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The impact of elevated phosphorus inputs on flora and fauna in riverine systems

Literature Review

I. Introduction

Aquatic plants need a cocktail of chemical elements to grow. Many, like oxygen, hydrogen and carbon, are easily attainable. Some however are not so abundant and may limit the growth or development of organisms. These potentially limiting substances are known as micronutrients or, more simply, nutrients. Two such micronutrients are: nitrogen and phosphorus. As a generalisation, the growth of higher plants and algae tends to be limited by phosphorus levels in fresh waters, whereas nitrogen is most commonly the limiting nutrient in marine and estuarine waters. Widely varying nutrient concentrations are found in aquatic systems dependent upon catchment geology, location within catchment and a variety of other factors (Smith *et al*. 2003). The combined natural 'external load' of P and N reaching watercourses originate from the air, the water table, as well as upstream (Smith *et al*. 1999).

The two main routes by which nutrients enter rivers are in dissolved form either as point source emissions, from concentrated outlets typically effluent pipes, or diffuse sources, where dispersed pollution enters rivers from groundwater, overland and through flow (Nijboer and Verdonschot, 2004). Point source emissions are characterised by little variability and their input to rivers has little dependence on flow characteristics through the catchment, whereas diffuse source emissions are dictated by the catchment hydrology (de Villiers, 2007). The main sources of excess human-derived nutrients entering waterways are from:

- Urban and industrial point sources: domestic treated sewage, industrial discharges, storm drainage
- Agricultural or urban diffuse source runoff: fertilisers and animal effluent, road and pavement runoff
- Delivery from agricultural sources can be exacerbated by type of farming method (e.g. ploughing), crops, fallow land and deforestation
- Phosphates used in detergents, liberated from mining activities and a variety of other complex sources can enter fresh waters via a variety of these routes

Phosphorus has become a management focus due to its contribution to eutrophication through the proliferation of nuisance phytoplankton as well as epiphytic and benthic algae (Mainstone & Parr, 2002) (Fig. 1). This can pose a major threat to water quality and environmental integrity (Smith and Schindler, 2009; Nijboer and Verdonschot, 2004). In extreme cases, excess nutrients can lead to a completely de-oxygenated (anoxic) environment, supporting nothing except a few species of bacteria (anaerobic bacteria) able to survive without oxygen. Numerous factors can increase the impact of eutrophication within a water system, for example 1) low water velocity increases the nutrients residence time and sequestration within the system, allowing nutrients to attach to river deposits, exacerbating eutrophication (Martins *et al*. 2001: Nijboer and Verdonschot, 2004) and 2) Aquatic macrophytes can halt the progress of nutrients downstream, increasing the nutrient retention period and thus encouraging eutrophication (Martins *et al*. 1999). Hydraulics, biological activity and sediment dynamics all have a key bearing on nutrient cycling and abundance in rivers and these processes must be considered in detail when assessing the cause of eutrophication and the necessary action to be taken.

