Science of the Total Environment 579 (2017) 702–708

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Mains water leakage: Implications for phosphorus source apportionment and policy responses in catchments

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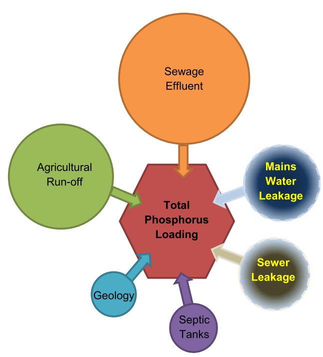
HIGHLIGHTS

- Mains water leakage of phosphate (MWL-P) dosed drinking water is currently not included in P budgets
- A new approach to estimate the spatial distribution and time-variant flux of MWL-P is demonstrated in an exemplar catchment
- Measures to reduce P from agricultural and sewage mean MWL-P could become a relatively more significant source of P
- There is a need to balance human health with ecological health
- New research is needed to better constrain the ultimate fate of MWL-P and the role of MWL-P within aquatic ecosystems

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 13 September 2016
Received in revised form 2 November 2016
Accepted 6 November 2016
Available online 14 November 2016

Editor: Simon Pollard

Keywords:
Phosphate
Eutrophication
Mains water leakage
Health
Policy

ABSTRACT

Effective strategies to reduce phosphorus (P)-enrichment of aquatic ecosystems require accurate quantification of the absolute and relative importance of individual sources of P. In this paper, we quantify the potential significance of a source of P that has been neglected to date. Phosphate dosing of raw water supplies to reduce lead and copper concentrations in drinking water is a common practice globally. However, mains water leakage (MWL) potentially leads to a direct input of P into the environment, bypassing wastewater treatment. We develop a new approach to estimate the spatial distribution and time-variant flux of MWL-P, demonstrating this approach for a 30-year period within the exemplar of the River Thames catchment in the UK. Our analyses suggest that MWL-P could be equivalent to up to c.24% of the P load entering the River Thames from sewage treatment works and up to c.16% of the riverine P load derived from agricultural non-point sources. We consider a range of policy responses that could reduce MWL-P loads to the environment, including incorporating the environmental damage costs associated with P in setting targets for MWL reduction, alongside inclusion of MWL-P within catchment-wide P permits.

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http://dx.doi.org/10.1016/j.scitotenv.2016.11.038
0048-9697/

1. Introduction

Phosphorus (P) is a vital element for all life. However, P is also the subject of an environmental paradox. One the one hand, world food security and the growing production of biofuels rely on enhanced P inputs to ecosystems, largely through the application of inorganic fertilisers and feed supplements manufactured from finite phosphorite deposits. Volatility in global markets can lead to dramatic increases in the price of P fertiliser, for example by 800% in 2008 (Cordell and White, 2011), meaning that parsimonious use and management of P resources is judicious (Cordell et al., 2009; Elser and Bennett, 2011; Elser et al., 2014; Jarvie et al., 2015). On the other hand and in parallel with increased mining and processing of phosphate rock, widespread enrichment of aquatic ecosystems with P has occurred in many parts of the globe (Carpenter, 2005). Anthropogenic inputs of P to these ecosystems have far-reaching effects, impairing water quality through stimulation of eutrophication with profound impacts on ecosystem function and health (Smith and Schindler, 2009). In turn, these ecosystem impacts can be directly linked to significant economic costs (Dodds et al., 2009; Pretty et al., 2003).

Research has attempted to quantify the absolute and relative contribution of agriculture (diffuse source) and sewage treatment work (STW) effluent (point source) to total P loadings in aquatic ecosystems (e.g. Jarvie et al., 2006; White and Hammond, 2009; Haygarth et al., 2014). Further work has considered the potential P load from other sources, including septic tank systems (e.g. Withers et al., 2012) and atmospheric deposition (Tipping et al., 2014). In response to P enrichment of aquatic ecosystems, policy and mitigation practices have predominantly targeted reductions in the export of P from agricultural land and from sources of waste water (i.e. sewage treatment works), involving changes in fertiliser, manure/slurry and other land management practices alongside the introduction of tertiary treatment technologies for P removal at sewage treatment works. However, these responses have had varying success with respect to improving water quality and reversing eutrophication within aquatic ecosystems (Hukari et al., 2016; Jarvie et al., 2013b; Lewis et al., 2011; Sharpley et al., 2015).

