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Agrochemicals in freshwater systems and their potential as endocrine disrupting chemicals: A South African context

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1. Introduction

South Africa’s semi-arid climate makes it ideal for the cultivation of various crops (Quinn et al., 2011; Yahaya et al., 2017), including maize, soybean, sunflower, wheat, sugar cane and a variety of citrus fruits. Approximately 15 million hectares of South Africa’s land is used for cultivation, accounting for 12–13% of the country’s total area of 122 million hectares (Van der Laan et al., 2017). South Africa has an estimated poverty rate of 28%, which is equivalent to more than 16 million individuals living in extreme poverty (World Poverty Clock, 2020). As a result, the country’s emerging economy is highly dependent on agriculture in terms of job creation and poverty alleviation (Olujimi et al., 2010). However, crop production in South Africa is not considered sustainable because it comes at an ecological cost due to the vast amount of agrochemicals applied (Van der Laan et al., 2017; Yahaya et al., 2017). In the 1970s, Africa’s pesticide load significantly increased when large quantities of dichlorodiphenyltrichloroethane (DDT), malathion and many other pesticides were “donated” to African countries for malaria vector control, following their ban in developed countries (Osibanjo et al., 2002; Manyilizu, 2019). Apart from disease vector control, the strive for food security has caused increased use of agrochemicals to reduce crop losses associated with a variety of pests (Archer et al., 2020). To date, South Africa is the largest pesticide user in sub-Saharan Africa (Ansara-Ross et al., 2012), with over 3000 pesticide products registered for use (Van der Laan et al., 2017), even though this is not internationally recognised: South Africa did not even feature in a publication by Sharma et al. (2019) who reviewed global pesticide usage and its effect on ecosystems.

South Africa’s subtropical location, warm temperate conditions and variable rainfall (~464 mm annually) makes the country especially vulnerable in terms of water availability (South African...
Government, 2020). Therefore, any form of water pollution puts additional pressure on this limited resource. Although pesticides benefit food production (De Souza et al., 2020) many agrochemicals reach and pollute non-target aquatic environments, posing a threat to human health and wildlife species. The National Water Act of South Africa (36:1998) (Department of Water Affairs and Forestry, 1998) forms the basis of protecting water bodies in South Africa against environmental pollution (Koreje et al., 2020). Nevertheless, the presence of agrochemicals in many South African rivers has been well-documented (e.g. the uMgeni River (Gakuba et al., 2018), Olifants River (Verhaert et al., 2017), Buffalo River (Yahaya et al., 2017), Luuvhuu and Letaba rivers (Gerber et al., 2015), Vaal River (Quinn et al., 2009), Lourens River (Bollomoh et al., 2007) and the Msunduzi River (Adeyinka et al., 2019)). Yet, there is little research regarding the ecological impact of many of the registered products in the environment (Van der Laan et al., 2017). This is worrisome as it is estimated that only 0.1% of pesticides applied reach target organisms (Pimentel and Levin, 1986; Pimentel, 1995; Gill and Garg, 2014). The remainder is free to move into non-target areas contaminating soil and watercourses (rivers, lakes, estuaries, dams, streams, and groundwater systems) through spray drift, leaching, and surface run-off (De Souza et al., 2020). In South Africa, the leaching of non-target agrochemicals to non-target aquatic environments via spray drift has been confirmed by Dalvie et al. (2014), who reported on the migration of pesticide residues towards pre- and primary schools bordering vineyards of a wine farm in the Western Cape province. Pesticides also end up in the aquatic environment when aquatic nuisance plants are sprayed (Mensah et al., 2012; Bornman et al., 2017). Other pesticide routes into the environment include effluent from wastewater treatment plants (WWTPs) (Van Zijl et al., 2017), leachates from municipal landfills (Ojemaye and Petrik, 2019), run-off and leaching from contaminated soils, and the careless disposal of pesticide containers (Olisah et al., 2020).

As a consequence of freshwater pollution, aquatic species are exposed to these pollutants through their food and surface areas (skin and/or gills) (Olujimi et al., 2010; Mahmood et al., 2016). This allows researchers to use many aquatic species including freshwater shrimp (Caridina nilotica) (Mensah et al., 2012), African sharptooth catfish (Clarias gariepinus) (Pheeffer et al., 2018), Mozambique tilapia (Oreochromis mossambicus) (Truter et al., 2016) and African clawed frog (Xenopus laevis) (Van Wyk et al., 2003; Viljoen et al., 2016) as indicators of effects of agrochemical pollution. Since pesticides are chronically present in the environment at low concentrations, humans and wildlife are consistently exposed (Damalas and Eleftherohorinos, 2011). Such chronic (or semi-chronic) exposure may lead to toxic effects.

One of the toxicological effects associated with chemical pollution of water is endocrine disruption (ED) where some compounds may interfere with the endocrine system. Compounds that include agrochemicals that cause ED effects are collectively known as endocrine disruptive compounds or chemicals (EDCs) (Mnif et al., 2011; Combarnous, 2017; Warner et al., 2020). The American Endocrine Society defines EDCs as environmental contaminants that can disturb or disrupt the normal functioning of the hormone system (Gore et al., 2015). Since the endocrine system plays a fundamental role in maintaining important physiological processes, EDCs can interfere with normal hormone function, changing the physiology of an organism and disrupting homeostasis (Gore et al., 2015) through various mechanisms of action. This can increase an individual’s disease susceptibility (Bornman et al., 2017) and lead to various adverse health effects. (Karthikeyan et al., 2019) such as diabetes in humans (Han et al., 2020), or respiratory problems and altered reproductive development (Jacobsen et al., 2012; Teng et al., 2020), and neurodegenerative disorders (Kass et al., 2020) in animals.

Although many studies quantified agrochemicals in South African water (Amdany et al., 2014; Odendaal et al., 2015; Wooding et al., 2017; Adeyinka et al., 2019; Ojemaye et al., 2020), few studies have also investigated the associated ED effects (Barnhoorn et al., 2010; Arukwe et al., 2016). Most studies that evaluated ED of agrochemicals in South Africa were laboratory-based using in vitro or in vivo bioassays of single active ingredients or pesticide mixtures (Archer and Van Wyk, 2015; Rimayi et al., 2018b). However, since only a few studies combined bioassays with chemical analysis of agrochemicals with their associated ED effects, we only have glimpses of the risks posed by current-use agrochemicals in South Africa.

The purpose of this review is to discuss the occurrence of agrochemicals in South African freshwater systems and their endocrine disruptive role based on peer-reviewed publications since 2011. A critical overview of the extent of pesticide pollution in our water and wildlife is given, with a specific focus on the most thoroughly investigated agrochemicals in South Africa: legacy organochlorine pesticides (OCPs) and current-use triazine herbicides. A summary is also provided on the presence of glyphosate-based herbicides (GBH), 2,4-dichlorophenoxyacetic acid (2,4-D) and chlorpyrifos in the aquatic environment. Lastly, the ED effects reported on in South African freshwater environments, including oestrogenic activity and intersex, are summarised.

