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Urban surface water quality and the potential of phytoremediation to improve water quality in peri-urban and urban areas in sub-Saharan Africa – a review

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ABSTRACT

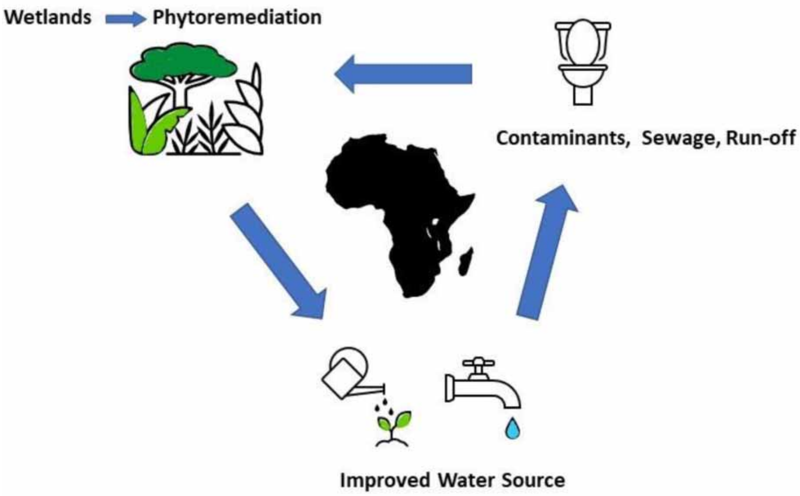
In 2017, 400 million people in sub-Saharan Africa (SSA) were still using unimproved drinking water sources, 80 million of whom relied on untreated surface water. Urban areas are vastly expanding all over the continent and many larger cities in SSA struggle to provide safely managed drinking water. Phytoremediation implemented in constructed wetlands (CWs) is a low-cost and sustainable alternative to highly costly and energy-consuming wastewater treatment plants. In addition, CWs offer the potential to be integrated into farming and aqua-culture systems and can therefore improve food quality and production. The most prominent pollutants in urban SSA surface waters and the pollutant removal efficiencies for microbial and chemical contaminations of different plant species were identified from the literature and the accumulation rates for Pb, Cr, and Cd were compared with each other. A strong focus was given to studies conducted in SSA or other (sub)tropical regions. This review identified a range of potential phytoremediators to treat contaminated surface water and highlights the need for further *in situ* studies in SSA. Plant species such as *Lemna minor, Ipomoea aquatica, Spirodela polyrhiza* and *Brachiaria mutica* show a high potential to phytoremediate the heavy metals Pb, Cr and Cd from surface water.

Key words: constructed wetlands, heavy metals, phytoremediation, sub-Saharan Africa, water quality

HIGHLIGHTS

- Review and analysis of water quality and wetland studies undertaken mainly in sub-Saharan Africa.
- Review and analysis of phytoremediators able to improve urban surface water quality impacted by microbial and heavy metal contamination in tropical settings.

GRAPHICAL ABSTRACT



1. INTRODUCTION

In 2017, 400 million people in sub-Saharan Africa (SSA) were still using unimproved water sources for drinking water, 80 million of whom relied on untreated surface water (WHO & UNICEF 2019). Currently, only around 61% of SSA has access to basic drinking water services (WHO & UNICEF 2019). For many people living in urban areas in SSA, collecting surface water directly from streams is their only access to freshwater, and as a result this untreated water is used for all their domestic purposes. A 2016 study of 30 SSA cities that depend on mainly surface water for their water supply found that one in five people face water quality risks (e.g., infections with diarrhoeal diseases) due to pollution and contamination of the water (The Nature Conservancy 2016).

Poor surface water quality can lead to human health issues and environmental damage. Surface water quality is affected by many factors, such as precipitation, climate, soils type, vegetation geology, flow conditions and human activities (Chaudhry & Malik 2017). Anthropogenic pollution from human activity is generally considered the most significant contributor to water quality in overcrowded urban areas. Surface water is of particular importance because it is generally more vulnerable to pollution than groundwater sources (Fawell & Nieuwenhuijsen 2003). Specific contaminants each have particular associated health risks. For example, consuming untreated or poorly treated water containing microbial contaminants can lead to illnesses such as cholera, potentially causing diarrhoea (Parsonnet *et al.* 1989; Ashbolt & Kirk 2006). Heavy metal contamination of water can lead to ingestion of heavy metals directly from drinking water or consuming contaminated food crops (Onakpa *et al.* 2018), which can cause life-threatening biotoxic effects on the human body (Duruibe *et al.* 2007). Contamination of surface water can also lead to contamination of groundwater (Winter *et al.* 1998), resulting in adversely affected groundwater drinking water supplies. This effect has the potential to further diminish the supply of safe drinking water in urban areas where water supplies are already strained. In addition, surface water is often the main source for irrigation water used for urban agriculture, potentially causing microbial contamination of the outside of the plant (e.g., leaves, fruits) and accumulation of heavy metals and other toxic elements in the inside of the plant (Gupta *et al.* 2010; Castro-Rosas *et al.* 2012).

Phytoremediation is the process of reducing the concentration of toxic contaminants in the environment using plants (Greipsson 2011). Traditional methods of removing contaminants, such as excavation and burial of contaminated soil, are very expensive and usually require heavy machinery (Purakayastha & Chhonkar 2010). As a result, traditional methods of soil remediation are estimated to cost a quarter of a million USD minimum per acre of land (Cunningham *et al.* 1995). Conventional methods for water treatment are therefore far too expensive for most places in SSA, which is home to some of the world's poorest and least developed communities. Phytoremediation, which is claimed to be 'a novel, cost-effective, efficient, environment- and eco-friendly, *in situ* applicable, and solar-driven remediation strategy' (Ali *et al.* 2013), may potentially be a very effective and beneficial approach for removing heavy metals, radionuclides and organic pollutants from the environment; based on this review of the relevant literature, however, it is currently used infrequently in SSA. A summary of phytoremediation techniques by Ali *et al.* (2013) is provided in Table 1.

Phytoremediation is an integral part of the function of CWs, which provide a range of other functions such as dilution, sedimentation, UV-radiation, coagulation or nutritional competition by microorganisms (Kadlec & Wallace 2009). Recent studies have demonstrated the potential of CWs for treating not only raw sewage, grey water or agricultural and urban run-off, but also polluted surface water (Ruan *et al.* 2006; Tang *et al.* 2009; Jia *et al.* 2014). CWs in low-income countries are an excellent low-cost and sustainable alternative to costly and energy consuming wastewater treatment plants which are often not affordable for the least developed countries (Denny 1997; Haberl 1999; Kadlec & Wallace 2009). CWs also have high pollutant removal rates, straight-forward operation and maintenance, high rates of water recycling, and the potential for providing significant wildlife habitats (Kadlec & Wallace 2009). In addition, CWs offer the potential to be integrated into farming and aqua-culture systems and can therefore improve food production. However, capacities to implement such techniques in low-income countries are often not available and adoption of this technology in low-income countries is still minimal (Denny 1997; Kivaisi 2001; Bojcevska & Tonderski 2007).

The bulk of the literature on phytoremediation is currently focused on temperate climates, although there are numerous phytoremediation studies on plants that grow in tropical climates. Furthermore, it is likely that additional phytoremediator species will be discovered, as there is a great wealth of plant biodiversity in tropical regions that are yet to be studied for phytoremediation potential (Guerra Sierra *et al.* 2021).

Monitoring the water quality of urban surface waters is necessary to identify dangerous contaminants and prevent illness and deaths from unsafe drinking water. Large numbers of untreated surface water sources are used for domestic water applications across SSA but are typically not assessed for water quality. The WHO *Guidelines for Safe Drinking Water* (WHO 2017) state that safe drinking water is required for all 'usual domestic purposes'. Therefore, this study presents a review of the currently available literature on urban surface water quality in SSA, with the aim of identifying the range of water contaminants while also identifying phytoremediators which have the potential to treat contaminated surface water.

2. METHODOLOGY

The literature review was conducted in two steps to assess literature with study locations in SSA for microbial and chemical urban surface water quality and potential phytoremediators. The first step of this review and meta-analysis was carried out by searching electronic search engines and databases: Google Scholar™, the Engineering Village Compendex™ and the University of Bath library. Keyword terms featured: 'surface water', 'water quality', 'assessment' and 'analysis' and multiple synonyms such as 'stream' or 'investigation', alongside the names of all SSA countries. Boolean operators were used to categorise the synonyms and searches of the same synonyms alongside subsets of SSA country names were iterated. The academic articles were then filtered by date (excluded if over 15 years old) and whether they featured readings in proximity to an urban or peri-urban setting. Studies were accepted if they mainly featured readings in urban and peri-urban locations. Results with separate wet and dry season readings were recorded as unique data sets. When the sampling dates were given but wet or dry was not specified, local rainfall data were used to find the season. Studies were rejected when there was a lack of specified units or when they only presented data as illegible figures. Quantitative results are summarised with the help of descriptive statistics such as box plots.

Table 1 | Summary of phytoremediation techniques (Ali *et al.* 2013)

Technique	Description
Phytoextraction	Accumulation of pollutants in harvestable biomass, i.e. shoots
Phytofiltration	Sequestration of pollutants from contaminated waters by plants
Phytostabilisation	Limiting the mobility and bioavailability of pollutants in soil by plant roots
Phytovolatilisation	Conversion of pollutants to volatile form and their subsequent release to the atmosphere
Phytodegradation	Degradation of organic xenobiotics by plant enzymes within plant tissues
Rhizodegradation	Degradation of organic xenobiotics in the rhizosphere by rhizospheric microorganisms
Phytodesalination	Removal of excess salts from saline soils by halophytes

During the second step, published literature was reviewed with the aim of identifying plants that could potentially remediate local water quality issues, especially when part of natural and CWs. In terms of pollutants, the focus was placed on microbiological and physiochemical contamination (total suspended solids (TSS), biological oxygen demand (BOD), chemical oxygen demand (COD)), and on chemical contamination by heavy metals, in particular Pb, Cd and Cr. The literature search employed the University of Bath library search engine and Google Scholar to find plants that have been identified as potential water quality phytoremediators in the tropics. This compiled list was not exhaustive but instead focused on the most commonly studied plants reported in the literature. Each plant species identified was then entered into the Kew Plants of the World Encyclopaedia (Kew 2021) to identify their native range and areas in which the species had been introduced. Any plants not native or not naturalised to multiple areas of SSA were removed from this review, as introducing potentially invasive plant species into an ecosystem can cause pervasive damage (Cronk & Fuller 2014). Once a list of plants was identified, phytoremediation studies for each plant were searched using again the University of Bath online library and Google Scholar. Selected studies conducted in any (sub)tropical country were included as long as the plant was native or naturalised in SSA. The keyword search included the botanical name of each plant species, along with keyword terms 'phytoremediation' and 'constructed wetland' and a list of the most problematic contaminants identified in the SSA water quality studies. Selected studies for each plant species were then reviewed to gain an understanding of the phytoremediation potential and efficiency with regards to removal and accumulation of microbiological and chemical contaminants.

3. RESULTS

3.1. Urban surface water quality in sub-Saharan Africa

In this section, a review of SSA surface water quality studies was conducted and the most critical water quality issues across the continent were identified. In total, the literature review identified 85 studies, comprising 131 data sets.

3.1.1. Locations of the analysed studies

Figure 1 shows a map of the distribution of the water quality studies found in this literature review across SSA. The map shows that a predominant number of studies was conducted in Nigeria (20) and Ethiopia (14), which are also the countries with the highest populations in SSA. Despite having the second and third highest urban populations in SSA (after Nigeria), South Africa and the Democratic Republic of the Congo only featured five and two urban surface water studies, respectively. Ghana, Kenya, and Uganda were the only other nations than those mentioned above to have five or more studies, despite having lower urban populations than, for example, the DRC and Angola.

3.1.2. Biological water quality

As shown in Figure 2, all studies measuring for presumptive *E. coli* found some level of contamination from it, recording amounts higher than the WHO drinking water guideline limit of none (zero). Presumptive faecal coliform measurements varied widely between 5 cfu/100 ml and 340,000 cfu/100 ml, with all measurements indicating some level of faecal contamination. Although many people rely on drinking untreated surface water, it should never be regarded as safe for human consumption.

Figure 3 compares the biological parameter results by dry and wet season. It appears that there is no strong correlation between season and biological parameter count. This further supports the notion that local factors have a greater influence on water quality than do seasonal factors. In the studies we reviewed for this paper, most of the average *E. coli* counts were below 100 cfu/100 mL whereas most of the faecal coliform and total coliform counts ranged between 100 cfu/100 mL and 10,000 cfu/100 mL indicating a high contamination caused by faecal matter. The average BOD measures were between 10 and 100 mg/L, also indicating a relatively high level of organic pollution.