Here, we argue that P loads to the environment from mains water leakage (MWL) could be important in the context of eutrophication in aquatic ecosystems, but have not been sufficiently well constrained to date. Current P loads from MWL are potentially significant, especially within highly populated areas. Further, without action, the relative importance of MWL-P is likely to grow as P loads from other sources decline following the introduction of appropriate policies and mitigation practices. Therefore, the need to address MWL-P in order to protect and to restore aquatic ecosystems in the face of eutrophication is likely to increase in the future (Doody et al., 2014; Lewis et al., 2011).

Phosphate (PO_4) dosing of mains water supplies was introduced in the USA during the first half of the 20th century to prevent calcite precipitation within distribution networks (Rice and Hatch, 1939). The additional benefits associated with reduced iron corrosion from distribution pipes were quickly established (Hatch and Rice, 1940). However, widespread dosing of mains water supplies with PO_4 in the UK, parts of Europe (Flem et al., 2015) and the USA was not adopted until the 1990s, largely in response to legislative requirements to reduce lead (Pb) and copper (Cu) concentrations in drinking water due to the impacts of heavy metal exposure on human health (Edwards et al., 2009). In the USA, a standard of 50 $\mu g\ L^{-1}$ for both Pb and Cu in drinking water was originally adopted. However, since 1991 an action level of 15 $\mu g\ L^{-1}$ Pb has been introduced under the lead and copper rule (LCR). If the LCR is exceeded, appropriate action must be taken by the relevant water utility, including introduction or optimisation of PO_4-dosing. As permitted concentrations of Pb in drinking water have been reduced across Europe, for example from 25 $\mu g\ L^{-1}$ to 10 $\mu g\ L^{-1}$ in 2013 (EU Drinking Water Directive, 1998), there has been an increase in both the concentration and the spatial extent of PO_4-dosing to ensure better compliance with these more stringent standards (CIWEM, 2011; Comber et al., 2011). Current PO_4-dosing for drinking waters in the UK typically achieves final P concentrations between 700 and 1900 $\mu g\ L^{-1}$ (UKWIR, 2012) and is essentially applied nationally (95% of sources). In the U.S., more than half of water utilities use a range of PO_4-based corrosion inhibitors (Dodrill and Edwards, 1995). Where applied and optimised, PO_4 dosing of mains water represents an effective technological solution to reduce Pb and Cu concentrations in drinking water (Comber et al., 2011).

However, leakage from mains drinking water networks is a globally-significant issue, with the volume of water that leaks costing water utilities worldwide an estimated $14 billion per year (World Bank, 2006). Mains leakage from the distribution network in England and Wales is currently estimated to be 22% of treated water, equivalent to around 3200 $ML\cdot day^{-1}$, which has declined considerably since the mid-1990s when leakage peaked at just over 30% of treated water (CIWEM, 2015). Pipe failure in drinking water distribution networks is also a major concern within North America, where recent data from the USA and Canada suggest a current failure rate of 11 failures 100 $miles^{-1}$ $year^{-1}$, with highest failure rates over 5 years for cast iron (28 failures 100 $miles^{-1}$ $year^{-1}$), ductile iron (6.15 failures 100 $miles^{-1}$ $year^{-1}$) and steel (5.9 failures 100 $miles^{-1}$ $year^{-1}$) pipes (Folkman, 2012). Further, there has been a significant deterioration in the overall condition of drinking water distribution networks over the last three decades in the USA, with 68% classified as excellent in 1980, 42% in 2000 and 32% in 2010 (EPA, 2002; Folkman, 2012). A recent assessment of utility water loss in China found that the average leakage rate was approximately 18%, with 40% of water utilities suffering leakage rates >20% whilst some smaller utilities had leakage in excess of 60% (Pan et al., 2009). Although Holman et al. (2008) noted that leakage of PO_4-dosed mains water could be an important source of P, research has only recently attempted to quantify the load of P delivered to the environment from MWL. Within the UK, Gooddy et al. (2015) estimated the total P load from MWL to be approximately 1000 tonnes $year^{-1}$. Subsequently, using a more sophisticated national-scale modelling approach, Ascott et al. (2016) revised this figure to 1200 $tonnes\cdot P\cdot yr^{-1}$.