2. Agrochemicals in South Africa

An agrochemical (agricultural chemical, agrichemical) refers to any chemical product used in agriculture and includes several classes such as antifeedants, antibiotics used in animal husbandry, bird repellents, chemosterilants, hormones, mating disruptors, plant activators, plant growth regulators, fumigants, pesticides, synthetic fertilisers and seed coatings (Speight et al., 2017; Ravichandra, 2018). Agricultural pesticides are the most widely applied class of agrochemicals and are used to prevent, control, repel, or kill any pest that interferes with human activities (related to food production) or animals (Department of Agriculture, Forestry and Fisheries, 2010; Manyilizu, 2019; Kass et al., 2020). They can be classified into different categories based on their target pests namely, acaricides, bactericides, fungicides, herbicides, insecticides, nematicides, molluscsicides, rodenticides, and viricides to name a selection (Speight et al., 2017; Kass et al., 2020).

Pesticide use dates back more than 4500 years ago when elemental sulphur dust was applied to crops in Mesopotamia (Hayes and Hansen, 2017; Manyilizu, 2019). During the 17th century, sulphur was used to fumigate insects and larvae, with tobacco being added to the mixture in the late 18th century (Smith and Secoy, 1976). It was the use of these inorganic substances which eventually led to the formulation of pesticides more than a century ago (Dallas and Day, 2004). These commercial pesticides not only contain the active ingredient (Mesnage et al., 2019), but are complex formulations that contain chemical additives or "inert" compounds (i.e. co-formulants, adjuvants, surfactants and stabilisers). The additives and inert compounds normally constitute 90–95% of the commercial formulation and in some cases may be more toxic than the active ingredient itself (Mefftaul et al., 2020). These additives increase the active compound’s efficacy (Mesnage et al., 2019), solubility (Mesnage et al., 2014) and stability (Mesnage and Antoniou, 2018). Although subject to acute toxicity testing, certain co-formulants are exempt from the requirement of chronic toxicity testing due to their “inert” designation (Mesnage and Antoniou, 2018; Mesnage et al., 2019). Unfortunately, as they are not regulated in the same manner as the active ingredient and/or complete commercial pesticide formulation, their toxic properties are generally overlooked (Mesnage and Antoniou, 2018).
In South Africa, the commercial sale of pesticides was first permitted after the Fertilisers, Farm Foods, Seeds and Pest Remedies Act was passed in 1917. Thirty years later the Act was replaced by the Fertilisers, Farm Feeds, Agricultural Remedies and Stock Remedies Act of South Africa (Act 36 of 1947) (Wiese and Bot, 1971), which is still in use today. When Act 36 of 1947 was revised in 2010 several new objectives were suggested, including improving South Africa’s legislation in terms of risks posed by pesticides, the development of alternative products (as opposed to agrochemicals), the integration of international agreements and to increase the transparency and public participation of pesticide registration. South Africa currently has water quality guidelines for the protection of aquatic environments for the following pesticides: atrazine (10 μg/L), endosulfan (0.01 μg/L), aldrin (10 μg/L), chlordane (25 μg/L), DDT (1.5 μg/L), dieldrin (5 μg/L), endrin (2 μg/L), lindane (15 μg/L), methoxychlor (20 μg/L) and mirex (1 μg/L) (Department of Water Affairs and Forestry, 1996; London et al., 2005). However, many of South Africa’s 3000 registered pesticide products with 500 active ingredients have not been re-evaluated since they were first approved for use (Department of Agriculture, Forestry and Fisheries, 2010). This highlights the absence of a regulatory framework for pesticide pollution in South African water (London et al. 2005). Moreover, although Dabrowski (2015) developed pesticide-use maps for South Africa, these maps only provide information about 206 active ingredients currently used in the country’s agricultural sector—only 41% of the 500 registered active ingredients (Horn et al., 2019). Unfortunately, knowledge gaps about the majority of current-use pesticides in South Africa, as well as the country’s poorly developed pesticide monitoring and control infrastructure (Ansara-Ross et al., 2012), make assessing their ecotoxicological risks challenging.

When Ansara-Ross et al. (2012) extensively reviewed the presence of pesticides in South African freshwater systems, fewer than 50 studies had been undertaken over a 34-year period (1977–2011). Since then, the number of studies investigating the occurrence of agrochemicals and/or endocrine disrupting effects in aquatic environments in South Africa has increased, with 41 publications in less than a decade (2011–2020) (Table 1). Of these studies, 66% solely investigated the presence of agrochemicals, 22% determined both the levels of pesticides present and associated ED effects, while the remaining 12% only evaluated ED effects. However, even though there has been an increase, many of these studies still focused on legacy OCPs, such as DDT, chlordane and γ-hexachlorocyclohexane (γ-HCH; also known as lindane), which have been banned for agricultural use throughout the world (including South Africa) (Bouwman et al., 2004; Pheiffer et al., 2018). Only a few studies have investigated the presence of currently used pesticides in South African freshwater (Curched et al., 2020). Of the 36 studies investigating the presence of agricultural pesticides, 75% were predominantly focused on legacy OCPs and only 25% on current-use pesticides (mostly triazine herbicides). Although the authors have primarily focused on South African freshwater systems, all of the country’s rivers flow out into the sea. As a result, the occurrence of agrochemicals in coastal regions and/or marine biota are included to illustrate the extent of agrochemical pollution in South Africa.

3. The presence of (legacy) organochlorine pesticides

Most OCPs have been extensively applied in the agriculture sector since the 1940s in South Africa (Araki et al., 2018) and are known EDCs (Oliash et al., 2019; Martyniuk, 2020). In 2004 South Africa became party to the Stockholm Convention on Persistent Organic Pollutants and banned the unregulated use of OCPs, although regulated use of DDT as a vector control for the eradication of malaria is still permitted (Bouwman et al., 2004, 2019; Buah-Kwofe and Humphries, 2017; Yahaya et al., 2017; Pheiffer et al., 2018). Organochlorine pesticides were the most widely applied pesticides and can be divided into three main classes: dichlorodiphenyletheranes (DDT and its metabolites, dicofol, perthane, metholachlor), chlorinated cyclodiene (aldrin, dieldrin, endrin, chlordane, endosulfan, heptachlor), chlorinated benzenes and cyclohexanes (chlordecone, hexachlorocyclohexane (HCH), hexachlorobenzene (HCB), mirex, and toxaphene) (Taiwo, 2019). These pesticides are highly lipophilic, not easily degraded by microorganisms and characterised by long half-lives (from 60 days up to 15 years) due to their high accumulation and slow degradation potential (Schlenk et al., 2005; Jayaraj et al., 2016; Adeyinka et al., 2019). This directly contributes to the environmental persistence and the presence of OCPs (predominantly DDT) have been well-documented in wildlife across South Africa (Table 1).

