Analytical screening of organic chemicals of emerging concern in western Kenya and their contribution to the prevalence of schistosomiasis

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"We ourselves feel that what we are doing is just a drop in the ocean. But the ocean would be less because of that missing drop."

Mother Teresa

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To God be the Glory!

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Abstract

In the past decades, the use and production of chemicals has been on the rise globally due to increasing industrialization and intensive agriculture; resulting in the occurrence and ecotoxicological risks of chemicals of emerging concern (CECs) in the aquatic compartments. Risks include changes in community structure resulting in the dominance of one species and ecosystem imbalance. When dominant disease-causing organisms are in the environment, the disease transmission is increased. For example, host snails for the schistosomiasis, a human trematode disease, are known to be tolerant to pesticide exposure compared to the predators. This would therefore result in an increased abundance of snails which consequently increase the disease transmission in the human population.

Kenya, being a low income country faces a lot of challenges with provision of clean water, diseases and sanitation facilities, and increasing population which results in intensive agriculture coupled with pesticide use. Although a lot of research has been carried out on the environmental occurrence and risk of CECs (Chapter 1), most of these studies have been done in developed countries with limited information from Africa. Additionally, research in Africa focused on urban areas with limited number of compounds analyzed and mostly in the water phase, and inadequate information on the effects of CECs on the aquatic organisms. In order to reduce this knowledge gap, this dissertation focused on identification and quantification of CECs present in water, sediment and snails from western Kenya, and the contribution of pesticides to the transmission of schistosomiasis.

Chapter 2 gives a summary of the results and discussion of the dissertation. In Chapter 3, a comprehensive chemical analysis was carried out on 48 water samples to identify compounds, spatial patterns and associated risks for fish, crustacean and algae using toxic unit (TU) approach. A total of 78 compounds were detected with pesticides and biocides being the compounds most frequently detected. Spatial pattern analysis revealed limited compound grouping based on land use. Acute risk for crustaceans and algae were driven by one to three individual compounds. These compounds responsible

for toxicity were prioritized as candidate compounds for monitoring and regulation in Kenya.

In Chapter 4, an extension of Chapter 3 was done to cover the CECs present in snails and sediment from the 48 sites. A total of 30 compounds were found in snails and 78 in sediments with 68 additional compounds being found which were not previously detected in water. Higher contaminant concentrations were found in agricultural sites than in areas without anthropogenic activities. The highest acute toxicity (TU 0.99) was determined for crustaceans based on compounds in sediment samples. The risk was driven by diazinon and pirimiphos-methyl. Acute and chronic risks to algae were driven by diuron whereas fish were found to be at low to no acute risk.

In Chapter 5, the effect of pesticide contamination on schistosomiasis transmission was evaluated by applying complimentary laboratory and field studies. In the field studies, the ecological mechanisms through which pesticides and physical chemical parameters affect host snails, predators and competitors were investigated. Pesticide data was obtained from the results in chapter 3. The overall distribution of grazers and predators was not affected by pesticide pollution. However, within the grazers, pesticide pollution increased dominance of host snails. On the contrary, the host-snail competitors were highly sensitive to pesticide exposure. For the laboratory studies, macroinvertebrates including *Schistosoma*-host snails, competitors and predators were exposed to 6 concentrations levels of imidacloprid and diazinon. Snails showed higher insecticide tolerance compared to competitors and predators.

Finally, Chapter 6 summarizes the conclusions of this dissertation, placing it in a broader context. In this dissertation, a comprehensive chemical characterization and risk assessment of CECs has been carried out in freshwater systems; together with the effects of pesticides on schistosomiasis transmission in rural western Kenya. Results of this dissertation showed that rural areas are contaminated posing a risk to aquatic organisms which contribute to schistosomiasis transmission. This shows the need for regular monitoring and policy formulation to reduce pollutant emissions which contributes negatively to both ecological and human health effects.

Zusammenfassung

Analytisches Screening organischer Chemikalien von wachsender und besorgniserregender Bedeutung in West-Kenia, sowie deren Beitrag zur Prävalenz der Bilharziose

Der Gebrauch und die Produktion von Chemikalien sind in den letzten Jahrzehnten weltweit gestiegen. Dieser Anstieg wurde vor allem durch die wachsende Industrialisierung und konventionelle Landwirtschaft als Folge der stetig wachsenden Bevölkerung hervorgerufen. Menschliche Aktivitäten haben dazu geführt, dass organische Mikroschadstoffe in aquatischen Kompartimenten vor allem in Wasser, Sedimenten und in Lebewesen auftreten. Bei diesen Stoffen handelt es sich unter anderem um Arzneimittel, Körperpflegeprodukte, Pestizide und Biozide, Steroide, Weichmacher und andere industriell eingesetzte organische Chemikalien.

Das Auftreten dieser Stoffe in der Umwelt könnte negative ökotoxikologische Auswirkungen auf aquatische Organismen zur Folge haben. Zum Beispiel wurden Änderungen in der Struktur der Lebensgemeinschaft beobachtet, wenn Organismen Mikroschadstoffen ausgesetzt werden. Diese Änderungen können zur Dominanz einer Art führen, was ein ökologisches Ungleichgewicht zur Folge hat. Zudem wird vermutet, dass solche Stukturänderungen auch für Krankheiten in Menschen, zum Beispiel für Bilharziose, verantwortlich sind. Bilharziose ist eine parasitäre Krankheit, die durch den Saugwurm des Geschlechts *Schistosoma* ausgelöst wird. Schnecken dienen hierbei als wichtige Wirte. Die Schnecken, die für die Übertragung des Parasiten wichtig sind, sind toleranter gegenüber Pestiziden als ihre Fressfeinde. Das führt zu einem erhöhten Aufkommen von Schnecken und wiederum zu einem erhöhten Übertragungsrisiko für Menschen.

Kenia steht als Niedriglohnland vor großen Herausforderung hinsichtlich der Bereitstellung sauberen Wassers sowie medizinischer und sanitärer Einrichtungen in den Gemeinden. Zudem wird immenser Druck durch die wachsende Bevölkerung auf die Landwirtschaft ausgeübt, was mit der Umwandlung von Grünland in Ackerfläche und dem intensiven Einsatz von Pestiziden einhergeht, um die Erträge zu steigern. Dies führt dazu,

dass Mikroschadstoffe in die Umwelt und folglich auch in die Oberflächengewässer gelangen. Es gibt viele wissenschaftliche Studien zum Vorkommen und Risiko von Mikroschadstoffen in der Umwelt (Kapitel 1). Jedoch wurden die meisten dieser Studien in Industrieländern durchgeführt, so dass es nur begrenzte Informationen zu diesen Stoffen für Afrika gibt. Die wenigen Studien aus Afrika beziehen sich meistens nur auf urbane Gebiete. Zudem wurden nur wenige Stoffe untersucht und auch nur in der Wasserphase und nicht im Sediment oder in Biota. Es gibt hier auch keine umfassenden Erkenntnisse über die Auswirkungen von Mikroschadstoffen auf die Gemeinschaften von aquatischen Organismen.

Um diese Wissenslücken zu füllen, befasst sich die vorliegende Dissertation mit der umfassenden Charakterisierung von organischen Mikroschadstoffen in Oberflächengewässern im ländlichen Raum in West-Kenia. Das Ziel der Dissertation war es, organische Mikroschadstoffe in Wasser-, Sediment- und Biotaproben aus West-Kenia zu identifizieren und zu quantifizieren (Kapitel 3 und 4) und den Beitrag von Pestiziden zu der Übertragung von Bilharziose zu bestimmen (Kapitel 5). Insgesamt wurden 48 Probenahmestellen in sieben Landkreisen einschließlich Narok, Kisumu, Homabay, Kericho, Nyamira, Kisii und Migori im südlichen Einzugsgebiet des Viktoriasees ausgewählt. Hier wurden verschiedene Wasserkörper, unter anderem Flüsse, Bäche, Drainagen, Altarme und Wasserreservoirs, beprobt. Die Probenahmen wurden zu zwei unterschiedlichen Zeitpunkten durchgeführt, die mit der Regenzeit übereinstimmten. Die erste Probenahmekampagne fand zwischen September und Oktober 2017 statt. Die zweite Kampagne wurde zwischen Oktober und November 2018 durchgeführt. Für die Identifizierung von Mikroschadstoffen wurde stets eine Analysemethode mittels Flüssigkeitschromatographie gekoppelt an hochauflösende Massenspektrometrie verwendet.

Kapitel 2 beinhaltet eine Zusammenfassung aller Ergebnisse und der Diskussion. In Kapitel 3 wurde eine umfassende chemische Analytik von Wasserproben, die während der ersten Probenahmekampagne genommen wurden, durchgeführt. Aus einer Liste von 428 zu untersuchenden Mikroschadstoffen wurden 75 Stoffe in den Wasserproben detektiert. Unter den detektierten Stoffen, war die Gruppe der Pestizide und Biozide mit

26 Ausgangssubstanzen und fünf Transformationsprodukten die am häufigsten vertretene Gruppe. In dieser Studie wurden zum ersten Mal Weichmacher in Wasserproben aus Kenia nachgewiesen. Die am häufigsten detektierten Substanzen waren das Antihistaminikum Diphenhydramin (90%), das Herbizid Simazin (88%) und die Industriechemikalie Triethylphosphat (85%). Unter den Transformationsprodukten wurde Acetylsulfamethoxazol am häufigsten detektiert (81%). Konzentrationen über 1 µg/L wurden für Acetylsulfamethoxazol (24 µg/L), Triethylcitrat (6,5 µg/L), N-ethyl-o-toluolsulfonamid (2 µg/L) und Hexazinon (1,5 µg/L) gefunden. Die Konzentrationen, die für Acetylsulfamethoxazol gemessen wurden, überstiegen die Grenzwerte der Europäischen Trinkwasserrichtlinie um den Faktor 2800. Über ein *Suspect Screening* wurden drei weitere Substanzen identifiziert und quantifiziert: Adenosin (bis zu 4 µg/L), und die antiretroviralen Medikamente Lamivudin (bis zu 1 µg/L) und Nevirapin (bis zu 0,5 µg/L). Eine Musteranalyse zeigte, dass es keinen Zusammenhang zwischen dem Vorkommen der Substanzen basierend auf der Landnutzung gab. Jedoch wurde ein Cluster detektiert, das Pestizide zusammenfasst, die in großen Zuckerrohrplantagen eingesetzt werden. Die ökotoxikologische Relevanz der detektierten Stoffkonzentrationen wurde für Fische, Algen und Krebstiere mittels toxischer Einheiten (engl. Toxic Units, TU) bewertet. Für Krebstiere wurde das größte potentielle akute Gesamtrisiko (TU bis zu 2) berechnet mit Diazinon als Treiber. Ein niedrigeres aber immer noch erhebliches akutes Gesamtrisiko (TU = 0,5) wurde für Algen ermittelt, was hauptsächlich auf Diuron zurückzuführen ist. Das niedrigste Gesamtrisiko wurde für Fische berechnet (TU = 0,001). Insgesamt sechzehn Substanzen wurden aufgrund ihrer Grenzwerte für akutes und chronisches Risiko in die Liste derer Substanzen aufgenommen, die in Zukunft reguliert und überwacht werden sollten. Ebenfalls in diese Liste aufgenommen wurden sieben weitere Substanzen, die anhand ihrer niedrigsten vorhergesagten Konzentration, bei der keine Effekte zu erwarten ist (engl. Predicted No Effect Concentration, PNEC) ausgewählt wurden.

In Kapitel 4 wurde die erste Studie um die Untersuchung von organischen Mikroschadstoffen in Schnecken und Sedimentproben, genommen an den 48 Probenahmestellen, erweitert. Die Schnecken wurden mit der QuEChERS Methode extrahiert. Die Sedimentproben wurden mit einer beschleunigten Lösemittelextraktion

(engl. pressurized liquid extraction (PLE)) aufbereitet. Insgesamt wurden 30 Mikroschadstoffe in Schnecken und 78 Mikorschadstoffe in den Sedimentproben gefunden. Durch die Analyse der Schnecken und Sedimentproben wurden 68 zusätzliche organische Mikroschadstoffe detektiert, die zuvor nicht in den Wasserproben gefunden wurden. In dieser Studie wurde zum ersten Mal das Krebsmedikament Anastrozol in kenianischen Umweltproben nachgewiesen. Wie in den Wasserproben, waren auch in den Schnecken- und Sedimentproben die Gruppe der Pestizide und Biozide die am häufigsten detektierte Gruppe. In den Schneckenproben wurden die Substanzen Cotinin und Tris(2-chlorethyl)phosphat am häufigsten detektiert, letztere war in allen Proben enthalten. In den Sedimentproben traf dies auf DEET zu, mit einer detektierten Häufigkeit von 98%.

Konzentrationen von Einzelsubstanzen lagen zwischen 0,2 und 481 ng/g wet weight (ww) in Schnecken und zwischen 0.2 – 111 ng/g organic carbon (OC) in Sedimentproben. Die höchsten Konzentrationen über 300 ng/g ww wurden für N-Ethyl-o-Toluolsulfonamid, Cotinin und Atrazin in Schnecken gefunden. In den Sedimentproben wurden Konzentrationen über 111 ng/g OC für Pirimiphos-methyl detektiert. Es wurden höhere Schadstoffkonzentrationen in Proben von Probenahmestellen in landwirtschaftlichen Gebieten gefunden als in Gebieten ohne menschliche Aktivität. Das höchste potentielle akute Gesamtrisiko (TU = 0,99) durch Schadstoffe in Sedimentproben wurde für Krebstiere berechnet. Die Höhe des Gesamtrisikos resultierte vor allem aus den detektierten Konzentrationen von Diazinon und Pirimiphos-methyl. Das akute und chronische Gesamtrisiko (TU bis zu 0,24) für Algen wurde durch Diuron bestimmt. Für Fische wurde ein niedriges beziehungsweise kein akutes Risiko festgestellt (TU bis zu 0,0007).

In Kapitel 5 wurden die Auswirkungen der Pestizidbelastung auf die Übertragung von Bilharziose durch eine Kombination von Feld- und Laborstudien untersucht. In den Feldstudien wurde untersucht, welche ökologischen Auswirkungen Pestizide und physikalisch-chemische Parameter auf die Wirtsschnecke, deren Fressfeinde und Nahrungskonkurrenten haben. Hierfür wurden Pestizidkonzentrationen aus Kapitel 3 verwendet. Die allgemeine Verteilung von Weidetieren und Räubern wurde nicht durch

die Pestizidbelastung beeinflusst. Jedoch wurde durch die Pestizidbelastung die Dominanz der Schnecken unter den Weidetieren erhöht. Hingegen reagierten die Nahrungskonkurrenten der Schnecken sehr empfindlich auf Pestizide. Für die Laborstudien wurden in der zweiten Feldprobenahme Makroinvertebrate einschließlich der Wirtsschnecken, deren Nahrungskonkurrenten und Fressfeinde gesammelt. Die Makroinvertebrate wurden im Labor sechs verschiedenen Konzentrationen von Imidacloprid und Diazinon ausgesetzt. Jeweils fünf Exemplare derselben Art wurden in 100 ml Glasbehältern exponiert. Die Testdauer lag zwischen 24 und 48 Stunden. Die untersuchten Endpunkte waren Immobilität und Letalität. Die Wirtsschnecken zeigten eine höhere Toleranz gegenüber den Insektiziden im Vergleich zu ihren Nahrungskonkurrenten und Fressfeinden. Der LC_{50} Wert für Imidacloprid lag zwischen 0,007 mg/L für Ruderwanzen und 599 mg/L für *Melanoides sp* (Turmdeckelschnecke). Die Letalität für Wirtsschnecken lag bei unter 10%. Für Diazinon lagen die LC_{50} Werte zwischen 0,5 mg/L Baetidae (Eintagsfliegen) für und 33 mg/L für die Wirtsschnecke *B. pfeifferi*. Daraus lässt sich schließen, dass Pestizide durch ihre negativen Auswirkungen auf Nahrungskonkurrenten und Fressfeinde indirekt die Verbreitung der Wirtsschnecken fördern und somit zu einem erhöhten Infektionsrisiko mit Bilharziose in tropischen Süßwasserkörpern beitragen.

In Kapitel 6 wurden die einzelnen Studien zusammengefasst, um ein generelles Fazit zu ziehen und die Ergebnisse in einen größeren Kontext zu setzen. Im Rahmen der vorliegenden Dissertation wurde eine umfangreiche chemische Charakterisierung und Risikobewertung von Chemikalien mit wachsender und besorgniserregender Relevanz am Beispiel verschiedener Frischwassersysteme durchgeführt – hinzu kam der Einfluss von Pestiziden auf die Verbreitung von Schistosomiasis in ländlichen Teilen West-Kenias. Die Ergebnisse dieser Dissertation zeigen, dass ländliche Räume kontaminiert sind und somit ein Risiko für Wasserorganismen darstellen, was wiederum zur Verbreitung von Schistosomiasis beiträgt. Aus diesem Grund bedarf es regelmäßiger Monitorings sowie politischer Konzepte, um Schadstoffemissionen zu reduzieren, die negative ökologische sowie für den Menschen gesundheitliche Effekte zur Folge haben.

List of Abbreviations

AChE	Acetylcholinesterase inhibition
APCI	Atmospheric pressure chemical ionization
API	Atmospheric pressure interfaces
APPI	Atmospheric pressure photoionization
ART	Acute risk threshold
ARVs	Antiretroviral
A-SMX	Acetyl-sulfamethoxazole
CA	Concentration addition
CECs	Chemicals of emerging concern
C_{ewsed}	Equilibrium water concentrations
CRT	Chronic risk threshold
C_{sed}	Sediment organic carbon
DCM	Dichloromethane
EC	Effect concentration
EE2	Ethynil-estradiol
EFSA	European Food Safety Authority
ESI	Electrospray ionization
GC	Gas chromatography
GLM	Generalized linear model
HRMS	High resolution mass spectrometry
IA	Independent action
K_{oc}	Organic carbon-water partitioning coefficients

LC	Liquid chromatography
$LogK_{ow}$	Octanol-water partitioning coefficient
LVSB	Lake Victoria South Basin
MDGs	Millennium Development Goals
MDL	Method detection limits
MEC	Measured environmental concentrations
MS	Mass spectrometry
MS-MS	Tandem mass spectrometry
m/z	Mass-to-charge ratio
MoA	Mode of action
msPAF	Multi-substance potentially affected fraction
NSAIDS	Non-steroidal anti-inflammatory drugs
NETS	N-ethyl-o-toluolsulfonamid
OMPs	Organic micropollutants
OTC	Over-the-counter
PaBs	Pesticides and biocides
PCA	Principal component analysis
PCPB	Pest Control Products Board
PERMANOVA	Permutational multivariate analysis of variance
PLE	Pressurized liquid extraction
PNEC	Predicted No-Effect Concentration
PPCPs	Pharmaceuticals and personal care products
PSA	Primary Secondary Amine

QuEChERS	Quick, Easy, Cheap, Effective, Rugged and Safe
RQ	Risk quotient
SSD	Species-sensitivity distribution
TMOA	Toxic mode of action
TOC	Organic carbon contents
TOF	Time-of-flight
TU	Toxic units
TU_{max}	Maximum toxic units
WWTPs	Wastewater treatment plants

Chapter 1

Background Information

1.1 Threats to water resources

Sustainable access to safe potable water and basic sanitation is one of the important targets set by the United Nations water-related Millennium Development Goals (MDGs) (Clasen, 2010). Water is a basic requirement for every living organism. Globally, extensive efforts have been put to ensure that everyone can access clean and safe water for daily life. Such efforts include legislations for the protection of aquatic ecosystems and riparian areas, and infrastructural developments for clean water supply (Sousa et al., 2018). However, WHO & UNICEF (2015) noted that approximately 663 million people globally still use unimproved drinking water sources including unprotected ground and surface water. Current practices on water use are insufficient to ensure that water resources are not depleted or polluted especially due to population growth and urbanization (Wang et al., 2014). In addition, effects of climate change results in water scarcity in some areas, while in other areas, flooding is a common phenomenon leading to contamination of water sources (Reuveny, 2007; Wu et al., 2015). The water crisis being experienced in most countries in the world may escalate due to increased water consumption, lack of efficient water reuse strategies and the dramatic climatic changes coupled with increased human activities (Sousa et al., 2018).

In Africa, insufficient infrastructure on water and wastewater treatment results in challenges on provision of clean water and sanitation to the growing population and urbanization (Wang et al., 2014). Africa's increasing population and rising economy has resulted in increasing consumption of water and discharge of wastewater which cause heavy pollution into water systems. It was projected that only 60% of the 783 million people in Sub-Saharan Africa could have access to improved drinking water sources by the year 2015 (United Nations, 2012). The lack of proper solid waste management, leakages of septic tanks, animal waste, storm water drainage and agriculture run off also contributes to pollution of aquatic ecosystems (Wang et al., 2014). Poor sanitation and contamination of aquatic ecosystems expose organisms to risks resulting in a community shift to tolerant organisms (Munz et al., 2017). On human health, disease burden is greatly increased leading to diseases such as cholera, schistosomiasis, and typhoid among others which negatively impacts the economy and increases dependency on medication.

1.2 Types of water pollution

There are four general categories of water pollution namely: thermal, biological, physical and chemical pollution. Thermal pollution results from effluents whose temperature is higher than the receiving natural water body. The effluent originates, for example, from cooling units in industries especially during the operation of thermal and nuclear power plants (Issakhov & Zhandaulet, 2020) where large amounts of water is used. Thermal pollution, unlike chemical pollution, results in a change in the physical properties of water. Heated water alters natural conditions in the receiving water body and affects aquatic flora and fauna (Issakhov & Zhandaulet, 2020). Biological water pollution refers to the contamination of water by pathogenic microorganisms such as bacteria, fungi and viruses (Elliott, 2003). In a broader context, it could also refer to the presence of invasive species such as water hyacinth. Introductions of these non-indigenous species into water systems can be on a large scale due to movement of species to higher latitudes as a result of global warming or could be on a small-scale due to wastewater or aquaculture discharges (Elliott, 2003). Physical pollution impacts primarily the physical appearance of the water caused by floating debris, foam and garbage (EPA, 2020). Erosion from agricultural areas or loose soil at the river banks, construction and logging sites contribute a great load of particulate matter in a water system (Heim & Schwarzbauer, 2013). Recently, (micro)plastics and (nano)plastics have emerged as pollutants in the aquatic environment which could be from consumer products such as exfoliating beads in facial or body scrubs (Andrady, 2011). (Micro)plastic and (nano)plastics pollution could also result from weathering processes of plastic waste resulting into microparticles that are released into the water system (Andrady, 2011).

Chemical pollution results from release of nutrients, pesticides, surfactants, pharmaceuticals and personal care products (PPCPs), trace metals, detergents, among others (Loos et al., 2009; Meffe & de Bustamante, 2014; Wanda et al., 2017). Chemical pollution can be divided further into two groups: conventional pollutants and emerging pollutants (Bu et al., 2013; K'oreje et al., 2020). During the past two centuries, the focus of environmental research has been partly turned from analysis of major ions (Schmidt, 2018) to the conventional pollutants (e.g. polychlorinated biphenyls, polycyclic aromatic

hydrocarbons, nutrients) (Gwenzi & Chaukura, 2018; Olatunde et al., 2014) to most recently the chemicals of emerging concern (CECs), among which PPCPs are particularly one of the most important groups (Bu et al., 2013). Chemicals of emerging concern include endocrine disrupting agents, PPCPs, hormones, UV filters, illicit (non-prescriptive) drugs, plasticizers, flame retardants, disinfection by-products, surfactants, pesticides and biocides (PaBs) among others (Petrie et al., 2014).

Ecotoxicological effects on organisms was first put to the public domain after the release of the book *Silent Spring* by Rachel Carson (Bernhardt et al., 2017; van Emden & Peakall, 2010) which raised concerns on the use of highly persistent pesticides in the United States causing acute environmental problems (van Emden & Peakall, 2010). The publication facilitated the introduction of ecotoxicology in order to conserve the ecosystem. Because of this, global research on CECs have been on the rise due to concerns on the ever increasing chemical production and the ecotoxicological effects on organisms in the ecosystem.

1.3 Organic chemicals of emerging concern in the environment

1.3.1 Sources of contaminants

The uncontrolled production, use and disposal of chemicals have greatly contributed to the widespread occurrence of organic pollutants (Schwarzbauer, 2006). Organic CECs are introduced to the environment due to anthropogenic activities. The sources of CECs in the environment (Figure 1.1) include untreated or partially treated effluents from wastewater treatment plants (WWTPs), surface run off in residential and industrial sites, atmospheric deposition, storm water drainage, effluent from hospitals, livestock production units, agricultural wash off, and solid waste dumpsites or landfills (Anh et al., 2019; Aus der Beek et al., 2016; Ccanccapa et al., 2016; Li, 2014). Effluents from WWTPs are one of the main pathways for the introduction of OMPs into the aquatic environment (Aus der Beek et al., 2016). The use of wastewater effluent for irrigation (Muñoz et al., 2009; Smit & Nasr, 1992) and sewage sludge or animal manure as a plant fertilizer introduces both human and veterinary pharmaceuticals, and biocides into the soil (Caracciolo et al., 2015). Industrial compounds including surfactants have multiple uses

including formulation of detergents and also as ingredients for other consumer products (Caliman & Gavrilescu, 2009). These non-regulated industrial compounds are directly released into surface water (Sousa et al., 2018).

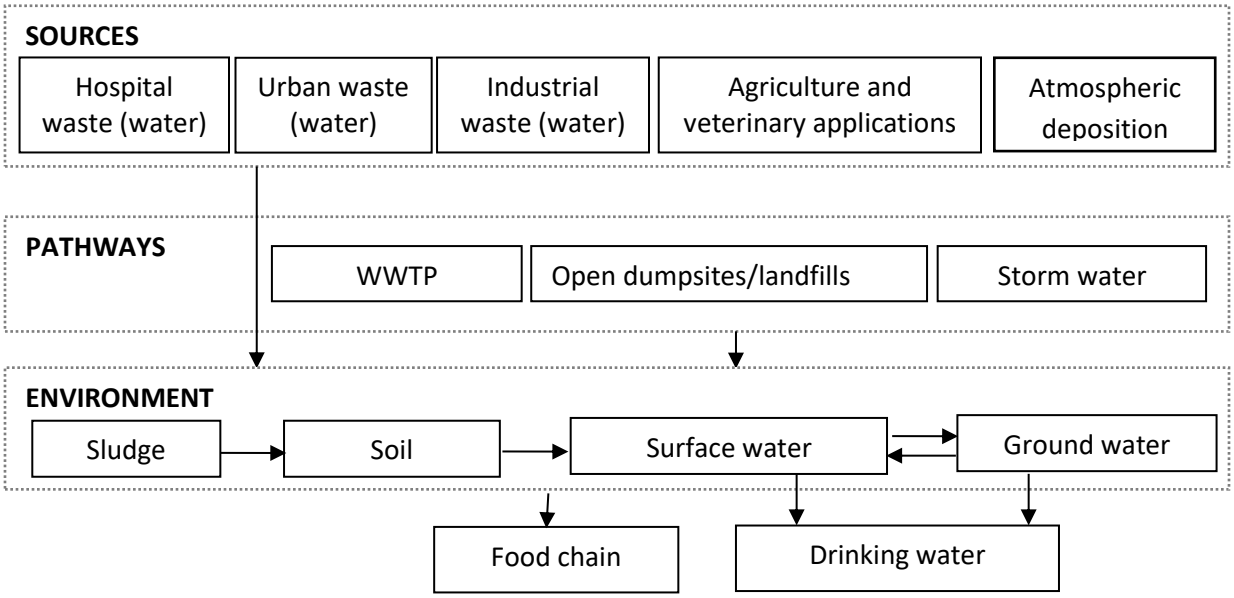

Figure 1.1: Sources and pathways of CECs into the environment. Modified from Vergeynst (2014).

For pharmaceuticals, when administered in human medical care, the compound is not completely eliminated in the body (Caracciolo et al., 2015; Heberer, 2002). Part of the medicine is excreted unaltered or as an active metabolite ending up in the WWTPs (Sui et al., 2015). Conventional municipal WWTPs are originally designed to remove particulates, organic matter, nutrients and pathogens (Luo et al., 2014). While these substances are removed efficiently, organic micropollutants are often insufficiently removed. While compounds such as acetaminophen have been shown to be removed completely during the treatment process (K'oreje et al., 2018), others such as carbamazepine are known to be recalcitrant and therefore known as biomarkers for wastewater effluents (Alvarino et al., 2016; Fram & Belitz, 2011; Gracia-Lor et al., 2017; Zhang et al., 2011). In addition, malfunctioning or lack of maintenance of WWTPs has resulted in the release of minimally treated or untreated wastewater loaded with CECs into the environment (Naddafi et al., 2009; Wood et al., 2015). When released into the environment, some pharmaceuticals such as antibiotics are only partially degraded in the

environment and as a result are likely to accumulate in water systems (Sui et al., 2015). This incomplete removal of compounds coupled with global large-scale consumption of CECs results in their frequent detection in effluents, surface waters, shallow wells and tap water (Jelic et al., 2011; Vasquez et al., 2014) . Further on in this thesis, the focus will be on emerging contaminants in the aquatic environment.

1.3.2 Consumption patterns

The occurrence of these compounds in the environment is influenced by the consumption patterns. The growing population, increasing disease burden by aging societies, and increasing investment of health and agricultural sectors have significantly increased the consumption of PPCPs (Aus der Beek et al., 2016), pesticides and industrial chemicals. The International Federation of Pharmaceutical Manufacturers and Associations (IFPMA) estimated the global pharmaceutical industry to represent $870,200 million in 2013 with North America (41%) and Europe (24%) leading in pharmaceutical market shares (IFPMA, 2013). As for pesticides, It is estimated that they are applied to one-third of global agricultural produce with Europe being the continent with the highest pesticide consumption (Zhang, 2018). For example, between 2007 to 2008, herbicides were mostly used (Zhang et al., 2011). The usage patterns may differ in different countries due to legislations, different disease burdens, agricultural practices and local conditions. As an example, diazinon is banned in the European Union as a plant protection product due to its very high and long term toxicity to aquatic organisms (European Commission for Food Safety, 2007) whereas it is registered for use in Kenya. In another example, the use of streptomycin in fruit growing is widespread in the United States of America while it is banned for such use in Germany (Kümmerer, 2009). As such, the consumption patterns of compounds may vary between countries in Africa and those in the rest of the world.

In Africa, crop damage by insects and disease is common which results in economic loss. To avoid these losses, high investment on pesticides has been done by African governments (Loha et al., 2018). For example in Tanzania, pesticide imports increased from 500 to 2500 tones with 18% of the pesticides used in public health sector while 81% were used in agricultural practices (Elibariki & Maguta, 2017). Recently, invasion by

desert locusts and fall army worms in Kenya has increased economic burden on farmers leading to the government investing more resources on pesticide coverage to eradicate this catastrophe. By 2018, Kenya had 1345 pesticides registered for crop protection, 145 pesticides for public health use, 101 technical grade materials for formulation purposes and 5 restricted pesticides in Kenya (Pest Control Products Board, 2018). For pharmaceuticals, Kenya has registered 687 compounds as essential medicines (Ministry of Health, 2016).

1.4 Fate and distribution of contaminants in aquatic ecosystems

When anthropogenic contaminants are uncontrollably discharged into the environment, they undergo different fates (Figure 1.2) in the aquatic compartments including water, biota and sediments with potential detrimental effects to living organisms. Depending on the compound's physical-chemical properties (Mayer & Holmstrup, 2008), the compound will partition into the different phases based on equilibrium partition which will determine the chemical's persistence, reaction, accumulation and route of exposure to organisms in the different compartments (Mackay & Arnot, 2011; Mayer & Holmstrup, 2008). Distribution of CECs in the water-biota-sediment environment is determined by physical, chemical and biological processes.

The behavior of compounds once they are released into the aquatic ecosystem is driven by chemical activity, a concept that quantifies a compound's energetic state (Reichenberg & Mayer, 2006). Organic chemicals move from high to low chemical activity until a thermodynamic equilibrium is reached at equal chemical activity (Mackay et al., 2020; Reichenberg & Mayer, 2006). Consequently, differences in chemical activity of a compound will determine the chemical fluxes and extent of diffusion, sorption and partitioning between the different aquatic compartments (Reichenberg & Mayer, 2006). In a multimedia systems where thermodynamic equilibrium has been reached, the chemical activity in one phase applies to the other phase therefore making this concept a convenient measure (Reichenberg & Mayer, 2006).

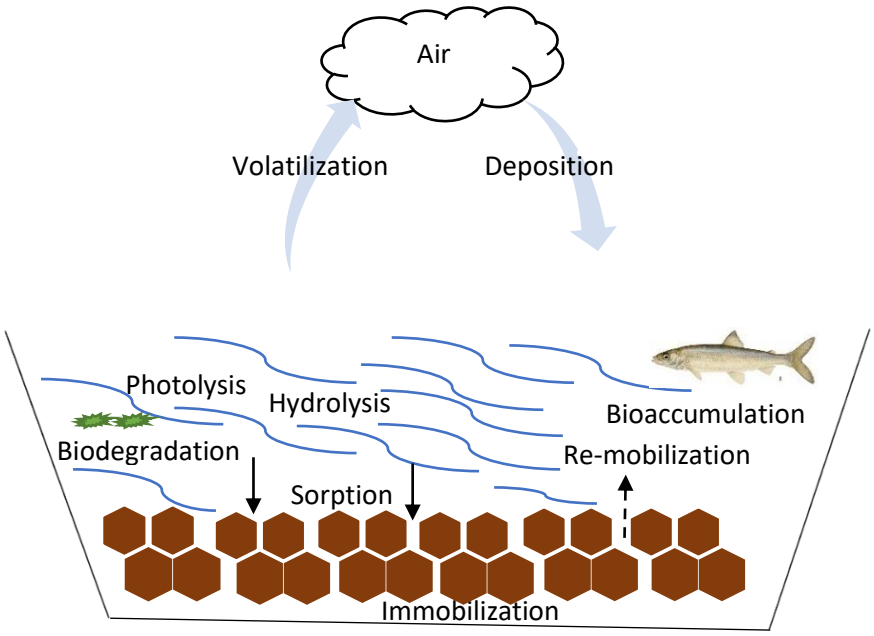

Figure 1.2: Fate of CECs into the aquatic environment.

Caliman and Gavrilescu (2009) complied a comprehensive table showing the different processes including sorption, precipitation, diffusion, complexation, acid dissolution, biotic and abiotic degradation, and bioaccumulation which compounds undergo when they are in the environment. These processes contribute to the bioavailability and bioaccessibility of compounds in the environment. Transfer of pollutants from water to air (volatilization) depends on the compound's Henry's law constant therefore, the fate of CECs through volatilization is minimal as most of the compounds have low volatility (Caliman & Gavrilescu, 2009).

The sediments in the aquatic ecosystem normally act as a source or sink to the contaminants and play an important role in the environmental fate of CECs in water bodies (Massei et al., 2018; Merhaby et., 2019). Hydrophobic contaminants tend to adsorb to the dissolved and suspended organic matter, and in the sediments (Luo et al., 2014) when released into the aquatic environment. Simultaneously, they could also be desorbed from the sediment into the water phase through re-mobilization (Figure 1.2). Bound residues of CECs decrease their leaching, runoff, uptake and bioavailability.

In the aqueous phase, contaminants such as ketoprofen, naproxen, ibuprofen among others are susceptible to photolysis (Petrie et al., 2014) since many of the compounds contain heteroatoms and aromatic rings that can absorb light (Zalenakova, 2018). Water temperatures, availability of sunlight, water depth, organic matter composition, latitude and seasonality are expected to affect the degradation and volatilization process of CECs in the environment (Ebele et al., 2017; Vryzas, 2018; Zalenakova, 2018). Photolysis would play an important role in the fate of contaminants within the study area, as high water and environmental temperature, lower depth, and longer sunlight duration is characteristic of this region. Photolysis results in the reduction of the parent compound concentrations but would also generate several transformation products. Hydrolysis is a transformation process whereby chemical bonds are cleaved by substitution in an organic compound of atoms by a water molecule (Margot et al., 2015). Hydrolysis is common for *β*-lactam, macrolide, and tetracycline antibiotics (Margot et al., 2015; Schwarzenbach et al., 2003).

The bioaccessible contaminants in the water can be taken up by the aquatic organisms through ingestion, respiration or absorption through the skin leading to bioaccumulation in the body of the organism (Yin et al., 2017; Zalenakova, 2018). For example, quinolones antibiotics have been shown to bioaccumulate in bivalves (Li et al., 2012). The contaminants can thereafter be biomagnified resulting in higher body burdens of contaminants in the predator than in the prey (Mackay et al., 2020). Another fate of compounds in water is through biodegradation involving microbial communities in the aquatic ecosystem (Zalenakova, 2018). Biotransformation requires the presence of growth substrates including carbon and energy sources (Margot et al., 2015).

From the processes discussed above, sediments and biota act as passive samplers reflecting long-term and mid-term exposure respectively, and allow also for the detection of compounds that are hardly detectable in the water phase. The detectability of contaminants in biota proves their bioavailability and bioaccumulation potential (Mackay et al., 2011; Yin et al., 2017). In fact, aquatic organisms are normally suitable for assessing micropollutants that have a high octanol-water partitioning coefficient ($\log K_{ow}$) which accumulates in their biological tissues. Highly lipophilic compounds with pseudo-

persistence and continuous input are normally accumulating in sediments (Xue et al., 2005) posing long-term exposure concerns to benthic species and ultimately resulting in a potential risk for the aquatic ecosystem (Li et al., 2017). For example, the persistence of antibiotics at low levels which could promote the rise of antibiotic resistant bacteria in the water system and may enhance the drug resistance of microorganisms (Sui et al., 2015). In order to fully understand the chemical status of an ecosystem, it is therefore essential to apply a multi-media approach involving the identification of compounds from multiple environmental matrices and understand their fate and partitioning in the environment.

1.5 Environmental occurrence of CECs

1.5.1 Global occurrence Of CEC

Contamination of water systems with CECs is a global concern and a lot of efforts have been put in identifying and quantifying these compounds especially from developed countries. The studies have shown the occurrence of these compounds with spatial patterns. For example, regional differences in the occurrence of these CECs are notable especially in Asia and Africa where antiretroviral drugs and some antibiotics were found which were not detected in Europe (Aus der Beek et al., 2016; Fekadu et al., 2019). This is in line with the different disease and consumption patterns in the continents. For example, some compounds such as the pharmaceutical chloramphenicol are banned for use in the European Union due to its carcinogenicity potential (Fekadu et al., 2019) but it is still in use in Kenya (Ministry of Health, 2016).

In the review by Aus der Beek et al. (2016), it was noted that most of the studies have been carried out on CECs in surface water (47% of the database) with majority being rivers and streams. The most commonly reported pharmaceuticals were analgesics, antibiotics and estrogens (Aus der Beek et al., 2016). Sousa et al. (2018) and Petrie et al. (2014) also found pesticides (20.3%), non-steroidal anti-inflammatory drugs (NSAIDS, 13.9%), beta-blockers (4.9%), lipid regulators (5.4%), psychiatric drugs (6.0%) to be among the compound classes mostly reported in monitoring campaigns. These CECs are incompletely removed during wastewater treatment resulting in several of them ending up in the environment, mainly in the final effluent and receiving water compartments.

In an EU-wide survey for polar organic compounds in water carried out by Loos et al. (2009), compounds such as benzotriazole, caffeine, carbamazepine, tolyltriazole, and nonylphenoxy acetic acid were the most frequently and at the highest concentration levels detected. Sediment samples had frequent detections of polychlorinated aromatic hydrocarbons (PAHs) up to 1208 ng/L in samples from Raba River, Hungary (Sousa et al. 2018). In the US, Australia and China bifenthrin was the most frequently detected pyrethroid at 78% in sediments (Li et al., 2017). Recently, there has been an increased detection of industrial compounds in environmental samples. A study performed on the Yangtze river delta in China reported concentrations of 1H-benzotriazole, lauramidopropyl betaine, methylchloroisothiazolinone and 2-naphthalene sulfonic acid higher than 1000 ng/L (Peng et al., 2018) in water which was linked to the 25 chemical industrial parks along the Yangtze River.

1.5.2 Occurrence of CECs in Africa

Monitoring of CECs in African water systems has drastically increased over the past years (Gwenzi & Chaukura, 2018; K'oreje et al., 2020; Madikizela et al., 2019) although data is still scarce. Majority of the research has been carried out in Nigeria, Tunisia, Kenya and South Africa with the latter being the highest contributor at 59% (K'oreje et al., 2020) (Figure 1.3). Other countries with monitoring studies include Tanzania (Damkjaer et al., 2018; Elibariki & Maguta, 2017) and Uganda (Nantaba et al., 2019). Similar to developed countries, most studies on the occurrence of CECs in African surface water have focused on rivers and streams traversing highly urbanized and industrialized areas (K'oreje et al., 2016; 2012; Kairigo et al., 2020; Matongo et al., 2015; Ngumba et al., 2016). These studies show more diffuse sources including urban and informal settlements in addition to wastewater treatment facilities as an important source of CECs in aquatic ecosystems.

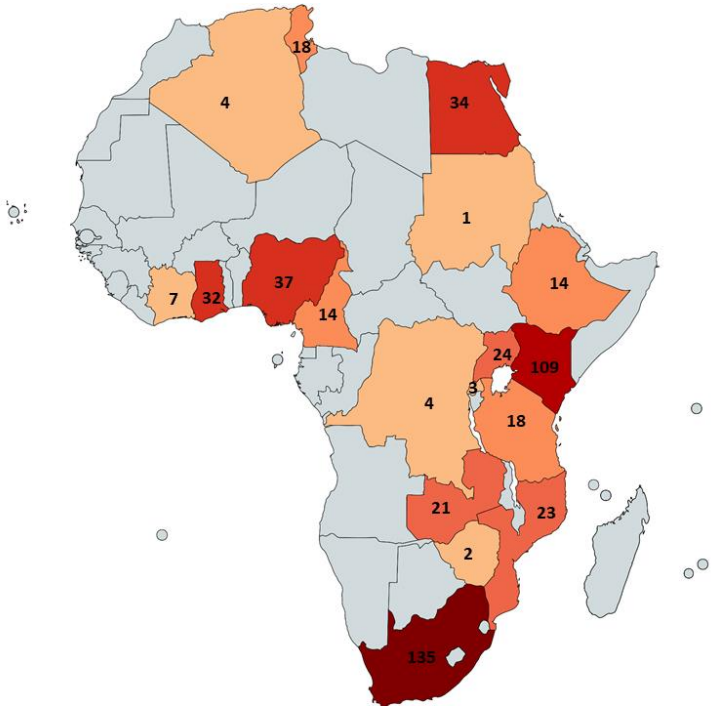

Figure 1.3: Number of organic micropollutants reported in wastewater, surface water, ground water, drinking water, sludge and sediments in Africa. Modified from K'oreje et al. (2020)

Difference on the pharmaceuticals detected in European and African aquatic ecosystems can be clearly observed. For example, acetaminophen is sold without prescription as an over-the-counter (OTC) drug and is the most consumed drug in Kenya (74.6%) (K'oreje et al., 2012). The maximum concentrations reported for acetaminophen in African environment was approximately 215 times higher than concentrations found in European studies (Fekadu et al., 2019) despite the medicine also being the most consumed in Europe. The high detections could be linked to poor sanitation especially in informal settlements (Fekadu et al., 2019; K'oreje et al., 2012) since acetaminophen is removed up to 99% in WWTPs (K'oreje et al., 2018). Contrary to antibiotics being frequently detected in developed countries, antiretroviral (ARVs) drugs are often detected at high concentrations (K'oreje et al., 2018, 2020; Madikizela et al., 2019; Ncube et al., 2018; Paul., 2015) in African environments. In a review done by Fekadu et al. (2019), lamivudine concentrations were reported at highest concentrations (up to 167 µg/L) in Kenyan wastewater (K'oreje et al., 2016) among all the surveyed pharmaceuticals in their literature. These differences could be linked to the national prevalence of HIV/AIDS in Kenya at 4.9% (National AIDS Control Council NACC, 2018). In South Africa, 7.1 million

of the population is infected with HIV. In addition, the country also has the biggest antiretroviral treatment program (Ncube et al., 2018).

In addition to the ARVs, analgesics/anti-inflammatory drugs and antibiotics also show high maximum concentrations in surface waters (K'oreje et al., 2020). The review done by Madikizela et al. (2019) reported naproxen, ibuprofen, diclofenac, ketoprofen and fenoprofen as the most prominent NSAIDs in the aquatic environment with ibuprofen being detected in higher concentrations in South African water resources. A study performed in the Ugandan part of Lake Victoria, which is the second largest freshwater lake in the world and Africa's largest, showed concentrations of up to 5.6 µg/L reported for sulfamethoxazole (Nantaba et al., 2019). Pharmaceuticals have been found in sludge and in sediment (Agunbiade & Moodley, 2015; Kimosop, et al., 2016; Matongo et al., 2015) with concentrations above 1000 µg/kg (sulfamethoxazole, sediment) and 43 mg/kg (efavirenz, sludge) (K'oreje et al., 2020).

Concerning pollution with pesticides, most of the research carried out in Africa mainly focused on organochlorine (Elibariki & Maguta, 2017; Lalah et al., 2003) and few organophosphate (K'oreje et al., 2018, 2020; Otieno et al., 2015) pesticides. However, the production and use of other new and more polar pesticides are on the rise and should be included in monitoring studies. As an example, neonicotinoid pesticides including imidacloprid, thiacloprid and acetamiprid are currently among the largest selling class of insecticides globally and also in Africa. Pesticides have been detected in ground water in Kenya (K'oreje et al., 2018), surface water in Rwanda (Houbraken et al., 2017), drinking water in Ethiopia up to 138 µg/L (fenproimorph) (Mekonen et al., 2016). Pesticides have also been found in biota. For example, brown fur seals (*Arctocephalus pusillus pusillus*) blubber from Cape Cross/Namibia had internal concentrations of p,p'-DDT, p,p'-DDD and p,p'-DDE ranging from 3-73 µg/kg, 2-59 µg/kg and 6-1030 µg/kg respectively with p,p'-DDT recording up to 80% dominance of the total DDT burden (Olisah et al., 2020).

Studies on the occurrence of industrial chemicals and endocrine disrupting compounds in African continent are scarce. A study done by Damkjaer et al. (2018) found 17-OH-pregnenolone (0.1 ng/L) and estrone (445 ng/L) in the influent, and lower concentrations of estrone (45 ng/L) at the effluent of WWTP in Tanzania. Chokwe & Okonkwo (2019)

found twelve organophosphorus flame retardants with concentrations ranging between 67.8 ng/g dry weight (dw) to 278 ng/g dw in sediments from Vaal River in South Africa.

1.6 Risk assessment

Aquatic biota are continuously or intermittently exposed to a mixture of CECs released into the environment causing potential risk to exposed organisms. Different approaches have been applied to investigate adverse effects on aquatic organisms. A chemical's mode of action would either take the independent action (IA) or concentration addition (CA) approach (Backhaus & Faust, 2012; Loewe, S. & Muischnek, 1926). The CA concept assumes that individual compounds have similar mode of action and act on similar biological target (Altenburger et al., 2003; Mansano et al., 2017). When a cocktail of chemicals is present in the aquatic environment, the compounds could be potentially harmful due to synergistic interactions (CA) which could lead to undesirable consequences to both humans and other organisms (Sousa et al., 2018). On the contrary, IA concept assumes that compound mixtures act on different subsystems of the organism and that the impaired subsystem affect the end point independently of each other (Backhaus & Faust, 2012). These two models have different approaches and concepts, however, deviations between the predicted effect concentrations are below a factor of 4 (Vighi et al., 2003). Compounds with the same mode of action can be accurately predicted by the CA model however it has also been known to be more conservative resulting in prediction of higher toxicity compared to the IA model (Belden et al., 2007; Vighi et al., 2003).