In this paper, we highlight the importance of properly accounting for MWL-P by developing an approach to quantify MWL contributions to P loads within the River Thames catchment over the past 30 years. The River Thames catchment is characterised by a high population density (~960 people km^2, Merrett, 2007) and variable mains leakage rates (between approximately 23 and 26% over the study period), and we compare estimates of MWL-P with P loads from both agricultural land and from STW effluent within the same catchment. Subsequently, we discuss how environmental policy could be adapted in the future to balance both protection of human health by minimising heavy metal exposure through drinking water and protection of aquatic ecosystems through reducing P loads derived from MWL.

2. Methods

2.1. Estimating the annual MWL-P load for 2011–2013

We estimated the annual load of MWL-P in the Thames catchment for 2011–2013 using published water company data. The Thames catchment is supplied by four water utilities: Thames Water; Affinity Water; Southeast Water; and Sutton and East Surrey Water. These companies also supply areas outside of the Thames catchment. Each water utility is divided into water resource zones (WRZs) in which the water supplied is largely self-contained in the area (Environment Agency, 2012). Within the Thames catchment, there are 10 WRZs and their boundaries coincide or very closely coincide with the topographical catchment boundary (Fig. 1). Published water company leakage rates ($ML\cdot day^{-1}$) for 2011–2013 for the WRZs were extracted from water company resource planning tables. These are publically available on water company websites (Affinity Water, 2014; Southeast Water, 2014; Sutton and East Surrey Water, 2014; Thames Water, 2014). The

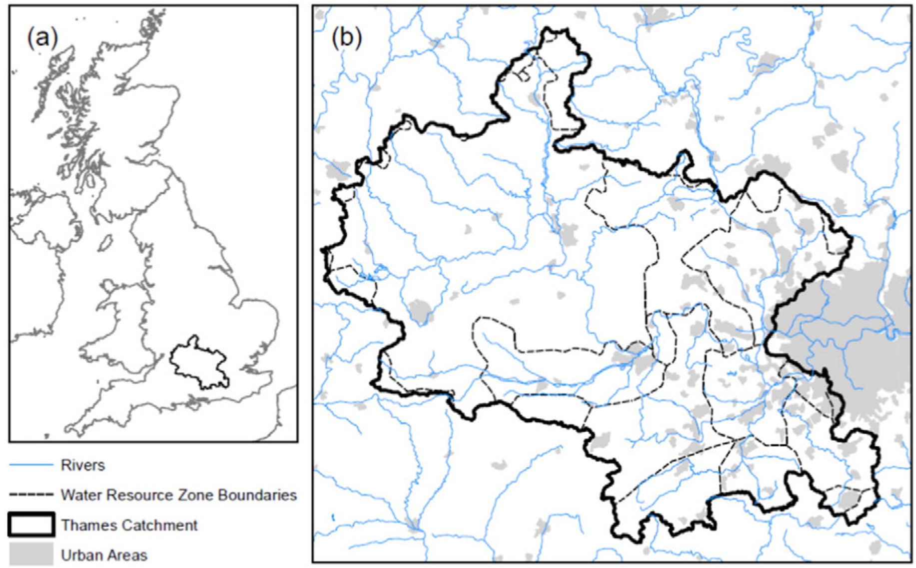

Fig. 1. (a) Location of the Thames study catchment and (b) location of the 10 water resources zones within the catchment. © UKP/Getmapping Licence No. UKP2006/01. Zones closely match the topographic boundary across the region.

