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An alternative approach to risk rank chemicals on the threat they pose to the aquatic environment

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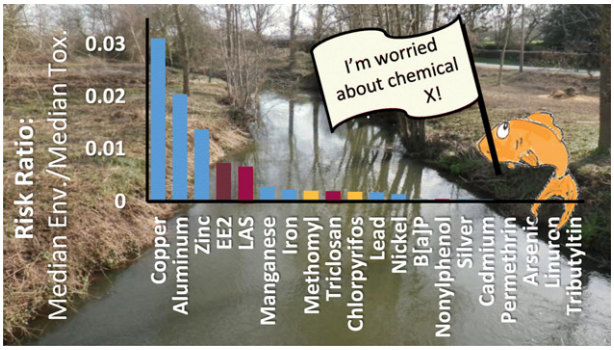
HIGHLIGHTS

- An analysis of the relative risk of 71 different chemicals was made
- Risk was based on the median or a percentile of the ecotoxicity values
- Ecotoxicity was compared with a median or a percentile of water concentrations
- The relative risk was highest for the metals copper, aluminium and zinc
- Ethinylestradiol, linear alkylbenzene sulfonate and triclosan were in the top ten

GRAPHICAL ABSTRACT



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ABSTRACT

This work presents a new and unbiased method of risk ranking chemicals based on the threat they pose to the aquatic environment. The study ranked 12 metals, 23 pesticides, 11 other persistent organic pollutants (POPs), 13 pharmaceuticals, 10 surfactants and similar compounds and 2 nanoparticles (total of 71) of concern against one another by comparing their median UK river water and median ecotoxicity effect concentrations. To complement this, by giving an assessment on potential wildlife impacts, risk ranking was also carried out by comparing the lowest 10th percentile of the effects data with the highest 90th percentile of the exposure data. In other words, risk was pared down to just toxicity versus exposure. Further modifications included incorporating bioconcentration factors, using only recent water measurements and excluding either lethal or sub-lethal effects. The top ten chemicals, based on the medians, which emerged as having the highest risk to organisms in UK surface waters using all the ecotoxicity data were copper, aluminium, zinc, ethinylestradiol (EE2), linear alkylbenzene sulfonate (LAS), triclosan, manganese, iron, methomyl and chlorpyrifos. By way of contrast, using current UK environmental quality standards as the comparator to median UK river water concentrations would have selected 6 different chemicals in the top ten. This approach revealed big differences in relative risk; for example, zinc presented a million times greater risk then metoprolol and LAS 550 times greater risk than nanosilver. With the exception of EE2, most pharmaceuticals were ranked as having a relatively low risk.

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1. Introduction

1.1. Overview

As society has developed over recent decades, so has the development and consumption of chemicals, so that now over 100,000 substances are marketed in Europe (Bu et al., 2013). Many of these enter water courses via direct or indirect routes. As the various chemical industries have developed, this has often led to an increase in freshwater contamination by chemicals over time (Schwarzenbach et al., 2006). Soil can also be exposed, both by design, such as with pesticides, and inadvertently through contaminants emerging from treated sewage sludge disposal to land and through atmospheric deposition. However, freshwater exposure is on a different scale (at least in countries where most of the population live inland), due to the discharge of all our domestic liquid waste, via sewage (wastewater) treatment plants, to river systems every day of the year. Given the relatively modest size of the environmental science community and the resources to support it, it is difficult to know on which chemicals research or regulatory efforts should be focused to best protect our aquatic wildlife. It is human nature for scientists and funders to switch their focus dramatically to new challenges, such as is currently occurring with nanoparticles, microplastics or neonicotinoid pesticides. But a succession of these transient obsessions could leave us with considerable knowledge on some substances whilst others remain un-studied (Anastas et al., 2010; Grandjean et al., 2011). Deciding which chemicals are of most concern is a global challenge (Brooks et al., 2013; Bu et al., 2013). To tease out the relative risk to wildlife of different chemicals, an approach which is as unbiased as possible is needed.

1.2. The current approaches to regulating and prioritising chemicals and their weaknesses

It is first necessary to try to distinguish chemical prioritisation from risk ranking. Chemical prioritisation is the traditional precursor to chemicals regulation. In this case chemicals are selected for control usually on the basis of possessing hazardous properties such as carcinogenicity, mutagenicity, toxicity, bioaccumulation potential or persistence. Sometimes this is accompanied by a risk assessment process, in which case the likely degree of exposure is also considered (Hansen et al., 1999; Wilkinson et al., 2007; Daginnus et al., 2010). In any case, the selected priority contaminants are lumped together for subsequent control without further indications as to their relative risk. In contrast, chemical risk ranking attempts to indicate the relative risk of each chemical against another. Sometimes some form of risk ranking may be used to generate the pool of chemicals for prioritisation (Swanson et al., 1997). Part of the reason for focusing on hazards such as carcinogenicity (where potency is not considered) is due to concerns over human health (Guillen et al., 2012). Thus, the chemical prioritisation process is claimed to cover both environment and human health. However, carcinogenicity would not be of great importance to most aquatic wildlife, which are short-lived. A persistent and bioaccumulative chemical might not be very toxic. Thus, such approaches could downplay toxic chemicals which, whilst neither persistent nor bioaccumulative, are 'pseudo-persistent', being perpetually present in rivers with a high wastewater exposure (Daughton, 2004). If hazard factors are to be used, then some subjectivity is inevitable in the scoring system. Should greater emphasis be given to whether a chemical is a suspected carcinogen rather than its toxicity to aquatic organisms?

The intention of risk assessment with regards to regulation is to derive an environmental quality standard (EQS) based on a predicted no effect concentration (PNEC). This method is designed from the outset to be precautionary and considerable attention is given to the lowest reported effect concentrations and highest reported/predicted river water concentrations (von der Ohe et al., 2011). The PNEC is commonly used as a value to compare toxicity between chemicals and the ideal method is to use a species sensitivity distribution (SSD), which may be employed when data are available on at least 20 different species. When a best fit line is plotted, the points at the bottom of the curve, where effects have been reported at the lowest concentrations, take on a great deal of significance. This approach can be used to identify a concentration above which 95% of species would be protected (HC5). Thus, we depend a great deal on a few studies showing effects at low concentrations being reliable and that the model drawing the curve is suitable. Without the benefit of an SSD, the lowest effect concentration for an aquatic species must be found. From such information an additional safety or adjustment factor (AF) is added to derive the PNEC, a level which, if not exceeded, should protect all aquatic wildlife in the absence of other pressures. Note, depending on our knowledge, or lack of it, different chemicals will receive different AFs, which may be up to 1000 (Hansen et al., 1999; von der Ohe et al., 2011).