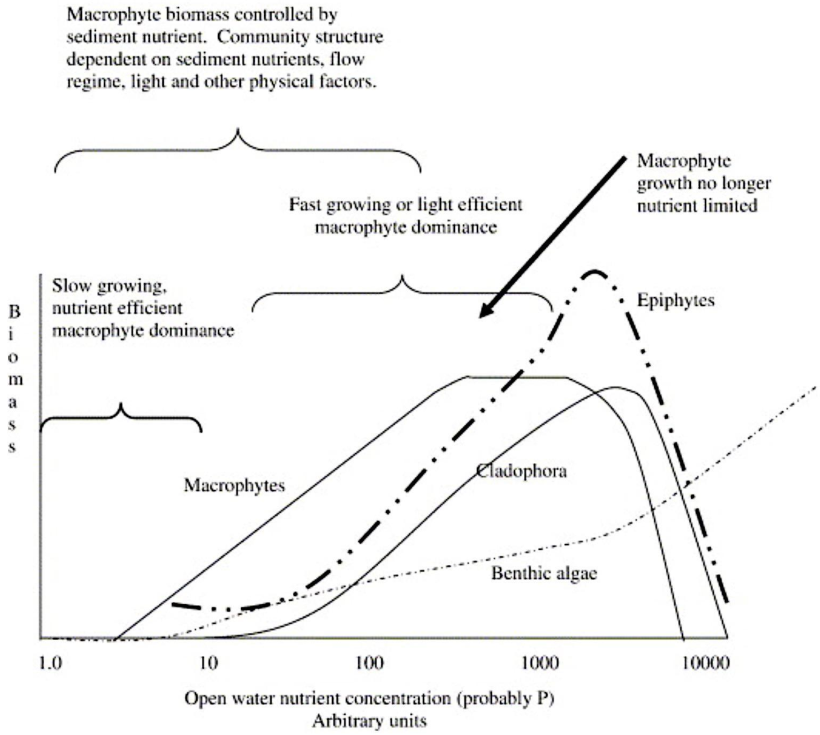

Figure 1: *Shift in aquatic flora towards nuisance algae and decline of macrophyte biomass as concentrations of phosphorus increase (Hilton et al. 2006).*

Soluble reactive phosphorus (SRP) is widely accepted as the main type of phosphorus readily available for aquatic organisms to use for growth. As a result managing in-river SRP levels is considered a key part of achieving good ecological health. However, recent work has highlighted the complexity of phosphorus in aquatic systems by examining the effects of particulate phosphorous stored in sediments. It is believed that phosphorus stored in the sediment itself can be a major contributor to nutrient enrichment due to resorption back into the water column (Koski-Vähälä & Hartikainen, 2001). In certain extreme conditions or after agitation this phosphorus can be released back into the water and contribute to eutrophication. Jarvie *et al*. (2005) showed under base flow conditions that high concentrations of SRP from sewage effluents in the tributaries of the Hampshire Avon and Herefordshire Wye ‘swamped out’ potential release of SRP from bed sediments. This suggests that rivers currently undergoing effluent P-stripping may not show reductions in SRP, as changes to the in-stream P-cycling mechanisms have the potential to switch bed sediments from net sinks to net sources of SRP. Conversely, Jarvie *et al.* (2006) demonstrated that particulate phosphorus retained as sediment on riverbeds had questionable relevance in regards to river eutrophication. Results indicated that bed sediments might be helping to reduce concentrations of SRP when the system is susceptible to eutrophication risk.

II. Impacts on aquatic flora

<u>Periphytes</u>

Phosphorus enrichment can quickly allow periphyton communities to develop at nuisance levels, in turn causing impairment of aquatic ecosystems. Excessive periphyton growth has been identified as the key ecological component driving the ecosystem degradation that can result from eutrophication (Hilton *et al.* 2006). Although studies have highlighted phosphorus as a key driver, understanding the additional ecological variables that work hand in hand with phosphorus to instigate proliferation of these nuisance organisms is critical for control. Rosemond *et al*. (2000) showed that irradiance limited periphyton biomass in summer and autumn, whereas nutrients limited periphyton biomass in seasons where light levels were higher. Nutrient enrichment had a greater scale effect in autumn and spring, indicating that phosphorous input at these times can trigger the greatest biomasses of nuisance periphyton and in turn the highest risk of eutrophication. Hill & Fanta (2008) looked at the combined effects of phosphorus and light on periphyton growth and found three lines of evidence indicating that light and phosphorus co-limit algal growth. Light predominantly accounted for the variance in growth and final biomass of periphyton, with phosphorus having more of a secondary role. Despite this, stimulatory effects on periphyton growth because of phosphorus did occur at relatively low irradiances and it was recommended by the authors that nutrient standards should be applied to streams irrespective of their light regimes.

Phosphorus input is not typically continuous, release from point sources such as fish farms and sewage treatment works is usually sporadic causing spikes of elevated phosphorus in river systems. Davies & Bothwell (2012) investigated biomass accrual of periphyton to these short-term fluxes. Results showed that periphyton reacted to hourly fluxes of phosphorus regardless of whether it was supplied in short concentrated pulses or continuously introduced at low concentrations. Compared with the control, there was also a substantial increase in biomass with pulses of only 1 min each hour. Conversely Bowes *et al*. (2010) previously demonstrated using within-river flumes in the river Kennet, UK that tripling the ambient phosphorus concentration over a short timescale of 9 days did not increase the accrual rate of periphyton. Therefore the excessive biofilm growth and algal dominance in the river Kennet was not because of intermittent peaks in SRP concentration, so regular failings of the sewage treatment works were not at fault. The authors concluded that the reason for this unresponsiveness was due to ambient levels of phosphorus being already in excess for algal growth.