load of P from MWL delivered to the Thames catchment (MWL-P_{Thames}, kt·P·year^{-1}) was then calculated as:

$$MWL-P_{Thames} = \sum L_{WRZ}.P_{WRZ}.\frac{365}{10^9}$$

where L_{WRZ} is the published WRZ leakage rate (ML·day^{-1}) and P_{WRZ} is the PO_4-P dosing concentration in mains water for a WRZ (taken as 1000 μg·P·L^{-1} based on Comber et al., 2011 and based on measurements in Gooddy et al., 2015).

2.2. Estimating the annual MWL-P load for 1994–2011

2.2.1. Historic leakage rates

A first estimate of the MWL-P load across the period 1994–2011 for the River Thames catchment was made using historic data for water company leakage rates, PO_4 dosing concentrations and dosing extents (Fig. 2). Annual water company level historic leakage rates (Fig. 2a) for the four water utilities above are available for 1998–2011 (Ofwat, 2003; Ofwat, 2006; Ofwat, 2010; Ofwat, 2015). Water resource zone level leakage rates (Fig. 2b) for this period were derived by back-extrapolating the observed WRZ data for 2011, assuming the same trend in leakage would occur at the WRZ and company level. For the period 1994–1998, historic leakage rates are only available for Thames Water (Ofwat, 2011). The same back-extrapolation approach was used to derive both company level and WRZ leakage trends for this period for all the water utilities in the Thames catchment. For 1994–1998, it was assumed that the trends in leakage for Thames Water are the same as the other three companies in the catchment.

2.2.2. Dosing concentrations and dosing extents

Very limited data are available to determine historical dosing extents (Fig. 2c) or dosing concentrations (Fig. 2d) for P in mains drinking water. On the basis that PO_4 dosing only began in earnest in 1994 (Comber et al., 2011), a linear increase in dosing concentration from zero to 646 μg·P·L^{-1} for 1994–2000 was assumed. The dosing concentrations reported by Comber et al. (2011) between 2000 and 2006 (which are based on mean values for 160 UK water resource zones) were then applied in our analysis. For the period 2006–2013, it was assumed that dosing concentrations remained constant at ~1000 μg·P·L^{-1}. This is likely to be a conservative estimate because the tightening of the Pb standard for drinking water in the EU in 2013 likely necessitated an increase in the concentration of P required within mains water in some areas (Comber et al., 2011). The spatial extent of PO_4 dosing has previously been estimated to have increased from 90 to 95% between 2007 and 2011 (CIWEM, 2011; UKWIR, 2012).

2.2.3. Dosing extent sensitivity analysis

Given the limited data available with which to constrain the extent of dosing, a sensitivity analysis was undertaken. Two further estimates of the temporal variation in MWL-P load for the River Thames catchment were derived: (1) Using a dosing extent 25% lower than the estimate above; and (2) Using a dosing extent 25% greater than the estimate above, limited to a maximum value of 100%.

3. Results and discussion

3.1. The importance of MWL-P to P loads and legacy P within the River Thames catchment

Table 1 and Fig. 3 summarise how P loads to the River Thames catchment from agriculture, STWs and MWL have changed over the past 30 years. Across the period 1981–2011, P loads from STWs within the catchment have decreased by 84%, whilst the load of P from agricultural land has fallen by 54% over the same period. These analyses indicate that policy and practice have successfully reduced the input of P to the River Thames from STW and agricultural sources. However, evidence suggests that despite this dramatic reduction in P loads, in-river P concentrations continue to exceed critical ecological thresholds and that the reduction in P loads has delivered little impact in terms of nuisance algal growth (Jarvie et al., 2013b; Bowes et al., 2012a, 2012b). Biological response to reduced P load and concentration seems to be delayed in many systems and/or P concentrations remain above biological thresholds despite significant reductions in P load (Jarvie et al., 2012; Jarvie et al., 2013b). The maintenance of P concentrations above thresholds that drive biological change may be due to the persistence of alternative sources of P that have not been properly accounted for to date in source apportionment work, including MWL-P.