3.1.3. Physicochemical water quality

Figure 4 shows that Cl and Turbidity were the only parameters which had a large amount of urban surface water locations with values that exceed health-based guidance from the WHO (2017). Although many of the chemical compounds such as NO_3, NH_4 and PO_4 had values which were within safe drinking water range, studies indicated that the levels of these chemical compounds would affect eutrophication in local ecosystems (Soares *et al.* 2011).

The presence of the wet and dry season appeared not to be a dominant factor on a macro-scale in terms of physicochemical water quality. However, every study that featured a wet and dry season comparison showed Cl to have a higher concentration

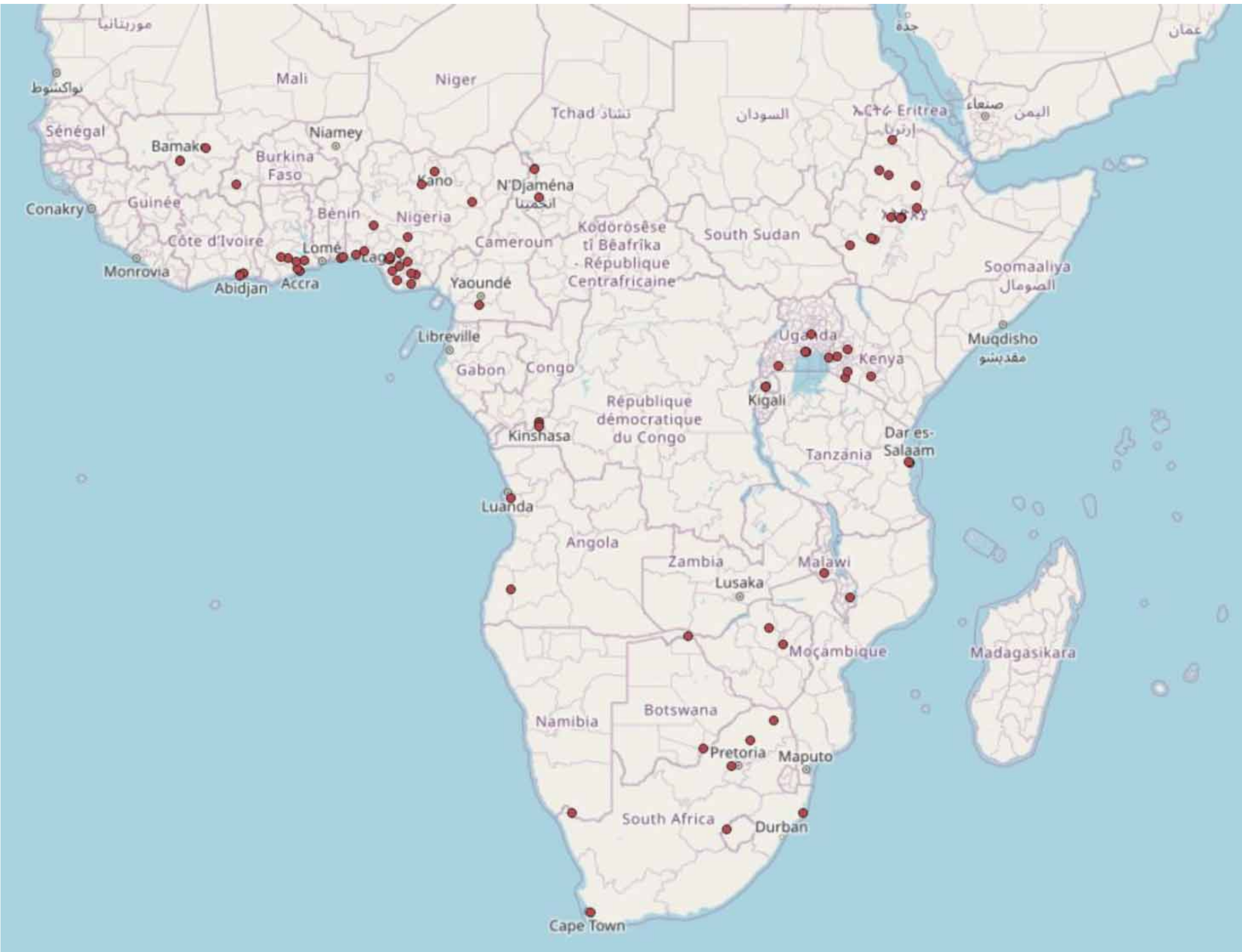


Figure 1 | SSA surface water study locations, marked by red dots.

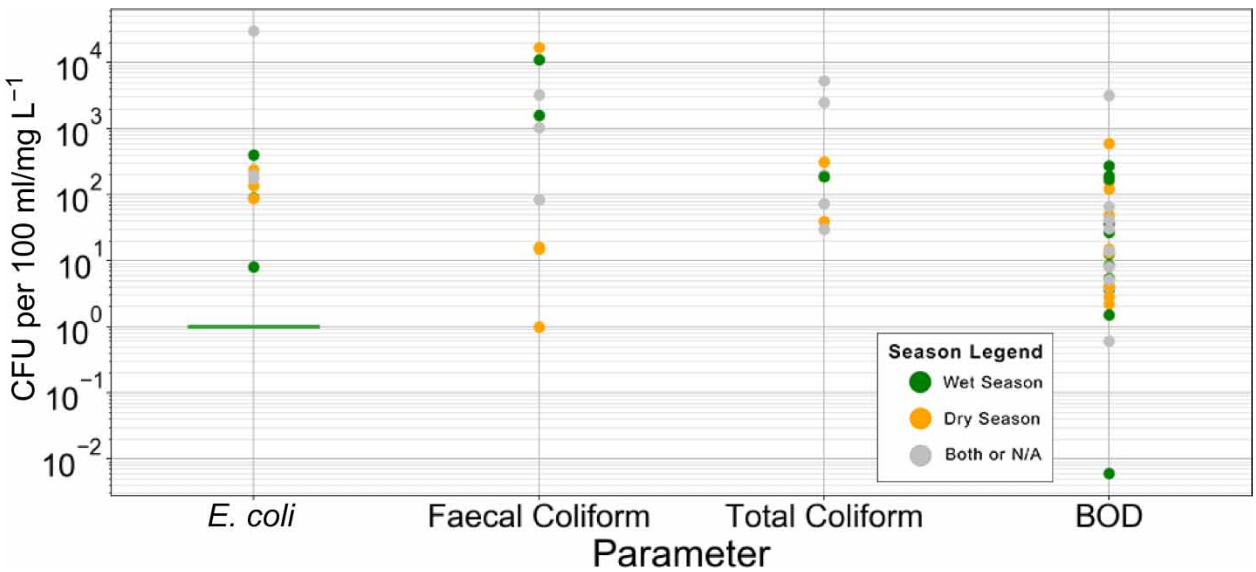


Figure 2 | Maximum concentration of various biological parameters for all studies collected, coloured by season. Green line defines the WHO limit of 0 cfu/100 mL. Please refer to the online version of this paper to see this figure in colour: http://dx.doi.org/10.2166/ws.2022.352.

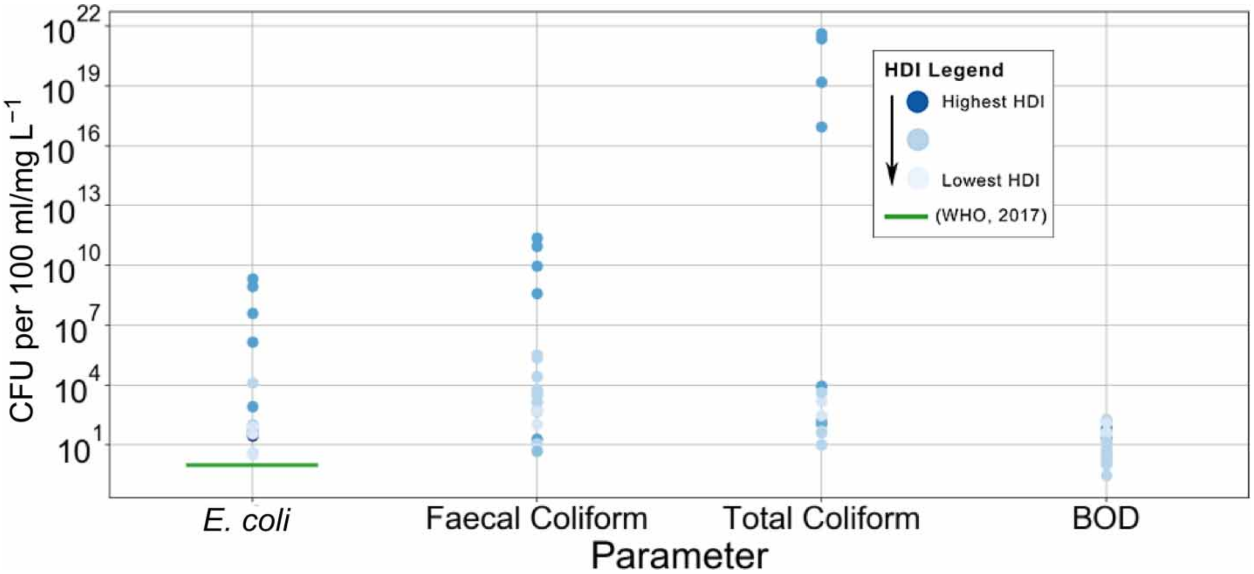


Figure 3 | Average concentration of various biological parameters for all studies collected. Green line defines the WHO limit of 0 cfu/100 mL. Please refer to the online version of this paper to see this figure in colour: http://dx.doi.org/10.2166/ws.2022.352.

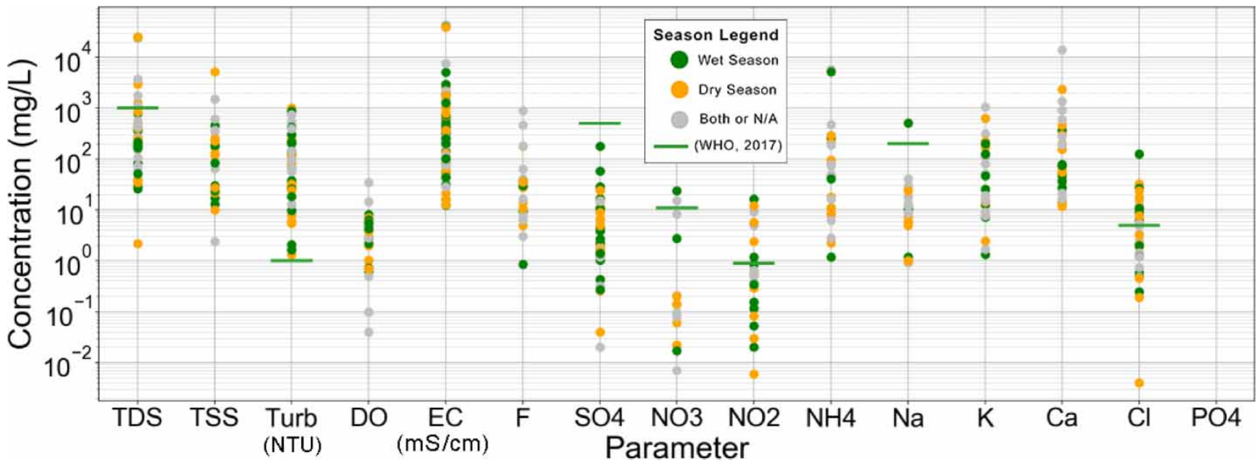


Figure 4 | Maximum concentration of various physiochemical parameters for all studies collected, coloured by season. Green lines define WHO limits. Please refer to the online version of this paper to see this figure in colour: http://dx.doi.org/10.2166/ws.2022.352.

in the dry season. Chemicals associated with fertilisers showed no clear trends within studies, probably due to differing levels of pollution caused by low solution volume in the dry season and high agricultural run-off in the wet season.

3.1.4. Extent of heavy metal pollution

Figure 5 highlights the average concentration of all recorded parameters for each study. It shows the extent of the current heavy metal surface water contamination problem in SSA by displaying metals exceeding the WHO guidelines (WHO 2017) at several surface water sampling points. In particular, Pb is a significant problem, with 31/34 studies exceeding the health-based WHO limit. Other heavy metals that frequently surpass health-based guidance include: Cd (19/30), Cr (16/29), and Ni (7/15). As with the biological and physicochemical parameters, it appears that there is no strong correlation between season and biological parameter count.

