

Review

Agriculture and Eutrophication: Where Do We Go from Here?

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Received: 25 June 2014; in revised form: 19 August 2014 / Accepted: 21 August 2014 /

Published: 2 September 2014

Abstract: The eutrophication of surface waters has become an endemic global problem. Nutrient loadings from agriculture are a major driver, but it remains very unclear what level of on-farm controls are necessary or can be justified to achieve water quality improvements. In this review article, we use the UK as an example of societies' multiple stressors on water quality to explore the uncertainties and challenges in achieving a sustainable balance between useable water resources, diverse aquatic ecosystems and a viable agriculture. Our analysis shows that nutrient loss from agriculture is a challenging issue if farm productivity and profitability is to be maintained and increased. Legacy stores of nitrogen (N) and phosphorus (P) in catchments may be sufficient to sustain algal blooms and murky waters for decades to come and more innovation is needed to drawdown and recover these nutrients. Agriculture's impact on eutrophication risk may also be overestimated in many catchments, and more accurate accounting of sources, their bioavailabilities and lag times is needed to direct proportioned mitigation efforts more effectively. Best practice farms may still be leaky and incompatible with good water quality in high-risk areas requiring some prioritization of society goals. All sectors of society must clearly use N and P more efficiently to develop long-term sustainable solutions to this complex issue and nutrient reduction strategies should take account of the whole catchment-to-coast continuum. However, the right balance of local interventions

(including additional biophysical controls) will need to be highly site specific and better informed by research that unravels the linkages between sustainable farming practices, patterns of nutrient delivery, biological response and recovery trajectories in different types of waterbodies.

Keywords: eutrophication; algal blooms; agriculture; wastewater; nitrogen; phosphorus; mitigation; society

1. Introduction

Harmful algal blooms and murky waters have become a common sight in many parts of the world [1], aquatic biodiversity is decreasing [2] and hypoxia and dead zones in coastal areas have expanded rapidly in recent decades [3]. A major driver of these endemic societal problems is eutrophication: the enrichment of inland and coastal waters with anthropogenically-driven inputs of nitrogen (N) and phosphorus (P) during the Anthropocene [4–7]. Increased fertilizer use required for agricultural intensification and the inevitable urbanization of a growing population have greatly accelerated the leakage of N and P across the land-water interface and transport to the coastal zone [8,9]. Freshwater, estuarine and marine ecosystems have all been degraded, but it is our freshwaters that are particularly vulnerable because they are so widely exploited. Symptoms of eutrophication vary in different types of waterbodies, but it is the excessive growth of aquatic weeds and phytoplankton (murky waters), blooms of harmful (toxic) algae and effects on fish populations that cause most public concern. The more general costs of eutrophication related to increased water treatment for drinking, reduced value of waterfront properties, loss in amenity value and biodiversity, and adverse impacts on tourism are also very high; UK: £75–114 million [10], USA: \$2.2 billion [11]. There is a clear need to restore and preserve the earth's water resources for future generations, and this is one of the major future challenges facing society. Some argue that the human interference of global N and P cycles has become so acute that we may have already crossed the boundaries of N and P enrichment in surface waters beyond which it will be extremely difficult to reverse [12,13].

It is the increased availability of P which has had the most influence on nuisance algal growth, especially in freshwaters, although it is widely recognized that strategic reductions in the inputs of both nutrients are required [14,15]. Significant ecological gains following reductions in major point source (*i.e.*, wastewater and industrial effluents rich in bioavailable P) discharges have been reported [16–19], but many surface waters remain polluted with non-point nutrient sources and are in poor ecological condition [3,20,21]. This is because the relationships between nutrient use, nutrient delivery, biological response and ecosystem resilience in space and time are highly complex making it difficult to accurately predict recovery trajectories based on nutrient load reduction [22,23]. For example, any remediation may well take long periods of time due to within catchment storage and fractal functioning, while complex feed-back mechanisms and the confounding effects of climate change make it difficult to predict the direction of biological improvement [24–27]. The need for a more holistic approach to improving water quality including biophysical restoration (*e.g.*, riparian management, flow regulation

and food web enhancement [28,29]) is increasingly being recognized, but there is no general recipe for success. These uncertainties present difficulties in developing socially and economically acceptable policies to mitigate the problems of eutrophication within the timeframe required by water regulators and conservation agencies.

Agriculture is now considered to be a major underlying and persistent cause of eutrophication in many catchments around the world [30–33]. However, nutrient loadings from agriculture are not easy to mitigate due to their storm-dependent and diffuse nature, and improvements in the chemical and/or ecological quality of many waterbodies impacted by farming still need to be achieved. This may be because (a) controls over nutrient transfers from agricultural land are not yet strict enough, or have not been implemented for long enough or sufficiently widely; (b) agricultural nutrient loads and/or their ecological relevance are overestimated relative to other sources; and (c) other site and environmental factors are more important than nutrient status in determining ecological status. Agriculture clearly needs to remain a viable, productive and profitable industry and it is important to establish clear evidence of the eutrophication impacts of farming so that sustainable solutions that do not unreasonably affect farm profitability can be found. Measures to reduce nutrient emissions to water may be costly to implement [34], and so it is important to take account of factors that affect their potential effectiveness when implemented [35]. As nutrient inputs to agricultural systems may increase in the future to grow more food and biofuels, and as hydrological patterns may become more extreme under climate change, it will also become increasingly important to identify where water quality and the provisioning of agricultural goods and services are incompatible.