With the lowest observable effect concentration (LOEC) and PNEC, great weight is placed on reports of the most sensitive species and effects which can lead to questions over the quality or reliability of these studies. For some chemicals these data may be well founded, but for others not. For example, a recent examination of studies on diclofenac and fish reveals some of the potential problems with using sub-lethal end-points (Wolf et al., 2014). There can be many potential weaknesses in ecotoxicity experiments (Harris et al., 2014). Indeed some may advocate that only studies which meet very strict standards may be used in a review (Klimisch et al., 1997). However, many ecotoxicology studies do not meet the highest standards, so dismissing such information may lead to few or no values remaining in a database!

There can be issues with defining exposure to the chemical in the environment too. Ideally the monitoring locations have covered all of the region, nation or continent for which the risk ranking is required. Ideally, the monitoring occurs frequently throughout the year and that the analytical methods are sufficiently sensitive as well as robust. However, the use of studies where high river concentrations are reported may be problematic: did analytical error occur or did the authors simply select locations very close to a pollution source which were not representative of the whole surface water body (Johnson et al., 2008)? Where many values were reported as below the limit of quantification (LOQ), was the chemical only just below the LOQ or entirely absent? A precautionary way to sidestep the problem is to use a high percentile of the measured values, such as the 90th or 95th percentile (von der Ohe et al., 2011).

To arrive at the final prioritisation result or risk rank, many studies and indeed the preferred European regulatory approach is to use a complex scoring system with equations to generate their result (Swanson et al., 1997; Wilkinson et al., 2007; Daginnus et al., 2010; von der Ohe et al., 2011; Kuzmanovic et al., 2015). Because of the many factors included and the complexity of the analysis involved this may give reassurance to some, but it can be criticised as being muddled and prone to subjectivity with its weightings. The intervention of the scoring systems has the effect of distancing the reader from the source data. So there is a strong argument to avoid this confusion by devising a transparent ranking system where the focus is solely on aquatic effects (toxicity) and exposure (Bu et al., 2013).

1.3. Devising an alternative approach to chemical risk ranking

An apparently attractive approach to compare different chemicals in a robust and reliable way might be to restrict the effects data only to tried and tested data obtained from Organisation for Economic Co-operation and Development (OECD) toxicity tests. However, there are three problems with adopting such an approach: firstly, OECD test data are not available for all chemicals, such as pharmaceuticals or nanoparticles. Secondly, some species may be more sensitive than the OECD test species (Wedekind et al., 2007). Thirdly, we are becoming increasingly aware that there could be end-points which might harm or disrupt different wildlife but which are not included in OECD tests such as endocrine disruption.

For a fair and reliable chemicals risk ranking protocol it would, therefore, seem safer to include as much ecotoxicity and surface water monitoring data as possible. There are two main reasons for this; firstly, to avoid having an insufficient quantity of data which would prevent the exercise taking place at all, and secondly, to include unusual evidence of harm identified by scientists using novel approaches. This leaves the question of what metric should be used as the comparator for the risk ranking? For this study both the median of the ecotoxicity data was compared with the median water concentration, as well as the lowest 10th percentile of the effects value being compared with the highest 90th percentile of the exposure data; the proximity of the two indicating the degree of risk. In this way, whilst even the more doubtful studies purporting to show effects at low concentration, or surprisingly high water concentrations are included, these do not have an excessive influence (as they might on the mean, or even more the extreme percentiles such as 5%ile, 95% etc.). Thus, the chemicals identified here as being of high risk imply a danger to a very wide range of organisms in a very wide range of locations.

2. Materials and methods

2.1. Overview

Whilst isolated industries or spills can damage wildlife, these local situations were not the topic of this research. Instead, the study focused on to what degree a chemical might be of widespread concern. The project was limited to exposure in the UK, so measured UK river concentrations were preferred. In the case of very limited data, other European measurements were included, and in the case of the pharmaceuticals and nanoparticles, modelled values were also incorporated and this method has been outlined before (Donnachie et al., 2016). The approach for the pharmaceuticals was to use reported effluent concentrations as the starting point and then apply a range of dilution factors appropriate for the UK. This is precautionary as it assumes no losses in the water. The model for the nanoparticles is somewhat more speculative as its starting point is assumed per capita consumption levels and predicted losses in wastewater treatment (Dumont et al., 2014). With respect to exposure to chemicals from the domestic population, the UK could be considered one of the most exposed countries in the developed Western world, due to a high population density and only relatively small rivers into which treated sewage is discharged (Keller et al., 2014). A detailed explanation of the data assimilation and application for chemical risk ranking has been reported previously (Donnachie et al., 2014). For the selected chemicals both the scientific literature and grey literature reports were reviewed. Whilst the literature review may not guarantee all the appropriate publications are found, the limitations were the same for all chemicals reviewed.

2.2. Chemicals selected for study

The selection of chemicals to represent each group was influenced by the degree of concern expressed in the scientific literature and to some extent by their ubiquity/prevalence in water (Table 1). In the case of the pharmaceuticals, these were selected on the basis of the frequency of reporting in pharmaceutical prioritisation papers (Donnachie et al., 2016). As for engineered nanoparticles, we have little information on the different types released to water or their concentrations to date, so the frequently used nano-Ag and nano-ZnO were assessed only. In their case and with the pharmaceuticals, river water modelled values were included (Dumont et al., 2014; Donnachie et al., 2016). A high priority was given to pesticides, given their prominence in a recent assessment on risks to European aquatic wildlife from organic chemicals (Malaj et al., 2014).

Table 1
The complete list of chemicals assessed against one another in the risk ranking study, together with the colour-coding used in Fig. 2 (numbers in brackets are identifiers for Fig. 2).