Figure 5 also highlights the maximum readings for all parameters in each study. The presence of wet or dry seasons was not a dominant factor in the quality of the water continentally. Seasonality appeared to be a more dominant determinant of water

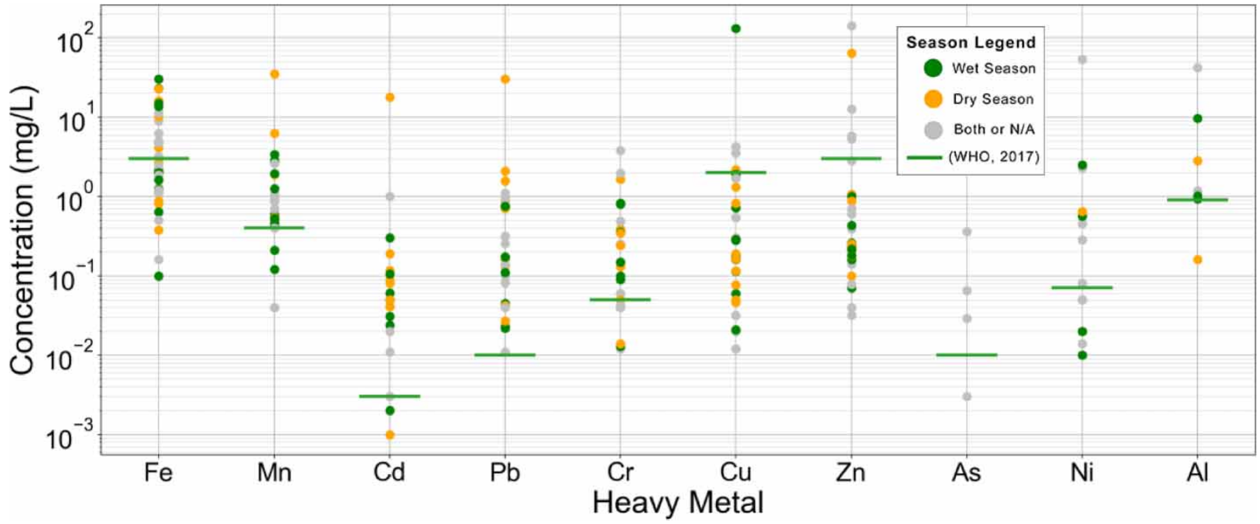


Figure 5 | Maximum concentration of various heavy metals for all studies collected, coloured by season. Green lines define WHO limits.

quality at a local level. However, on a continent-wide scale, other factors associated with the location of the study have a greater effect on pollution levels. For studies in which heavy metal results were given for both a dry and wet season, the dry season average concentration for the respective metal was higher than the wet season in 25 of 31 cases. Therefore, heavy metal concentrations are generally higher in the dry season at a given location; however, the effect is not great enough to be noticeable on the macro-scale. The number of studies exceeding the WHO guidelines are similar between the average concentration and maximum concentration data sets. For example, Pb exceeded the WHO guidelines in 31 of 34 studies for average concentration and 27 of 30 studies for maximum concentration. Cd (19/30 average, 18/30 maximum) and Cr (16/29 average, 17/26 maximum) showed similar trends. Parameters that showed a marked increase between average and maximum concentration were Mn (11/32 average, 25/31 maximum), Al (3/12 average, 6/8 maximum), and Fe (8/48 average, 15/35 maximum). Some parameters exceeded the WHO guidelines fewer times overall for maximum concentration than average concentration, due to some studies only measuring either average or maximum concentration and not both (WHO 2017).

3.2. Phytoremediation potential of native sub-Saharan African plants

This section reviews the current literature on phytoremediation to determine which plant species have the highest potential to remediate the most problematic contaminants identified above. For each water quality issue, a table of plants is provided (see Tables 2–5), detailing the findings of existing phytoremediation studies. Plant phytoremediation potential is generally measured by concentration of pollutants accumulated in various parts (e.g., roots, stem, leaves, fruits and seeds) of the plant and percentage removal of the pollutant from the water. Phytoremediation can also be measured by bioconcentration factors (BCFs), which is the ratio of concentration of a pollutant accumulated in the plant over the concentration of the pollutant in the environment. A high BCF indicates good phytoremediation potential because this suggests that the concentration accumulated in the roots is significantly higher than the environment (Landis *et al.* 2011). Translocation factors (TFs) indicate the proportion of pollutants accumulated in various parts of the plant over the pollutant concentration accumulated in the roots. A high TF to easily harvestable parts of a plant means that the species may be suitable for phytoextraction (Zhuang *et al.* 2007).

3.2.1. Phytoremediation of TSS, turbidity and microbial indicators and decrease of COD and BOD

Apart from heavy metals, there are other important water quality parameters that need to be assessed across SSA. These include, but are not limited to, COD, BOD, microbial pathogens, turbidity and TSS. Studies on the phytoremediation of these parameters are shown in Table 2. Most of the studies that looked at the remediation of physicochemical and biological remediation of water focused their analysis on (pilot-scale) CWs. The rationale for this is probably due to the fact that CWs provide a controlled environment and the process of phytoremediation can be much more easily investigated in comparison

Table 2 | Studies on the phytoremediation of physiochemical and bacteriological parameters by aquatic plant species native to SSA

Plant species	Habitat	Results	Environment	Citation
Arrowroot, *Thalia geniculata*	All SSA	BOD removal efficiency = 96%, COD removal performance = 69%	Container	Pincam *et al.* (2020)
		Maximum removal efficiencies for BOD5, TSS, FC and COD were 74.95, 92.56, 90.81 and 49.41%, respectively	CW	Katsenovich *et al.* (2009)
		Removal efficiencies of BOD, turbidity and bacterial biomass were 85, 88 and 91%, respectively	CW	López-Ocaña *et al.* (2018)
Water spinach, *Ipomoea aquatica*	All SSA but South Africa	Removal efficiencies of TSS and COD were 85.6 and 44.8%, respectively	Container	Aziz *et al.* (2020)
		Removal efficiencies in COD, BOD, TSS and organic carbon were 91, 75, 25 and 45%, respectively	*In situ* lagoon	Rane *et al.* (2016)
		Removal efficiencies for TSS and FC were 73 and 59%, respectively	CW	Fahim *et al.* (2019)
Water lettuce, *Pistia stratiotes*	All SSA	Removal efficiencies for turbidity and COD were 98.5 and 79.18%, respectively	Batch reactor	Schwantes *et al.* (2019)
		Removal performance of COD = 45.71%	CW	Qin *et al.* (2016)
		Maximum BOD, COD and total coliform removal efficiencies of 90.6, 85.5 and 97.9%, respectively were obtained in the 12th week of treatment	Container	Owamaha *et al.* (2014)
Water hyacinth, *Eichhornia crassipes*	South America, introduced to Africa	Removal efficiency of COD = 40.44%	CW	Qin *et al.* (2016)
Cattail reed, *Typha latifolia*	All northern hemisphere, East Africa, Congo, Nigeria, Morocco, Algeria, Tunisia	Removal efficiency of COD = 70.69%	CW	Rana & Maiti (2018)
		Removal efficiencies (no plant control in brackets): TSS = 64.3% (32.9%), BOD5 = 72.4% (49.6%), COD = 75.7% (49.2%), total coliforms = 88.8% (40.5%), faecal coliforms = 86.2% (34.6%), faecal streptococci = 82.1% (39.2%), *E. coli* = 89.5% (43.5%)	CW	Leto *et al.* (2013)
		Removal efficiency (after 48 h): *E. coli* 99.59%, *Salmonella enterica* serovar Typhimurium 99.04%	Container	Kipasika *et al.* (2016)
		Removal efficiency for total and faecal coliforms of 90%. COD reduced from 100.75 mg to 34.50 mg.	CW	Mashauri *et al.* (2000)
Common duckweed, *Lemna minor*	Most of SSA apart from West Africa	Removal efficiencies *L. minor* (no plant control in brackets): TDS = 68% (28%), turbidity = 97% (61%), COD = 92% (45%), BOD5 = 92% (41%), 400 cfu/100 ml faecal coliforms measured at the end of experiment	Container	Amare *et al.* (2018)
		Removal efficiencies of TSS and COD were 50.8 and 75% respectively	Container	Aziz *et al.* (2020)
Umbrella papyrus, *Cyperus alternifolius*	Most of SSA	Removal efficiencies (no plant control in brackets): TSS = 47.0% (32.9%), BOD5 = 64.8% (49.6%), COD = 66.6% (49.2%), total coliforms = 80.5% (40.5%), faecal coliforms = 79.4% (34.6%), faecal streptococci = 74.1% (39.2%), *E. coli* = 85.5% (43.5%)	CW	Leto *et al.* (2013)
			Container	Kipasika *et al.* (2016)

(*Continued.*)

Table 2 | Continued

Plant species	Habitat	Results	Environment	Citation
		Removal efficiency (after 48 h): *E. coli* 99.89%, *Salmonella enterica* serovar Typhimurium 99.31%		
Dwarf papyrus, *Cyperus isocladus*	Eastern and southern Africa	Reduction of *E. coli* populations from 4.5 log 100 ml^{-1} to 1.85 log 100 ml^{-1} in 48 h	CW	Neralla & Weaver (2010)
Papyrus sedge or paper reed, *Cyperus papyrus*	All SSA	Reduction efficiency for faecal coliforms of 98%	In-situ natural wetland	Kansiime & Mwesigye (2012)
Large reed grass, *Phragmites Mauritianus*	Southern and eastern Africa, Madagascar and Mauritius	Modelling nitrogen transformation and removal in horizontal subsurface flow CWs. Nitrogen removal was only 15% of incoming nitrogen load	CW	Senzia *et al.* (2004)
		Removal efficiencies (after 48 h): *E. coli* 98.88%, *Salmonella enterica* serovar Typhimurium 98.55%	Container	Kipasika *et al.* (2016)
		Removal efficiencies for *E. coli* strain ATCC 13706 (96%) *Cryptosporidium parvum* oocysts (98%) *Giardia lamblia* cysts (97%) total coliforms (84%) and faecal streptococci (89%).	CW	Reinoso *et al.* (2008)
Common reed, *Phragmites australis*	Worldwide	Reduction of BOD (80.69%), COD (69.87%) and faecal coliforms (95.61%)	CW	García-Ávila *et al.* (2019)
African arrowroot or Indian shot, *Canna indica*	South America, naturalised in most of SSA	Reduction of 68% BOD, 61.8% COD, 71.7% TDS, and 73.3% TSS	CW	Suganya & Sebastian (2017)
		Removal efficiencies of COD (87%) and for BOD5 (91%)	CW	Pinninti *et al.* (2022)
Para grass, *Brachiaria mutica*	North-, Central-, West Africa	Removal efficiency for faecal coliforms >64%	CW	Kaushal *et al.* (2016)
		Removal efficiency for COD of 60%	Container	Van *et al.* (2020)
Lemon grass, *Cymbopogon citratus*	South Asia, naturalised all over the tropics	Removal efficiencies for TSS of 93.7–97.3%, for COD up to 83% and for total coliforms 97%	CW	de Rozari *et al.* (2021)
Napier or elephant grass, *Cenchrus purpureus/ Pennisetum purpureum*	All SSA	Reduction of COD 872 mg/L^{-1} (influent) to 112 mg/L^{-1} (effluent) –87.16% and for BOD 210 mg/L^{-1} (influent) to 15 mg/L^{-1} (effluent) –2.86% from swine wastewater	CW	Klomjek (2016)

Table 3 | Studies on the phytoremediation of Pb by aquatic plant species native to SSA

Plant species	Habitat	Results	Environment	Citation
Arrowroot, *Thalia geniculata*	Tropical Africa	Maximum accumulation = 4.2 mg/kg	CW	Anning *et al.* (2013)
Water spinach, *Ipomoea aquatica*	All SSA but South Africa	Root accumulation range = 162.66 mg/kg to 9,950.78 mg/kg Stem accumulation range = 11.97 mg/kg to 2,792.37 mg/kg Leaf accumulation range = 43.18 mg/kg to 479.51 mg/kg	Container	Chanu & Gupta (2016)
		Removal performance = 27.72% (control = 7%), BCF = 0.23	Container	Saad *et al.* (2020)
		Removal performance = 27%	*In situ* lagoon	Rane *et al.* (2016)
Pondweed, *Potamogeton* spp.	All SSA but Benin, Togo and Djibouti	Average accumulation = 0.7 mg/kg	Natural samples	Shehata (2019)
Water lettuce, *Pistia stratiotes*	All of SSA	Maximum accumulation in roots = 0.309 mg/kg, BCF = 0.9 Maximum accumulation in leaves = 0.29 7 mg/kg, BCF = 1.53	Container	Kumar *et al.* (2019)
		Maximum accumulation = 14.9 mg/kg BCF range = 0.13–2.39	Container	Ergönül *et al.* (2020)
		Removal performance = 28%	CW	Kodituwakku & Yatawara (2020)
		Mean accumulation = 73.1 mg/kg Maximum accumulation = 96.2 mg/kg	Natural samples	Galal *et al.* (2018)
Water hyacinth, *Eichhornia crassipes*	West, East and Central Africa	Maximum accumulation = 0.3 ppm (0.3 mg/kg), BCF = 9.08	Natural samples	Ndimele & Jimoh (2011)
		Shoot accumulation = 115.7 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
		Removal performance = 46%	CW	Kodituwakku & Yatawara (2020)
		Average accumulation in shoots = 390 mg/kg Average accumulation in roots = 650 mg/kg	Natural samples	Agunbiade *et al.* (2009)
Cattail reed, *Typha latifolia*	Native to DRC, Kenya, Nigeria & Uganda	Removal performance = 71.38% Average accumulation in roots = 24.38 mg/kg Average accumulation in shoots = 19.38 mg/kg	CW	Rana & Maiti (2018)
		Removal performance = 53%	Container	Kumari & Tripathi (2015)
		Accumulation across whole plant = 5.6 mg/kg	CW	Anning *et al.* (2013)
Common reed, *Phragmites australis*	Most of SSA, excluding some of central and west Africa	Removal performance = 63%	Container	Kumari & Tripathi (2015)
		Average accumulation = 15.53 mg/kg Root to stem translocation factor = 0.6	Natural samples	Ahmad *et al.* (2014)
Coontail, *Ceratophyllum demersum*	Most of SSA, introduced to South Africa	Accumulation range = 9,805 mg/kg to 22,504 mg/kg Maximum BCF = 645.43	Container	Dogan *et al.* (2018)
		Average removal performance = 100%	Container	Foroughi *et al.* (2011)
		Accumulation range = 677.3 mg/kg to 1,772.8 mg/kg	Container	Chen *et al.* (2015)

