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A review of the conservation status of the Nile crocodile (*Crocodylus niloticus* Laurenti, 1768) in aquatic systems of Zimbabwe

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ABSTRACT

Nile crocodile survival is threatened by water pollution, habitat loss, extensive water abstraction for irrigation, domestic use and industrial development and overexploitation of fisheries resources in water systems in Zimbabwe. This review assessed the abundance, distribution and population trends of Nile crocodiles (*Crocodylus niloticus* Laurent, 1768), and explored the effects of ranching, trophy hunting and human-crocodile conflicts (HCC) on its conservation status in water systems of Zimbabwe. Scoping reviews of available literature and analysis of recurrent themes indicated that crocodile censuses were concentrated in the warmer northern and southern parts of the country. Ranching and trophy hunting have contributed to the increases in crocodile populations. Human encroachment and wetland degradation have increased HCC in fringe communities proximate to protected areas consequently inducing negative perceptions and hurtrage which threatens crocodile populations. Overall, there is an increase in crocodile populations in sampled areas. Nonetheless, there is a need to assess the abundance, distribution and population trends, and delineate hotspots of suitable habitats and contextual challenges in less sampled areas before stating the national crocodile population estimate. Implementing astute crocodile conservation efforts involving locals is key in HCC mitigation. However, it implores the need for development of pro human-crocodile co-existence and circumstantial HCC resolution policies. For posterity, the conservation status of crocodiles in Zimbabwe should be ascribed as Vulnerable or Near Threatened rather than the current Least Concern or Low Risk status.

1. Introduction

The Nile crocodile (hereafter referred to as crocodiles), *Crocodylus niloticus* Laurent, 1768, which belongs to the subfamily Crocodylinae, or true crocodiles, is an apex predator ubiquitous in wetland systems of over twenty six Sub-Saharan African countries (Games and Moreau, 1997; CSG, 2009; Combrink et al., 2011; van Asch et al., 2019). As an apex predator targeting mainly fish, other reptiles, aquatic birds, mammals, amphibians, insects and molluscs the species is vital in maintaining the integrity of freshwater ecosystems (Roff and Zacharias, 2001; Glen et al., 2007). The ubiquity of the Nile crocodile has led to debates on its nominal and sub-species and related differences in morphology in geographically distinct regions in Africa (Fuchs et al., 1974; van Asch et al.,

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2019). However, the species is a large crocodilian averaging 5 m in length but reportedly reaching 6–7 m in rare instances (CSG, 2009; IUCN, 2017).

Sexual maturity is attained at 2.6 m for females and 3.1 m for males (Cott, 1961). Females lay around 40–60 eggs in guarded nests and the incubation times ranges from 70 to 100 days, after which they open the nest and carry juveniles to the shallow edges of wetlands (Cott, 1961; Kofron, 1993). Socially, crocodiles are gregarious and observe hierarchy (based on age and sex) but do not actually form clusters rather they congregate in shallow sections of wetlands to feed, defecate, bask, court and mate (Hutton, 1987; CSG, 2009). Typically, aggression occurs among males when establishing territoriality and fighting for mates, and among females when guarding hatchlings at the edges of wetlands (Kofron, 1993). For a comprehensive insight into the other ecological aspects on diet, thermoregulation, reproduction, social behaviour, habitat preferences and population dynamics see (Cott, 1961; Hutton, 1984, 1987; Kofron, 1993; Fergusson, 2010).

Habitat ranges of crocodiles traverses the Nile River in the north, Senegal and Benin Rivers in the west to the east in the Congo Basin (IUCN, 2017). The habitat ranges also stretches across the southern most limits of Lower Kunene River in Namibia (Griffin, 2003), through the Okavango Delta and Makgadikgadi Pans in Botswana, Zambezi River in Zimbabwe and Zambia (Wallace et al., 2013), and Lake Sibaya (Combrink, 2004) the St Lucia Wetlands in South Africa (Leslie, 1997; Feeley, 2010; Combrink, 2014; Marais, 2014), and right to the Island of Madagascar (Pooley, 2016a; Pooley et al., 2019). The fact that the species is widespread and abundant in wetland systems of Africa has led to its conservation status classification as a Least Concern/ Low Risk animal (CSG, 2009; IUCN, 2017). Furthermore, the estimated global population of the species which lies between 250 000–500 000 individuals buttressed the belief that it was in abundance, widespread and thus relatively secure and not threatened by intrinsic and exogenous factors (Kyalo, 2008). Fergusson (2006, 2010) indicated that the crocodile population in the Eastern and Southern African regions were relatively secure. Consequently, most countries including Zimbabwe have adopted the IUCN conservation status at face value without revision, consolidation and consideration of the abundance, distribution and contextual opportunities and challenges facing the species (Zisadza-Gandiwa et al., 2013, 2016). However, the lack of clarity on the methods used to estimate the local and even regional crocodile populations reduces the validity, reliability and applicability of the estimates in conservation efforts (CSG, 2009; IUCN, 2017).