The risk for individual species can be accessed using two approaches: Toxic units (TU) (Altenburger et al., 2003; Sprague, 1970) or risk quotient (RQ) (European Commission, 2003). The TU is applied for the ecological risk assessment of measured environmental concentrations (MEC) which is normalized by the effective concentration (required to induce 50% effect on the tested organism, Equation 1.1) (Ccanccapa et al., 2016). A TU value greater than 1 results in an effect on half of the exposed organism. Risk assessment based on TU is limited on the toxicity of single species and is not regarded as protective threshold (Munz et al., 2017). Summing up the individual TUs for each site could be used to show site specific toxic stress which is important in mixture toxicity. For RQ (Equation

1.2), the risk is expressed as a ratio of the MEC to the predicted no-effect concentration (PNEC) for the compound. This risk could be classified in three levels: low risk (RQs 0.01-0.1), moderate risk (RQs 0.1-1) and high risk (RQs>1) (Hanna et al., 2018). Most standard acute toxicity test are based on *Daphnia magna* representing crustaceans, fish (*Pimephales promelas*) and algae (*Selenastrum capricornutum*) for ecotoxicological evaluations of chemicals (Ccanccapa et al., 2016).

Equation 1.1:

$$TU = \frac{MEC(mg\ L^{-1})}{EFFECT\ CONCENTRATION\ (mg\ L^{-1})}$$

Equation 1.2:

$$RQ = \frac{MEC(mg\ L^{-1})}{PNEC\ (mg\ L^{-1})}$$

For communities, the risk could be predicted based on multi-substance potentially affected fraction (msPAF) (De Zwart & Posthuma, 2005; Posthuma & De Zwart, 2006, 2012) which describes the effect of chemical mixtures on species abundance. In the msPAF model, CA is applied to calculate single risk of compounds with a shared toxic mode of action (TMOA) followed by IA to sum the toxicity risks of each TMOA (Rämö et al., 2018). This approach assumes that species in an ecosystem have varied sensitivity to contaminants, therefore the risk to a single compound can be expressed using species-sensitivity distribution (SSD) and the proportion of the species likely to be affected (De Zwart & Posthuma, 2005). An SSD describes compound exposure and the potentially affected fraction (PAF) of species in a community (Munz et al., 2017; Posthuma & De Zwart, 2012) therefore integrating the effect of a substance on several species (Munz et al., 2017) in risk assessment.

1.7 Ecotoxicological effects of organic emerging chemicals of concern in the environment

Continuous exposure to low or sub-lethal concentrations of some CECs can result in adverse effects also to non-target species. For example, the introduction of neonicotinoids in the mid-1990s raised concern of potential negative effects on honey

bees and bumblebees. This prompted the European Food Safety Authority (EFSA) to undertake risk assessment on clothianidin, imidacloprid and thiamethoxam and found that their use on certain flowering plants posed a risk to bees and therefore a partial ban was imposed by the European Union (Wood & Goulson, 2017). In the aquatic ecosystem, these contaminants have been shown to cause adverse effects on biota resulting in abnormal regulatory responses in exposed organisms (Bernal-Rey et al., 2020). There is insufficient knowledge especially from Africa with regards to toxicity, impacts and behavior of CECs for monitoring and regulation. However, long-term ecological risk is increasingly recognized and regulations will be expected over the next decades (Sui et al., 2015).

Pesticides such as aldrin, DDTs and dieldrin are considered as persistent organic pollutants (POPs) which remain in the environment for many years, are toxic to organisms, and bioaccumulate resulting in carcinogenicity, neurological and developmental toxicity (Sousa et al., 2018). In the risk assessment carried out by Duquesne & Küster (2010), exposure to organophosphate paraoxon-methyl resulted to cholinesterase inhibition and reduced rate of algae consumption of *D. magna* at 1 and 1.5 µg/L respectively. A study carried out by Pham & Bui (2018) showed that the tropical *D. lumholtzi* was more sensitive (LC_{50} 3.41 µg/L) to the organophosphate diazinon compared to the temperate *D. magna* (LC_{50} 4.63 µg/L). Pyrethroids have also been found to be highly toxic to crustaceans since they induce neurotoxic effects through modulation of sodium channels (Rose et al., 2016). Among the pharmaceuticals tested on *D. magna* by Kim et al. (2007), the oral medication diltiazem used to treat hypertension was most toxic with an EC_{50} of 8.2 mg/L compared to carbamazepine (EC_{50} 76.3 mg/L) after 96 hours of exposure. Mixtures of pharmaceuticals have also been shown to exhibit greater effect than individual compounds in a study carried out by Petrie et al. (2014). For example, carbamazepine and clofibric acid exhibited stronger effects to *D. magna* during immobilization tests than single compounds at the same concentration (Petrie et al., 2014).

Algae play an important role in aquatic ecosystems as producers and also for carbon fixation. However, biocides including antifouling agents, herbicides and fungicides are

known to be toxic to algae (Cedergreen & Streibig, 2005). These compounds interfere with the photosynthetic ability of the microalgae in the aquatic ecosystem (Booij et al., 2015). In a bioassay performed by de Almeida et al. (2017), the biocides bifenox, metribuzin, dichloflunid, aclonifen and triclosan inhibited growth of algae with bifenox being most toxic (EC_{50}=10nM). In addition, metribuzin and dichloflunid exerted a significant effect on photosystem II efficiency (EC_{50}=70 nM and 481 nM respectively). The herbicide diuron was also found to be toxic to algae as it binds to plastoquinone on D1 protein therefore blocking the electron transfer in Photosystem II resulting to photosynthesis inhibition (Mansano et al., 2017). Microalgae are also affected by antibiotics (Kümmerer, 2009). Among the pharmaceuticals studied by Grung et al. (2008) on the algae bioassay, sulfamethoxazole (RQ 0.14) contributed to the high toxicity. Pharmaceuticals were also shown to cause additive effect on growth inhibition when exposed to sulfa-antibiotics (217–784 µg/L) in combination with trimethoprim and pyrimethamine (Vasquez et al., 2014). Wilson et al. (2003) observed a shift in community structure and decrease in algal richness with increasing concentrations of the antibiotic ciprofloxacin, the antimicrobial agent triclosan and the surfactant tergitol NP10.

Fish are vulnerable to water quality changes that occur in the water ecosystem especially due to pollutants which are regularly discharged. Pesticides, such as organophosphates and carbamates, have been shown to cause acetylcholinesterase inhibition in fish. Acetylcholinesterase activity is important for several physiological functions including locomotor activity, predation evasion, feeding orientation and social interaction (Bernal-Rey et al., 2020). Recently, Femi et al. (2020) showed highest toxicity on *Poecilia reticulate* when exposed for 24 hours to cypermethrin (LC_{50} 29.12 µg/L), deltamethrin (LC_{50} 36.62 µg/L) and lambda-cyhalothrin (LC_{50} 83.35 µg/L). Additionally, estrogenic exposure to fish (fathead minnows or Japanese medaka) have been widely studied (Farré et al., 2008; Kidd et al., 2007). A study reported vitellogenin (VTG) induction in fathead minnows after exposure to ethynil-estradiol (EE2) at concentrations below 1 ng/L (Farré et al., 2008). This indicates a risk towards feminisation of male fish, at effective concentrations of the steroid hormone (Damkjaer et al., 2018). Additionally effects on reproduction, hormonal responses and xenobiotic biotransformation system on fish have also been reported (Farré et al., 2008). Pharmaceuticals have the potential to alter

physiological reactions in fish by binding to nuclear receptors and mediate the effects on transcription or translation levels (Burkina et al., 2015). For example, diclofenac exposure to rainbow trout at 5 µg/L resulted in alterations of the kidney and necrosis on the gills (Schwaiger et al., 2004). Exposure of bass fish to the antidepressant venlafaxine for 6 days significantly decreased the brain serotonin levels leading to reduced predation ability thereby indicating the appetite suppression effect of venlafaxine (Bisesi et al., 2014).

1.8 Analytical identification of CECs

Screening of environmental samples including water, sediments and biota for the presence of emerging contaminants is an essential step in the control of contaminants in the water cycle from source to use. In the recent decades, there has been numerous innovations in instrumentation and method development to support the analysis of trace organic compounds present in the environment (Perez-Fernandez et al., 2017). Since these compounds occur at sub ng/L concentrations, sensitive instrumentation are required in combination with efficient sample preparation methods. High throughput by automation and short analysis time have also been major improvements in instrumentation. Consequently, this has resulted in broadening of the spectrum of compounds that can be analyzed in the environment (Hird et al., 2014).

Over the last decades, gas chromatography (GC) and liquid chromatography (LC) have been used in the characterization of these compounds in environmental samples. The choice of chromatographic technique to be applied is dependent on volatility, polarity and thermal stability of the analytes of interest (Hogenboom et al., 2009). Mass spectrometry (MS) has been increasingly applied in environmental chemistry with the aim of identification of the growing number of CECs (Masiá et al., 2014). Over the years, MS techniques have evolved from popular GC or LC coupled to an MS with a single quadrupole mass analyser, to tandem mass spectrometry (MS/MS) and, more recently, high-resolution mass spectrometry (HRMS) with time-of-flight (TOF) and Orbitrap mass analyzers especially for LC-MS (Hernández et al., 2012). The use of alternative mass analyzers and their combinations such as the quadrupole time-of-flight (QTOF) has improved the capabilities of the instruments available. This has been brought about by

the need to have a detection method that is more specific and sensitive for accurate identification and quantification of compounds at very low concentrations and with lower method detection limits.

Liquid chromatography - mass spectrometry methods (LC-MS) are used mostly for non-volatile, polar and thermolabile compounds (Hogenboom et al., 2009). Like the GC, conventional detectors such as ultra violet (UV), diode array or fluorescence (Tadeo, 2008) were not selective enough for trace contaminant identification and could only provide limited confirmatory evidence (Tadeo, 2008). Advancements in detection and column technology have enabled the wide use of LC-MS in contaminant characterization comparable to that offered by GC-MS (Perez-Fernandez et al., 2017). Most compounds that are not GC amenable could be separated by the LC. These developments in chromatography has enabled more rapid, efficient LC separations and provide possibilities for analysis of ionic or polar compounds (Hird et al., 2014). LC injections are much more simple and precise compared to GC-MS (Tadeo, 2008).

Tadeo (2008) detailed the working principle of LC-MS. Briefly in LC- MS analysis, a sample prepared in suitable solvent is injected by an automated syringe into the LC column via a port consisting an injection valve and sample loop. It is however important to note that environmental matrices are quite complex and the CEC occur at trace concentrations. Sample pre-treatment and removal of matrix interference (matrix effect) is essential before chromatographic analysis. In the column, separation is based on differences in hydrophobicity by partitioning between the stationary phase and the mobile phase. Reserved-phase (RP) columns are the most widely used separation techniques in LC (Tadeo, 2008) with C_8 and C_{18} stationary phases on silica being commonly used. In RP separation, the stationary phase is apolar while the mobile phase is polar. Operational properties such as solvent pH, temperature, composition and flow rate are important parameters for good separation.

After the column, the sample moves to the MS system where it is first ionized to generate gas-phase ions from the LC eluent. Ionization of the analyte could be thermally, by electric fields or by impacting energetic electrons, ions or photons (Gross, 2011). The MS system

consists of an ion source, a mass analyzer, and a detector operating under high vacuum conditions. The principle of mass spectrometry involves the use of an ion source to generate ions, which then undergoes separation with a mass analyzer based on their mass-to-charge (m/z) ratio, and finally, the ions get detected both qualitatively and quantitatively by their m/z ratio and abundance (Gross, 2011; Hernández et al., 2012). The process of ionization is achieved through evaporation, pressure reduction and ionization (Ho et al., 2003). Several ionization techniques exist including atmospheric pressure interfaces (API), electrospray ionization (ESI), atmospheric pressure chemical ionization (APCI), and the less commonly used atmospheric pressure photoionization (APPI). Electrospray ionization coupled to MS/MS remains the most common technique employed for the determination of chemical contaminants (Tadeo, 2008) due to its specificity and sensitivity (Ho et al., 2003). Recent instruments have a data system that processes and stores data from the detector.

There has been an increased interest on the development of HRMS systems in environmental science for targeted and non-targeted analysis owing to its sensitivity in full-scan acquisition mode, high resolving power and high mass accuracy (Hernández et al., 2012). Since not all compounds can be identified through target analysis, suspect screening and non-target screening approaches have been developed for compound identification. Recent technological advancement have enabled the use of a single instrument to perform several tasks including pre- and post-target analysis, retrospective analysis, discovery of metabolite and transformation products, and non-target analysis (Hernández et al., 2012).

Screening techniques in identification of compounds

Target analysis/screening

Target analysis is the most conventional approach based on a pre-determined list of known analytes. It involves the use of reference standards (Moschet et al., 2013), acquisition methods of known analytes and applying already validated methods (Hird et al., 2014), and therefore limited to few compounds with optimized methods. Sometimes it could be referred to as target screening when it involves analysis of hundreds of

compounds using a multi-residue approach capable of detecting several compounds in the sample. Characteristic ions for the analytes are selected prior to sample analysis. A list of compounds of interest is prepared from already existing information such as pesticides or pharmaceuticals registered/banned for use, compounds frequently detected in the environment (e.g NORMAN list, www.norman-network.com) or those put in priority list based on legislation. Although the list could be extensive when performing target screening, positive identification of a chemical requires reference compounds. The focus during data processing is only limited to the available standard and any unexpected compounds cannot be detected if its ions are not selected and therefore missed even if it is present in the sample (Tadeo, 2008). Due to growing interest in the field of CECs in the environment, the list of target analysis is always changing to cover all possible compounds.

Suspect screening

Since it might be impossible to have all reference standards to perform target screening, new approaches have been developed for identification of these contaminants using prior knowledge from various sources and without reference compounds (Chiaia-Hernandez et al., 2014). Similar to the list used in determining contaminants of interest in target screening, these lists could be applied during suspect screening to identify candidate compounds. Suspect screening requires the knowledge on compounds such as chemical structure and exact mass. Exact mass and isotope pattern is used in calculating the molecular formula plus or minus the expected adduct(s) of the suspect substance thereby enabling the screening of this substance in the sample (Schymanski et al., 2015). Identification of peaks during suspect screening can be aided by comparison with MS/MS spectra with libraries from vendors (Hird et al., 2014) or in-silico fragmentation tools available from various softwares such as MetFrag (Ruttkies et al., 2016). In addition, online substance databases including PubChem, ChemSpider can be used in combination with spectral databases such as Massbank and mzCloud for compound identification.

Non-target screening

In order to reduce the gap of unknown compounds in the samples, non-target screening approach is implemented which aides in detecting all signals detectable with the analytical instrument (Gago-Ferrero et al., 2015; Hird et al., 2014). Contrary to suspect screening, no prior information on the exact masses is available during non-target screening (Moschet et al., 2013). Non-target identification starts from the exact mass, isotope, adduct and fragmentation information since no structural information exists (Schymanski et al., 2015). A full spectral acquisition is carried out with high selectivity from the high mass-resolving power or MS/MS. This makes it possible to extend environmental monitoring to compounds which are no longer in production or use, certain metabolites, recent compounds not included in monitoring, or compounds whose analytical standards are not available. For this reason, non-target analysis provides a greater scope than target approach (Hird et al., 2014). It results into thousands of peaks which can be used for different purposes including pattern analysis and peak clustering. In addition, using statistical tools, the peaks generated can be prioritized based on frequency, site specific, intensity, and newly emerging contaminants for further identification (Krauss et al., 2019). In addition, the peaks can be deposited in mass spectra repositories such as MassBank (Horai et al., 2010) making it accessible to the public for use in identification of contaminants.

Performing non-target screening requires highly sensitive instruments capable of generating sufficient information for structure elucidation by either mass spectra for interpretation or accurate mass information for deducing empirical formulae (Tadeo, 2008). This therefore limits the approach to HRMS instruments. Data generated from non-target acquisition could be processed by exact mass filtering or by searching databases relating to molecular formulae. Exact masses are extracted from the total ion current (TIC) with a narrow mass window or time window if information on retention time is available and the results are reported as “hits”. The development of non-target analysis techniques has not only benefitted environmental scientists but has been a key component in analysis of food contamination (Hird et al., 2014) by providing rapid and accurate screening of unknown compounds.

1.9 Schistosomiasis and pollution

Risk assessment involves evaluating current status and changes that occur after exposure of organisms to chemical contaminants. The continuous exposure to these chemicals may gradually lead to irreversible changes including change in species composition and abundance due to a decrease in most sensitive species or increase of the resistant ones (Liess & Von Der Ohe, 2005). Furthermore, occurrence of these contaminants in the environment may lead to either direct or indirect effects on human health. Several studies have shown the impact on pesticides on the trematodes and their vectors such as snails which cause morbidity on human beings (Bakry et al., 2015; Halstead et al., 2017; Mohamed, 2011; Monde et al., 2016).

Human schistosomiasis is an infectious snail-borne disease caused by the trematode of the genus *Schistosoma* (Sang et al., 2014). It affects approximately 243 million people globally (Halstead et al., 2017) with 85% of this population in sub-Saharan Africa including Kenya (Sang et al., 2014). The adult schistosomes reside in the intestines or bladder within the human body (Figure 1.4).

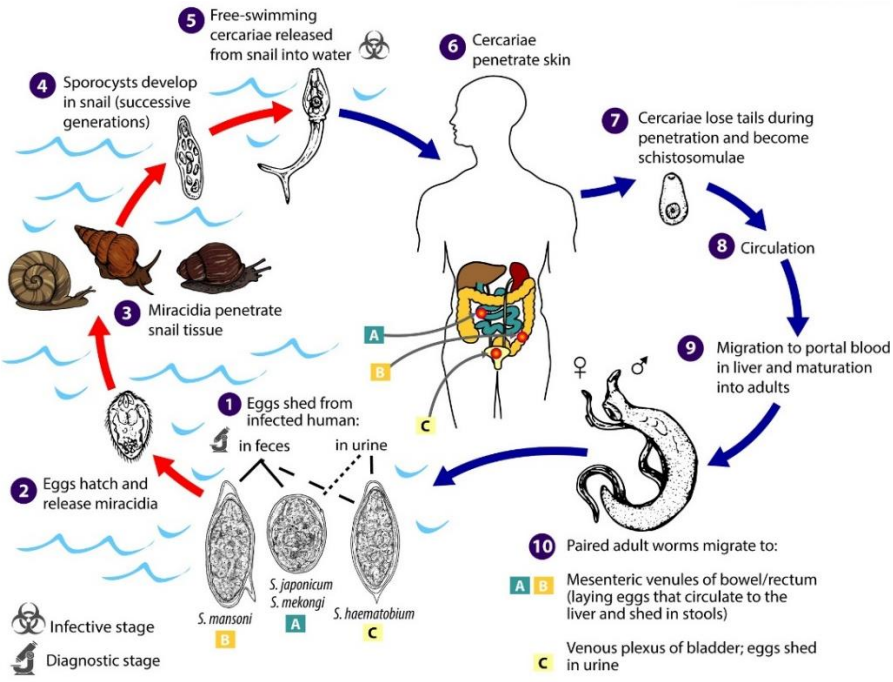

Figure 1.4: Life cycle image of the *schistosoma spp.* parasite. Image taken from center for disease control (CDC, https://www.cdc.gov/dpdx/schistosomiasis/index.html)

Briefly, the lifecycle of the parasite (Figure 1.4) begins when the parasite eggs are released into the environment via human urine or feces during defecation. When the eggs get into contact with water, they hatch as miracidia and penetrate into aquatic snail tissue (intermediate host). It reproduces asexually shedding free swimming cercariae, which then infect humans through skin contact with infested water (Arostegui et al., 2019). Infection with the parasite causes mild to severe systemic diseases including anemia, diarrhea, and organ-specific pathologies including kidney disease (Grimes et al., 2015).

The occurrence of schistosomiasis, which is endemic within our study area (Sang et al., 2014), is modulated by the biological component, socio-cultural processes and disorderly urban spaces (de Souza Gomes et al., 2014). In the study area, the disease burden is exacerbated by low hygiene standards, poverty and poor sanitation. Sang et al. (2014) reported an overall prevalence of 13% of the children population studied in the Lake Victoria Basin. In the same study, they noted an infection hotspot in Migori with prevalence up to 80%. The high incidences reported in this region could be associated with occupational and recreational activities including fishing, rice cultivation in paddies, sand harvesting, swimming and performing domestic chores.

Current control strategies, both globally and in Kenya, include the use of antischistosomal drug praziquantel (Grimes et al., 2015; Hanelt et al., 2010). However, without improving the environmental conditions, the population is at risk of reinfections. Therefore, this necessitates the periodic mass administration of the drug annually or biannually (Grimes et al., 2015). Biological control strategies involving the use of natural snail predators could be a targeted and effective way of reducing disease transmission, although only for native species (Arostegui et al., 2019). Non-native species are also potentially effective but might result in undesirable impacts such as outcompeting the local communities.

Snails responsible for disease transmission live in water and are exposed to the physical-chemical and biological conditions including host-parasite-environment interactions which may influence transmission. A study performed by Halstead et al. (2017) showed that fertilizer and atrazine increased snail densities by increasing the algae which snails feed on and in addition, the chlorpyrifos decreased snail predators. Monde et al. (2016) showed that biological control of *Schistosoma*-host snails using catfish may be negatively

affected when exposed to endosulfan-polluted water at concentrations between 0.03 and 1 µg/L. This could result in an increase of the disease prevalence. On the contrary, Mohamed (2011) found that exposure of *B. alexandrina* snails to the organophosphorous profenophos reduced infection with the parasite *Schistosoma mansoni* hence reducing disease burden. This knowledge gap prompted the formulation of this dissertation in order to fully understand the effects of pollution on trophic interactions among the *Schistosoma* transmitters (snails), predators (water bug, *Belostoma flumineum*) and competitors.

1.10 The Lake Victoria South Basin - a case study

The study area is located in the south western Kenya within the Lake Victoria Basin in East Africa (Figure 1.5).

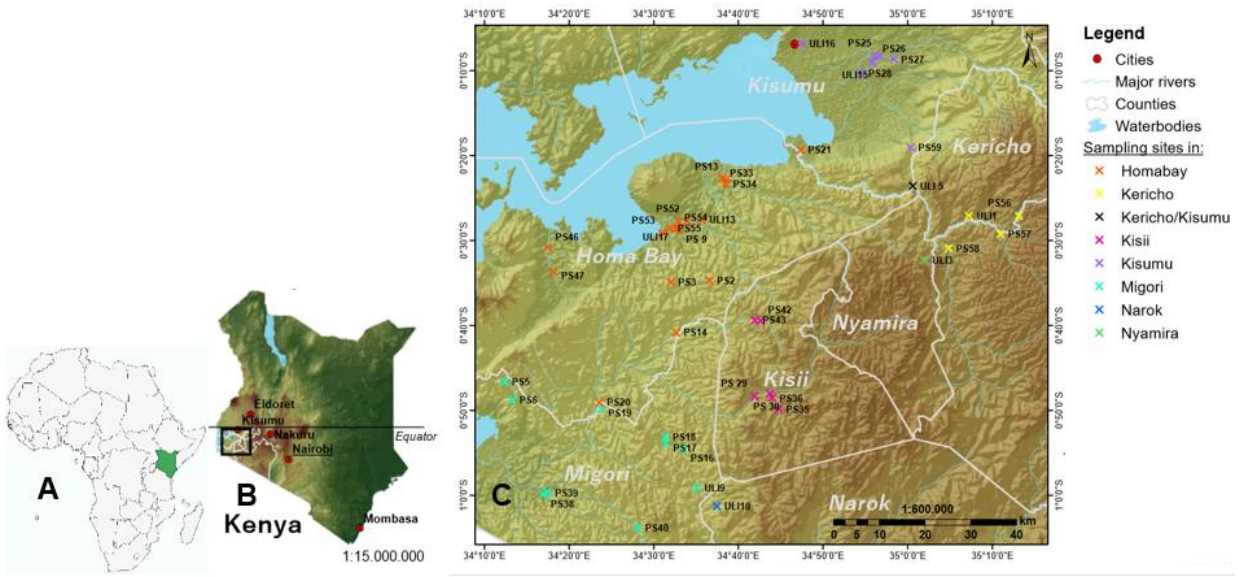

Figure 1.5: Map of Africa (A) showing the country of study Kenya (B) showing an overview of study area and C) Sampling area within the Lake Victoria South Catchment areas. Data used for mapping was extracted from https://africaopendata.org.

The major rivers in this area drain into Lake Victoria, which is the second-largest fresh water lake in the world and largest in Africa by area. The Lake Victoria Basin is further subdivided into two: North and South catchment areas. The Lake Victoria South Basin (LVSB), which is also the study area, is located in western Kenya and is estimated to have a surface area of 21,720 Km^2. It covers counties including Kericho, Kisumu, Kisii,

Nyamira, Migori, Narok, Bomet and Homa-Bay (Figure 1.5). The catchment comprises six major drainage systems of Nyando, Sondu, Gucha-Migori, North and South Awach, and Mara which is a transboundary resource shared between Kenya and Tanzania.

The study area experiences hot and humid climate with bi-modal rainfall patterns having long rains from March to May and short rains from October to December. The local population depends on natural resources for their livelihoods with agriculture, fisheries, sand harvesting and quarrying as the main sources. Agricultural land use include both subsistence farming of food crops such as maize and commercial farming of cash crops including sugarcane, rice and tea. Water and sanitation services are mostly provided to residents living in major towns within the LVSB, however, rural areas are not able to access these services. The residents not connected to piped water therefore go to reservoirs and rivers to perform household chores such as cleaning utensils, washing clothes, taking a bath, and thereafter fetch water for use at home. These activities coupled with lack of sanitation facilities results in input of pollutants into the aquatic ecosystem and at the same time predisposes the locals to the *schistosoma* parasite due to prolonged contact with potentially infested water. This results in emergence of infection hotspots within the area. This study area was chosen since schistosomiasis is endemic in this region with high morbidity especially in school going children. Additionally, information related to the water quality with regards to micropollutants is insufficient.

1.11 Research design

The research involved field sample collection and laboratory experiments. Field monitoring was performed in 48 different surface water systems including rivers, rice growing channels, reservoirs and ox-bow lakes in different land use systems including urban towns, industrial and agricultural sites. This was done to be able to get different contamination patterns in the area. Field monitoring was performed in two phases: first phase focused on collection of water, sediments and macroinvetebrates while second phase involved collection of macroinvertebrates only for acute toxicity tests. Water samples were directly analysed without prior sample preparation. Contaminants in the snails were extracted using a modified QuEChERS (Quick, Easy, Cheap, Effective, Rugged and Safe) while pressurized liquid extraction (PLE) was performed on the

sediments. All chemical analysis was performed using liquid chromatography coupled to a high resolution mass spectrometer (HRMS, QExactive Plus MS) from Thermo Scientific. The detailed research design and methodology is included in the respective chapters.

1.12 Thesis vision and objectives

A lot of research on environmental occurrence of CECs has been performed in developed countries. However, there is insufficient knowledge on the same from Africa (see Section 1.5). The few studies existing from Africa are limited in the number of target compounds that could be analyzed. These studies focus mostly on pesticides and pharmaceutical but not on industrial chemicals which have also been shown to occur in the environment and pose hazardous effect of aquatic biota. Additionally, most of the studies have been carried out in urban areas and on wastewater treatment sites. The fate of compounds in African aquatic ecosystems has yet to be fully understood and require investigation. Knowledge gaps also exist on the contribution of CECs on the abundance of S*chistosoma*–host snails and the transmission of schistosomiasis in endemic areas. Therefore, the main scope of the research is to perform a multi-compartment chemical analysis on CECs to gain knowledge on their occurrence and fate in rural western Kenya, and to determine the contribution of pesticides on S*chistosoma*–host snails abundance in this endemic region.

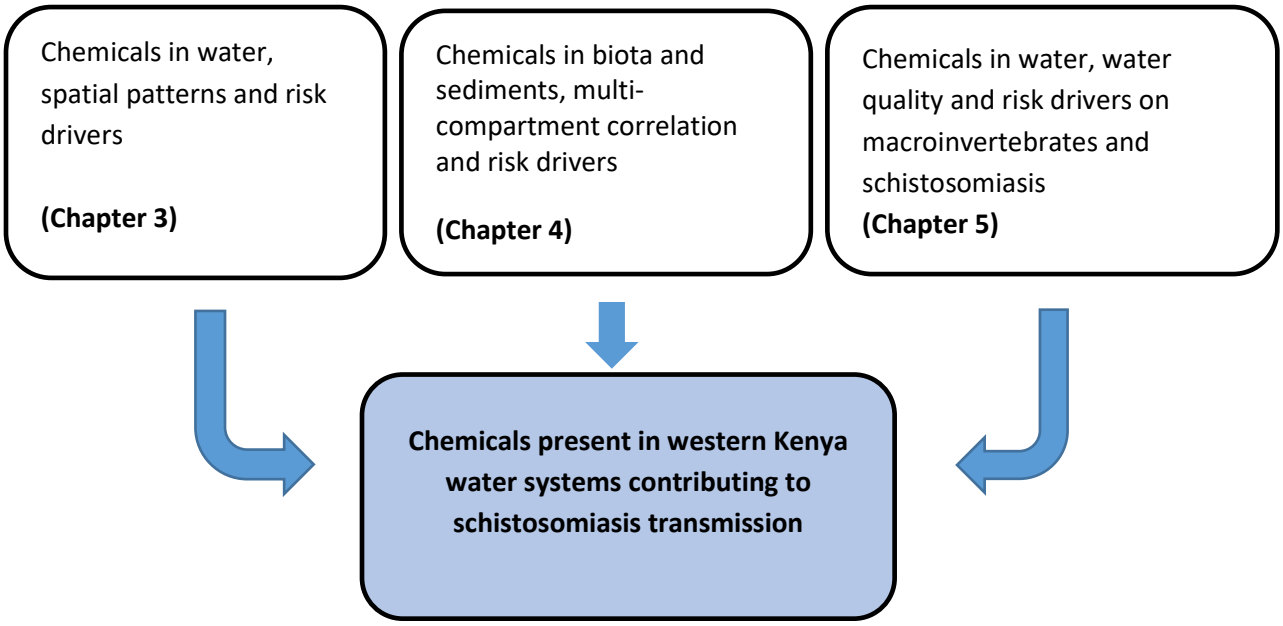

Figure 1.6: The concept of the dissertation divided into three chapters

In order to achieve this goal, three objectives were set for this study:

I. To perform a comprehensive chemical characterization and risk assessment on water samples from surface water systems in rural western Kenya. This objective was achieved in Chapter 3 where an in-depth view of the water chemistry using both target and suspect screening was performed. A multi-residue approach was applied and risk assessment done on the water samples to bridge the knowledge gap on occurrence, risk, and prioritization of CECs in Kenya.

II. To perform a multi-compartment chemical analysis of CECs to investigate the distribution of compounds in aquatic systems of western Kenya. This objective was achieved in Chapter 4 by performing chemical analysis on snail and sediment samples to get a holistic view on contamination patterns in the study area. Risk assessment was also performed on standard test organisms to check for toxicity when exposed to compounds present in the sediment. The compounds found in snails and sediments were compared to those detected in water samples.

III. To investigate the effect of pesticides on the transmission of schistosomiasis in western Kenya. In Chapter 5, the effect of pesticides on the abundance and survival of *Schistosoma*-host snails together with other aquatic organisms was evaluated. Little consideration has been given to understand the ecological consequences of pesticide exposure on host snails which result in schistosomiasis transmission. To achieve this objective, the abundance of snails and other macroinvertebrates found in the field sampling were compared to the pesticide concentrations found in Chapter 3. In addition, acute toxicity tests were performed on snails and macroinvertebrates using two pesticides selected from Chapter 3.

Chapter 2

Summary results and discussion

2.1 Summary results

This section highlights the approaches and summary discussion (2.2 to 2.5) of the individual chapters in this dissertation within the context of CECs monitoring in western Kenyan aquatic systems and the contribution to the prevalence of schistosomiasis.

The main objective of this PhD thesis was to perform a multi-compartment chemical analysis of CECs to gain knowledge on their occurrence and distribution in rural western Kenya, and to determine the contribution of pesticides on the transmission of schistosomiasis. This dissertation comprehensively determined the occurrence and risk assessment of CEC in water (Chapter 3), and in snails and sediments (Chapter 4). In addition, the contribution of pesticides on the abundance of *Schistosoma*-host snails and other macroinvertebrates was investigated (Chapter 5). The specific objectives were addressed in this dissertation:

Objective 1 (Chapter 3): To perform a comprehensive chemical characterization and risk assessment on water samples from surface water systems in rural western Kenya.

Pesticides, pharmaceuticals and personal care products (PPCPs) and industrial compounds were found in the water samples. In addition, two anti-retroviral drugs were quantified through suspect screening which could be linked to the prevalence of HIV/AIDS in the region. Emissions of CECs could be attributed to the local land use; specifically with pesticides which were found in the large agro-industrial sugarcane plantations. Based on risk assessment, a candidate list of compounds was formulated for regulation and monitoring.

Objective 2 (Chapter 4): To perform a multi-compartment chemical analysis of CECs to investigate the fate of compounds in aquatic systems of western Kenya.

By performing a multi-compartment chemical analysis, additional compounds were found in snail tissues and sediment samples. Some compounds were found to be present in all the three compartments, while other compounds were matrix-specific depending on octanol-water partitioning of the compounds. Similar to water results, snail and sediment samples from sugarcane plantation areas were highly contaminated.

Objective 3 (Chapter 5): To investigate the effect of pesticides on the abundance of *Schistosoma*-host snails and other aquatic organisms in the schistosomiasis endemic region of western Kenya.

Schistosoma-host snails were more tolerant when exposed to selected pesticides compared to the sensitive macroinvertebrates. The field and laboratory results showed that pesticide pollution is a major driver to the increased abundance of the host snails which results in higher risk of schistosomiasis transmission.

2.2 Multi-compartment analysis of chemicals of emerging concern in freshwater systems within the Lake Victoria South Basin, Kenya

In the past decades, advancements in technology have enabled the quantification of trace compounds at sub ng/L concentrations. Such advancements include the application of high resolution mass spectrometry (HRMS) due to its improved selectivity, sensitivity and full scan analysis mode (Schymanski et al., 2014, 2015; Vergeynst et al., 2015). A multi-residue method was applied on 48 water, snail and sediment samples to quantify 429 compounds using LC-HRMS.

In the present study, a first comprehensive identification and quantification of CECs including pesticides, biocides, PPCPs and industrial compound has been reported in rural areas within the Lake Victoria South Basin. From the chemical analysis done in Chapter 3 and Chapter 4, 75 compounds were found in water, 30 in snail tissues and 79 in sediments through target screening. An additional three compounds were identified and quantified using suspect screening performed on water samples (Chapter 3). Among the compounds were two anti-retro viral drugs (ARVs, lamivudine and nevirapine). The ARV efavirenz was also present in the snail and sediment samples (Chapter 4). These ARVs are commonly used in the first line of treatment among HIV/AIDS infected individuals. The frequent detection of these ARVs shows evidence of their ubiquitous occurrence in Kenyan waters and may be associated with the high prevalence of the disease among the populations (K'oreje et al., 2018; Ngumba et al., 2016).

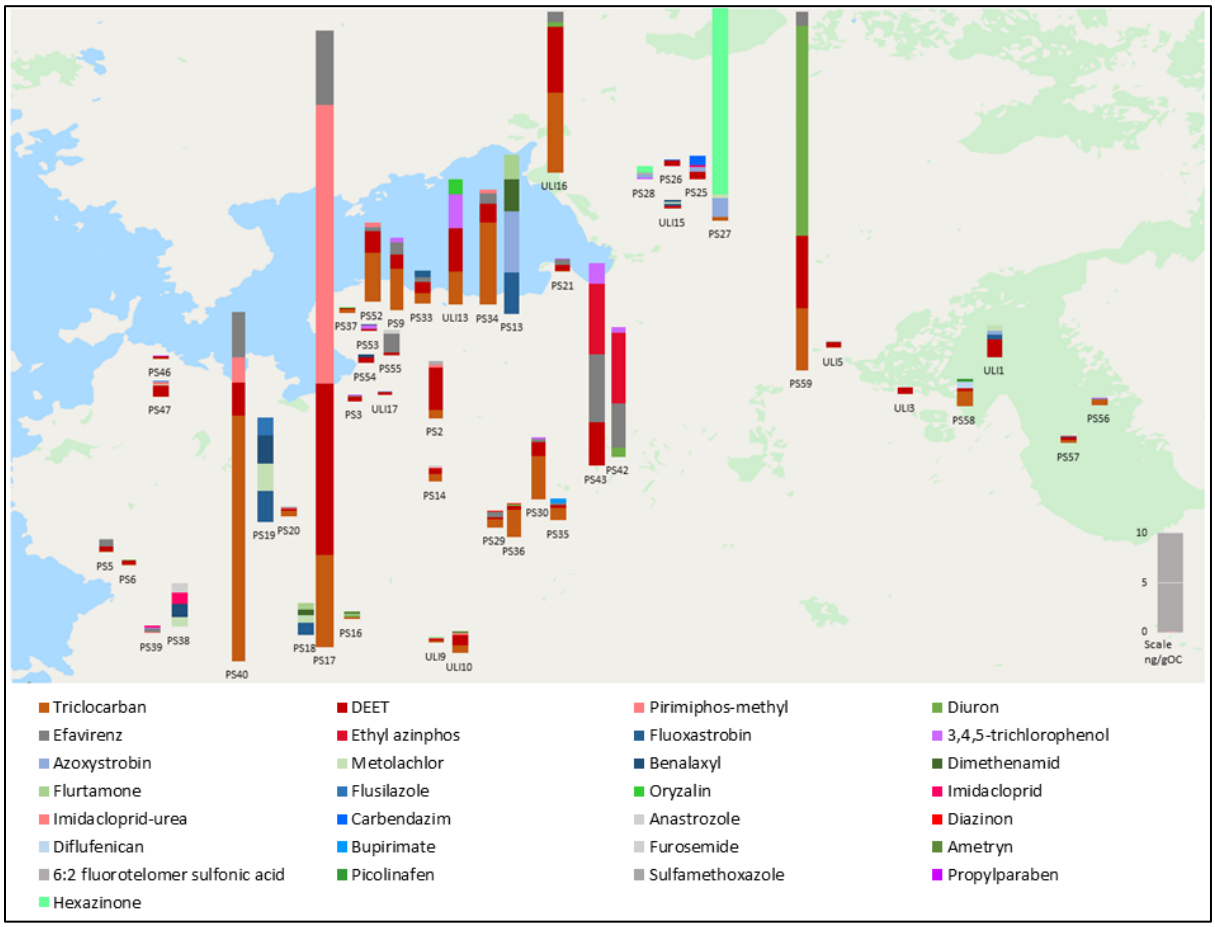

Figure 2.1: Map showing the spatial distribution of four compounds with the highest concentrations found in sediments in each site.

From the samples analyzed in Chapter 3 and 4, pesticides and biocides were the compound class mostly detected in all the three compartments. Individual pesticide concentrations were reported up to 1.5 µg/L in water (hexazinone), 375 ng/g wet weight in snail (bupirimate) and 111 ng/g organic carbon in sediments (pirimiphos-methyl, Figure 2.1). The high detections of pesticides could be due to the presence of large agro-industrial plantations within the study area. Due to surface run off, pesticides are washed into surface water bodies which are typically not protected by riparian strips. The uncontrolled use of pesticides in Kenya also contributes to the occurrence of these compounds in the environment. The concentrations of compounds found (Chapter 3 and 4) could be linked to existing land use in the study area. For example, agricultural land use, especially sugarcane and rice growing plantation, were the greatest input source of pesticides whereas non-point sources from urban centers and informal settlements were

responsible for pharmaceutical inputs in surface water. The concentrations detected in the rural areas from this study were lower than those reported in previous studies for major towns in Kenya (Bagnis et al., 2020; K'oreje et al., 2020; Kairigo et al., 2020a; Kimosop et al., 2016; Ngumba et al., 2016).

By extending the chemical analysis to snails and sediments (Chapter 4), an additional 68 compounds were found that were not previously detected in water such as the fungicides difenoconazole, bupirimate, flusilazole, and tebuconazole. Compounds with a high octanol-water partitioning coefficient ($\log K_{ow}$) majorly accumulate in sediments and lipid-rich biological tissues resulting in long-term exposure of benthic species and substantial risk for the aquatic ecosystem (Li et al., 2017). On the contrary, the highly hydrophilic compound such as acetyl-sulfamethoxazole (A-SMX, log K_{ow} 0.86) was only found in water samples (Chapter 3). A-SMX is a transformational product of the antibiotic sulfamethoxazole used to treat bacterial infections in both humans and veterinary (Fekadu et al., 2019). It is excreted through the urine (Nouws et al., 1991; Zhang et al., 2015) and prone to deconjugation under environmental conditions during wastewater treatment (Bischel et al., 2015; Zhang et al., 2015).

The study resulted in the comprehensive chemical characterization of water, sediment and snail samples thereby bridging the knowledge gap on the occurrence and distribution of CECs in Kenyan aquatic systems, and also in Africa at large.

2.3 Risk assessment of emerging chemicals of concern in freshwater systems

Risk assessment was evaluated in order to understand the ecological relevance of the concentrations found in the aquatic environment in western Kenya. Due to the lack of toxicity data in African species, effect concentrations from the data rich European monitoring and assessment data from Busch et al. (2016) was used to calculate the toxic risk. Thus, the actual risk to native species may deviate from the risk to the standard test species. Acute and chronic risk was evaluated for standard test organisms including fish, crustaceans and algae by calculating TU for water and sediments. Since insufficient data exists on sediment effect concentrations, equilibrium water concentrations were derived from concentrations in sediment organic carbon assuming equilibrium partitioning

(Massei et al., 2018). Toxic units were calculated by normalizing environmental concentrations for water and equilibrium water concentrations for sediment compounds with the effect concentrations which were extracted from Busch et al. (2016).

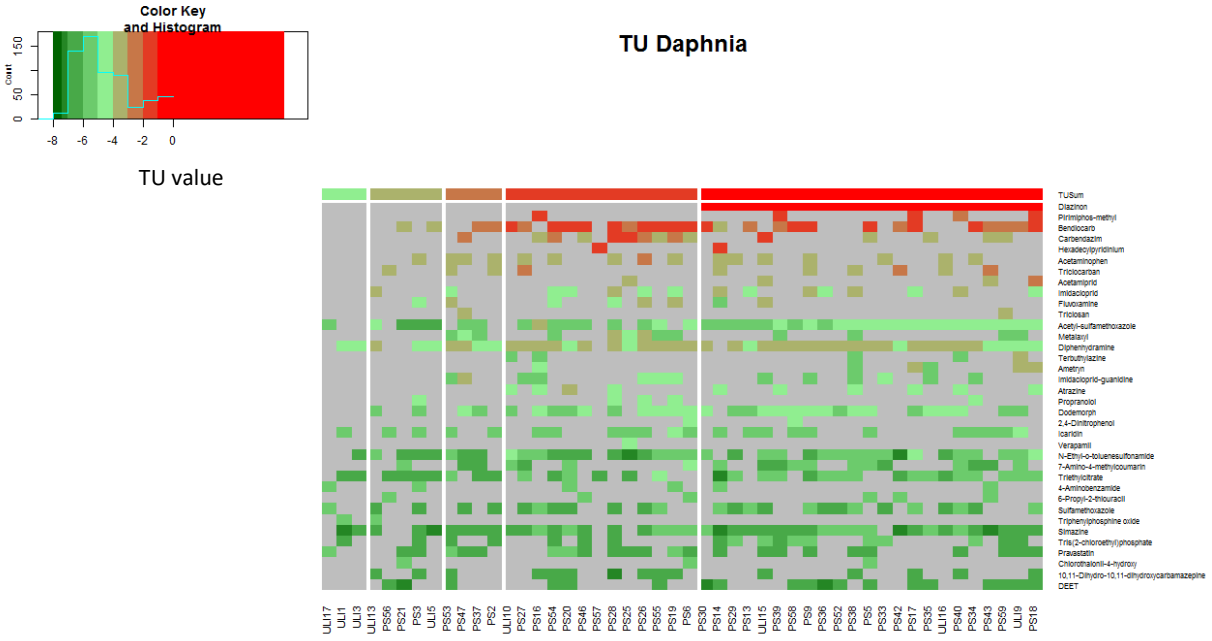

Figure 2.2: Heatmap showing the compounds found in water contributing to the overall risk (TU_{sum}) for acute toxicity in *D. Magna*. Grey color represent sites where the individual compound was not detected.

Generally, risk assessment from compound concentrations found in water (Chapter 3) and sediment (Chapter 4) revealed acute and chronic toxicity for crustaceans while fish and algae experience majorly chronic risk. The risk on crustaceans was driven by a low number of strongly dominating pesticides including diazinon, pirimiphos-methyl, bendiocarb, carbendazim in water (Figure 2.2, Chapter 3) and diazinon, fipronil sulfone and pirimiphos-methyl in sediment (Chapter 4). Since these compounds pose a risk to aquatic organisms, the government should come up with policies on regulating the use of these compounds (see Chapter 6). Acute risk threshold was exceeded in sites impacted by the large agro-industrial sugarcane plantations due to the use of pesticides in the farms. Algae had cumulatively lower risk than crustaceans, with the herbicides hexazinon (from water) and diuron (from sediments) driving the risk for algae toxicity. In Chapter 3 and 4, acute and chronic risk for fish was lower than crustaceans and algae.

Sediments may act as a secondary source of water contamination with pesticides due to remobilization processes resulting in detrimental effects on aquatic species. It is therefore important to select pesticides which meet the needs of farmers while limiting risks to environmental health (Jepson et al., 2020). In agreement with other studies, pharmaceuticals including antibiotics were of limited relevance concerning acute toxicity. However, their occurrence at high concentrations should be of concern for sublethal endpoints with potential effects on fitness, reproduction and behavior of aquatic organisms which are not reflected in the current assessment. The high concentrations found for antibiotics could contribute to potential creation of antibiotic resistance to exposed organisms including human beings (Ebele et al., 2017). There is potential risk on human health considering that the rivers and reservoirs are used by the local communities for daily household activities and also as a source of drinking water. Further research should focus also on the health effects on human beings when they are exposed to these contaminants considering that most household water sources in Kenya are from surface water.

2.4 Pesticide contribution on transmission of schistosomiasis

A lot of efforts have been carried out in order to reduce the morbidity due to schistosomiasis infections. Schistosomiasis control largely focuses on large-scale treatment of school-aged children and the population at risk in endemic areas with the drug praziquantel (Grimes et al., 2015; Mwangi et al., 2014; Onkanga et al., 2016; Utzinger et al., 2015). However, treatment with praziquantel is not effective in long-term relief from infection with the parasite as it does not prevent re-infections in an individual (Mwangi et al., 2014). In order to understand better the disease ecology, we investigated the contribution of pesticides on the prevalence of the disease and, by extension, effect on human health (Chapter 5). Our study in Chapter 5, to the best of our knowledge, is the first study to show ecological effects of pesticides contributing to the transmission of schistosomiasis in a field and laboratory scale.

In Chapter 5, acute toxicity tests using imidacloprid and diazinon concentrations obtained in Chapter 3 was performed on macroinvertebrates and snails. The results showed high tolerance of the host snails while the other macroinvertebrates were very sensitive to the

pesticides. From the field experiment results (Chapter 5), pesticide pollution led to increased incidence of host snails and a decrease in potential competitors. In both field and laboratory setting, the presence of pesticides resulted in a shift of the community composition as lethal and sub-lethal effects were observed in sensitive macroinvertebrates at low concentrations (Chapter 5). Additionally, it was determined that the measured physical chemical parameters of the water had no significant effect on the abundance of host snails. The results in Chapter 5 confirms that pesticides facilitates schistosomiasis transmission by indirectly favoring the tolerant host snails through affecting the competitors but not the predators. Snails are mandatory to close the gap on the schistosomiasis life cycle, therefore their abundance directly results in increased transmission of the disease. From our study, we show that a more comprehensive approach in reducing the prevalence of schistosomiasis would involve the provision of potable water in order to reduce human contact with the vector. Buffer strips should be adhered to in order to reduce pesticide run off into the water system. Additionally, since transmission begins with an infected person defecating or urinating close to a river, adequate sanitation facilities are required especially in endemic areas to reduce release of *Schistosoma* eggs into water bodies.

Schistosomiasis is no longer only a tropical disease but has also been reported in Corsica, France (Quilichini et al., 2019; Utzinger et al., 2015). With the changing climate, increasing pesticide pollution and open borders for human movements, the disease can thrive and spread to various locations worldwide. It is therefore essential that comprehensive studies are performed to understand the disease ecology and prevent an outbreak of the disease.