Over the period 1994–2013, the relative and absolute contribution of MWL-P to the River Thames catchment have increased substantially.

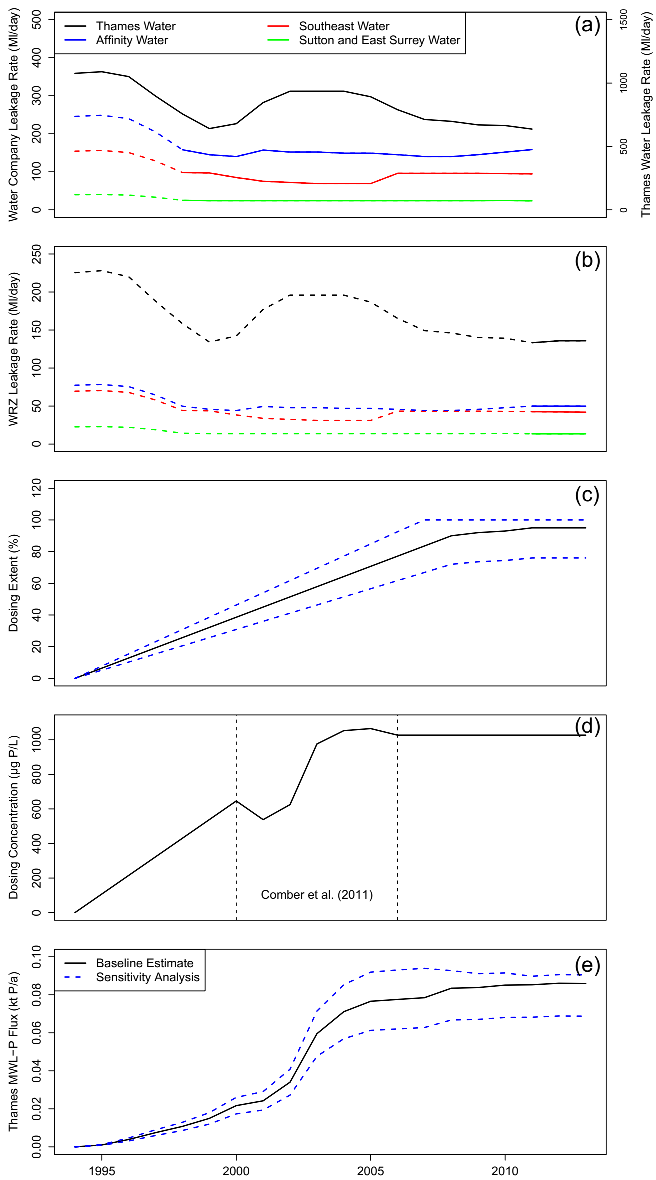

Fig. 2. (a) Water company level leakage rates, (b) derived (dashed lines) water resource zone (WRZ) leakage rates for the Thames catchment, (c) estimates of phosphate dosing extents, (d) estimates of phosphate dosing concentrations and (e) the derived MWL-P flux for the Thames catchment.