(*Continued.*)

Table 3 | Continued

Plant species	Habitat	Results	Environment	Citation
		Removal performance range = 92–95% Accumulation range = 708 mg/kg to 2,816 mg/kg	Container	Abdallah (2012)
		Removal performance = 48.54% (control = 27.48%)	CW	Johnson *et al.* (2019)
		Average accumulation = 18 mg/kg, average BCF = 294.84	Natural samples	Ahmad *et al.* (2016)
Common duckweed, *Lemna minor*	Some of southern and western Africa	BCF range = 18.93–523.1	Container	Bokhari *et al.* (2016)
		Maximum accumulation = 2.58 mg/kg BCF = 86.26	Container	Khan *et al.* (2020)
		Maximum accumulation = 561 mg/g Maximum accumulation with nutrients = 129.8 mg/kg	Container	Leblebici & Aksoy (2011)
		Average accumulation = 704.5 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
Greater duckweed, *Spirodela polyrhiza*	Around half of SSA	Maximum accumulation = 330 mg/kg Maximum accumulation with nutrients = 68.7 mg/kg	Container	Leblebici & Aksoy (2011)
		Accumulation range = 4,446 mg/kg – 7,806 mg/kg Maximum removal performance = 93.19%	Container	Goswami *et al.* (2018)
Umbrella papyrus, *Cyperus alternifolius*	Most of SSA	Average accumulation = 304.6 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
Papyrus sedge or paper reed, *Cyperus papyrus*	All SSA	Max accumulation in roots = 61.6 mg/kg Max accumulation in stem = 6.2 mg/kg Water Pb concentration = 83 μg/L	Natural Samples	Sekomo *et al.* (2011)
African arrowroot or Indian shot, *Canna indica*	South America, naturalised in most of SSA	Pb was the least accumulated of 8 tested metals. Highest concentration was 60 mg/kg in the roots.	Container	Bose *et al.* (2008)
		Accumulation in roots after 21 days = 2,480 mg/kg Accumulation in leaf after 21 days = 12.2 mg/kg	Container	Cule *et al.* (2016)
Para grass, *Brachiaria mutica*	North-, Central-, West Africa	Maximum accumulation in shoots = 1,290 mg/kg Maximum accumulation in roots = 8,880 mg/kg	Container	Khan *et al.* (2018)
Lemon grass, *Cymbopogon citratus*	South Asia, naturalised all over the tropics	Average accumulation = 52.47 mg/kg TF > 1	Container	Gautam *et al.* (2017)
		BCF = 1.1, TF = 1.1 Accumulation in leaves = 64 mg/kg Accumulation in roots = 58 mg/kg	*In situ*	Israila *et al.* (2015)
Napier or elephant grass, *Pennisetum purpureum*	All SSA	Accumulation in roots = 255 mg/kg Accumulation in shoots = 61 mg/kg BCF = 1.8	Container	Ng *et al.* (2016)

Table 4 | Studies on the phytoremediation of Cd by aquatic plant species native to SSA

Plant species	Habitat	Results	Environment	Citation
Water spinach, *Ipomoea aquatica*	All SSA but South Africa	Maximum accumulation in roots = 1,200 mg/kg, Maximum accumulation in shoots = 138 mg/kg BCF range = 375–2,227	Container	Wang *et al.* (2008)
Pondweed, *Potamogeton* spp.	All SSA but Benin, Togo and Djibouti	Highest average accumulation from 6 sites = 5.1 mg/kg	Natural samples	Shehata (2019)
Water lettuce, *Pistia stratiotes*	All of SSA	Maximum concentration in roots = 0.214 mg/kg, BCF = 0.55 Maximum concentration in leaves = 0.247 mg/kg, BCF = 0.84	Container	Kumar *et al.* (2019)
		Maximum accumulation = 2.21 mg/kg BCF range = 0.08–1.24	Container	Ergönül *et al.* (2020)
		Removal performance = 32%	CW	Kodituwakku & Yatawara (2020)
		Mean accumulation = 2.2 mg/kg Maximum accumulation = 3.4 mg/kg	Natural samples	Galal *et al.* (2018)
Water hyacinth, *Eichhornia crassipes*	West, East and Central Africa	Average accumulation in shoots = 173.9 mg/kg Average accumulation in roots = 19.93 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
		Removal performance = 27.1%	CW	Kodituwakku & Yatawara (2020)
		Average accumulation in roots = 0.19 mg/kg Average accumulation in shoots = 0.5 mg/kg	Natural samples	Agunbiade *et al.* (2009)
Cattail reed, *Typha latifolia*	Native to DRC, Kenya, Nigeria & Uganda	Cd removal performance = 83.09% Average concentration in roots = 14.68 mg/kg Average concentration in shoots = 11.84 mg/kg	CW	Rana & Maiti (2018)
		Removal performance = 95–96%	Container	Putri & Moersidik (2021)
		Removal performance = 49%	Container	Kumari & Tripathi (2015)
Common reed, *Phragmites australis*	Most of SSA, excluding some of central and west Africa	Removal performance = 58%	Container	Kumari & Tripathi (2015)
		Accumulation in roots = 2.93 mg/kg Roots to shoots translocation factor = 0.87	Natural samples	Ahmad *et al.* (2014)
Coontail, *Ceratophyllum demersum*	Most of SSA, introduced to South Africa	Accumulation range = 354.17 mg/kg to 2,668.33 mg/kg Maximum BCF = 1,358	Container	Dogan *et al.* (2018), Foroughi *et al.* (2011)
		Removal performance range = 97.77–100%	Container	Foroughi *et al.* (2011)
		Average accumulation = 5 mg/kg, average BCF = 1,332.50	Natural samples	Ahmad *et al.* (2016)

(*Continued.*)

Table 4 | Continued

Plant species	Habitat	Results	Environment	Citation
Common duckweed, *Lemna minor*	Some of southern and western Africa	BCF range = 35.08–455.5	Container	Bokhari *et al.* (2016)
		Removal performance = 34.43%, BCF = 0.6	Container	Seifi & Dehghani (2021)
		Maximum accumulation = 4,734.56 mg/kg, BCF = 3,295.61 Removal performance = 42–78%	Container	Chaudhuri *et al.* (2014)
		Maximum accumulation = 0.332 ppm	Container	Khan *et al.* (2020)
		Removal performance range = 19–70% (control 5–17%)	Container	Amare *et al.* (2018)
		Accumulation in shoots = 880.4 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
Greater duckweed, *Spirodela polyrhiza*	Around half of SSA	Maximum accumulation = 7,711 mg/kg, BCF = 42–78%	Container	Chaudhuri *et al.* (2014)
Umbrella papyrus, *Cyperus alternifolius*	Most of SSA	Accumulation in shoots = 96.76 mg/kg, Accumulation in roots = 96.76 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
Papyrus sedge or paper reed, *Cyperus papyrus*	All SSA	Accumulation in roots = 4.2 mg/kg Accumulation in stem = 0.2 mg/kg Water Cd concentration = 18 μg/L	Natural sample	Sekomo *et al.* (2011)
African arrowroot or Indian shot, *Canna indica*	South America, naturalised in most of SSA	Accumulation in roots = 58.69 mg/kg Accumulation in shoots = 10.13 mg/kg	Container	Solanki *et al.* (2018)
		Accumulation in roots = 62 mg/kg Accumulation in shoots = 20 mg/kg	Container	Bose *et al.* (2008)
Para grass, *Brachiaria mutica*	North-, Central-, West Africa	Accumulation in roots = 734 mg/kg Accumulation in shoots = 67 mg/kg Accumulation in leaves = 59 mg/kg	Container	Ahsan *et al.* (2019)
Lemon grass, *Cymbopogon citratus*	South Asia, naturalised all over the tropics	Average accumulation = 79 mg/kg TF > 1	Container	Gautam *et al.* (2017)
		BCF = 1.3, TF = 1.9	*In situ*	Israila *et al.* (2015)
Napier or elephant grass, *Pennisetum purpureum*	All SSA	Accumulation in roots = 72 mg/kg BCF = 1.02–10.18, depending on growth stage and treatment	Container	Zhou *et al.* (2020)
		Accumulation in roots = 44.2 mg/kg Accumulation in shoots = 32.5 mg/kg BCF = 2.9	Container	Ng *et al.* (2016)
		Accumulation in roots = 206 mg/kg Accumulation in stem = 68 mg/kg Accumulation in leaves = 43 mg/kg	Container	Zhang *et al.* (2014)

Table 5 | Studies on the phytoremediation of Cr by aquatic plant species

Plant species	Habitat	Results	Environment	Citation
Water spinach, *Ipomoea aquatica*	All SSA but South Africa	Maximum accumulation = 23 ppm Removal performance = >75% (up to 28 ppm solution concentration)	Container	Weerasinghe *et al.* (2008)
		Removal performance = 50%	*In situ* lagoon	Rane *et al.* (2016)
Water lettuce, *Pistia stratiotes*	All of SSA	Removal performance = concentration increased	CW	Kodituwakku & Yatawara (2020)
		Mean accumulation = 311.6 mg/kg Maximum accumulation = 365.1 mg/kg	Natural samples	Galal *et al.* (2018)
		Removal performance = 77.3%	Container	Tabinda *et al.* (2020)
Water hyacinth, *Eichhornia crassipes*	West, East and Central Africa	Removal performance range = 63–84%	Container	Mishra & Tripathi (2009)
		Removal performance = 88%	Container	Singh & Sinha (2011)
		Removal performance = 80.9%	Container	Tabinda *et al.* (2020)
		Average accumulation in shoots = 21.78 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
		Removal performance = 23%	CW	Kodituwakku & Yatawara (2020)
		Average accumulation in shoots = 5.05 mg/kg Average accumulation in roots = 10.12 mg/kg	Natural samples	Agunbiade *et al.* (2009)
Cattail reed, *Typha latifolia*	Native to DRC, Kenya, Nigeria & Uganda	Removal performance = 73.64% Average accumulation in roots = 10.72 mg/kg Average accumulation in shoots = 11.46 mg/kg	CW	Rana & Maiti (2018)
		Removal performance = 57% *Typha latifolia* and *Phragmites australis* combined removal performance = 68%	Container	Kumari & Tripathi (2015)
Common reed, *Phragmites australis*	Most of SSA, excluding some of central and west Africa	Removal performance = 64% *Typha latifolia* and *Phragmites australis* combined removal performance = 68%	Container	Kumari & Tripathi (2015)
		Accumulation in roots = 11.06 mg/kg Accumulation in shoots = 8.82 mg/kg Root to stem translocation factor = 0.8	Natural samples	Ahmad *et al.* (2014)
Coontail, *Ceratophyllum demersum*	Most of SSA, introduced to South Africa	Removal performance = 27–84.3% Accumulation range = 175 mg/kg to 803 mg/kg	Container	Abdallah (2012)
		Removal performance = 56.4% (control = 33.1%)	CW	Johnson *et al.* (2019)
		Average accumulation = 17 mg/kg, BCF = 689.2	Natural samples	Ahmad *et al.* (2016)
Common duckweed, *Lemna minor*	Some of southern and western Africa	Removal performance = 26% (control = 5–17%)	Container	Amare *et al.* (2018)
		Removal performance range = 25–64%	Container	Nassar & Ibrahim (2021)
		Accumulation in shoots = 64.59 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
Greater duckweed, *Spirodela polyrhiza*	Around half of SSA	Maximum accumulation in plant = 2,850 mg/kg	Container	Liu *et al.* (2017)

(*Continued.*)