Macrophytes

Macrophytes are key components of aquatic ecosystems and examining the assemblages can give invaluable insight into how well a river is functioning. For example, the improvement of sewage treatment works emissions at Ashford, England, resulted in the return of more generalist macrophyte species, replacing stress tolerant species indicating improved water quality and river health (Kelly and Wilson, 2004). Phosphorus loading has the ability to affect macrophytes directly and indirectly. Direct effects include alteration of biological traits. Mony *et al.* (2007) explored the adaptability of morphological and physiological traits in the submerged rooted macrophyte *Ranunculus peltatus* in response to phosphorus availability. Four physiological and four morphological traits assumed to reflect plant nutritional and functional changes according to nutrient availability were selected. Physiological traits depended widely on phosphorus availability. *R. peltatus* tended to adapt its phosphorus uptake rate during the elongation stage. In April increasing uptake rates occurred in low P-concentrations whilst the reverse was found for high P-availability. This indicates that the plants affinity for P was changed, with transportation through the roots increasing when the nutrient was limited. The results showed a lack of morphological functional changes and this was attributed to the selected traits potentially being weak indicators of plant plasticity responses to nutrient stress. The only morphological trait that did show notable changes was sexual reproduction. In low phosphorus availability sexual reproduction was enhanced. It was hypothesised that production of seeds in nutrient poor situations would give the plant better ability to regenerate when conditions were favourable. The most detrimental indirect effect of phosphorus pollution to macrophytes is the stimulation of epiphytic growth. A study by Wade *et al*. (2002) used modelling to explore macrophyte and epiphyte dynamics in the river Kennet, UK. Model simulations indicated that smothering by epiphytes was a critical limitation to the growth of macrophytes. Increases of soluble reactive phosphorus will initially stimulate macrophytic growth, but following this growth epiphyte biomass will peak from increased SRP and increased availability of macrophyte surface area to grow on. Excess epiphytic growth causes shading of macrophyte leaves, reducing the ability of the plant to photosynthesise.

A vast amount of literature highlights the capability of macrophytes to help shift eutrophic freshwater systems from turbid phytoplankton-dominated systems to clear, macrophyte-dominated systems through phosphorus uptake. Although the potential of macrophytes for phytoremediation of phosphorus in eutrophic lakes and wetlands has been well documented (Gao *et al*. 2009; Song *et al.* 2013; Di Luca *et al.* 2015), successful applications and recommendations for use in eutrophic river systems are scarce. A study by O'Brien *et al*. (2014) showed that in a stream environment phosphate uptake velocity was negatively related to primary production, suggesting that macrophyte activity actually slowed phosphorus uptake. As macrophyte phosphorus demand was 10 times lower than overall phosphorus uptake the authors suggested that other stream biota had much greater demands for phosphorus. It was concluded from this work that macrophytes have limited influence on water column nutrient concentrations, as macrophyte demand is much lower than the supply available from the water column. Therefore, in this case macrophytes offer little potential as a management solution for nutrient removal. This study complemented the findings by Garbey *et al.* (2004) well, as these authors found that *Ranunculus peltatus* had an inability to accumulate phosphorus in eutrophic environments, which suggested that the species could have restricted growth when exposed to high nutrient conditions. As nutrient loading initially increases the standing crop of the most competitive macrophytes until they are out-competed by epiphytic growth, excessive growth of macrophytes can also be problematic in river systems. O'Hare *et al.* (2010) highlighted the implications of increased standing crops of *Ranunculus* (subgenus *Batrachium*) triggered by phosphorus loading. After sampling across the UK it was found that standing crops increased significantly with P availability as soluble reactive phosphorus (filtered SRP) and total phosphorus (TP). These proliferations in standing crop can exacerbate flood risk by impeding river flow.

