38*), delayed by its passage through groundwater. This was augmented by increases in nitrogen fertilizer inputs (*36*) and other changes associated with the conversion to arable crops (*10, 37*). Furthermore, even where grassland had not been converted to arable, management intensity increased markedly (*41*). Intensive pastures (cut or grazed) in the UK can often receive as much fertilizer nitrogen as arable land (Figure 3). Although the influence of groundwater is most marked for the Frome, it is also partly explains the upward trend in the Stour.

Prognosis

Is the situation getting better? Nitrate concentrations in many UK rivers now appear to have leveled off (*37*) (Figure 2) although concentrations in groundwater-dominated watersheds, such as the River Frome, are still rising (*15*). In some rivers, concentrations appear to have been declining since the early 1980s (*42*), albeit at a very low rate (e.g., Stour: 0.05 mg NO_3−N L^{-1} y^{-1}), before any nitrate protection zones were introduced (Nitrate Sensitive Area pilot scheme, 1990; *6*). The stabilization of nitrate concentrations in the 1980s is a combination of three effects: first, nitrogen fertilizer use stabilized after the dramatic postwar increases (*36*) (Figure 3) as better advice and rising costs encouraged farmers to stop applying excessive amounts. Second, some of the postwar upward trend was the result of enhanced mineralization caused by large-scale grassland to arable conversion (mainly in the 1940s but also in the 1960s) which generates elevated nitrate leaching for more than two decades after plowing (*38*). Third, surface water concentrations partly reflect delayed transmission of nitrate in groundwater (depending on the type and extent of groundwater contributions in any particular watershed). Thus, the leveling off partly reflects breakthrough of nitrate leached from soils many years earlier, when agricultural intensification was at its height (*15*).

So, what of the contrast between 1970 and 2010? In both the U.S. and UK, farming in 1970 was focused on maximizing yield: use of inorganic nitrogen fertilizer was increasing rapidly, farm machinery was becoming ever more powerful, and land was being drained to bring it into arable production. Off-farm impacts of nutrients, on drinking water and the aquatic environment, were given scant attention. Forty years later, there is now a strong focus on addressing diffuse pollution from agriculture through a variety of measures, both voluntary and statutory. This appears to be having some effect on nitrate concentrations at the watershed scale, although the rate of concentration decline is often disappointingly low. Nevertheless, the intensity and spatial extent of arable and managed grassland agriculture in the UK means that agriculture continues to be the principal source of nitrate. It is, therefore, predicted that the EU Water Framework Directive's requirement for "good ecological status" of all water bodies by 2015 will not be achieved, in large part because of continued diffuse pollution from agriculture (*43*).

Policy Options for Nitrate Reduction

Nitrate concentrations in water are directly related to land use in the upstream watershed (*6, 33*). In most intensive agricultural systems, grassland or arable, fertilizer inputs make up a significant component of the nitrogen balance. This means that limiting the nutrients applied to farmland should improve local water quality, sooner or later. Fertilizer application is becoming more precise (location, timing, amount) as farmers better account for other sources of nitrate in the soil and tailor applications to specific crop needs. This should decrease the amount of excess nitrate in the soil available for leaching to surface or groundwater. In watersheds where there is good connectivity between soil and stream (i.e., those dominated by surface and near-surface runoff (*44*), there should be an immediate response to source control programs. Unfortunately, it may take a long time for measures to have significant effect in groundwater-dominated watersheds: the modal residence time for a Chalk watershed (236 km^2) is around 80 years, implying that it might take 100 years for the system to reach an equilibrium level (*45*). Nonetheless, this must not deter watershed managers from implementing source control schemes, even in those watersheds where rapid adjustments are not expected: benefits will be seen eventually. Further modeling work is required to better define how much improvement can be expected and when. This remains a major issue in the UK given the importance of groundwater for public water supply and the fact that all the major aquifers lie beneath intensively farmed land.