Table 5 | Continued

Plant species	Habitat	Results	Environment	Citation
Rice grass, *Leersia hexandra*	All of SSA	Maximum accumulation in natural samples = 2,978 mg/kg Hydroponics culture Cr(III) accumulation = 5,868 mg/kg Hydroponics culture Cr(VI) accumulation = 597 mg/kg	Natural samples and container	Zhang *et al.* (2007)
		Maximum accumulation in leaves = 5,430 mg/kg Maximum accumulation in stems = 1,956 mg/kg Maximum accumulation in roots = 40,599 mg/kg	Container	Zhang *et al.* (2009)
		Wetlands efficient at removing Cr(VI) influent at 5 mg/L	CW	Liu *et al.* (2015)
Umbrella papyrus, *Cyperus alternifolius*	Most of SSA	Accumulation in shoots = 20.56 mg/kg	Container	Shirinpur-Valadi *et al.* (2019)
Dwarf papyrus, *Cyperus isocladus*	Eastern and southern Africa	More alkaline solution allows faster uptake of Cr Removal of 44% on day 12	Container	Wirosoedarmo *et al.* (2020)
Papyrus sedge or paper reed, *Cyperus papyrus*	All SSA	Accumulation of Cr(III) in roots = 400 mg/kg Accumulation of Cr(VI) in roots = 340 mg/kg Removal efficiency of Cr(III) solution = 76.4% Removal efficiency of Cr(VI) solution = 67.4%	Container	Kassaye *et al.* (2017)
		Max accumulation in roots = 45.8 mg/kg Max accumulation in stem = 6.4 mg/kg Water Cr concentration = 115 μg/L	Natural samples	Sekomo *et al.* (2011)
Large reed grass, Phragmites Mauritianus	Southern and eastern Africa, Madagascar and Mauritius	Removal efficiency of wetland = 99.83% with 16.7 mg/kg influent	CW	Kaseva & Mbuligwe (2010)
African arrowroot or Indian shot, *Canna indica*	South America, naturalised in most of SSA	Accumulation in roots = 300 mg/kg Accumulation in shoots = 250 mg/kg Translocation factor decreases over exposure time	Container	Taufikurahman *et al.* (2019)
		Removal efficiency of wetland = 62.1, 71.4% with recirculation	CW	Maine *et al.* (2022)
		Accumulation in roots = 240 mg/kg Accumulation in shoots = 125 mg/kg	Container	Bose *et al.* (2008)
Para grass, *Brachiaria mutica*	North-, Central-, West Africa	Accumulation in roots = 130 mg/kg (with soil at 100 mg/kg) Accumulation in shoots = 1.2 mg/kg BCF = 1.28 when soil = 100 mg/kg	Container	Ullah *et al.* (2021)
		Accumulation in roots = 2,110 mg/kg Accumulation in shoots = 8.5 mg/kg Accumulation in leaves = 138 mg/kg (after 125 days)	*In situ*	Mohanty *et al.* (2012)
		Accumulation in roots (Cr(VI)) = 42 mg/kg Accumulation in shoots (Cr(VI)) = 22 mg/kg	Container	Kullu *et al.* (2020)
Lemon grass, *Cymbopogon citratus*	South Asia, naturalised all over the tropics	Average accumulation = 41 mg/kg TF > 1	Container	Gautam *et al.* (2017)

to natural wetlands *in situ*. Moreover, most studies used control ponds without plants to compare the impact to planted wetlands.

Microbial contamination is frequently one of the more critical water quality issues in SSA and globally. *Thalia geniculata* was reported by Katsenovich *et al.* (2009) to achieve a faecal coliform (FC) removal rate of 49.41% in a CW with an average hydraulic retention time of 9.8 days. *Ipomoea aquatica*, on the other hand, achieved an average FC removal rate of 59% across a 12-week experiment (Fahim *et al.* 2019). In one CW study, *Typha latifolia* achieved removal rates of 88.8, 86.2, 82.1 and 89.5% for TC, FC, faecal streptococci and *E. coli*, respectively (Leto *et al.* 2013). These removal rates were approximately double those of the control system that contained no plants. The same study also investigated *Cyperus alternifolius* as a potential microbial remediator, which achieved removal percentages of 80.5, 79.4, 74.1 and 85.5% for TC, FC, faecal streptococci and *E. coli*, respectively. Kipasika *et al.* (2016) archived in a batch experiment removal rates for *E. coli* of 99.89% with *Cyperus alternifolius* and 99.59% with *Typha latifolia*. Gersberg *et al.* (1989) reported that CWs can remove pathogens from secondary wastewater with efficiencies of 90–99%. However, with raw wastewater, further disinfection was necessary to achieve such removal rates. While reeds such as *Typha latifolia* and *Phragmites australis* and sedges such as *Cyperus papyrus* are well known for their phytoremediation properties, de Rozari *et al.* (2021) highlighted the removal efficiency of lemon grass (*Cymbopogon citratus*), with a removal efficiency for TSS of 93.7–97.3%, for COD up to 83% and for total coliforms of 97%.

3.2.2. Phytoremediation of lead

Out of all surface water studies reviewed in Table 2, Pb was measured in 25 data sets. The average concentration of Pb in SSA surface water studies exceeded the WHO guidelines for drinking water quality of 0.01 mg/L in 13 out of 25 data sets (WHO 2017). This indicates that throughout SSA Pb pollution is a widespread water quality issue. Studies of the phytoremediation potential of aquatic plants in remediating Pb are summarised in Table 3. The table is not exhaustive and includes mainly studies conducted in tropical settings. Figure 6 highlights the maximum accumulation of Pb by each plant species listed in Table 3 and includes only studies conducted in a controlled environment (e.g., greenhouse or CW).

Pb accumulation has been recorded in a wide number of aquatic plants. However, as is evident from Figure 6, some plant species such as *Brachiaria mutica*, *Cyperus alternifolia*, *Spirodela polyrhiza*, *Lemna minor*, *Ceratophyllum demersum*,

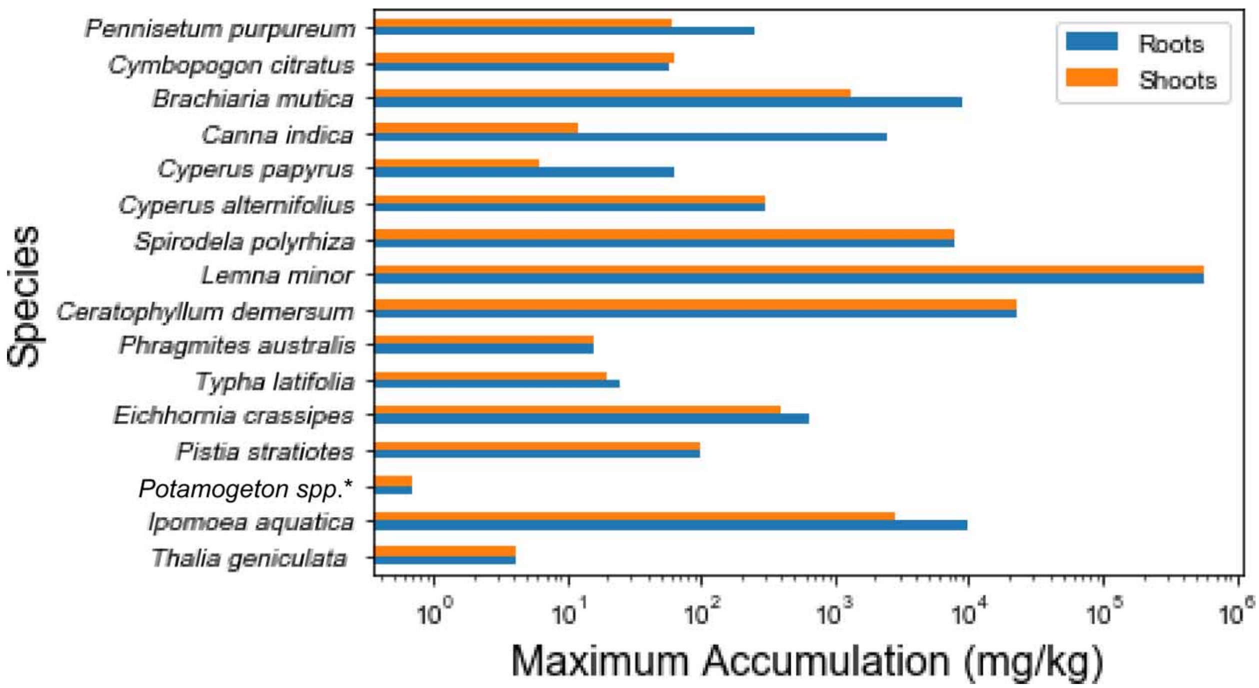


Figure 6 | Maximum accumulation of Pb by different tropical wetland plant species (*indicates average accumulation).

Eichhornia crassipes and *Ipomoea aquatica* are particularly well suited to remediate Pb from surface water sources. All these species demonstrated accumulation rates of 100 mg/kg Pb or higher.

In a 15-day plant pot study, *Ipomoea aquatica* was reported to accumulate up to 9,951 mg/kg and 2,792 mg/kg in the roots and shoots, respectively (Chanu & Gupta 2016), suggesting that it could be a potential hyperaccumulator. However, this study exposed the plant to very high concentrations of Pb, and significant toxicity was observed when treating the plant with over 20 mg/L Pb. In contrast, a recent study by Saad *et al.* (2020) reported a BCF of only 0.23, and a 27% removal rate of Pb when the plant was treated with lower concentrations of Pb. Furthermore, a test carried out in a 60,000 L constructed lagoon only achieved a 27% removal rate of Pb in 8 days (Rane *et al.* 2016). Therefore, *Ipomoea aquatica* may be suitable for remediation of heavily polluted soils; however, a low removal performance and low BCF when Pb pollution levels are low indicate that it might not be as effective in such situations.

Multiple studies reported that *Eichhornia crassipes* has the potential to remediate Pb (Agunbiade *et al.* 2009; Ndimele & Jimoh 2011). For example, Ndimele & Jimoh (2011) reported that for water sources with low pollution, the BCF is reasonable, recording a maximum BCF of 9.08. Alternatively, Agunbiade *et al.* (2009) reported a high TF for all metals (>1.6), indicating that high concentrations of toxic metals can be accumulated in the plant stems and leaves, and therefore Pb is likely to be easily phytoextracted by harvesting. A maximum concentration of 115.7 mg/kg was recorded in a recent plant pot experiment by Shirinpur-Valadi *et al.* (2019); however, natural samples of the species collected in two other studies report much lower Pb accumulations (Agunbiade *et al.* 2009; Ndimele & Jimoh 2011). This is perhaps due to much lower exposure to concentrations of Pb. One CW trial using *Eichhornia crassipes* found that the plant remediated 46% of the Pb concentration in diluted sewage sludge over a period of 28 days (Kodituwakku & Yatawara 2020).

Some studies have suggested that *L. minor* can accumulate high levels of Pb. However, at high levels the plant exhibits symptoms of toxicity (Leblebici & Aksoy 2011). The same study reported Pb accumulation of up to 561 mg/g, indicating that Pb accounts for over half the dry weight of the plant, which is somewhat doubtful. However, when other nutrients were added to the influent, the Pb accumulation decreased to a maximum of 128.7 mg/g and the phytotoxic effects on the plant were reduced. This implies that, in areas with a wide range of contaminants, the accumulation of Pb may be somewhat inhibited, due to the uptake of other nutritional and chemical compounds. Shirinpur-Valadi *et al.* (2019) reported a slightly lower average accumulation rate of 704.5 mg/kg in the plant's shoots. Pb toxicity to the plant at high concentrations is also supported by Khan *et al.* (2020), who reported phytotoxic effects when *Lemna minor* was exposed to concentrations of Pb exceeding 8 mg/L. Overall, a high BCF and reasonable high tolerance to Pb toxicity suggests that *L. minor* could be a particularly potent remediator of Pb.

Both studies reviewed on Pb accumulation in *Spirodela polyrhiza* indicated very high levels of accumulation (Leblebici & Aksoy 2011; Goswami *et al.* 2018). Goswami *et al.* (2018) reported a maximum Pb accumulation of 7,806 mg/kg and 93% removal rate with an initial solution Pb concentration of 0.91 mg/L. The removal rate was found to decrease with an increase in Pb concentration. Leblebici & Aksoy (2011) reported up to 330 mg/g Pb uptake, which reduced to 68.7 mg/g when other nutrients were introduced into the solution. Like *L. minor*, an increase in nutrients decreased the phytotoxic effects on *Spirodela polyrhiza*. This species also showed a high potential for Pb phytoremediation.