III. Impacts on aquatic fauna

Invertebrates

It is know a well-known fact that nutrient enrichment decreases richness of macroinvertebrates through the elimination of sensitive taxa comprised mostly of the insect orders Ephemeroptera, Plecoptera and

Trichoptera. Ortiz & Puig (2007) showed that above a nutrient rich point source effluent benthic invertebrate taxa richness was 20% higher than downstream, with mayflies, stoneflies and caddisflies being present only in the upstream reach. Friberg *et al.* (2010) also investigated occurrence of macroinvertebrate taxa, but this time correlated it to more specific gradients in water chemistry. The results showed overall negative logarithmic relationships for invertebrate taxa with increasing levels of total phosphorus and BOD5 (a measure of the quantity of labile organic matter while the nutrients express the level of eutrophication). Sharp declines with increasing BOD5 levels were found for the trichopteran families *Sericostomatidae* and *Glossosomatidae*. The plecopteran genus *Leuctra* had the greatest sensitivity to increased BOD5. Some taxa exhibited a stronger negative relationship with total phosphorus than BOD5 including the plecopteran genus *Nemoura* and the ephemeropteran family Leptophlebiidae. Grantham *et al*. (2012) highlighted the importance of conducting future work to evaluate how invertebrate communities respond to pulse impacts of nutrient rich effluent discharge. Their experiment found that continuous exposure as low as 5% caused moderate declines in diversity metrics and a loss of sensitive species, but as nutrient exposures are typically sporadic from wastewater treatment plant overflows during rain events, evaluating the effects from this perspective would be more beneficial for river management.

Fish

The oxygen deficient environment, created by excessive microbial activity utilising oxygen in the decomposition of aquatic fauna, increases fish mortality rates (Jarvie *et al*. 2005). Although fish are mobile organisms and can move away from regions of low-oxygen, the stressed system created by excess nutrients does result in less fish biomass and species diversity, with rarer, less-tolerant species suffering the most. Further threats to fish populations are magnified by eutrophication as it encourages harmful myxobacteria and pseudomorics, as well as unnatural pH readings (Snieszko, 1974), which can lower fish survival rates. Physical factors of the environment are also affected. Juvenile salmonids can be deferred from anoxic regions due to raised water temperatures caused by eutrophication (Lappalainen, 2002), thus disrupting natural migrations. Linked via the food web, the alteration in the producer community can have consequences for the community of consumers (Martins *et al.* 1997). The depletion of the plant population stimulates a rise in the diatom (plankton algae) populations and a decrease in cyanophytes (bacteria algae) (Schriver *et al*. 1995) altering the dynamics of the food web and availability of different food types for fish. Hill *et al.* (2010) showed how elevated periphyton production triggered community shifts in invertebrates and fish species in East Fork Poplar Creek, Tennessee USA. Excessive addition of nutrients placed the upstream region of in a state of eutrophy and in this region over 50% of the biomass of invertebrates and fish were grazers, yet downstream and at other reference sites less than 10% were grazers. Fish can be sensitive to changes elsewhere in the trophic system, losing a source of nutrition when invertebrate populations suffer from eutrophication (Lappalainen, 2002).

IV. Current Policy

Soil nutrient balances give an indication of the potential risk associated with losses of nutrients to the environment. The results published by DEFRA this year for the 2015 levels show that the phosphorus balance for England was in a surplus of 3 kg/ha of managed agricultural land, a decrease of less than 1 kg/ha (-10%) compared to 2014 and a 6 kg/ha reduction (-63%) compared to 2000 (DEFRA, 2016). The Environment Agency also claims that since 2009 around 350 river water bodies have improved in status for phosphorus. This improvement is attributed largely to phosphate stripping but work to address nutrient run-off from agriculture is also playing a part (Environment Agency, 2015).

Although these reductions look positive, according to DEFRA farming is still contributing more than 25% of the phosphorus getting into water sources (Edwards *et al*. 2015) and agricultural pollution is still costing the water and tourism industries, and the taxpayer and angling groups, between £758 million to £1.3 billion each year (DEFRA, 2015). The European Commission (EC) has even issued legal guidance warning the United Kingdom to ensure that the Water Framework Directive is correctly enacted in national law. The EC asked for more precision in the enactment of some central elements in the Water Framework Directive such as environmental objectives and programmes of measures required to attain them. The Commission also highlighted how under the existing UK arrangements national authorities may lack the necessary powers to tackle negative water impacts (European Commission, 2015). To tackle agricultural diffuse pollution in England DEFRA consulted on new basic rules for farmers. Eleven basic rules and actions were set in the consultation. DEFRA proposed introducing a minimum of seven of these rules and sought after views on a further four. Examples of these rules include: Properly calibrated and maintained machinery and the storage of field manure to be located at least 10m from the nearest watercourse. Analysis indicated that the first seven rules would trigger a 2.4% reduction in phosphorus and if the other four rules and actions were undertaken a 6.6% reduction in phosphorus would occur (DEFRA, 2015). DEFRA's aim is to introduce legislation during 2016/17.