Here we review the issue of farming as a eutrophication source in freshwaters and discuss current uncertainties over what level of mitigation of non-point source nutrient inputs might be required as society strives towards sustainable intensification. We take the UK as our main example because of the multiple pressures from population growth and agriculture on water supplies and amenity resource. The UK is one of the most densely populated regions of Europe (250 capita·km⁻²) and intensive arable and productive grassland occupy 70% of the total utilizable agricultural area [36]. UK freshwaters have consequently become heavily enriched with both N and P [37–39]. Eutrophication control policy in the UK is still developing and provisional targets have been set for annual average P concentrations according to waterbody type [40]. These targets (and associated improvements in ecological status) are expected to be met through source load reductions, but without a clear understanding or evidence base that they are achievable. Following the regulation of wastewater P discharges from large sewage treatment works (STW), nutrient input controls are now being directed at agriculture, largely through the adoption of increasingly coercive on-farm measures [41,42]. Source apportionment estimates still rely on annual nutrient loadings and do not take account of the link between temporal patterns of source load delivery and ecological response, which for rivers is critical [23]. The policy risk is that agriculture is being targeted more than is necessary and wastewater sources not tackled strictly enough, with the overall result that water quality improvements are not achieved.

Our paper raises a number of important issues that need to be taken account of in developing sustainable and achievable policies for eutrophication control. We contend that to have a better chance of success in reducing the growing problem of eutrophication, nutrient reduction strategies must be more accurately apportioned, appropriately targeted and take account of wider societal goals. The balance of necessary controls targeted at farming will vary between catchments and waterbodies;

stricter nutrient reduction controls may be justified in some catchments, whilst trade-offs between environmental and farm productivity objectives will need to be carefully considered in others [43]. In some catchments, more stringent controls on wastewater discharges will be more likely to deliver the required ecological improvements than the current emphasis on agriculture. Ultimately, aquatic eutrophication is a societal problem which cannot be resolved easily or quickly, and will need long-term sustainable solutions involving all stakeholders.

2. Nutrient Legacies

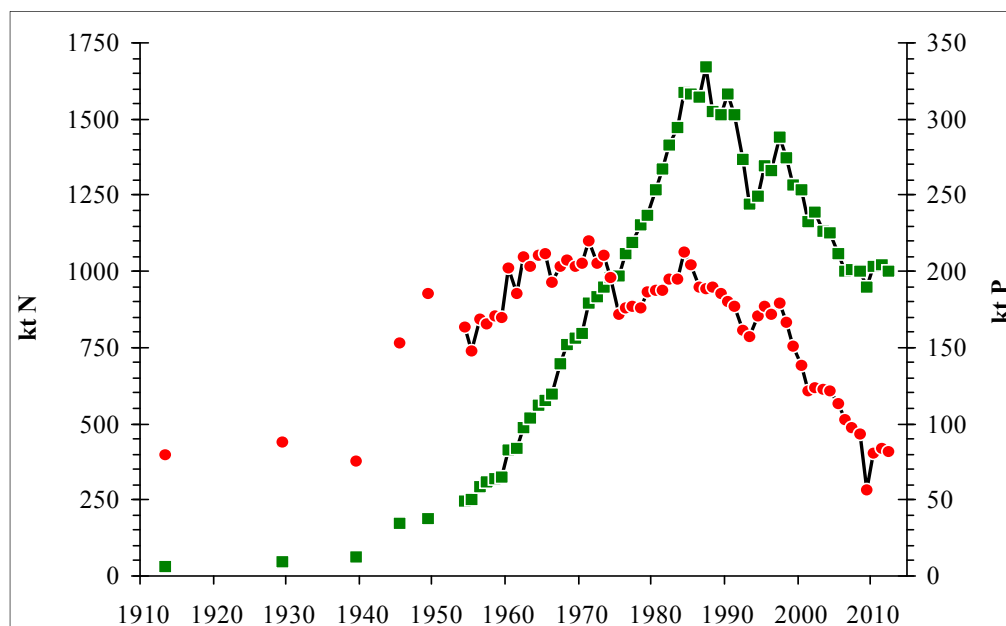
Agriculture's contribution to eutrophication relates not only to current farming practices, but to the legacies of previous nutrient inputs and adoption of management regimes that were streamlined for production goals and not environmental protection. Highly significant loads and concentrations of N and P can be washed directly into surface waters when runoff occurs shortly after the application of fertilizers and manures, or during livestock grazing [44,45], but only occur on the relatively few occasions when these sources are applied or present. Of equal relevance for eutrophication is the N and P that has accumulated in groundwaters, soils and sediments from past applications of fertilizers and manures to land [32,46–48]. For example, in the UK, farmers were actively encouraged by post-war government policies to intensify and apply greater amounts of relatively cheap fertilizers and feeds for over 50 years supported directly by free on-farm advice and subsidy payments based upon a national need for more food (Table 1). Fertilizer use consequently increased rapidly after the war, especially for N (Figure 1), but the utilization of N and P in fertilizers and feeds in agriculture is inherently inefficient, not least because farmers tend to oversupply nutrients to their crops and livestock to insure against uncertainties in growth rates and soil nutrient supply.

Table 1. Timeline of government support for UK farmers over the last 50 years. Adapted from Garforth [49].