Ceratophyllum demersum has been reported to accumulate very high levels of Pb. A 2018 study by Dogan *et al.* (2018) reported a maximum Pb uptake of 22,504 mg/kg, while another study carried out by Chen *et al.* (2015) indicated uptake of up to 1,772.8 mg/kg Pb. Both studies reported accumulation greater than 1,000 mg/kg, suggesting that *C. demersum* is potentially a hyperaccumulator of Pb. It should be noted that both studies were laboratory experiments, where the plant was exposed to very high concentrations of Pb. A study on the remediation potential of *C. demersum* in treating municipal wastewater reported a 100% removal of Pb from both treated and untreated wastewater with low initial concentrations of 0.058–0.059 mg/L (Foroughi *et al.* 2011). This implies that *C. demersum* may be appropriate for Pb phytoremediation in both heavily and lightly polluted water sources. Greenhouse experiments with *B. mutica* and *Canna indica* showed high accumulations of Pb while also translocating very little from the roots, suggesting that they would perform well at phytostabilisation or rhizofiltration (Bose *et al.* 2008; Cule *et al.* 2016; Khan *et al.* 2018).

In summary of the studies we reviewed, several plant species showed high potential for remediation of Pb. Out of the studies reviewed, *C. demersum, B. mutica, Canna indica* and *L. minor* appear to have the highest potential for phytoremediation, as high uptake and BCFs have been reported consistently across several studies. *Spirodela polyrhiza* may also be another very effective remediator of Pb. *Phragmites australis, Pistia stratiotes, Eichhornia crassipes* showed some promise for remediation of Pb; however, *Ipomoea aquatica* may not be effective due to the low removal efficiency of Pb reported by multiple studies. *Pennisetum purpureum*, *Cyperus papyrus* and *Cymbopogon citratus* have not demonstrated high enough accumulation of Pb to be considered suitable for Pb phytoremediation. However, they have shown an ability to tolerate this heavy metal.

3.2.3. Phytoremediation of cadmium

Out of all surface water studies reviewed, Cd was measured in 30 data sets. The average concentration of Cd in SSA surface water studies exceeded the WHO guidelines for drinking water quality of 0.003 mg/L in 19 out of 30 data sets (WHO 2017). This suggests that throughout SSA, Cd pollution is a widespread water quality issue. Studies on the phytoremediation potential of aquatic plants in remediating Cd are summarised in Table 4. The table is not exhaustive and it mainly includes studies conducted in tropical settings. Figure 7 highlights the maximum accumulation of Pb by each plant species listed in Table 4 and includes only studies conducted in a controlled environment (e.g., greenhouse or CW).

Cd accumulation has been recorded in a wide number of aquatic plants; however, referring to Figure 7, it is evident that some plant species, such as *Pennisetum purpurium*, *B. mutica*, *Cyperus alternifolia*, *Spirodela polyrhiza*, *L. minor*, *C. demersum*, *Eichhornia crassipes* and *Ipomoea aquatica*, are particularly well suited to remediating Cd from surface water sources. All these species demonstrated accumulation rates of 100 mg/kg Pb or higher. These results are in line with the analysis done for Pb.

C. demersum was reported to have very high potential as a remediator of Cd. One laboratory study reported Cd accumulation of between 354 mg/kg and 2,668 mg/kg when exposed to concentrations of Cd ranging from 0.5 to 2 mg/L (Dogan *et al.* 2018). This suggests that *C. demersum* may potentially be a hyperaccumulator of Cd. Furthermore, a study on the remediation of municipal wastewater found that *C. demersum* remediated between 98 and 100% of Cd (Foroughi *et al.* 2011).

Many studies have reported on the Cd accumulation properties of *L. minor*. For example, Shirinpur-Valadi *et al.* (2019) reported a Cd concentration of 880 mg/kg in the shoots, and Chaudhuri *et al.* (2014) reported a maximum concentration of 4,734 mg/kg. Removal efficiencies varied between studies, with reported removal rates between 42 and 78% (Seifi & Dehghani 2021), 34.43% (Chaudhuri *et al.* 2014), and between 19 and 70% (Amare *et al.* 2018). One study reported that the bioconcentration factor of *L. minor* ranged between 35 and 455 mg/kg (Bokhari *et al.* 2016), suggesting that it can accumulate large amounts of Cd. Therefore, *L. minor* seems to be very promising as a Cd phytoremediator.

S. polyrhiza has been reported to accumulate up to >7,700 mg/kg Cd at an initial concentration of 3 mg/L, with removal rates varying between 42 and 72%, depending on the initial concentration of Cd (Chaudhuri *et al.* 2014). This shows that

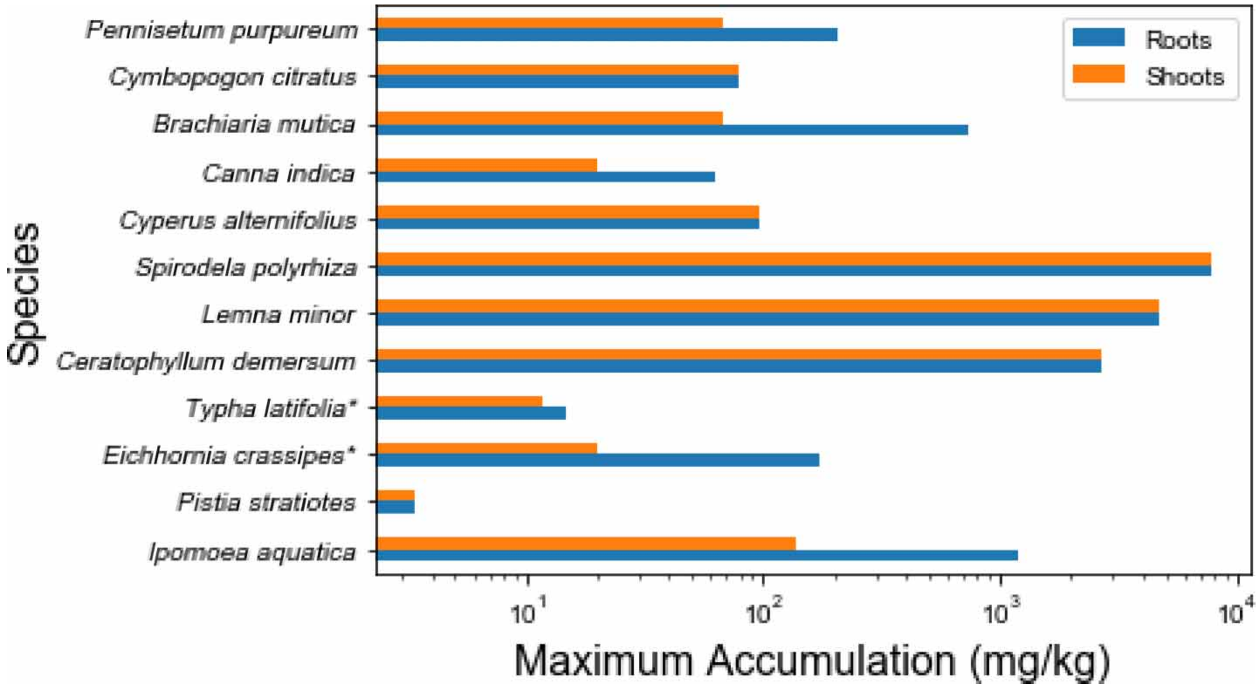


Figure 7 | Maximum accumulation of Cd by different tropical wetland plant species (*indicates average accumulation).

S. polyrhiza could be a potential hyperaccumulator of Cd. However, more studies are needed to support this hypothesis. *Cana indica*, *Cyperus papyrus* and *Cymbopogon citratus* generally accumulated low levels of Cd (all below 80 mg/kg; Bose *et al.* 2008; Sekomo *et al.* 2011; Israila *et al.* 2015; Gautam *et al.* 2017; Solanki *et al.* 2018). *Brachiaria mutica,* however, accumulated over 700 mg/kg of Cd in its roots (Ahsan *et al.* 2019) suggesting it would also do well in stabilising and absorbing Cd below the soil surface.

*Eichhornia crassipe*s was reported to accumulate up to 173.9 mg/kg and 19.93 mg/kg in the roots and shoots, respectively (Shirinpur-Valadi *et al.* 2019). However, in a study conducted *in situ*, the Cd uptake in the plants collected was much lower (Agunbiade *et al.* 2009), possibly due to the low availability of Cd for uptake. Both studies note high translocation factors to the stems of the *Eichhornia crassipe*s, leading to much higher concentrations of Cd in the shoots of the plants than the roots. A CW study noted a 27.1% remediation of Cd in 28 days (Kodituwakku & Yatawara 2020).

A study on *Ipomoea aquatica* reported that the roots and shoots of the species accumulated up to 1,200 mg/kg and 138 mg/kg Cd, respectively (Wang *et al.* 2008). This suggests that there is significant potential for this plant to accumulate Cd and to act as a hyperaccumulator. In contrast, *Pistia stratiotes* accumulated relatively little Cd in several studies, reporting a lower range of accumulation from 0.214 mg/kg to 3.4 mg/kg (Galal *et al.* 2018; Kumar *et al.* 2019; Ergönül *et al.* 2020). The BCFs reported were also low. Kodituwakku & Yatawara (2020) reported a Cd removal rate of 32% from sewage sludge in a CW over 28 days. *Pistia stratiotes* may have some long-term phytoremediation potential in a low polluted environment; however, it is unlikely to be the most effective species for Cd remediation.

Typha latifolia has been reported to have a high Cd removal rate in multiple studies. Separate studies on Cd removal from raw municipal wastewater, acid mine drainage and sewage effluent achieved 83, 95 and 49% removal rate, respectively (Kumari & Tripathi 2015; Rana & Maiti 2018; Putri & Moersidik 2021). Average concentrations of 14.68 mg/kg and 11.84 mg/kg were recorded in shoots and roots, respectively, in the municipal wastewater study (Putri & Moersidik 2021). Overall, studies suggested that *Typha latifolia* has a reasonably high remediation potential for Cd, although these results are difficult to compare with maximum accumulation rates.

3.2.4. Phytoremediation of chromium

Out of all surface water studies reviewed, Cr was measured in 29 data sets. The average concentration of Cr in SSA surface water studies exceeded the WHO guidelines for drinking water quality of 0.05 mg/L in 16 out of 29 data sets (WHO 2017). Studies on the phytoremediation potential of aquatic plants in remediating Cr are summarised in Table 5. The table is not exhaustive and includes mainly studies conducted in tropical settings. Figure 7 highlights the maximum accumulation of Cr by each plant species listed in Table 5 and includes only studies conducted in a controlled environment (e.g., greenhouse or CW).

Similar to Pb and Cd, Cr accumulation has been recorded in a wide number of aquatic plants. But referring to Figure 8, it is evident that some plant species such as *B. mutica*, *Cana indica*, *Leersia hexandra*, *Spirodela polyrhiza*, and *Pistia stratiotes* are particularly well suited to remediating Cd from surface water sources. All of these species demonstrated accumulation rates of 100 mg/kg Pb or higher.

B. mutica reported an accumulation in its roots of over 2,000 mg/kg in one study (Mohanty *et al.* 2012), although it was unable to replicate such high numbers in other experiments (Kullu *et al.* 2020; Ullah *et al.* 2021). The plant showed a strong preference for accumulating Cr in its roots. *Cyperus papyrus* also showed low amounts of translocation to upper parts of the plant (Sekomo *et al.* 2011), while providing a reasonable level of accumulation (up to 400 mg/kg) (Kassaye *et al.* 2017). Also, *Canna indica* displayed reasonable levels of phytoremediation potential with Cr accumulation masses above 200 mg/kg in its roots in two studies (Bose *et al.* 2008; Taufikurahman *et al.* 2019). Its TF was, however, below one.

Leersia hexandra is a hyperaccumulator of Cr, according to a study carried out by Zhang *et al.* (2007) which reported that plants growing naturally by the side of an electroplating facility accumulated up to a maximum of 2,978 mg/kg Cr (Zhang *et al.* 2007). Hydroponic experiments in the same study accumulated up to 6,868 mg/kg Cr(III) and 597 mg/kg Cr(IV). A separate study by Zhang *et al.* (2009) reported the maximum Cr accumulation of the plant in the leaves, stems and roots as 5,430 mg/kg, 1,956 mg/kg, and 40,599 mg/kg, respectively. Liu *et al.* (2015) found that CWs were efficient at removing 5 mg/L Cr(VI) from a solution, with total concentrations in the shoots, roots and sediments being 563 mg/kg, 1,754 mg/kg and 298 mg/kg, respectively.