Metals Blue	Pesticides* Orange	Other persistent organic pollutants* Green	Pharmaceuticals Purple	Surfactants & similar Red	Nano particles Grey
Aluminium (2)	Bentazone (70)	Benzo[*a*]pyrene (B[*a*]P) (14)	Aspirin (71)	Alcohol ethoxylated sulphates (AES) (13)	Nano silver (44)
Arsenic (19)	Beta–hexachlorocyclohexane (β-HCH) (61)	Dibutyltin (39)	Atenolol (63)	Alkyl sulphate (42)	Nano zinc oxide (31)
Cadmium (17)	Carbofuran (24)	Dichlorobenzene (DCB) (57)	Carbamazepine (58)	Benzotriazole (37)	
Chromium (40)	Chlorpyrifos (10)	Fluoranthene (28)	Diclofenac (51)	Bisphenol A (25)	
Copper (1)	Diazinon (32)	Hexachlorobutadiene (45)	Ethinyl estradiol (EE2) (4)	Bis(2-ethylhexyl) phthalate (DEHP) (41)	
Iron (8)	Dichlorodiphenyl-dichloroethylene (DDE) (26)	Polychlorinated biphenyl 52 (PCB 52) (36)	Fluoxetine (35)	Linear alkylbenzene sulfonate (LAS) (5)	
Lead (11)	Glyphosate (60)	Polychlorinated biphenyl 153 (PCB 153) (55)	Ibuprofen (53)	Nonylphenol (15)	
Manganese (7)	Imidacloprid (52)	Polychlorinated biphenyl 180 (PCB 180) (29)	Metoprolol (69)	Octylphenol (20)	
Mercury (34)	Lenacil (21)	Perfluorooctanesulfonate (PFOS) (49)	Naproxen (64)	Sucralose (48)	
Nickel (12)	Lindane (γ-HCH) (38)	Trichlorobenzene (TCB) (56)	Ofloxacin (67)	Triclosan (6)	
Silver (16)	Linuron (22)	Trichloromethane (TCM) (62)	Paracetamol (47)		
Zinc (3)	Malathion (30)		Propranolol (43)		
	2-Methyl-4-chlorophenoxyacetic acid (MCPA) (66)		Sulfamethoxazole (65)		
	Mecoprop (59)				
	Metaldehyde (68)				
	Metolachlor (33)				
	Methomyl (9)				
	Pendimethalin (46)				
	Permethrin (18)				
	Pirimicarb (50)				
	Simazine (54)				
	Terbuthylazine (27)				
	Tributlytin				

* It will be recognised that many pesticides are POPs and vice versa. For the purposes of this study we have made arbitrary decisions into what category to place some chemicals.

2.3. Environmental toxicity information gathering

For all chemicals, publications were searched for using the same series of key words in the Web of Science™ database of scientific journals (Table S1) (Donnachie et al., 2014). The search focused on retrieving studies of the effects of a chemical on aquatic organisms and the concentration of a chemical in the UK aquatic environment. With ecotoxicology, the preference was for studies reporting measured concentrations, rather than nominal. A range of effect reports were collected from the literature including LOEC, EC50, LC50, lethal and sub-lethal toxicity. The widest range of species (preferably from the UK or Europe) and end-points were considered, to ensure that as representative a picture of species and possible effects as possible was obtained. Reviews, cross-referencing and consensus within the literature on which were the most sensitive organisms and end-points for a chemical were noted. The number of different species for which toxicity information was available for each chemical was on average 23, but ranged from 4 to 94 (Table S2, Fig. 2). For those chemicals with limited ecotoxicology information every single report could be incorporated in the database, such as for the pharmaceuticals and some POPs. However, with chemicals with an abundance of ecotoxicity values, such as metals with many 1000s' of reports, judgement was used in selecting a representative range of values, particularly not ignoring effects at low concentrations (Donnachie et al., 2014). With metals, the species will change according to the pH. Most UK rivers have neutral or slightly alkaline pH (Neal and Robson, 2000; Salminen et al., 2005; De Vos et al., 2006). Therefore, all ecotoxicity data for metals carried out at pH values below 6.5 or above pH 8.5 were discarded. The aim of the literature search on each chemical was to get an overall impression of the effects of a chemical on a range of aquatic organisms, and so provide a fair reflection of the hazard they represent. In total 4396 separate ecotoxicity data points were included.

2.4. Gathering information on environmental concentrations

For the chemicals studied, measurements of concentrations made in the UK were the primary objective. These included total measured river water concentrations reported in the literature, but also from other available databases (Table S2, Fig. 2). These other databases included the Forum of European Geological Surveys (FOREGS, now EuroGeoSurveys) (Salminen et al., 2005; De Vos et al., 2006) and UK Environment Agency monitoring data ("WIMS" data, mainly from river monitoring but also including values from some lakes (Environment Agency, downloaded 3/2013), using 2010–2012 data if available, but going back to 2000 when there were no or little recent data). Unfortunately, for some pesticides known to be used in the UK (terbuthylazine, metolachlor, lenacil and imidacloprid), no measurements appear to have been undertaken. In these cases, values from across Europe were collected using the publically available WaterBase rivers database, now aggregated with groundwater and lakes data (European Environment Agency, downloaded 3/2015, at the time of download data up to 2012 were included).

In some cases metal measurements for rivers were reported as total and others as dissolved. As the ecotoxicity of metals pertains to their dissolved concentration, only dissolved metal measurements in the environment were used. However, it is acknowledged that the actual toxicity of a metal in water is linked to many complex chemical interactions, including competition between metals for the binding sites on ligands or target organs (Rüdel et al., 2015). Thus, as a test case the biotic ligand model (Bio-met bioavailability tool, version 1.4., downloaded 11/2011) was used to examine Cu, Zn and Mn using the differing chemistry of typical UK lowland rivers (ca. 40–120 mg/L, DOC 5.1–8.1 mg/L, pH 7.4–8.1 (Neal and Robson, 2000)).

No single data source provides an ideal balance of measurements from around the country. Measurements of concentrations reported in the scientific literature often focus on locations believed to be hot spots, such as rivers receiving mine waste contaminated with metals, whilst the FOREGS database focuses on second order, drainage basins and the WIMS data (Environment Agency, downloaded 3/2013) contains many entries where the concentrations were found to be below the limit of detection. These non-detects were included in this study by reporting them as half the LOQ. Overall, 309,049 separate data points for environmental concentrations were included.

2.5. Risk analysis

The information from the ecotoxicology and environmental concentrations was plotted and the medians noted (Fig. 1). The difference between these medians can be described as a risk ratio, which can be used to rank concern; the larger the value, the greater the concern (Eq. (1)).

$$\text{Risk} = \frac{mW}{mT} \quad (1)$$

where mW is the median river water concentration (μg/L) and mT is the median effect (i.e. toxicity) concentration (μg/L).