Fish and other aquatic species are especially vulnerable since they are directly exposed to OCPs through water and their diets, allowing for the concentration of pesticides in their tissue (Taiwo, 2019). Barnhoorn et al. (2015) evaluated the presence of OCPs in C. gariepinus inhabiting three severely polluted impoundments, the Rooihoos-, Rietvei- and Hartbeespoort reservoirs (Table 1). Across all three sampling sites, 52 of the 60 fish sampled contained OCPs. Firth et al. (2019), on the other hand, detected eight OCPs in two mussel species, the black mussel (Choromytilus meridionalis) and the Mediterranean mussel (Mytilus galloprovincialis), farmed at aquaculture facilities in Saldanha Bay on the south coast of South Africa (Table 1). The presence of agrochemicals in fish, results in bioaccumulation to higher trophic levels (Buah-Kwofe et al., 2018; Islam et al., 2018), including apex predators such as the great white shark (Carcharodon carcharias) most frequently found in the inshore regions of False Bay and Mossel Bay on the south coast and KwaZulu-Natal (east coast) (Kock et al., 2013). Although the South African government announced regulations in 1991 for protection against overexploitation by fisheries (Compagn, 1991), sharks are now facing threats of another anthropogenic origin: agrochemicals. In a study conducted by Schlenk et al. (2005), all OCPs investigated (HCB, lindane, heptachlor, DDT and DDE), except for α,α’-DDT, were detected in the muscle and liver tissue samples.

The country is known for its rich biodiversity, containing 7% of the world’s bird, mammal and reptile species; 8% of its plant species and 15% of its coastal marine species (Cherry, 2005). The country is also home to ten World Heritage Sites, including the iSimangaliso Wetland Park on the east coast of South Africa (Table 1). Yet, OCPs have been found here (Humphries, 2013; Buah-Kwofe and Humphries, 2017). Humphries (2013) determined DDT residue concentrations in sediment samples of Lake Sibaya, Southern Africa’s largest natural freshwater lake and a constituent of the iSimangaliso Wetland Park World Heritage Site. The majority of the total DDT concentrations in sediment samples exceeded Canadian sediment quality guidelines. A few years later, sediment samples were collected from Lake Sibaya and three additional main wetland systems within the park (Lake St Lucia, Mkuuze, and Kosi Bay) (Buah-Kwofe and Humphries, 2017). The OCPs were widely distributed throughout the park’s sediment and when published (2017), these concentrations were not only the highest recorded in South Africa, but some of the highest reported globally.

Apart from the presence of OCPs in the sediment of South African biodiversity hotspots these contaminants have also been documented in fish inhabiting other ecologically important regions. Organochlorine pesticides were found in fish from South Africa’s largest floodplain, the Pongola River Floodplain (Table 1) (Volschenk et al., 2019). The region comprises no less than 90 floodplain-associated pans and due to its rich biodiversity, is of high ecological and socio-economic importance (Dube et al., 2017).
Table 1
Studies which investigated agrochemicals and/or endocrine disrupting effects in freshwater aquatic environments across South Africa for the period of 2011–2020.