Alternatively, *L. minor* has been reported to accumulate >64 mg/kg Cr in 10 weeks (Shirinpur-Valadi *et al.* 2019). Removal rates of Cr reported in separate studies were 26, and 25 to 64% (Amare *et al.* 2018; Kebede *et al.* 2020; Nassar & Ibrahim

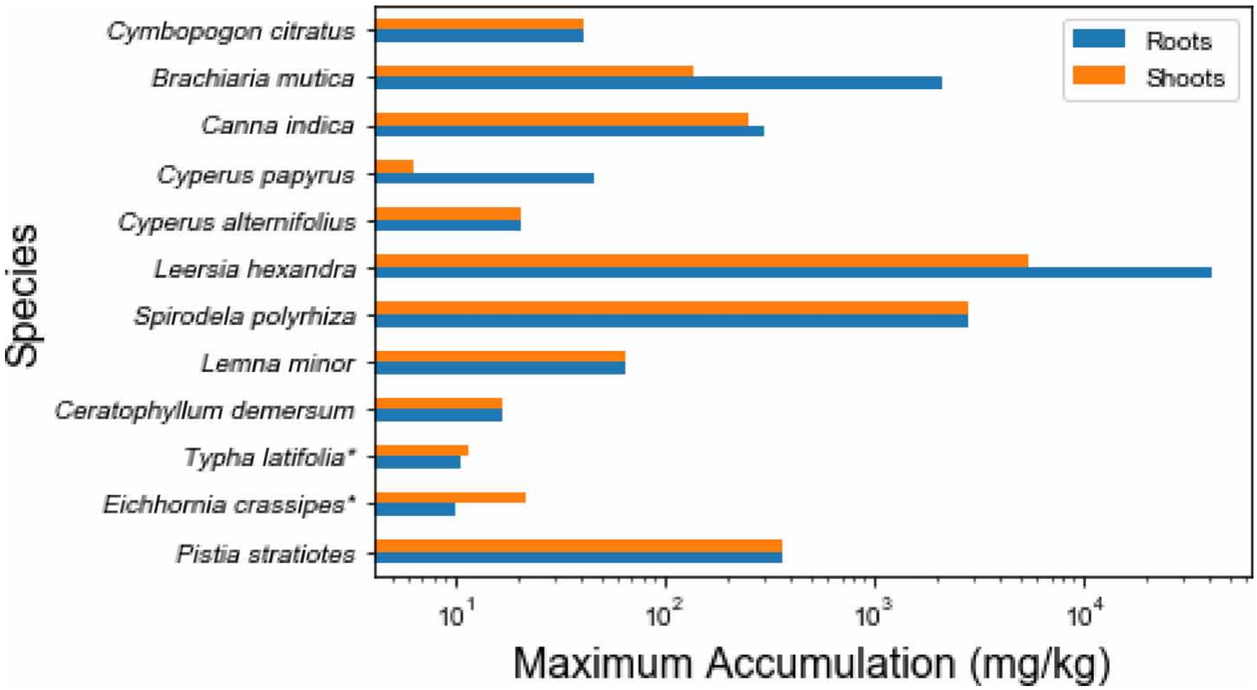


Figure 8 | Maximum accumulation of Cr by different tropical wetland plant species (*indicates average accumulation).

2021). The 64% removal rate occurred under laboratory conditions at an acidic pH of 3.8, implying that it is unlikely that this removal efficiency could be achieved if the plant was used for phytoremediation purposes in a natural environment. Overall, *L. minor* may have some potential to remediate Cr; however, the low removal rates from existing studies suggest it may not be the most efficient choice in comparison to other aquatic plant species.

Cyperus isocladus had few published literature supporting its use as a phytoremediator of Cr and its use in a CW did not give impressive results, with a 41% removal efficiency after 12 days (Wirosoedarmo *et al.* 2020). Similarly, *Cymbopogon citratus* had few supporting studies but was reported to accumulate over 40 mg/kg in one instance (Gautam *et al.* 2017), although this is much lower that many results for other plants.

Phragmites mauritianus shows great potential as part of a CW, with one study reporting removal efficiencies of above 99% for Cr (Kaseva & Mbuligwe 2010). However, in this study, accumulation masses were not found and further research is required to determine the full potential of *P. mauritianus*.

In summary, according to the studies reviewed, *L. hexandra* is the most promising plant species in remediating Cr, due to its ability to uptake very high concentrations of Cr. However, Cr is mostly accumulated in the roots of the plant, which may make phytoextraction difficult. High removal rates across several studies also indicate *Eichhornia crassipes* and *B. mutica* are strong candidates for remediation. *Typha latifolia*, *P. mauritianus*, *Phragmites australis* and *Ipomoea aquatica* appear to be reasonably effective at remediating Cr, although further studies are needed to confirm this assumption. *Pistia stratiotes* has shown good remediation potential in several studies. *Cyperus isocladus* and *Cymbopogon citratus* have also proven ability to accumulate Cr, but not as effectively as the other candidates.

4. DISCUSSION

4.1. Urban surface water pollution in SSA and its pollution sources

According to the studies reviewed, the surface water quality in urban areas of SSA is considered poor. The averages and maximums for parameters turbidity, chlorine, faecal coliforms, *E. coli*, Cr and Pb all failed to meet the WHO *Guidelines for Drinking Water* (2017) in the majority of studies reviewed. Since the land-use dynamics in many SSA cities lie on a trajectory of increasing densification (Mulugeta *et al.* 2017; Gbanie *et al.* 2018), rapidly expanding urban areas are likely to increase contamination and further disrupt the natural hydrological cycle (McGrane 2016). Out of the 133 data sets reviewed, only pH and turbidity were measured in more than half, indicating that water quality studies in SSA tended to focus on a few specific parameters.

There was considerable inconsistency in the water parameters measured across studies. This is likely to be a result of missing national drinking water guidelines in many low-income countries and/or poor enforcement of the existing guidelines. The last issued international guidance on water quality testing in low-income countries was issued in 1997 by the WHO (WHO 1997). However, this guidance may no longer be comprehensive enough in its scope due to its inherent focus on bacteria rather than the broad range of chemical as well as microbial contaminants affecting water quality. Hence, it is possible that there are, for example, chemical contaminants which are detrimental to human health that are not currently being measured across SSA. For example, a review by Fawell & Nieuwenhuijsen (2003) discussed As, Se and U as problematic drinking water pollutants. However, this review found that, in practice, these substances were either entirely absent or being measured very infrequently across studies. However, it should be noted that laboratory capacities are low in Africa and often absent in more remote regions (Peletz *et al.* 2018).

It is also worth noting that microbial measurements were found to be presented inconsistently across studies. Some studies tested specifically for indicator organisms such as *E. coli*, whereas others tested generally for faecal coliforms or total coliforms. This is expected, as these are the three most common indicator organisms to test for faecal contamination in low-income countries (Tortorello 2003). Other studies tested for alternative indicator organisms such as faecal streptococci, *Clostridium* spp. or *Pseudomonas* spp. The presence of these indicator organisms in a water sample suggests that other faecal-related pathogens in the water are also present but does not guarantee it (Ashbolt *et al.* 2001).

However, most of the existing work on phytoremediation in tropical environments has been based on the use of temperate indicator organisms, models and mechanisms, regardless of their applicability or appropriateness for tropical environments. Some studies observed a high re-grow behaviour of *E. coli,* especially in tropical environments, and suggested measuring for faecal enterococci instead. Conversely, other studies suggested a weak correlation between indicator bacteria and pathogenic bacteria (Wright 1989; Rochelle-Newall *et al.* 2015). Moreover, national water quality regulations across SSA and worldwide differ with respect to which indicator organisms are most suitable, which of course is also subject to local environmental conditions. Nevertheless, *E. coli* is widely considered to be the best indicator of pathogens in a water sample (in the absence of better substitutes), as it is found in almost all faecal matter of mammals, yet is rarely sourced from other natural sources and does not substantially replicate whilst in the environment (Edberg *et al.* 2000). Across the studies assessed, there were very few samples taken for *E. coli*. However, this may be due to the fact that it is often easier to test for a group of bacteria (i.e., total coliforms) rather than for the specific organism. As mentioned above, there is also often confusion about which indicator organism is more appropriate, or whether to focus on faecal coliforms or *E. coli* or both. Only four out of the 60 studies we reviewed measured *E. coli*, and the same was true of total coliforms. Faecal coliforms were measured in 17 studies, suggesting that this is commonly used as an alternative to *E. coli* measurements in SSA. However, with *E. coli* being one of the few thermotolerant faecal coliforms, there is a strong correlation between *E. coli* and faecal coliforms which is estimated to be between 77 and 84.3% (*E. coli*/thermotolerant coliforms ratio) depending on the method and the culture medium used (Hachich *et al.* 2012). If we can rely on the assumption that about 80% of thermotolerant faecal coliforms are *E. coli,* we have a reliable reference point for comparing studies using *E. coli* or thermotolerant faecal coliforms.

Since an estimated 10% of the population of SSA drinks from surface water sources as compared to 72% who have basic or limited use of improved sources including borehole wells (WHO & UNICEF 2017), it is unsurprising that there were noticeably more studies focused on groundwater drinking wells than surface water. Nevertheless, surface water is rarely safe for human consumption and generally can be much more easily contaminated. According to the studies reviewed, the main sources of microbial contamination are from human and animal waste (e.g., Cabral 2010). Open defecation causes an increase in microbial contamination in water supplies (Rajgire 2013). Also, most urban areas in SSA have no functional liquid and solid waste disposal systems in place (Remigios 2010). Urban run-off created during rain events flushes the uncontained liquid and solid waste into the next water body. This suggests that improved containment and treatment of sewage is key to reducing pathogens in a water source. Therefore, most people have concentrated on alternative water sources such as groundwater, if these are available. Lapworth *et al.*'s (2017) review of urban groundwater sources in SSA found that water from untreated wells can also be of very poor quality. Many other studies have cited similar problematic parameters to those found in this surface water quality review, including NO_3, Cl, Turbidity and PO_4.

The way in which some data are reported in the reviewed studies caused a discrepancy in what the 'maximum' for a parameter meant. For 70 out of 135 data sets, the maximum was found to be the largest value recorded across all sampling data for a given parameter (i.e., the sample maximum). For 35 data sets, the maximum was taken as the highest mean value across the sampling sites (i.e., the maximum of the means), as the studies only provided average values. 30 of the reviewed studies did not give a maximum value in any sense. This discrepancy means that around one-third of the maximum values given by studies are an underestimate of the actual highest value measured.

Conversely, the accumulation of heavy metals and toxic chemical compounds in the human body is a growing and adverse health concern. Multiple studies have shown that urbanisation increases the exposure of citizens to toxic materials (Hu *et al.* 2013; Rehman *et al.* 2018; Zhang *et al.* 2020). Since the African continent currently faces an unprecedented rapid urbanisation rate (Heinrigs 2020) its challenges regarding heavy metal pollution are only likely to increase.

The literature reviewed for this study demonstrates that pollution can occur from a wide range of sources, and the exact water quality issues caused by the source vary greatly. Different industrial processes produce different water quality issues. For example, Cr pollution has been attributed to effluents from ferrochrome production and tanneries (Loock-Hattingh *et al.* 2015; Suryani *et al.* 2017), Na and chlorides from paper production (Ambani & Annegarn 2015), Mn has been linked to steel manufacturing (Adeniyi & Emile 2016) and Zn is associated with ore processing and electroplating (Adeniyi & Emile 2016). A variety of different water quality issues have also been attributed to mining operations. For example, U, Hg and cyanide pollution has been attributed to gold mining (Winde *et al.* 2004; Kone *et al.* 2019), Cu pollution has been linked to copper mining (Weissenstein & Sinkala 2011), Fe, Mn and NO_3 pollution is associated with the mining of alkaline rocks (Meck *et al.* 2009) and the many problems associated with acid mine drainage are connected to the mining of coal and metal ores (Ochieng *et al.* 2010).

Although there are some natural sources of water pollution, such as saltwater intrusion, mineralisation of rocks releasing naturally occurring toxic minerals and faecal pollution caused by animals, the most significant pollution sources found in the studies we reviewed were anthropogenic. Heavy metal pollution in particular is caused by a wide range of sources, including vehicle pollution, domestic waste, structural corrosion, distribution systems, mining and other industrial activities (Briffa *et al.* 2020).

4.2. Phytoremediation of microbiological and chemical contaminants

Sunlight (UV light) has been reported to be the most important factor in the remediation of pathogens in water (El-Sharkawi *et al.* 1989). Nevertheless, aquatic plants have been shown to improve water quality in CW experiments in comparison to controls without plants (Leto *et al.* 2013; Amare *et al.* 2018; Fahim *et al.* 2019). Our survey of the available literature revealed that there are relatively few studies looking at the removal of microbial contaminants using phytoremediation. Most research is generally focused on the phytoremediation of heavy metals or nutrients such as nitrates, phosphates and ammonia.