Year	Intervention
1946	National Agricultural Advisory Service (NAAS) set up to give free technical advice to farmers to boost agricultural production
1957	Treaty of Rome established the principle of the Common Market (CM) to safeguard European food security
1962	A Common Agricultural Policy (CAP) implemented across member states to provide commodity price support (import quotas and levies, intervention prices)
1971	NAAS widens the free services offered to farmers and becomes the Agricultural Development and Advisory Service (ADAS)
1984	Dairy quotas introduced to help limit over-production of milk
	ADAS started to charge farmers for advice eventually leading to full privatization in 1997—the era of free advice was over.
1986	The first agri-environment scheme involving long-term voluntary agreements with farmers to adopt practices that would help protect environmentally-sensitive areas was introduced and funded by the taxpayer
1992–1993	MacSharry reforms designed to limit over-production led to a switch from commodity-based support to direct farmer support (arable area and livestock headage payments) and set-aside was introduced

Table 1. *Cont.*

Year	Intervention
1991	Countryside Stewardship Scheme (CSS) involving long-term subsidised agreements with farmers was introduced to protect valuable habitats paid by the taxpayer
2000	Reform of CAP under AGENDA 2000 led to two pillars of support: farmers and rural development
2003–2005	Further reform of CAP to encourage resource protection led to a system of Single Income Payments (SIP) to farmers based on farmed area and Cross-compliance measures which gradually became more ecosystem service oriented. Set-aside was abolished.
2009	New countryside stewardship scheme introduced to help protect the rural environment and comply with requirements of the Water Framework Directive (WFD)- Entry Level Scheme (ELS) and Higher Level Scheme (HLS)
2014	Further reforms of CAP in preparation

Figure 1. Trends in N (green squares) and P (red circles) fertilizer use in the UK over the last 100 years (data before 1955 are sparse; adapted from Johnston and Dawson [50]).

For example, a crop recovers no more than approximately 60% of fertilizer N and 30% of fertilizer P in the year of application under UK conditions, and often it is much less [51]. The recycling of feed and crop nutrients in livestock manures, and other urban bioresources, applied to land has also contributed additional amounts of reactive N and P to soils that are poorly utilized [52]. Any unused nutrient is either lost to the atmosphere (NH_3 and NO_x) and/or temporarily (N), or more permanently (P), immobilized as surplus nutrient in the soil. These surplus nutrients are transported to groundwater in percolating leachate (nitrate) and to surface waters in runoff (ammonium N, dissolved and particulate P) generated by rainfall events [53,54]. Soil-accumulated “legacy” nutrients therefore provide a ubiquitous source of background nutrient loss to water from farmed land every time land runoff is generated. The widespread installation of subsidized field drainage in the UK during the 1960s,

70s and 80s has further exacerbated rates of nutrient leakage by creating a more rapid connectivity between the field and the waterbody [54,55].

The past over-use of inorganic fertilizers and feed supplements associated with the intensification of animal and crop production has therefore left a legacy of increased background leakage of N and P into inland waters, which has taken a generation to become evident. Nitrogen is still continuing to increase in some lowland aquifer regions due to fertilizer N inputs from over 50 years ago [47], and groundwater stores of nitrate provide a long-term source of readily-available N to many waterbodies during the ecologically-active summer period when land runoff is less frequent [17,25,56]. Together with industrial N emissions, agriculture has also contributed to increased background deposition of atmospheric N across the land-water interface, especially for larger waterbodies and the ocean [57]. The legacy P stored in soils from past fertilizer and manure inputs also represents a large and potentially long-term P source for eutrophication in standing waters with long residence times [30,48,58]. It will take many decades to drawdown these legacy N (groundwater) and P (soil) stores, even if N and P fertilizers were no longer applied. In one UK catchment, Howden *et al.* [59] predicted that any reductions in fertilizer N use implemented now are unlikely to impact on river nitrate levels for at least three decades due to long travel times to the aquifer. Similarly, at current average levels of P removal by crops of about $20 \text{ kg} \cdot \text{P} \cdot \text{ha}^{-1}$, it could take up to 50 years to drawdown the surplus (legacy) P that has accumulated in UK soils since the war [60].

Although introduced in the best interests of society, the cumulative environmental impacts of post-war policies to intensify farming are only now being fully realized and have surfaced too late to be able to reverse quickly. Farmers now face the legislative burden and associated costs of redressing the environmental damage caused by legacy nutrients, a generational injustice. It is clear that farmers must implement measures that not only reduce nutrient losses from current fertilizer and manure N and P inputs, but also address the mobilization and delivery of legacy soil N and P in leachate, runoff and erosion. In particular, the stores of legacy P are a resource that could be recovered to reduce reliance on inorganic fertilizers and the environmental damage they cause [48,58,61]. However, what is unclear is the extent to which today's eutrophication problems are due more to nutrient losses from current farming activities, or to legacy nutrient effects from past farming activities. This is of fundamental importance for policy development and expectations regarding the impact of current nutrient reduction strategies, and will clearly vary according to waterbody type and between catchments. Legacy nutrient stores and their impact on waterbody recovery trajectories therefore need to be better quantified to inform nutrient load reduction strategies.

3. Uncertainties in Ecological Outcomes from Non-Point Source Controls

The rationale for reducing nutrient inputs from agriculture to aquatic ecosystems assumes that there is a direct relationship between non-point N and P loadings, increases in waterbody nutrient concentrations and deterioration in ecological status. In reality, this linkage is very uncertain [23], and difficult to demonstrate [62,63], especially where agriculture is not the major contributor to nutrient loadings, and/or where the precise source land areas, or farming practices, responsible for the nutrient loss have not been accurately identified. Lotic waters also differ from lentic waters in their response to nutrient loadings due to large differences in residence times, and the relative importance of the

numerous site factors that influence algal growth in all waterbodies [64–67]. These uncertainties in outcome are discussed in more detail below with particular reference to the UK.