2.6. Modifying factors applied to the first tier risk ranking

The first tier analysis described above could be considered as an uncritical compilation of the ecotoxicity and river water concentration datasets. The tier two or refined risk analysis used the same data as used in tier one, but became more selective by excluding certain values in a search for greater refinement or realism. Thus, for all tier two analysis the following items were applied:

- Environmental measurements collected before the year 2000 were excluded. Due to economic changes or the introduction of legislation/control measures, older chemical measurements are likely to be less relevant for the current or future risk of chemicals. An exception was made for LAS, since although most of the available data were from 1995 to 1998, it is known to remain a popular surfactant, so concentrations are likely to be the same or conceivably higher.
- As the study was focused on the UK, all non-UK measurements were also excluded, which meant some compounds fell out entirely from the risk-ranking, thereby reducing the number of investigated chemicals to 60.

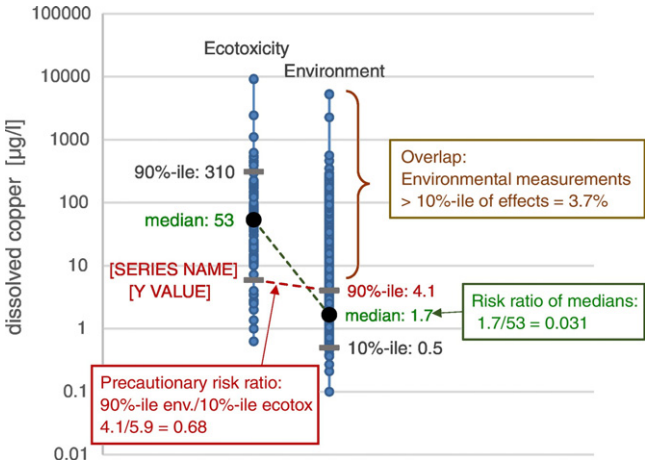

Fig. 1. Example of generating a risk ratio for one chemical, in this case copper. All the ecotoxicity data are plotted on the left and the environmental concentrations on the right. The relative risk value is taken from the ratio of the two median values. Also shown is the more precautionary approach using the ratio between the highest 90%ile of the environmental measurements and the lowest 10%ile of the toxicology data. The percentage of measurements above the 10%ile toxicology value is also shown.

Within these tier two restrictions a number of 'experiments' were carried out to examine what impact they might have on what the highest risk ranked chemicals might be, as follows:

- *Excluding sub-lethal effects data*

With lethality there is no ambiguity, or controversy, over the end-point and the danger is clear. It is possible to argue that chemicals of lethal toxicity are therefore of the most concern. Thus, in this exercise all sub-lethal toxic effects were excluded. In addition, lethal (acute) toxicity might be considered most appropriate for chemicals where only transient exposure would occur, such as for some insecticides.

- *Including only sub-lethal effects data*

As the study of ecotoxicity has become more sophisticated and scientific techniques have developed, the ability to measure sub-lethal effects has become possible. Sub-lethal effects can include molecular, biochemical, physiological, reproductive and behavioural effects on organisms. These sub-lethal effects typically occur at concentrations below those required to kill the organisms, thus, basing a risk ranking on them may be considered a more precautionary approach.

- *Excluding chemicals with a bioconcentration factor (BCF) below 500*

There is considerable consensus on the undesirability of lipophilic chemicals entering the environment, since they can accumulate in wildlife, with the highest concentrations being found in wildlife at the top of the food chain. A simple start to gauging the potential for a chemical to have this undesirable property is to record its bioconcentration factor (BCF). The BCF is calculated by dividing the 'steady-state' wet tissue concentration by the 'steady-state' water concentration of a particular substance (Chapman et al., 1996). BCF values were collected from the literature and the median BCF values were used to select the most bioaccumulative chemicals from this selection (Table S5). All chemicals with a BCF factor below 500 were omitted, leaving only 21 of the 60 chemicals in tier 2 to be ranked in the normal manner by comparing their median ecotoxicity value with median river water measured value.

- *Ranking chemicals on the degree of overlap of ecotoxicity and environmental concentration data*

Using the medians as a comparator provides a robust method to compare the relative risk of chemicals. However, this does not reveal to what degree any of the chemicals might actually be harming wildlife. Particularly from the regulators point of view, such information could be seen as the most valuable in any prioritisation exercise. Given concerns over the potential quality of reports on the most sensitive effects on wildlife and also for the highest measurements in rivers (the extremes) with their danger to mislead, the ratio of the lowest 10%ile of the ecotoxicity data to the upper 90%ile of the environmental measurement data was compared (note: not the extremes of minimum and maximum). An alternative is to provide a percentage for the number of environmental concentrations which exceed the lowest 10%ile of the ecotoxicity data (however, this can only be provided for the few chemicals where this overlap actually occurs).

2.7. Comparing ranking methods from this study with established EQS used in regulation

As a way of comparing how the risk ranking method developed here might differ from those chemicals that would be the focus of regulatory concern, the EQS applied in the UK by the Environment Agency of England and Wales were collected (see Table SI 2). The EQS applied in the UK are mostly EU wide EQS specified under the European Water Framework Directive (European Union, 2013), but a few chemicals have additional UK standards summarized in http://evidence.environment-agency.gov.uk/ChemicalStandards/ (website checked 4/2017, last updated 4/2011). Out of the 71 chemicals of concern examined in this study, 36 had an EQS for the annual average concentrations, one (glyphosate) had a proposed EQS and the relevant drinking water standards (0.1 μg/L) for seven pesticides and for aluminium (200 μg/L) were applied as they do not have an EQS.

3. Results and discussion

3.1. Ranking

The objective was to risk rank chemicals on the basis of their potential to harm the greatest range of wildlife at the widest range of locations. To help illuminate the approach used, the ecotoxicity data and reported environmental measurement data for one chemical (copper) are shown as an example, showing the different ratios or percentage overlap on which the ranking could be based (Fig. 1).