Depending on the proportion of MWL-P delivered to receiving waters, MWL-P loads may now be approaching a comparable order-of-magnitude to P loads from diffuse agricultural sources and from STW effluent (Table 1). Based on national figures for dosing concentrations of between 0.5 and 2 mg/L (Hayes, 2010), the relative proportion of MWL-P could be from 12% to 47% of sewage treatment effluent. Fig. 3 compares a worst case scenario, in which the maximum possible

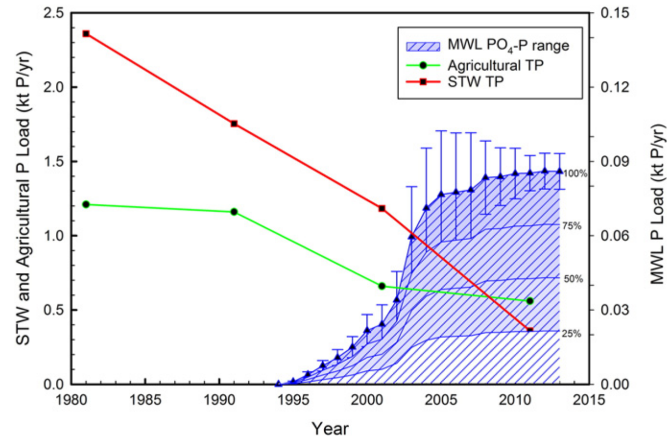

Fig. 3. Absolute and relative contribution of sewage treatment work effluent, agriculture and mains water leakage to P loads in the River Thames catchment, 1981–2013. Shaded area represents mains water leakage-P load assuming 0–100% of the total load from leakage is delivered to the river. Note factor of 10 difference in scale between individual y-axes.

contribution of MWL-P to the River Thames P budget occurs, assuming conservative transport of P and therefore 100% delivery of MWL-P to the river network, with scenarios of 25%, 50% and 75% delivery of MWL-P. The scenario of 100% MWL-P delivery reflects two conditions: (1) no return of MWL back into the sewer network (where MWL-P would be subjected to P removal technologies at STWs, where these have been implemented); and (2) no retention of P along hydrological pathways between the point of leakage from a mains distribution network and the catchment outlet. Based on this worst case scenario, our analyses suggest that if the trend in declining STW effluent P concentrations continues, for example due to more stringent and/or more widespread consents on final effluent P concentrations, the relative contribution of MWL to P loads within the River Thames could exceed STW P by the end of 2016.

River P concentrations that persist above biological thresholds may also be due to legacy P in catchments, associated with the accumulation and subsequent chronic release of P from environmental pools along the land-water continuum (Jarvie et al., 2013a; Sharpley et al., 2013; Powers et al., 2016). In a scenario where 100% of MWL-P arrives at the river network without significant storage in the catchment, there would be no contribution from MWL to legacy P. However in other more probable scenarios with < 100% transfer of MWL-P to the river network, some MWL-P will be retained within the catchment and could be released at a later date, thereby contributing to legacy P effects within catchments (Jarvie et al., 2014). By integrating average annual MWL-P loads from 1994 when P-dosing first started until our most recent estimates of MWL-P (Fig. 3), it is possible to estimate the total mass of P that has been released into the River Thames catchment due to MWL. Assuming a scenario in which 50% of MWL-P arrives at the river network without accumulation in the catchment, and the remaining MWL-P is stored within one or more environmental pool (e.g. soil, deeper sediments, groundwater), the legacy contribution to P within the River Thames catchment from MWL is approximately 1 kt P over the period

Table 1
Absolute and relative P loads from sewage treatment work effluent, agricultural sources and mains water leakage in the Thames catchment. Data for sewage treatment work effluent and agriculture are from Haygarth et al. (2014), data for agriculture are based on corrected fertiliser loads using the export coefficient approach for the Thames catchment from White and Hammond (2009), data for mains water leakage (MWL) are from the approach reported in the current paper.

Year	Sewage treatment effluent kt/year	Agriculture kt/year	Mains water leakage kt/year	MWL as % sewage treatment effluent	MWL as % agriculture
1981	2.36	1.21	0	0	0
1991	1.77	1.16	0	0	0
2001	1.18	0.66	0.03	2.5	5
2011	0.38	0.56	0.09	24	16

1994–2013. Ascott et al. (2016) estimated that ~15% of MWL fluxes may recharge to groundwater in the River Thames catchment. Given water residence times (around 25 years) within the shallow groundwater that MWL is likely to recharge within this catchment (Gooddy et al., 2006), it is probable that any MWL-P recharged to groundwater 20 years ago may now be discharging into river networks, assuming relatively conservative transport of P in shallow groundwater systems.