| Sampling year | Location | Sample matrix | Were agrochemicals investigated? | Agrochemicals found to be present in samples | Were endocrine disrupting effects investigated? | Reference |
|---------------|----------|---------------|---------------------------------|---------------------------------------------|-----------------------------------------------|-----------|
| 1 2006/2007   | Eerste River, Western Cape | Surface water & sewage effluent | No                              | Not applicable                              | Yes: Oestrogenic activity                      | Swart et al. (2011) |
| 2 2007        | Roodeplaat Dam and Retiefie Dam, Gauteng, and Hartbeespoort Dam, North West | Fish tissue | Yes                             | p,p’-DDD, p,p’-DDE, p,p’- & o,p’-DDT; aldrin; dieldrin; endosulfan I & II; endrin; endrin aldehyde; heptachlor; α-, β-, γ- & γ-HCH (indane); methoxychlor | No | Barnhoorn et al. (2015) |
| 3 2009        | Kruger National Park | Crocodile eggs | Yes                             | p,p’-DDE, p,p’-DDD, p,p’-DDT, β- HCH, HCB, oxychlordane; mirex; trans-nonachlor | Yes: Eggshell thinning | Bouwer et al. (2014) |
| 4 2009        | Vhembe District, Limpopo | Frog tissue | Yes                             | p,p’- & o,p’-DDE, p,p’-DDE, o,p’-DDD | Yes: Asymmetric testicular morphology | Viljoen et al. (2016) |
| 5 2009        | Vaal River, Gauteng | Surface water; sediment & fish tissue | Yes                             | HCBs, DDTs; oxychlordane; trans- & cis-nonachlor; HCHs | No | Wegener et al. (2011) |
| 6 2009/2010   | Thohoyandou, Limpopo | Aquatic & terrestrial bird eggs | Yes                             | HCB, β-, γ- & γ-HCH, oxychlordane; cis-chlordane; trans-nonachlor; mirex; p,p’- & o,p’-DDT, p,p’- & o,p’-DDE | Yes: Eggshell thinning | Bouwer et al. (2013) |
| 7 2009/2010; 2013/2014 | Crocodile River, North West and Limpopo | Surface water & crocodile tissue | Yes                             | Atazine, γ-HCH, metazachlor, metalchlor, promethrin, propazine; simazine; terbuthlazine, α-, β-, & δ-HCH | Yes: Aromatase activity; 17β-oestradiol, testosterone and 11α-ketotestosterone levels | Ayuke et al. (2016) |
| 8 2009–2011   | Olifants River, Letaba River and Luvuvhu River, Limpopo | Sediment | Yes                             | α-, β-, γ- & γ-HCH, heptachlor; heptachlor epoxide; oxychlordane; cis- & trans-chlordane; cis- & trans-nonachlor; aldrin; endrin; dieldrin; HCB, p,p’- & o,p’-DDE, p,p’- & o,p’-DDD, p,p’-DDT | No | Gerber et al. (2015) |
| 9 Unknown     | Luvuvhu River, Limpopo | Surface water; sediment & fish tissue | Yes                             | p,p’- & o,p’-DDE, p,p’- & o,p’-DDD, p,p’- & o,p’-DDE, γ-HCH, dieldrin; endrin aldehyde; endosulfan I; endosulfan sulfate | Yes: Ethyl-oestradiol, oestra- didol, oestraone and oestral levels in surface water; gonadal abnormalities; retines | Barnhoorn et al. (2010) |
| 10 2011       | Olifants River, Mpumalanga | Surface water | No                              | Not applicable                              | Yes: 17β-oestradiol and 17α ethynil-oestra diol levels: (anti-oestrogenic activity; (anti-oestrogenic activity) | Truter et al. (2015) |
| 11 2011/2012  | Hartbeespoort Dam, North West | Surface water | Yes                             | HCB, α-, β-, γ- & γ-HCH; p,p’- & o,p’-DDT; o,p’- & p,p’-DDT | No | Amdary et al. (2014) |
| 12 2011/2012  | Robben Island, Atlantic Ocean and Bird Island, Indian Ocean | African penguin eggs | Yes                             | HCB, α-, β-, γ-HCH; oxychlordane; trans-nonachlor; mirex; p,p’-DDE; p,p’-DDD, p,p’-DDT | Yes: Eggshell thinning | *Bouwer et al. (2015) |
| 13 2012       | Lake Sibaya, KwaZulu-Natal | Surface sediment | Yes                             | p,p’-DDE, p,p’-DDD, p,p’-DDT | No | Humphries (2013) |
| 14 2012       | Cape Town, Western Cape; Port Elizabeth, Eastern Cape; Durban; and Pietermaritzburg, KwaZulu-Natal and Drinking water | Yes                             | p,p’-DDE, p,p’-DDD, p,p’-DDT | No | Odendaal et al. (2015) |
| Location | Year | Pollutants | Tissues | Concentrations | Remarks |
|----------|------|------------|---------|----------------|---------|
| Pretoria, Gauteng; and Bloemfontein, Free-State | 2012 | Nelson Mandela municipality, Eastern Cape; eThekwini municipality, KwaZulu-Natal; and City of Cape Town municipality, Western Cape | Surface water | No | Not applicable | Yes: (anti-)oestrogenic activity; (anti-)androgenic activity; 17β-oestradiol levels |
| Offents River Basin, Limpopo | 2012 | Surface water; sediment; invertebrates & fish tissue | Yes | p,p'-DDE; p,p'-o,p'-DDT; o,p'- & p,p'-DDT; α, β & γ-HCH; HCB; cis- & trans-chlordane | Verhaert et al. (2017) |
| Durban, KwaZulu-Natal | 2012 | Sediment | Yes | DDT & its metabolites; endosulfan I & II; endosulfan sulphate; α & γ-chlordane; cis- & trans-chlordane; heptachlor; heptachlor epoxide; methoxychlor | Vogt et al. (2018) |
| Ndumo Game Reserve, KwaZulu-Natal | 2012/2013 | Fish tissue | Yes | α, β & γ-HCH; trans-nonachlor; trans-chlordane; HCBs; p,p'- & o,p'-DDE; p,p'- & o,p'-DDT; p,p'- & o,p'-DDT | No |
| Pongola River Floodplain, KwaZulu-Natal | 2012/2013, 2013/2014 | Frog tissue | Yes | Aldrin; trans-heptachlor epoxide; cis-chlordane; p,p'-DDE; p,p'-DDT; p,p'- & o,p'-DDT; α, β, β & γ-HCH | No |
| Roodeplaat Dam, Gauteng | 2013 | Water & fish tissue | Yes | *Dieldrin; p,p'-DDE; p,p',o,p'-DDT; cis & trans-chlordane; *chlorpyrifos; *chlorpyrifos-methyl; *terbutylazine; *dieldrin; *heptachlor epoxide | No |
| Msunduzi River, KwaZulu-Natal | 2013 | Surface water; soil & sediment | Yes | Mirex; p,p'- & o,p'-DDE; p,p'- & o,p'-DDT; dieldrin; heptachlor; HCH; aldrin; endrin; HCB | No |
| Northern KwaZulu-Natal | 2013 | Aquatic bird eggs | Yes | p,p'-DDE; p,p'- & o,p'-DDT | Yes: Eggshell thinning |
| Fleurhof Dam, Lenasia Dam and Orlando Dam, Gauteng | 2013 | Fish tissue | Yes | HCB; α & γ-HCH; trans-heptachlor epoxide; cis- & trans-nonachlor; cis- & trans-nonachlor; dieldrin; p,p'-DDE; p,p'- & o,p'-DDT; p,p'- & o,p'-DDT | No |
| Unknown Rietvlei Nature Reserve, Gauteng | 2014/2015 | Fish tissue | Yes | p,p'- & o,p'-DDE; p,p'- & o,p'-DDT; p,p'- & p,p'-DDT; aldrin, endrin; HCH; 2,4-D | No |
| Renoster River and Vaal River, North West and Free-State | 2014/2015 | Surface water | Yes | Glyphosate; 2,4-D | No |
| Herbeepoor Dam, North West and Jukaskei River, Gauteng | 2014/2015 | Surface & groundwater & fish tissue | Yes | Atrazine; simazine; terbutylazine; atrazine; prometryn; prometon; propazine | No |
| City of Ekurhuleni Metropolitan, Gauteng | 2015/2016 | Wastewater effluent & river water | No | Not applicable | Yes: (anti-)oestrogenic activity |
| Orlando Dam and Klipspruit wetland catchment, Gauteng | 2015/2016 | Surface water; sediment & fish tissue | No | Not applicable | Yes: Intersex; abnormal urogenital papillae; abnormal gonads |
| Simangaliso Wetland Park, KwaZulu-Natal | 2015/2016 | Surface sediment | Yes | α, β, γ-HCH; aldrin; dieldrin; endrin; endrin ketone; heptachlor; heptachlor epoxide | No |