The analysis for all the chemicals is shown in Fig. 2. Using this type of figure makes it possible to gauge at a glance the amount and range of ecotoxicity and environmental data available. With this representation it is also possible to see overlaps within individual chemicals, where some reported environmental concentrations exceed reported effect concentrations. It will be noted that many of the metals (in blue) are at the high risk end and the pharmaceuticals (in purple) at the low risk end of the graph (see Table 1 or S3 for key). We can then add greater realism by eliminating chemicals not measured in the UK at or before the year 2000, reducing the total number of chemicals assessed now to 60 (Fig. 3). In this case the risk ranking was done on the basis of using all the ecotoxicity data (lethal and sub-lethal) and for comparison the ratios obtained using only sub-lethal ecotoxicity data for each chemical are also shown. Using either all the ecotoxicity data or only the sub-lethal ecotoxicity data produced a similar result, with eight of the top ten chemicals being the same in both cases, though not in the same order (Fig. 3, Table 2). This is an important observation as some consider sub-lethal or chronic ecotoxicity data as the most critical and that it should not be mixed with acute data for fear of missing crucial sensitive species or end-points. However, another school of thought is that there is a continuum between the two (Lange et al., 1998). In this case, by using all ecotoxicity data or sub-lethal ecotoxicity data alone, the ranking was similar. It is noteworthy that most of the selected pharmaceuticals, with the exception of EE2, do not rank as a high risk using this approach. For example, diclofenac appears to be a 10,000-times lower risk than Cu. Similarly, engineered nano-Ag and nano-ZnO are 500 times less of concern than LAS (Fig. 3, Table S4).

Another concern may be that toxicity in the real world is modified by other factors, for example water hardness. When the influence of typical lowland river chemistry (based on the UK rivers Thames, Trent and Calder) was examined via a biotic ligand model, it was found that the toxicity of Cu decreased 3-fold, Mn 2-fold and Zn by 14%, values which are all less than an order of magnitude. These relatively modest changes suggested that introducing extra realism would not drastically change the high ranking position of the metals. It must be acknowledged that exposure via water may not be the most relevant route for very hydrophobic substances, but it is nevertheless possible to focus on the highly bioaccumulative substances for a separate risk ranking. Restricting the choice of chemicals to those with BCF > 500 unsurprisingly brought a lot more organics, such as POPs and insecticides, into the top ten, yet Cu and Zn, which were highlighted in the other approaches, are top of this grouping too (Table 2, Table S4). When BCF is not a consideration, then some of the more polar chemicals join the metals to become ranked highly, such as LAS, triclosan and EE2 (Tables 2 and S4).

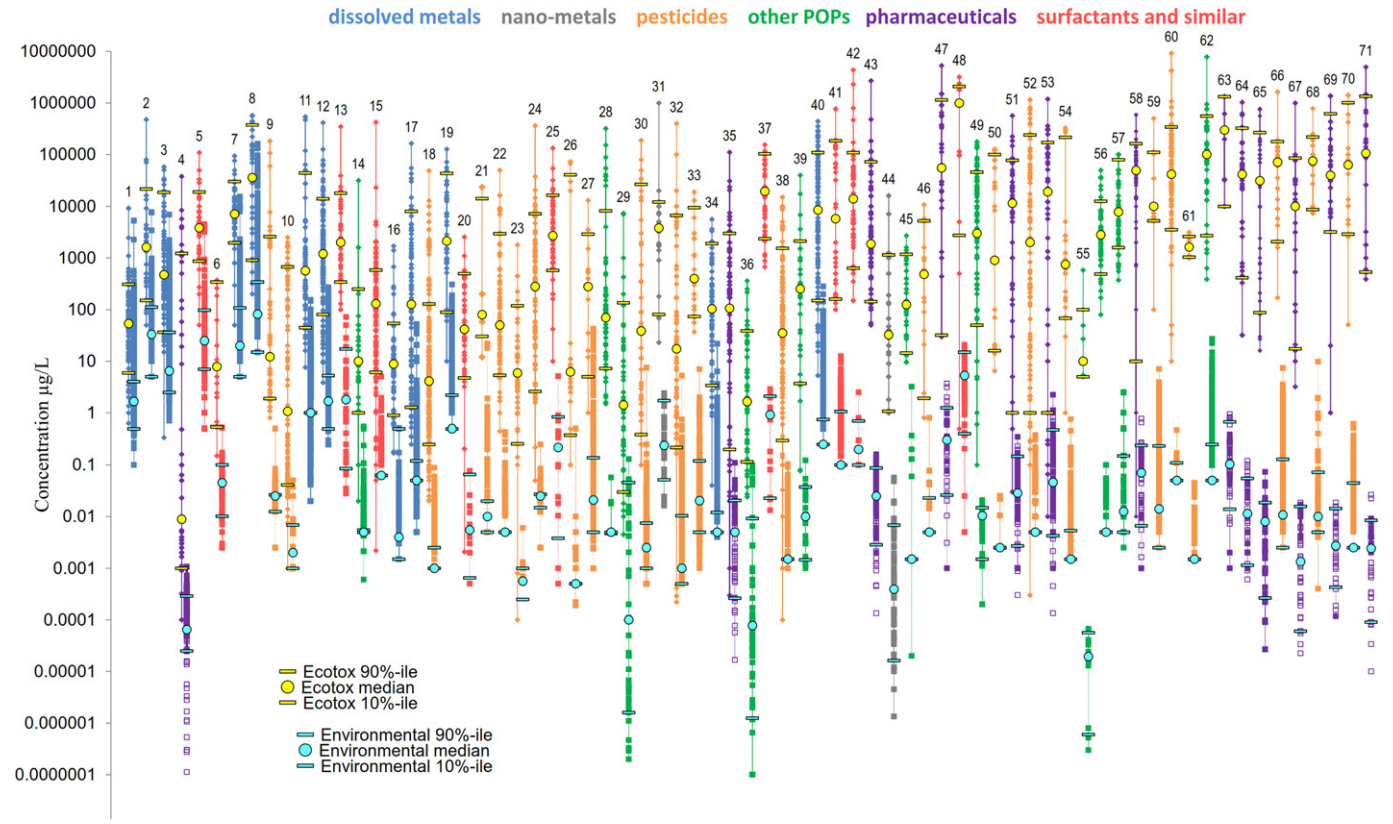

Fig. 2. Paired data of all the collected ecotoxicity effect and measured river concentrations for 71 chemicals. For each chemical the two datasets are plotted side by side in the same colour with ecotoxicity on the left and environmental data on the right. The 10%ile, median and 90%ile for each dataset are also marked. For the pharmaceuticals and nanoparticles both modelled (open symbols) and measured (closed symbols) data are included. The highest risk chemicals are on the left and the lowest risk on the right. The colours refer to the chemical groups as defined in Table 1 and the numbers are the rank on the basis of median environmental/median ecotoxicity concentrations. See Tables 1, and S2 for key and Fig. S1 for a larger scale.