2.5 Prioritization of CECs for regulation and monitoring in Kenya

Globally, there is an increasing debate over the legislation of CECs due to potential human and environmental health risks (K'oreje et al., 2020). There has already been extensive research on adverse effects of pesticides on the environment mostly in developed countries. Consequently, a lot of data is available to use for legislation including restricting the use of certain pesticides (Hladik et al., 2018; Wood & Goulson, 2017). Legislation for PPCPs, surfactants and other CECs is challenging due to the

insufficient toxicity data (Eckstein, 2012). Nonetheless, developed countries such as the USA and those in the Europe Union have come up with guidelines for monitoring of selected compounds such as steroid hormones (Van Zijl et al., 2017) and pharmaceuticals such as diclofenac (Lapworth et al., 2012). However, such legislation and guidelines in Africa are lacking due to insufficient monitoring and toxicity data on CECs (K'oreje et al., 2020).

In this study, a first prioritization list of candidate compounds for regular monitoring and regulation was developed based on the risk assessment (Chapter 3). For comparison, two approaches for calculating risk was applied based on: i) toxic units (TUs) and ii) risk quotients (RQs). Based on the extent and frequency of exceedance of toxicity threshold values, 16 compounds majorly pesticides starting with diazinon, followed by bendiocarb and hexazinon were ranked highly in the priority list for monitoring. For the RQ criteria, 7 compounds were ranked as a result of exceedance of the threshold values. Diazinon ranked highly in both criteria indicating the need for constant monitoring and formulation of guidelines for its usage (Chapter 3).

In Chapter 5, the sensitive macroinvertebrates were negatively affected while the tolerant host snails dominated the ecosystem when exposed to pesticides. Additionally, human health was negatively affected as pesticide pollution contributed to the dominance of *Schistosoma* host snails which are important in the life cycle of the disease. The occurrence of pesticides in the aquatic ecosystem in the schistosomiasis-endemic areas would result in creation of infection hotspots as the snail abundance would increase the disease transmission. The risk on crustaceans, fish and algae, (Chapter 3 and 4) and on human health (Chapter 5) which is driven by pesticides should be avoided by formulating policies and guidelines on the use of these compounds. Pesticides have been prioritized among the CECs causing adverse effects and hence regular monitoring should be done to avoid the risks on aquatic ecosystem and human health.

This study generated a large amount of data on the occurrence and risk assessment of CECs in Kenyan aquatic ecosystems (Chapters 3, 4 and 5) that could be used by policy makers in formulation of regulations. Extensive financial input and expertise is needed in

monitoring these compounds, and since a lot of compounds have been found in the environment, it becomes a challenge to decide which compounds to be prioritized for monitoring. This study proposes an approach (Chapter 3) that could be easily implemented to prioritize the extensive list of compounds for regular monitoring and legislation especially in a third world country with limited resources.

Chapter 3

Occurrence and risk assessment of organic micropollutants in freshwater systems within the Lake Victoria South Basin, Kenya

Author's contribution statement

Declaration of author contributions to the publication:
Occurrence and risk assessment of organic micropollutants in freshwater systems within the Lake Victoria South Basin, Kenya

Status: Published in Science of the Total Environment

Contributing authors:

Kandie J. F, Krauss M, Beckers LM, Massei R, Fillinger U, Becker J, Liess M, Torto B, Brack W

What are the contributions of the doctoral candidate and his co-authors?

(1) Concept and design

Doctoral candidate: 40%
Co-author last: 50%
Co-author rest of the authors: 10%

(2) Conducting tests and experiments

Doctoral candidate: 90% (organized and carried out field work, sample preparation, chemical analysis)
Co-author 2,9: 10% (supervised and supported chemical analysis)

(3) Compilation of data sets and figures

Doctoral candidate: 75 % (Raw data evaluation, toxic unit calculation, drawing of figures)
Co-author 2,9: 10 %(Supervised data evaluation, Final data evaluation)
Co-author 3 and 4: 15% (Grouping datasets, drawing figures)

(4) Analysis and interpretation of data

Doctoral candidate: 75% (interpretation, Excel analysis)
Co-author 3 and 4: 15% (statistical analysis, Pattern analysis)
Co-authors 2 and last: 10% (Interpretation of chemical and risk data, Advise on risk assessment)

(5) Drafting of manuscript

Doctoral candidate: 80%
Co-author all: 20%

I hereby certify that the information above is correct.

_____

Date and place Signature doctoral candidate

_____

Date and place Signature supervisor

Abstract

The unintended release of chemicals to the environment has led to global concern on water quality prompting widespread research on the occurrence of these compounds in water. While increasing information on organic micropollutants (OMPs) in European water resources is available, there is still limited information on the occurrence of OMPs in African water systems. In this study, a multi-residue analysis covering 428 chemicals using liquid chromatography coupled to high resolution mass spectrometry (LC-HRMS) was performed on water samples collected from 48 surface water sites within the Lake Victoria South Basin, Kenya. A total of 75 compounds including pharmaceuticals, personal care products (PPCPs), pesticides, and industrial chemicals were detected and an additional three compounds (nevirapine, lamivudine and adenosine) were identified through suspect screening. Four compounds including diphenhydramine, simazine, triethylphosphate and acetyl-sulfamethoxazole (A-SMX) were detected in more than 80% of the sites showing their ubiquitous nature in the study area. Individual compound concentrations were detected up to 24 µg L^{-1}. Concentrations above 1 µg L^{-1} were also reported for triethylcitrate, N-ethyl-o-toluenesulfonamide, hexazinone, nevirapine, adenosine and carbendazim. While crustaceans were potentially the taxon at risk for acute toxicity (toxic unit (TU) up to 2) with diazinon driving this risk, lower but substantial acute risk (TU 0.5) was observed for algae. Chronic risks were observed in 11 sites for algae (TU > 0.02) and in 5 sites for fish (TU > 0.01). A total of 16 compounds were prioritized based on frequency and extent of the exceedance of thresholds for acute and chronic risks to algae, crustaceans and fish and another 7 compounds prioritized by applying lowest Predicted No-Effect Concentrations (PNEC). Based on these indicators, this study provides candidate priority compounds for monitoring, assessment and abatement in western Kenya.

3.1 Introduction

The widespread and intensive use of chemicals such as pesticides, pharmaceuticals and personal care products (PPCPs) has given rise to concern on their occurrence in and impact on aquatic ecosystems (Posthuma et al., 2019). There has been an increase in the use of pesticides and PPCPs due to the increasing population and diseases especially in developing countries (Bernhardt et al., 2017; Peng et al., 2018). In this context, Kenya being a developing country, faces great challenges to cater for food, clean water and health needs of its growing population. Agriculture is a main economic branch contributing more than 70% of Kenya's foreign trade which increases 10% on average annually (Moya et al., 2019). This has led to an increased demand for plant protection products. Between 2006 and 2010, the Ministry of Environment, Water and Natural resources reported approximately 36 thousand tons of pesticide importation into Kenya which increased to 54 thousand tons by 2013 (Loha et al., 2018). In addition, (re)emergence of diseases and epidemics (Berger et al., 2010) has led to increased use of pharmaceutical products in the country and to an increased release of these compounds into the environment. Many organic micropollutants (OMPs) are persistent in the environment including carbendazim, clothianidin, diuron and atrazine with several studies showing that exposure to these compounds results in acute and chronic effects to aquatic organisms (Ccanccapa et al., 2016; Liess & Von Der Ohe, 2005; Shahid et al., 2018; Velki et al., 2019).

Monitoring of emerging OMPs has been increasingly done in the western world; however, there is still a big lack of data for Africa (Aus der Beek et al., 2016; Fekadu et al., 2019; Madikizela et al., 2017). Most studies on water quality monitoring in Africa are based on environmental or drinking water guidelines which cover only few OMPs (Gwenzi & Chaukura, 2018). The lack of state-of-the-art analytical equipment to detect concentrations in the ng L^{-1} range is a major obstacle to monitoring of hazardous environmental contamination in many developing countries. Although the occurrence of some pesticides in environmental matrices in Kenya has been monitored since 1987 (Kahunyo et al., 1988), the compounds analyzed were generally low in number. Only very few studies have been performed in Kenyan water systems characterizing pharmaceutical pollution patterns in surface waters. These studies focused on a few

pharmaceuticals (Bagnis et al., 2020; K'oreje et al., 2018; 2016; 2012; Kimosop et al., 2016; Ngumba et al., 2016) and only three studies (Bagnis et al., 2020; K'oreje et al., 2018; Ngumba et al., 2016) reported potential risks on aquatic organisms based on the measured environmental concentrations.

To reduce this knowledge gap, the present study focused on the assessment of surface waters including rivers, drainage channels and dams to obtain information on the extent of pesticides, PPCPs and industrial compound pollution within the Lake Victoria South Basin (LVSB) in Kenya. The aim of this study was to (1) determine the level of OMPs pollution in various surface water systems within LVSB (2), to perform suspect screening for a comprehensive characterization of multi-residue pollution, (3) to undertake risk assessment on aquatic organisms based on toxic units (TU) and (4) for the first time, to prioritize compounds for regulation and monitoring in Kenya.

3.2 Materials and methods

3.2.1 Chemicals

LC-MS grade methanol, formic acid and ammonium formate were obtained from Honeywell, while LC-MS grade water was purchased from Thermo-Fisher. Analytical standards were obtained from various suppliers and at least of 97% purity. More information on the compounds analyzed is presented in appendix A (Table SI-1).

3.2.2 Study area and sampling

The study was performed within the LVSB, Kenya (Figure 3.1) covering Kericho, Kisumu, Kisii, Nyamira, Migori, Narok and Homabay counties. LVSB has an estimated area of 21,720 km^2. A total of 48 sites were sampled including main rivers (11), tributaries (20), dams (8), irrigation field channels and associated rice fields (6), and ox-bow lakes (3). The study sites were selected to cover various types of surrounding land use including agricultural (tea, sugarcane, maize and rice plantations), industrial (sugarcane factory), natural (grassland) and mixed urban and residential areas. Variations in compound detections and concentrations is hypothesized to be related to these different land use systems within the study area.

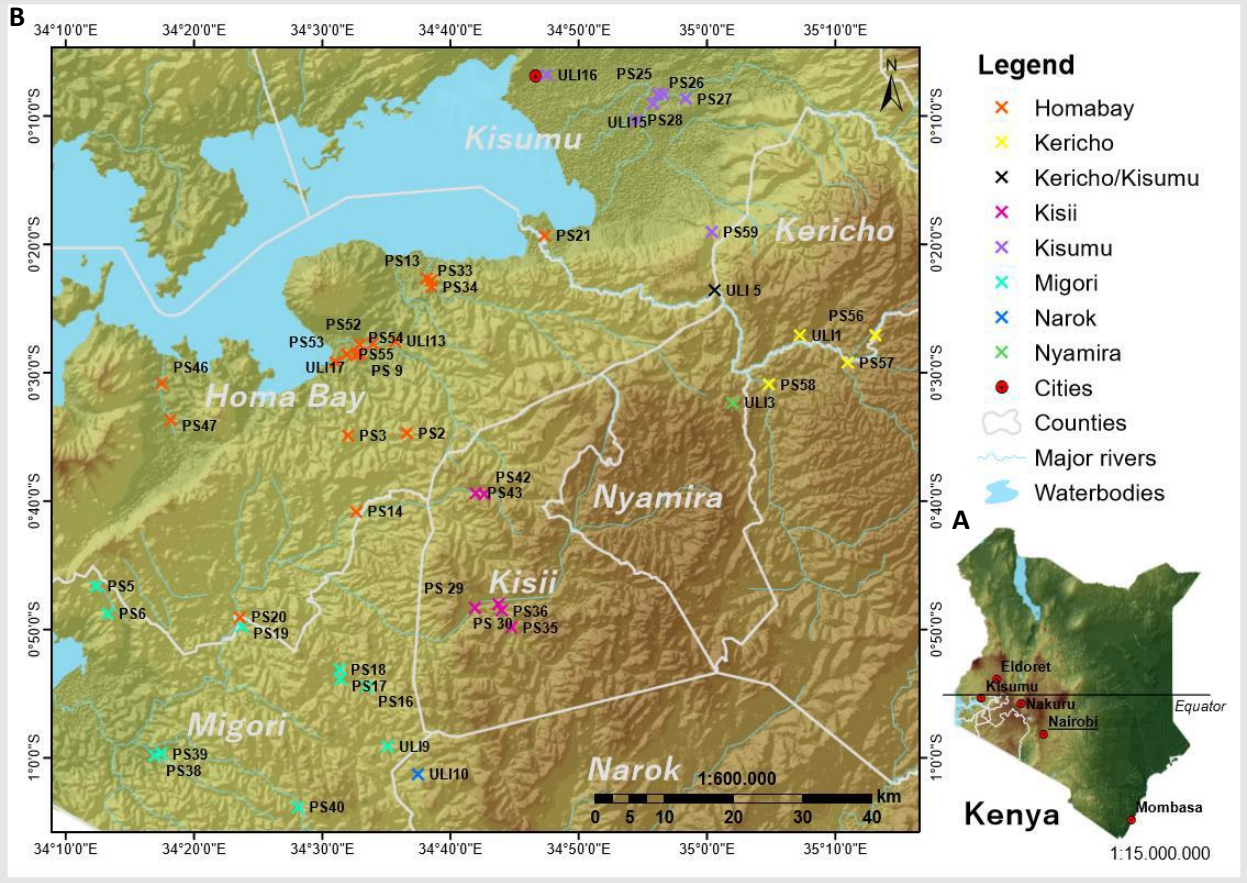

Figure 3.1: Map of the study area A) Kenya and B) Sampling area within the Lake Victoria South Catchment areas. Data used for mapping was extracted from https://africaopendata.org.

Sampling was done between September and October 2017. This coincides with the short rainy season in western Kenya and thus with expected peaks in surface runoff and washout of pesticides. Detailed site descriptions are given in appendix A (Table SI-2). Where feasible, samples were taken upstream and downstream of each land use type in a river system.

At each site, a 500-mL grab sample was taken in a pre-cleaned glass beaker and solids were allowed to settle for 1 minute. An aliquot of 1 mL was transferred to a 2-mL amber auto sampler glass vial (Phenomenex) for chemical analysis. To check for contamination during sampling, a total of 15 trip blanks and 95 sampling blanks (1 mL LC-MS grade water) were taken for all sampling trips and sites, respectively. All samples were stored immediately in a portable freezer at below -4 ˚C and transported to the laboratory. Once in the laboratory, they were stored at -20 ˚C until analysis.

In-situ physico-chemical characterization (flow, temperature, total conductivity, dissolved oxygen (DO), phosphates, pH, and turbidity) was performed during sampling. Additional water samples were collected in pre-cleaned 500-mL Nalgene bottles for further physico-chemical analysis. Laboratory measurements of nitrate, nitrite, ammonium and carbonate hardness were performed using Merck test kits (Merck KGaA, Darmstadt Germany) at the laboratories of the International Centre of Insect Physiology and Ecology (Icipe), Thomas Odhiambo Campus, Mbita, Kenya.

3.2.3 Target and suspect chemical analysis

Due to limited data availability on contaminants in Africa, the selection of compounds for chemical analysis was based on their occurrence in European surface waters and on local use. A target list of 428 compounds (Table SI-1A) including pesticides, PPCPs and industrial compounds covering a wide range of physico-chemical properties were selected for chemical analysis.

In order to allow for a more comprehensive analysis, a suspect screening on additional 233 compounds including pesticides and pharmaceuticals known to be used in Kenya was conducted. Candidate pesticides for suspect screening were selected based on a published list of pesticides registered for use in Kenya by the Pest Control Products Board (PCPB, 2018). A suspect list for pharmaceuticals was compiled based on a list from the Ministry of Health on essential pharmaceuticals for use in Kenya (Ministry of Health, 2016). The compounds were characterized based on log K_{ow}, log D and compound structure (Chemaxon, Budapest, Hungary). Based on their functional groups according to Moschet et al. (2013), the suspect list was filtered for liquid chromatography-electrospray ionization (LC–ESI) amenable compounds. Since water samples were analyzed, the focus was on hydrophilic organic compounds excluding any compound above log K_{ow} 5 (Moschet et al., 2013). Only $[M+H]^+$ and $[M-H]^-$ adducts were included in the screening workflow.

3.2.4 Instrumental analysis

The 1-mL water aliquots were spiked with 25 µL of an internal standard mixture containing 40 isotope-labelled compounds at 40 ng mL^{-1}, 25 µL of methanol and 10 µL of a 2 M

ammonium formate buffer at pH 3.5. Chemical analysis was performed by direct sample injection (100 µL) into a high performance liquid chromatography system (Thermo Ultimate 3000 LC) coupled to a high resolution mass spectrometer (QExactive Plus, Thermo). Chromatographic separation was done at 40°C using a C18 column (Phenomenex Kinetex C18 EVO, 50 x 2.1 mm, 2.6 µm particle size), equipped with a pre-column (5.0 x 2.1 mm) and 0.2 µm in-line filter. The mobile phase comprised water (A) and methanol (B) both with 0.1% v/v formic acid. Analytes were ionized using a heated electrospray ionization (ESI) source with separate runs in positive and negative ion mode. Details on the elution gradient and the mass spectrometer set up are shown in appendix A (Table SI-3, SI-4).

Matrix-matched calibration standards were prepared using filtered water from a pristine reference stream (Wormsgraben) without anthropogenic pollution from the Harz Mountains (Germany), standards of the target substances dissolved in methanol and isotope-labelled internal standards. Eleven calibration levels ranging from 1 ng L^{-1} to 2000 ng L^{-1} were used. Analyte concentrations were determined using internal quantification against internal standards. Due to a limited number of isotope labeled standards, internal quantification was performed using the internal standard whose retention time was closest to that of the analyte. For concentrations > 2000 ng L^{-1}, samples were re-run after appropriate dilution with LC-MS grade water.

Raw data files were first converted into mzML files using ProteoWizard (mscovert frontend version 2.1.0) and processed using MZmine (Version 2.38, Pluskal et al., 2010) including the ADAP chromatogram builder module (Myers et al., 2017). The settings used for MZmine data processing are included in appendix A (Table SI-5). Target compounds were annotated in MZmine (Version 2.38) using a custom database search. Target compounds detected in MZmine were further confirmed and quantified using TraceFinder 4.1 (Thermo, Appendix A Table SI-6). Method detection limits (MDLs) were determined based on a replicate analysis of calibration standards based on US EPA (2011).

For suspect screening, peaks were tentatively annotated using MZmine (Version 2.38). A peak intensity threshold of 10^5 was set in both modes and any peak below this threshold was excluded from the subsequent data processing. In total, 79 peaks were tentatively

annotated in positive and 50 peaks above 10% a.u. intensity in negative mode. Based on the peak shape and baseline noise, the candidate list contained 38 compounds (24 in positive and 14 in negative ionization mode). Data dependent acquisition was then performed on the environmental samples using the LC-HRMS to obtain mass fragments in both ionization modes in separate runs. The MS/MS spectra were extracted in XCalibur (Thermo) and ions with an intensity below 5000 were omitted before further processing. Extracted accurate masses where checked for plausible fragment ion molecular formulas using SIRIUS (V4.0.1, (Böcker et al., 2009; Böcker & Rasche, 2008; Dührkop & Böcker, 2016)). For compound identification, MS/MS spectra were searched against the MassBank (Horai et al., 2010) and mzCloud (www.mzcloud.org) spectral libraries and matched to *in-silico* predicted spectra of the compounds using the MetFrag (Ruttkies et al., 2016) and CFM-ID (Dührkop et al., 2015; Shen et al., 2014; Heinonen et al., 2012) software. In MetFrag (settings: search ppm: 5, PubChem, KEGG and CompTox search) compounds were considered as a plausible candidates based on fragment match and spectral similarity (above 0.5, maximum 2). If a match was obtained between the measured spectrum, predicted spectrum (MetFrag and CFM-ID) and spectral libraries (MassBank and mzCloud) a reference standard was acquired to confirm the identity with MS/MS and retention time. Six out of the 38 compounds were tentatively identified and reference standards were acquired for confirmation (Table SI-7A). The confirmed compounds in the samples were quantified (retrospective analysis) using 12-level standard calibration solutions (1-5000 ng L^{-1}) spiked with internal standards.

3.2.5 Risk assessment

In order to assess the toxic risks posed by the compounds detected, toxic units (TU) (Sprague, 1970) were calculated for three different trophic levels (fish, crustaceans and algae). The measured environmental concentrations (MEC) were normalized by lethal and sublethal concentrations causing effect in these organisms (Equation 1.1). The effect values were retrieved from the ECOTOX database and selected as described by (Beckers et al., 2018; Busch et al., 2016). Where ECOTOX data was not available, the values were predicted from Structure Activity Relationships models using the ECOSAR database as described in Busch et al. (2016) (Table SI-8A).

Mixture risks in environmental samples were calculated using the model of concentration addition (CA) (Loewe and Muischnek, 1926) as the sum of TUs (TU_{sum}, Equation 3.1). This model is designed for compounds with similar modes of action, while mixture effects of dissimilarly acting compounds are better described with the model of independent action (IA). However, it has been shown that CA and IA predictions do not differ by more than a factor of 2 in almost 90% of the cases (Belden et al., 2007). Thus, using the more conservative CA model has been suggested to give good predictions in most environmental mixtures irrespective of the modes of action of the components (Backhaus and Faust, 2012) since full concentration-response data as required for IA are available only for a limited number of compounds.

Equation 3.1:

$$TU\ sum = \sum TU$$

3.2.6 Prioritization of pollutants based on potential risk

Compounds of particular concern for monitoring and regulation in western Kenya were prioritized based on exceedance of risk threshold values calculated using two methods: (i) based on toxic units (TUs) and (ii) based on risk quotients (RQs). Based on TUs, compounds were prioritized according to the exceedance of the risk threshold values suggested by (Malaj et al., 2014): acute toxic risk (0.1 TUs) for all organisms and chronic risk for fish (0.01 TUs), *Daphnia*, (0.001 TUs) and algae (0.02 TUs). The second method was applied by calculating the RQ derived by normalizing concentrations to the lowest Predicted No-Effect Concentration (PNEC) across three trophic levels available from the NORMAN network (www.norman-network.net). Here, the risk threshold value was 1. Using these two methods independently, prioritization was based on two indicators: (i) the frequency of exceedance of TU and RQ-based thresholds (Equation 3.2 with n as the number of sites (*n*) where the TU or RQ of a specific compound exceeded the risk thresholds and N as the total number of sites sampled) and (ii) the maximum extent of

exceedance (Equation 3.3) of these thresholds ranking compounds addressing the maximum intensity of the risk (von der Ohe et al., 2011).

Equation 3.2:

$$Frequency\ of\ Exceedance = \frac{\sum n}{\mathrm{N}}$$

The extent of exceedance was calculated by normalizing the maximum toxic unit (TU_{max}) or risk quotient (RQ_{max}) per compound across all sites to the respective threshold (Equation 3.3). Results were then scaled from 0 to 1 as proposed by (von der Ohe et al., 2011). Results from the two indicators were summed up (maximum is 2) resulting in a priority score which was the basis of the compound ranking.

Equation 3.3:

$$Extent\ of\ Exceedance = \frac{\mathrm{TUmax}}{\mathrm{Threshold\ Value}} \quad or \quad \frac{\mathrm{RQmax}}{\mathrm{Threshold\ Value}}$$

3.2.7 Data analysis

Data analysis and visualization was performed using Microsoft Excel 2013, R version 3.5.0 and R studio (version 1.1.383). For reporting concentrations, data below the method detection limits (<MDLs) were considered as zeros. Cluster analysis was used to determine spatial pollution patterns. Due to variations in the concentration of detected compounds, the data was log transformed and scaled to reduce skewedness prior to analysis as performed by Beckers et al. (2018). Heatmaps (R package ‘gplots’, function heatmap.2, linkage = “complete”, dist= “euclidean”) were used to display spatial patterns of the detected compounds in the study area (Beckers et al., 2018).

3.3 Results and discussion

The results from the analysis of general water quality parameters are given in appendix A (Table SI-9).

3.3.1 Chemical analysis

3.3.1.1 Target screening

A total of 75 out of 428 compounds were detected in the water samples. The full list of detected compounds and concentrations are presented in appendix A (Table SI-10). Figure 3.2 shows the detection frequency of the 75 compounds while Figure 3.3 shows the 20 compounds with the highest concentrations.

Pesticides and biocides were the most frequently detected chemical classes with the highest number of compounds (26 parent compounds and 5 transformation products (TP)). Simazine was the pesticide with the highest detection frequency of 88% followed by dodemorph and bendiocarb with 65% and 60%, respectively (Figure 3.2). Individual pesticide concentrations ranged from non-detects to 1.5 µg L^{-1}. Highest concentrations were found for hexazinone and carbendazim with >1 µg L^{-1}. Hexazinone is a non-selective post emergence herbicide used for the control of grasses and broad leaf weeds in sugarcane plantations while carbendazim is a broad spectrum benzimidazole fungicide used for the control of blight and powdery mildew (PCPB, 2018). The presence of agro-industrial farms, particularly growing rice and sugarcane, could be a plausible reason for the high concentrations of pesticides detected. In addition, sampling was carried out when the sugarcane plants were at the growing stage and during the spraying season.

Additionally, the neonicotinoid imidacloprid and its TP imidacloprid-guanidine were detected at concentrations ranging up to 32 and 152 ng L^{-1}, respectively. The use of neonicotinoids has been increasing globally due to the ban on most organophosphates and organochlorine compounds with imidacloprid being the most widely used substance (Calvo-Agudo et al., 2019; Wood & Goulson, 2017). The concentrations of most pesticides in this study are within the range of concentrations reported by K'oreje et al. (2018).

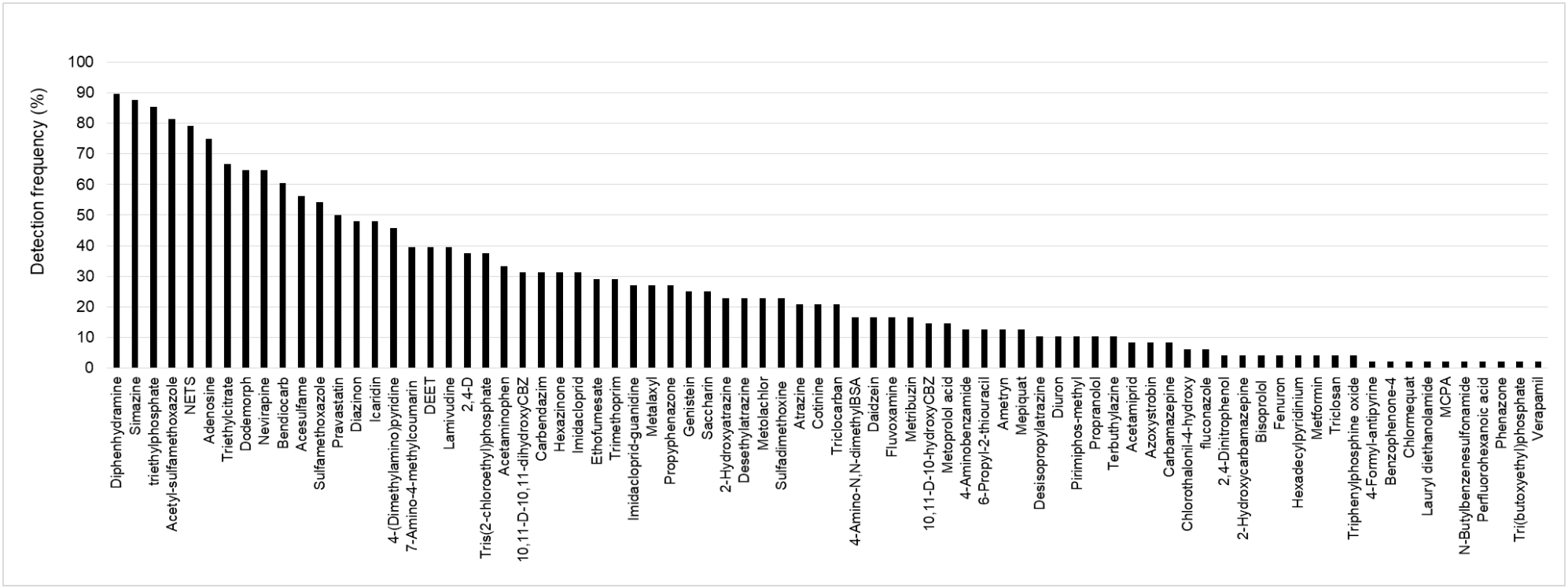

Figure 3.2: Frequency of detection for target compounds in the Lake Victoria South Basin. 10,11-D-10,11-dihydroxyCBZ: 10,11-Dihydro-10,11-dihydroxycarbamazepine, 4-Amino-N,N-dimethylBSA: 4-Amino-N,N-dimethylbenzenesulfonamide, 10,11-D-10-hydroxyCBZ: 10,11-Dihydro-10-hydroxycarbamazepine, 2,4-D: 2,4-Dichlorophenoxyacetic acid, NETS: N-Ethyl-o-toluenesulfonamide.

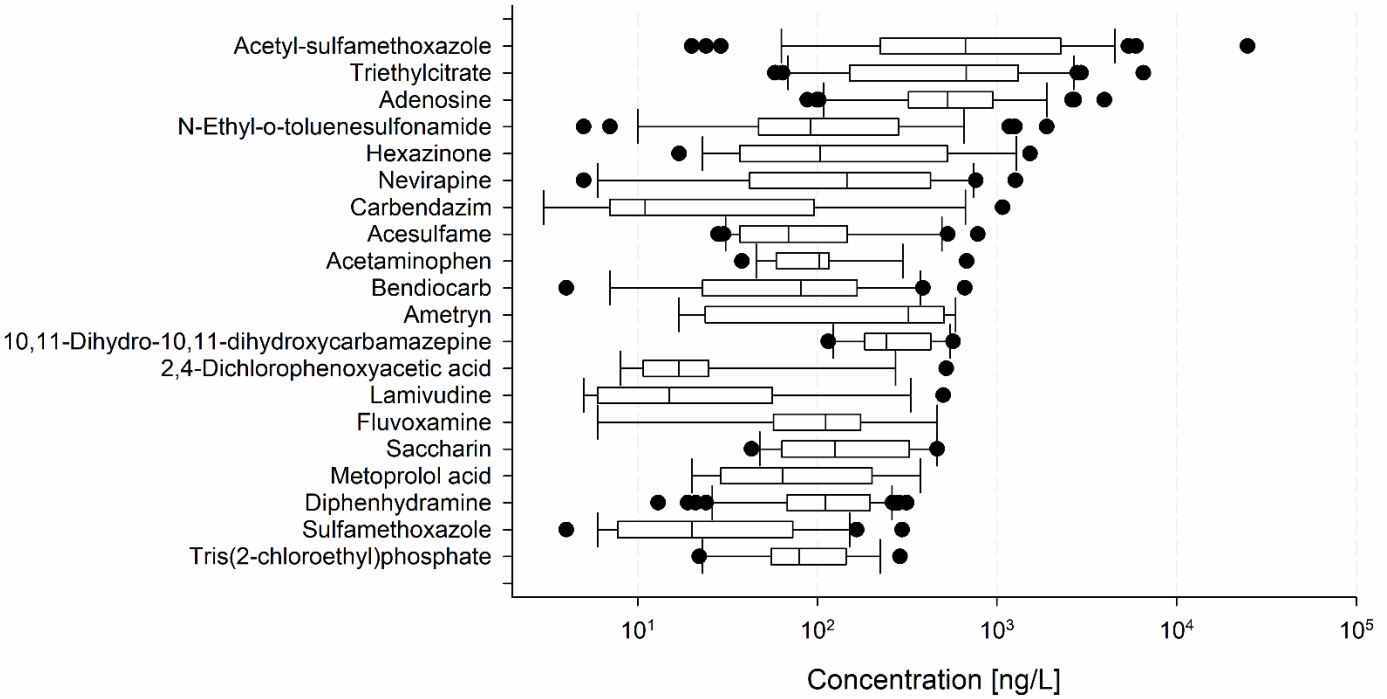

Figure 3.3: Box plots of concentration ranges of twenty compounds detected with the highest concentration within the different study sites. ● outlier (Q1–1.5xIQR) with Q1: lower quartile and IQR: interquartile range. The center line is the median concentration of the individual compound."

PPCPs were reported with a detection frequency ranging from 2 to 90%. The most frequently detected compound was the anti-allergic drug diphenhydramine, which was detected in 90% of the sites. This could be attributed to the availability of several over the counter drugs that contain diphenhydramine as one of the main active ingredients. Such medications are cheap, readily available and widely used for treatment of coughs and cold (Kigen et al., 2015). Among the reported metabolites, acetyl-sulfamethoxazole (A-SMX) was the most frequently detected one (81%). This result is in agreement with previous studies carried out on water samples obtained from Lake Victoria North and Nairobi Basins, Kenya (K'oreje et al., 2016; 2018; Ngumba et al., 2016) and Lake Victoria Basin, Uganda (Nantaba et al., 2019) which reported sulfamethoxazole to be the most frequently detected pharmaceutical in surface water.

Individually, the PPCP concentrations varied from 0.001 to 24 µg L^{-1} (Table SI-10A) with A-SMX exhibiting the highest maximum concentration (Figure 3.3). This concentration exceeds the European drinking water guideline values by a factor of 2800. A-SMX is a TP of sulfamethoxazole, an antibiotic used to treat various bacterial infections in both humans and livestock (Aus der Beek et al., 2016; Fekadu et al., 2019). Sulfamethoxazole, usually used in combination with trimethoprim as co-trimoxazole, is of relatively low cost compared to other antibiotics. It is easily accessible over the counter and used as a first line treatment drug for several ailments including HIV opportunistic infections which are prevalent in Kenya (National AIDS Control Council (NACC), 2018). Noticeably, the highest A-SMX, sulfamethoxazole (297 ng L^{-1}) and trimethoprim (110 ng L^{-1}) concentrations were detected (PS16) at the same site. These concentrations were lower than those previously reported in cities in Kenya (K'oreje et al., 2016; 2018; Ngumba et al., 2016) but higher (factor of 8-14) compared to those reported in a review by Aus der Beek et al. (2016) for European waters. The common insect repellants DEET and icaridin were found at concentrations of 28 and 67 ng L^{-1}, respectively. The co-occurrence of pharmaceuticals and pesticides in agricultural sites is probably due to the influence of wastewater treatment plants, untreated wastewater and poor waste disposal (Gwenzi & Chaukura, 2018) with no clear demarcation between agricultural and residential areas in the study area.

To the best of our knowledge, we report for the first time concentrations of plasticizers (Figure 3.2, Table SI-10A) in Kenya. The most frequently detected compounds include triethylphosphate (85%), N-ethyl-o-toluenesulfonamide (79%) and triethylcitrate (67%), detected at maximum concentrations of 0.3 µg L^{-1}, 2 µg L^{-1} and 6.5 µg L^{-1} respectively. These compounds have been recently found in high frequency but about one order of magnitude lower concentrations at sites along Yangtze River Delta in eastern China (Peng et al., 2018). Triethylphosphate is a multiple use industrial chemical applied in the plastics industry as a catalyzer, plasticizer, flame retardant, in polyester resins and polyurethane foam (Wei et al., 2015). N-ethyl-o-toluenesulfonamide is of similarly wide spread use in industrial products including plastics, inks and cosmetics but also as inert ingredient in pesticide formulations. Triethylcitrate is used as a plasticizer and replacement of phthalates but also used as a food additive. A probable source of these compounds would be from the traffic, waste dumpsites and car and motor bike washing activities at the river banks rampant in the area.

3.3.1.2 Suspect screening

Using the suspect screening workflow outlined in subsection 3.2.4, six compounds (rimantidine, adenosine, nevirapine, pencycuron, lamivudine and flupyradifurone) from the candidate list were tentatively identified and selected for further confirmation (Table SI-11A). After instrumental analysis with respective standards, nevirapine, lamivudine and adenosine were confirmed (Appendix A Figures SI-1.1, SI-1.2 and SI-1.3). Retrospective quantification revealed concentrations of up to 4 µg L^{-1} (adenosine), 0.5 µg L^{-1} (lamivudine) and 1 µg L^{-1} (Nevirapine). Nevirapine and lamivudine are anti-retroviral (ARV) drugs used in Kenya for the treatment of HIV/AIDS and to prevent mother-to-child transmission in pregnant infected females. The detection of these ARVs shows evidence of the national occurrence of the compounds in surface water as they had previously been reported in western (K'oreje et al., 2016; 2018) and central Kenya (K'oreje et al., 2016; Ngumba et al., 2016). This ubiquitous occurrence could be attributed to the relatively high national prevalence (4.9%) of HIV/AIDS in Kenya and specifically between 4 and 21% within the study area (NACC, 2018). In contrast to Europe, these compounds are frequently reported in African waters in line with the consumption pattern of the ARVs

(Fekadu et al., 2019; Madikizela et al., 2019; Ncube et al., 2018;). Lamivudine and nevirapine are known to be rather photostable and poorly biodegradable. Their detection in the water samples receiving treated wastewater suggest that they are poorly removed (11-59%) during wastewater treatment (K'oreje et al., 2018). Pencycuron and flupyradifurone had a high MetFrag score above 1 (Table SI-10A), but the MS/MS of the reference standard and the measured spectrum did not match. Rimantidine had a hit in MassBank, MetFrag and mzCloud, however, the retention time differed from that of the reference standard. The software used for *in-silico* fragmentation and candidate identification are helpful to exclude candidates however, verification with reference standards was shown to be essential.

3.3.2 Spatial pattern analysis

We hypothesized that chemicals will group depending on land use cover. To determine the pollution patterns occurring in the LVSB, cluster analysis was computed in order to identify spatial patterns in chemical exposure. A hierarchical clustering was performed considering all com- pounds detected at every site. However, only limited grouping of compounds and sites could be obtained in the present study (Figure 3.4). This is due to the high and rather random variability in pollutant concentrations and mixed land use systems. No clear demarcation exists between agricultural and non-agricultural areas. The most interesting cluster comprises majorly pesticides including2,4-D, diuron, diazinon, atrazine, hexazinone, ametryn and pirimiphos methyl. This group contains site-specific compounds which were detected at high concentrations in Uli9, PS16, PS17 and PS18 within Migori County. This could be attributed to the presence of large-scale agro-industrial sugarcane plantations within this county. Additionally, a cluster exists consisting of compounds which have been detected at similar concentrations in all the sites. These compounds include imidacloprid, dodemorph, simazine, diphenhydramine and N-ethyl-o-toluenesulfonamide.

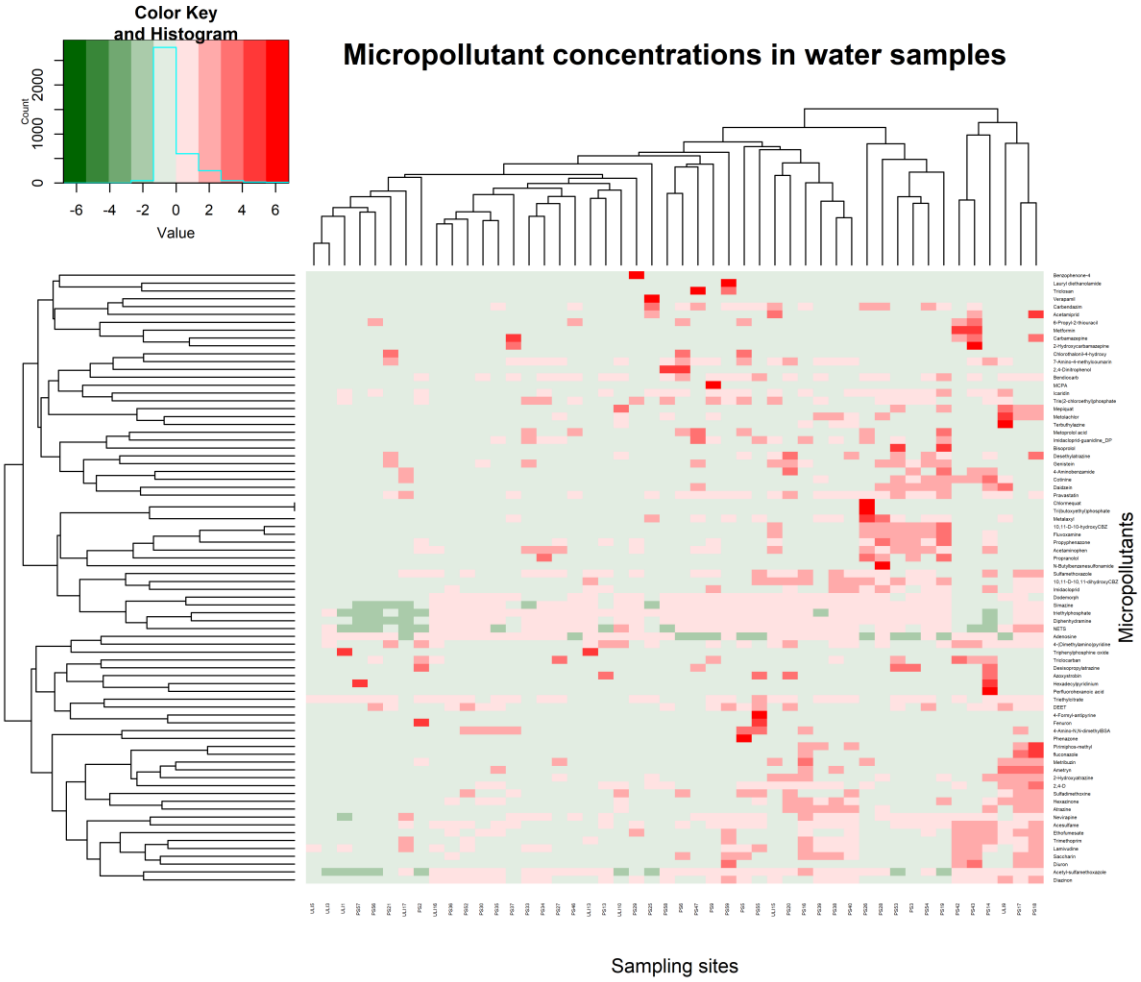

Figure 3.4: Spatial pollution patterns in Lake Victoria South Basin based on compound concentrations (ng L^{-1}) detected in the study sites (data scaled and centered). 10,11-D-10,11-dihydroxyCBZ: 10,11-Dihydro-10,11-dihydroxycarbamazepine, 4-Amino-N,N-dimethylBSA: 4-Amino-N,N-dimethylbenzenesulfonamide, 10,11-D-10-hydroxyCBZ: 10,11-Dihydro-10-hydroxycarbamazepine, 2,4-D: 2,4-Dichlorophenoxyacetic acid, NETS: N-Ethyl-o-toluenesulfonamide.

3.3.3 Risk assessment

The potential for risk was evaluated comparing the sum of TUs based on the measured concentrations of all contaminants to the acute and chronic risk thresholds for fish, crustaceans and algae as explained above (Figures 3.5, 3.6, 3.7 and SI-11A). The highest risk was found for invertebrates with 23 out of 48 sites (Figure 3.5) exhibiting acute toxic risk (TU > 0.1) for these organisms. Two of them even showed TU_{sum} > 1 indicating concentrations above the EC values. For 39 out of48 sites, chronic toxic risks (TU >

0.001) are detected typically related to changes in the invertebrate communities and losses of sensitive species (Liess & Von Der Ohe, 2005, Malaj et al., 2014, Shahid et al., 2018). In all 23 sites exhibiting acute toxic risks these risks are predominated by the neurotoxic insecticide diazinon. At all other sites, diazinon fell below the MDL of 2 ng L^{-1} corresponds to 0.2 TU. Thus, diazinon probably causes acute and chronic toxic risks also at other sites, which is masked by the non-detectability at very low but biologically active concentrations. This high toxicity of diazinon is influenced by the low effect value of diazinon reported by Bouldin et al. (2007) in their study performed on a constructed wetland. The results from this study are in agreement with several studies in Europe identifying diazinon as a major risk driver for invertebrates represented by the sensitive standard test organism *Daphnia magna* wherever the compound is applied. This has been reported by Ccanccapa et al. (2016) for the Ebro river basin. The compound was also shown to contribute greatly to the overall toxicity in waste and surface waters in Germany and Switzerland (Beckers et al. (2018) and Munz et al. (2017)) despite diazinon being banned for use as a plant protection product in Europe. In Kenya, it is still sold legally over the counter (PCPB, 2018). In the 17 sites where a chronic risk to invertebrates was found while diazinon was not detected, the carbamates bendiocarb and carbendazim, pirimiphos-methyl (phosphoric ester insecticide) and the quaternary ammonium disinfection agent hexadecylpyridinium were major the standard test organism *Daphnia magna*.

For algae, the acute toxicity threshold (TU > 0.1) was exceeded by 4 out of 48 samples in Migori County (Figure 3.6). This could be attributed to the large-scale agro-industrial sugarcane plantations in this region. In all cases this risk was driven by the herbicide, hexazinone with TU up to 0.51 and TU_{sum} up to 0.55. At 11 out of 48 sites chronic risk was observed (TU > 0.02). In addition to hexazinone, the biocide triclosan, the photosynthesis inhibiting herbicides metribuzin, simazine, atrazine and its transformation product desisopropylatrazine and the pharmaceutical acetaminophen were compounds responsible for toxic risk to algae.

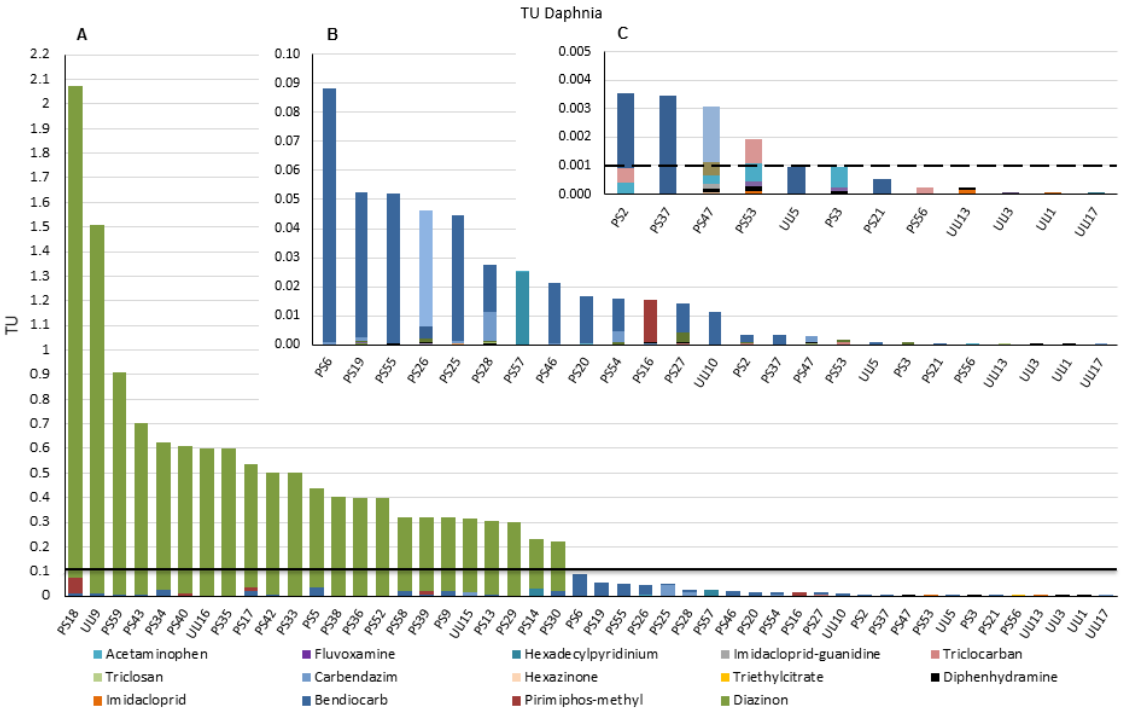

Figure 3.5: Compounds driving acute and chronic risk on D. Magna in the Lake Victoria South Basin based on compound toxic unit value (TU). Figure A includes all sites sampled, Figure B shows the sites which are below the acute threshold but exceeded the chronic threshold. Figure C shows the compounds which exceeded the chronic threshold. The bold line (in Figure A) indicates the threshold for acute risk (0.1) and the dotted line (in Figure C) indicates the chronic threshold (0.001) for D. Magna

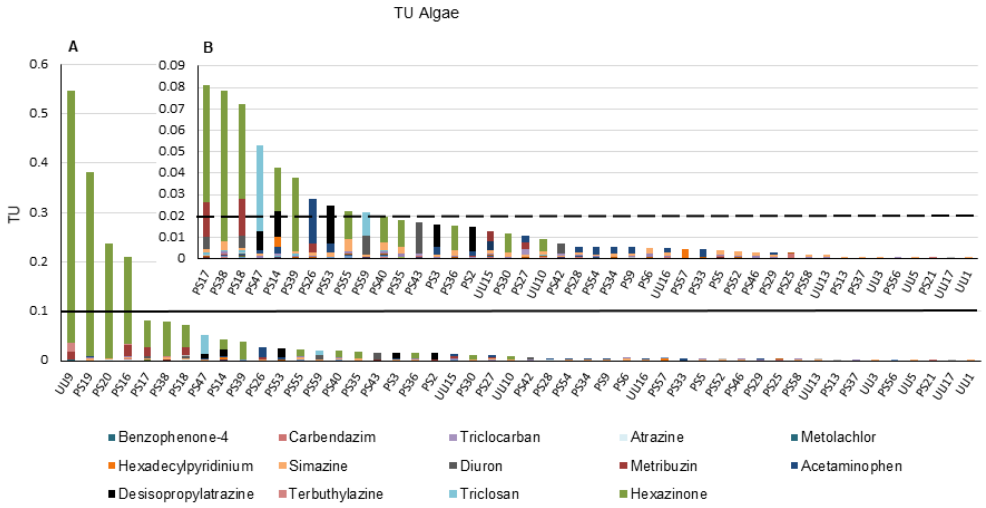

Figure 3.6: Compounds driving acute and chronic risk on algae in the Lake Victoria South Basin based on compound toxic unit value (TU). The bold line (in Figure A) indicates the threshold for acute risk (0.1) and the dotted line (in Figure B) indicates the chronic threshold (0.02) for algae.