*Source: Horak, S., Horn, S. and Pieters, R. (2021). doi:10.1016/j.envpol.2021.115718*
| No. | Year | Location | Sample Type | Presence | Chemicals Detected |
|-----|------|----------|-------------|----------|-------------------|
| 31  | 2015/2016 | iSimangaliso Wetland Park, KwaZulu-Natal | Fish tissue | Yes | α-, β-, δ- & γ-HCH; aldrin; dieldrin; endrin; endrin aldehyde; endrin ketone; p,p'-DDE; p,p'-DDD; p,p'-DDT; heptachlor; heptachlor epoxide; methoxychlor; α- & β-endo-sulfan; endosulfan sulphate |
| 32  | 2015/2016 | Buffalo River, Eastern Cape | Surface water | Yes | α-, β-, δ- & γ-BHC; heptachlor; heptachlor epoxide; aldrin; dieldrin; endrin aldehyde; endosulfan I & II; endosulfan sulphate; p,p'-DDD; p,p'-DDE; p,p'-DDT; methoxychlor |
| 33  | 2015–2017 | Saldanha Bay, Western Cape | Mussel tissue | Yes | Cis- & trans-permethrin; dieldrin; α-chlordane; chlorobenzilate; endosulfan II; trans-nonachlor; p,p'-DDE |
| 34  | 2016/2017 | iSimangaliso Wetland Park, KwaZulu-Natal | Crocodile tissue | Yes | p,p'-DDE; p,p'-DDD; p,p'-DDT; γ-HCH; aldrin; dieldrin; endrin; heptachlor; methoxychlor; α- & β-endo-sulfan |
| 35  | 2016/2017 | Maputaland coast, KwaZulu-Natal | Coral reef invertebrates | Yes | p,p'-DDE; p,p'-DDD; p,p'-DDT; α-, β-, δ- & γ-HCH; aldrin; endrin; endrin ketone; endrin aldehyde; dieldrin; heptachlor; heptachlor epoxide; methoxychlor; α- & β-endo-sulfan sulphate |
| 36  | 2017 | Camps Bay, Western Cape | Seawater; sediment; beach sand; seaweed & marine organisms | Yes | Atrazine; alachlor; simazine; metolachlor; butachlor |
| 37  | 2017 | South Atlantic and Indian Ocean coasts of South African | Chokkie squid | Yes | DDT; HCHs; HCB |
| 38  | 2017/2018 | Grabouw, Piketberg and Hottentots-Holland Mountains, Western Cape | Surface water | Yes | Carbendazim; fluopyram; fludioxonil; metalaxyl; myclobutanil; epoxiconazole; tebuconazole; atrazine; S-metolachlor; prometryn; simazine; terbutylazine; terbutryn; acetamiprid; chlorantraniliprole; chlorpyrifos; diazinon; imidacloprid; methoxyfenozide; thiacloprid; azoxystrobin; bosalid; epoxiconazole; propiconazole; atrazine; diuron; propyzamide; cyromazine; oxamyl |
| 39  | 2017/2018 | Swartkops River and Sundays River estuaries, Eastern Cape | Fish tissue | Yes | α-, β-, γ- & δ-HCH; p,p'-DDE; p,p'-DDD; p,p'-DDT; heptachlor; heptachlor epoxide; aldrin; dieldrin; endrin; endrin aldehyde; endrin ketone; endosulfan I & II; endosulfan sulphate |
| 40  | 2018 | Boulkra River, Tiyume River, Buffalo River and Swartkops River, Eastern Cape | Wastewater influent & effluent | Yes | Atrazine |
| 41  | Unknown | Unknown | Surface & drinking water, treated wastewater | Yes | Carbendazim; diuron; limuron; Yes: Oesothrogen receptor activity; androgen receptor activity; glucocorticoid receptor activity; proges- terone receptor activity; thyroid receptor activity; retinoid X receptor activity; retinoid X receptor activity; Leuscha et al. (2018) |

*Agrochemical only detected, levels not quantified.

*Studies that detected agrochemicals in coastal areas and/or marine biota are included to illustrate the extent of freshwater pollution.

c: alpha; β: beta; δ: delta; ε: epsilon; γ: gamma; δ: delta; p: para; o: ortho; BHC: benzenehexachloride; DDD: dichlorodiphenyldichloroethylene; DDE: dichlorodiphenyldichloroethane; DDT: dichlorodiphenyltrichloroethane; HCB: hexachlorobenzene; HCH: hexachlorocyclohexane; 2,4-D: 2,4-dichlorophenoxyacetic acid;

Studies which have only investigated the level of agrochemicals.

Studies which have only investigated ED effects.

Studies which have investigated both the level of agrochemicals present and associated ED effects.
Likewise, OCPs have been detected in three invertebrate coral reef species (Theonella swinhoei, Sarcophyton glaucum, Sinularia gravis) along the Maputaland coast in South Africa, with results further indicating widespread contamination (Porter et al., 2018) throughout our country’s biodiversity hotspots (Table 1). Lindane, aldrin, heptachlor, DDT, DDE and DDD were detected in the fat tissue of wild Nile crocodiles (Crocodylus niloticus) from the iSimangaliso Wetland Park. Halogenated compounds (DDT, HCB, 2,4,5-T, HCH and mirex) have been also detected in C. niloticus eggs from the Kruger National Park (Table 1), indicating exposure via food and/or water (Bouwman et al., 2014).

The presence of OCPs in marine and freshwater biota is of concern. Fish play an important role in different trophic levels, as well as in the cycling of essential nutrients in aquatic environments (Islam et al., 2016). Sharks and crocodiles are both important apex predators within their respective food webs. They fulfill important ecological roles such as limiting prey populations and preventing bottom-up driven ecosystems (Wallach et al., 2015). Coral reefs are productive ecosystems that form nurseries for a wide variety of fish species and prevent coastal erosion (World Wildlife Fund, 2020), while mussels are an important part of the South African aquaculture industry, accounting for 1140 tonnes or 37.4% of the total marine catches in 2013 (Department of Agriculture, Forestry and Fisheries, 2017). Moreover, the presence of OCPs in ecologically important regions in South Africa is worrisome as many of these areas are not surrounded by intense agricultural or domestic activities. In many cases, the fish species from these areas serve as a food source for local people from the surrounding villages. The OCPs therefore not only pose a threat to wildlife and biodiversity but also the health of local communities.

4. The threat of current-use triazine herbicides

A well-known class of pesticides currently used in South Africa are triazine herbicides. They are used to control weeds in both agricultural and non-agricultural applications (Kunene and Mahlambi, 2020) and include atrazine, simazine, prometryn, propazine, terbuthylazine, ametryn, prometon, terbutryn and atratran (Quinn et al., 2011; Odendaal et al., 2015) (Table 1). Atrazine (2-chloro-4-ethylamino-6-isopropyl-amino-s-triazine) is probably the best-known herbicide in the triazine class and is widely used in both pre- and early post-emergent applications (World Health Organization, 2011) for the control of broadleaf and grassy weeds (Gre et al., 2015; Asouzu Johnson et al., 2019). Moreover, it is amongst South Africa’s top 25 used pesticides (Farounb and Ngqwalwa, 2020). Since maize is an important staple food and livestock fodder in South Africa, it is also the country’s number one crop in terms of production mass and area harvested (Fourie et al., 2017). As a result, maize production consumes 88% of the country’s applied pesticide (Dabrowski, 2015; Rimayi et al., 2018a). This herbicide has a half-life of 30 days in soil and water. Its high water solubility enables it to move into surface- and groundwater, resulting in the contamination of non-target areas (Asouzu Johnson et al., 2019). Some studies reported that atrazine remained in the soil for up to four years after application, entering surface- and groundwater (Farounb and Ngqwalwa, 2020). Since atrazine is a known EDC (Machete and Shadung, 2019) causing feminisation in X. laevis (Hayes et al., 2010) and reproductive dysfunction in zebrafish (Danio rerio) (Wirbisky et al., 2016), its environmental persistence (Kass et al., 2020; Kunene and Mahlambi, 2020) can lead to potential toxic effects in other wildlife species.