The Environment Agency's NEP (National Environment Programme) lists environmental improvement schemes that companies can incorporate as part of their business plans. The business plans are then submitted to Ofwat (the water services regulation authority), where 5-year price limits were determined. In the most recent NEP 434 actions were proposed for the water companies of England and Wales under the Water Quality Water Framework Directive. Out of these actions 36 improvements and 7 investigations were proposed related to phosphorus (Environment Agency, 2010). However, the final outcome in the Ofwat price review for 2010-2015 showed price limits for 87 improvements and 52 investigations for water quality overall (including phosphorus, ammonia, BOD and dissolved O_2 standards), a total of 139 outputs for compliance with the Water Framework Directive (Ofwat, 2009).

Additionally, in 2013 the UK Technical Advisory Group (UKTAG) published recommendations to revise the standards for phosphorus in rivers, as it was determined that the standards set in 2009 were not sufficiently stringent (UK Technical Advisory Group, 2013). In 75% of rivers with clear ecological impacts of nutrient enrichment, the existing standards produce phosphorus classifications of good or even high status. Taking on board these recommendations DEFRA updated the standards in the second cycle of the Water Framework Directive river basin management planning process (DEFRA, 2014). The new phosphorus standards were designed to take into account the latest scientific evidence on how elevated phosphorus affects aquatic plant communities and have standards tailored to specific conditions of river sites that consider natural variation of nutrient concentrations along rivers and site-to-site differences in the ecological response to elevated concentrations (Table 1). Due to the uncertainty of the standards matching the ecology at individual sites the report emphasises that the proposal is not to seek costly actions for phosphorus reduction at individual sites. Therefore, costly actions will only be undertaken at individual sites with appropriate ecological evidence of nutrient-related impacts.

Type (for existing standards)	Annual mean of reactive phosphorus (µg per litre)							
	High		Good		Moderate		Poor	
	Existing	New	Existing	New	Existing	New	Existing	New
Lowland, low alkalinity	30	19 (13-26)	50	40 (28-52)	150	114 (87-140)	500	842 (752-918)
Upland, low alkalinity	20	13 (13-20)	40	28 (28-41)	150	87 (87-117)	500	752 (752-851)
Lowland, high alkalinity	50	36 (27-50)	120	69 (52-91)	250	173 (141-215)	1000	1003 (921-1098)
Upland, high alkalinity	50	24 (18-37)	120	48 (28-70)	250	132 (109-177)	1000	898 (829-1012)

Notes:
1. The revised standards illustrated are the medians from, respectively, 456 lowland, high alkalinity sites; 129 upland high alkalinity sites; 137, lowland, low alkalinity sites; and 97 upland, low alkalinity sites. The numbers in parentheses are the upper and lower 5th and 95th percentiles of the standards for the sites in each type.
2. "Lowland" means less than or equal to 80 metres above mean sea.
   "Upland" means more than 80 metres above mean sea level.
   "Low alkalinity" with a concentration $CaCO_3$ of less than 50 mg per litre.
   "High alkalinity" with a concentration $CaCO_3$ of greater than or equal to 50 mg per litre.

***Table 1:** Summary of existing and revised standards for phosphorus in rivers.*

V. Concluding Remarks

Excess nutrients can lead to eutrophication, when rapid algal growth and its consequent decay remove oxygen from the water. This can cause a severe decline in species diversity, including fish diversity. Current management strategies are failing to deliver the reductions required in nutrients to achieve the objectives of Water Framework Directive. Action must focus on reducing pollution at source, rather than relying on expensive 'end of pipe' nutrient removal strategies.

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VI. References

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