1.1. *Contextual situation for Nile crocodiles in wetland systems of Zimbabwe*

The Nile crocodile occurs throughout Zimbabwean wetland systems comprising marshes, rivers, pools, swamps, and reservoirs, and it is the only true crocodilian species found in the country (Child, 1987; Kofron, 1993). Fergusson (2006, 2010) indicated that crocodile populations were relatively secure in Zimbabwe with over 12,000 wild crocodiles in the northern parts of the country. Nile crocodiles are important to the ecology of the water systems as an apex predator regulating the aquatic food web, competing with fisheries for the fish resources and most importantly the species generates the much needed foreign currency through exports of skins, meat and other ancillary parts such as the tail and gall bladder that are in demand in the Asian markets (Crocodile Farmers Association of Zimbabwe, (CFAZ, 2016). Moreso, crocodile ranches and trophy hunting in safari areas leads to employment, income and regulation of the species numbers in captivity and wilderness in Zimbabwe (ZPWMA, 2015).

The concerns on crocodile populations arise from a number of factors in Zimbabwe. Firstly, there has been an actual increase in the number of human-crocodile and crocodile-livestock conflicts especially in wetlands outside of protected areas (ZPWMA, 2015; Musiwa and Mhlanga, 2020). The locals have perceived an increase in the numbers of crocodile sightings in rivers and dams in Zimbabwe and thus, believe the species has been increasing in numbers (Gandiwa, 2011; Zisadza-Gandiwa et al., 2013, 2016). Consequently, there has been an increased human and livestock casualties and fatalities frequently reported in the national media leading to negative local perceptions towards the species in most water systems. Humans have in turn retaliated by persecuting, killing and illegally collecting crocodile eggs for sale and live specimens for medicinal and ritual purposes especially in wetland systems outside of protected areas (Gandiwa, 2011). Sadly, the persecution of crocodiles at best is perceived to be part of Problem Animal Control (PAC), and is hardly reported with the same frequency in the national media, but it has led to decreases in the numbers of the keystone apex predators in water systems of Zimbabwe (Chihona, 2014; Sai et al., 2016; Utete, 2020). Coupled with the on-going urbanisation, technological development and agricultural expansion, there has been a systematic destruction of suitable wetland habitats for crocodiles and hippos among other aquatic species (Utete, 2020). Illegal encroachment leading to siltation and its impacts on potential habitats for the Nile crocodiles has been discussed fully by Zisadza-Gandiwa et al., (2013, 2016) for the Gonarezhou National Park and is examined later in this study. The problem has been lack of concerted data on the impacts of other drivers of global environmental change mainly climate change e.g. through altered rainfall patterns, temperature and evapotranspiration leading to water level fluctuations and other impacts on wetland systems and the crocodiles therein (Utete et al., 2017). The literature is heavily biased towards fish stocks and fisheries dependent livelihoods in water systems of Zimbabwe as summarized by Kupika et al. (2017) and Utete et al. (2018). This leads to inferential speculation that may not offer the best practical conservation solutions for the Nile crocodiles in Zimbabwe. Available literature on drivers of global environmental change such as diseases on the crocodile populations in Zimbabwe has been limited to the reports of outbreaks of the mycoplasmosis by Foggin (1987). In fact, Foggin (1987) indicated that there was an outbreak of mycoplasmosis reported annually, from about 1/3 of Zimbabwean crocodile farms but there were no clear cut actual mortality numbers besides that about 20% of the crocodiles on the assessed crocodile ranches died. There was no (follow up) study to even assess the outbreaks in wild Nile crocodile populations in and outside of protected areas. Thus, the overall effects of diseases on the crocodile populations in Zimbabwe is yet to be quantified.