Unfortunately, several studies have detected triazine herbicides in the South African environment (Table 1), while other laboratory-based studies have evaluated the effects of environmentally-relevant concentrations on amphibians (Rimayi et al., 2018b; Asouzu Johnson et al., 2019), Du Preez et al. (2005) determined triazine pesticide concentrations in the surface waters of amphibian habitats in the western Highveld region of South Africa, indicating herbicide movement to non-target areas as was also observed by Curchod et al. (2020) (Table 1). The presence of pesticides in the surface water of three rivers in the Western Cape was compared with spray records, with the majority of pesticides detected, but not listed, being herbicides such as atrazine, simazine, terbutryn, prometryn and terbutylazine. Rimayi et al. (2018a) on the other hand, detected atrazine in groundwater samples from the Hartbeespoort Dam catchment (Table 1). Since the geohydrological data showed no evidence of interaction between the surface- and groundwater, the study concluded that groundwater is being polluted by an unknown source. Recently, Barnhoorn and Van Dyk (2020) (Table 1) were the first to report on the presence of certain triazine herbicides (including simazine, atrazine and terbuthylazine) in the freshwater impoundment of the Roodeplaat Dam. This water source supports several agricultural and residential activities that are most likely responsible for the pesticide contamination.

Although triazine herbicides in drinking water have been detected at different South African localities, the concentrations differ. Higher atrazine levels were recorded in drinking water from Johannesburg throughout the year and Bloemfontein in autumn and spring (13–185 ng/L), when compared with Cape Town, Port Elizabeth, Durban, and Pietermaritzburg (not detected to 23 ng/L) (Odendaal et al., 2015). The highest level of atrazine detected in river and tap water from the Vals and Renoster catchments (Machete and Shadung, 2019) was more than twice (350 ng/L) the concentration reported by Odendaal et al. (2015). However, in both studies, the levels were still below the atrazine drinking water guideline values proposed by the World Health Organization (WHO) (100 μg/L) (World Health Organization, 2011). Although this indicates no acute toxicity, the long-term effects of low-level chronic exposure of many triazine herbicides remain largely unknown (Odendaal et al., 2015). This highlights the importance of laboratory-based studies that evaluate the effects of chronic exposure to environmentally-relevant herbicide concentrations.

Since the embryonic and larval stages of amphibians are highly sensitive to xenobiotics and spend their early life stages in water bodies, amphibians are often used to study the effects of agrochemicals on aquatic ecosystems (Islam et al., 2018). Laboratory studies evidenced that environmental concentrations of atrazine are toxic towards tadpoles and male African clawed frogs (X. laevis) (Asouzu Johnson et al., 2019), Rimayi et al. (2018b) reported that environmentally relevant concentrations of atrazine caused ED effects such as reduced tadpole mass, gonadal atrophy, damage to the seminiferous tubule’s structure, damaged mitochondria in Leydig and Sertoli cells and disruption of germ cell lines in tadpoles and adult male Africa clawed frogs (X. laevis).

5. Glyphosate-based herbicides, 2,4-D and chlorpyrifos

In addition to OCPs and triazine herbicides, the use of other pesticides (glyphosate-based herbicides (GBHs), 2,4-dichlorophenoxyacetic acid (2,4-D), organophosphate insecticides) in South Africa should also be cautioned. These are current-use pesticides which have been detected in the environment (Morgan et al., 2008; Rendón-Von Osten and Dzul-Caamil, 2017; Horn et al., 2019). Glyphosate [N-(phosphonomethyl) glycine] is the active ingredient in many non-selective herbicide formulations used globally for the control of weeds (Davouk et al., 2013). Glyphosate-based herbicides are also the most commonly applied type of herbicide in South Africa (Mensah et al., 2013) and formulations have various trade names, including the well-known Roundup. In 2012, an estimated 23 million litres of GBHs were
sold in South Africa alone (Horn et al., 2019). Although GBHs are widely used throughout South Africa, Horn et al. (2019) were the first to report on the levels of glyphosate, 2,4-D and Cry proteins from Bt maize in South African water systems (Table 1). Although glyphosate could only be quantified at one of the sampling locations, this preliminary assessment highlighted the importance of regularly monitoring the presence of pesticides in surface waters.

Mensah et al. (2012) investigated growth parameters of freshwater shrimp (C. nilotica) as a potential biomarker of Roundup in South African freshwaters. All of the parameters tested (percentage mass gain, percentage length gain, specific growth rate, feed intake, feed conversion ratio and feed conversion efficiency) proved to be potential biomarkers. Moulting frequency, however, was considered a clear indicator of sub-lethal effects associated with Roundup. Since the neuroendocrine system regulates moulting in crustaceans, the dose-dependent manner in which Roundup affected moulting suggests that the Roundup formulation exhibits endocrine-like activities. In 2013, the same authors developed South African water quality guidelines for glyphosate using species sensitivity distributions (SSD). Both short- and long-term trigger values (0.250 and 0.002 mg/L glyphosate, respectively) were derived and can be used in monitoring programs to ensure the safety of South African aquatic biota (Mensah et al., 2013).

Another commonly used pesticide is the broad-spectrum herbicide, 2,4-D, which is used pre- or post-emergence to control broadleaf weeds. It was introduced in the 1940s during World War II (Hayes and Hansen, 2017) and is still used in many countries today, including South Africa, to reduce crop losses (Horn et al., 2019). Islam et al. (2018) recently critically reviewed the potential impact of 2,4-D on humans and the environment focusing on its environmental accumulation, interaction with soil, the movement to non-target areas via leaching, volatilisation and drift, and effect on living organisms (fish, amphibians, insects, earthworm, rodents, bacteria) and plants (terrestrial and aquatic). Although 2,4-D is the active ingredient in over 1500 formulated herbicides (Hendricks et al., 2009) nobody reported on its presence in South African water until Horn et al. (2019) detected concentrations exceeding European drinking water guidelines. This is of concern as 2,4-D is directly applied to both aquatic and conventional farming systems (Islam et al., 2018) and can therefore easily reach non-target aquatic environments, posing a risk to both wildlife and human communities that use the water for household activities.

Organophosphate pesticides (OPs) are another class of agrochemicals of importance found in the South African environment. Although OPs are not as environmentally persistent as OCPs, they can bioaccumulate (Quinn et al., 2011). Chlorpyrifos is a well-known OP that is commonly applied as a broad-spectrum insecticide. It is characterised by a long half-life in soil, as well as being volatile. This allows the insecticide to enter the atmosphere and reach non-target aquatic environments when rain- and washout occurs (Vogel et al., 2008; Fuhrmann et al., 2020). Based on probabilistic risk estimates, concentrations of particle-associated pesticides and physicochemical parameters, chlorpyrifos was found to be one of two pesticides posing the highest acute risk to marine and freshwater organisms present in the Lourens River estuary in the Western Cape (Bollmohr et al., 2007). Based on the Predicted Relative Risk of their geographic information system (GIS)-based pesticide risk indicator, Dabrowski and Baldacchini (2013) came to the same conclusion. Furthermore, chlorpyrifos has been detected in the surface waters of Rietvlei Nature Reserve and Albasini Dam (Wooding et al., 2017) (Table 1), as well as in sampling points across Grabouw in the Western Cape (Dalvie et al., 2003).