3.1. Accurate Source Apportionment

Farming practices have an important potential influence on N and P loadings and concentrations in freshwaters because the majority of water discharge originates in headwater areas adjacent agricultural land [68]. Farming also contributes significantly to the nutrient loadings transported by rivers to downstream standing waters and to the coastal zone causing hypoxia [6]. However, agriculture is only one of a number of sources contributing nutrients for algal growth [69]. It is often difficult to distinguish a direct link between agriculture and ecological impacts in waterbodies receiving nutrient inputs from multiple sources, and where the eutrophication problems occur downstream [70–72]. For example, in most UK lowland river catchments, groundwater and storm-driven inputs of N and P from agriculture are supplemented by direct and near continuous discharges of household and industrial wastewater from a large number of STW and in storm-driven urban runoff [73–75]. Whilst agriculture is undoubtedly the major source of N, wastewater continues to be a major source of P (the main limiting nutrient for nuisance algal growth), although relative contributions will clearly differ between regions and catchments [76]. In lowland areas with low rainfall, effluent discharges from sewage treatment works (STW) during summer periods have a major impact on streamwater quality and ecology, because STW contribute maximal bioavailable P concentrations at times when baseflow dilution is low, residence times are long and temperatures are high, providing the ideal conditions for maximum in-stream P retention and rapid biological growth [67,77]. In many lakes, wastewater nutrient inputs have been the main drivers of eutrophication problems [78], and agricultural contributions have only become apparent as wastewater sources have been reduced [79,80].

In the U.S., UK and Europe, there is a stronger correlation between river P concentrations and urban population densities than with land use [9,81,82], and the number and distribution of STW in river networks therefore have a large influence on bioavailable P supply in the water column during periods of active algal growth [18,70]. Even in rural areas, the ecological impacts of discharges from septic tank systems (STS) may be much greater than previously thought because their location, condition and effectiveness remain largely unknown [83]. Correspondingly, it is not simply agriculture that discharges P to surface waters during high flow events as storms will also generate overflow from STW, combined sewer overflows (CSOs) and STS and mobilize stores of wastewater retained in the landscape from previous effluent discharges. For example, high-flow remobilization of wastewater-derived P, which had previously been retained within the river channel during low flows, accounted for between 20% and 50% of the annual average P loads measured in a mixed land-use U.S. watershed [84]. The common belief that in all cases high-flow P loads are derived from agricultural sources is simply unfounded and may lead to an overestimation of the real agricultural contribution to catchment P loads and an underestimation of the impact of wastewater on river P dynamics and fluxes to lakes, reservoirs and the coastal zone.

For UK freshwaters, this overestimation may be highly significant because of the large number of STW discharging concentrated effluent into UK surface waters [76]. There are a total of 9000 STW in the UK, but only about 1900 (21%) are large enough (serving >2000 p.e.) to warrant secondary or

tertiary treatment to lower organic and P loadings (Table 2). Agglomerations above 2000 p.e. must receive secondary treatment but with no limit on P discharge, whilst agglomerations of >10,000 p.e. in areas sensitive to eutrophication must receive tertiary treatment to lower discharge P concentrations to <1 mg/L as orthophosphate. There are about 450 STW in the UK receiving tertiary treatment. Overall, 98% of STW are compliant with these regulations, but a large proportion of small and medium-sized STW (with or without secondary treatment) are still discharging high concentrations (up to 10 mg·P·L⁻¹) of highly bioavailable P into rural headwaters and rivers [85]). There are also a large number of pumping stations that discharge raw effluent into nearby watercourses to overcome issues of overflow at STW during storm events. The high P concentrations discharged, relative to dilution within the receiving waters, are not only a key determinant of eutrophication risk in rivers [66], but also become adsorbed onto eroded agricultural sediments leading to overestimation of diffuse P loadings when these sediments are remobilised during storm events.

Table 2. Numbers of sewage treatment works serving different population agglomerations in the UK in normal areas and areas that have been classified as sensitive to eutrophication (from Defra [86]).

Agglomerations p.e.	Normal Areas ¹		Sensitive Areas ¹		Total	Percent of Total
	Freshwaters and Estuaries	Coastal Waters	Freshwaters and Estuaries	Coastal Waters		
2000–10,000	422	26	594	1	1043	56
10,001–15,000	65	16	110	0	191	10
15,001–150,000	190	65	302	5	562	30
>150,000	33	10	36	2	81	4
Total	710	117	1042	8	1877	

p.e.—person equivalent; ¹ A waterbody is classified as sensitive if it is (a) eutrophic or could become so in the near future without tertiary protection; (b) an abstraction source that has or could have high nitrate levels without tertiary protection and (c) requires or could require tertiary protection under other EU Directives (e.g., bathing quality). A normal waterbody is one which is not classified as sensitive [86].

There are three important arising implications:

- (1) In densely populated regions, wastewater sources can have a much more dominant role in P cycling within river networks than has hitherto been appreciated and agriculture's contribution to river and lake eutrophication needs to be re-evaluated within this context.
- (2) More sophisticated source apportionment models are necessary to fully and more accurately apportion the relative nutrient contributions from wastewater, agriculture and urban sources in catchments with multiple pressures to direct proportioned mitigation efforts more effectively.
- (3) Where the inputs of wastewater sources have been underestimated, efforts to mitigate freshwater eutrophication through control of non-point P inputs from agriculture are likely to have far less impact than is currently predicted by nutrient export models calibrated and validated by catchment P flux data that do not take account of remobilized point source inputs.