3.2. Policy responses to minimise MWL-P

Two highly significant challenges define the context for MWL-P. Firstly, minimising human health risks associated with exposure to contaminants such as Pb and Cu in drinking water. Secondly, minimising the contribution made by MWL-P to nutrient enrichment within the environment. Water utilities have invested significant capital and operating resources in reducing P loads delivered to receiving waters, both by working with land owners and land managers to mitigate P losses from agricultural land as well as by enhancing P removal at STWs (Kinniburgh and Barnett, 2010). As illustrated by the recent public health crisis in Flint, USA that was partly associated with inadequate PO_4 dosing of raw water sources (Torrice, 2016), fundamental human health, social and reputational effects mean that cessation of PO_4 dosing within distribution networks in which lead piping remains will never be an option to reduce P loads delivered to the environment. Below, we consider alternative responses to the challenge of reducing MWL-P whilst continuing to ensure that human health risks associated with drinking water are minimised.

The most obvious alternative to continued P-dosing of raw water sources is wholesale replacement of lead piping within drinking water distribution networks. This should include not only the communication pipes that are owned by water utilities, but also the below-ground supply pipes within the boundary of land that is the responsibility of homeowners and plumbing within domestic properties up to the final point of distribution at the domestic tap. Partial replacement of lead pipes within drinking water distribution networks is not a suitable response. Partial replacement has the potential to exacerbate corrosion of lead pipes and thereby increase human exposure to lead within drinking water, due to galvanic corrosion between the original lead piping and the replacement pipe that is often constructed from copper (St. Clair et al., 2016). However, full replacement of lead pipes has very significant cost implications. For example, in the USA the American Water Works Association has estimated the cost of replacing drinking water infrastructure at around \$1 trillion over the next 25 years (Shanaghan, 2012), whilst a \$70 billion programme under the True LEADership Act has recently been proposed by US senators that would include lead service line replacement. In the UK, it has been estimated that the market price of P used to dose raw water must go up by a factor of 20 before the replacement of lead piping would be financially viable (UKWIR, 2012). Lead rehabilitation has previously been tested and new methods are market-ready for deployment, including lining the internal walls of pipes with non-lead bearing materials (3M Infrastructure Protection Division, 2013). Rehabilitation would be cheaper than replacement as fewer excavations are required (UKWIR, 2012). However, the cost limited life-span of liners and the timescales involved in widespread lining or replacement may ultimately make these actions an unlikely solution to MWL-P in anything but the long-term.