6. Endocrine disrupting effects observed in South African water

The term “endocrine disruptor” was first introduced globally in 1991 at the Wingspread Conference in Racine, Wisconsin (Soto et al., 2006; Papalou et al., 2019), with Colborn et al. (1993) publishing one of the first papers on the topic. Natural EDCs do occur. However, the majority are synthetic and over 1000 synthesised chemicals released into the environment through anthropogenic activities are considered to be EDCs (Olujimi et al., 2010; DeWitt and Patisaul, 2018; Papalou et al., 2019). These include plasticisers, pharmaceuticals, personal and healthcare products, industrial chemicals (solvent and lubricants), flame retardants and agrochemicals (Papalou et al., 2019). Agrochemicals are an important class of EDCs, with various studies noting the ED effects associated with certain insecticides (De Angelis et al., 2009), herbicides (Harada et al., 2016), fungicides (Kackar et al., 1997), algaeicides (Hiromori et al., 2016), nematicides, acaricides (Yang et al., 2019), fumigants (Goldman et al., 2007), fertilisers (Poulson et al., 2018), and plant growth regulators (Xiagedeer et al., 2020). Therefore, when these agrochemicals reach non-target aquatic environments they can elicit ED effects on wildlife species through different modes of action.

Complex mixtures of EDCs from different sources (industrial chemicals, agrochemicals, pharmaceuticals) end up in aquatic environments, making it near impossible to pinpoint their origins. The response to the COVID-19 pandemic is likely to contribute additional pharmaceuticals to the aquatic environment (Horn et al., 2020). When environmental matrices are assessed for ED activity, any of the hormone-active chemicals present in environmental extracts can be responsible for the observed activity. Different environmental mixtures of agrochemical EDCs can (Jacobsen et al., 2012; Ma et al., 2019; Papalou et al., 2019) but do not always elicit synergistic, additive or combined effects (Archer and Van Wyk, 2015). Different EDCs also elicit an effect through different modes of actions. The majority of ED related studies in South Africa have investigated disruption of reproductive systems (Barnhoorn et al., 2010; Kruger et al., 2013; Arukwe et al., 2016; Viljoen et al., 2016; Leusch et al., 2018) (Table 1) and when EDCs elicit an effect on the reproductive system, the response (mode of action) can either be (anti)-oestrogenic or (anti)-androgenic (Archer and Van Wyk, 2015).

The majority of studies conducted in South Africa with regards to ED have focused on two forms of hormonal disruption: chemicals that mimic the in vitro and/or in vivo actions of natural endogenous oestrogen(s) (oestrogenic activity) in water (Swart et al., 2011; Bittner et al., 2014; Truter et al., 2015, 2016; Archer et al., 2020), and intersex in fish (Kruger et al., 2013; Bengu et al., 2017) (Table 1). The former has been well-documented in South African water systems, including the Rietvlei and Marais dams (Aneck-Hahn et al., 2008), effluent from the Daaspoort WWTP in Pretoria West that enters the Apies River (Mahomed et al., 2008), a groundwater system near Mokopane (Aneck-Hahn et al., 2009), sites surrounding the Eerste River (Swart et al., 2011), and the mouth of the Salt River, Durban Bay and the Isipingo estuary (Truter et al., 2015) (Table 1).

Aneck-Hahn et al. (2009) reported oestrogenic activity in borehole samples from rural communities in the Waterberg district (Limpopo province), indicating oestrogenic pollution of the groundwater system the communities rely on for potable water. Archer and Van Wyk (2015) investigated the ability of various fungicides and one insecticide commonly used in the Western Cape for agricultural purposes to alter binding of the androgen dihydrotestosterone to the human androgen receptor using a recombinant yeast androgen screen (Table 1). Mancozeb was the most
anti-androgenic with the highest relative potency (19.449%) in comparison to the control fungicide, vinclozolin (100%). Recombinant yeast bioassays were also employed by Truter et al. (2015) to assess the (anti-)oestrogenic and (anti-)androgenic activity of water sampled from rivers, harbours and estuaries in three major South African metropolitan municipalities (Table 1). The mouth of the Salt River in Cape Town not only accounted for the highest 17β-oestradiol concentration but also the most oestrogenic. Oestrogenicity was detected in eThekwini, while none of the Port Elizabeth harbour samples exhibited oestrogenic activity. Overall, anti-androgenicity was more prevalent than oestrogenicity and based on the results obtained there is a potential for ED in marine and estuarine wildlife in eThekwini and the City of Cape Town.

Endocrine disrupting chemicals can also influence the regulation and expression of important genes associated with normal hormone function in fish species. Truter et al. (2016) followed an in vitro and in vivo approach, combined with enzyme-linked immunosorbent assays to establish whether surface water from the upper Olifants River in Mpumalanga contained steroid oestrogens (17β-oestradiol and 17α-ethinylestradiol); showed (anti-) oestrogenic and (anti-)androgenic activity; or potentially caused ED in juvenile fish species by affecting the expression of genes associated with reproductive, thyroid and adrenal signalling pathways (Table 1). Both 17β-oestradiol and 17α-ethinylestradiol were detected in the upper Olifants River, with the highest concentrations for both oestrogens detected at a WWTP-impacted site during summer. Oestrogenic, androgenic, and anti-androgenic activity was observed for some of the sites sampled, while no anti-oestrogenic activity was observed. Based on the gene expression results the aromatase coding gene (cyp19a1b) was downregulated and the thyroid-linked gene (type-2 deiodinase) upregulated in the juvenile O. mossambicus, indicating potential aromatase and type-2 deiodinase disruption due to anthropogenic stressors. Endocrine disrupting effects have also been observed in frog species. Viljoen et al. (2016) reported on the asymmetric testicular morphology of frogs that naturally occur in dams and ponds in Limpopo where DDT is still sprayed for malaria control. This phenomenon was attributed to possible ED caused by DDT (Table 1).

The prevalence of intersex (an individual exhibiting both male and female reproductive features simultaneously due to exposure to pollutants during embryogenesis) as a form of ED has been reported in fish species throughout South Africa (Barnhoorn et al., 2004; Barnhoorn et al., 2010, 2015; Kruger et al., 2013). Some agrochemicals (linuron, diuron and vinclozolin metabolites) have the potential to block the androgen receptor, resulting in the feminisation of wildlife species (Quinn et al., 2011). The first histological evidence of intersex in fish species found in South African water bodies was reported by Barnhoorn et al. (2004) more than 15 years ago. Based on the gonad-somatic index values obtained, the results suggested that the intersex was a result of the feminisation of male catfish present in the Marais and Rietvlei dams. More recently, intersex was found in 13.6% of C. gariepinus from the Orlando Dam and 50% from a Klipspruit pond. In addition, 80% of the male African sharptooth catfish collected here displayed abnormally shaped urogenital papillae, and discoloration of liver tissue was observed in some fish (Bengu et al., 2017).