3.2. Comparing ranking methods from this study with established EQS used in regulation

There may be a criticism that going back to first principles to risk rank threats to the environment from chemicals is simply not necessary, since this topic is already in the capable hands of regulators. As an exercise to test this, the current UK EQS concentrations were used as a comparator to the median UK river water concentrations collected in this study (Fig. 4). Whilst the two different approaches would both put copper, zinc, manganese and the pesticide methomyl in the top 10, for the other places the methods diverge (Table 2, and Fig. 4). If median UK river concentrations were compared to the EQS (or drinking water standards) a top 10 (or 12, due to three pesticides ranking equally in 10th place) prioritised by a regulator would include benzo[*a*]pyrene (B[*a*]P), PFOS, TBT, 5 metals, the PAH fluoranthene and another three pesticides. Both B[*a*]P and PFOS environmental values exceed the EQS by a considerable margin so would appear to be the greatest threat to UK wildlife using this approach (Table 2). In contrast, the median ecotoxicity value, as proposed in this study, would prioritise aluminium and iron together with EE2, LAS and triclosan in a top 10. Similarly, the ratio of highest 90th percentile environmental measurement was compared to the EQS value with that from the highest 90th percentile environmental measurement and lowest 10th percentile ecotoxicity value would prioritise the metals (Figs. 3 and S3). This is not a dry academic activity. If the regulatory approach with their EQS values is correct, then the environment is protected, but if not, then there could be damage to the environment from chemicals that are receiving insufficient attention. The method proposed here cannot of course claim to be pre-eminent, but it suggests that regulators should carefully examine and review whether some chemicals have been unfairly neglected (and would remain so) because of the vagaries of prioritisation methods where the focus is on hazard.

3.3. Limitations

3.3.1. Data availability

The confidence we can put in this or any risk ranking/prioritisation exercise is limited by the quantity and quality of available data. It will be noted from Table S2 that a lot of the measurements for some chemicals were below the LOQ, leading in some cases to the medians being half the LOQ. Clearly this is not ideal. But this is likely to overestimate risk rather than underestimate it, which is preferable. With some chemicals we are fortunate to have both considerable ecotoxicity and measurement data, but this is not the case for all. This project required considerable manual data inputting and the possibility of some errors being introduced cannot be dismissed. We would urge others to repeat our method for the same chemicals using their own monitoring data or for other chemicals, to see whether different conclusions are reached. This risk ranking method is now being applied to rivers in China (Su et al., 2017; Zhang et al., 2017a; Zhang et al., 2017b). The risk ranking here was limited to only 71 chemicals, which we selected as of high concern out of the many thousands of chemicals that are present in our rivers. Developments in techniques can change the information available per chemical. In addition, even if sufficient information is available, the quality of reported information is currently not assessed and may influence the ranking of a chemical. However, in this case, given that the medians or the 10th and 90th percentile were used, this is unlikely to be a major issue. It will be noted that metals featured strongly as being of the highest risk, so a more thorough re-analysis of their position following careful bioavailability considerations will be necessary.