For fish, the overall risk for acute toxicity was low compared to crustaceans and algae. While no acute toxic risks were found, chronic toxic risks (TU > 0.01) were observed for 5 out of 48 sites (Figure 3.7). Risk drivers were the fungicide carbendazim with a maximum TU of 0.09 and the disinfectant hexadecylpyridiniumwith a maximum TU of 0.04. Despite the relatively low toxic risk to fish based on the compounds analyzed in this study, it should be mentioned that this assessment does not include the risk of endocrine disruption for example by steroid hormones, which are not detectable at effect concentrations in the applied screening analysis but require a more targeted approach. It should be considered that the assessment was based on toxicity data for data rich test species used in European monitoring and assessment due to the lack of African species with a sufficient data basis. Thus, the actual risk to native species may deviate from the risk to the standard test species. It should also be mentioned that the involvement of toxicity to insects, which are for example particularly sensitive to neonicotinoids but are data poor in general and thus hard to use in a general assessment might unravel additional risky compounds.

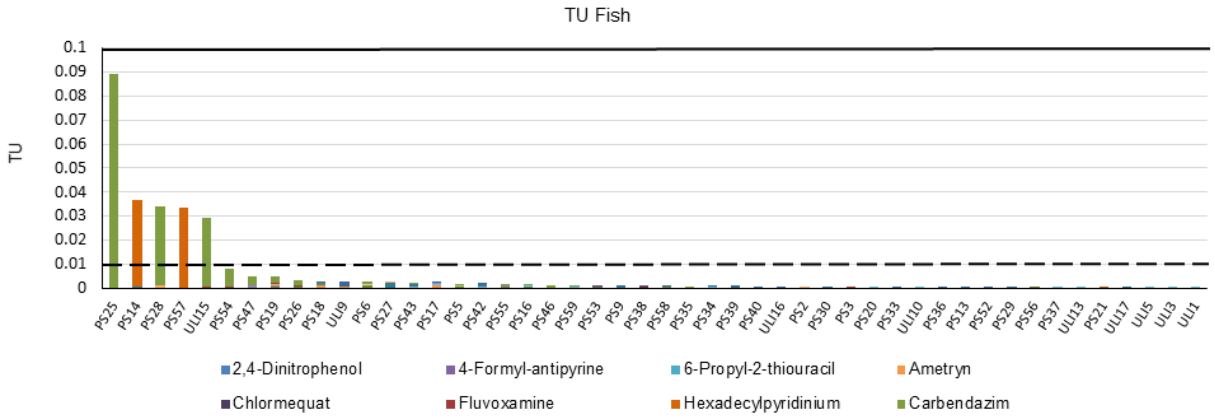

Figure 3.7: Compounds driving chronic risk on fish in the Lake Victoria South Basin based on compound toxic unit value (TU). The bold line indicates the threshold for acute risk (0.1) and the dotted line indicates the chronic threshold (0.01) for fish.

Based on the acute and chronic thresholds, 16 compounds were ranked ranging from 0.1–1.5 (Table 3.1). The compounds with the highest ranking include diazinon, bendiocarb, hexazinon, carbandazim and pirimiphos-methyl. According to the PNEC criteria, 7 compounds were ranked ranging from 0.1 to 0.7 (Table 3.2). In addition to

compounds already considered such as diazinon, three additional compounds are flagged as posing a significant risk including A-SMX, ametryn and terbuthylazine which had a priority score of 0.4, 0.3 and 0.2 respectively based on toxicity to the algae (*Selenastrum capricornutum*). Irrespective of the approach applied, diazinon ranked the highest in priority indicating a need for constant monitoring in order to protect aquatic ecosystems.

Table 3.1: Compounds prioritized based on acute and chronic risk thresholds, compound class, trophic level, threshold level (acute 0.1), frequency and extent of exceedance of the threshold, priority ranking.

Compound	Class	Organism	Level	Extent of exceedance	Frequency of exceedance	Priority ranking
Diazinon	Pesticide	*D. magna*	Chronic	1.0	0.5	1.5
Bendiocarb	Pesticide	*D. magna*	Chronic	0.2	0.5	0.7
Diazinon	Pesticide	*D. magna*	Acute	0.5	0.2	0.7
Hexazinone	Pesticide	Algae	Chronic	0.2	0.2	0.4
Carbendazim	Pesticide	*D. magna*	Chronic	0.2	0.1	0.3
Pirimiphos-methyl	Pesticide	*D. magna*	Chronic	0.2	0.1	0.3
Carbendazim	Pesticide	Fish	Chronic	0.2	0.1	0.3
Hexadecylpyridinium	Biocide	*D. magna*	Chronic	0.2	0.0	0.2
Hexazinone	Pesticide	Algae	Acute	0.1	0.1	0.2
Triclocarban	Biocide	*D. magna*	Chronic	0.1	0.1	0.2
Hexadecylpyridinium	Biocide	Fish	Chronic	0.1	0.04	0.1
Acetaminophen	Pharmaceutical	*D. magna*	Chronic	0.1	0.02	0.1
Acetamiprid	Pesticide	*D. magna*	Chronic	0.1	0.02	0.1
Acetaminophen	Pharmaceutical	Algae	Chronic	0.1	0.0	0.1
Metribuzin	Pesticide	Algae	Chronic	0.1	0.02	0.1
Triclosan	Biocide	Algae	Chronic	0.1	0.02	0.1

Table 3.2: Compounds prioritized based on Predicted No-Effect Concentration (PNEC) threshold, compound class, trophic level, threshold level (acute 0.1), frequency and extent of exceedance of the threshold, priority ranking.

Compound	Class	Organism	Extent of exceedance	Frequency of exceedance	Priority ranking
Diazinon	Pesticide	*D. magna*	0.2	0.5	0.7
A-SMX	TP Pharma	*S. capricornutum*	0.2	0.2	0.4
Ametryn	Pesticide	*S.capricornutum*	0.2	0.1	0.3
Pirimiphos-methyl	Pesticide	*D. magna*	0.2	0.1	0.3
Simazine	Pesticide	*S.capricornutum*	0.1	0.1	0.2
Hexadecylpyridinium	Biocide	*P.promelas*	0.1	0.04	0.1
Terbuthylazine	Pesticide	*S.capricornutum*	0.1	0.02	0.1

TP Pharma: Transformation product from pharmaceutical; *S. capricornutum: Selenastrum capricornutum; P. promelas: Pimephales promelas*

3.4 Conclusions

The present study has provided a comprehensive identification and risk assessment of emerging OMPs in freshwater systems in rural areas within the Lake Victoria South Basin for three representative organism groups including crustaceans, algae and fish. Particularly high concentrations (> 1 µg L^{-1}) were detected for some pharmaceuticals and pesticides with A-SMX being reported at 24 µg L^{-1}. Suspect screening supported the detection of two common antiretroviral drugs used in the first line of treatment among HIV/AIDS infected individuals. Risk assessment revealed a high risk of acute and chronic toxicity for crustaceans while fish and algae experience majorly chronic risk. This risk was driven by a low number of strongly dominating pesticides including diazinon, bendiocarb, hexazinone, carbendazim, which should be highly prioritized for monitoring, regulation and abatement. There is a potential risk on human health considering that the rivers and reservoirs are used by the local communities for daily household activities including source of drinking water. The maximum concentration of A-SMX reported in this study exceeds European drinking water guideline values by a factor of 2800. Thus, the assessment of contamination in Africa and other developing countries based on acute toxicity data for aquatic organisms as done in the present study is an important first step

but demands for extension particularly with respect to endpoints and areas of concern as a basis for efficient abatement for a non-toxic environment in Africa for humans and ecosystems. Further research should focus on seasonal variability of contaminants within the region.

Acknowledgements

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Chapter 4

Multi-compartment chemical characterization and risk assessment of chemicals of emerging concern in freshwater systems of western Kenya

Author's contribution statement

Declaration of author contributions to the publication:

Multi-compartment chemical characterization and risk assessment of chemicals of emerging concern in freshwater systems of western Kenya

Status: Accepted in Environmental Sciences Europe

Contributing authors:

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What are the contributions of the doctoral candidate and his co-authors?

(1) Concept and design

Doctoral candidate: 40%
Co-author last: 50%
Co-author rest of the authors: 10%

(2) Conducting tests and experiments

Doctoral candidate: 80% (organized and carried out field work, sample preparation, chemical analysis)
Co-author 2, 8 and 9: 10% (supervised sample preparation and supported chemical analysis)
Co-author 3: 10% (snail sampling and identification)

(3) Compilation of data sets and figures

Doctoral candidate: 75 % (Raw data evaluation, toxic unit calculation, drawing of figures)
Co-author 2, 9: 10 %(Supervised data evaluation, Final data evaluation)
Co-author 3: 15% (Grouping datasets, drawing figures)

(4) Analysis and interpretation of data

Doctoral candidate: 75% (interpretation, Excel analysis)
Co-author 3: 15% (statistical analysis, Pattern analysis)
Co-authors 2 and last: 10% (Interpretation of chemical and risk data, Advise on risk assessment)

(5) Drafting of manuscript

Doctoral candidate: 80%
Co-author all: 20%

I hereby certify that the information above is correct.

_____

Date and place Signature doctoral candidate

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Date and place Signature supervisor

Abstract

Within the last decades, there has been increasing research on the occurrence of chemicals of emerging concern (CECs) in aquatic ecosystems due to their potential adverse effects on freshwater organisms and risk to human health. However, information on CECs in freshwater environments in sub-Saharan countries is very limited. Here, we investigated the occurrence of CECs in snails and sediments collected from 48 sites within the Lake Victoria South Basin, Kenya, which have been previously investigated for water contamination. Samples were analyzed by liquid chromatography coupled to high resolution mass spectrometry (LC-HRMS) with a target list of 429 compounds.

In total, 30 compounds have been detected in snails and 78 in sediment samples, compared to 79 previously identified compounds in water. By extending the monitoring of CECs to snails and sediments, we found 68 compounds that were not previously detected in water. These compounds include the anti-cancer drug anastrozole, detected for the first time in the Kenyan environment. Individual compound concentrations were detected up to 480 ng/g wet weight (N-ethyl-o-toluenesulfonamide) in snails and 110 ng/g organic carbon (pirimiphos-methyl) in sediments. Higher contaminant concentrations were found in agricultural sites than in areas not impacted by anthropogenic activities. Crustaceans were the organisms at greatest toxic risk from sediment contamination (toxic unit (TU) up to 0.99) with diazinon and pirimiphos-methyl driving this risk. Acute and chronic risks to algae were driven by diuron (TU up to 0.24), whereas fish were found to be at low to no acute risk (TU up to 0.007).

The compound classes present at highest frequencies in all matrices were pesticides and biocides. This study shows substantial contamination of surface water in rural western Kenya. By filling data gaps on contamination of sediments and aquatic biota, our study reveals that CECs pose a substantial risk on environmental health in Kenya demanding for monitoring and mitigation.

4.1 Introduction

Chemicals of emerging concern (CECs) such as pesticides, pharmaceuticals, and personal care products (PPCPs), surfactants and other industrial chemicals are of global environmental concern due to their toxic risk on ecosystems and human health. The occurrence of CECs in freshwater ecosystems has been shown to cause adverse effects including long-term changes in the aquatic community composition (Liess & Von Der Ohe, 2005). Several studies have documented the global occurrence of CECs in freshwater resources (Luo et al. 2014; Sui et al. 2015; Aus der Beek et al. 2016; Li et al. 2017). However, most research is focused on high-income countries. Although many African laboratories are increasing their efforts to monitor the occurrence of CECs in African aquatic ecosystems (Mzukisi et al. 2017), this research field is yet to be fully explored. Many previous studies from Kenya focused largely on urban areas with a limited number of compounds investigated (Bagnis et al. 2020; Getenga et al. 2004; K'oreje et al. 2018; Kairigo et al. 2020; Kimosop et al. 2016; Ngumba et al. 2016; Otieno et al. 2015). Among the studies on pesticides in the environment, previous work largely focused on organochlorine and organophosphates compounds (Wandiga et al. 2002; Musa et al. 2011; Osoro et al. 2016) although the next generation of pesticides including phosphoric esters and neonicotinoids are currently in use and have been found in Kenyan surface water (K'oreje et al. 2018). There is need to increase the number of compounds under research and expanding to rural areas.

When CECs are discharged into the aquatic ecosystem, they undergo equilibrium partitioning between water and sediments and may accumulate in biota according to their hydrophobicity and toxicokinetics in the organism (Di Toro et al. 1991; Mayer and Holmstrup 2008; Ashauer et al. 2011; Inostroza et al. 2017). In this context, sediments and biota act as passive samplers reflecting long and mid-term exposure, respectively, and allow also for the detection of compounds that are hardly detectable in the water phase. Detection of contaminants in biota proves their bioavailability and bioaccumulation potential (Mackay et al. 2011; Yin et al. 2017). Mollusks such as snails and mussels are particularly suitable as passive samplers due to their low metabolization competence (Nhan et al. 2001; Oehlmann and Schulte-Oehlmann 2003). Persistent organic pollutants

(POPs) with a high octanol-water partitioning coefficient (logK_{ow}) are particularly prone to accumulate in sediments and lipid-rich biological tissues. This results in long-term exposure particularly of benthic species and poses a substantial risk to the aquatic ecosystem (Li et al. 2017). Thus, a large body of literature on the bioaccumulation of legacy POPs including organochlorine pesticides, polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) exist (Heim & Schwarzbauer, 2013; Merhaby et al. 2019). However, currently used organic chemicals including modern pesticides are designed to be more polar and less persistent. These chemicals may still accumulate in sediments and may even pose higher toxic risks to aquatic ecosystems due to their higher bioavailability (Bandow et al. 2009). Many CECs can be considered as pseudo-persistent with emission exceeding degradation rates and have been shown to accumulate in sediments and biota (Masiá et al. 2013; Ccanccapa et al. 2016). These chemicals are in the focus of this present study.

The objectives of this study were: (1) to investigate the occurrence and distribution of CECs in aquatic ecosystems in snails and sediments in the Lake Victoria South Basin and to reveal whether analyzing these matrices can substantially increase the number of chemicals detected compared to the water analysis only, (2) to determine the spatial distribution of contaminants dependent on land use within the Lake Victoria South Basin, and 3) to assess risks of sediment-associated chemicals on fish, crustaceans and algae by calculating equilibrium water concentrations from sediment concentrations. For the water phase, data were obtained from our previous study (Kandie et al. 2020) and used for comparison of compound concentrations in snail and sediments.

4.2 Materials and methods

4.2.1 Chemicals

Analytical standards were purchased from various suppliers with purity of above 97%. The 429 target compounds included pesticides and biocides (PaBs), PPCPs, industrial compounds and transformation products which have been detected in surface waters and sediments. More information on the target compounds and internal standards used is presented in the appendix (Appendix A Table SI-1 and Appendix B Table SI-1). Methanol

(LC-MS grade), ethyl acetate, dichloromethane (DCM) and acetone (all LC grade) were obtained from Sigma Aldrich (Germany), while LC-MS grade water was purchased from Thermo-Fisher (Germany). Primary Secondary Amine (PSA), Sodium chloride and magnesium sulfate were supplied by Sigma-Aldrich.

4.2.2 Description of the study area and sampling

The study area was located within the Lake Victoria South Basin (LVSB) in western Kenya covering Kisii, Nyamira, Migori, Homabay, Kericho, Kisumu and Narok counties. Sampling sites have been described in detail in Kandie et al. (2020) and Becker et al. (2020). Forty-eight sites were sampled for sediments and snails between September and October 2017 in parallel to water samples analyzed previously Kandie et al. (2020). For rivers or drainage canals, 50-m transects were sampled with four sampling spots distributed equally along the transect, with bias on suitable snail habitats. For reservoirs, the four edges were taken as sampling spots in each site. Snails were sampled using a handheld stainless-steel snail catcher. Approximately 200 g of sediment were sampled using a pre-cleaned stainless-steel scoop, homogenized, sieved (2 mm) and transferred to a zip lock bag. Sediment samples were covered with aluminum foil to avoid photo degradation and immediately put into a portable freezer (-4˚C) and transferred to -20˚C in the laboratory. Snails were kept alive in falcon tubes for identification for up to 15 hours. After identification in the laboratory, the samples were stored at -20˚C prior to extraction.

4.2.3 Sample preparation and extraction

4.2.3.1 Snail extraction

Snails were found in 20 out of the 48 sites sampled and separated according to sites and species. The snails sampled were from the genera *Bulinus, Biomphalaria, Melanoides, Lymnaea, Physa* and *Ceratophallus*. The snails were found in shallow slow-moving water. Except for *Physa* which were much larger (approximately 3 cm), all the other genus had an average size of 5mm. Most of the snails were attached to aquatic vegetation within streams and drainage canals. The snails from each site belonging to the same species were pooled. Chemicals were extracted from the snail tissues by applying a modified

QuEChERS (Quick, Easy, Cheap, Effective, Rugged and Safe) method as described by Inostroza et al. (2016). Briefly, 1 g of crushed snails was homogenized in 5 mL acetonitrile (LC-MS grade) using a Stuart handheld homogenizer (SHM1/UK) for 1 minute. For salting out, 2 g of anhydrous magnesium sulfate and 500 mg of sodium chloride were added, vortexed immediately on highest intensity for 2 minutes and centrifuged at 3000 x g for 5 minutes. An aliquot (1 mL) of the acetonitrile phase was transferred into a pre-assembled 2-ml QuEChERS tube containing 150 mg anhydrous magnesium sulfate and 50 mg Primary Secondary Amine (PSA) for clean-up, vortexed on highest intensity for 1 minute and centrifuged for 5 minutes at 17,000 x g. The supernatant extract was then vacuum filtered (0.2 µm PTFE Samplicity system filter, Merck Millipore, Germany) into a 1.5 mL autosampler vial. The filtered extract (400 µL) was transferred into amber autosampler vials and stored at -20˚C until instrumental analysis. A method blank was prepared using water from a pristine reference stream within the Harz Mountains (Wormsgraben, Germany) and processed similar to the samples. From recovery experiments carried out in a pre-study, 80% of the compounds had recoveries between 70% and 120% showing a good overall performance of the QuEChERS method.

4.2.3.2 Sediment extraction and clean up

Sediment samples were freeze-dried (SP Scientific, Advantage EL-85) and the sediment mass extracted was adjusted according to the total organic carbon (TOC) contents to obtain a comparable matrix load of the extracts (Massei et al. 2018). The TOC content of the sediments was determined on a Flash 200 organic elemental analyzer coupled to a Delta V Advantage Isotope Ratio mass spectrometer (Thermo). The analyzer was operated at 1000°C to ensure complete combustion. Details of the TOC procedure are described in SI-2B.

Compounds were extracted from sediments according to the method described by Massei et al. (2018) with minor modifications. Briefly, pressurized liquid extraction (PLE, ASE 200 device, Dionex) was performed by taking approximately 5 to 10 mg of freeze-dried sediment (corresponding to 100 mg of TOC content) and mixed with pure diatomaceous earth (25% of sediment weight, Hydromatrix, Restek). This mixture was transferred to stainless steel PLE cells fitted with 27 mm glass fibre filters (Dionex) and extracted at

100°C with ethyl acetate and acetone (50:50, v/v) in two static cycles at a pressure of 100 bar. A method blank containing only hydromatrix was prepared for each batch of samples to evaluate instrument background contamination. The extract was concentrated to 0.5 mL using a gentle stream of nitrogen (Xcelvap, Thermo), and the solvent was exchanged for DCM. For extract clean-up, flash chromatography was performed using a pre-packed chromatography column (Chromabond Flash RS 4 SiOH, 4 g, Macherey-Nagel) and an Agilent 1260 binary pump. Conditioning of the column was done using DCM prior to clean up. A corresponding volume of internal standard (1 µg/mL) was added to the concentrated sample extract in the vials (final concentration 50 ng/mL). Using a glass pipette, the extract was transferred to the flash column for clean up using DCM and methanol at 5 mL min^{-1} (Appendix B SI-3) and collected in separate vials. The DCM and methanol fractions were combined (50:50, w/w) and concentrated to 1 mL using a gentle stream of nitrogen while rinsing the vial walls with methanol and blowing down to 0.5 mL to exchange the solvent completely for methanol.

4.2.4 Instrumental analysis

Aliquots of 100 µL snail extracts were transferred into 2 mL vials with insert together with 10 µL of internal standard mix (1 µg/mL) containing 40 isotope-labelled compounds (Table SI-1B). Eleven method-matched calibration levels were prepared ranging from 0.05 to 500 ng/mL in vial. For the calibration samples, an appropriate volume of the standard solution was added to 1 mL of water and processed in the same way as the samples.

Sediment extracts were analyzed in 100-µL aliquots transferred to a 2-mL vial with insert. Method-matched calibrations were prepared at twelve concentration levels corresponding to 0.2 to 2000 ng/mL in vial by adding standard solutions to 5 mL of ethyl acetate:acetone (50:50 v/v) to mimick the PLE extracts. The solvent was evaporated using a gentle stream of nitrogen (Xcelvap, Thermo) and exchanged to DCM. Calibration solutions were subjected to clean-up with flash chromatography after adding 50 µL of internal standard (50 ng/mL).

All the samples were analysed using liquid chromatography (Thermo Ultimate 3000 LC) coupled to high resolution mass spectrometry (QExactive Plus, Thermo) equipped with a heated electrospray ion source. Snail and sediment extracts were analysed by injecting 10-µL and 5-µL aliquots into the instrument, respectively. Instrument settings and the chromatographic conditions have been described by Kandie et al. (2020). A water (A) and methanol (B) solvent gradient, both with 0.1% v/v formic acid, was applied in separate positive and negative ionization analytical runs. A combination of full scan (m/z range 100-1500) at a nominal resolving power of 70,000 (referenced to m/z 200) and data-independent MS/MS fragmentation at a resolving power of 35,000 was used for both positive and negative mode. The mass spectrometer settings are described in Kandie et al. (2020).

4.2.5 Risk assessment based on Toxic Units (TU)

The ecotoxicological relevance of the obtained compound concentrations was evaluated by performing risk assessment based on toxic units (TU) (Sprague 1970). Equilibrium water concentrations (C_{ewsed}) were derived from concentrations in sediment organic carbon (C_{sed}) assuming equilibrium partitioning (Equation 4.1). Organic carbon-water partitioning coefficients (K_{oc}) were derived using linear solvation energy relationships (LSER) applying the open access UFZ-LSERs database (Ulrich et al. 2017) as described by Inostroza et al. (2017).

Equation 4.1:

$$C_{ewsed} = \frac{Csed}{Koc}$$

The TU was calculated for each chemical by normalizing the equilibrium water concentrations (C_{ewsed}) to the effect concentrations (EC) for fish, crustaceans and algae according to Equation 4.2. Effect concentrations were derived from Busch et al. (2016). The EC values were retrieved from United States Environmental Protection Agency's (USEPA) ECOTOX database and were based on the 5th percentile of measured acute values (Busch et al. 2016), predicted read-across or ECOSAR.

Equation 4.2:

$$\text{Toxic unit (TU)} = \frac{C_{ewsed}}{EC}$$

The calculated TUs were compared to acute and chronic risk threshold values as proposed by Malaj et al. (2014). The acute risk thresholds (ART) for all organisms is 0.1 TUs while chronic risk thresholds (CRT) for fish (0.01 TUs), *Daphnia* (0.001 TUs) and algae (0.02 TUs) were applied.

To predict mixture toxicity, individual TU values were summed up (TU_{sum}) based on the concentration addition (CA) model (Loewe & Muischnek, 1926) designed for compounds with similar mode of action (Altenburger et al., 2003; de Almeida et al., 2017) but being also a reasonable estimate for mixtures of environmental compounds without knowing their individual modes of action (Backhaus and Faust 2012).

4.2.6 Data analysis

Peak detection and annotation of target compounds was performed using MZmine (Version 2.38, Pluskal et al., 2010) and detected target compounds were further confirmed and quantified using TraceFinder 4.1 (Thermo). The MZmine and TraceFinder settings were applied as shown in Kandie et al. (2020). Jchem was used for structure-based determination of compound properties including molecular formula and exact mass. The method detection limits (MDLs) were determined using calibration standards based on USEPA (2011) guidelines. Graphs and statistical analysis were performed using Microsoft Excel 2013 and SigmaPlot 13.0.

4.3 Results and discussion

4.3.1 Body burden of pollutants in snails

Out of the 429 compounds targeted, 30 compounds including PaBs, PPCPs and industrial compounds were detected in snail tissues (Figure 4.1). Concentrations detected in snail tissues on wet weight basis (ng/g ww) are given in the appendix B (Table SI-5 and Figure SI-1).

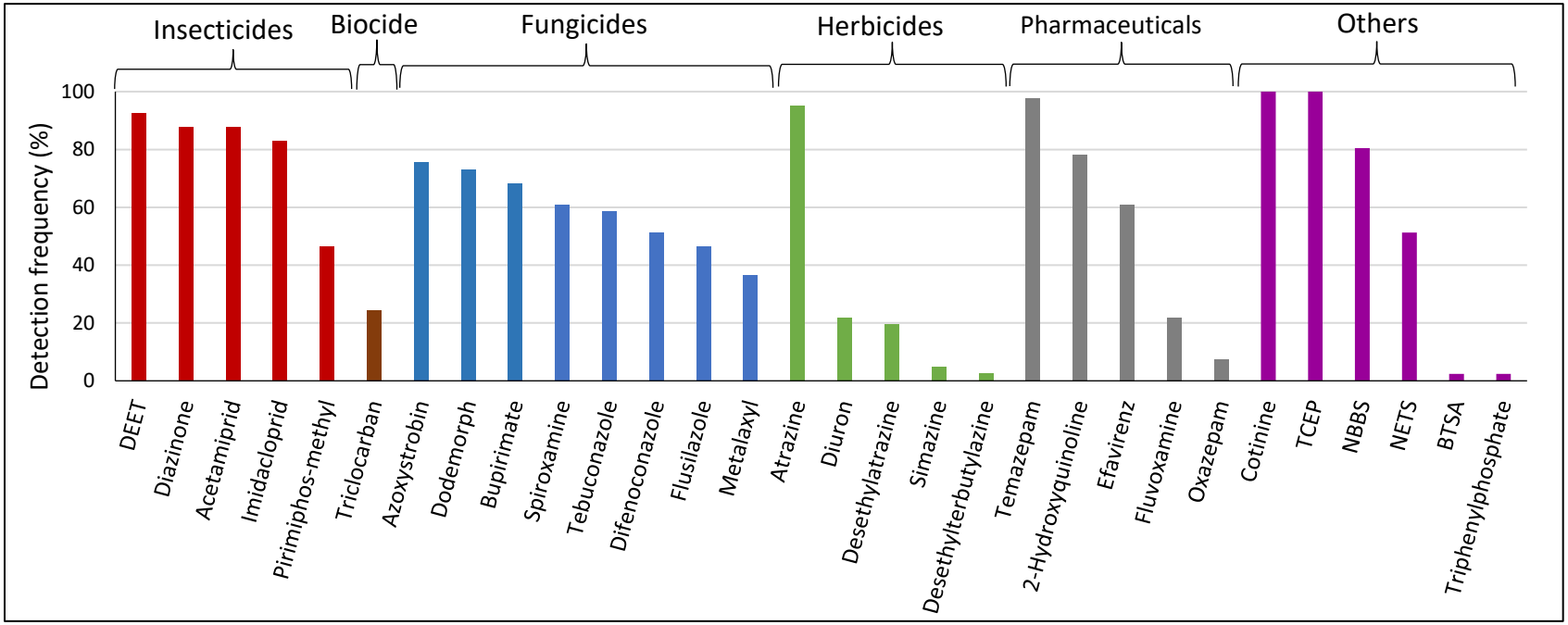


Figure 4.1: Frequency of detection for compounds in snail tissue samples from the Lake Victoria South Basin. TCEP: Tris(2-chloroethyl)phosphate; NBBS: N-Butylbenzenesulfonamide; NETS: N-Ethyl-o-toluenesulfonamide; BTSA: 2-Benzothiazolesulfonic acid.

Almost two thirds of the chemicals detected in snails were PaBs with 19 out of 30 compounds (Figure 4.1). Atrazine, a pre-emergence and post-emergence herbicide still in use in Kenya for the control of broadleaf weeds and grasses, was detected in 95% of the snail samples followed by the insect repellant diethyltoluamide (DEET) with 93%. The frequent detection of DEET may be explained by the high mosquito infestation and the high prevalence of malaria infecting 20 to 40% of the population in counties within the lake endemic region of western Kenya (Bashir et al. 2019). DEET is used as a topical insect repellant for the control of mosquitoes. It could be released into the water when performing water-related activities such as taking a bath in the rivers and reservoirs (Thomas et al. 2014), a common practice which was observed during sampling in the region. Individual pesticide concentrations ranged from 0.2 ng/g ww to 375 ng/g ww with maximum concentrations found for the herbicide atrazine at 375 ng/g ww, the fungicide bupirimate at 97 ng/g ww and the insecticide diazinon (37 ng/g ww). The neonicotinoids, acetamiprid and imidacloprid were present in the snail tissues in concentrations up to 27 ng/g ww and 21 ng/g ww, respectively.

Among PPCPs, temazepam (98%) and 2-hydroxyquinoline (78%) had the highest detection frequencies (Figure 4.1). Individual compound concentrations ranged from 0.8 ng/g ww to 137 ng/g ww. Maximum concentrations were recorded for efavirenz (137 ng/g ww) and 2-hydroxyquinoline (115 ng/g ww). This highest concentration for efavirenz was found in PS28 (Figure SI-1B) located within Kisumu County and could be attributed to the high HIV/AIDS prevalence (16.3%) and the access to antiretroviral therapy (ART) with a coverage of 90% (National AIDS Control Council NACC 2018) in the region. Efavirenz is a non-nucleoside reverse transcriptase inhibitor (NNRTI) used in combination with other medications as antiretroviral treatment for HIV/AIDS (Kreitchmann et al. 2019).

Plasticizers and flame retardants were also present in snail tissues with highest detection frequencies reported for tris(2-chloroethyl)phosphate (100%) and N-butylbenzenesulfonamide (NBS, 80%). Individual compound concentrations ranged from 2.8 ng/g ww to 481 ng/g ww. The highest concentration was found for N-ethyl-o-toluene sulfonamide (NETS, 481 ng g^{-1} ww), a compound with wide-spread use in industrial products and as an ingredient to pesticide formulations. In addition, cotinine was present

in all the snails sampled (detection frequency 100%) with concentrations up to 311 ng/g ww (Figure SI-1B). Due to the compound being the most abundant metabolite of nicotine, its high stability and half-life, cotinine has been suggested to be the ideal biomarker of tobacco exposure and smoking status (Petersen et al. 2010). Cotinine is excreted by human mainly through the urine and may end up in surface water through effluent discharge of domestic waste.

Variability in compound concentrations was observed in the different snail species sampled from the same site. Example sites include PS17 and PS28 (Figure SI-1B) where total concentrations in *Ceratophallus sp.* and *Melanoides tuberculata* were lower than those in other species. A plausible explanation for this observation may be different lipid contents and age which could not be considered during sample preparation. A study carried out by Duncan et al. (1987) noted considerable variation in total lipid for *Biomphalaria glabrata* (5%) and *B. alexandrina* (2%) snails and, in addition, intraspecific differences in total lipids (1-10%) in *B. glabrata.* Since most pollutants accumulate primarily in lipid tissue, variation in lipid content may lead to differences in the body burden of contaminants.

The predators of the snails investigated in this study include cray fish (Procambarus alleni), water bug (Belostoma flumineum), waterfowl and cichlid fishes (Monde et al. 2016; Halstead et al. 2018). These predators could biomagnify and bioaccumulate certain contaminants during feeding resulting in elevated levels of contaminants in the food chain. The consumption of contaminated food is a major source for xenobiotics in predating birds and mammals (Nendza et al. 1997). Effects of pesticide exposure in birds have been linked to neurotoxicity and endocrine disruption (Köhler and Triebskorn 2013). Other effects of pesticides include impaired foraging and chick rearing, eggs shell thinning and reproductive failure (Köhler and Triebskorn 2013). In a study carried out by Guo et al. (2016) on compound prioritization based on potential of secondary poisoning in fish-eating birds and mammals, diazepam was ranked the highest (risk score 0.1–1) among the pharmaceuticals tested. In addition, gregarious animals including fish and birds could biomagnify and bioaccumulate contaminants then migrate becoming a predominant pathway for contaminants in the environment (Blais et al. 2007).

4.3.2 Occurrence and distribution of pollutants in sediments

Out of the 429 targeted compounds analyzed in the sediment samples from the LVSB, 78 compounds were detected (Figure 4.2 and Table SI-6B).

Similar to snail samples, PaBs were the dominant chemical class with 71% of compounds detected in the sediments. Compounds frequently detected included DEET (98%), triclocarban (71%), diuron (65%), pirimiphos methyl (58%) and diazinon (56%) (Figure 4.2). High detection frequencies of diazinon and pirimiphos-methyl are in line with a study performed by Musa et al. (2011) who noted that diazinon and pirimiphos-methyl were among the commonly used pesticides in the Nyando catchment area which is within the LVSB. Individual compound concentrations reached up to 111 ng/g organic carbon (OC) with highest concentrations recorded for pirimiphos-methyl (111 ng/g OC), diuron (93 ng/g OC) and DEET (68 ng/g OC) (Figure 4.3). Pirimiphos-methyl is a broad spectrum insecticide used for the control of pest during storage. In addition, it is approved as an insecticide for indoor spraying against mosquitoes, cockroaches and houseflies (Pest Control Products Board 2018). Diuron is a selective herbicide for the control of weeds in sugarcane plantations. Concentrations reported in this study are within the range reported in a review by K'oreje et al. (2020) on the occurrence of pesticides in African river sediments.

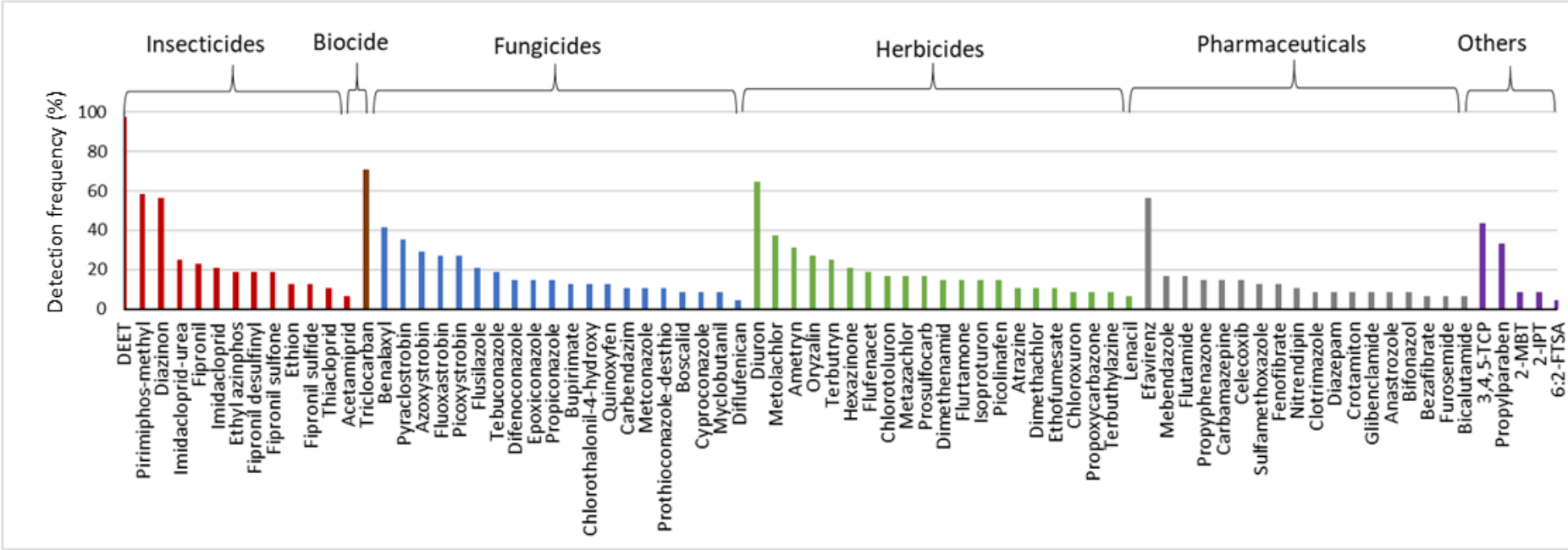
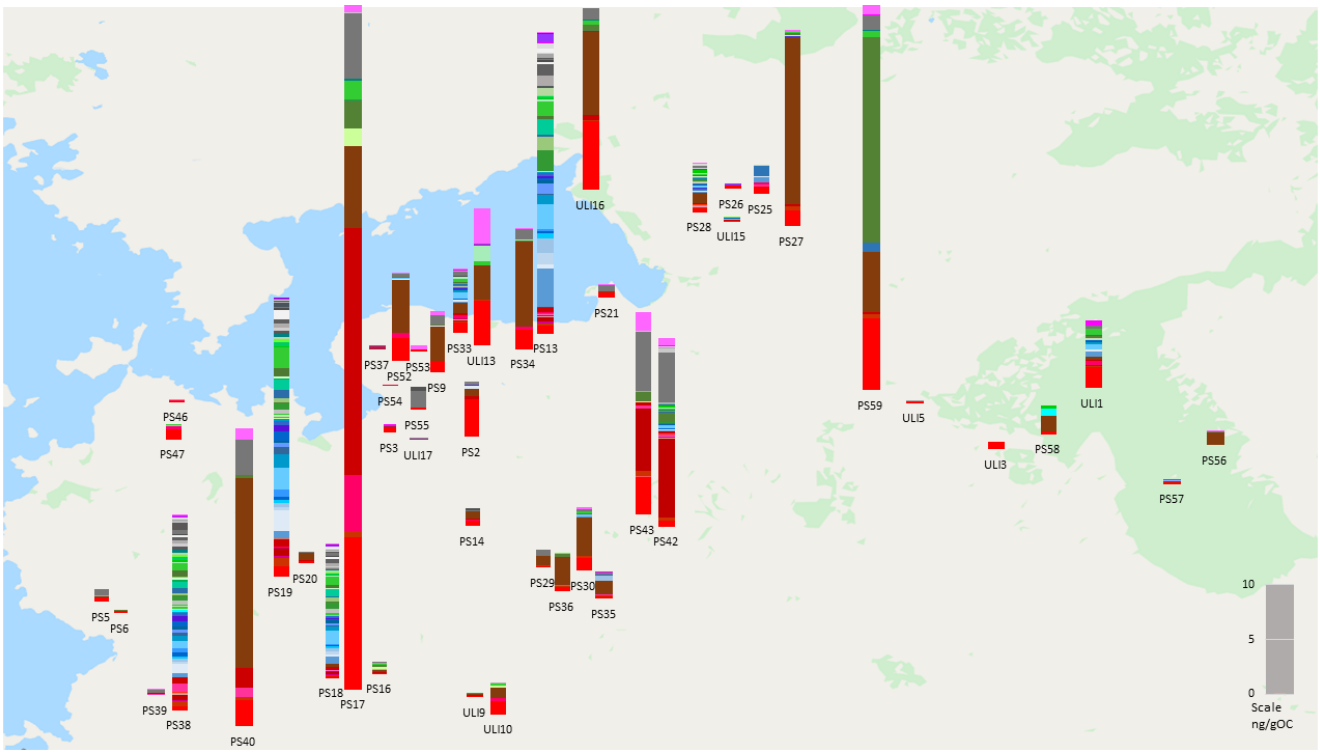


Figure 4.2: Frequency of detection for compounds in sediment samples from the Lake Victoria South Basin. 3,4,5-TCP: 3,4,5-trichlorophenol; 2-MBT: 2-morpholinothiobenzothiazole; 2-IPT: 2-Isopropylthioxanthone; 6:2-FTSA: 6:2 fluorotelomer sulfonic acid



DEET, Diazinon, Ethion, Ethyl azinphos, Fipronil
Fipronil desulfinyl, Fipronil sulfide, Fipronil sulfone, Imidacloprid, Imidacloprid-urea
Pirimiphos-methyl, Triclocarban, Azoxystrobin, Benalaxyl, Boscalid
Bupirimate, Carbendazim, Chlorothalonil-4-hydroxy, Cyproconazole, Difenoconazole
Diflufenican, Epoxiconazole, Fluoxastrobin, Flusilazole, Metconazole
Myclobutanil, Picoxystrobin, Propiconazole, Prothioconazole-desthio, Pyraclostrobin
Quinoxyfen, Tebuconazole, Ametryn, Atrazine, Dimethachlor
Dimethenamid, Diuron, Ethofumesate, Flufenacet, Flurtamone
Hexazinone, Isoproturon, Metazachlor, Metolachlor, Picolinafen
Terbuthylazine, 6:2 fluorotelomer sulfonic acid, Bezafibrate, Clotrimazole, Propyphenazone
Diazepam, Sulfamethoxazole, Carbamazepine, Crotamiton, Glibenclamide
Fenofibrate, Efavirenz, Anastrozole, Mebendazole, Bifonazol
Furosemide, Flutamide, Celecoxib, Bicalutamide, Nitrendipin

Figure 4.3: Compound concentrations and spatial distribution in sediment samples

A total of 19 PPCPs were detected with frequencies ranging from 6% (bezafibrate) to 56% (efavirenz). We also found the preservative propylparaben (33%) and the anti-cancer drug anastrozole (8%). To the best of our knowledge, this is the first study to report anastrozole occurrence in Kenyan aquatic ecosystems. The detection of anastrozole could be linked to the rising diagnoses and treatment of breast cancer in the continent and particularly in Kenya (Ekpe et al., 2019). Anastrozole is applied for hormone therapy during breast cancer treatment. Highest PPCP concentrations were detected for efavirenz with up to 29 ng/g OC. Other compounds detected at higher concentrations include crotamiton (1.8 ng/g OC) and diazepam (1.5 ng/g OC). The concentrations of the antibiotic sulfamethoxazole measured in this study (0.21 ng/g OC) fall below the concentrations reported by Kairigo et al. (2020) in sediments from Mwania river in Kenya by about one order of magnitude. This is probably due to differences in consumption patterns and the impact of municipal waste in the study area since their study was performed in an urban setting.

4.3.3 Comparison of the incidence of CECs in different environmental matrices of western Kenya

The chemical data from the present study were compared with the compounds found in the water phase in Kandie et al. (2020). In total, 142 compounds were detected in the study area with 79 compounds in water (Figure SI-2B), 30 compounds in snails and 78 compounds in sediments. Among these compounds, only nine were common in all three matrices (Figure 4.4) including acetamiprid, atrazine, azoxystrobin, DEET, diazinon, diuron, imidacloprid, pirimiphos-methyl and triclocarban. Although these compounds were present in all matrices, their ranking with respect to detection frequency and concentrations in the individual matrices was quite different. For example, atrazine and diazinon were among the compounds frequently detected at high concentrations in biota, whereas this was not the case in water and sediments. As expected, rather hydrophobic compounds such as pirimiphos-methyl (log K_{ow} 4.12) were found in higher concentrations in sediments and snails (PS17) than in water.

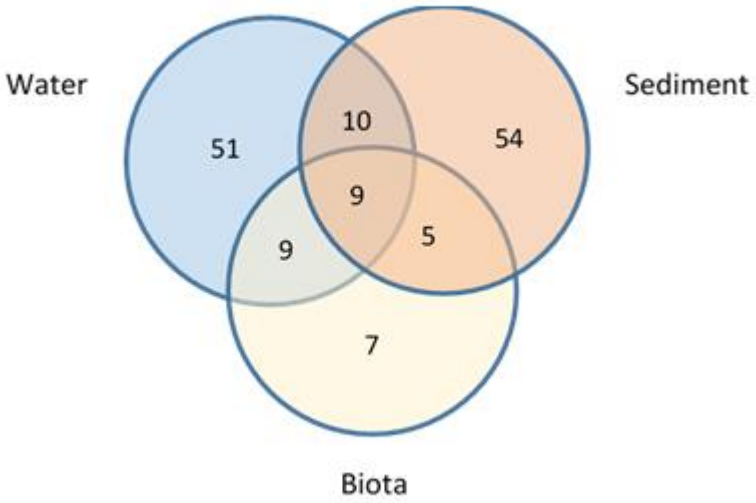


Figure 4.4: Numbers of detected compounds in water, sediment and snail samples within the study area

Correlations between concentrations in water, snails and sediments have been found only for few compounds. The strongest correlation was obtained for pirimiphos-methyl (r = 0.53) for concentrations in sediments and snails followed by water and snail concentrations of DEET (r = 0.35) and diazinon (r = 0.31) in sediment and water. The low correlations observed between biota and the other matrices suggest a quite complex bioaccumulation regime involving different uptake pathways (Contardo-Jara et al. 2011), complex temporal exposure patterns (e.g. due to pesticide peaks) and high small-scale variance of exposure.

Among the compounds present, 54 compounds in sediments, 51 in water and 7 in snails were specific to the individual phases (Figure 4.4), indicating the need to consider different complementary matrices in order to get a more comprehensive picture of contamination. As an example, the pharmaceutical efavirenz and fungicides difenoconazole, bupirimate, flusilazole and tebuconazole were not detected in grab water samples but could be quantified in snail and sediment samples. Efavirenz is moderately hydrophobic (log K_{ow} = 4.7) and is likely to partition to snail and sediment phases. Also,

the snapshot character of water samples or the need of enrichment in order to exceed detection limits could influence compound detections. In addition, the intermittent release of chemicals into the environment through a runoff event after spraying or emission events could influence the presence of a compound in the aquatic environment.

4.3.4 Impact of land use on contamination patterns in different environmental compartments

Among the 48 sampling sites, 17 sites could be clearly connected to specific land uses (Appendix B Figure SI-3), including agricultural areas (i.e., sugarcane, tea and rice) and reservoirs characterized by low anthropogenic inputs. The other sites showed mixed land use patterns and were not taken into consideration. Overall, sugarcane growing areas showed the highest concentrations of PaBs of all investigated types of land use (Figure 4.3, Appendix B Figures SI-1, SI-2 and SI-3).