The use of riparian buffer zones to protect river water quality is often criticized for being a form of "end-of-pipe" technology, curative rather than preventive. Nevertheless, the complexity of process interactions controlling nitrate leaching and transport to surface waters necessitates a multifaceted management regime in which "pathway" mitigation measures such as conserving or reinstating buffer zones can augment the benefits provided by source control. Riparian zones act as biogeochemical "hot spots", particularly for nitrogen cycling (*46*). This is because reduced topographic gradients and high upslope contributing area result in frequent near-surface soil saturation, especially in winter. Nitrate-rich water moving from the hillslope to the stream via the riparian zone comes into contact with organic horizons in the floodplain soil, which results in anaerobiosis and the promotion of denitrification (*40, 47*). Denitrification is the main process responsible for the nitrogen buffering capacity of riparian zones and can significantly reduce nitrate concentrations in water moving to the stream, as long as the floodplain is wide enough to provide sufficient residence time for denitrification to operate effectively (*44*). One important element of riparian management schemes may be to fill in open ditches and block under-drainage; this will restore high water tables and encourage denitrification. As noted above, land drainage helped the postwar arable cropping expansion; in reversing the process, this may enable farming to continue throughout most of the watershed, albeit with careful control of fertilizer use, and none in the near-stream zone. It is important to note that riparian protection measures must be implemented at the watershed scale, focused on river channel networks not individual farms. Set against the loss of high agricultural productivity, restoring floodplain functionality would bring benefits across the watershed in addition to water quality improvement: storage of flood waters, enhanced biodiversity, better access for recreation and a more diverse landscape. Locally relevant and targeted agri-environment options can help to balance production and environmental protection and may be able to offer the greatest combined output of ecosystem goods and services (*48*).

Trends in nitrate concentrations are difficult to detect with time series shorter than 20−30 years (*10*). Variations over shorter time scales often reflect short-term climatic variability rather than longer-term responses to policy initiatives or changes in land management practice (Figure 2). Judgements about the success or otherwise of decisions made on the basis of short data sets (even up to a decade) could, therefore, be misleading. For example, policy could be judged "successful" (i.e., nitrate concentrations have fallen) after a few years when all that has happened is a shift from wetter to drier years (less runoff is usually associated with lower concentrations). On the other hand, interventions could be judged "unsuccessful", even if they have had significant positive impact by limiting upward trends at a time when the climate moves from drier to wetter years. Any policy-related changes are bound to be contingent on the precise time at which they are introduced in relation to pre-existing trends or particularly influential hydrological events. Inter-annual fluctuations in river water nitrate concentration are more likely to reflect climatic variation than the impact of any changes in policy or farming practice therefore (*37*). If the future UK climate means greater interannual variability, with wetter winters but more frequent droughts too, then contrast between dry and wet years could become greater, for flow, flux, and concentration.

It follows that, when scientists and government agencies evaluate policy-driven schemes, this must be done within an appropriate historical context (i.e., a sufficiently long record) to distinguish the impact of policy-driven changes from shorter-term variability. In turn, this means that monitoring systems must be adequately maintained over long periods—records of less than thirty years may well be of little use (*10, 49*). No long-term record should be discontinued without proper consideration of its ongoing merits, and protection of a few benchmark sites is essential (*37*). The current environmental agenda is dominated by the climate change question: sustainable watershed management must be appropriate, both for the present and for a future in which mitigation, adaptation, and restoration strategies may be needed as the climate changes.

It is clear that future agricultural policy must be carefully constructed to avoid long-lasting deleterious effects. Tension between the business of farming and efforts to control diffuse pollution is inevitable and may intensify if domestic food security rises up the political agenda in Europe and North America; an integrated approach is certainly needed. We need to avoid situations such as that which happened in Europe in 2008 where reform of agricultural policy abandoned "set aside" (introduced in the 1980s to remove 15% of land from production), counteracting efforts to reduce diffuse-source pollution under the Nitrates' Directive and WFD. Prior experience has shown that widespread land use change can have long-term consequences for water quality and so should be approached very carefully. We now have a much better, although still incomplete, understanding of both the causes and the consequences of diffuse source pollution than we did in 1970. We also have a much better appreciation of the wide range of ecosystem services that can be provided from a watershed. What is needed now is to employ this awareness in "joining up" hitherto fragmented policy themes and funding mechanisms to design and promote new approaches to watershed management that can deliver intended outcomes cost-effectively, and reward land managers for providing the desired range of beneficial services (*48*). Above all, it will be important to combine knowledge from local stakeholders, policy makers, and scientists to anticipate and sustainably manage our watersheds (*50*).

Finally, in relation to the current situation, we can do no better than quote the following (*33*): "No 'quick' fixes' of long-term nutrient excesses should be expected. Ground water moves slowly, and waters of improved quality may take thirty years or more to move from the surface into nearby streams and wells. A long-term view must be taken. Understanding the regional distribution and key scientific factors that affect nutrient concentrations in ground and surface waters is critical to implementing and evaluating cost-effective programs to manage and protect our water resources."

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ES101395S