3.2. Waterbody Characteristics

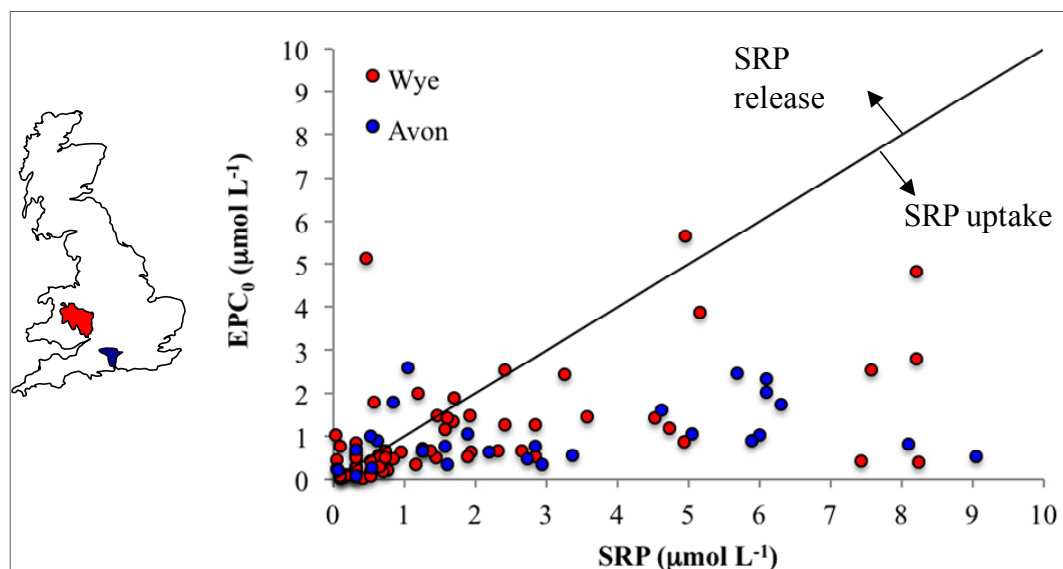
Where agriculture does dominate the nutrient loading in catchments, annual loading is not necessarily synonymous with ecological impact in rivers, especially for P. Unlike nitrate, which is unreactive and highly mobile in soils, much of the (legacy) P discharged to surface waters from cultivated land during high flow events is in a particulate and less bioavailable form [87–89]. Even in grassland regions, the higher rates of soluble P exported to water may be adsorbed to river bank or river bed sediments and/or diluted during high flow periods reducing any direct ecological impact of this fraction for much of the year. For rivers with low water residence times, the predominance of agriculturally-derived P fluxes during high flows in autumn and winter, P-sediment interactions and sediment dynamics (deposition/remobilization) within the river channel provide an important ecological disconnect to the requirement by algae for bioavailable dissolved P concentrations in the water column during low flows in spring and summer [67,77]. This is in sharp contrast to lakes and reservoirs, where higher water and sediment residence times mean that soluble and sediment-bound P inputs are more likely to remain a direct, or seasonally-recycled source of available P for algal growth.

Agriculturally-derived inputs of P may therefore have much less of an impact on algal growth in rivers than might be suggested from a consideration of the relative contribution agriculture makes to annual P loadings in catchments. Agriculture's contribution to river eutrophication also depends on whether P is released from channel sediments during low flow periods, which are the times of greatest ecological sensitivity. Research from UK rivers on P exchange between bed sediments and the overlying water column suggests that, provided the sediment-water interface remains oxic, bed sediments can continue to act as a P sink during low-flow conditions, even when impacted by sewage effluent. Data for two large catchments representative of major farming landscapes of the UK are shown in Figure 2. Situations where there is potential for P release from the sediment tend to occur (a) when the sediment is already saturated with P and is reaching equilibrium with the overlying water-column [90]; (b) after a reduction in dissolved P concentrations in the overlying water column (e.g., after point source reductions) and before the sediment has been flushed downstream by subsequent storm events [91,92]; (c) a high sediment organic C content (e.g., from livestock or septic tanks) promoting migration of the redox boundary to, or above, the sediment-water interface and reductive dissolution of the Fe-oxyhydroxides in the surface sediments [91,93]; and (d) mechanical disturbance of the "oxidised cap" of surface sediment, which may release P-rich pore-waters from subsurface anoxic sediment into the overlying river water [94].

The lack of tangible ecological benefits from nutrient reduction strategies, whether they are targeted at point sources or agriculture, may also be related to the large number of site-specific environmental variables (e.g., flow, temperature, shade), aquatic processes (e.g., abiotic and biotic partitioning), food web interactions (e.g., grazing communities) and hydrological processes operating in different types of surface water that govern ecological response to nutrient inputs and subsequent recovery trajectories [22,23,64,95]. For example, there are multiple anthropogenic physical pressures that have occurred over similar timescales to nutrient enrichment, such as drainage of riparian wetlands, river channel modifications and water abstraction, which have had important impacts on channel morphology and hydroecology [96,97]. Algal growth is consequently limited by factors other than nutrients and untangling the relative importance of these multiple contributing factors is a major

barrier to developing successful policies for eutrophication control. For example, Bowes *et al.* [95] found that shading had far more impact on algal growth in the River Thames in England than reducing in-stream P concentrations. These uncertainties have led to more holistic approaches to eutrophication control in the hope that combined actions will have more chance of success [31].

Figure 2. The relationship between the equilibrium P concentration at zero P sorption (EPC_0) of river bed sediments and the concentration of soluble reactive P (SRP) in the overlying water column for lowland tributaries of the Avon (1715 km²) and Wye (4017 km²) catchments with arable, livestock and mixed land use (adapted from Jarvie *et al.* [98] and Palmer-Felgate *et al.* [99]). A 1:1 line shows where sediments and river water would be in equilibrium, *i.e.*, no net uptake or release of SRP. The majority of points lie below this line indicating potential for SRP uptake by bed sediments. The points lying above the equilibrium line indicate potential for SRP release by sediments, but mostly at low SRP concentrations ($<1 \mu\text{mol}\cdot\text{L}^{-1}$), which are typically close to limiting concentrations for algal growth in UK lowland rivers [95].



There are three important arising implications:

- (1) Nutrient control strategies must take account of waterbody characteristics (waterbody type and site characteristics) since these characteristics will have a large influence on ecological state and therefore on recovery trajectories.
- (2) Since catchments cross diverse landscapes, may contain more than one waterbody type and will extend to the coastal zone, catchment-based management plans to reduce nutrient loadings should take account of the location of eutrophication problems and which types of waterbodies are affected.
- (3) While nutrient controls over agriculture may be justified to reduce downstream eutrophication from an accumulating nutrient pool, the ecological response of rivers to non-point P controls may be less than expected because of the limited bioavailability and accessibility of the P delivered.