Sites located in areas without evident anthropogenic influence (PS21, PS39, PS46, PS47 and PS58) were generally less contaminated although total CEC concentrations (1 µg/L) in water from PS 39 indicates hidden wastewater impact as a source for the pharmaceuticals acetyl-sulfamethoxazole (A-SMX) and diphenhydramine, and the industrial compound triethycitrate (Figure 4.3, Appendix B Figures SI-1, SI-2 and SI-3). Total CEC concentrations were up to 2 ng/g OC (mainly triclocarban and DEET) and 228 ng/g ww in sediments and snails, respectively. Cotinine, efavirenz, atrazine, N-ethyl-o-toluene sulfonamide and N-butylbenzenesulfonamide were the compounds contributing to the pollution found in snails. Low contaminant concentrations were observed in sites such as in PS46 located within Homabay County with no immediate impact from anthropogenic activities. In addition, a lot of the farmers in this region practice subsistence farming for household consumption, therefore less pesticide inputs may be used in the agricultural practices. The site PS 47 is located closely downstream of the Ruma national park, a wildlife reserve with limited human activities.

Agricultural sites such as PS17 and PS18 within sugarcane plantations had high contaminant concentrations in all the three matrices (Figure 4.3, Appendix B Figures SI-1, SI-2 and SI-3). Total compound concentrations were up to 11 µg/L in water, 555 ng/g

ww in snails, 304 ng/g OC in sediments. Notably, high concentrations were obtained for the PaBs atrazine in snails and pirimiphos-methyl, DEET and triclocarban in sediments, while 2,4-dichlorophenoxyacetic acid (2,4-D) and hexazinone predominated in water (Kandie et al. 2020). The high concentrations of pesticides are in agreement with sampling during the spraying period in sugarcane plantations (September-October). Other compounds that contributed to pollution included efavirenz and NETs (in both snails and sediments) while the hydrophilic CECs A-SMX, the sweetener acesulfame and triethycitrate were found only in water samples from these two sites. The presence of Awendo town nearby could contribute as an important source of these compounds into the aquatic system.

In rice plantation fields (PS25, PS26, PS27, PS28 and Uli15), high total compound concentrations of up to 3.3 µg/L in water, 598 ng/g ww in snails, 22 ng/g OC in sediments were detected. Similar to sugarcane sites, PaBs contributed most to the overall pollution with atrazine predominating in snails and triclocarban in sediments. Carbendazim and bendiocarb contributed most to water contamination (Kandie et al., 2020).

Among agricultural areas, sites within tea plantations (PS56, PS57, Uli1, Uli3) were least impacted with pollutants. Total CEC concentrations reached up to 6 ng/g OC in sediments with the insect repellant DEET as major contributor, while no snails were found at these sites. A plausible explanation for the low pesticide contamination is the mismatch of sampling (September and October) and the spraying period in tea plantations (June). Additionally, the streams in tea growing areas are protected by wide buffer zones while in other agricultural areas the farmland is much closer to the waterbodies leading to higher run-off potential.

In general, the occurrence of PPCPS in agricultural sites shows evidences of municipal waste impact on the local water bodies. For example, the pharmaceuticals efavirenz and 2-hydroxyquinoline were found in high concentrations in snails collected from rice fields. A plausible explanation for the high concentrations found could be the direct discharge of untreated domestic wastewater from residential areas, lack of sanitation facilities and effluent discharge from wastewater treatment facilities into the river.

4.3.5 Risk assessment based on sediment concentrations

Based on equilibrium water concentrations calculated from sediment, toxic risks were estimated for fish, crustacean and algae. For fish, the TU_{sum} ranged from $9.1x10^{-8}$ to $1.4x10^{-2}$ (Appendix B Table SI-7, Figure SI-4). Pirimiphos-methyl and imidacloprid-urea were identified as risk drivers for fish exposed to sediments in PS17, while diuron is predominating in PS59. Ethyl azinphos contributed greatly to the risk in PS42 and PS43. Maximum TUs were observed for pirimiphos-methyl (TU = 0.007) and the transformation product imidacloprid urea (TU = 0.005), but did not exceed the acute and chronic risk thresholds. However, with this low toxic risk on fish observed, it should be mentioned that natural and synthetic estrogenic hormones were not measured in this study. These hormones have been shown to drive effects on fish reproduction in the ng/L range resulting in the collapse of whole fish populations (Kidd et al. 2007). Estrogens are often emitted with untreated wastewater (König et al. 2017).

Cumulative TUs for crustaceans were generally higher than those obtained for fish (range: $9.6X10^{-8}$ to 1.1) with diazinon, fipronil sulfone and pirimiphos-methyl driving the overall risk (Figure 4.5, Appendix B Table SI-8,). Maximum TU_{sum} was reported for the site PS17 (TU_{sum} 1.1) impacted by the large agro-industrial sugarcane plantation with pirimiphos-methyl (TU = 0.99) and diazinon (TU = 0.11) driving the risk, both exceeding the ART (TU > 0.1) for crustaceans. The chronic risk threshold of TU > 0.001 was exceeded at 20 sites for diazinon, at 14 sites for pirimiphos-methyl and at four sites for fipronil sulfone.

For algae, the cumulative risk was higher than for fish but lower compared to crustaceans. The TU_{sum} ranged from $8.7X10^{-7}$ to 0.24, with the photosynthesis inhibitor diuron driving the risk to algae at most sites (Table SI-9, Figure 4.6). Diuron is a selective herbicide for the control of weeds in sugarcane plantations (Pest Control Products Board 2018). Highest TU values were obtained in sediments from PS59 (TU = 0.24) exceeding ART and PS17 (TU = 0.03) exceeding CRT.

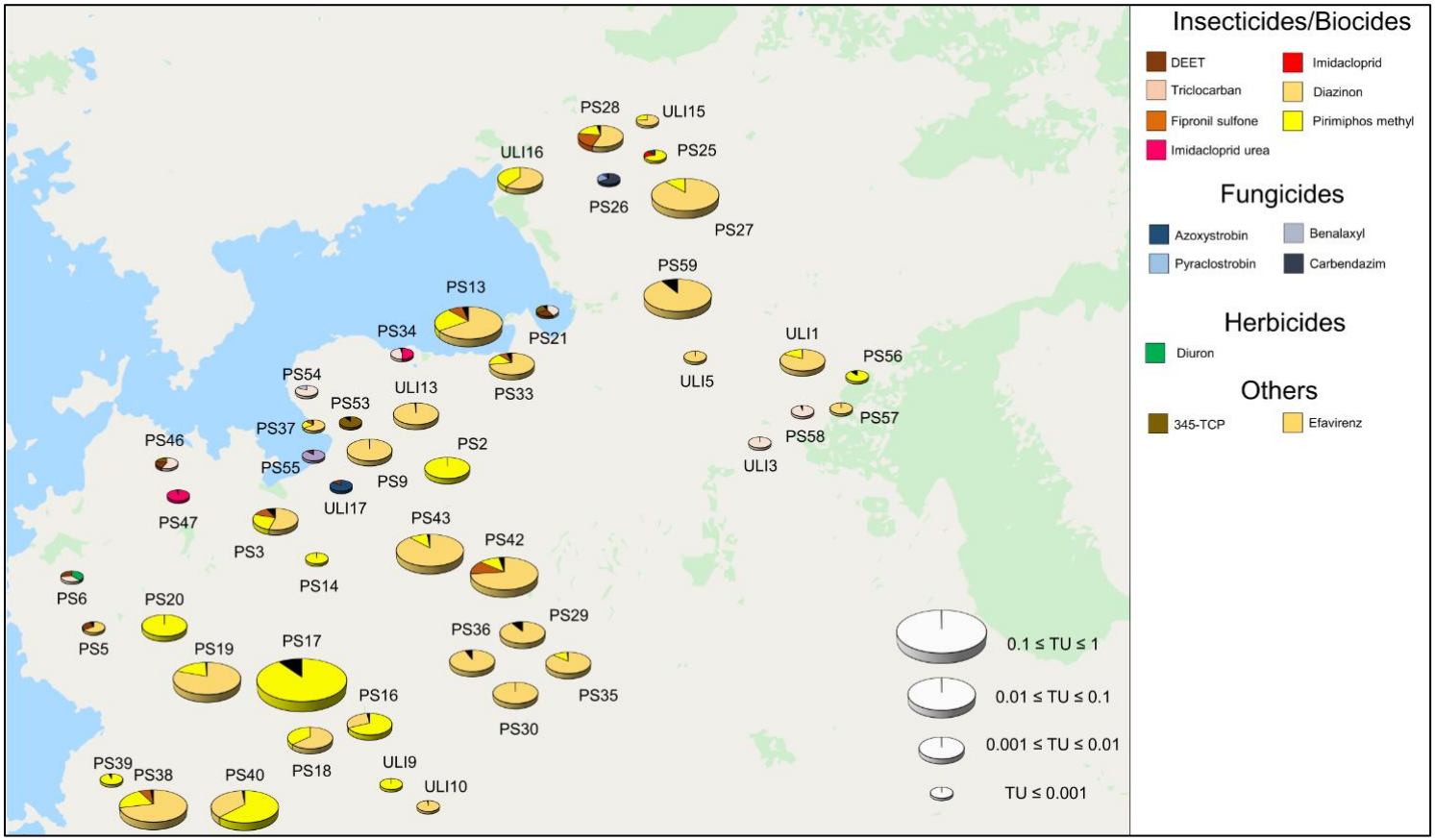


Figure 4.5: Distribution of risk of toxicity for crustaceans from compounds present in sediment extract based on equilibrium water concentrations. 345-TCP:3,4,5 trichlorophenol

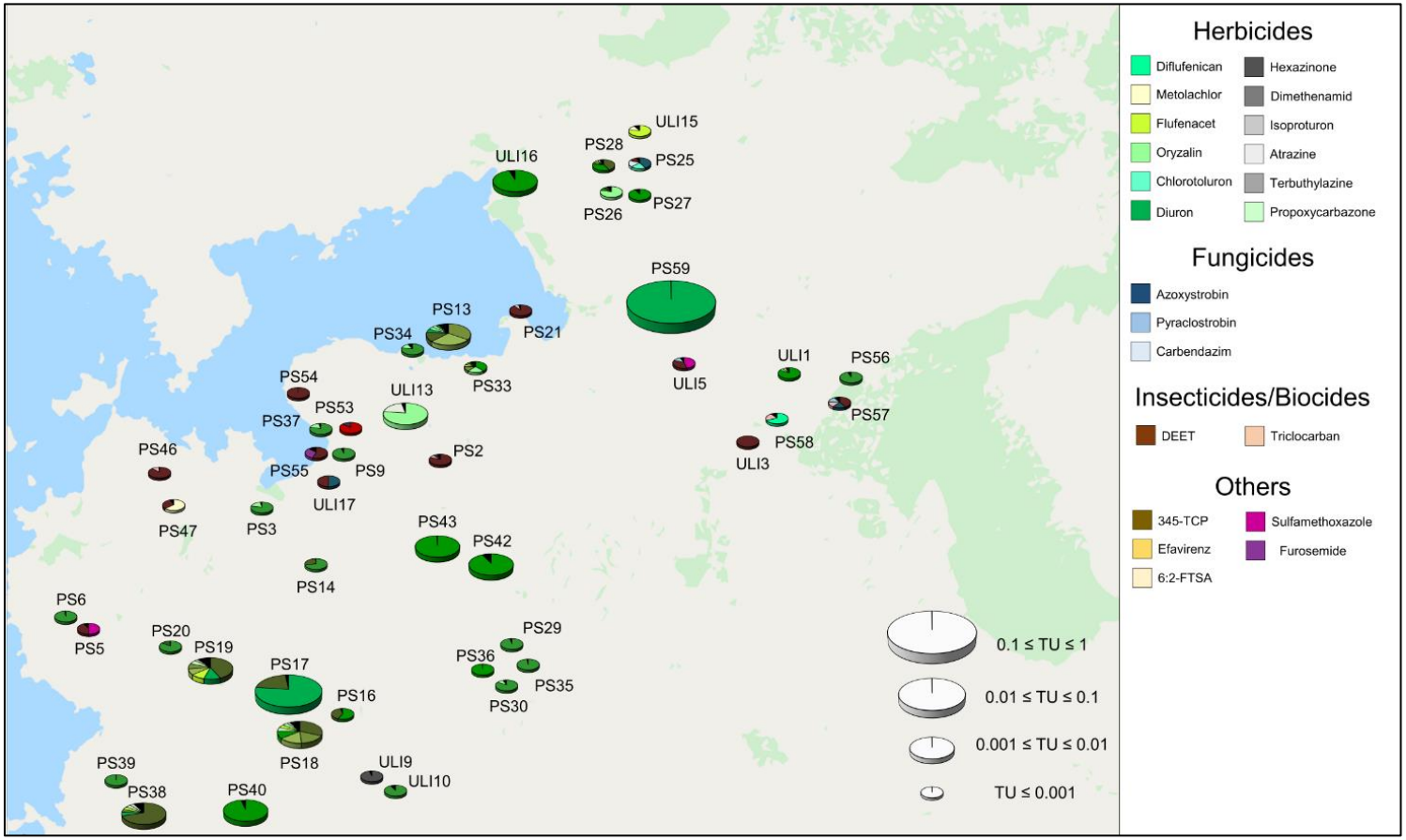


Figure 4.6: Distribution of risk of toxicity for algae from compounds present in sediment extract based on equilibrium water concentrations. 345-TCP:3,4,5 trichlorophenol; 6:2-FTSA: 6:2-fluorotelomer sulfonic acid

4.4 Conclusions

The present study bridges substantial data gaps on contamination of freshwater habitats in rural areas within western Kenya with CECs. The study demonstrates that complementing water monitoring with the analysis of sediments and biota may strongly increase the number of detectable contaminants allowing for a more comprehensive assessment of pollution. In this study 66 chemicals were detected only in snails and sediments while 51 compounds were found exclusively in water. Although common hot spots of contamination were identified in all three matrices, sediments and snails provide a picture that is rather independent from water concentrations measured in randomly taken grab samples. These findings support the complementary monitoring of biota and sediments as time-integrated samplers of pollution.

Although partly confounded by complex land-use structures, contamination of sediments and biota could be linked to specific agricultural production. Pesticide application in sugarcane plantations and rice fields were important sources of contamination and toxic risks to aquatic organisms. Poorly treated and untreated municipal wastewater and a lack of sanitation may be seen as another highly relevant source of pollution with pharmaceuticals and personal care products.

Very high toxic risks due to contaminated sediments were found for crustaceans in some of the sites, with diazinon and pirimiphos-methyl driving this risk. Substantial acute and chronic risk was also observed for algae mainly driven by diuron, while fish suffered only low toxic risk. Sublethal effects such as endocrine disruption and other specific effects by natural and synthetic steroids and pharmaceuticals discharged with untreated wastewater could not be considered in this study but might be more relevant for fish populations than acute toxicity. Future studies should focus on potential toxicological risk to humans and wildlife resulting from bioaccumulation and biomagnification of certain contaminants due to consumption of contaminated food.

Overall, this study indicates substantial contamination of rural areas in western Kenya and promotes systematic monitoring and assessment of CECs in different matrices in order to characterize and mitigate risks to ecosystems but also human health.

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Chapter 5

Pesticide pollution in freshwater paves the way for schistosomiasis transmission

Author's contribution statement

Declaration of author contributions to the publication:

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Contributing authors:

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What are the contributions of the doctoral candidate and his co-authors?

(1) Concept and design

Doctoral candidate: 30%
Co-author rest of the authors: 70%

(2) Conducting tests and experiments

Doctoral candidate: 50% (organized and carried out field work, sampling, sample preparation, chemical analysis)
Co-author 1, 2, 4, 5: 50% (organized and carried out field work, bioassay experiments)

(3) Compilation of data sets and figures

Doctoral candidate: 50 % (chemical and physical-chemical data evaluation)
Co-author 1,2,4: 50% (Acute tests data evaluation)

(4) Analysis and interpretation of data

Doctoral candidate: 30% (Chemical data interpretation)
Co-author 1,2: 60% (statistical analysis, risk assessment calculation, results interpretation)
Co-authors rest: 10% (Supervised data evaluation, final data evaluation, Advise on risk assessment)

(5) Drafting of manuscript

Doctoral candidate: 40%
Co-author all: 60%

I hereby certify that the information above is correct.

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Date and place Signature doctoral candidate

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Date and place Signature supervisor

Abstract

Schistosomiasis is a severe neglected tropical disease caused by trematodes and transmitted by freshwater snails. Snails are known to be highly tolerant to agricultural pesticides. However, little attention has been paid to the ecological consequences of pesticide pollution in areas endemic for schistosomiasis, where people live in close contact with non-sanitized freshwaters. In complementary laboratory and field studies on Kenyan inland areas along Lake Victoria, we show that pesticide pollution is a major driver in increasing the occurrence of host snails and thus the risk of schistosomiasis transmission. In the laboratory, snails showed higher insecticide tolerance to commonly found pesticides than associated invertebrates, in particular to the neonicotinoid imidacloprid and the organophosphate diazinon. In the field, we demonstrated at 48 sites that snails were present exclusively in habitats characterized by pesticide pollution and eutrophication. Our analysis revealed that insensitive snails dominated over their less tolerant competitors. The study shows for the first time that in the field, pesticide concentrations considered "safe" in environmental risk assessment have indirect effects on human health. Thus we conclude there is a need for rethinking the environmental risk of low pesticide concentrations and of integrating agricultural mitigation measures in the control of schistosomiasis.

5.1 Introduction

Schistosomiasis, also called bilharzia, is among the tropical diseases with the highest impact on socio-economic development, only exceeded by malaria. Approximately 218 million people are infected worldwide (WHO, 2017). Infection has been strongly associated with long-term disabilities (King & Dangerfield-Cha, 2008). The number of deaths due to schistosomiasis is poorly documented with estimates ranging between 11,700 (Lozano et al. 2012) to 280,000 each year (King & Bertino, 2008) because of hidden pathologies such as liver and kidney failure (Colley et al. 2014). Schistosomiasis is caused by flatworms of the genus *Schistosoma* which parasitize humans as their definitive host (supporting the adult life stage of the parasite). The intermediate hosts are freshwater snails of the family planorbidae which release infective larval stages (cercariae) into the water. Transmission occurs when humans are exposed to water containing infected host snails; direct infection from person to person is not possible (CDC 2018). People are infected during routine agricultural, domestic, occupational and recreational activities, which expose them to infested water. Over 80% of afflicted people live in sub-Saharan Africa (Hotez et al. 2014), but the disease concerns public health in most (sub)tropical countries worldwide (Hotez et al. 2014) and has recently established in Europe (Pennisi 2018).

Control strategies against schistosomiasis focus on the treatment with praziquantel that kills the adult worms in the human host. However, even mass drug administration does not prevent re-infection in infested water, and schistosomiasis has been observed to rebound within short time (Gurarie et al. 2018). For the sustainable control of schistosomiasis, it is also essential to interrupt the infection cycle by control of the intermediate hosts (Secor, 2014; Lo et al. 2018; Chadeka et al. 2019). Host snails are susceptible to predation by organisms such as shrimps (Sokolow et al. 2014) and ostracods (Yousif, 2013) which have been applied as biological control agents. Additionally, host snails are susceptible to competition from other snails and insects that feed on periphyton (microbes attached to surfaces), detritus and water plants (Mone et al. 2003; Yeung & Dudgeon 2013; Barbosa et al. 2014). Spreading of schistosomiasis

has been often linked to the loss of biodiversity and the ecological degradation of freshwater habitats (Johnson & Thieltges, 2010; Secor, 2014; Sokolow et al. 2017). The findings suggest that host snails are significantly controlled by antagonistic species in natural habitats but that this ecosystem service is sensitive to anthropogenic impact. Therefore, it is essential to identify key environmental factors that drive the interactions of host snails with their associated community.

Recently, agricultural pesticides have returned to the focus of public attention as causes for the worldwide decline in insects and biodiversity (Liess & von der Ohe, 2005; Geiger et al. 2010; Beketov et al. 2013). Tropical regions, characterized by extensive agriculture and heavy rainfalls, are known areas of endemicity of schistosomiasis. In such conditions there is a high risk of surface run-off that washes pesticides from agricultural fields into adjacent freshwater systems (Liess & Schulz, 1999). However, information on pesticide concentrations in tropical freshwaters and their effects on the macroinvertebrate community are often fragmented and inadequate (London et al. 2005; Musa et al. 2011). In temperate latitudes, snails are known as one of the macroinvertebrate taxa being most tolerant to pesticides (Von der Ohe & Liess, 2004). In mesocosms, high concentrations of insecticides and herbicides favoured host snails indirectly through the reduction of predators and through the replacement of suspended algae with periphyton that serves as food for snails (Halstead et al. 2018). Additionally, even low levels of agricultural pesticides result in a typical replacement of sensitive macroinvertebrates by more tolerant taxa in mesocosms (Van den Brink et al. 2000) and in natural streams (Knillmann et al. 2018; Liess & von der Ohe, 2005). These effects are usually driven by insecticides that are most toxic to many macroinvertebrates (Liess & von der Ohe, 2005). Therefore, we hypothesized that pesticide pollution may favour highly tolerant snails that host human-pathogenic schistosomes over their more sensitive natural enemies and thus increase the risk of schistosomiasis transmission.

5.2 Materials and methods

5.2.1 Acute toxicity tests

We investigated acute insecticide sensitivity for all macroinvertebrate taxa that could be found in sufficient quantities in October/November 2018 from 6 sites in the study area. Site selection was based on host snail availability, high macroinvertebrate diversity, and low presumed pesticide pollution as indicated by buffer strips to minimize testing populations that may have developed pesticide resistance (Becker & Liess, 2017). Test organisms were collected using sweep nets, standard pint dippers and snail catchers; they were sorted and identified to family level in the field. The organisms were placed in plastic boxes filled with water from the sampling site and aerated using battery-operated air pumps. The containers were cooled in a portable fridge at 18 °C in order to prevent mortality during transport to the laboratory. The organisms were acclimatized to test conditions overnight. Tests were performed in a shaded screen-house with temperatures ranging from 20 to 33 °C. Commonly applied agricultural insecticides comprise three major classes with distinct modes of action: Organophosphates/carbamates, neonicotinoids and pyrethroids. If sufficient test organisms were available, we tested the acute sensitivity to one of the most toxic substances among the organophosphates (diazinon) and neonicotinoids (imidacloprid) measured at the study sites, respectively. To increase environmental realism, we applied local plant protection products containing the active substance and additional carriers that might have affected the toxicity.

Tests were performed according to the Rapid Test protocol for field-collected organisms (Kefford, 2013) with minor modifications. Imidacloprid was applied as a 70% wettable granule formulation (Loyalty, manufactured by Shandong United Pesticide Industry Co., Ltd. Jinan city, China) and diazinon was applied as a 60% emulsified liquid formulation (Diazol, repacked and distributed by Laibuta Chemicals Ltd). Fresh stock solutions were prepared a few hours before each test dissolving the formulations in a 1:1 mixture of bottled water and activated carbon filtered stream water. This mixture was a compromise to minimize adverse effects from water to which the organisms had not been adapted,

and to minimize potential effects from residual toxicity and from dissolved solids in the stream water which can absorb pesticides. Stock solutions of 165 mg active substance/L were left to stir overnight in amber vials; no additional solvents were applied.

Test organisms were exposed to 6 test pesticide concentrations including a control. For each taxon test concentrations were selected from the following geometric series such that they covered the expected range of a partial response from <5% mortality in the lowest concentrations to >95% mortality in the highest concentration: 0.001; 0.004; 0.014; 0.055; 0.209; 0.792; 3.01; 11.4; 43.5, 165 mg/L.The ranges expected to result in a partial response were identified from data bases (Lewis et al. 2019; USEPA 2019) and previous studies (Becker & Liess, 2017) for related taxa and substances. The tests were performed in 100 mL glass vessels containing up to 5 individuals of the same species (predators were kept individually to avoid cannibalism). The test medium was constantly aerated using aquarium pumps connected to glass pipettes via a silicone tube. Only the glass pipette had contact with the test solution and the air flow was controlled through a clamp on the silicone tube. After 24 h and = 48 h, mobile, immobilized and dead individuals were counted. Individuals were considered immobilized when no movement was observed within 10 s of undisturbed observation or after probing with a rod; fanning of gills was not considered movement.

5.2.2 Field sampling

The 48 study sites located in Homa Bay, Kericho, Kisii, Kisumu, Migori and Nyamira in Western Kenya were investigated from September to November 2017. The aquatic habitats were chosen from areas characterised by different types of land use and crops grown, identified using aerial photos from Google Maps. We classified the sampling sites according to habitat types (major tributary, minor tributary, irrigation channel, oxbow lake, reservoir or rice field) and the surrounding dominant land use within 50-100 m (natural, agricultural, semi-urban, urban or industrial). Agricultural land use was classified by farm type, subsistence or irrigation schemes and crop type (maize, tea, sugarcane or rice).

Streams, rivers and oxbow lakes were sampled across a 50-metre section whereas dams and irrigation schemes were sampled at four sub-sites. The aquatic habitats were surveyed for the presence of submerged, emerging or floating vegetation as well as algal bloom and the percentage of detritus cover. Depth was measured at the bank at point of sampling using a metre rule or was indicated as >1 m. Flow velocity was estimated with the drift approach. Additionally, we measured physicochemical parameters (temperature, conductivity, pH, dissolved oxygen, carbonate hardness, ammonium, phosphate, nitrate, nitrite and nitrite) and the turbidity on site using cholorimetric test kits (MACHEREY-NAGEL Quantofix, Düren, Germany), a turbidimeter (WTW TURB 355 IR, Weilheim, Germany), a multi-measurement probe (EXTECH ExStick EC500, Boston, USA) and an oxygen probe (EXTECH ExStick DO600, Boston, USA).

For pesticide analyses, grab samples were taken using pre-cleaned glass beakers. Briefly, oven dried 500 mL beakers were rinsed three times with the sample water and filled up to the top. After suspended solids settled, 1 mL aliquots were taken into five 2 mL autosampler amber glass vials (Phenomenex, Germany) using a volumetric pipette. All samples were immediately stored in a portable freezer (Waeco Compressor Cooler Box - 50 litres #CF-50) at -4 °C and transferred to the laboratory where they were stored at -20 °C until analysis. For quality control, sampling and trip blanks were taken during each sampling campaign.

Macroinvertebrates were sampled along four equal sections of the water body. Banks were sampled in a criss-cross fashion along the sampling points using littoral sweep nets, dippers and snail catchers; a standardised sampling procedure was predetermined to collect macroinvertebrates comprehensively within the different microhabitats and habitat types. In brief, each site was sampled for 30 minutes by two persons in parallel. Collected macroinvertebrates were sorted and counted in white plastic trays and preserved in 70% ethanol. Some host snails were transported to the laboratory and checked for *Schistosoma* infection. The snails were kept individually in a 24-well plate (Nunc 142475 Nunclon) and exposed to sunlight for a minimum of 30 minutes, and shed cercariae were = identified under a dissecting microscope (Zeiss AxioCam5 100-400x) and an

identification key for cercariae (Frandsen & Christensen, 1984). Macroinvertebrates were identified under a dissecting microscope (Zeiss AxioCam5 100-400x) to the lowest taxonomic level possible with the available identification keys (Brown, 1994; Day et al. 1999: 2001a: 2002a: 2001b: 2002b: 2002c; de Moor 2003a:2003b; Harrison 2009). Based on these data we calculated the following biological indices: the overall macroinvertebrate individual number, the species richness, Pielou's species evenness, the Shannon index for species diversity, the ASPT indicator for stream health from the South African Scoring System SASS (Dickens & Graham 2002), and the dominance (relative abundance) of potential predator and competitor species of the host snails. We considered all taxa as potential predators that comprise more than a marginal proportion of predatory species in the study region that may feed on freshwater snails or their eggs. Similarly, we considered all taxa as potential competitors that comprise more than a marginal proportion of periphyton feeders or herbivores in the study area.

5.2.3 Pesticide analysis

Details on the analysis of pesticide residues in water samples have been described in Kandie et al. 2020. In brief, 25 µL of an internal standard solution containing 40 isotope-labelled compounds (40 ng/mL, 25 µL of methanol and 10 µL of 2M NH4-formate buffer (pH 3.5) was added to each sample prior to instrumental analysis. Analysis was performed using high-performance liquid chromatography (HPLC, Ultimate 3000 LC) coupled to high resolution mass spectrometry (HRMS, QExactive Plus MS) from Thermo Scientific. The sample (100 µL) was directly injected for chromatographic separation (Phenomenex Kinetex c18 EVO, 50 x 2.1 mm, 2.6 µm particle size), equipped with a pre-column (5.0 x 2.1 mm) and 0.2 µm in-line filter using a methanol/water gradient containing 0.1% formic acid. Heated electrospray ionisation (ESI) was performed for both the negative and positive modes with combined full scan run (100−1500 m/z) at a nominal resolving power of 70,000 (referenced to m/z 200) and data-independent MS/MS fragmentation (DIA) at a nominal resolving power of 35,000. An isolation mass window of m/z = 50 (m/z range 122-860) or m/z = 260 (m/z range 860–1370) was used in DIA analysis. Matrix matched calibration standards were prepared for 11 calibration levels

(ranging from 1 to 2,000 ng/L) using 1 mL filtered water from a pristine reference stream (Wormsgraben, Harz Mountains, Germany). Quantification of detected pesticides was performed using isotope–labelled internal standards of compounds with closest retention time to the target compound. Data evaluation was performed using MZmine (Version 2.38 http://mzmine.github.io/) and trace finder (Thermo, version 4.1).

5.2.4 Data analysis

All data were analyzed using the software R 3.5.2 (R, 2018). From the mortality observed in the acute toxicity tests we calculated the acute lethal median concentrations after exposure for 24 h ($LC_{50,24h}$) with 4-parameter log-logistic non-linear regression available with the package drc 3.0-1 (Ritz et al. 2015). The parameters for the upper and lower boundary were fixed to 1 and 0, respectively. If a taxon had been tested at more than one date or from more than one site, data were merged prior to the analysis. Tests which showed > 30% control mortality were excluded from analyses. The resulting LC_{50} values were ranked in ascending order to obtain the species sensitivity distribution (SSD). This increase in the proportion of affected taxa with pesticide concentration was described using a quasibinomial generalized linear model (GLM) with a logit link function.

For all GLMs in this publication, *p*-values were obtained from likelihood-ratio tests that compared each model to a null model without the environmental variable. Depicted means and 95% confidence intervals were extracted from (generalized) linear models using the package effects 4.1-0 (Fox & Weisberg, 2018). Normally distributed residuals and homoscedasticity were confirmed using normal Q-Q plots and plotting residuals vs. fitted values; GLMs were inspected using scaled residuals available with the package DHARMa 0.2.0 (DHARMa, 2018). The effects of each environmental variable measured on the incidence of host snails were analyzed using one-way binomial GLMs (binary regression) with a complementary log-log link function which allows for a non-symmetric dose-response curve. The effects of each environmental variable measured on the density of existing host snail populations were analyzed using one-way GLMs with a zero-truncated negative binomial distribution of residuals and a log link function available with the package VGAM 1.0-6 (VGAM, 2018). This way we dealt with overdispersion and with

the missing possibility for the population density to be zero. Because many effects on the population density of host snails were driven by a single site (site 39) with extraordinarily high numbers of host snails and other macroinvertebrates, we repeated the analysis on the effects of host snail density with that site excluded. Only those environmental variables were considered in further analyses which significantly ($p < 0.05$) explained the population density after site 39 had been excluded.

Environmental variables that explained the incidence or population density of host snails were combined in a hurdle model available with the package pscl 1.5.2 (Zeileis et al. 2008). Hurdle models consist of two connected generalized linear models to simultaneously fit the incidence (zero part) and the population density (count part). Prior to modeling, the environmental variables were standardized (normalized and centered) to make the model parameters comparable. Environmental variables that significantly ($p < 0.05$) explained the incidence were incorporated in the zero part of the model (binomial GLM with complementary log link), and environmental variables that explained the population density were incorporated in the count part (zero-truncated negative binomial GLM with log link). To avoid overfitting, we applied an additive model without interactions. Then we sequentially removed all non-significant environmental variables based on a likelihood ratio test (backward elimination). Each time a variable had been removed, we started testing again with the least-significant of the remaining variables according to the model statistics.

Additionally, we fitted a hurdle model with all the environmental variables that on their own significantly explained the incidence or the population density included in both the zero and the count part. Hurdle models consist of two connected generalized linear models to simultaneously fit the incidence (zero part) and the population density (count part) (Zeileis et al. 2008). We removed all non-significant effects from this full model in a stepwise backward-elimination process and then sorted the remaining effects based on the magnitude of their regression coefficients. Due to multicollinearity, selecting a single minimum adequate model can lead to different results depending on the method of model selection (Mac Nally, 1996). Therefore, this model was subjected to hierarchical

partitioning. Because this procedure is currently not available for hurdle models in R, we extended the code of the function hier.part from the package hier.part 1.0-4 (R hier.part, 2013). The modified function started with a null model and each step included an environmental variable to both the zero and the count part at the same time. The improvement of the goodness-of-fit that resulted from the inclusion of an environmental variable was quantified using the log-likelihood.

Relations among the environmental explanatory variables were visualized using a principal component analysis (PCA) available with the function prcomp in basic R. A PCA reduces complexity by combining correlated environmental variables to few "supervariables" called principal components. The data were standardized prior to the analysis. Additionally, the association of the second principal component with the log-transformed number of macroinvertebrate individuals was analyzed using ordinary one-way linear regression.

The effect of pesticide pollution (TU_{max}) on the overall community composition consisting of grazers, potential predators and other macroinvertebrates was analyzed using a permutational multivariate analysis of variance (PERMANOVA) available with the package vegan 2.5-4 (R Vegan 2019). We also investigated the effect of pesticide pollution on the taxonomic composition of potential predators using a PERMANOVA. To investigate effects of the species composition of potential predators on the grazer composition, we performed a PCA on the predator composition. The first to fifth principal component was then fitted vs. the dominance of snails within grazers using quasi-binomial GLMs with a logit link function to account for the possibility of overdispersion. The proportions were weighted with the numbers of observed grazers. Similarly, the effects of pesticide pollution on the dominance of all grazers, of potential predators, and of snails within the guild of grazers were analyzed using quasi-binomial GLMs with a logit link function. The proportions were weighted with the numbers of observed individuals or of observed grazers, respectively. All data were analyzed using the software R 3.5.2 (R, 2018).

5.3 Results

We studied how pesticide pollution and additional environmental factors affect the macroinvertebrate community composition in a typical endemic region of schistosomiasis. For this, we sampled 48 freshwater sites in the Kenyan Lake Victoria Basin (Figure 5.1). The habitats ranged from small and medium-sized streams to irrigation channels, oxbow lakes, reservoirs and rice fields, and thus covered the main inland transmission sites in the study area (Sang et al. 2014). Each site was monitored once during the rainy season in October 2017. To confirm the hypothesized high pesticide tolerance of host snails, we collected host snails and other common invertebrates and tested their acute sensitivity to two insecticides covering different modes of action.

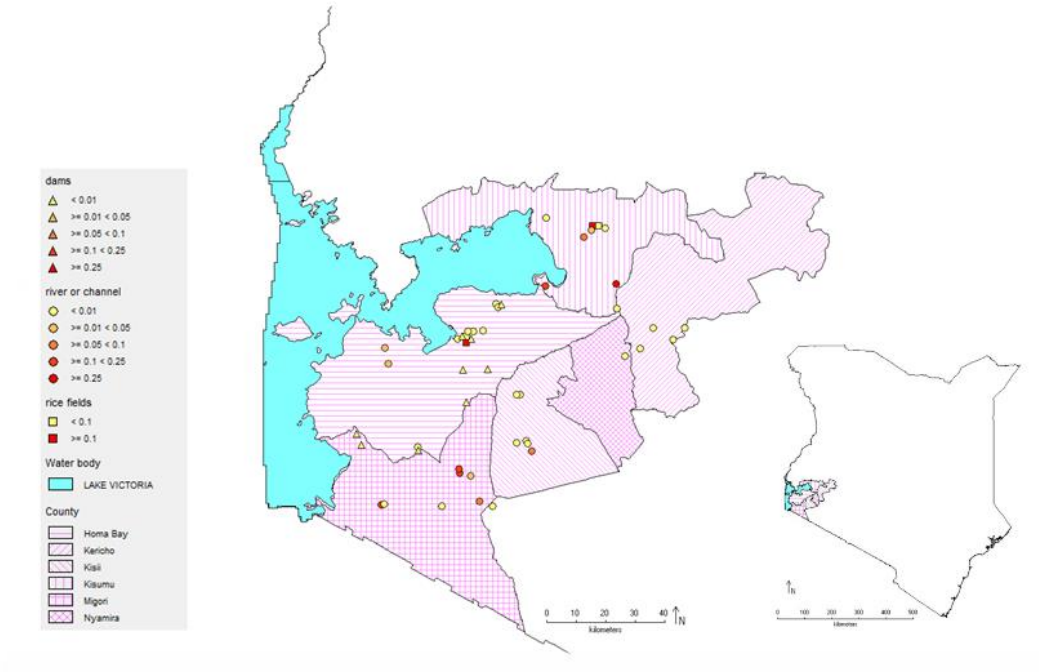

Figure 5.1: Dominance of host snails of the pathogens of human schistosomiasis among the study sites. Sites were depicted by the shapes of the icons as either reservoirs (triangles), streams/channels (circles) or rice fields (squares). The dominance of host snails (number of transmitting planorbid snails / total number of individuals per site) is represented by the shade of the icon. Maps created using DIVA-GIS 7.5.0. https://diva-gis.org/.

5.3.1 Pesticide tolerance of *Schistosoma* host snails

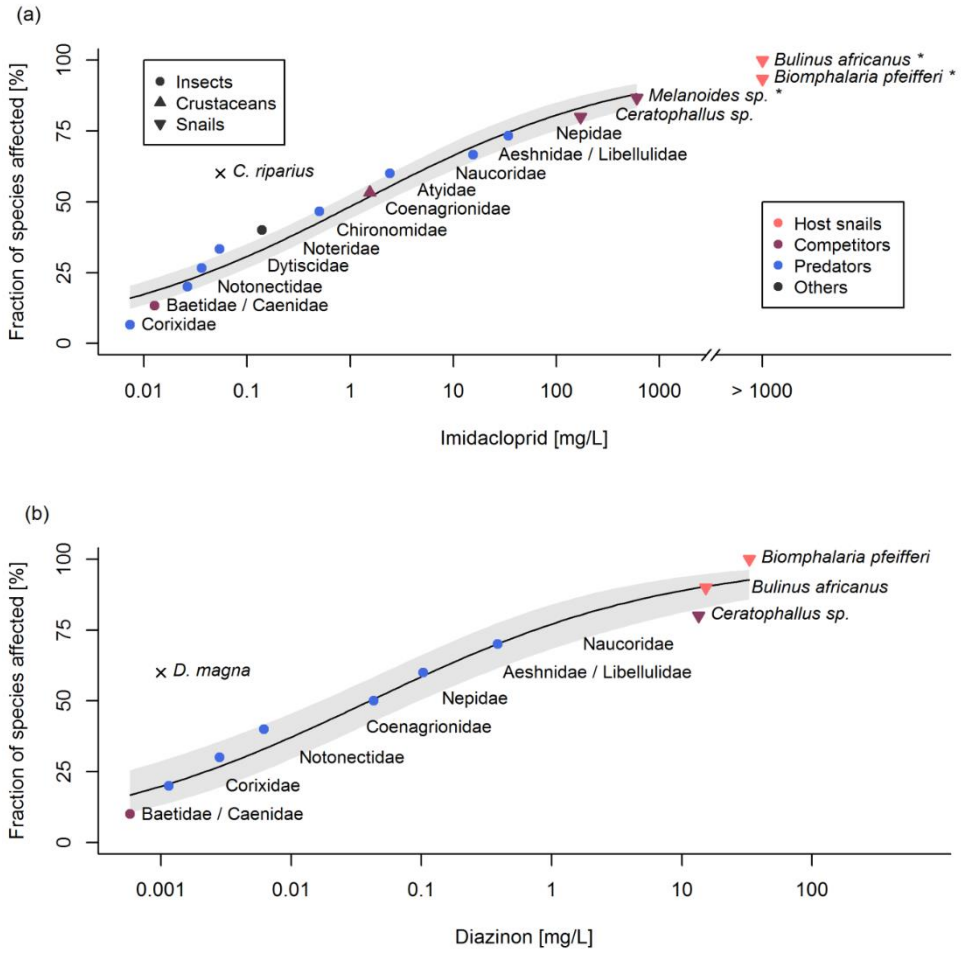

Figure 5.2: Species sensitivity distribution (SSD) of freshwater macroinvertebrates from the study region to common agricultural insecticides. Data points show the acute LC_{5024h} for various species. The SSD curves were fitted using a quasibinomial GLM with logit-link; means ± 95 % confidence intervals are shown. (a) Sensitivity distribution to the neonicotinoid insecticide imidacloprid. χ^2 = 230.69, res. df = 11, p < 0.001, McKelvey-Zavoina's pseudo-R^2 = 0.29. For *Melanoides sp.*, *Bulinus africanus* and *Biomphalaria pfeifferi* the LC_{50}exceeded the highest test concentration and was extrapolated from non-linear regression (*Melanoides sp.*) or estimated. (b) Sensitivity distribution to the organophosphorus insecticide Diazinon. χ^2 = 115.89, res. df = 8, p < 0.001, McKelvey-Zavoina's pseudo-R^2 = 0.40. (a, b) The acute $LC_{50\ 48h}$ of the most sensitive standard reference taxa (*Chironomus riparius* and *Daphnia magna*) was added for comparison (Mwinzi et al. 2012; Lewis et al. 2019) and used for the calculation of toxic units (see text).

Among all macroinvertebrates tested, host snails of human-pathogenic schistosomes showed the highest tolerance to the neonicotinoid insecticide imidacloprid and to the organophosphorus insecticide diazinon (Figure 5.2a and 5.2b), both of which were

commonly found at the study sites. The acute median lethal concentration of imidacloprid after exposure for 24 h ($LC_{50\ 24h}$, concentration that killed 50 % of test organisms) ranged from 0.007 mg/L for corixidae (true bugs) to 599 mg/L for the non-host snail *Melanoides sp*. The mortality of the host snails *Bulinus africanus* and *Biomphalaria pfeifferi* remained below 10% even at the highest concentration tested (165 mg/L) so that we could only estimate their respective LC_{50}. This test concentration was close to the solubility limit of imidacloprid in water (610 mg/L, Lewis et al. 2019), indicating very high insecticide tolerance of the host snails. The $LC_{50,24h}$ of diazinon for other taxa ranged from 0.5 µg/L for baetidae and caenidae (mayflies) to 13.5 mg/L for the non-host snail *Ceratophallus sp*. Again, the host snails *B. africanus* (15.2 mg/L) and *B. pfeifferi* (33.0 mg/L) were the most tolerant species.

5.3.2 Pesticide pollution in the study area

The surveyed aquatic habitats showed considerable agricultural pesticide pollution. We analyzed 28 commonly applied active substances and degradation products and detected all the compounds in water samples, ranging from 5 to 27 (median = 20) substances per site (Appendix C Table SI-1). To quantify the toxicity, pesticide concentrations were converted to toxic units (TU) using the formula $TU = log_{10}(\frac{Concentration}{LC50_{reference}})$ with *Concentration* being the measured concentration of a pesticide and $LC_{50reference}$ being the LC_{50} of that pesticide for a standard reference organism (typically *Daphnia magna*, Appendix C Table SI-1) (Tomlin, 2000). Toxic units of -1 or -2 represent pesticide concentrations of 1/10 or 1/100 of the LC_{50}, respectively. Toxic units of pesticides in agricultural European, Siberian and Australian streams typically reach from ≤ -5 to -1 (Schaefer et al. 2012; Becker & Liess 2017).

In two sites all pesticides detected were below the limit of quantification such that the toxic unit of the most toxic compound (TU_{max}) could not be calculated. However, in the other sites, the TU_{max} ranged from -6.4 to -1.2 with a median TU_{max} of -2.3 (Appendix C Table SI-2). The most toxic substances found were the organophosphorous and carbamate insecticides bendiocarb (most toxic substance at 17 sites, TU up to -1.66, Table SI-3C),

diazinon (most toxic at 15 sites, TU up to -1.71) and pirimiphos-methyl (most toxic at 5 sites, TU up to -1.21). Due to difficulties in the quantification of very low pesticide concentrations, the minimum TU_{max} was set to -5 for further analyses, a threshold at which typically no ecological effects have been observed in field studies (Schaefer et al. 2012; Becker & Liess 2017; Knillmann et al. 2018).

5.3.3 Environmental factors driving host snail abundance

We investigated the influence of 27 environmental variables on the abundance of host snails, covering habitat type, land use, water chemistry and the composition of the macroinvertebrate community (Table SI-4C). Host snails were found in 9 out of a total of 48 sites investigated in 2017; at one site they were infected with human-pathogenic schistosomes. The abundance of host snails encompasses the incidence, i.e. the probability of a population to occur at a given site, and the density of existing populations. Both endpoints can be driven by different environmental factors and thus were analyzed separately in a first step.

When each environmental variable was considered individually, the incidence of host snails increased significantly with pesticide toxicity (n = 48, χ^2 = 7.71, res. df = 46, p = 0.005, Figure 5.3), species diversity (χ^2 = 4.42, res. df = 46, p = 0.035) and species richness (χ^2 = 4.39, res. df= 046, p = 0.036). Additionally, the incidence of host snails decreased with increasing dissolved oxygen (χ^2 = 8.06, res. df = 46, p = 0.004) and with the increasing dominance (proportion on all macroinvertebrates) of other grazers and herbivores that act as potential competitor species (χ^2 = 9.09, res. df = 46, p = 0.003; Table SI-4C).

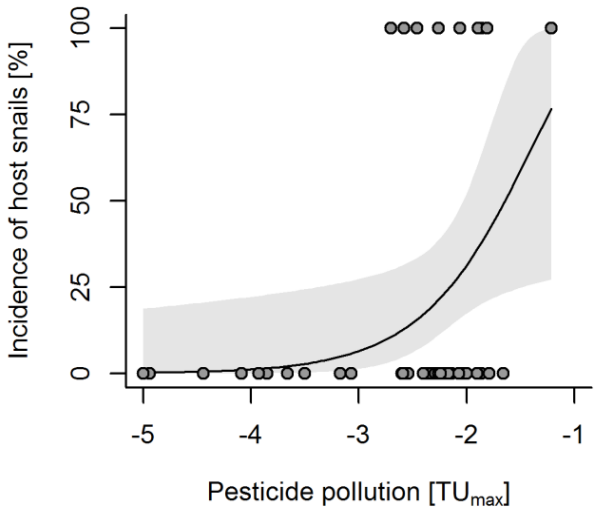

Figure 5.3: Pesticide pollution increases the incidence (probability of occurrence) of snails that act as hosts of schistosomiasis. Binomial GLM with complementary log-log link function; $\chi^2 = 7.60$, res. df = 46, $p = 0.006$, McFadden's pseudo-$R^2 = 0.16$. Means ± 95 % confidence intervals are shown. Pesticide pollution was quantified as $\log_{10}$ of the maximum ratio of a pesticide concentration measured in a grab sample of water vs. the acute $LC_{50}$of that pesticide for a standard reference organism (TU_{max}). TU_{max} of marginally polluted sites ($n = 4$) was set to a minimum of TU -5.

Environmental effects on the density of host snail populations were driven by a single stream (Table SI-5C). This stream was characterized by extraordinarily high numbers of host snails and other macroinvertebrates. The site was located 100 m downstream of a bathing and washing area and was the only site at which infected host snails were found. When we excluded this site as an outlier, population density was explained only by a decrease in density with increasing turbidity ($n = 8$, $\chi^2 = 4.50$, res. df = 6, $p = 0.034$).

In a second step, we combined the effects identified on the incidence and population density in a hurdle model in order to rank the relevance of the environmental variables in explaining the overall abundance of host snails. Stepwise regression identified that the incidence of host snails increased primarily with pesticide pollution, followed by an increase with the decreasing dominance of potential competitors, with increasing species richness and with a decreasing amount of dissolved oxygen (Table 5.1). The density of host snail populations only decreased with turbidity.

Table 5.1: Minimal adequate model for environmental effects on the abundance of *Schistosoma* host snails. We selected all environmental variables that on their own showed a significant effect on the incidence or on the population density of host snails and combined them in an additive hurdle model. Using backward elimination based on likelihood ratio tests, non-significant environmental variables (species diversity) were removed. Because data have been standardized, importance of the environmental variables on the incidence or on the population density of snails can be compared within each part of the model based on their regression coefficients; coefficients far from zero indicate high (positive or negative) impact. Log-likelihood = -42.43 on 8 df and 40 res. df; McFadden's pseudo-R^2 = 0.28.