Suffice to indicate that several studies e.g. Magadza (2010) and Marshall (2011) among others have indicated water pollution and its impacts on fish stocks, macroinvetebrates in rivers and impoundments in Zimbabwe. However, there are few direct studies linking water pollution and its effects (positive and negative) on various ecological aspects of the crocodiles in Zimbabwe. Available studies by Wessels et al. (1980) indicated high levels of chlorinated hydrocarbon insecticide residues in eggs of Nile crocodiles in Lake Kariba. Moreso, Phelps et al., (1986, 1989) reported on organic micro pollutants and mercury (Hg), selenium (Se), cadmium (Cd), lead (Pb), and zinc (Zn) in 26 Nile crocodile eggs from ten sites in Zimbabwe. Phelps et al. (1986) indicated high concentrations of Zn, Pb and Hg in the shells of Nile crocodiles in Lake Kariba that were attributed to the high use of DDT in the control of tsetse flies and malaria mosquitoes in the western parts of Zimbabwe. In a follow up study Phelps et al. (1989) reflected on high levels of DDT in the fat of the apex predator Nile crocodiles in Lake Kariba that were attributed to the short-term persistence or residence time of the organochlorine in the water and sediments, influenced by the tropical nature of the lake, leading to its mobility and accumulation in biological organisms especially those at the top of the food chains or webs. What is significant is that the three documented studies on the impacts of pollutants on Nile crocodiles are dated occurring over 30 years ago. Limnological conditions in Lake Kariba have changed drastically since then (Magadza, 2010; Marshall, 2011), and thus, it necessitates for concerted continuous studies on pollution as an environmental change driver in wetland systems and its effects on ecological aspects of Nile crocodiles in Zimbabwe in general.

A glaring observation in all available crocodile population information is that, most (80%) of the data were concentrated in the Zambezi Valley region including Lake Kariba in northern Zimbabwe (ZPWMA, 2015). Solid data were obtained by Zisadza-Gandiwa et al. (2013) who estimated a combined total of 280–300 crocodiles in the Save (240–250) and Mwenezi and Runde (45–50) Rivers between 2008 and 2011. Hutton (1987) and Fergusson (2010) suggested an approximate population of 228 crocodiles in Ngezi Dam. Sai et al. (2016) suggested a total of 82 crocodiles over a 36 km stretch in Sengwa and Kove Rivers. There are other anecdotal and fragmented studies e.g. Chihona (2014) in Ruti Dam that focussed more on human-crocodile conflicts without really estimating crocodile populations in the wetland systems.

The few examples used above exposed a lack of concrete reliable population data on the Nile crocodile, a keystone species, in wetland systems inside and outside protected areas (Zisadza-Gandiwa et al., 2013). This is an area which needs examination in the context lens of the actual conservation status of the Nile crocodile in the nation which still prescribes to the Least Concern/ Low Risk category of the IUCN Red Data List without evidence from reliable and valid comprehensive survey data (Fergusson, 2010; IUCN,