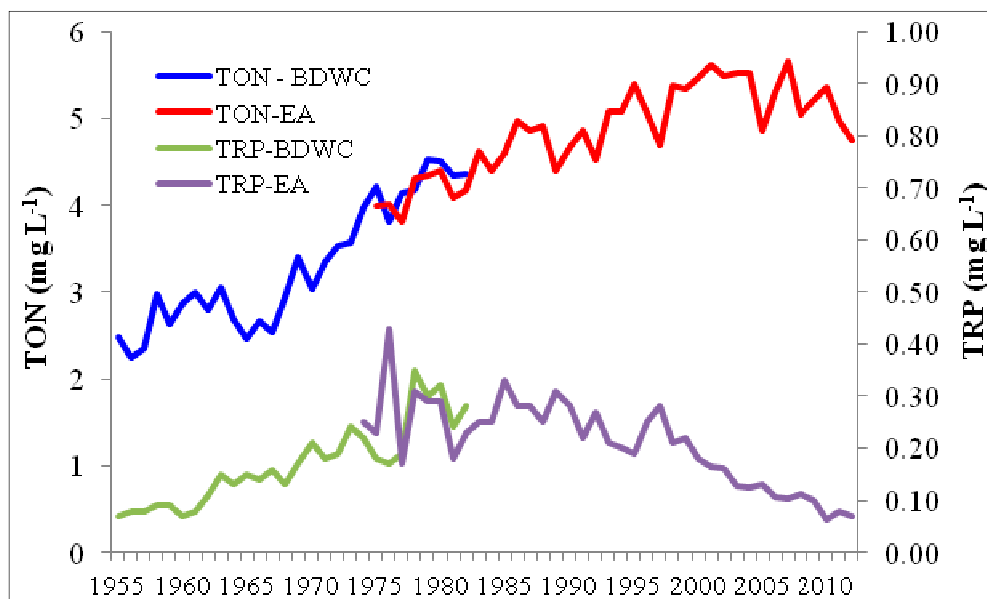
3.3. Mitigating Nutrient Pressures from Farming

There are many examples of where agriculture has had a direct impact on the quality and ecological status of rivers [100,101], lakes [27,102], estuaries [24] and coastal systems [103]. A wide range of on-farm measures are now being recommended to reduce N and P loadings from agriculture to water in order to make potable water safe and to restore or maintain good or high ecological status. These measures include legislative, voluntary and economic levers that are delivered via national regulations and codes of practice, or stakeholder-led catchment-based approaches [42]. They range from placing limits on the amount, timing and methods of nutrient application to land through to containing runoff and nutrient delivery [104]. In the UK, there is currently far less regulation for P than for N, and catchment management initiatives are necessarily holistic to cater for unforeseen and multiple outcomes and allow control over a wider range of pollutants than just nutrients. However, subsidy payments to farmers in the EU are now subject to increasing levels of compliance with adoption of general measures to protect valuable habitats, reduce the risk of environmental pollution and degradation of the rural countryside. In Northern Ireland, a national P surplus target of 10 kg-P-ha^{-1} applies to a small number of derogated farms under the EU Nitrates Directive regulations. There are also additional direct payments to buy environmental services from farmers through agri-environment schemes and countryside stewardship to help meet the requirements of the Water Framework Directive (WFD), (e.g., Table 1 for the UK). However, there is little evidence so far that this legislative burden and/or suite of stewardship measures have reduced farming intensity or improved water quality [24,35,72,105–107], despite model projections. This contrasts markedly with the success of point source controls; as illustrated in Figure 3 the progressive reduction in river P concentrations in the River Avon, southern England due to point source controls contrasts with the more modest and delayed slow-down of river N concentrations in response to the reductions in N fertilizer use since 1988 (see Figure 1).

Mitigation options targeted at agriculture are unlikely to be (cost)-effective if the precise source land areas, or farming practices, responsible for the nutrient loss have not been accurately identified, or if they are not implemented successfully, or sufficiently widely, over the waterbody catchment area. The dynamic and unpredictable nature of non-point nutrient export in catchments makes this source tracking a very difficult task. The success of non-point source measures also relies heavily on farmer engagement and skill, and needs to be tailored to suit specific site requirements, which will vary from farm to farm [108]. Soils, fresh application of fertilizers/manures and farmyards are all potential sources of nutrients that will deliver variable N and P loads depending on the type of farming system, soil type and site hydrology [63]. Transfers of legacy nutrients will dilute the beneficial impacts of controls over current activities, and strategies to reduce legacy nutrient inputs will clearly not bring immediate benefits [35,48]. Controlling nutrient loads from agriculture therefore depend not only on how much the inputs can be reduced, but how those inputs are managed on the farm and how cultivation and cropping practices can be adapted to reduce the mobilization and transport of legacy soil nutrients through runoff and erosion [32,109,110]. Critical source areas and delivery pathways of P transfer on farms are numerous, dynamic and complex, and will clearly differ between landscapes with permeable and impermeable soils, and only their accurate identification will provide a sound basis

for the implementation of effective options to mitigate P transport [111–113]. End-of-pipe (retention) solutions are arguably not a sustainable method of reducing eutrophication risk [58].

Figure 3. Trends in the concentrations of total oxidized N (TON) and total reactive P (TRP) in the River Avon, Knapp Mill, England over the last 60 years. Since 2001, tertiary treatment to remove P from the effluent has been installed at 18 STW in the Avon catchment. (Data courtesy of the Bournemouth District Water Company ((BDWC) and the Environment Agency (EA) after Mainstone *et al.* [41]).