Term	**Coefficient**	**Std. error**	***z***	***p***	
Count part (zero-truncated negative binomial with log-link; models population density)					
Intercept	1.92	0.60	4.58	0.001	**
Turbidity	-3.12	1.17	-2.68	0.007	**
ln(Distribution coefficient)	-0.58	0.79	-0.74	0.461	
Zero part (binomial with complementary log-log-link; models incidence)					
Intercept	-5.11	1.90	-2.69	0.007	**
Pesticide pollution	2.73	1.29	2.12	0.034	*
Dominance competitors	-2.30	1.09	-2.12	0.034	*
Species richness	1.81	0.92	1.97	0.048	*
Dissolved oxygen	-0.92	0.45	-2.02	0.044	*

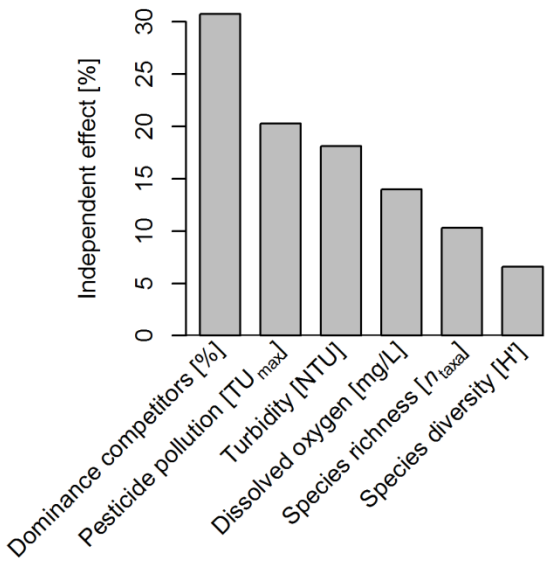

Figure 5.4: Ranking the relevance of environmental variables in driving the abundance of host snails. We combined all environmental variables that on their own showed a significant effect on the incidence or on the population density of host snails and combined them in a hurdle model. The model was subjected to hierarchical partitioning to identify the independent contribution of each environmental variable to the goodness-of-fit (quantified as log-likelihood of the hurdle model). Each step, an environmental variable was either included both in the zero and the count part of the model or excluded completely.

Results from stepwise regression are sensitive to the method used for model selection. Therefore, we additionally applied a multi-model approach by subjecting the full hurdle model to hierarchical partitioning (Figure 5.4). Here the abundance of host snails was most strongly affected by the dominance of potential competitors, followed by the effect of pesticide pollution. In accordance with the results from backward selection, turbidity, species richness and dissolved oxygen showed intermediate effects, and species diversity was least important again.

In the next step, we confirmed the main drivers that potentially underlie the identified environmental variables using a principal component analysis (PCA). The first principal component explained 33.0 % of the total variation among the sites and was associated with typical effects of surface run-off after heavy rainfall (Figure 5.5). It increased with pesticide pollution and turbidity and with decreasing dominance of potential competitor species of the host snails. The second principal component additionally explained 29.4%

of the variation and increased with species richness, with a decreasing amount of dissolved oxygen (indicating increasing oxygen consumption) and with decreasing dominance of potential competitors. Thus, the second principle component likely reflected an increase of host snails with eutrophication that supports more taxa but results in oxygen depletion. Moreover, the second principal component increased with the overall number of macroinvertebrate individuals as an indicator of productivity (n = 48, F = 8.12, res. df = 46, p = 0.007, R^2 = 0.15) which further supported its interpretation as eutrophication. Host snails were only found when both the effects of run-off and eutrophication were high which resulted in a decreased dominance of potential competitors (Figure 5.5).

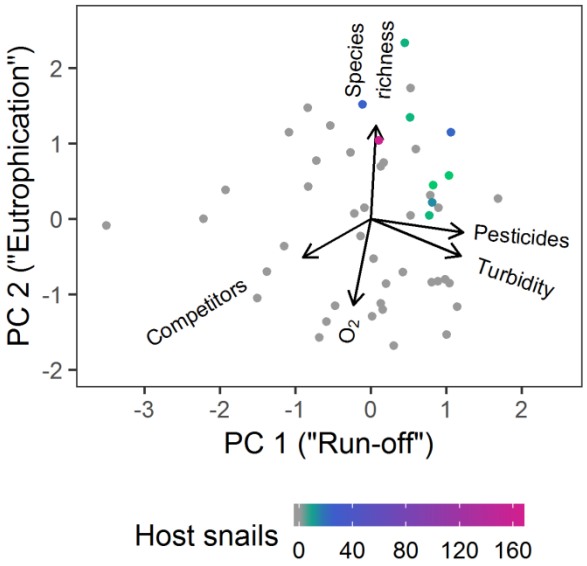

Figure 5.5: Principal component analysis of the environmental variables that drive the abundance of *Schistosoma* hosts. The 1st principal component explains 33.0 % of the variation and is associated with pesticide pollution, turbidity and the dominance of potential competitor species of the host snails. The 2nd principal component explains 29.4 % of the variation and is associated with the species richness, dissolved oxygen and again with the dominance of competitors. Colors indicate the number of host snails collected.

5.3.4 Ecological mechanisms

To better understand the ecological mechanisms through which pesticides affect host snails, we investigated effects of pesticides on the macroinvertebrate community composition.

Pesticide pollution affected neither the dominance of all grazers (host snails and their potential competitors; n = 48, χ^2 = 0.41, res. df = 46, p = 0.520) nor of predators (χ^2 = 0.37, res. df = 46, p = 0.541) or other macroinvertebrates (χ^2 = 2.63, res. df = 46, p = 0.105). Thus, the overall distribution of grazers, predators and other taxa within the community did not significantly change with pesticide pollution (Figure 5.6a). However, within the guild of grazers, pesticide pollution increased the dominance of snails (Figure 5.6b) which were much more tolerant to pesticides than their highly sensitive insect competitors (Figure 4.2). In contrast, pesticide pollution did not affect the composition of predatory macroinvertebrates (PERMANOVA; n = 48, F = 0.78, res. df = 46, p = 0.679) which generally showed intermediate sensitivity to pesticides (Figure 5.2). Additionally, the taxonomic composition of potential predators had no effect on the balance of snails vs. potential competitors: The first principal component of a PCA on the composition of predators did not explain the dominance of snails within the grazers (n = 47, χ^2 = 0.02, res. df = 45, p = 0.890); the same was observed for higher principal components. Therefore, we conclude that pesticides indirectly favored host snails through negative effects on their competitors but not on their predators.

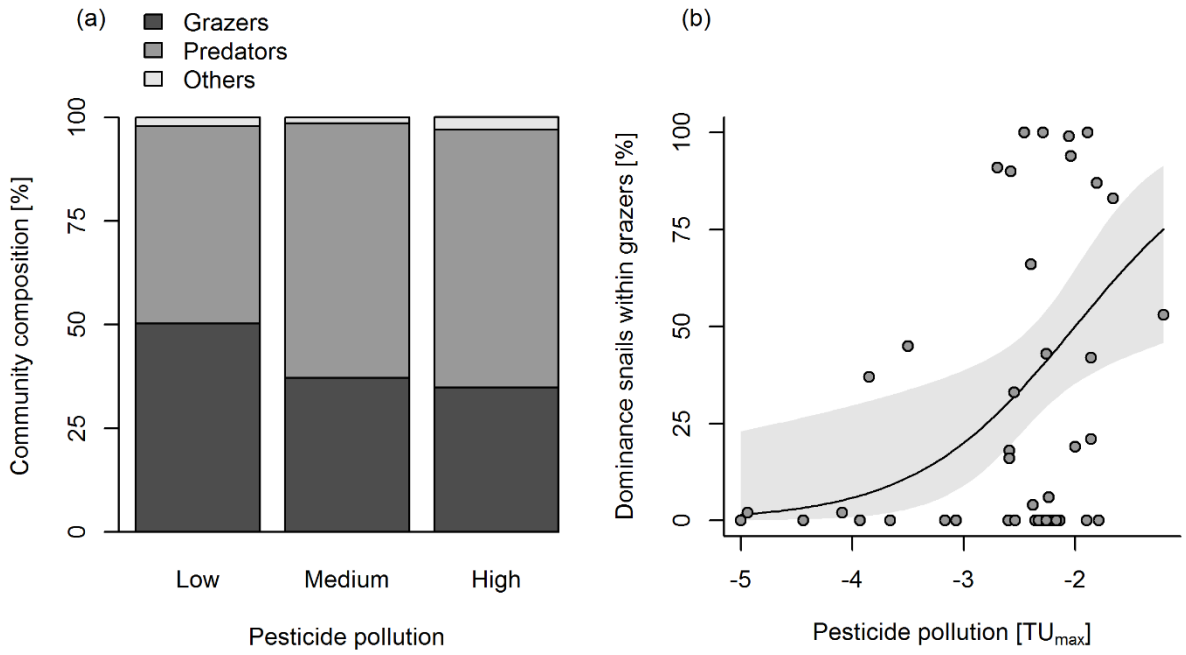

Figure 5.6: Pesticide pollution favors tolerant snails over less tolerant competitors. (a) No significant change in the community composition of grazers, predators and other taxa with pesticide pollution (PERMANOVA; n = 48, F = 0.80, res. df = 46, p = 0.502, R^2 = 0.02). For the graph, the range of TU_{max} values was evenly split in three categories, and for each pollution category the mean proportion of each guild on the macroinvertebrate community is shown. Because some taxa belong to more than one guild, we calculated proportions as the individuals in a guild divided by the summed-up individuals in all guilds (≠ the total individual number) so that the proportions sum up to 1. (b) Within the guild of grazers, the dominance of snails increases with pesticide pollution (n = 47, χ^2 = 13.82, res. df = 45 p < 0.001, McKelvey-Zavoina's pseudo-R^2 = 0.37). One site was omitted because no grazers were found. Quasi-binomial GLM with logitlink function; means ± 95 % confidence intervals are shown.

5.4 Discussion

To our knowledge this is the first field study providing evidence that ecological effects of agricultural pesticides can pose a serious risk to human health. Laboratory studies have already shown how pesticide pollution can increase risk of schistosomiasis (Halstead, 2018). We demonstrate a second mechanism with both field and laboratory data showing how such conditions favor the disease by benefitting the host snail. Host snails of schistosomiasis showed highest tolerance to insecticides amongst all tested macroinvertebrate taxa. Host snails were also solely found in habitats that were at least

moderately affected by pesticides and eutrophication. In these conditions, snails replaced more sensitive potential competitor species. The study thus sheds light on an important risk factor for the transmission of schistosomiasis that has been largely overlooked in previous research on the ecology of host snails and in public health programs.

5.4.1 Pesticide pollution in the study area

Concentrations of the highest exposure (TU_{max}) ranged from <-5 to -1.21, representing a 10^{th} to < $100{,}000^{th}$ of the acute lethal median concentration for standard test organisms. This range of pesticide exposure is comparable to several previous studies conducted in agricultural streams of Europe and Australia (Liess & von der Ohe, 2005; Beketov, 2013; Becker & Liess, 2017).

Given the predominance of subsistence farming in the study area, the results illustrate that pesticide pollution in freshwater is not limited to intensified agriculture. Apparently, the risk of pesticide runoff from agricultural fields is high in the study region. During the rainy seasons in April-June and in October-December, heavy rainfalls erode the cleared land, as indicated by the high turbidity observed in streams during sampling. Because the water bodies are typically not protected by riparian strips from surface run-off, pesticides are washed from flooded agricultural fields into the streams and reservoirs (Liess et al. 1999). In addition, plant protection products are sold in Kenya at comparably low prices without the need for a certificate of competence. This makes pesticides available even for small farmers who may lack enough training and equipment to comply with the proposed environmentally safe use.

The study focused on a broad set of agricultural pesticides typically detected in water. It cannot be excluded that the overall pesticide toxicity might have been even higher due to additional compounds such as pyrethroid insecticides that require different analytical methods. However, pyrethroids typically occur concurrently with the compounds detected and show a similar range of toxicity (Becker & Liess, 2017; Münze et al. 2017). In previous studies (Liess & von der Ohe, 2005; Beketov 2013; Becker & Liess, 2017) pesticides have been sampled during the peak exposure following run-off events after heavy rainfalls.

Event-triggered sampling was not feasible in our study area. However, we sampled during the main rainy season at which we expected the highest pesticide exposure from run-off. Therefore, the TU_{max} values determined to characterize the toxic pressure in this study are comparable to those of previous investigations.

5.4.2 Effects of pesticides on schistosomiasis infection

Host snails of human-pathogenic schistosomes were found exclusively in freshwaters that were at least moderately polluted with pesticides ($TU_{max} \geq -3$) and at least mesotrophic. Physicochemical and land-use parameters had no significant effect on the abundance of host snails, and additional pollutants such as pharmaceuticals, personal care products and industrial chemicals have been shown to cause considerably lower environmental risk than pesticides at the study sites (Kandie et al. 2020). This was observed across various habitats ranging from reservoirs to irrigation channels and streams.

The results support our hypothesis that agricultural pesticide pollution in tropical freshwaters increases the risk of infection with schistosomiasis: Snails as intermediate hosts of human-pathogenic schistosomes are mandatory to close the infection cycle, and humans can become infected only from larval forms (cercariae) released by snails into the water (CDC, 2018). Besides freshwater contamination with infected human excreta and human contact with freshwater infested with cercariae, presence of host snails is a major risk factor for transmission (International Agency for Research on Cancer, 2012). Two human-pathogenic trematodes occur in the study region: *Schistosoma mansoni* parasitizes snails of the genus *Biomphalaria sp.* and causes intestinal schistosomiasis, whereas *S. haematobium* parasitizes certain snails of the *Bulinus africanus* complex and causes urogenital schistosomiasis (King et al. 2015). Access to sanitation is often insufficient in the densely populated study area (Odhiambo et al. 2014; Chadeka et al. 2019), and therefore many people are exposed to non-sanitized freshwater during activities such as bathing (particularly school children), washing and field work (International Agency for Research on Cancer, 2012). This is especially pronounced at the shore of Lake Victoria which suffers from a high disease burden (Odhiambo et al. 2014; Mwandawiro et al. 2019). When traveling, schistosomiasis transmission may be

imported to inland areas if host snails are present (Bruun & Aagaard-Hansen, 2008). In these conditions, we expect that the risk of infection is influenced by the occurrence of intermediate host snails.

Our finding of pesticide-induced shifts in the community composition towards more snails are in line with various studies that reported significant ecological effects of pesticides even in streams with low concentrations resembling a TU_{max} of -2 to -4. These effects include changes in the macroinvertebrate community composition towards more tolerant taxa (Liess & von der Ohe, 2005), reduced leaf litter breakdown (Münze et al. 2017) and the development of pesticide resistance (Becker & Liess, 2017). The results, however, contrast the common perception of the environmental risk of pesticides. For example, according to the European framework for the registration of plant protection products, environmental concentrations <1 % of the acute LC_{50} of the most sensitive standard reference organism are generally considered safe (EFSA, 2013); this would resemble a TU_{max} up to -2. No such threshold concentrations have been defined in Kenya, but pesticides need to be considered environmentally safe by the national Pest Control Product Board before registration (PCPB, 2006). The present study shows that pesticides nevertheless affect the community composition of freshwater macroinvertebrates and that these ecological effects can have serious consequences for human health, hence the need for revision of acceptable regulatory concentrations.

5.4.3 Ecological mechanisms supporting host snails

Given the very high pesticide tolerance of snails compared to the pesticide concentrations measured in the environment, a direct effect of pesticides on the observed host snails appears unlikely. Instead, pesticides may indirectly favor host snails through adverse effects on their antagonistic species such as predators and competitors. A recent mesocosm study showed that herbicides and insecticides can favor host snails of human-pathogenic schistosomes through effects on predators and the support of periphyton as food source for snails (through effects on antagonistic planktonic algae) (Halstead, 2018). However, the study did not collect field data, nor did it discuss potential effects on competitor species (Halstead, 2018). The observed effects on predators in the mesocosm

study contrast our results, probably because pesticide concentrations were 3 to 4 orders of magnitude higher than those observed at our study sites. Such concentrations may affect even taxa such as predators that showed generally intermediate pesticide tolerance in our tests and in previous studies (Wogram& Liess, 2001). We observed that with increasing pesticide pollution snails replaced grazing insects that are known to compete with the more tolerant snails (Yeung & Dudgeon, 2013) and are generally highly sensitive to insecticides and some fungicides (Wogram& Liess, 2001; Beketov & Liess, 2008). In fact, pesticides have been shown to affect the survival and emergence of aquatic insects at concentrations down to 0.005 % (4 orders of magnitude below) of their acute LC_{50} (Liess & Schulz, 1996; Beketov & Liess, 2005). Moreover, the calculation of toxic units for additional trophic levels revealed highest risk for insects and crustaceans compared to algae and vertebrates in our study sites (Kandie et al. 2020). Therefore, we focused on the toxicity of pesticides to invertebrates. Our results indicate that in the field, pesticides favor snails mainly through negative effects on more sensitive competitors. Pesticide pollution was closely related with turbidity, as both factors increase with rainfall induced flooding (Liess & Schulz 1999; Shen 2018). Nevertheless, these factors showed contrasting effects on the abundance of host snails (increased incidence vs. decreased population density). We hypothesize that flooding results in a short-term reduction of host snail populations due to increased flow velocity (Woolhouse & Chandiwana, 1990), whereas pesticide exposure in the long-term facilitates the establishment of tolerant taxa such as snails (Liess & von der Ohe, 2005; Liess et al. 2013). This may explain the generally low population densities of host snails observed during the rainy season and the more obvious link of pesticide pollution with incidence than with population density.

5.5 Conclusions

The present case study illustrates that serious consequences of agricultural pesticide pollution arises for public health, even at concentrations considered safe within the traditional risk assessment. Given that pesticide application, particularly in developing countries, is predicted to increase 2- to 5-fold from 2000 to 2050 to meet the food demand of a growing human population, freshwater pollution and its ecological effects will

aggravate (Alexandratos & Bruinsma, 2012; Tilman et al. 2001). The results underline the urgent need for reassessing the environmental risk of low pesticide concentrations and for integrated disease management that includes a focus on the regulation and management of pesticides in areas where schistosomiasis is endemic or might be introduced due to potentially favorable ecological conditions.

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Author contributions:

ML conceived the approach; ML, JB, UF, WB, BT, EA, HH, FM, AA and FK designed the research; AG, FK, LM, JB and JA conducted the research; JB and AG, analysed and interpreted the data; JB and AG, drafted the initial version; JB, AG, FK, ML, UF, WB, BT, EA, HH contributed to the final version - all approved the final version of the publication.

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Chapter 6

General conclusions and future outlook on CEC research

General conclusions and future outlook on CEC research

The contents of this section are drawn from the synthesis of the results discussed in detail in Chapters 3-5. Within this chapter, recommendations for future research are proposed.

Through this dissertation, attempts to narrow the knowledge gap on occurrence of CECs in African aquatic ecosystems have been made. The dissertation shows evidence of contamination of surface water systems in rural western Kenya with a cocktail of CECs. It also shows the risk of aquatic organisms especially crustaceans when exposed to these pollutants. This dissertation also shows evidences of pesticides increasing schistosomiasis transmission by negatively affecting sensitive competitors and having no impact on the tolerant host snails in water systems. It is important that such data exist to prevent adverse effects on organisms due to exposure and also help to formulate policies and guidelines on chemical usage.

Human activities including land clearing for cultivation, cultivation at the river banks, poor pesticide practice in handling and disposal, open defecation, lack of domestic sewer systems, and open dumpsites mainly contribute to the CEC occurrence in the environment. All these activities are driven by the ever growing population (Bhatt et al., 2019; Loha et al., 2018; Weber et al., 2014; WHO & UNICEF, 2015). The rivers within the study area flow into Lake Victoria, world's second largest freshwater lake, spreading across Kenya, Tanzania and Uganda. Poor agricultural and waste management releases chemicals into highland rivers which drain into the lake, thereby depositing these contaminants into the lake. Additionally, open defecation greatly contributes to release of PPCPs and also the schistome eggs into the environment. This poses great risk on both the surface water systems in the highlands and the lake to which they drain into. Dedicated efforts are needed for comprehensive monitoring not only for screening but also for determination of cause–effect relationships (Tadeo, 2008) between CECs and environmental risk in the compartment.

Environmental occurrence and ecotoxicological data on African species is still insufficient despite the growing research in the region. Currently, comprehensive risk assessment on aquatic organisms is only possible with the data rich European species. This data

limitation is mainly driven by financial and technological challenges faced by African scientists in order to undertake such comprehensive research within Africa. Regular monitoring of CECs require advanced technology and financial support. Therefore, more international collaborations for example, North-South collaborations should be encouraged to facilitate the generation of chemical and risk data from Africa. It is important that more efforts are invested on toxicity tests on African native species.

Additionally, future research should focus on:

I. Risk to public health. Considering most of the rural population in Africa use surface water or ground water for domestic use without prior treatment. Antibiotics were found in the samples in water systems which are used as a source of drinking water. Since antibiotic resistance is getting much attention due to the risk (Kairigo et al. , 2020b), further research should focus on identification of resistant bacteria, antibiotic resistance genes and any gene transfer through water consumption.
II. Another public health problem is the high prevalence of schistosomiasis in sub-Saharan Africa including Kenya. Our study showed that pesticide pollution contributed to the transmission of the disease. Future research should be done to understand the disease ecology of schistosomiasis and transmission competence of the miracidia after exposure to pesticides. This would help to reduce morbidity and mortalities associated with this disease especially in endemic areas such as in Africa.
III. Source specific contamination patterns should be investigated in African setting. More data should be acquired based on input sources of CECs including dumpsites, latrines, industrial effluent and surface run off. In addition, integrative sampling techniques such as use of large volume solid phase extraction (LVSPE) could be applied to have an alternative approach to the grab samples. Integrative samples allow a more continuous monitoring of CECs which also capture short-term (pulse) or long term changes in pollutant concentrations.
IV. Most countries in Africa focus on conventional pollutants in water with only a few studies existing on CECs. Additionally, less data exist on multi-compartment analysis of CECs. Future research should focus on the fate and behavior of CECs

in the environment. Considering the different seasonal changes and climatic conditions in Africa, research should evaluate influence of these factors on the distribution of CECs when released into the water system.

v. Target and suspect screening techniques were applied in this study. Although a large compound list from Europe monitoring and those registered for use in Kenya were included, this approach limited the scope of compounds only to the available standards for these compounds. Further research should include non-target screening (Chiaia-Hernandez et al., 2014; Gago-Ferrero et al., 2015; Schymanski et al., 2015) to be able to identify and quantify unknown compounds which may be present in the samples. A large number of CECs for environmental monitoring is relevant especially in the African continent where analytical costs need to be minimized while expanding the research scope of CECs in the environment.

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Appendix A

Table SI-1: List of target compounds analyzed in the samples

Compound	Formula	ion mode 1	main adduct	Exact mass	Expected ion
Pesticides, Biocides and their transformational products					
1-(3,4-Dichlorophenyl)urea	C7H6Cl2N2O	ESI+	M+H	203.9857	204.9930
1,2-Benzisothiazolinone	C7H5NOS	ESI+	M+H	151.0092	152.0165
2-Isopropyl-6-methyl-pyrimidin-4-ol	C8H12N2O	ESI+	M+H	152.0950	153.1022
2-Octyl-4-isothiazolin-3-one	C11H19NOS	ESI+	M+H	213.1187	214.1260
3,4-Dichlorophenylurea	C7H6Cl2N2O	ESI+	M+H	203.9857	204.9930
3-Iodopropynyl butylcarbamate	C8H12INO2	ESI+	M+H	280.9913	281.9985
Benzethonium	C27H42NO2	ESI+	M+	412.3210	412.3210
Benzyldimethyldodecylammonium	C21H38N	ESI+	M+	304.2999	304.2999
Benzyldimethylhexadecylammonium	C25H46N	ESI+	M+	360.3625	360.3625
Bromochlorophen	C13H8Br2Cl2O2	ESI-	M-H	423.8268	422.8195
Butylparaben	C11H14O3	ESI-	M-H	194.0943	193.0870
Chlorophene	C13H11ClO	ESI-	M-H	218.0498	217.0426
DCOIT	C11H17Cl2NOS	ESI+	M+H	281.0408	282.0481
Diazinon	C12H21N2O3PS	ESI+	M+H	304.1011	305.1083
Dichlorophene	C13H10Cl2O2	ESI-	M-H	268.0058	266.9985
Didecyldimethylammonium	C22H48N	ESI+	M+	326.3781	326.3781
Diuron	C9H10Cl2N2O	ESI+	M+H	232.0170	233.0243
Ethylparaben	C9H10O3	ESI-	M-H	166.0630	165.0557
Fipronil	C12H4Cl2F6N4OS	ESI-	M-H	435.9387	434.9314
Fipronil desulfinyl	C12H4Cl2F6N4	ESI-	M-H	387.9717	386.9644
Fipronil sulfide	C12H4Cl2F6N4S	ESI-	M-H	419.9438	418.9365
Fipronil sulfone	C12H4Cl2F6N4O2S	ESI-	M-H	451.9336	450.9263
Hexadecylpyridinium	C21H38N	ESI+	M+	304.2999	304.2999
Hexadecyltrimethylammonium	C19H42N	ESI+	M+	284.3312	284.3312
Imazalil	C14H14Cl2N2O	ESI+	M+H	296.0483	297.0556
Irgarol	C11H19N5S	ESI+	M+H	253.1361	254.1434
Methylchloroisothiazolinone	C4H4ClNOS	ESI+	M+H	148.9702	149.9775
Methylparaben	C8H8O3	ESI-	M-H	152.0473	151.0401
Propylparaben	C10H12O3	ESI-	M-H	180.0786	179.0714
Terbutryn	C10H19N5S	ESI+	M+H	241.1361	242.1434
Thiabendazole	C10H7N3S	ESI+	M+H	201.0361	202.0433
Triclocarban	C13H9Cl3N2O	ESI-	M-H	313.9780	314.9853
Triclosan	C12H7Cl3O2	ESI-	M-H	287.9512	286.9439
Denatonium	C21H29N2O	ESI+	M+	325.2274	325.2274
2,4-Dichlorophenoxyacetic acid	C8H6Cl2O3	ESI-	M-H	219.9694	218.9621
2,6-Dichlorobenzamide	C7H5Cl2NO	ESI+	M+H	188.9748	189.9821
2,6-Xylidine	C8H11N	ESI+	M+H	121.0891	122.0964

2-Aminobenzimidazole	C7H7N3	ESI+	M+H	133.0640	134.0713
3,5,6-Trichloro-2-pyridinol	C5H2Cl3NO	ESI-	M-H	196.9202	195.9129
4-Isopropylaniline	C9H13N	ESI+	M+H	135.1048	136.1121
Abamectin	C48H72O14	ESI+	M+Na	872.4922	895.4820
Acetamiprid	C10H11ClN4	ESI+	M+H	222.0672	223.0745
Acetochlor	C14H20ClNO2	ESI+	M+H	269.1183	270.1255
Ametryn	C9H17N5S	ESI+	M+H	227.1205	228.1277
Atrazine	C8H14ClN5	ESI+	M+H	215.0938	216.1010
Azinphos methyl	C10H12N3O3PS2	ESI+	M+H	317.0058	318.0130
Azoxystrobin	C22H17N3O5	ESI+	M+H	403.1168	404.1241
Benalaxyl	C20H23NO3	ESI+	M+H	325.1678	326.1751
Bendiocarb	C11H13NO4	ESI+	M+H	223.0845	224.0917
Bentazone	C10H12N2O3S	ESI-	M-H	240.0569	239.0496
Bifenox free acid	C13H7Cl2NO5	ESI-	M-H	326.9701	325.9629
Boscalid	C18H12Cl2N2O	ESI+	M+H	342.0327	343.0399
Bromoxynil	C7H3Br2NO	ESI-	M-H	274.8581	273.8509
Bupirimate	C13H24N4O3S	ESI+	M+H	316.1569	317.1642
Carbaryl	C12H11NO2	ESI+	M+H	201.0790	202.0863
Carbendazim	C9H9N3O2	ESI+	M+H	191.0695	192.0768
Chlorfenvinphos	C12H14Cl3O4P	ESI+	M+H	357.9695	358.9768
Chloridazon	C10H8ClN3O	ESI+	M+H	221.0356	222.0429
Chlormequat	C5H13ClN	ESI+	M+	122.0731	122.0731
Chlorotoluron	C10H13ClN2O	ESI+	M+H	212.0716	213.0789
Chloroxuron	C15H15ClN2O2	ESI+	M+H	290.0822	291.0895
Chlorpropham	C10H12ClNO2	ESI+	M+H	213.0557	214.0629
Clomazone	C12H14ClNO2	ESI+	M+H	239.0713	240.0786
Clothianidin	C6H8ClN5O2S	ESI+	M+H	249.0087	250.0160
Cyproconazole	C15H18ClN3O	ESI+	M+H	291.1138	292.1211
Cyromazine	C6H10N6	ESI+	M+H	166.0967	167.1040
Dichlorprop	C9H8Cl2O3	ESI-	M-H	233.9850	232.9778
Dichlorvos	C4H7Cl2O4P	ESI+	M+H	219.9459	220.9532
Difenoconazole	C19H17Cl2N3O3	ESI+	M+H	405.0647	406.0720
Diflubenzuron	C14H9ClF2N2O2	ESI+	M+H	310.0321	311.0393
Diflufenican	C19H11F5N2O2	ESI+	M+H	394.0741	395.0813
Dimethachlor	C13H18ClNO2	ESI+	M+H	255.1026	256.1099
Dimethachlor ESA	C13H19NO5S	ESI-	M-H	301.0984	300.0911
Dimethachlor OA	C13H17NO4	ESI-	M-H	251.1158	250.1085
Dimethenamid ESA	C12H18ClNO2S	ESI+	M+H	275.0747	276.0820
Dimethoate	C5H12NO3PS2	ESI+	M+H	228.9996	230.0069
Dinoseb	C10H12N2O5	ESI-	M-H	240.0746	239.0673
Dodemorph	C18H35NO	ESI+	M+H	281.2719	282.2791
Epoxiconazole	C17H13ClFN3O	ESI+	M+H	329.0731	330.0804
Ethion	C9H22O4P2S4	ESI+	M+H	383.9876	384.9949

Ethofumesate	C13H18O5S	ESI+	M+H	286.0875	287.0948
Ethyl azinphos	C12H16N3O3PS2	ESI+	M+H	345.0371	346.0443
Ethylenethiourea	C3H6N2S	ESI+	M+H	102.0252	103.0324
Fenoxycarb	C17H19NO4	ESI+	M+H	301.1314	302.1387
Fenpropidin	C19H31N	ESI+	M+H	273.2457	274.2529
Fenpropimorph	C20H33NO	ESI+	M+H	303.2562	304.2635
Fenthion	C10H15O3PS2	ESI+	M+H	278.0200	279.0273
Fenuron	C9H12N2O	ESI+	M+H	164.0950	165.1022
Flufenacet	C14H13F4N3O2S	ESI+	M+H	363.0665	364.0737
Flufenoxuron	C21H11ClF6N2O3	ESI+	M+H	488.0362	489.0435
Fluoxastrobin	C21H16ClFN4O5	ESI+	M+H	458.0793	459.0866
Flurtamone	C18H14F3NO2	ESI+	M+H	333.0977	334.1049
Flusilazole	C16H15F2N3Si	ESI+	M+H	315.1003	316.1076
Hexazinone	C12H20N4O2	ESI+	M+H	252.1586	253.1659
Imidacloprid	C9H10ClN5O2	ESI+	M+H	255.0523	256.0596
Imidacloprid-urea	C9H10ClN3O	ESI+	M+H	211.0512	212.0585
Isoproturon	C12H18N2O	ESI+	M+H	206.1419	207.1492
Lenacil	C13H18N2O2	ESI+	M+H	234.1368	235.1441
Linuron	C9H10Cl2N2O2	ESI+	M+H	248.0119	249.0192
MCPA	C9H9ClO3	ESI-	M-H	200.0240	199.0167
Mecoprop	C10H11ClO3	ESI-	M-H	214.0397	213.0324
Mepiquat	C7H16N	ESI+	M+	114.1277	114.1277
Metalaxyl	C15H21NO4	ESI+	M+H	279.1471	280.1543
Metamitron	C10H10N4O	ESI+	M+H	202.0855	203.0927
Metazachlor	C14H16ClN3O	ESI+	M+H	277.0982	278.1055
Metazachlor ESA	C14H17N3O4S	ESI-	M-H	323.0940	322.0867
Metazachlor OA	C14H15N3O3	ESI-	M-H	273.1113	272.1041
Metconazole	C17H22ClN3O	ESI+	M+H	319.1451	320.1524
Methiocarb	C11H15NO2S	ESI+	M+H	225.0823	226.0896
Methyldesphenyl chloridazon	C4H4ClN3O	ESI+	M+H	145.0043	146.0116
Metolachlor	C15H22ClNO2	ESI+	M+H	283.1339	284.1412
Metolachlor ESA	C15H23NO5S	ESI-	M-H	329.1297	328.1224
Metolachlor OA	C15H21NO4	ESI-	M-H	279.1471	278.1398
Metribuzin	C8H14N4OS	ESI+	M+H	214.0888	215.0961
Myclobutanil	C15H17ClN4	ESI+	M+H	288.1142	289.1215
N,N-Dimethylsulfamide	C2H8N2O2S	ESI+	M+H	124.0306	125.0379
Oryzalin	C12H18N4O6S	ESI+	M+H	346.0947	347.1020
Oxadiazone	C15H18Cl2N2O3	ESI+	M+H	344.0694	345.0767
Parathion-methyl	C8H10NO5PS	ESI+	M+H	263.0017	264.009
Pendimethalin	C13H19N3O4	ESI+	M+H	281.1376	282.1448
Pethoxamid	C16H22ClNO2	ESI+	M+H	295.1339	296.1412
Phthalamic acid	C8H7NO3	ESI-	M-H	165.0426	164.0353
Picolinafen	C19H12F4N2O2	ESI+	M+H	376.0835	377.0908

Picoxystrobin	C18H16F3NO4	ESI+	M+H	367.1031	368.1104
Piperonyl butoxide	C19H30O5	ESI+	M+NH4	338.2093	356.2431
Pirimicarb	C11H18N4O2	ESI+	M+H	238.1430	239.1503
Pirimiphos-methyl	C11H20N3O3PS	ESI+	M+H	305.0963	306.1036
Prochloraz	C15H16Cl3N3O2	ESI+	M+H	375.0308	376.0381
Promethazin	C17H20N2S	ESI+	M+H	284.1347	285.1420
Propachlor	C11H14ClNO	ESI+	M+H	211.0764	212.0837
Propamocarb	C9H20N2O2	ESI+	M+H	188.1525	189.1598
Propanil	C9H9Cl2NO	ESI+	M+H	217.0061	218.0134
Propiconazole	C15H17Cl2N3O2	ESI+	M+H	341.0698	342.0771
Propoxycarbazone	C15H18N4O7S	ESI+	M+H	398.0896	399.0969
Propyzamide	C12H11Cl2NO	ESI+	M+H	255.0218	256.0290
Prosulfocarb	C14H21NOS	ESI+	M+H	251.1344	252.1417
Prothioconazole-desthio	C14H15Cl2N3O	ESI+	M+H	311.0592	312.0665
Pyraclostrobin	C19H18ClN3O4	ESI+	M+H	387.0986	388.1059
Pyrazophos	C14H20N3O5PS	ESI+	M+H	373.0861	374.0934
Quinmerac	C11H8ClNO2	ESI+	M+H	221.0244	222.0316
Quinoxyfen	C15H8Cl2FNO	ESI+	M+H	306.9967	308.0040
Simazine	C7H12ClN5	ESI+	M+H	201.0781	202.0854
Simetryn	C8H15N5S	ESI+	M+H	213.1048	214.1121
Spiroxamine	C18H35NO2	ESI+	M+H	297.2668	298.2741
Sulcotrione	C14H13ClO5S	ESI+	M+H	328.0172	329.0245
Tebuconazole	C16H22ClN3O	ESI+	M+H	307.1451	308.1524
Terbuthylazine	C9H16ClN5	ESI+	M+H	229.1094	230.1167
Terbuthylazine-2-hydroxy	C9H17N5O	ESI+	M+H	211.1433	212.1506
Thiacloprid	C10H9ClN4S	ESI+	M+H	252.0236	253.0309
Thiacloprid amide	C10H11ClN4OS	ESI+	M+H	270.0342	271.0415
Thiamethoxam	C8H10ClN5O3S	ESI+	M+H	291.0193	292.0266
Triadimenol	C14H18ClN3O2	ESI+	M+H	295.1088	296.1160
Triallate	C10H16Cl3NOS	ESI+	M+H	303.0018	304.0091
Trifloxystrobin	C20H19F3N2O4	ESI+	M+H	408.1297	409.1370
DEET	C12H17NO	ESI+	M+H	191.1310	192.1383
2-Hydroxyatrazine	C8H15N5O	ESI+	M+H	197.1277	198.1349
Chlorothalonil-4-hydroxy	C8HCl3N2O	ESI-	M-H	245.9154	244.9082
Desethylatrazine	C6H10ClN5	ESI+	M+H	187.0625	188.0697
Desethylterbutylazine	C7H12ClN5	ESI+	M+H	201.0781	202.0854
Desisopropylatrazine	C5H8ClN5	ESI+	M+H	173.0468	174.0541
Imidacloprid-guanidine	C9H11ClN4	ESI+	M+H	210.0672	211.0745
Icaridin	C12H23NO3	ESI+	M+H	229.1678	230.1751
Industrial chemicals					
1H-Benzotriazole	C6H5N3	ESI+	M+H	119.0483	120.0556
4-Hydroxybenzotriazole	C6H5N3O	ESI+	M+H	135.0433	136.0505
5-Methyl-1H-benzotriazole	C7H7N3	ESI+	M+H	133.0640	134.0713

7-Amino-4-methylcoumarin	C10H9NO2	ESI+	M+H	175.0633	176.0706
7-Diethylamino-4-methylcoumarin	C14H17NO2	ESI+	M+H	231.1259	232.1332
Tris(2-chloroethyl)phosphate	C6H12Cl3O4P	ESI+	M+H	283.9539	284.9612
Daidzein	C15H10O4	ESI+	M+H	254.0579	255.0652
Triethylcitrate	C12H20O7	ESI+	M+H	276.1209	277.1282
(4-Sulfophenyl)acetic acid	C8H8O5S	ESI-	M-H	216.0092	215.0020
2,3-Epoxypropyltrimethylammonium	C6H14NO	ESI+	M+	116.1070	116.1070
2,4-Diaminobenzenesulfonic acid	C6H8N2O3S	ESI-	M-H	188.0256	187.0183
2,4-Dichlorophenol	C6H4Cl2O	ESI-	M-H	161.9639	160.9566
2,4-Dinitrophenol	C6H4N2O5	ESI-	M-H	184.0120	183.0047
2,7-Naphthalenedisulfonic acid	C10H8O6S2	ESI-	M-H	287.9762	286.9690
2-Isopropylthioxanthone	C16H14OS	ESI+	M+H	254.0765	255.0838
2-Methylbenzothiazole	C8H7NS	ESI+	M+H	149.0299	150.0372
2-Naphthalene sulfonic acid	C10H8O3S	ESI-	M-H	208.0194	207.0121
3,4,5-Trichlorophenol	C6H3Cl3O	ESI-	M-H	195.9249	194.9177
4-(Dimethylamino)pyridine	C7H10N2	ESI+	M+H	122.0844	123.0917
4'-Aminoacetanilide	C8H10N2O	ESI+	M+H	150.0793	151.0866
4-Aminobenzamide	C7H8N2O	ESI+	M+H	136.0637	137.0709
4-Amino-N,N-dimethylbenzenesulfonamide	C8H12N2O2S	ESI+	M+H	200.0619	201.0692
4-Bromophenol	C6H5BrO	ESI-	M-H	171.9524	170.9451
4-Chlorophenol	C6H5ClO	ESI-	M-H	128.0029	126.9956
4-Nitrophenol	C6H5NO3	ESI-	M-H	139.0269	138.0197
Benzenesulfonic acid	C6H6O3S	ESI-	M-H	158.0038	156.9965
Bisphenol S	C12H10O4S	ESI-	M-H	250.0300	249.0227
Diglyme	C6H14O3	ESI+	M+H	134.0943	135.1016
Ethyl 4-(dimethylamino)benzoate	C11H15NO2	ESI+	M+H	193.1103	194.1176
Hexa(methoxymethyl)melamine	C15H30N6O6	ESI+	M+H	390.2227	391.2300
Iminostilbene	C14H11N	ESI+	M+H	193.0891	194.0964
Isophorone diamine	C10H22N2	ESI+	M+H	170.1783	171.1856
Lauryl diethanolamide	C16H33NO3	ESI+	M+H	287.2460	288.2533
Melamine	C3H6N6	ESI+	M+H	126.0654	127.0727
m-Xylene-4-sulfonic acid	C8H10O3S	ESI-	M-H	186.0351	185.0278
N-Butylbenzenesulfonamide	C10H15NO2S	ESI+	M+H	213.0823	214.0896
N-Ethyl-o-toluenesulfonamide	C9H13NO2S	ESI+	M+H	199.0667	200.0740
Perfluorobutanoic acid	C4HF7O2	ESI-	M-H	213.9865	212.9792
Perfluorodecanoic acid	C10HF19O2	ESI-	M-H	513.9673	512.9600
Perfluoroheptanoic acid	C7HF13O2	ESI-	M-H	363.9769	362.9696
Perfluorohexanoic acid	C6HF11O2	ESI-	M-H	313.9801	312.9728
Perfluorooctanesulfonamide	C8H2F17NO2S	ESI-	M-H	498.9535	497.9462
Perfluorooctanesulfonic acid	C8HF17O3S	ESI-	M-H	499.9375	498.9302
Perfluorooctanoic acid	C8HF15O2	ESI-	M-H	413.9737	412.9664

Perfluorotetradecanoic acid	C14HF27O2	ESI-	M-H	713.9545	712.9473
p-Toluenesulfonamide	C7H9NO2S	ESI+	M+NH4	171.0354	189.0692
Tetrachlorosalicylanilide	C13H7Cl4NO2	ESI-	M-H	348.9231	347.9158
Tetraglyme	C10H22O5	ESI+	M+NH4	240.1804	222.1467
TMDD	C14H26O2	ESI+	M+NH4	226.1933	244.2271
Tri(butoxyethyl)phosphate	C18H39O7P	ESI+	M+H	398.2433	399.2506
Triglyme	C8H18O4	ESI+	M+NH4	178.1205	196.1542
Triphenylphosphate	C18H15O4P	ESI+	M+H	326.0708	327.0781
Bis(2-ethylhexyl)phosphate	C16H35O4P	ESI+	M+H	322.2273	323.2346
Di-n-butyl phosphate	C8H19O4P	ESI+	M+H	210.1021	211.1094
Diphenylphosphate	C12H11O4P	ESI+	M+H	250.0395	251.0468
TDCPP	C9H15Cl6O4P	ESI+	M+H	427.8839	428.8912
Tetrabromobisphenol A	C15H12Br4O2	ESI-	M-H	539.7571	538.7498
Tricresylphosphate	C21H21O4P	ESI+	M+H	368.1177	369.1250
Triethylphosphate	C6H15O4P	ESI+	M+H	182.0708	183.0781
Tri-isobutylphosphate	C12H27O4P	ESI+	M+H	266.1647	267.1720
Tris(1-chloro-2-propyl)phosphate	C9H18Cl3O4P	ESI+	M+H	326.0008	327.0081
2(4-morpholinyl)benzothiazole	C11H12N2OS	ESI+	M+H	220.0670	221.0743
2-(Methylthio)benzothiazole	C8H7NS2	ESI+	M+H	181.0020	182.0093
2-Benzothiazolesulfonic acid	C7H5NO3S2	ESI-	M-H	214.9711	213.9638
2-Hydroxybenzothiazole	C7H5NOS	ESI-	M-H	151.0092	150.0019
2-Morpholinothiobenzothiazole	C11H12N2OS2	ESI+	M+H	252.0391	253.0464
Benzothiazole	C7H5NS	ESI+	M+H	135.0143	136.0215
N-Cyclohexyl-2-benzothiazole-amine	C13H16N2S	ESI+	M+H	232.1034	233.1107
N-Cyclohexyl-2-benzothiazole-sulfenamide	C13H16N2S2	ESI+	M+H	264.0755	265.0828
Caffeine	C8H10N4O2	ESI+	M+H	194.0804	195.0877
Theophyllin	C7H8N4O2	ESI+	M+H	180.0647	181.0720
2,4-Dichlorobenzoic acid	C7H4Cl2O2	ESI-	M-H	189.9588	188.9516
Amidosulfobetaine-14	C22H47N2O4S	ESI+	M+	435.3251	435.3251
Dioctylsulfosuccinate	C20H38O7S	ESI-	M-H	422.2338	421.2265
Dodecyl sulfate	C12H26O4S	ESI-	M-H	266.1552	265.1479
Dodecylbenzenesulfonic acid	C18H30O3S	ESI-	M-H	326.1916	325.1843
Lauramidopropylbetaine	C19H39N2O3	ESI+	M+	343.2955	343.2955
Lauric isopropanolamide	C15H31NO2	ESI+	M+H	257.2355	258.2428
N,N-Dimethyldodecylamine N-oxide	C14H31NO	ESI+	M+H	229.2406	230.2478
Tetradecylsulfate	C14H30O4S	ESI-	M-H	294.1865	293.1792
Trimethyloctylammonium	C11H26N	ESI+	M+	172.2060	172.2060
Acesulfame	C4H5NO4S	ESI-	M-H	162.9939	161.9867
Cyclamate	C6H13NO3S	ESI-	M-H	179.0616	178.0543
Saccharin	C7H5NO3S	ESI-	M-H	182.9990	181.9917
Sucralose	C12H19Cl3O8	ESI-	M-H+FA	396.0146	441.0123
6:2 fluorotelomer sulfonic acid	C8H5F13O3S	ESI-	M-H	427.9752	426.9679

Triphenylphosphine oxide	C18H15OP	ESI+	M+H	278.0861	279.0933
Human metabolites, steroids and hormones					
Cotinine	C10H12N2O	ESI+	M+H	176.0950	177.1022
Creatinine	C4H7N3O	ESI+	M+H	113.0589	114.0662
Genistein	C15H10O5	ESI+	M+H	270.0528	271.0601
Cortisone	C21H28O5	ESI+	M+H	360.1937	361.2010
Progesterone	C21H30O2	ESI+	M+H	314.2246	315.2319
Testosterone	C19H28O2	ESI+	M+H	288.2089	289.2162
4-Androstene-3,17-dione	C19H26O2	ESI+	M+H	286.1933	287.2006
Androsterone	C19H30O2	ESI+	M+H	290.2246	291.2319
Hydrocortisone	C21H30O5	ESI+	M+H	362.2093	363.2166
Pharmaceuticals, Personal Care products and their transformational products					
2-(2-(Chlorophenyl)amino)benzaldehyde	C13H10ClNO	ESI+	M+H	231.0451	232.0524
2-Thiouracil	C4H4N2OS	ESI+	M+H	128.0044	129.0117
4-Aminoantipyrine	C11H13N3O	ESI+	M+H	203.1059	204.1131
4-Fluorobenzoylpropionic acid	C10H9FO3	ESI-	M-H	196.0536	195.0463
4-Hydroxytamoxifen	C26H29NO2	ESI+	M+H	387.2198	388.2271
5-Fluorouracil	C4H3FN2O2	ESI-	M-H	130.0179	129.0106
6-Mercaptopurine	C5H4N4S	ESI-	M-H	152.0157	151.0084
7-Hydroxymethotrexate	C20H22N8O6	ESI+	M+H	470.1662	471.1735
Acetaminophen	C8H9NO2	ESI+	M+H	151.0633	152.0706
Acyclovir	C8H11N5O3	ESI-	M-H	225.0862	224.0789
Albendazole	C12H15N3O2S	ESI+	M+H	265.0885	266.0958
Amantadine	C10H17N	ESI+	M+H	151.1361	152.1434
Ambroxol	C13H18Br2N2O	ESI+	M+H	375.9786	376.9859
Amiodarone	C25H29I2NO3	ESI+	M+H	645.0237	646.0310
Amitriptyline	C20H23N	ESI+	M+H	277.1830	278.1903
Amoxicillin	C16H19N3O5S	ESI+	M+H	365.1045	366.1118
Anastrozole	C17H19N5	ESI+	M+H	293.1640	294.1713
Atenolol	C14H22N2O3	ESI+	M+H	266.1630	267.1703
Atorvastatin	C33H35FN2O5	ESI-	M-H	558.2530	557.2457
Azelastine	C22H24ClN3O	ESI+	M+H	381.1608	382.1681
Azithromycin	C38H72N2O12	ESI+	M+H	748.5085	749.5158
Benzocain	C9H11NO2	ESI+	M+H	165.0790	166.0863
Bethamethasone	C22H29FO5	ESI+	M+H	392.1999	393.2072
Bezafibrate	C19H20ClNO4	ESI-	M-H	361.1081	360.1008
Bicalutamide	C18H14F4N2O4S	ESI-	M-H	430.0610	429.0538
Bifonazol	C22H18N2	ESI+	M+H	310.1470	311.1543
Bisoprolol	C18H31NO4	ESI+	M+H	325.2253	326.2326
Bosentan	C27H29N5O6S	ESI+	M+H	551.1839	552.1911
Bupropion	C13H18ClNO	ESI+	M+H	239.1077	240.1150
Canrenone	C22H28O3	ESI+	M+H	340.2038	341.2111