Apart from C. gariepinus, evidence of intersex was also found in O. mossambicus sampled from three locations in the Luvuvhu River, Limpopo (Barnhoorn et al., 2010) (Table 1). The number of intersex fish in the Luvuvhu River was higher than the number found in the Rietvlei Nature Reserve a few years earlier (Barnhoorn et al., 2004). The highest percentage of intersex was recorded for Nandoni Dam (63%), followed by Xikundu Weir (58%), and Albasini Dam (48%). Fat samples were also collected from the fish and the following pesticides were quantified: DDT, DDE, DDD, endosulfan, dieldrin, endrin, and lindane. Environmentally relevant concentrations of herbicides have also been found to cause oestrogenic and anti-androgenic effects in tilapia species elsewhere, including Nile tilapia (O. niloticus) native to northern Africa (De Almeida et al., 2018).

Other ED effects have also been reported in South African wildlife. Bornman et al. (2010) investigated the formation of calcifications inside the seminiferous tubules, known as testicular microlithiasis, in wild eland (Tragelaphus oryx) from two well-known nature reserves in Gauteng (Rietvlei Nature Reserve and Suikerbosrand Nature Reserve). Overall, spermatogenesis was impaired, intra-tubular dystrophic calcifications were present and sloughing of the seminiferous epithelium was found. Some of the eland from Suikerbosrand Nature Reserve had DDT, DDE or DDE present in their fat samples. The authors associated the presence of these testicular lesions with associated body-burdens of EDCs. This was one of the first studies in South Africa showing that environmental EDCs not only affect fish species but also mammalian wildlife. Moreover, infertile and banded eggs have been reported in C. niloticus, with authors concluding that the crocodiles experienced contaminant-induced ED effects (Arukwe et al., 2016) (Table 1).

Although ED effects have been investigated in South African freshwater, few studies follow a combined approach of investigating both agrochemicals and ED effects. This is of concern as the ED effects of many current-use pesticides in South Africa are unknown. This was evidenced by Dabrowski et al. (2014). As part of an agricultural pesticide prioritisation study, an initial screening process yielded 152 active ingredients which are applied to South African crops on a national scale. Around 9% of the active ingredients were known EDCs. Of the remaining 91%, there was no data available for ~57%, only possible evidence for ~22% and ~13% were known to not elicit ED effects. Moreover, in South Africa, most studies related to the ED effects associated with agrochemicals have focused on the adverse effects of DDT and its metabolites in the Vhembe district of Limpopo. Although the highest number of malaria cases were reported in South Africa in 2017 since 2000, a lot of progress has been made towards KwaZulu-Natal’s goal of eliminating malaria by 2023 (Maharaj et al., 2019). However, even if the country implements alternative methods for vector control, South Africa’s extensive history of DDT application has left a permanent mark on the country’s wildlife.

Patrick et al. (2016) assessed the ED effects associated with lifetime exposure (in utero-, lactational- and direct) to DDT and DDE at environmentally relevant concentrations. Endocrine disrupting effects were evaluated by measuring the response of different male reproductive endpoints (anogenital distance, prostate mass) and testicular histology in Sprague-Dawley rats. Rats exposed to DDT and DDE exhibited a higher prostate and testicular mass, respectively. Exposure to a mixture of pesticides resulted in a shorter anogenital distance, compared to the control group. An imbalance in thyroid and vitamin A homeostasis has also been found to be associated with DDT (Delport et al., 2011).

7. Conclusion

Based on the literature reviewed, the country’s aquatic environment and wildlife are considerably burdened by agrochemicals of which OCPs and triazine herbicides have been investigated to the greatest extent. However, other pesticides such as GBH, 2,4-D and chlorpyrifos also remain important chemicals which could potentially impede South Africa’s progress towards environmental sustainability in the agricultural sector. Although DDT has been banned in developed countries and its use restricted in South Africa for malaria vector control only, remnants of its legacy-use are still detected in South African waters. Moreover, agrochemicals have
even been detected in the Pongola River Floodplain—South Africa’s largest floodplain—and the iSimangaliso Wetland Park World Heritage site, emphasising the threat posed to the country’s rich biodiversity. As the largest pesticide user in sub-Saharan Africa, South Africa has used and possibly misused many of the chemicals now classified as EDCs. This is evidenced by studies detecting (anti-)oestrogenic and (anti-)androgenic activity in South African rivers, groundwater systems and drinking water. In addition, a variety of ED effects have been found in wildlife species. Since agrochemicals are widespread throughout South African water and many pesticides are known EDCs, their presence poses a risk to human health, wildlife and the environment.

To safeguard and ensure healthy aquatic ecosystems, researchers, managers and government bodies will have to work together to develop and implement better and continuous monitoring programs. Farmers and relevant stakeholders in the agriculture sector will have to work with the Department of Agriculture, Forestry and Fisheries to ensure that detailed spray records of all agrochemicals applied to crops are kept to facilitate the monitoring of pesticides that reach non-target areas. Where possible, biological control should be considered as an alternative to agrochemicals. In terms of research, stronger collaborations are required to follow a multidisciplinary approach of effects-directed research in which in vitro and/or in vivo bioassays and chemical screening are combined. If target areas exhibiting high ED activity have been identified, high-throughput chemical screening can be used to investigate which chemicals are involved. Such an approach will aid in assessing the ecotoxicological risks posed by many current-use agrochemicals, as well as facilitate the development of water quality guidelines for pesticide products commonly applied in the South African agricultural sector. Consequently, this will reduce agrochemical pollution of surface- and groundwater systems, enhancing the ecological integrity of aquatic environments and biota.

Author contributions
Ilize Horak, Conceptualisation, Writing - original draft. Suriarie Horn, Conceptualisation, Writing - review & editing, Supervision, Funding acquisition. Rialet Pieters, Conceptualisation, Writing - review & editing, Supervision, Funding acquisition.

Declaration of competing interest
The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements
The financial support of the National Research Foundation (NRF) South Africa in the form of bursaries and funding (grant numbers 103487, 121713 and 116716), and financial support in the form of project funding provided by the Water Research Commission (WRC) South Africa (project number C2020/2021-0095) are hereby acknowledged. Opinions expressed and conclusions arrived at are those of the authors and are not necessarily to be attributed to the NRF and/or WRC. The authors would like to acknowledge that the graphical abstract was created with BioRender.com.