There are three important arising implications:

- (1) It is extremely difficult to accurately quantify the degree of change in agricultural practices needed to achieve the reductions in N and P loadings necessary for eutrophication control. Nutrient reduction measures targeted at farming may therefore be less effective, or take much longer, than expected.
- (2) The dynamic and unpredictable nature of nutrient loss from agriculture (e.g., extreme events) makes it very difficult to implement fully effective mitigation actions. At best, accurately targeted measures to reduce runoff, soil erosion and direct losses from fertilizer and manure sources will reduce land vulnerability and frequency of high loss events, but some loss is inevitable.
- (3) Strategic reductions in inorganic N and P inputs to farming systems are essential for drawing down legacy N and P stores for long-term gains. Farmers and the agricultural industry must embrace the concept of sustainable intensification by improving N and P use efficiency on the farm.

4. Wider Societal Goals

The socio-economic impacts of reducing nutrient loads from agriculture to the levels required to achieve statutory limits (N) and/or good ecological status (P), may be neither sustainable nor

acceptable in many sectors of society. In some lowland areas of the UK where annual rainfall (and therefore the capacity to dilute nutrient inputs) is low (ca. 500 mm), it has been estimated that about 50% of the arable land area would need to be taken out of production to achieve the statutory limit of nitrate in drinking water of $11.3 \text{ mg}\cdot\text{N}\cdot\text{L}^{-1}$ required by the EU Nitrates Directive [114]. Even greater areas of agricultural land would need to be taken out of production to achieve the target concentrations of $1\text{--}2 \text{ mg}\cdot\text{N}\cdot\text{L}^{-1}$ considered necessary to avoid deterioration in lake macrophyte communities [16,31], or reduce benthic and sestonic algae in rivers [115,116]. These levels of intervention would clearly cause major socio-economic issues for rural communities and regional agricultural output. Similarly, runoff P concentrations from both arable and grassland farming systems adopting best agricultural practice and operating within recommended ranges of soil P fertility can exceed the P target concentration of $30\text{--}35 \text{ }\mu\text{g TP or SRP L}^{-1}$ required for control of algal growth in standing and flowing waters [115,116] by some considerable margin. Two examples representative of large areas of arable and grassland soils in the UK are given in Figure 4. Even the somewhat higher target SRP concentrations ($40\text{--}110 \text{ }\mu\text{g}\cdot\text{L}^{-1}$) set for UK waters are very challenging in relation to the degree of P enrichment in many anthropogenically-impacted rivers. Such challenging concentration targets are very difficult to meet if we are to maintain a viable and profitable agricultural industry.

Figure 4. Instantaneous concentrations of soluble reactive P (SRP) measured in drainflow from (a) the Foxbridge drain at Rosemaund from 1997–2000. The drain catchment area was farmed according to best practice with recommended soil P fertility levels [88]. The average flow-weighted concentrations ($\text{mg}\cdot\text{L}^{-1}$) of SRP and total P (TP) over the monitoring period are also shown; (b) hydrologically isolated 0.2 ha plots with optimum soil P fertility in Northern Ireland grazed by cattle but receiving no fertilizer P from 2001–2005 [117]. The average flow-weighted concentrations of SRP and TP in both drainflow and surface runoff over the monitoring period at this site are also shown.

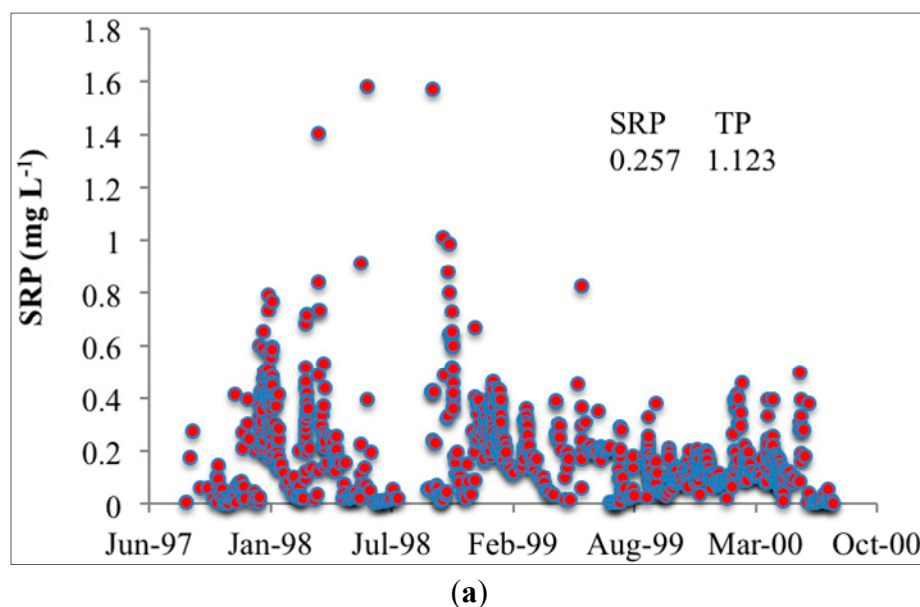
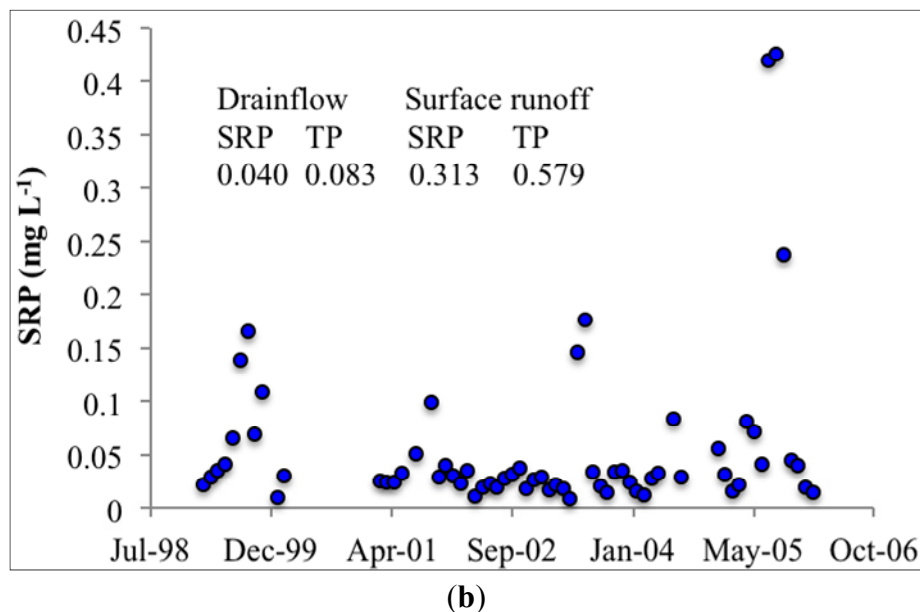