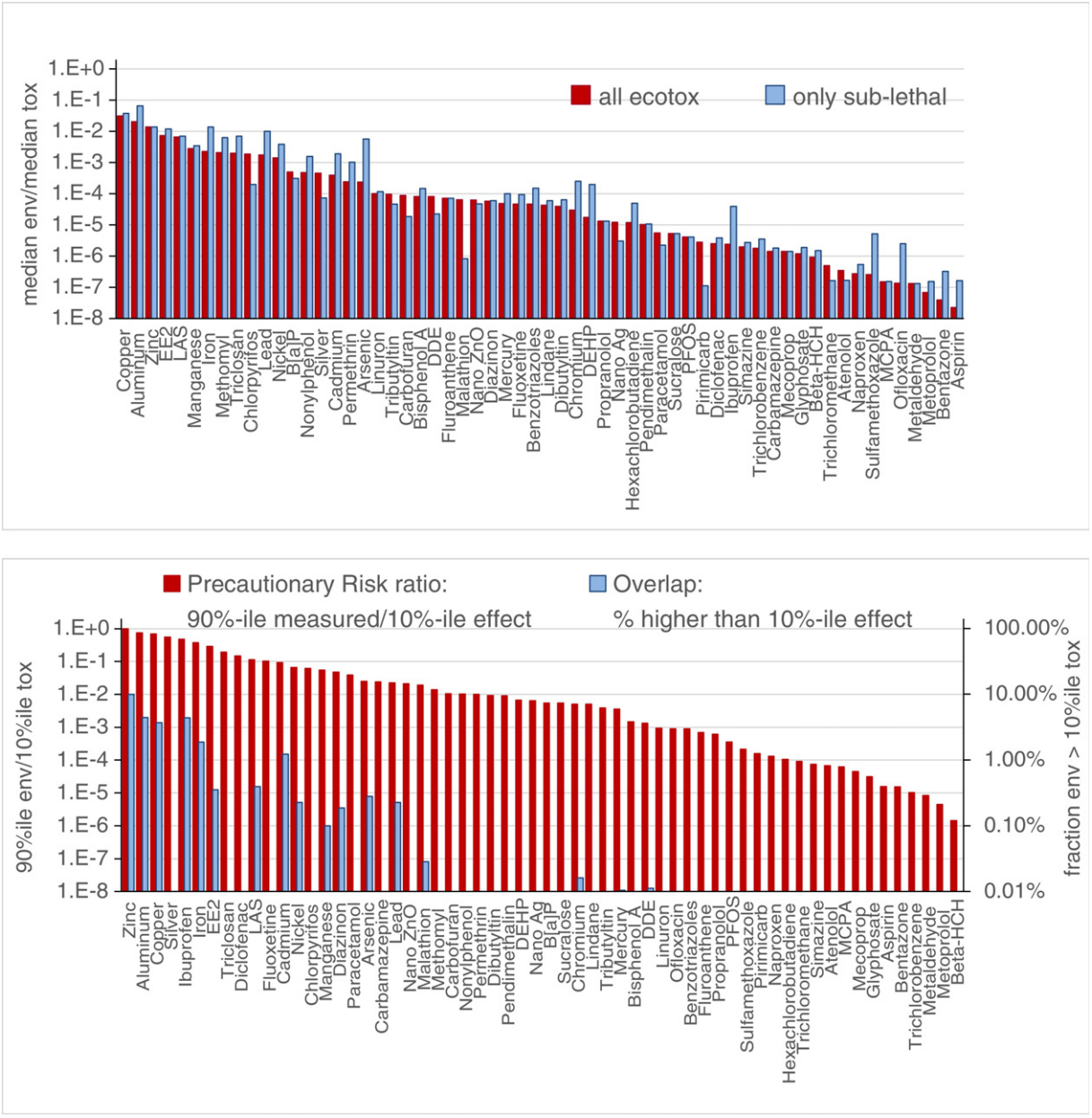

Fig. 3. Risk ranking of all 60 chemicals in tier 2. Top: based on the ratio of the median environmental concentrations versus the median of all ecotoxicity data, compared with the ratios if only the sub-lethal ecotoxicity data are used. Bottom: Assessing the potential impact on UK aquatic wildlife by either comparing the ratio of the highest 90%ile environmental data with the lowest 10%ile ecotoxicity data or by investigating what percentage of environmental measurements are above the 10%ile ecotoxicity data point. Only UK measured data since 2000 were considered. Note the same data are plotted on a natural scale and larger format as Fig. S2.

Furthermore, chemicals for which we have limited or no ecotoxicity or measurement data cannot be assessed by this method (or other methods). As it is not feasible to test every chemical on every endpoint in every species, we would argue that the best way to prepare for unexpected chemical problems is to maintain consistent wildlife monitoring schemes. That way population level effects can be spotted early and hopefully the cause(s) found and eliminated before it is too late (Johnson and Sumpter, 2016).

3.3.2. Exposure route

The recognised method for assessing aquatic ecotoxicity is to utilise laboratory tests to expose organisms to the test chemicals in the water. There are two problems with this approach where hydrophobic pollutants are concerned; firstly, the realistic route of exposure in the wild would largely be via contaminated food and possibly sediments (not water), and secondly, the water concentration of the chemical, due to its high partition coefficient, would be very low and difficult to measure with confidence. Thus, both the hazards and presence of such POPs may be underestimated.

3.3.3. Detection limits

There are problems in dealing with highly toxic but rarely detectable chemicals. In this analysis insecticides like methomyl and chlorpyrifos have entered the top ten, yet they are rarely present above the LOQ (see Table S2). Arguably, most monitoring networks are not 'fit for purpose' to report concentrations of these chemicals, due to their often limited use and short-term applications in agriculture. Although there is some evidence that such insecticides may be harming invertebrates in streams running through agricultural areas (Liess and von der Ohe, 2005) they are unlikely to be a problem in the main parts of the river network due to dilution.

Table 2
The top ten most highly ranked chemicals of concern using the refined approaches (tier 2). All the refined rankings shown here were based on UK river measurements after 2000 (the figures in parentheses are the risk ratios). The approaches favoured in this study are highlighted in grey.

Relative ranking	Ratio of medians using all ecotoxicity data	Ratio of medians using sub-lethal ecotoxicity data only	Ratio of medians using lethal ecotoxicity data only	Ratio of medians for BCF>500, using all ecotoxicity data	Precautionary ratio 90%ile environment/ 10%ile all ecotoxicity data	The % of environment values exceeding 10%ile of all ecotoxicity	Ratio of UK EQS and median environmental concentration
1	Cu (0.0311)	Al (0.0663)	Cu (0.0252)	Cu (0.0311)	Zn (0.993)	Zn (9.9%)	B[a]P (29.4)
2	Al (0.0205)	Cu (0.0377)	Al (0.0180)	Zn (0.0137)	Al (0.741)	Al (4.4%)	PFOS (18.8)
3	Zn (0.0137)	Triclosan (0.0203)	Zn (0.0137)	EE2 (0.0073)	Cu (0.681)	Ibuprofen (4.4%)	Tributyltin (2.82)
4	EE2 (0.0073)	Fe (0.0137)	Chlorpyrifos (0.0083)	Triclosan (0.0057)	Ag (0.556)	Cu (3.7%)	Cu**(0.06-1.66)
5	LAS* (0.0066)	Zn (0.0137)	LAS* (0.0046)	Chlorpyrifos (0.0019)	Ibuprofen (0.480)	Fe (1.9%)	Zn** (0.05-0.81)
6	Triclosan (0.057)	EE2 (0.0120)	Mn (0.0023)	Pb (0.0018)	Fe (0.377)	Cd (1.2%)	Fluoranthene (0.794)
7	Mn (0.0028)	Pb (0.0100)	Pb (0.0014)	B[a]P (0.00050)	EE2 (0.290)	LAS* (0.40%)	Mn (0.663)
8	Fe (0.0023)	LAS* (0.0070)	Ni (0.0013)	Ag (0.00045)	Triclosan (0.191)	EE2 (0.35%)	Cd (0.200-0.625)
9	Methomyl (0.0021)	Methomyl (0.0063)	Fe (0.0009)	Cd (0.00040)	Diclofenac (0.147)	As (0.28%)	Ni (0.423)
10	Chlorpyrifos (0.0019)	As (0.0057)	B[a]P (0.0006)	Permethrin (0.00024)	LAS*(0.113)	Ni (0.23%)	Methomyl, Carbofuran, Malathion (0.250)

*Although not measured in the UK after 2000, LAS was included as it has not been regulated and is still widely used, thus the environmental measurements used here were UK pre-2000.
**EQS depends on water hardness.

3.3.4. Considering mixture effects

It is often stated that a chemical by chemical analysis of risk to the environment ignores mixture effects. The problem at first seems open-ended and unlimited, apparently magnifying many times the chemical challenge we thought we faced. Typically, mixture effect (concentration addition) curves which identify the relative contribution of the chemicals in the group are dominated by only one or two of the chemicals present (Backhaus, 2014). When considering the results here, such as those shown in Fig. 3, there is a strong indication that doing mixture studies on, for example, several of the pharmaceuticals assessed here, would be irrelevant to the protection of the environment. A wiser course might be to do mixture studies with the top five or ten chemicals identified in the risk ranking (Table 2). This is because the apparent risk of, for example, Zn is 10,000 times greater than metoprolol! It would seem unlikely that a combination of pharmaceuticals alone in a mixture would elevate the risk by five orders of magnitude. So the results from this study might actually help focus research on which mixtures should be studied first.