Captopril	C9H15NO3S	ESI+	M+H	217.0773	218.0845
Carbamazepine	C15H12N2O	ESI+	M+H	236.0950	237.1022
Celecoxib	C17H14F3N3O2S	ESI+	M+H	381.0759	382.0832
Cetirizine	C21H25ClN2O3	ESI+	M+H	388.1554	389.1626
Ciprofloxacin	C17H18FN3O3	ESI+	M+H	331.1332	332.1405
Citalopram	C20H21FN2O	ESI+	M+H	324.1638	325.1711
Clarithromycin	C38H69NO13	ESI+	M+H	747.4769	748.4842
Clofibrate	C12H15ClO3	ESI+	M+H	242.0710	243.0782
Clofibric acid	C10H11ClO3	ESI-	M-H	214.0397	213.0324
Clonidine	C9H9Cl2N3	ESI+	M+H	229.0174	230.0246
Clopidogrel	C16H16ClNO2S	ESI+	M+H	321.0590	322.0663
Clotrimazole	C22H17ClN2	ESI+	M+H	344.1080	345.1153
Clozapine	C18H19ClN4	ESI+	M+H	326.1298	327.1371
Crotamiton	C13H17NO	ESI+	M+H	203.1310	204.1383
Cyclophosphamide	C7H15Cl2N2O2P	ESI+	M+H	260.0248	261.0321
Cyproterone	C22H27ClO3	ESI+	M+H	374.1649	375.1721
Desloratadine	C19H19ClN2	ESI+	M+H	310.1237	311.1310
Dexamethasone	C22H29FO5	ESI+	M+H	392.1999	393.2072
Diatrizoate	C11H9I3N2O4	ESI+	M+NH4	613.7696	631.8035
Diazepam	C16H13ClN2O	ESI+	M+H	284.0716	285.0789
Diclofenac	C14H11Cl2NO2	ESI+	M+H	295.0167	296.0240
Dimethylaminophenazone	C13H17N3O	ESI+	M+H	231.1372	232.1444
Diphenhydramine	C17H21NO	ESI+	M+H	255.1623	256.1696
Domperidone	C22H24ClN5O2	ESI+	M+H	425.1619	426.1691
Drospirenone	C24H30O3	ESI+	M+H	366.2195	367.2268
Duloxetine	C18H19NOS	ESI+	M+H	297.1187	298.1260
Dydrogesterone	C21H28O2	ESI+	M+H	312.2089	313.2162
Ebastin	C32H39NO2	ESI+	M+H	469.2981	470.3054
EDDP	C20H23N	ESI+	M+H	277.1830	278.1903
Efavirenz	C14H9ClF3NO2	ESI+	M+H	315.0274	316.0347
Enalapril	C20H28N2O5	ESI+	M+H	376.1998	377.2071
Enrofloxacin	C19H22FN3O3	ESI+	M+H	359.1645	360.1718
Erythromycin	C37H67NO13	ESI+	M+H	733.4612	734.4685
Fenofibrate	C20H21ClO4	ESI+	M+H	360.1128	361.1201
Finasteride	C23H36N2O2	ESI+	M+H	372.2777	373.2850
Fluconazole	C13H12F2N6O	ESI+	M+H	306.1041	307.1113
Flumequine	C14H12FNO3	ESI+	M+H	261.0801	262.0874
Flutamide	C11H11F3N2O3	ESI-	M-H	276.0722	275.0649
Fluvoxamine	C15H21F3N2O2	ESI+	M+H	318.1555	319.1628
Furosemide	C12H11ClN2O5S	ESI-	M-H	330.0077	329.0004
Gabapentin	C9H17NO2	ESI+	M+H	171.1259	172.1332
Gemfibrozil	C15H22O3	ESI+	M+H	250.1569	251.1642
Glibenclamide	C23H28ClN3O5S	ESI+	M+H	493.1438	494.1511

Glimepiride	C24H34N4O5S	ESI-	M-H	490.2250	489.2177
Guanylurea	C2H6N4O	ESI+	M+H	102.0542	103.0614
Hydrochlorothiazide	C7H8ClN3O4S2	ESI-	M-H	296.9645	295.9572
Hydrocortisonacetate	C23H32O6	ESI+	M+H	404.2199	405.2272
Hydroxychloroquine	C18H26ClN3O	ESI+	M+H	335.1764	336.1837
Ifosfamide	C7H15Cl2N2O2P	ESI+	M+H	260.0248	261.0321
Imatinib	C29H31N7O	ESI+	M+H	493.2590	494.2663
Indometacin	C19H16ClNO4	ESI+	M+H	357.0768	358.0841
Ketamine	C13H16ClNO	ESI+	M+H	237.0920	238.0993
Ketoconazole	C26H28Cl2N4O4	ESI+	M+H	530.1488	531.1560
Ketoprofen	C16H14O3	ESI+	M+H	254.0943	255.1016
Lidocaine	C14H22N2O	ESI+	M+H	234.1732	235.1805
Lincomycin	C18H34N2O6S	ESI+	M+H	406.2138	407.2210
Loperamide	C29H33ClN2O2	ESI+	M+H	476.2231	477.2303
Lorazepam	C15H10Cl2N2O2	ESI+	M+H	320.0119	321.0192
Losartan	C22H23ClN6O	ESI+	M+H	422.1622	423.1695
L-Thyroxine	C15H11I4NO4	ESI+	M+H	776.6867	777.6940
Mebendazole	C16H13N3O3	ESI+	M+H	295.0957	296.1030
Mebeverine	C25H35NO5	ESI+	M+H	429.2515	430.2588
Medroxyprogesterone	C22H32O3	ESI+	M+H	344.2351	345.2424
Medroxyprogesteroneacetate	C24H34O4	ESI+	M+H	386.2457	387.2530
Mefenamic acid	C15H15NO2	ESI-	M-H	241.1103	240.1030
Megestrol-17-acetate	C24H32O4	ESI+	M+H	384.2301	385.2373
Melperon	C16H22FNO	ESI+	M+H	263.1685	264.1758
Memantine	C12H21N	ESI+	M+H	179.1674	180.1747
Metformin	C4H11N5	ESI+	M+H	129.1014	130.1087
Methimazol	C4H6N2S	ESI+	M+H	114.0252	115.0324
Methotrexate	C20H22N8O5	ESI+	M+H	454.1713	455.1786
Metoprolol	C15H25NO3	ESI+	M+H	267.1834	268.1907
Miconazole	C18H14Cl4N2O	ESI+	M+H	413.9860	414.9933
Mirtazapine	C17H19N3	ESI+	M+H	265.1579	266.1652
Monensin	C36H62O11	ESI+	M+Na	670.4292	693.4190
Montelukast	C35H36ClNO3S	ESI+	M+H	585.2104	586.2177
Mycophenolic acid	C17H20O6	ESI+	M+H	320.1260	321.1333
N-Acetyl mesalazine	C9H9NO4	ESI-	M-H	195.0532	194.0459
N-Acetyl-4-aminoantipyrine	C13H15N3O2	ESI+	M+H	245.1164	246.1237
Naproxen	C14H14O3	ESI+	M+H	230.0943	231.1016
N-Formyl-4-aminoantipyrine	C12H13N3O2	ESI+	M+H	231.1008	232.1081
Nitrendipin	C18H20N2O6	ESI+	M+H	360.1321	361.1394
Nitrofurantoin	C8H6N4O5	ESI-	M-H	238.0338	237.0265
Norfloxacin	C16H18FN3O3	ESI+	M+H	319.1332	320.1405
Norgestimate	C23H31NO3	ESI+	M+H	369.2304	370.2377
Norgestrel	C21H28O2	ESI+	M+H	312.2089	313.2162

Ofloxacin	C18H20FN3O4	ESI+	M+H	361.1438	362.1511
Ondansetron	C18H19N3O	ESI+	M+H	293.1528	294.1601
Orlistat	C29H53NO5	ESI+	M+H	495.3924	496.3996
Oxazepam	C15H11ClN2O2	ESI+	M+H	286.0509	287.0582
Oxybutynin	C22H31NO3	ESI+	M+H	357.2304	358.2377
Oxypurinol	C5H4N4O2	ESI-	M-H	152.0334	151.0261
Paroxetine	C19H20FNO3	ESI+	M+H	329.1427	330.1500
Pentoxifylline	C13H18N4O3	ESI+	M+H	278.1379	279.1452
Phenazone	C11H12N2O	ESI+	M+H	188.0950	189.1022
Pindolol	C14H20N2O2	ESI+	M+H	248.1525	249.1598
Pioglitazone	C19H20N2O3S	ESI+	M+H	356.1195	357.1267
Pipamperone	C21H30FN3O2	ESI+	M+H	375.2322	376.2395
Pravastatin	C23H36O7	ESI-	M-H	424.2461	423.2388
Prednisolone	C21H28O5	ESI+	M+H	360.1937	361.2010
Prednisone	C21H26O5	ESI+	M+H	358.1780	359.1853
Primidone	C12H14N2O2	ESI+	M+H	218.1055	219.1128
Propranolol	C16H21NO2	ESI+	M+H	259.1572	260.1645
Propyphenazone	C14H18N2O	ESI+	M+H	230.1419	231.1492
Raloxifene	C28H27NO4S	ESI+	M+H	473.1661	474.1734
Ranitidine	C13H22N4O3S	ESI+	M+H	314.1413	315.1485
Risperidone	C23H27FN4O2	ESI+	M+H	410.2118	411.2191
Ropinirole	C16H24N2O	ESI+	M+H	260.1889	261.1961
Roxithromycin	C41H76N2O15	ESI+	M+H	836.5246	837.5318
Scopolamine-N-butyl	C21H30NO4	ESI+	M+	360.2169	360.2169
Sertraline	C17H17Cl2N	ESI+	M+H	305.0738	306.0811
Sotalol	C12H20N2O3S	ESI+	M+H	272.1195	273.1267
Sulfadimethoxine	C12H14N4O4S	ESI+	M+H	310.0736	311.0809
Sulfamethazine	C12H14N4O2S	ESI+	M+H	278.0837	279.0910
Sulfamethoxazole	C10H11N3O3S	ESI+	M+H	253.0521	254.0594
Sulfapyridine	C11H11N3O2S	ESI+	M+H	249.0572	250.0645
Sulfathiazole	C9H9N3O2S2	ESI+	M+H	255.0136	256.0209
Tacrolimus	C44H69NO12	ESI+	M+Na	803.4820	826.4718
Tamoxifen	C26H29NO	ESI+	M+H	371.2249	372.2322
Temazepam	C16H13ClN2O2	ESI+	M+H	300.0666	301.0738
Terbinafine	C21H25N	ESI+	M+H	291.1987	292.2060
Tetracain	C15H24N2O2	ESI+	M+H	264.1838	265.1911
Tramadol	C16H25NO2	ESI+	M+H	263.1885	264.1958
Trenbolone	C18H22O2	ESI+	M+H	270.1620	271.1693
Triamcinolone	C21H27FO6	ESI+	M+H	394.1792	395.1864
Trimethoprim	C14H18N4O3	ESI+	M+H	290.1379	291.1452
Valsartan	C24H29N5O3	ESI-	M-H	435.2270	434.2198
Vardenafil	C23H32N6O4S	ESI+	M+H	488.2206	489.2279
Verapamil	C27H38N2O4	ESI+	M+H	454.2832	455.2904

Ziprasidone	C21H21ClN4OS	ESI+	M+H	412.1125	413.1197
10,11-Dihydro-10,11-dihydroxycarbamazepine	C15H14N2O3	ESI+	M+H	270.1004	271.1077
10,11-Dihydro-10-hydroxycarbamazepine	C15H14N2O2	ESI+	M+H	254.1055	255.1128
2-Hydroxycarbamazepine	C15H12N2O2	ESI+	M+H	252.0899	253.0972
4-Formyl-antipyrine	C12H12N2O2	ESI+	M+H	216.0899	217.0972
6-Propyl-2-thiouracil	C7H10N2OS	ESI-	M-H	170.0514	169.0441
Acetyl-sulfamethoxazole	C12H13N3O4S	ESI+	M+H	295.0627	296.0700
Metoprolol acid	C14H21NO4	ESI+	M+H	267.1471	268.1543
Benzophenone-3	C14H12O3	ESI+	M+H	228.0786	229.0859
Benzophenone-4	C14H12O6S	ESI-	M-H	308.0355	307.0282
Phenylbenzimidazole sulfonic acid	C13H10N2O3S	ESI-	M-H	274.0412	273.0339

Table SI-2: List of sites sampled in the study area.

Sampling Location	Coordinates		County	Land use	Crop grown	Water system
	Southings	Eastings				
PS2	0°34'43.43"S	34°36'35.40"E	Homabay	Natural	None	Dam
PS3	0°34'53.96"S	34°32'0.10"E	Homabay	Agriculture	Maize	Dam
PS5	0°46'39.89"S	34°12'21.19"E	Migori	Agriculture	Maize	Dam
PS6	0°48'47.38"S	34°13'15.06"E	Migori	Agriculture	Maize	Dam
PS 9	0°28'29.67"S	34°32'56.95"E	Homabay	Agriculture, urban	Rice	Dam
PS13	0°22'48.01"S	34°38'30.92"E	Homabay	Agriculture	Maize	Dam
PS14	0°40'50.77"S	34°32'39.07"E	Homabay	Agriculture, urban	Maize	Dam
PS16	0°54'29.02"S	34°33'28.89"E	Migori	Agriculture	Sugarcane	Tributary
PS17	0°53'54.15"S	34°31'24.54"E	Migori	Agriculture , urban	Sugarcane	Tributary
PS18	0°53'8.15"S	34°31'20.35"E	Migori	Agriculture, urban ,industrial	Sugarcane	Tributary
PS19	0°49'46.82"S	34°23'44.12"E	Migori	Agriculture	Sugarcane	Ox-bow
PS20	0°49'4.79"S	34°23'34.97"E	Homabay	Agriculture	Sugarcane	Ox-bow
PS21	0°19'20.39"S	34°47'20.33"E	Homabay	Natural	None	Ox-bow
PS25	0° 8'14.32"S	34°56'35.97"E	Kisumu	Agriculture	Rice	Rice channel
PS26	0° 8'17.02"S	34°56'8.82"E	Kisumu	Agriculture	Rice	Rice channel
PS27	0° 8'38.67"S	34°58'20.86"E	Kisumu	Agriculture	Rice	Tributary
PS28	0°10'18.07"S	34°54'28.44"E	Kisumu	Agriculture	Rice	Tributary
PS 29	0°48'19.63"S	34°41'54.93"E	Kisii	urban	None	Main River
PS 30	0°48'2.29"S	34°43'45.10"E	Kisii	Agriculture	Tea,sugarcane	Main River
PS33	0°22'38.69"S	34°38'6.98"E	Homabay	Agriculture	Maize	Main River
PS34	0°23'17.85"S	34°38'30.02"E	Homabay	Agriculture	Maize	Main River
PS35	0°49'50.42"S	34°44'44.98"E	Kisii	Agriculture	Tea	Tributary
PS36	0°48'31.95"S	34°43'58.93"E	Kisii	Agriculture	Tea,sugarcane	Tributary
PS 37	0°27'47.58"S	34°32'55.31"E	Homabay	Agriculture	maize	Tributary
PS38	0°59'50.59"S	34°16'55.57"E	Migori	Agriculture	Maize	Main river

Sampling Location	Coordinates		County	Land use	Crop grown	Water system
	Southings	Eastings				
PS39	0°59'40.04"S	34°17'26.47"E	Migori	Natural	None	Main river
PS40	1° 3'53.69"S	34°28'5.52"E	Migori	urban	None	Main river
PS42	0°39'27.61"S	34°42'37.10"E	Kisii	Agriculture, Urban	Maize	Tributary
PS43	0°39'24.17"S	34°41'57.84"E	Kisii	Agriculture, Urban	Maize	Tributary
PS46	0°30'48.53"S	34°17'29.08"E	Homabay	Natural	None	Tributary
PS47	0°33'40.22"S	34°18'9.70"E	Homabay	Natural	None	Tributary
PS52	0°27'45.4"S	34°33'55.1"E	Homabay	Natural	None	channel
PS53	0°29'9.67"S	34°31'2.85"E	Homabay	Agriculture	Rice	Tributary
PS54	0°28'34.93"S	34°32'41.36"E	Homabay	Agriculture	Rice	Rice channel
PS55	0°28'34.33"S	34°31'56.51"E	Homabay	Agriculture	Rice, maize	Dam
PS56	0°27'5.46"S	35°13'8.39"E	Kericho	Agriculture	Tea	Tributary
PS57	0°29'12.72"S	35°10'58.85"E	Kericho	Agriculture	Tea	Tributary
PS58	0°30'54.82"S	35° 4'49.80"E	Kericho	urban, natural	None	Main river
PS59	0°19'1.31"S	35° 0'22.66"E	Kisumu	Agriculture	Maize,	Tributary
ULI1	0°27'5.24"S	35° 7'15.27"E	Kericho	Agriculture	Tea, urban	Main River
ULI3	0°32'20.58"S	35° 2'0.91"E	Nyamira	Agriculture	Tea	Tributary
ULI 5	0°23'35.22"S	35° 0'35.88"E	Kericho/ Kisumu	Urban	None	Main River
ULI9	0°59'9.30"S	34°35'5.25"E	Migori	Agriculture	Sugarcane, maize	Tributary
ULI10	1° 1'18.24"S	34°37'27.29"E	Narok	Agriculture	Sugarcane	Tributary
ULI13	0°27'34.68"S	34°35'40.87"E	Homabay	Agriculture	Maize	Tributary
ULI15	0° 9'0.91"S	34°55'49.30"E	Kisumu	Agriculture	Rice	Rice channel
ULI16	0° 6'46.67"S	34°47'29.77"E	Kisumu	urban	None	Main river
ULI17	0°28'38.92"S	34°32'38.87"E	Homabay	Agriculture	Rice	Rice channel

Quality control

The trip blank consisting of LC-MS grade water was taken on the sampling trip without opening and stored and shipped with the samples. To check possible contamination during sampling from the

pipette, a sampling blank also containing 1 mL LC-MS grade water was used. During sampling, the sampling blank vial was opened and using the sampling pipette, 1 mL of the water was taken and released again into the same vial. The sampling and trip blanks were transported to the laboratory together with the environmental samples. The blanks were also processed and analysed in a similar manner as the samples.

Table SI-3: Gradient for instrumental analysis. Data acquisition on the MS was done from 0 to 24 minutes. Solvent C was used for cleaning the column from hydrophobic matrix residues prior to re-equilibration.

Time [min]	**Flow rate [mL]**	**Solvent A [%] Water + 0.1% formic acid**	**Solvent B [%] MeOH + 0.1% formic acid**	**Solvent C [%] Acetone/ Isopropanol (50:50)**
0	0.3	95	5	0
1	0.3	95	5	0
13	0.3	0	100	0
24	0.3	0	100	0
24.1	0.35	5	10	85
26.2	0.35	5	10	85
26.3	0.35	95	5	0
31.9	0.35	95	5	0
32.0	0.3	95	5	0

Table SI-4: Ion source parameters setting in positive and negative ionization mode

Parameter	Positive mode	Negative mode
Sheath gas flow rate	45	25
Aux gas flow rate	1	1
Spray voltage [kV]	3.8	3.5
Capillary temperature [°C]	300	300
S-lens RF level	70	70
Aux gas heater temperature [°C]	300	280

For both modes, a combination of full scan (m/z range 100 - 1500) at a nominal resolving power of 70,000 (referenced to m/z 200) and data-independent MS/MS fragmentation (DIA) with an isolation window of m/z = 50 (m/z range 97-476) or m/z = 260 (m/z range 473-1501) at a resolving power of 35,000 and centroid spectrum data was used. For suspect screening, dd-MS2 experiments were performed for confirmation with an inclusion list of the candidate ions. In full scan mode, the nominal resolving power was 70,000 (referenced to m/z 200) and 35,000 (referenced to m/z 200) in dd-MS^2 scans.

Table SI-5: MZmine settings used in the data processing.

Step	Parameter	Setting
Mass detection		
	Mass detector	Centroid
	Noise level	5e3
ADAP chromatogram building		
	Min group size of # of scans	8
	Group intensity threshold	1e4
	Min highest intensity	5e3
	m/z tolerance	0.001 m/z or 7 ppm
Smoothing		
	Filter width	7
Chromatogram deconvolution		
	Algorithm	Local minimum search
	Chromatographic threshold	60 %
	Search minimum in RT range	0.10 min
	Minimum relative height	30 %
	Minimum absolute height	5e4
	Min ration of peak top/edge	2.3
	Peak duration range	0.1 – 5 min
Join aligner		
	m/z tolerance	0.001 m/z or 7 ppm
	Weight for m/z	70
	Retention time tolerance	0.3 (absolute) min
	Weight for RT	30
Custom database search		
	m/z tolerance	0.001 m/z or 7 ppm
	Retention time tolerance	0.4 (absolute) min
Gap filling		
	Intensity tolerance	30 %
	m/z tolerance	0.001 m/z or 7 ppm
	Retention time tolerance	0.15 (absolute) min
	RT correction	yes

Table SI-6: Settings of Trace Finder parameters

Step	Parameter	Setting
Processing	m/z tolerance	7 ppm
	Sensitivity	ICIS
	Detection method	Nearest RT
	Smoothing	3
Isotopes	Fit Threshold (%)	75
	Allowed Mass Deviation (ppm)	5
	Allowed Intensity Deviation (%)	10
	Use Internal Mass Calibration	Yes
Qualitative peak processing	Enable peak threshold	Yes
	% of largest peak by height	10
	Only select top peaks by height	10
	ISTD matching (+/- min)	0.025
	Exclude matching quan peaks window	0.025
Fragments		
	Ignore if not defined	Yes
	Minimum number of fragments	1
	Intensity threshold	1e4
	Mass tolerance	8 ppm

Table SI-7: List of 6 suspect candidates with the criteria used for confirmation and identification

Compound	Molecular formula	Class	Ionization	Log K_{ow}	△Rt (min)	Metfrag Fragment march	Similarity	Confirmation with standard
Rimantidine	$C_{12}H_{21}N$	Pharmaceutical	Positive	3.34[1]	4.1	1.98	MassBank and MetFrag match	No
Adenosine	$C_{10}H_{13}N_5O_4$	Natural/ Xenobiotic	Positive	-1.05[1]	0.08	2.0	MassBank match (Record: Adenosine, EQ330401), SIRIUS 92%, Presence of characteristic fragment: 136.0619, 73.0275	Yes
Nevirapine	$C_{15}H_{14}N_4O$	Pharmaceutical	Positive	3.89[1]	-0.07	1.99	MetFrag, mzCloud 82%, SIRIUS 99%	Yes
Pencycuron	$C_{19}H_{21}ClN_2O$	Pesticide	Negative	4.82[2]	-0.05	2.0	MetFrag	No
Lamivudine	$C_8H_{11}N_3O_3S$	Pharmaceutical	Positive	-9.54[1]	-0.01	1.97	mzCloud 71%, MetFrag	Yes
Flupyradifurone	$C_{12}H_{11}ClF_2N_2O_2$	Pesticide	Negative	1.2[3]	-6.55	2.0	MetFrag	No

Log K_{ow} values extracted from [1]Pubchem database, [2]Chemspider Database (Royal society of chemistry) and [3]European Chemical Agency (ECHA) MassBank search settings: relative intensity: 7, tolerance: 0.1, MS Type: MS2, Instrument Type: ESI)

Table SI-8: Effect data for fish, daphnia and algae used in the calculation of toxic units of individual compound

	Effect data values (mg/l)		
Compound	Fish	Daphnia	Algae
Tris(2-chloroethyl)phosphate	6.60E+01 (E)	1.13E+02 (PE)	2.69E+00 (PE)
Triethylcitrate	2.34E+02 (PE)	5.75E+02 (PE)	5.81E+00 (PE)
N-Butylbenzenesulfonamide	3.92E+01 (PE)	2.06E+01 (PE)	1.23E+01 (PE)
Cotinine	2.71E+02 (PE)	1.29E+01 (PA)	8.09E+00 (PA)
N-Ethyl-o-toluenesulfonamide	6.78E+01 (PE)	6.98E+01 (PE)	6.66E+01 (PE)
2,4-Dichlorophenoxyacetic acid	3.15E+00 (E)	2.74E+00 (E)	2.00E+00 (E)
2-Hydroxyatrazine	6.33E+01 (PE)	1.35E+02 (PE)	8.97E+01 (PE)
Acetamiprid	1.32E+01 (E)	1.40E-02 (E)	1.01E+00 (E)
Ametryn	5.00E-01 (E)	3.53E+00 (E)	8.83E+00 (PE)
Atrazine	9.13E-01 (E)	1.24E-01 (E)	7.86E-03 (E)
Azoxystrobin	4.87E-01 (E)	5.94E-02 (E)	2.09E-02 (E)
Bendiocarb	5.61E-01 (E)	7.58E-03 (E)	4.38E+04 (PE)
Carbendazim	1.21E-02 (E)	2.51E-02 (E)	4.20E-01 (E)
Chlormequat	1.12E+03 (PE)	2.92E+00 (PE)	1.47E+02 (PE)
Chlorothalonil-4-hydroxy	4.79E+00 (E)	4.77E-01 (PE)	5.52E-01 (PE)
DEET	7.20E+01 (E)	6.12E+01 (PA)	4.05E+00 (PA)
Desethylatrazine	5.60E+01 (PE)	1.26E+02 (PA)	3.08E-02 (PA)
Desisopropylatrazine	8.19E+01 (PE)	2.07E+02 (PA)	6.77E-04 (E)
Diazinon	3.00E-02 (E)	1.00E-05 (E)	1.00E+00 (E)
Diuron	2.06E-01 (E)	3.82E-01 (E)	6.94E-04 (E)
Dodemorph	4.08E+00 (PE)	3.34E+00 (PE)	8.16E-02 (PE)
Ethofumesate	5.65E+00 (E)	2.5E+00 (E)	2.90E+00 (E)
Fenuron	9.39E+01 (PE)	1.29E+00 (PA)	1.13E+00 (PA)
Hexazinone	1.20E+02 (E)	8.17E+01 (E)	3.00E-03 (E)
Imidacloprid	5.30E+01 (E)	5.43E-02 (E)	1.22E+01 (PE)
Imidacloprid-guanidine_DP	1.97E-01 (PE)	9.74E-01 (PE)	2.41E+01 (PE)
MCPA	1.50E+00 (E)	2.51E+00 (E)	4.48E+00 (E)
Mepiquat	1.58E+03 (E)	1.06E+02 (E)	1.99E+01 (PE)
Metalaxyl	2.19E+01 (E)	3.92E-01 (E)	2.08E+00 (E)
Metolachlor	4.10E-02 (E)	5.05E+00 (E)	2.60E-02 (E)
Metribuzin	3.72E+00 (E)	3.71E+01 (E)	8.00E-03 (E)
Pirimiphos-methyl	2.63E-02 (E)	2.09E-04 (E)	2.62E+01 (E)
Simazine	2.50E+00 (E)	5.19E+01 (PA)	3.08E-02 (E)
Terbuthylazine	2.71E+00 (E)	1.09E-01 (E)	1.30E-03 (E)
6-Propyl-2-thiouracil	1.78E+01 (PE)	1.04E+01 (PE)	1.30E+00 (PE)
7-Amino-4-methylcoumarin	2.95E+00 (PE)	8.31E+00 (PE)	2.29E+01 (PE)
Fluvoxamine	6.43E-01 (E)	8.40E-01 (E)	3.76E+00 (E)
Metformin	2.14E+02 (PE)	1.35E+03	3.20E+02 (E)
Pravastatin	7.93E+00 (PE)	7.02E+01 (PE)	4.96E+01 (PE)
10,11-Dihydro-10,11-dihydroxycarbamazepine	2.78E+02 (PE)	1.18E+03 (PE)	2.29E+00 (PA)
10,11-Dihydro-10-hydroxycarbamazepine	6.50E+01 (PE)	2.64E+02 (PE)	1.61E+00 (PA)
2-Hydroxycarbamazepine	8.26E+00 (PE)	5.90E+01 (PA)	1.12E+01 (PA)

4-Formyl-antipyrine	3.34E-01 (PE)	8.29E+00 (PE)	9.57E+01 (PE)
Acetaminophen	1.88E+02 (E)	1.60E-01 (E)	3.20E-02 (E)
Acetyl-sulfamethoxazole	1.36E+02 (PE)	6.03E+01 (PE)	1.93E+01 (PA)
Bisoprolol	3.75E+01 (PE)	4.01E+00 (PE)	2.39E+00 (PA)
Carbamazepine	3.63E+01 (E)	1.11E+02 (E)	1.65E+01 (E)
Diphenhydramine	6.97E+00 (PE)	9.25E-01 (PE)	6.19E-01 (PE)
fluconazole	3.06E+01 (E)	6.97E+02 (PE)	2.54E+01 (PE)
Metoprolol acid	1.46E+02 (PE)	1.24E+01 (PE)	4.10E+00 (PE)
Phenazone	1.78E-01 (PE)	6.52E+00 (PE)	2.97E+00 (PA)
Propranolol	1.18E+01 (E)	9.19E-01 (E)	5.80E+00 (E)
Propyphenazone	2.20E-02 (PE)	5.74E+00 (PE)	7.05E+00 (PE)
Sulfadimethoxine	1.63E+02 (PE)	2.48E+02 (PE)	1.85E+00 (PE)
Sulfamethoxazole	5.79E+02 (E)	3.54E+01 (E)	5.79E-01 (E)
Trimethoprim	1.00E+02 (E)	1.00E+02 (E)	2.67E+01 (E)
Verapamil	1.00E+01 (E)	1.43E-01 (PE)	1.83E+00 (PE)
Triclosan	2.55E-01 (E)	8.30E-02 (E)	1.00E-03 (E)
Hexadecylpyridinium	3.00E-03 (PE)	4.00E-03 (PE)	2.20E-02 (PE)
Lauryl diethanolamide	3.94E+00 (PE)	3.39E+00 (PE)	2.87E-01 (PE)
Perfluorohexanoic acid	5.54E+01 (PE)	3.48E+01 (PE)	1.04E+03 (E)
Triphenylphosphine oxide	5.37E+01 (E)	5.70E+00 (PE)	4.90E+00 (PE)
2,4-Dinitrophenol	5.55E-01 (E)	3.09E+00 (E)	1.24E+01 (E)
4-(Dimethylamino)pyridine	5.53E+02 (PE)	2.74E+02 (PE)	2.55E+00 (PE)
4-Aminobenzamide	3.88+02 (PE)	5.12E+00 (PE)	1.95E+01 (PE)
4-Amino-N,N-dimethylbenzenesulfonamide	1.91E+02 (PE)	1.48E+02 (PE)	1.33E+01 (PE)
Acesulfame	2.57E+03 (PE)	5.82E+02 (PE)	9.93E-01 (PA)
Benzophenone-4	1.18E+02 (PE)	3.34E+01 (PE)	1.62E+02 (PE)
Daidzein	3.99E-01 (PE)	3.38E+01 (PE)	1.94E+01 (PE)
Genistein	1.90E+00 (E)	6.59E+01 (PE)	3.56E+01 (PE)
Icaridin	3.21E+01 (PE)	2.39E+00 (PE)	5.65E+01 (PE)
Tri(butoxyethyl)phosphate	6.80E+00 (E)	1.04E+01 (PE)	1.02E+00 (PE)
triethylphosphate	1.00E+02 (PE)	9.50E+02 (E)	3.73E+00 (PE)
Saccharin	1.07E+00 (PE)	2.56E+01 (PA)	1.29E+00 (PA)
Triclocarban	1.30E-02 (E)	8.00E-03 (E)	1.00E-02 (E)

(1)Experimental data retrieved from the EPA ECOTOX database (E), (2) predicted read-across data (PA), (3) predicted ECOSAR database (Version 2.0.2) (PE). Derived from Busch et al., 2016

Nevirapine $[M+H]^+$: m/z 267.1241
Smiles: CC1=C2C(=NC=C1)N(C3=C(C=CC=N3)C(=O)N2)C4CC4

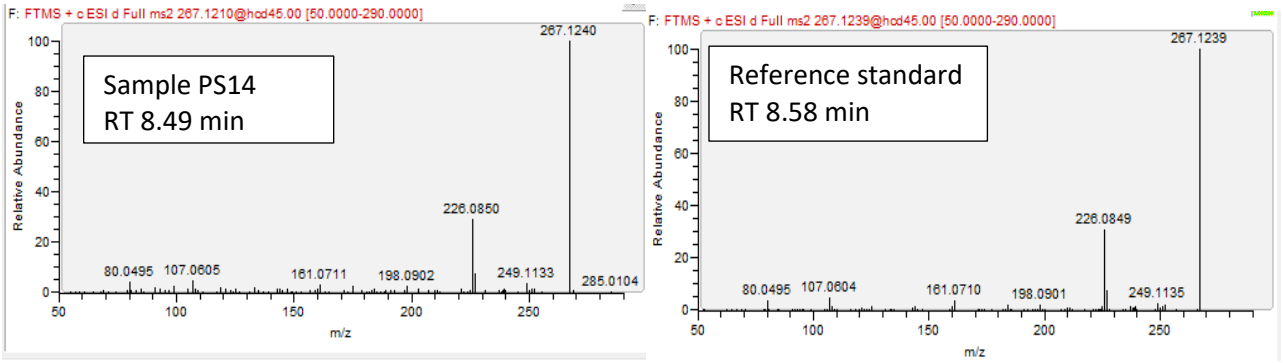


Figure SI-1.1: MS/MS spectra of Nevirapine in environmental sample PS14 and in reference standard with the retention time.

Lamivudine $[M+H]^+$: m/z 230.0594
Smiles: C1C(OC(S1)CO)N2C=CC(=NC2=O)N

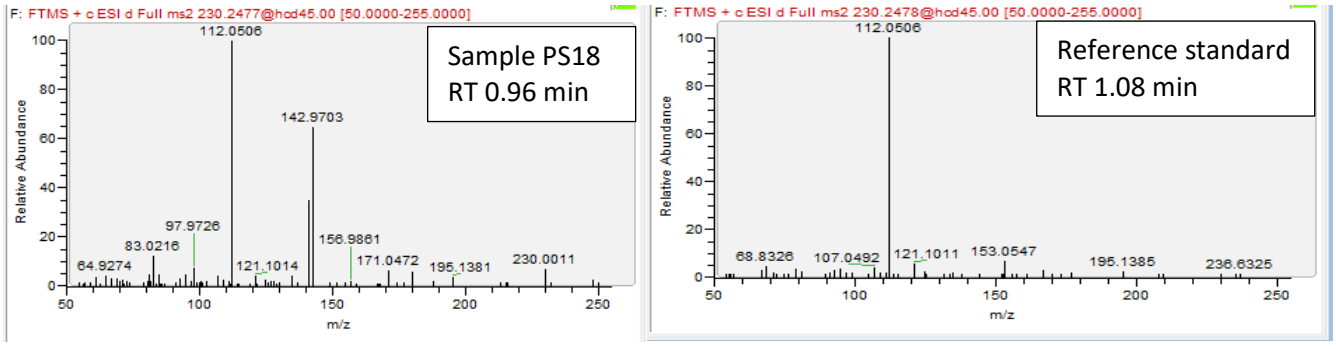


Figure SI-1.2: MS/MS spectra of lamivudine in environmental sample PS18 and in reference standard with the retention time.

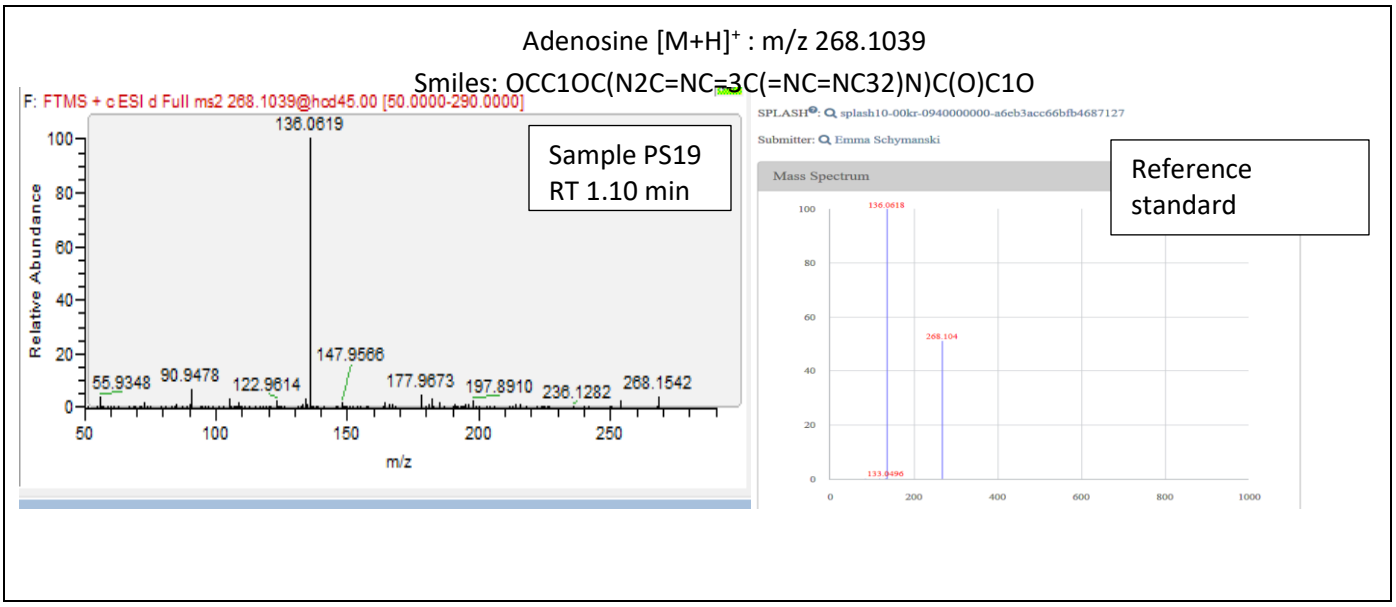


Figure SI-1.3: MS/MS spectra of adenosine in environmental sample PS19 and in reference standard with the retention time.

The MS^2 spectrum for adenosine was obtained from MassBank (Horai et al., 2010) (Record EQ330401) under splash10-00kr-0940000000-a6eb3acc66bfb4687127. Analysis was performed by Otto et al. on LC-ESI-QFT (Q Exactive Orbitrap, Thermo scientific)

Table SI-9: Results of the physical-chemical parameters for water samples from individual sampling sites.

Sampling site	Temp(°C)	Total Conductivity (µs/cm)	pH @ specified temp	Dissolved Oxygen (mg/l O_2)	Turbidity (NTU)	Ammonium (mg/l $NH4^{+)}$	Phosphate(mg/l PO_4^{3-})	Nitrates(mg/l NO_3^-)	Carbonate hardness (°d)	Nitrites(mg/l NO_2^-)	Field Storage (4°C, -20°C):
PS 37	23.1	237	7.21	14.95	72	ND	10	10	7.8	0.025	23
PS 9	34.3	109	7.8	5.10	425	ND	25	10	3.0	0.05	5
ULI 17	37.1	478	8.41	7.75	376	ND	25	25	14.4	ND	1
PS 47	27.6	159	7.27	2.33	791	0.2	25	25	5.4	0.025	-3
PS 14	24.3	152	6.94	2.28	32	ND	3	10	4.0	0.025	-6
PS 3	30.5	141	7.84	6.19	379	ND	10	10	4.0	ND	-5
PS 2	31.5	168	7.36	5.93	21	0.2	25	10	5.0	0.2	-4
PS 13	27.0	101	7.94	4.25	788	0.2	10	10	4.0	0.025	-15
PS 34	24.7	74	7.4	6.46	308	ND	25	10	2.0	0.05	-12
PS 33	24.4	80	7.61	6.09	951	ND	25	25	2.0	0.025	-7
PS 5	24.9	123	7.17	2.48	943	0.2	10	10	3.8	0.025	-3
PS 6	33.7	165	8.34	7.60	2010	0.2	25	10	5.2	0.15	1
PS 20	32.6	172	7.52	5.20	144	ND	10	10	4.0	ND	-4
PS 19	31.4	87	8.92	11.43	414	0.2	10	10	2.2	0.025	-5
PS 42	19.9	116	7.62	6.02	1050	0.2	25	10	3.6	0.15	-5
PS 43	21.4	135	7.52	5.51	313	0.6	25	25	2.0	0.3	-10

Sampling site	Temp(°C)	Total Conductivity (µs/cm)	pH @ specified temp	Dissolved Oxygen (mg/l O_2)	Turbidity (NTU)	Ammonium (mg/l $NH4^{+)}$	Phosphate(mg/l PO_4^{3-})	Nitrates(mg/l NO_3^-)	Carbonate hardness (°d)	Nitrites(mg/l NO_2^-)	Field Storage (4°C, -20°C):
PS 46	26.9	224	6.67	3.36	369	0.2	25	10	7.2	ND	-6
ULI 5	19.4	52	5.62	7.62	26	ND	25	25	1.6	ND	-7
ULI 3	20.8	75	5.37	6.72	32	ND	10	10	2.0	0.025	-5
ULI 1	21.8	46	7.04	5.35	12	ND	10	10	2.4	ND	-5
PS 30	18.9	57	7.45	6.32	116	ND	10	10	6.0	ND	-10
PS 36	21.2	74	7.52	7.34	1569	ND	10	10	6.0	ND	-4
PS 35	22.9	80	7.55	5.81	356	ND	10	10	6.0	ND	-2
PS 29	23.5	59	7.76	6.92	150	ND	25	10	6.0	ND	-1
PS 21	25.9	86	6.77	1.77	452	ND	10	ND	3.0	ND	-2
Uli 13	26.2	114	7.65	5.81	80	ND	25	10	4.0	0.025	-2
Uli 15	27.3	133	6.81	3.35	402	0.2	25	10	5.6	0.025	-3
Uli 16	25.4	102	7.46	5.90	135	ND	25	ND	2.6	0.025	-2
PS 27	23.3	212	7.93	6.15	448	ND	10	ND	7.0	0.025	-10
PS 25	25.5	344	7.08	2.11	24	ND	25	ND	9.0	0.025	-6
PS 26	33.9	251	7.65	5.13	2990	ND	25	10	9.0	0.025	-3
PS 28	27.2	260	7.24	NM	177	ND	25	10	8.0	0.025	-2
PS 39	26.3	185	7.93	2.10	1915	ND	10	10	6.0	0.025	-5

Sampling site	Temp(°C)	Total Conductivity (µs/cm)	pH @ specified temp	Dissolved Oxygen (mg/l O_2)	Turbidity (NTU)	Ammonium (mg/l $NH4^{+)}$	Phosphate(mg/l PO_4^{3-})	Nitrates(mg/l NO_3^-)	Carbonate hardness (°d)	Nitrites(mg/l NO_2^-)	Field Storage (4°C, -20°C):
PS 38	27.5	215	8.02	1.68	503	ND	25	10	6.0	0.025	-2
PS 40	27.5	172	7.85	6.12	558	ND	25	ND	5.8	0.05	-2
PS 17	21.0	142	7.65	3.66	116	0.2	10	10	5.6	0.025	-5
PS 18	23.4	170	7.48	2.94	139	0.6	10	10	6.4	0.3	-3
PS 16	26.7	100	7.42	5.88	228	ND	10	ND	3.4	0.025	-2
ULI 10	20.6	128	7.74	4.59	85	0.2	10	ND	6.2	0.2	-14
ULI 9	22.8	287	7.28	1.85	77	ND	25	ND	10.0	0.025	-10
PS 52	23.6	97	7.64	4.01	1625	ND	10	ND	4.2	0.025	-7
PS 53	26.1	256	7.03	1.58	62	ND	25	ND	11.0	ND	-5
PS 54	32.3	337	7.46	3.95	728	ND	25	ND	15.0	ND	-3
PS 55	30.7	113	7.97	4.2	151	ND	25	ND	5.0	0.025	-2
PS 56	17.9	39	8.27	2.95	0.9	ND	3	ND	2.0	ND	-7
PS 57	16.9	37	7.60	5.14	65	0.2	3	ND	2.2	ND	-3
PS 58	20	68	7.50	4.50	763	ND	10	ND	1.8	0.025	-2
PS 59	23.8	215	4.92	3.20	86	ND	10	ND	7.0	0.025	-2

NTU: Nephelometric Turbidity Units. ND: not detected. NM: Not measured due to instrument error.

Table SI-10: Results of the compound concentrations found in the water samples (In separate Excel file)

This dataset can also be found online using the link Faith Jebiwot Kandie. (2020). Appendix A [Data set]. Zenodo. http://doi.org/10.5281/zenodo.4010825

Table SI-11: Toxic unit values of the compounds found in the water samples (In separate Excel file)

This dataset can also be found online using the link Faith Jebiwot Kandie. (2020). Appendix A [Data set]. Zenodo. http://doi.org/10.5281/zenodo.4010825

Appendix B

Table SI-1: List of internal standards used

Internal standards				
Compound	**Formula**	**Ion mode**	**Adduct**	**Expected ion**
IS_1-Naphthol-D7	C10H1[2]H7O1	ESI-	M-H	150.0942
IS_4-Nitrophenol-D4	C6H1[2]H4N1O3	ESI-	M-H	142.0448
IS_Acesulfame-D4	C4H1[2]H4N1O14S	ESI-	M-H	166.0118
IS_Atenolol-D7	C14H15[2]H7N2O3	ESI+	M+H+	274.2143
IS_Atrazine-13C3	C5[13]C3H14Cl1N5	ESI+	M+H+	219.1111
IS_Bentazone-D6	C10H6[2]H6N2O3S1	ESI-	M-H	245.0872
IS_Benzophenone-3-D5	C14H7[2]H5O3	ESI+	M+H+	234.1173
IS_Benzotriazole-D4	C6H1[2]H4N3	ESI+	M+H+	124.0807
IS_Bezafibrate-D4	C19H16[2]H4Cl1N1O4	ESI+	M+H+	366.1405
IS_Bezafibrate-D4	C19H16[2]H4Cl1N1O4	ESI-	M-H	364.1259
IS_Caffeine-D3	C8H7[2]H3N4O2	ESI+	M+H+	198.1065
IS_Carbamazepine-D10	C15H2[2]H10N2O1	ESI+	M+H+	247.165
IS_Carbendazim-D4	C9H5[2]H4N3O2	ESI+	M+H+	196.1019
IS_Chlormequat-D9	C5H4[2]H9Cl1N1	ESI+	M+	131.1296
IS_Clarithromycin-D3	C38H66[2]H3N1O13	ESI+	M+H+	751.503
IS_Cotinine-D3	C10H9[2]H3N2O1	ESI+	M+H+	180.1211
IS_Creatinine-D3	C4H4[2]H3N3O1	ESI+	M+H+	117.085
IS_Cyclamate-D11	C6H2[2]H11N1O3S1	ESI-	M-H	189.1234
IS_Decyltrimethylammonium-D3	C13[2]H30N1	ESI+	M+	230.4256
IS_DEET-D7	C12H10[2]H7N1O1	ESI+	M+H+	199.1822
IS_Desisopropylatrazine-D5	C5H3[2]H5Cl1N5	ESI+	M+H+	179.0855
IS_Diazinon-D10	C12H11[2]H10N2O3P1S1	ESI+	M+H+	315.1711
IS_Diclofenac-D4	C14H7[2]H4Cl2N1O2	ESI+	M+H+	300.0491
IS_Diclofenac-D4	C14H7[2]H4Cl2N1O2	ESI-	M-H	298.0345
IS_Diglyme-D6	C6H8[2]H6O3	ESI+	M+H+	141.1392
IS_Hydrochlorothiazide-13C6	C1[13]C6H8Cl1N3O4S2	ESI-	M-H	301.9773
IS_Imidacloprid-D4	C9H6[2]H4Cl1N5O2	ESI+	M+H+	260.0847
IS_Isoproturon-D3	C12H15[2]H3N2O1	ESI+	M+H+	210.168
IS_Laurylsulfate-D25	C12H1[2]H25O4S	ESI-	M-H	290.3048
IS_Mecoprop-D3	C10H8[2]H3Cl1O3	ESI-	M-H	216.0512
IS_Metolachlor-D6	C15H16[2]H6Cl1N1O2	ESI+	M+H+	290.1788
IS_Mono-isobutylphthalate-D4	C6H1[2]H4N1O3	ESI+	M+H+	227.1216
IS_Mono-isobutylphthalate-D4	C12H10[2]H4O4	ESI-	M-H	225.107
IS_Progesterone-D9	C21H21[2]H9O2	ESI+	M+H+	324.2883
IS_p-Toluene-sulfonamide-D4	C7H5[2]H4NO2S1	ESI+	M+NH4+	176.0678
IS_Sulfamethoxazole-D4	C10H7[2]H4N3O3S1	ESI+	M+H+	258.0845
IS_Tebuconazole-D9	C16H13[2]H9Cl1N3O1	ESI+	M+H+	317.2089

Internal standards				
Compound	**Formula**	**Ion mode**	**Adduct**	**Expected ion**
IS_Triclosan-D3	C12H4[2]H3Cl3O2	ESI-	M-H	289.9627
IS_Tri-n-butylphosphate-D27	C12[2]H27O4P1	ESI+	M+H+	294.3414
IS_Verapamil-D6	C27H32[2]H6N2O4	ESI+	M+H+	461.3281

SI-2: Analysis of total organic carbon performed on sediment samples.