Figure 4. Cont.



Doody *et al.* [102] detailed the impact of extensive farming on water quality in the Lough Melvin catchment in Northern Ireland, where TP concentrations in the lake increased from $19 \mu\text{g}\cdot\text{L}^{-1}$ to $30 \mu\text{g}\cdot\text{L}^{-1}$ over a decade. This was despite stocking rates delivering N loading rates well below the current EU Nitrates Directive limit of 170 kg N ha^{-1} throughout this period. This paradox highlights that balancing agriculture intensification and water quality may be unrealistic in catchments with impermeable soils, where flashier hydrology overrides the impact of low nutrient inputs on farms [118]. If the majority of agricultural land in a catchment is exporting higher background N and P concentrations than the water quality targets to control eutrophication, the degree of interventions required may be substantial and either very costly, unachievable or unacceptable to society. Achieving the ambitious growth targets for agriculture embodied in the need for sustainable intensification may not be possible in all catchments if the primary goal is the protection of our water resources; for example, within the constraints of the EU Water Framework Directive. The current national focus on restoring all water bodies to good ecological status may need to be reconsidered if the balance between agriculture and water quality cannot be achieved.

5. Conclusions

Algal blooms and murky waters are a societal problem that has arisen due to a number of simultaneous evolutionary and policy-driven anthropogenic interventions, and it is not easy to disentangle the relative contribution of these pressures to inform mitigation strategies. Long-term legacy storage of N and P in our landscape, complex patterns of nutrient delivery and in-stream processing in catchments suggests that these problems will persist for decades at least [35,47,48]. It has taken a generation to acknowledge the environmental impacts of agricultural intensification and urbanization and it will likely take another generation to resolve these impacts. The mitigation efforts that must be directed at agriculture to achieve lasting improvements in water quality and ecological status for the next generation remain very unclear.

Our analysis here suggests that these mitigation efforts should (a) be more accurately apportioned in relation to other sources; (b) take account of the ecological relevance of agricultural discharges; (c) take account of legacy nutrient stores and likely recovery trajectories and (d) consider the impacts on agricultural productivity. This requires the development of more sophisticated catchment-based apportionment tools and indicators for identifying which ecologically-relevant sources to tackle first [23], and likely time-lags in waterbody response [59]. A framework is needed for prioritizing those waterbodies that can recover quickly from the implementation of nutrient reduction measures combined with other restoration approaches [43]. The rationale for the prioritization of catchments for protection of current ecological status is supported by uncertainty in the recovery trajectories of impacted catchments, the incompatibility between agriculture and water quality in some catchments and the potential long-term impact of legacy N and P on water quality. In catchments that are currently below good ecological status, the high economic and social costs of tighter nutrient limits on agriculture may make achieving good ecological status prohibitive, and prioritizing agricultural production over water quality protection may be a more realistic option. In other catchments, the effluent discharges from STW and unmanaged STS will be the more urgent problem to tackle, requiring more stringent controls over wastewater inputs in both urban and rural environments.

Further population growth and urbanization, demographic redistributions, an increased demand for food and biofuel on the same land area (*i.e.*, greater fertilizer use), limited water resources and climate change are going to place even greater anthropogenic pressure on our future water security. Concerted and comprehensive actions are therefore urgently needed to protect our water resources for future generations, and these actions must encompass a societal response involving all stakeholders: the general public, the water industries, agricultural communities and rural and urban planners. Since fertilizers are the primary source of reactive N and P stored, circulating and leaking in our environment, there is a clear strategic need to improve the efficiency of fertilizer N and P use, drawdown the legacy nutrient reserves of N and P in soils, sediments and groundwater and more effectively recover nutrients from urban areas where they become concentrated [61,119–121]. These goals are supported by other policy drivers related to mitigation of the effects of N fertilizer manufacture on climate change and the potential future scarcity of rock phosphate [122,123]. Such general strategies should be supported by site-specific sustainable practices on the farm, but these will necessarily vary between waterbodies according to agriculture's contribution to the eutrophication problem. More science is needed to clarify this contribution on a catchment specific basis and accurately assess recovery rates and biological impacts (and indicators *e.g.*, [124]) at a time of greater climate instability. A greater reliance on evidence-based policies will ensure that the multiple objectives of clean water, diverse aquatic communities and sustainable intensification of agriculture are achievable for the benefit of future generations.

Acknowledgments

We are grateful for the assistance of Chris Dawson and Associates for providing historic data on fertilizer use, Rachael Dils and Iain Harris of the Environment Agency for supplying river nutrient data at Knapp Mill.

Author Contributions

Withers and Neal both conceived the need for this perspectives paper and equally contributed text. Withers contributed data for Figure 3b, co-ordinated and finalized the paper. Jarvie contributed comment, text and data for Figure 2; Doody contributed comment, text and data for Figure 3a.

Conflicts of Interest

The authors declare no conflict of interest.

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