4. Conclusions

4.1. Regarding the environment

We don't know whether, in reality, any of these chemicals are actually harming wildlife in UK rivers. But the dramatic difference in risk and hence potential impacts on wildlife revealed by this analysis of data seems to make a complex situation very much simpler. We cannot do everything. Focusing our control measures on metals such as Cu and Zn and organic chemicals such as LAS, triclosan, estrogens and some of the insecticides might improve the conditions for wildlife. Conversely, focusing on the low-risk chemicals such as glyphosate, metaldehyde, metoprolol, bentazone and diclofenac will not significantly help protect aquatic wildlife.

4.2. Observations regarding regulations

If we are to focus our limited resources correctly on the chemicals having the greatest impacts on aquatic wildlife, it is highly desirable to carry out some form of risk ranking exercise. The PNEC may be an unstable comparator because of its reliance on a limited number of studies and the variable range of assessment factors that may be employed. Bringing in many different additional hazard factors will inevitably introduce our own biases. It would be a mistake to muddle human health concerns with harm to aquatic wildlife. A focus on persistent and bioaccumulative chemicals may cause us to overlook polar and labile chemicals which are nevertheless toxic and ever-present thanks to pseudo-persistence. It will be noted that, for example, LAS features as having a relatively high risk in this analysis (albeit based on pre-2000 measurements), since we do not consider persistence or bioaccumulation, let alone carcinogenicity. It currently does not appear to be considered by any regulator as a chemical of concern but this study would suggest it deserves a second look. We argue for the pre-eminence of only two factors; toxicity (in all its forms) and exposure.

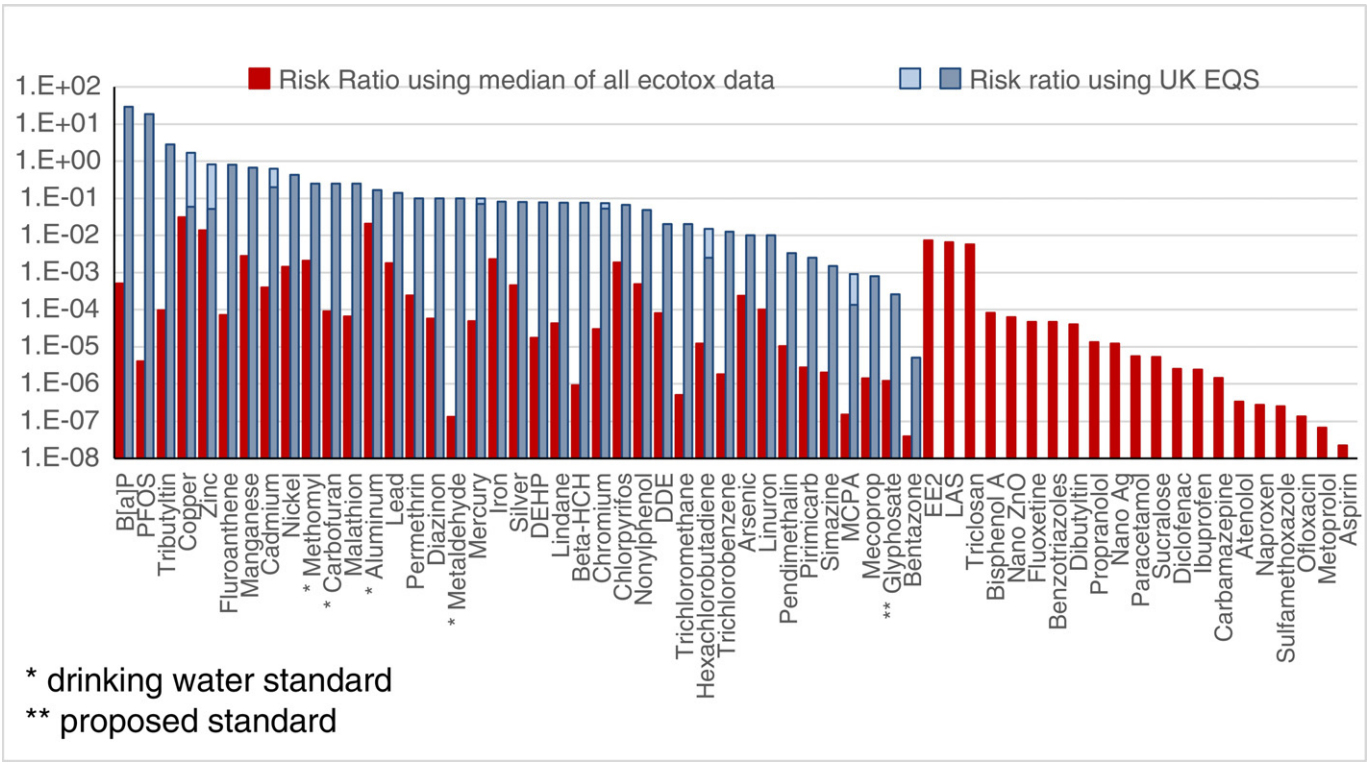

Fig. 4. Ranking by firstly comparing the median environmental concentration with the EQS (blue), and secondly by comparing the median environmental concentration against median ecotoxicology concentration as advocated in this study (red). Note, where two shades of blue are used for the EQS bar this illustrates that there is an EQS range depending on hardness/pH of the river water and the different shades represent minimum and maximum of that range. In the case of mercury and hexachlorobutadiene, the annual average water EQS no longer applies, so the two shades of blue represent the annual average EQS used before 2013 (European Union, 2008) and the maximum water concentration EQS which is still in use.

4.3. Funding to research the environmental effects of chemicals

We, and probably the wider public, would assume that the primary stimulus for funding research in this area is about protecting the environment. Whilst we should always examine the risk posed by new substances, it would be rational that our research efforts are proportional to the relative risk and evidence of harm. To take one example, if nano-Ag were to represent only 0.1% of the risk of say Cu, LAS or triclosan to our aquatic wildlife, what proportion of our scarce research funds should it receive?

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2017.05.039.

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