Based on the assumption that contaminants adsorb to the organic carbon phase rather than the inorganic components, the total organic carbon (TOC) analysis was essential to normalize the contaminant content on the organic carbon content. For the analysis, triplicate 5 mg sediment samples were folded completely in aluminum foil to remove any air spaces. The samples were then combusted at 1000ºC on a Flash 200 organic elemental analyzer coupled to a Delta V Advantage Isotope Ratio mass spectrometer (Thermo). A temperature conducting detector (TCD) was used to measure the signal after the gas chromatography. Six level calibration points between 0.1 and 2 mg was used to develop the calibration curve with acetanilide as the standard.

SI-3: LC program for the extract clean-up by flash column chromatography.

Prior to extract clean up using flash column chromatography, 10.5 mL of dichloromethane (DCM, LC grade) was used to condition the flash column for 2.1 minutes. Sediment clean-up was first performed using DCM followed by methanol and collected separately in 20 mL pre-cleaned evaporation vials.

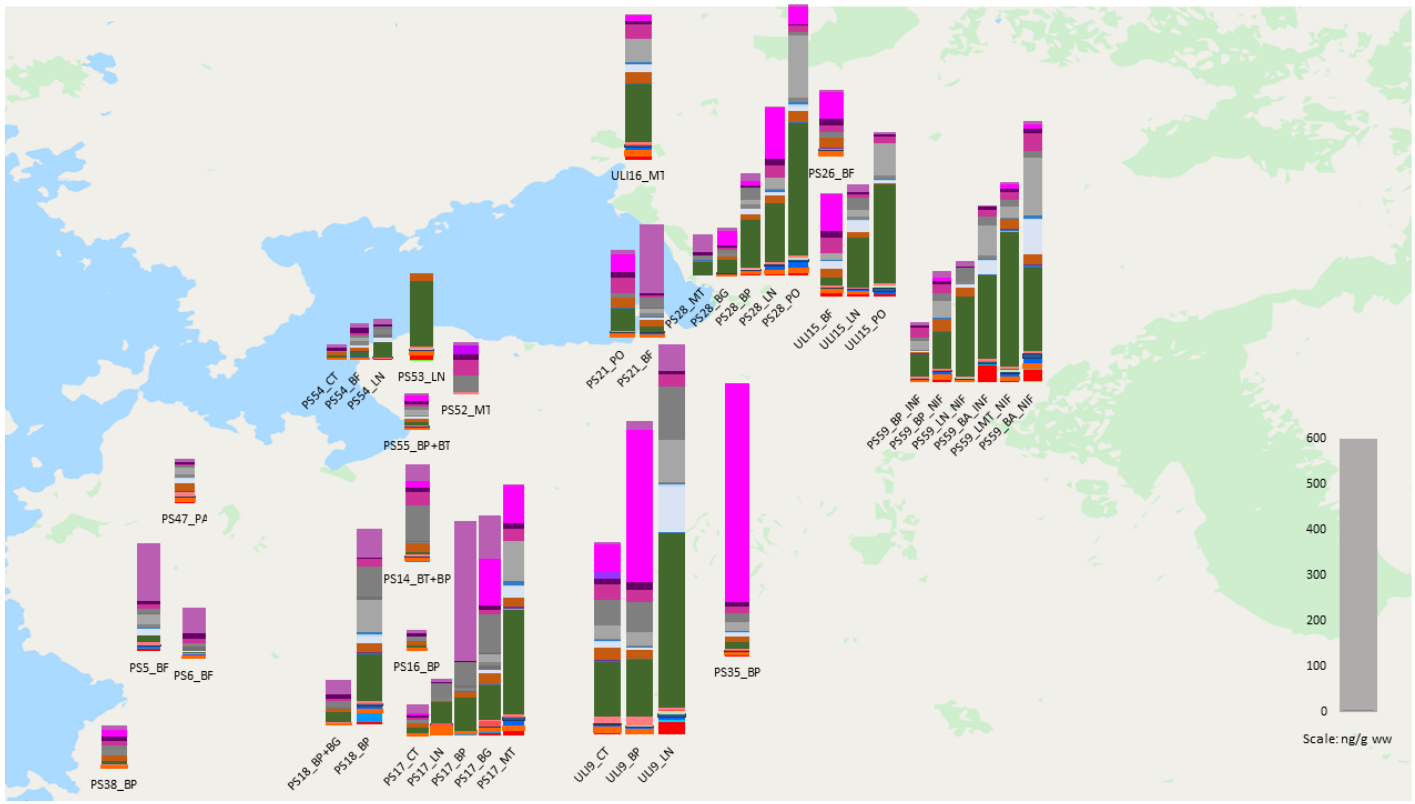
Channel	**Time (min)**	**Flow rate (mL min^{-1})**
DCM	0 - 0.5	0 - 5
DCM	0.5 – 3.5	5
DCM	5.5 - 5	5 - 0

Channel	**Time (min)**	**Flow rate (mL min^{-1})**
Methanol	0 - 0.5	0 - 5
Methanol	0.5 – 3.5	5
Methanol	3.5 – 3.6	5 - 0

Table SI-4: List of compounds concentrations found in snail tissue samples (in Excel file)

Table SI-5: List of compounds concentrations found in sediment samples (in Excel file)

Table SI-4 and SI-5 can be found online under Faith Jebiwot Kandie. (2020). Appendix B [Data set]. Zenodo. http://doi.org/10.5281/zenodo.4010840



- Diazinon
- Imidacloprid
- Pirimiphos-methyl
- DEET
- Acetamiprid
- Triclocarban
- Desethylatrazine
- Atrazine
- Flusilazole
- Metalaxyl
- Spiroxamine
- Azoxystrobin
- Tebuconazole
- Bupirimate
- Dodemorph
- Oxazepam
- Temazepam
- Efavirenz
- Fluvoxamine
- 2-Hydroxyquinoline
- N-Butylbenzenesulfonamide
- Tris(2-chloroethyl)phosphate
- Triphenylphosphate
- N-Ethyl-o-toluenesulfonamide
- 2-Benzothiazolesulfonic acid
- Cotinine

BA: *Bulinus africanus (nasutus);* BF: *Bulinus forskalii;* BT: *Bulinus truncates*; BP: *Biomphalaria pfeifferi*; BG: *Bulinus globusus*; CT: *Ceratophallus sp.*; LN:*Lymnea natalensis;* MT: *Melanoides tuberculata*; PO: *Pila ovata*; PA: *Physa acuta.*

Figure SI-1: Compound concentrations and spatial distribution in snail tissue samples.

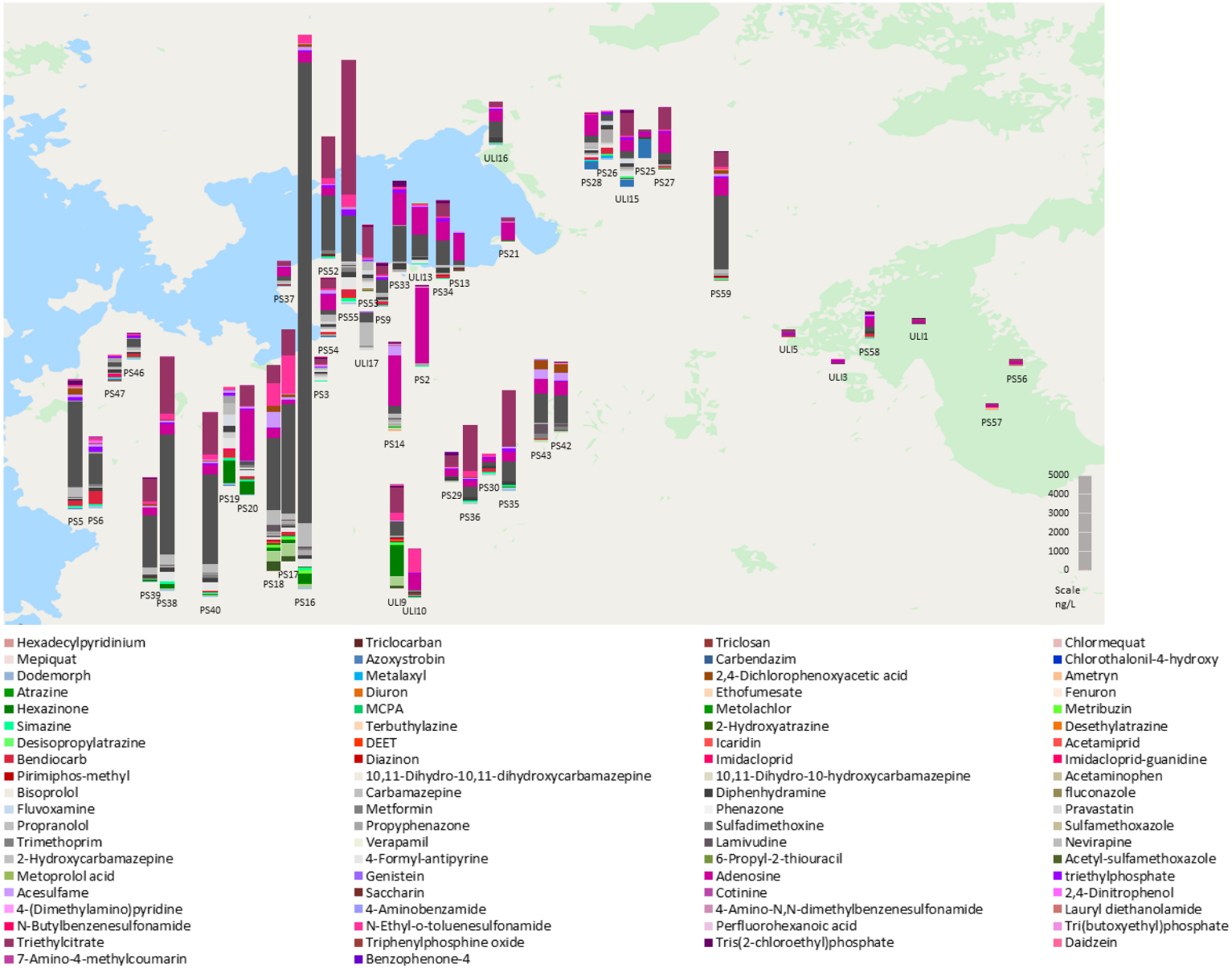


Figure SI-2: Compound concentrations and spatial distribution in water samples

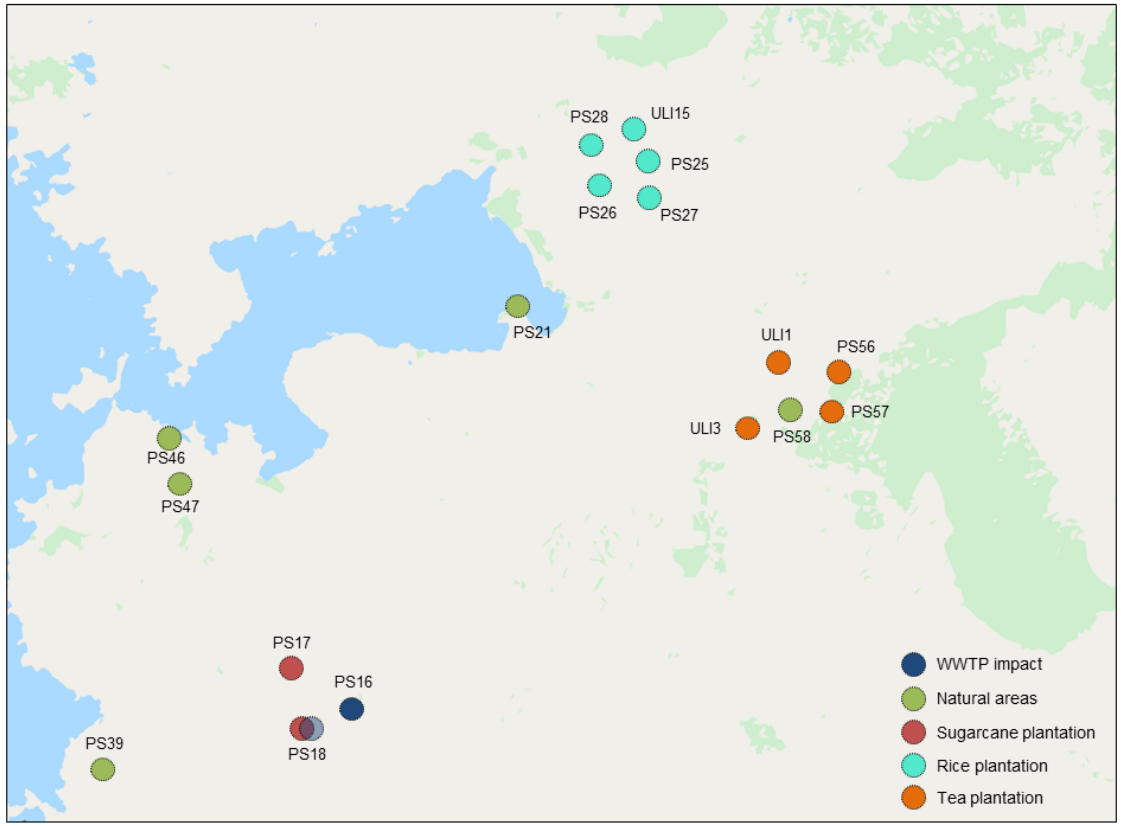


Figure SI-3: Map showing selected sites with dominant land use within the study area

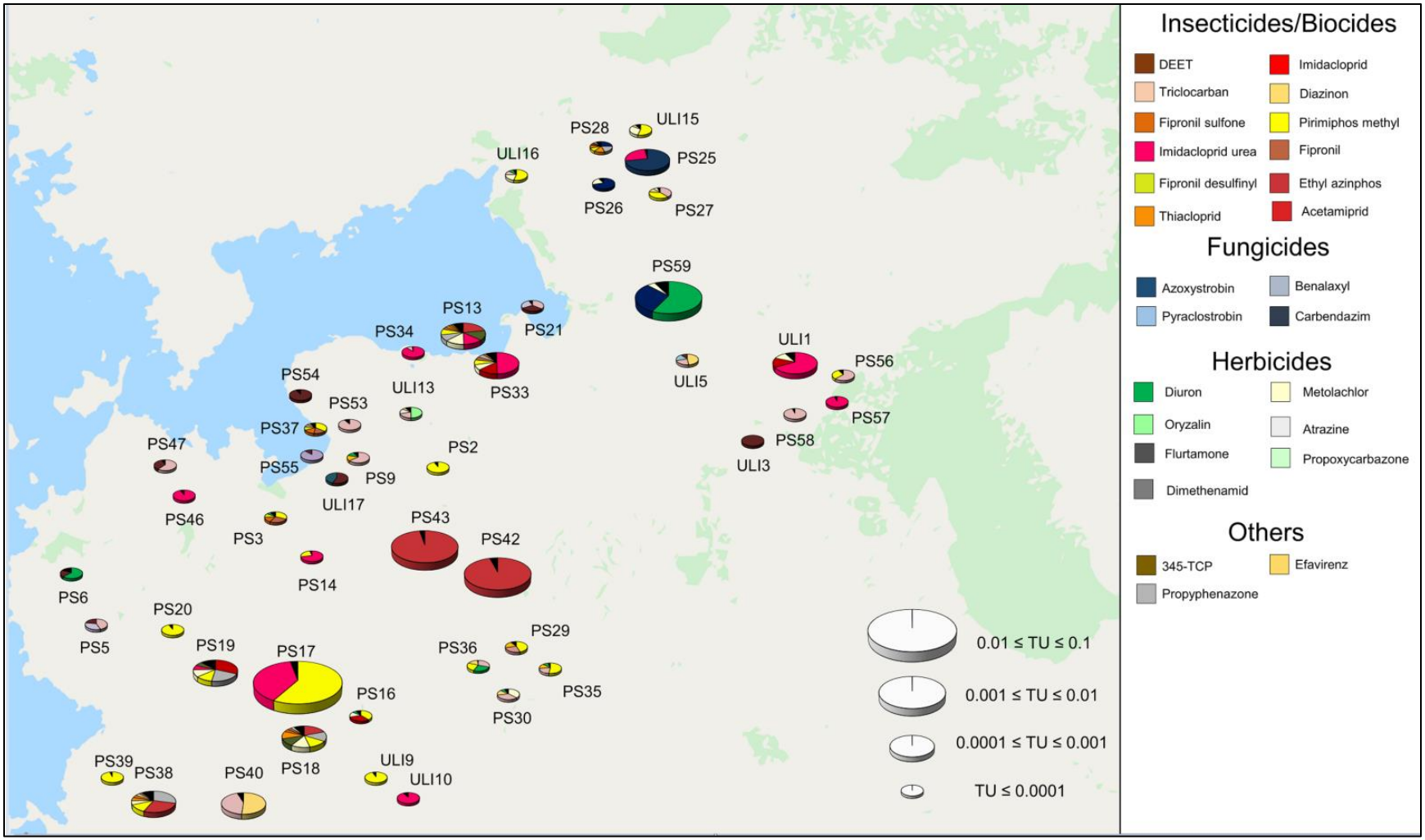


Figure SI-4: Distribution of risk of toxicity for fish from compounds present in sediment extract based equilibrium water concentrations

Table SI-6a,b and c: Toxic unit of compounds in sediment samples calculated for fish, crustaceans, and algae (in Excel file)

Table SI-6 can be found online under Faith Jebiwot Kandie. (2020). Appendix B [Data set]. Zenodo. http://doi.org/10.5281/zenodo.4010840

Appendix C

Table SI-1: Pesticides analyzed in the water samples. The acute LC50 values for *Daphnia magna*, *Chironomus sp.*, *Chironomus riparius* and *Hyalella azteca* were obtained from the PPDB and the ECOTOX data base[31,57]. In the PPDB data base LC50 values for 48 h exposure on *Chironomus* and *D. magna* were compared and the lowest value was used. If no values were available or if the tolerance of *D. magna* was exceptionally high (neonicotinoids), LC50 values for the same species or *H. azteca* were searched in the ECOTOX data base and the median value was used.

Name	CAS Nr.	Application	Class	LC50 [µg/L]	Test species	Duration	Source
2,4-Dichlorophenox. acid	94-75-7	Herbicide	Auxine	11,020.00	*C. sp.*	48 h	ECOTOX
Acetamiprid	135410-20-7	Insecticide	Neonicotinoid	11.56	*C. riparius*	96 h	PPDB
Ametryn	834-12-8	Herbicide	Triazine	28,000.00	*D. magna*	48 h	PPDB
Atrazine	1912-24-9	Herbicide	Triazine	85,000.00	*D. magna*	48 h	PPDB
Azoxystrobin	131860-33-8	Fungicide	Strobilurine	230.00	*D. magna*	48 h	PPDB
Bendiocarb	22781-23-3	Insecticide	Carbamate	30.00	*D. magna*	48 h	PPDB
Carbendazim	10605-21-7	Fungicide	Carbamate	150.00	*D. magna*	48 h	PPDB
Chlormequat	999-81-5	Herbicide	Growth regulator	31,700.00	*D. magna*	48 h	PPDB
Chlorothalonil-4-hydroxy	28343-61-5	Fungicide	Chloronitrile	76,000.00	*C. riparius*	48 h	PPDB
Desethylatrazine	6190-65-4	Metabolite	Triazine	5,100.00	*H. azteca*	96 h	ECOTOX
Desisopropylatrazine	1007-28-9	Metabolite	Triazine	7,200.00	*H. azteca*	96 h	ECOTOX
Diazinon	333-41-5	Insecticide	Organophosphate	1.00	*D. magna*	48 h	PPDB
Diethyltoluamid (DEET)	134-62-3	Repellent	Methylbenzamide	75,000.00	*D. magna*	48 h	PPDB

Diuron	330-54-1	Herbicide	Phenylurea	5,700.00	*D. magna*	48 h	PPDB
Dodemorph	1593-77-7	Fungicide	Morpholine	3,340.00	*D. magna*	48 h	PPDB
Ethofumesate	26225-79-6	Herbicide	Benzofuran	13,520.00	*D. magna*	48 h	PPDB
Fenuron	101-42-8	Herbicide	Acetylurea	502,000.00	*D. magna*	48 h	PPDB
Hexazinone	51235-04-2	Herbicide	Triazine	85,000.00	*D. magna*	48 h	PPDB
Icaridin	119515-38-7	Repellent		100,000.00	*D. magna*	48 h	PPDB
Imidacloprid	138261-41-3	Insecticide	Neonicotinoid	55.00	*C. sp.*	96 h	PPDB
MCPA	94-74-6	Herbicide	Auxine	190,000.00	*D. magna*	48 h	PPDB
Mepiquat	15302-91-7	Herbicide	Piperidine	68,500.00	*D. magna*	48 h	PPDB
Metalaxyl	57837-19-1	Fungicide	Acetylalanine	3,470	*D. magna*	48 h	PPDB
Metolachlor	51218-45-2	Herbicide	Chloroacetanilide	23,500.00	*D. magna*	48 h	PPDB
Metribuzin	21087-64-9	Herbicide	Triazinone	49,000.00	*D. magna*	48 h	PPDB
Pirimiphos-methyl	29232-93-7	Insecticide	Organophosphate	0.21	*D. magna*	48 h	PPDB
Simazine	122-34-9	Herbicide	Triazine	1,100.00	*D. magna*	48 h	PPDB
Terbuthylazine	5915-41-3	Herbicide	Chlorotriazine	21,200.00	*D. magna*	48 h	PPDB

Table SI-2: Pesticide pollution in water samples of the study sites. For each site, the maximum (TU_{max}) and the summed up (TU_{sum}) toxic unit of all measured substances is shown.

Study site	Coordinates	TU_{max}	TU_{sum}	Most toxic substance	Nr. detected	Nr. quantified
1	0°34'43.43"S; 34°36'35.40"E	-3.17	-3.17	Bendiocarb	14	6
2	0°34'53.96"S; 34°32'0.10"E	-4.44	-4.36	Simazine	15	4
3	0°28'29.67"S; 34°32'56.95"E	-2.04	-1.89	Bendiocarb	22	10
4	0°22'48.01"S; 34°38'30.92"E	-1.66	-1.65	Bendiocarb	16	6
5	0°40'50.77"S; 34°32'39.07"E	-2.30	-2.09	Bendiocarb	24	9
6	0°49'46.82"S; 34°23'44.12"E	-2.59	-2.39	Diazinon	17	8
7	0° 8'17.02"S; 34°56'8.82"E	-2.70	-2.63	Diazinon	23	12
8	0° 8'38.67"S; 34°58'20.86"E	-1.86	-1.85	Pirimiphos-meth.	24	12
9	0°10'18.07"S; 34°54'28.44"E	-1.86	-1.63	Pirimiphos-meth.	28	16
10	0°48'19.63"S; 34°41'54.93"E	-1.21	-1.06	Pirimiphos-meth.	27	17
11	0°22'38.69"S; 34°38'6.98"E	-1.90	-1.89	Bendiocarb	20	10
12	0°23'17.85"S; 34°38'30.02"E	-2.38	-2.36	Bendiocarb	21	10
13	0°33'40.22"S; 34°18'9.70"E	-3.85	-3.85	Bendiocarb	16	4
14	0°29'9.67"S; 34°31'2.85"E	-2.14	-2.11	Carbendazim	12	6
15	0°28'34.93"S; 34°32'41.36"E	-2.00	-1.98	Bendiocarb	28	10
16	0°27'5.24"S; 35°7'15.27"E	-2.60	-2.59	Bendiocarb	18	4
17	0° 9'0.91"S; 34°55'49.30"E	-2.58	-2.24	Carbendazim	18	9
18	0° 6'46.67"S; 34°47'29.77"E	-2.54	-2.53	Diazinon	15	4
19	0°28'38.92"S; 34°32'38.87"E	-2.25	-2.09	Bendiocarb	17	8
20	0°48'2.29"S; 34°43'45.10"E	-2.34	-2.34	Diazinon	14	1
21	0°32'20.58"S; 35°2'0.91"E	-2.21	-1.92	Diazinon	16	4
22	0°27'45.4"S; 34°33'55.1"E	-2.26	-2.25	Diazinon	20	9
23	0°59'50.59"S; 34°16'55.57"E	-2.36	-2.34	Diazinon	15	7
24	0°19'20.39"S; 34°47'20.33"E	-3.07	-3.06	Bendiocarb	13	4
25	0°53'54.15"S; 34°31'24.54"E	-2.46	-2.39	Diazinon	23	11
26	0°54'29.02"S; 34°33'28.89"E	-1.79	-1.67	Pirimiphos-meth.	18	10
27	0°28'34.33"S; 34°31'56.51"E	-2.07	-1.83	Pirimiphos-meth.	21	12
28	1° 3'53.69"S; 34°28'5.52"E	-2.33	-2.30	Diazinon	15	6
29	0°48'31.95"S; 34°43'58.93"E	-2.17	-2.10	Diazinon	19	9
30	0°53'8.15"S; 34°31'20.35"E	-2.29	-2.27	Bendiocarb	14	4
31	0°27'34.68"S; 34°35'40.87"E	-3.50	-3.43	Carbendazim	14	5
32	1° 1'18.24"S; 34°37'27.29"E	-2.40	-2.39	Diazinon	17	4
33	0°49'50.42"S; 34°44'44.98"E	-4.09	-3.88	Imidacloprid	19	8
34	0°49'4.79"S; 34°23'34.97"E	-2.55	-2.45	Bendiocarb	21	10
35	0°27'47.58"S; 34°32'55.31"E	-1.89	-1.88	Bendiocarb	20	10
36	0°46'39.89"S; 34°12'21.19"E	-6.44	-6.34	Diethyltoluamid	4	2
37	0°48'47.38"S; 34°13'15.06"E	/	/	Ametryn	6	0
38	0°39'27.61"S; 34°42'37.10"E	-2.26	-2.06	Bendiocarb	16	8
39	0°39'24.17"S; 34°41'57.84"E	-2.06	-1.98	Diazinon	17	10
40	0°30'48.53"S; 34°17'29.08"E	-5.81	-5.80	Simazine	8	2
41	0°23'35.22"S; 35° 0'35.88"E	-4.94	-4.94	Simazine	8	1

42	0°59'40.04"S; 34°17'26.47"E	-3.66	-3.66	Bendiocarb	9	2
43	0°59'40.04"S; 34°17'26.47"E	-1.81	-1.76	Diazinon	23	15
44	0°59'9.30"S; 34°35'5.25"E	-2.54	-2.53	Bendiocarb	20	10
45	0°27'5.46"S; 35°13'8.39"E	-3.93	-3.81	Imidacloprid	10	3
46	0°29'12.72"S; 35°10'58.85"E	-2.59	-2.24	Diazinon	17	9
47	0°30'54.82"S; 35° 4'49.80"E	-2.24	-2.23	Diazinon	12	3
48	0°19'1.31"S; 35° 0'22.66"E	/	/	Acetamiprid	1	0

Table SI-3: Ranking of the analyzed pesticides according to the environmental toxicity observed.

Pesticide	CAS Nr.	Application	Class	Max. TU	Most toxic at nr. of sites	Detected at nr. of sites
Pirimiphos-methyl	29232-93-7	Insecticide	Organophosphate	-1.21	5	6
Bendiocarb	22781-23-3	Insecticide	Carbamate	-1.66	17	31
Diazinon	333-41-5	Insecticide	Organophosphate	-1.71	15	36
Carbendazim	10605-21-7	Insecticide	Carbamate	-2.14	3	46
Acetamiprid	135410-20-7	Insecticide	Neonicotinoid	-2.80	0	39
Imidacloprid	138261-41-3	Insecticide	Neonicotinoid	-3.24	0	16
Simazine	122-34-9	Herbicide	Triazine	-3.83	3	46
2,4-Dichlorophenoxyacetic acid	94-75-7	Herbicide	Auxine	-4.32	0	48
Metalaxyl	57837-19-1	Fungicide	Acetylalanine	-4.40	0	47
Dodemorph	1593-77-7	Fungicide	Morpholine	-4.42	0	40
Ametryn	834-12-8	Herbicide	Triazine	-4.68	0	6
Hexazinone	51235-04-2	Herbicide	Triazine	-4.74	0	15
Atrazine	1912-24-9	Herbicide	Triazine	-4.88	0	45
Azoxystrobin	131860-33-8	Fungicide	Strobilurine	-4.98	0	32
Ethofumesate	26225-79-6	Herbicide	Benzofuran	-5.09	0	31
Metribuzin	21087-64-9	Herbicide	Triazinone	-5.42	0	8
Desethylatrazine	6190-65-4	Metabolite	Triazine	-5.61	0	47
Metolachlor	51218-45-2	Herbicide	Chloroacetanilide	-5.70	0	27
Diuron	330-54-1	Herbicide	Phenylurea	-5.76	0	15
Desisopropylatrazine	1007-28-9	Metabolite	Triazine	-5.77	0	17
Terbuthylazine	5915-41-3	Herbicide	Chlorotriazine	-5.96	0	31
Mepiquat	15302-91-7	Herbicide	Piperidine	-6.17	0	47
Icaridin	119515-38-7	Repellent		-6.17		47
Chlormequat	999-81-5	Herbicide	Growth regulator	-6.43	0	1
Diethyltoluamid (DEET)	134-62-3	Repellent	Methylbenzamide	-6.43	0	48
MCPA	94-74-6	Herbicide	Auxine	-7.15	0	48
Chlorothalonil-4-hydroxy	28343-61-5	Fungicide	Chloronitrile	-7.80	0	13
Fenuron	101-42-8	Herbicide	Acetylurea	-8.38	0	47

Table SI-4: Environmental effects on the incidence of schistosomiasis hosts in surface waters of the study area**.** Each environmental variable was fitted using a one-way binomial GLM with cloglog-link. The unit and the transformation of each environmental variable prior to analysis is given in squared brackets. For numerical variables, model coefficients are reported together with their standard error; n = 48 for each numeric model. For categorical variables, the back-transformed mean of each factor level is reported together with 95 % confidence intervals and the number of observations. χ^2 and p are reported from a likelihood ratio test against the null model without explanatory variables.

Numerical variable	Intercept	Slope	Res. df	X^2	p	
Flow velocity [ln(m/s)]	-1.64 ± 0.63	-0.02 ± 0.17	46	0.02	0.900	
Depth [ln(cm)]	-3.81 ± 2.65	0.57 ± 0.65	46	0.84	0.359	
Temperature [°C]	-1.50 ± 1.87	< -0.01 ± 0.07	46	< 0.01	0.969	
Conductivity [ln(µS/cm)]	-5.47 ± 3.07	0.79 ± 0.60	46	1.90	0.170	
Acidity [pH]	1.65 ± 2.85	-0.44 ± 0.39	46	1.02	0.312	
Dissolved oxygen [ln(mg/L)]	0.47 ± 0.70	-1.60 ± 0.58	46	8.06	0.004	**
Turbidity [ln(NTU)]	-0.92 ± 1.07	-0.13 ± 0.20	46	0.41	0.520	
Carbonate hardness [ln(°dH)]	-3.03 ± 1.16	0.89 ± 0.63	46	2.25	0.134	
NH_4 [mg/L]	-1.51 ± 0.38	-0.88 ± 2.69	46	0.11	0.746	
PO_4 [mg/L]	-1.24 ± 0.73	-0.02 ± 0.04	46	0.23	0.629	
NO_{3^-} [mg/L]	-1.13 ± 0.45	-0.06 ± 0.05	46	1.59	0.207	
NO_{2^-} [ln(mg/L)]	-0.72 ± 1.16	0.26 ± 0.35	46	0.54	0.463	
Species richness) [ln(*n* taxa)]	-5.29 ± 2.11	1.62 ± 0.86	46	4.39	0.036	*
Evenness [J']	-2.07 ± 1.20	0.93 ± 2.14	46	0.16	0.688	
Species diversity [H']	-4.21 ± 1.53	1.64 ± 0.84	46	4.42	0.035	*
SASS 5 [ln(ASPT)]	2.12 ± 3.11	-2.33 ± 1.99	46	1.56	0.211	
Macroinv. abund. [ln(*n* ind.)]	-2.47 ± 2.16	0.20 ± 0.48	46	0.18	0.672	
Dominance predators [%]	-2.05 ± 0.95	0.73 ± 1.31	46	0.33	0.564	
Dominance competitors [%]	-0.36 ± 0.45	-4.58 ± 1.87	46	9.09	0.003	**
Pesticide pollution [TU_{max}]	2.46 ± 1.69	1.72 ± 0.79	46	7.60	0.006	**
Emerged vegetation cover [%]	-1.37 ± 0.36	-1.73 ± 1.84	46	1.33	0.249	
Floating vegetation cover [%]	-1.61 ± 0.35	0.70 ± 1.91	46	0.13	0.716	

Numerical variable	Intercept	Slope	Res. df	X^2	*p*	
Submerged vegetation [%]	-1.69 ± 0.36	5.65 ± 3.95	46	1.26	0.262	
Detritus cover [%]	-1.57 ± 0.36	-0.03 ± 1.68	46	< 0.01	0.984	

Categorical variable		Mean (95 % CI)	*n*	Res. df	X^2	*p*	
Habitat:	Main tributary	0.11 (0.03 – 0.38)	18	42	6.31	0.277	
	Minor tributary	0.40 (0.17 – 0.74)	10				
	Irrigation channel	0.20 (0.03 – 0.79)	5				
	Oxbow lake	< 0.01 (0.00 – 1.00)	3				
	Reservoir	0.25 (0.07 – 0.68)	8				
	Rice field	< 0.01 (0.00 – 1.00)	4				
Land use:	Natural	0.30 (0.11 – 0.67)	10	44	6.37	0.095	·
	Agricultural	0.16 (0.07 – 0.34)	32				
	Semi-urban	< 0.01 (0.00 – 1.00)	5				
	Industrial	1 (0.00 – 1.00)	1				
Farm type:	Natural	0.27 (0 .10– 0.63)	11	43	1.37	0.850	
	Subsistence	0.21 (0.07 – 0.53)	14				
	Agroforestry	< 0.01 (0.00 – 1.00)	1				
	Commercial	0.13 (0.04 – 0.44)	15				
	Irrigation scheme	0.14 (0.02 – 0.67)	7				
Crop type:	Maize	0.15 (0.04 – 0.49)	13	35	3.93	0.415	
	Rice	0.09 (0.01 – 0.49)	11				
	Sugar cane	0.42 (0.16 – 0.83)	7				
	Tea	0.14 (0.02 – 0.67)	7				
	Other	< 0.01 (0.00 – 1.00)	2				

Table SI-5: Environmental effects on the population density of schistosomiasis hosts in surface waters of the study area. Each environmental variable was fitted using a one-way GLM with a zero-truncated negative-binomial distribution and log-link. In the upper part of the table results are reported with all study sites considered where hosts snails had been found; n = 9 and res. df = 7 in all models. Only those environmental variables are presented that showed a (marginally) significant effect ($p < 0.1$); see Tab. S4 for a complete list of environmental variables tested and for the units of measurement. Below, effects of the same variables are shown when site 39 (with extraordinary mass development of host snails) was excluded as a highly influential outlier; here n = 8 and res. df = 6 in all reported models. Model coefficients are reported together with their standard error and McFadden's pseudo-R^2. χ^2 and p values are reported from a likelihood ratio test against the null model without explanatory variables

Environmental variable	Intercept	Slope	Pseudo-R^2	X^2	p	
	For all study sites					
Depth	11.66 ± 3.23	-2.26 ± 0.81	0.09	6.25	0.012	*
Acidity	10.37 ± 1.99	-1.09 ± 0.28	0.14	10.41	0.001	**
Turbidity	12.60 ± 2.95	-2.01 ± 0.60	0.07	5.00	0.025	*
PO_4	4.75 ± 1.09	-0.13 ± 0.06	0.04	2.79	0.095	·
Species richness	-5.98 ± 3.74	3.52 ± 1.49	0.06	4.47	0.034	*
Evenness	5.96 ± 1.33	-6.23 ± 2.41	0.06	4.02	0.045	*
Macroinvertebrate abundance	-3.92 ± 2.36	1.43 ± 0.50	0.11	7.90	0.005	**
Dominance predators	5.49 ± 0.84	-4.39 ± 1.21	0.13	9.08	0.003	**
Dominance competitors	1.52 ± 0.59	6.94 ± 2.68	0.09	6.22	0.013	*
	Without site 39					
Depth	2.93 ± 3.39	-0.19 ± 0.82	< 0.01	0.03	0.871	
Acidity	7.55 ± 6.72	-0.72 ± 0.90	0.01	0.72	0.395	
Turbidity	5.73 ± 1.43	-0.73 ± 0.29	0.09	4.50	0.034	*
PO_4	2.83 ± 0.58	-0.04 ± 0.03	0.03	1.73	0.188	
Species richness	-0.80 ± 2.54	1.22 ± 1.04	0.02	1.16	0.281	
Evenness	3.21 ± 0.86	-1.94 ± 1.50	0.02	1.07	0.302	
Macroinvertebrate abundance	-0.83 ± 1.83	0.68 ± 0.41	0.04	2.27	0.132	

Dominance predators	3.65 ± 1.02	-2.10 ± 1.37	0.05	2.33	0.127
Dominance competitors	2.16 ± 0.51	0.02 ± 3.17	< 0.01	< 0.01	0.996

Table SI-6: Classification of macroinvertebrate taxa in relation to the host snails of schistosomiasis.

Order	Family	Feeding type	Relation to snails
Ephemeroptera	Baetidae	Grazer	Competitor
Ephemeroptera	Caenidae	Grazer	Competitor
Ephemeroptera	Ephemeridae	Grazer	Competitor
Ephemeroptera	Heptageniidae	Grazer	Competitor
Ephemeroptera	Leptophlebiidae	Grazer	Competitor
Ephemeroptera	Oligoneuridae	Grazer	Competitor
Ephemeroptera	Polymitarcyidae	Grazer	Competitor
Ephemeroptera	Prosopistomatidae	Grazer	Competitor
Ephemeroptera	Teloganodidae	Grazer	Competitor
Plecoptera	Perlidae	Predator	Predator
Trichoptera	Ecnomidae	Predator, Grazer	Predator, Competitor
Trichoptera	Hydropsychidae	Filterer	Other
Trichoptera	Pisuliidae	Grazer	Competitor
Trichoptera	Polycentropodidae	Filterer	Other
Zygoptera	Chlorocyphidae	Predator	Predator
Zygoptera	Chlorolestidae	Predator	Predator
Zygoptera	Coenagriidae	Predator	Predator
Zygoptera	Lestidae	Predator	Predator
Anisoptera	Aeshnidae	Predator	Predator
Anisoptera	Corduliidae	Predator	Predator
Anisoptera	Gomphidae	Predator	Predator
Anisoptera	Libellulidae	Predator	Predator
Heteroptera	Belostomatidae	Predator	Predator
Heteroptera	Corixidae	Predator, Grazer	Predator, Competitor
Heteroptera	Gerridae	Predator	Predator
Heteroptera	Hydrometridae	Predator	Predator
Heteroptera	Naucoridae	Predator	Predator
Heteroptera	Nepidae	Predator	Predator
Heteroptera	Notonectidae	Predator	Predator
Heteroptera	Veliidae	Predator	Predator
Coleoptera	Dytiscidae	Predator	Predator
Coleoptera	Elmidae / Dryopidae	Grazer	Competitor
Coleoptera	Gyrinidae	Predator	Predator
Coleoptera	Haliplidae	Predator, Plant sucker	Predator, Other
Coleoptera	Hydrophilidae	Predator	Predator
Coleoptera	Noteridae	Predator	Predator

Coleoptera	Scirtidae	Predator	Predator
Diptera	Chironomidae	Filterer, Grazer	Other
Diptera	Culicidae*	Filterer, Grazer	Competitor, Other
Diptera	Muscidae	Predator	Predator
Diptera	Simulidae	Filterer	Other
Diptera	Tipulidae	Shredder	Other
Lepidoptera	Pyralidae	Herbivor	Competitor
Crustacea	Atyidae	Grazer, Filterer	Competitor
Gastropoda	Ampulariidae	Grazer, Herbivor, Predator	Snail
Gastropoda	Ancylidae	Grazer	Snail
Gastropoda	Hydrobiidae	Grazer	Snail
Gastropoda	Lymnaeidae	Grazer, Herbivor	Snail
Gastropoda	Physidae	Grazer	Snail
Gastropoda	Planorbidae	Grazer, Herbivor	Snail
Gastropoda	Thiaridae	Grazer	Snail
Gastropoda	Viviparidae	Grazer	Snail
Annelida	Hirudinea	Predator	Predator
Annelida	Oligochaeta	Detritivor	Other

Curriculum vitae

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18.08.1989 Uasin Gishu, Kenya

Employment

Since 08/2017 **PhD student**
International Centre of Insect Physiology and Ecology
Nairobi Kenya
Department of Human Health

Helmholtz Centre for Environmental Research
Leipzig Germany
Department of Effect-Directed Analysis

Since 08/2014 **Graduate Assistant**
Moi University, School of biological and physical sciences
Eldoret, Kenya
Department of Biological Sciences

03/2013 to 07/2014 **Environmental Consultant and EIA Expert**
Safe Environment Consultancy Limited
Nairobi, Kenya

09/2012 to 02/2013 **Intern Officer**
National Environment Management Authority – Uasin Gishu County.
Eldoret, Kenya

01/2012 to 08/2012 **Research Assistant**
University of Eldoret (UoE)- Commission of Higher Education funded project on urban wastewater reuse
Eldoret, Kenya.

Education

Since 02/2018 **PhD student**
Helmholtz Centre for Environmental Research, Leipzig, Germany
Department of Effect-Directed Analysis

International Centre of Insect Physiology and Ecology
Department of Human Health

Goethe University Frankfurt
Faculty of Biosciences

Title: “Analytical screening of organic chemicals of emerging concern in western Kenya and their contribution to the prevalence of schistosomiasis”

08/2014 to 09/2018 **Master student**

Ghent University, Gent, Belgium
Faculty of Bioscience Engineering: Masters in Environmental Sanitation
Final grade: Magna cum Laude (with great distinction)

Title: “Occurrence and removal of pharmaceutical residues in Kenyan wastewater treatment plants”

08/2014 to 09/2016 **Diploma student**
Premese Africa Development Institute
Community Health and Development-Disaster management
Final grade: Credit

08/2008 to 12/2012 **Bachelor student**
Moi University, Eldoret, Kenya
School of Environmental studies: Bachelor of Environmental studies-Environmental Biology and Health major
Final grade: First class honors

02/2004 to 11/2007 **Kenya Certificate of Secondary Education**
Loreto High School-Matunda, Eldoret, Kenya
Final grade: B

02/1996 to 11/2003 **Kenya Certificate of Primary Education**
Moi University Primary School- Eldoret, Kenya
Final grade: 379/500

Awards and scholarships

2020 Second Phase funding for the SENTINEL project from DFG (German Research Foundation) for three years

2017	Scholarship holder from DFG (German Research Foundation) for three years at the Helmholtz Centre for Environmental Research, Leipzig, Germany and International Centre of Insect Physiology and Ecology
2016	TNAV Thesis Award for the best thesis on water technology, Belgium
2014 to 2016	VLIR-UOS Scholarship holder for two years at Ghent University, Belgium.

Scientific contributions

Publications:

*Kandie FJ, Krauss M, Beckers L-M, Massei R, Fillinger U, Becker J, et al. Occurrence and risk assessment of organic micropollutants in freshwater systems within the Lake Victoria South Basin, Kenya. *Science of the Total Environment* 2020; 714: 136748.

*Becker J.M., Ganatra A.A, Kandie F., Mühlbauer L., Ahlheim J., Brack W., Torto B., Agola E., McOdimba F., Hollert H., Fillinger U., Liess M. Pesticide pollution in freshwater paves the way for schistosomiasis transmission. *Scientific Reports* 10, no. 1 2020: 1-13.

* Kandie J. F, Krauss M, Massei R, Ganatra A, Fillinger U, Becker J, Liess M, Torto B, Brack W. Multi-compartment chemical characterization and risk assessment of chemicals of emerging concern in freshwater systems of western Kenya. *Environmental Science Europe* 2020.

K'oreje, K. O., Kandie, F. J., Vergeynst, L., Abira, M. A., Van Langenhove, H., Okoth, M., & Demeestere, K. (2018). Occurrence, fate and removal of pharmaceuticals, personal care products and pesticides in wastewater stabilization ponds and receiving rivers in the Nzoia Basin, Kenya. *Science of the Total Environment*, *637–638*, 336–348.

*Publication which is part of this dissertation.

Platform presentations:

Faith Kandie , Riccardo Massei, Ulrike Fillinger, Akbar Ganatra, Jeremias Becker, Mathias Liess, Baldwyn Torto, Martin Krauss, Werner Brack. Occurrence, risk assessment and prioritization of organic micropollutants: A multi-compartment analysis of freshwater systems in western Kenya. *SETAC SciCon* Europe, 2020

Faith Kandie, Riccardo Massei, Martin Krauss, Baldwyn Torto, Werner Brack. Effect-Directed analyses (EDA) of water extract from western Kenya: the role and effect of local

organic micropollutants on *Schistosoma* snail predators. *International conference on neglected tropical disease (IncoNTD)*. 2019, Nairobi

Faith Kandie Liza-Marie Becker, Riccardo Massei, Ulrike Fillinger, Akbar Ganatra, Jeremias Becker, Mathias Liess, Baldwyn Torto, Martin Krauss, Werner Brack. Occurrence and distribution of organic micropollutants in water, sediments and snail tissues in freshwater systems of western Kenya. *SETAC* Europe 2019, Helsinki

A Ganatra, J Becker, F Kandie, L Muehlbauer, F McOdimba, B Torto, E Lelo, W Brack, U Fillinger, M Liess. Agrochemical pollution increases abundance of schistosoma host snails in western Kenya. *Transactions of the Royal Society of Tropical Medicine and Hygiene,* 2019 Liverpool

Faith Kandie, Werner Brack, Baldwyn Torto. Neonicotinoids in Freshwater systems of Western Kenya. Academy of Science of South Africa (ASSAF) meeting on Neonicotinoids Ecosystem Services for Agriculture and Biodiversity in Africa. 2019 Nairobi

Underlined are the presenting authors.