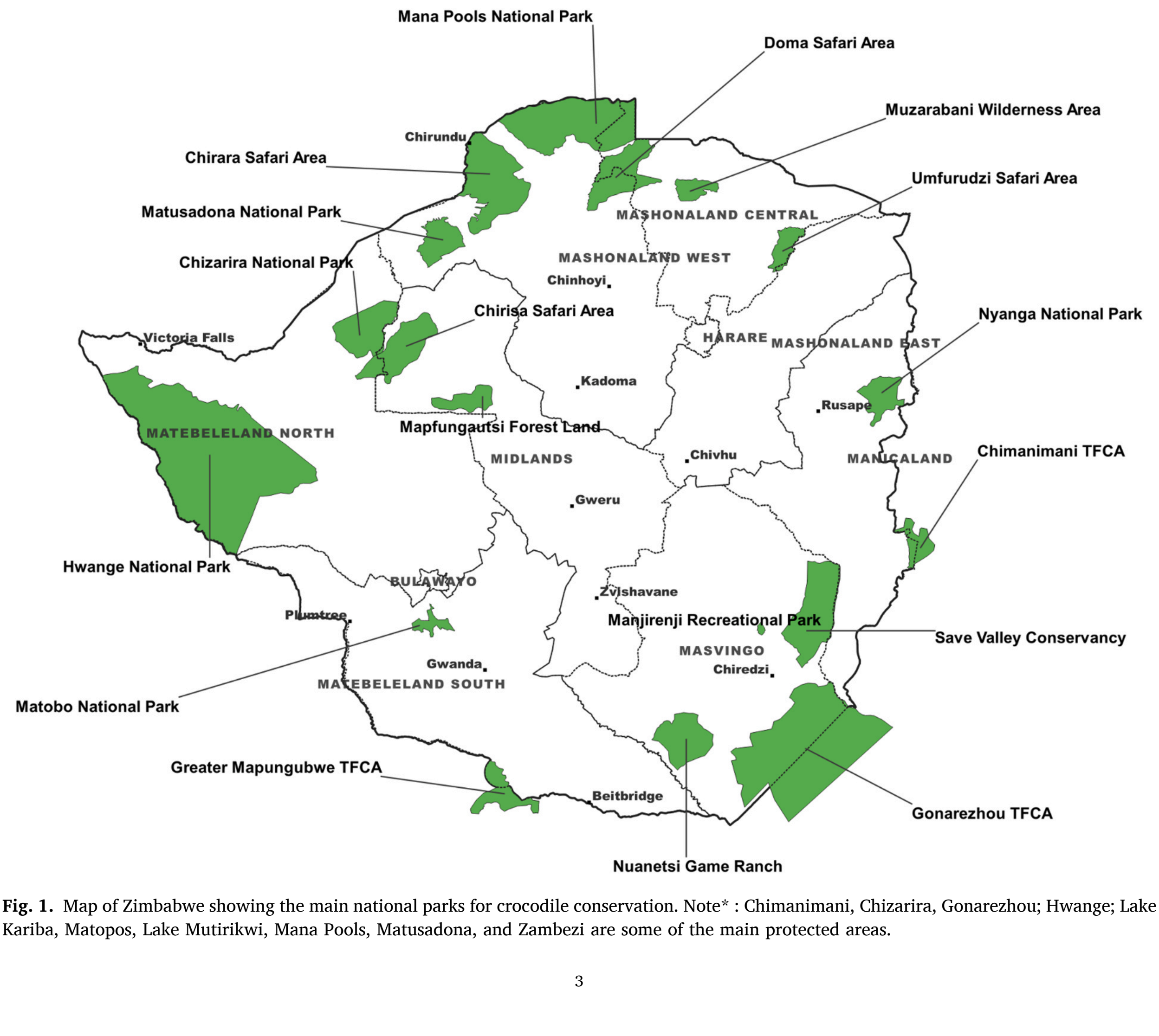

Fig. 1. Map of Zimbabwe showing the main national parks for crocodile conservation. Note* : Chimanimani, Chizarira, Gonarezhou; Hwange; Lake Kariba, Matopos, Lake Mutirikwi, Mana Pools, Matusadona, and Zambezi are some of the main protected areas.

2017). The country's Nile crocodile population is listed on Appendix II of CITES (IUCN, 2017) which denotes that the species is not necessarily threatened with extinction but it may become vulnerable, threatened, endangered and eventually go extinct if risk factors such as overexploitation (through uncontrolled trade, unsustainable harvesting of eggs), and habitat degradation, persecution and poaching are not curtailed (Wallace et al., 2011, 2013; Zisadza-Gandiwa et al., 2013). There is a risk of induced complacency and reduced attention and vigilance on the conservation of Least Concern/ Low Risk species with only late and often ineffective corrective measures applied after dire aberrations in populations (Fergusson, 2010). A cross examination of literature elsewhere indicated that because of specific and unique concerns some countries e.g. Namibia and South Africa actually disregarded the general IUCN classification of the Low Risk status for the Nile crocodiles and have contextually re-classified the species as peripherally endangered and vulnerable respectively after nationwide surveys (Griffin, 2003; Marais, 2014). Thus, there is a need to review the current status of the species and proffer scientific based and contextual recommendations for the conservation of the Nile crocodiles in Zimbabwe.

Moreso, increasing human-crocodile conflicts (HCC) and their devastating effects on the socioeconomic and health aspects of livelihoods in the country implores a need for a review of the conservation status of the species (Dickman, 2010; Fergusson, 2010; Chihona, 2014). Currently, in the ZIMPARKS Act 2014 [20; 14] 8th Schedule Section 121 there are six species that are classified as dangerous animals comprising the Southwest African lion (*Panthera leo*), common hippopotamus (*Hippopotamus amphibius*), African elephant (*Loxodonta africana*), Cape buffalo (*Syncerus caffer*), African rhinoceros, both square lipped (*Ceratotherium simum*) and black (*Diceros bicornis*), and African leopards (*Panthera pardus*). There is a surprise omission of the Nile crocodile which is as equally dangerous as the hippopotamus, and a key protagonist and victim in HCC in Zimbabwe (Dickman, 2010; Utete, 2020). The argument is that the population of the species is large and not threatened in and outside of protected wetland systems (Fergusson, 2010; ZPWMA, 2015). Regardless, increased HCC incidences reflected that this is a key area that needs urgent revision in the ZIMPARKS Act.