Assuming that the present-day extent and final concentration of PO_4 dosing for drinking water is likely to continue, at least in the short to medium term, alternative approaches to reducing MWL-P loads merit consideration. Such approaches should focus on how MWL can be minimised, with the consequence that reductions in MWL will result in lower P loads being delivered to the environment from mains water. The (sustainable) economic level of leakage (SELL in the UK, or simply ELL in other countries), the leakage rate at which it would cost more to make further reductions in leakage than to produce replacement water from another source, is an important factor in long term investment planning within the water industry. The SELL also represents a minimum level of leakage against which the performance of water companies can be assessed. Preliminary research carried out two decades ago suggests that the (S)ELL is highly sensitive to assumed water cost, for example a 1% increase in the value of the lost water could lead to the (S)ELL falling by 10% (POST, 1995). The current methodology for calculating the SELL in the UK incorporates estimates for a number of externalities associated with MWL, for example the carbon costs, the interruption to water supplies, the disturbance to vehicle movement and the impact of noise pollution due to leakage, alongside the environmental benefits of reduced water abstraction following reductions in MWL. However, the SELL methodology does not currently include any estimate of the environmental damage costs associated with MWL-P. Given that the methodology for calculating the SELL is currently under review ahead of the next price review of water customer's bills in England and Wales, there is a timely opportunity to consider whether to incorporate MWL-P as an externality within a revised SELL methodology. For example, assuming a damage cost of c.\$47 (£33) per kg of P (Pretty et al., 2003) and the estimate of 1200 tonnes of MWL-P yr^{-1} in the UK from Ascott et al. (2016), multiplying these figures gives the total damage costs associated with P from MWL, which would be approximately \$57 million yr^{-1} (£39 million) in the UK. Clearly, this estimate assumes that all MWL-P remains within the environment and contributes to environmental damage. Significant uncertainty surrounds these assumptions, emphasising the need to better constrain the ultimate fate of MWL-P if more accurate assessments of the damage costs associated with this source of P are to be made. The consequence of incorporating MWL-P as an externality would be to lower the SELL and thereby to reduce P loads delivered to the environment from MWL, assuming that SELL targets were met. However, a proportion of any additional capital or operating costs associated with meeting a lower SELL target would be borne by water customers, which would require approval from the economic regulator in England and Wales and may well meet resistance from water customers.

Finally, the (S)ELL framework could be broadened to encompass a sustainable environmental/economic level of P release, thereby recognising MWL-P as a source of P that must be quantified and managed as part of landscape-scale controls on P delivery to the environment. The basis for such a framework already exists in the form of the Total Maximum Daily Load (TMDL) approach developed in the USA to deliver the requirements of the Clean Water Act. In the UK, initial trials of catchment-wide P permits, led by the Environment Agency in collaboration with the water industry although currently focussed solely on STWs, provide a similar opportunity to incorporate MWL-P within landscape-scale controls on P export to the environment. Within either a TMDL or catchment-wide P permit, MWL-P could be quantified and subsequently allocated a proportion of the TMDL (as a non-point source, therefore given a load allocation), or a proportion of the catchment P permit where a catchment permitting framework was extended beyond STWs. Where a TMDL or catchment P permit was exceeded following incorporation of current levels of MWL-P, a number of options would be available to water companies. Firstly, reductions in MWL and thereby in MWL-P could be specifically proposed by the water company in order to meet the TMDL or catchment-wide P permit. Secondly, a water company may choose to offset MWL-P by delivering an equal reduction in P load from other sources that fall within their remit, particularly through enhancing P removal at STWs. Finally, the potential to trade the reduction in P due to MWL-P required in order to meet a TMDL or a catchment P permit could be considered, for example by water companies contributing financially towards reductions in P export from agricultural land that matched this requirement. However, incorporating MWL-P within either the TMDL or catchment P permit framework would require accurate estimates of MWL-P loads that are derived from mains distribution networks. Accurate quantification of the ultimate fate of MWL-P would also be required, to constrain the proportion of MWL-P that is delivered to receiving waters as opposed to being returned to sewer or entering long-term storage within a catchment.

4. Conclusions

Effective strategies to reduce phosphorus enrichment of aquatic ecosystems require accurate quantification of the absolute and relative importance of individual sources of P. Assuming that mains water supplies will continue to be dosed with PO_4, MWL-P loads must be quantified more widely and the ultimate fate of MWL-P within the environment better understood. Addressing these challenges would underpin more accurate P source apportionment models, enabling policy and investment to be effectively targeted in order to protect and restore aquatic ecosystems facing the risk of eutrophication. Perhaps more fundamentally, this information will provide insight into the way in which finite P resources are used to maintain drinking water supplies, supporting optimisation of this demand for P in the future.

Acknowledgements

The research was funded by the UKs Natural Environment Research Council (NERC) National Capability resources devolved to the British Geological Survey and through the NERC Macronutrient Cycles Programme (LTLS project, Grant No. NE/J011533/1). This paper is published with permission of the Executive Director, British Geological Survey (NERC).

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