Whilst known to be effective, the mechanisms by which plants remediate pathogens from water are not fully understood (Dhir 2020). The removal rates of microbes from a water supply are affected by complex factors (Perkins & Hunter 2000). For example, suspended solids and ammonia interfere with efficient disinfection of water (Gersberg *et al.* 1989). Perkins & Hunter (2000) list the rate of effluent flow, die-off rate, rate of removal by filtration and sedimentation, rate of addition from animal sources and microbial competition as factors that influence bacterial removal efficiency. This study also reported that the flow rate or hydraulic load negatively correlates with removal efficiency, which suggests that fast flowing wetlands may be very poor at removing pathogens. A recent study conducted in New Zealand further indicated that some plants exhibited bacteriostatic characteristics (i.e., preventing the growth of bacteria), thus leading to the decrease in one or more of the human pathogens and indicator organisms tested (Gutierrez-Gines *et al.* 2021). A study comparing the use of water lettuce in CWs to non-organic subsurface flow limestone and sawdust filtration systems found that *Pistia stratiotes* performed slightly better at removing total nitrogen and total phosphorus. However, the limestone substrate wetland performed better at removing FC and TSS, possibly due to increased alkalinity which inactivates various bacteria and viruses (Hijikata *et al.* 2016; Fahim *et al.* 2019). This confirms the assumption that coliform bacteria prefer a slightly acidic medium for growth. This study also revealed that high root surface area was important in trapping bacteria to remove them from water.

According to Stottmeister *et al.* (2003), the effect of a plant's root system is principally explained by the supply of oxygen to the roots, which plays a crucial role in the activity and type of metabolism performed by microorganisms in the root zone. This is especially so for the grazing pathogens like protozoan, nematodes, zooplankton, lytic bacteria and viruses (Vymazal 2005). Another process of removal via phytoremediation is the adsorption by biofilms on the sediment media and rhizosphere (Stevik *et al.* 2004; Stott & Tanner 2005). Macrophytes also can affect pathogens by excreting toxic antimicrobial substances from their roots (Sundaravadivel & Vigneswaran 2001; Stottmeister *et al.* 2003). The roots, which grow vertically and horizontally, favour the removal by enhancing hydraulic pathways and increasing the contact time (Vymazal *et al.* 1998;

Stottmeister *et al.* 2003). Other factors known to reduce pathogenic organisms in natural and CWs include temperature, unfavourable pH, the presence of toxic chemicals (Vymazal *et al.* 1998; Stevik *et al.* 2004) sedimentation (Stott 2003), microbial competition for nutrients, interaction with the plant's root microbiome and interaction of pathogens with sediments (Searcy *et al.* 2006). Removal may also depend on water type and salinity, which significantly affect pathogens settling (Hogan *et al.* 2013). In contrast, some pathogenic organisms are halophilic or thrive in slightly saline water. These include *Vibrio cholerae* or faecal streptococci (Holt 1993).

The rate of metal uptake in plants varies widely depending on both the species of plant and the metal being accumulated. In many studies referenced in this review paper, the bioaccumulation of heavy metals was often studied under controlled laboratory conditions. Most of the results in Tables 2–5 use experimental setups that consist of CWs, or batch experiments in which plants are kept in a container with selected plant grow media, nutrients and contaminants. In many studies, plants were exposed to high concentrations of heavy metals. High heavy metal concentrations often resulted in toxic effects being exhibited by the plants, leading to a reduction in plant parameters such as root length, shoot length and overall biomass. Many heavy metal ions such as e.g., Co, Cu, Fe, Mn, Mo, Ni, and Zn are essential for plant growth and development. While others soluble elements are absorbed as part of the plant's physiological hydration process (Arif *et al.* 2016).

In the SSA water quality studies evaluated for this review, the concentration of these heavy metals was generally significantly lower than the heavy metal concentrations that caused toxic effects to plants. Average concentrations of Pb, Cd and Cr in surface water measurements only exceeded 1 mg/L in a single study for each metal, with Cd concentration only exceeding 0.25 mg/L in two out of 24 studies. Therefore, the relatively low concentration of heavy metals in surface waters in comparison to many laboratory studies means that heavy metal toxicity is unlikely to damage any of the studied aquatic plants if they are used for phytoremediation of contaminated water, except in circumstances in which contamination of one or more heavy metals is extremely severe.

C. demersum appears to be one of the most effective accumulators of both Pb and Cd, so it is a strong candidate for phytoremediation. *L. hexandra* has very high phytoremediation potential for Cr; however, there is a lack of phytoremediation focused on other heavy metals. A study by Lin *et al.* (2016) found that the plant *L. hexandra* can uptake very high concentrations of Cu, particularly in the roots. This suggests that the species may also be a good remediator of other heavy metal contaminants. Some of the other plants that appear promising for phytoremediation are *Typha latifolia* (Cd and Cr), *S. polyrhiza* (Pb and Cd), *Ipomoea aquatica* (Cd and Cr) and *L. minor* (Pb). However, this review has emphasised the need for more field-based, practically applied studies to further advance the field of phytoremediation.

Several studies identified specific plants as potential 'hyperaccumulators' of heavy metals. The criteria required for a plant to be a hyperaccumulator are well defined for some heavy metals, but more ambiguous for others. The threshold for hyperaccumulator status was originally defined for Ni as 1,000 mg/kg concentration in the dry leaf mass (Brooks *et al.* 1977); however, this differs for other metals depending on the typical concentrations observed to be accumulated naturally by most plant species. The typical thresholds defined for hyperaccumulator status are defined as 1,000 mg/kg for Pb, 300 mg/kg for Cr and 100 mg/kg for Cd (van der Ent *et al.* 2013). This study also noted that hyperaccumulator status should only be accredited to species of plants that have been found to accumulate over the threshold value of heavy metal in the leaves or fronds of a species in its natural habitat. Hydroponic experiments are not considered sufficient to claim hyperaccumulator status for a species because it is doubtful that the plant could accumulate such levels of a heavy metal and survive in their natural environment. This is supported by Gerhardt *et al.* (2009), who state that certain plant stress factors present in the natural environment, but insufficiently captured by laboratory and greenhouse studies, may cause considerable variation and challenge for field applications. The study by van der Ent *et al.* (2013) also clarified that accumulation of heavy metals in the root alone is not sufficient to attribute hyperaccumulator status to a plant species, as this is a much more common phenomenon and limits the species' phytoextraction potential. Therefore, this study claimed that hyperaccumulation has not yet been proven beyond doubt for Pb and Cr, as the existing studies are potentially erroneous. Plants suitable for phytoextraction need to have a high translocation from the roots to the easily harvestable parts of the plant (Nanda Kumar *et al.* 1995), because it is difficult to remove roots from the soil when harvesting. If the roots of a plant cannot be harvested, although the metal is stabilised in the roots of the plants, when the plant roots die and decay the metals stored in the rhizosphere of the plant can be released back into the soil (Liu *et al.* 2015). However, many aquatic plants either have no roots, or roots that are present in water, which makes their removal easier. Therefore, aquatic plants that absorb large concentrations of heavy metals in the roots but do not translocate metals well may still be suitable for phytoextraction. This can be determined on a plant-by-plant basis, by investigating the harvesting methods for each individual plant species.

Many aquatic invasive species are often hyperaccumulators due to their rapid biomass growth and strong resistance to toxic metals (Yang *et al.* 2007; Prabakaran *et al.* 2019). Plants popular in SSA phytoremediation studies include multiple invasive aquatic species, such as *Typha latifolia* (Anning *et al.* 2013; Kipasika *et al.* 2016; Worku *et al.* 2018) and *Eichhornia crassipes* (Ndimele & Jimoh 2011; Kassaye *et al.* 2016; Auchterlonie *et al.* 2021). These species spread less easily when in isolated CWs, but as invasive alien species they can disrupt biodiversity and deplete ecosystem productivity *in situ* (Kateregga & Sterner 2007; Vilà & Hulme 2017). There is optimism that invasive species can be managed effectively in order to benefit communities (Hill & Coetzee 2017; Yan *et al.* 2017). However, there are examples all across SSA of alien aquatic macrophytes causing ecosystem service problems (Coetzee *et al.* 2014; Ombwa *et al.* 2015; Honlah *et al.* 2019; Mukarugwiro *et al.* 2019; Enyew *et al.* 2020). Further research into the potential issues of invasive or noxious plant species is therefore necessary to avoid well meaning phytoremediation schemes from becoming tainted.

In regard to the way forward and future work, it is established that CWs may have the potential to provide clean drinking water to communities by removing organic content, pathogens, and heavy metals from the water. However, more *in situ* pilot testing is needed for measuring specifically for microbial indicators in CWs to determine if plants are capable of effectively remediating pathogens from water sources. Also, studies are yet to be carried out on larger water bodies to determine the feasibility of using plants to help remediate important water sources, such as lakes and reservoirs. While this may be challenging to implement, the complex nature of how contaminants are removed using plants in their natural environments means that it is difficult to specifically and confidently predict how a system will behave without implementing a full-scale phytoremediation strategy and assessing the results. Even though there are plenty of laboratory experiments on phytoremediation, there are still relatively few field trials (Beans 2017). Full-scale testing of different phytoremediation processes and applications is therefore the critical and timely next step in promoting this technique in SSA.

The efficacy of phytoaccumulation is influenced by various environmental and soil chemistry factors (Neilson & Rajakaruna 2015); therefore, trialling in the field provides great benefit scientifically (Beans 2017). Despite this, many SSA studies currently excavate soil from a location to then test in the laboratory (Abaga *et al.* 2014; Waziri *et al.* 2016; Opoku *et al.* 2020). Where CWs are being used to remediate water or soil in urban areas, it is more often for treatment of industrial pollution or effluent (Worku *et al.* 2018; Gajaje *et al.* 2021) rather than in poor residential areas to directly improve water supply.

Studies of urban agriculture in SSA focus mainly on the health impacts caused by the pollutant uptake and contamination of edible crops and many recent studies only consider whether the soil and water conditions will make the plants toxic (Opaluwa *et al.* 2012; Conteh *et al.* 2017; Eliku & Leta 2017; Woldetsadik *et al.* 2017; Kacholi & Sahu 2018; Oppong *et al.* 2018; Gebeyehu & Bayissa 2020), rather than exploring the promising symbiosis of intercropping and phytoremediation; this area of research is currently well studied in China (Hong *et al.* 2017; Bian *et al.* 2021; Cui *et al.* 2021; Yang *et al.* 2021). In order for this transition to occur in Africa, knowledge about practicalities must expand, especially in regard to the end-of-life disposal of contaminated phytoextraction plants. Hyperaccumulator plants must be treated as toxic waste at the end of their life (Neilson & Rajakaruna 2015) and current 'phytomining' methods which remove the toxic metal substances are often energy intensive and require industrial equipment (Akinbile *et al.* 2021). The intrinsic laboratory bias in the literature (Beans 2017) means that different waste disposal methods in underprivileged societies are not being widely explored. Nevertheless, there is an enormous potential for phytoremediators such as e.g., bamboo, giant reed, or hemp after they have fulfilled their remediation purpose, to be used as non-food items such as energy sources, paper pulp, and wooden building materials (Fernando *et al.* 2016).

Given the potential of small-scale phytoremediation to elevate the quality of life of the 80 million Africans drinking untreated surface water (Dwumfour-Asare *et al.* 2018; WHO & UNICEF 2019), prompt and practical implementation of phytoremediation is essential. Countries such as Nigeria and Ghana are actively pursuing new research in this area.

5. CONCLUSION

The current and relevant literature has been reviewed to gain an understanding of urban surface water quality in SSA, major contaminants and the potential of prospective native plant options for phytoremediation. The low-cost and off-grid nature of phytoremediation makes the technique very well suited to SSA (Anning *et al.* 2013; Odoh *et al.* 2019), and most of the research reviewed was based around attempting to find the phytoremediation potential of certain plant species for various microbial and metal pollutants. Of the 85 studies reviewed, a considerable majority were laboratory and greenhouse-based pilot studies rather than tests of full-scale implementation or investigations conducted *in situ* – a trend seen globally (Stephenson

& Black 2014; Beans 2017). The focus on laboratory research highlights that SSA is still at the stage of discovering potential plant species, rather than being at a wide-scale implementation stage. Thus, there is little assessment of the technicalities and implementation of CWs and phytoremediation in SSA to date. Parameters measured vary greatly depending on the study, with almost no parameters being measured consistently across more than half of the studies. Therefore, development of a guideline for water testing in low-resource settings that outlines the most important water quality parameters and their methods for testing could help water quality data to be reported more consistently. A greater emphasis should be placed on testing for *E. coli* or an alternative indicator bacteria considered more reliable in the tropics such as e.g., faecal streptococci. A range of aquatic plants has been identified as potential phytoremediators of various water quality issues, particularly for heavy metals. Phytoremediation of pathogens using CWs shows some potential; however, more studies are needed to develop this technology. Also, more data are needed to assess which phytoremediation approach relates best to the treatment of different forms of surface water contamination. A general emphasis on laboratory studies has resulted in a lack of field studies, which currently limits the practical application of phytoremediation and understanding of the full extent of its capabilities. Therefore, to advance this sustainable and nature-driven technology to the next stage, field trials and full-scale phytoremediation strategies need to be implemented to gauge the true global potential of the technology and further refine the methods to be applicable in a sub-Saharan setting.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

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