Wild breeding stocks of crocodiles form the basis and source of eggs for the lucrative crocodile ranching industry (Wallace et al., 2013; Marais, 2014). However, it has been noted that continuous collection of eggs and breeding stocks in the wild without baseline scientific studies may have detrimental effects on the wild crocodile populations (Siamudaala et al., 2004). Hence a review of the conservation status of the species will enable a conflexion and properly informed scientific and legal protection of the Nile crocodile and its eggs in the wilderness. It will also stimulate regulation of sustainable harvesting by crocodile ranchers who play a prominent role in the economy of the country through earning foreign currency, generating employment and enhancing food security (Revol, 1995; Wallace et al., 2013). This review study aimed to assess: a) abundance and distribution and population trends of crocodiles, b) crocodile ranching and trophy hunting and their impacts on crocodile populations, and c) human-crocodile conflicts and their effects on crocodile population trends and conservation. The ultimate aim was to establish the current conservation status of the Nile crocodiles in the aquatic systems of Zimbabwe.

2. Materials and methods

2.1. Study area

Sharing borders with South Africa, Botswana, Mozambique and Zambia, Zimbabwe (Fig. 1), is a landlocked developing Southern African country with a human population close to 14 million (ZIMSTAT, 2012). Resources dependent industries such as agriculture, mining and wildlife-based tourism are the mainstay of the economy (ZIMSTAT, 2012). It has a fluctuating Gross Domestic Product (GDP) ranging between 3% in 2009 to 5.1% in 2020, because of the constant change in national currency, which normally consists of simultaneous circulation of bond notes and a basket of international currencies. Majority of the population (approximately 89% of the people) are unemployed and live on less than one American dollar per day (ZIMSTAT, 2012). The country waged a protracted liberation struggle which intensified from 1972 to 1979, faced intermittent droughts in 1984, 1992–1994, and embarked on a chaotic land redress/reform program from 2000 which is still ongoing. This resulted in unplanned resettlement areas some of which encroached into national protected areas in the country (ZPWMA, 2015). The net effect has been destruction and irregular monitoring of wetlands especially those outside of the protected areas and this has reduced crocodile habitats, crocodile recruitment, exacerbated HCC, and increased human fatalities (ZPWMA, 2015). However, the threats tend to differ in magnitude and intensity. Hence there is a requirement for a comprehensive review of the abundance, distribution and risks facing the crocodiles in wetlands of Zimbabwe.

3. Data collection

The study applied a scoping review method, a synthesis based approach, to examine Nile crocodile conservation in Zimbabwe from rigorous analysis and examination of existing literature following methods by Gough et al. (2012). The initial point was formulating the research question: what is the conservation status of crocodiles in Zimbabwe? Afterwards, a protocol was generated and a systematic selection of relevant information was carried out across all the ten provinces of Zimbabwe i.e. Bulawayo, Harare, Manicaland, Mashonaland Central, Mashonaland East, Mashonaland West, Masvingo, Matabeleland North, Matabeleland South and Midlands Provinces (ZIMSTAT, 2012) indicated in Fig. 1. Then critical appraisal of results, data extraction and contextual synthesis and dissemination were done. The section below summarises each of the step carried out for the research:

3.1. Defining the search strategy and predocument selection

After situating the study in the formulated main research question, the first exploratory searches of relevant literature were done in Google Scholar, Scopus, and Bing and GiveWater, and the Boolean search engines in order to combine the words AND, NOT, OR and the commonly used ISI Web of Knowledge (ISI WoK) databases with no historical cut off dates for Zimbabwe (and its time as Rhodesia before independence where relevant) see (Utete, 2020). The choice of the search engines was based on their extensive and interdisciplinary coverage and high recognition as standardised databases for conducting meta-analyses. This study explicitly searched for literature focusing on Nile crocodiles, crocodile conservation, crocodile farming with further searches for human-crocodile conflicts in all coupled (using AND, NOT, OR) subgroups which comprised: ''limnology-crocodiles'', ''water resources conservation-crocodile farming'', "fisheries-crocodiles'', ''aquatic organisms-Nile crocodiles'', ''aquatic resources-crocodiles'', and aquaculture researches including ''aquaculture-Nile crocodiles'', ''crocodiles-rivers '', ''crocodile-human deaths'', ''crocodiles-human injury'', together with technical reports on crocodiles in Zimbabwe following recommended methods by Arksey and O'Malley (2005) and Woodhead et al. (2018). In the literature search coupled confounding terms e.g. ''fisheries-crocodiles'', ''human-wildlife conflict'', and ''water resources conservation-crocodiles'' produced a lot of background noise and conjoined other non-relevant information for the study and were removed. There were no additional terms that related to Nile crocodiles in wetland systems of Zimbabwe. Thus, the final search terms used were as follows: (((Nile crocodiles, human-crocodile conflicts, injury, deaths, fishers, fisherfolks, fishermen, fisheries, water systems, rivers, lakes, crocodile farming AND (''crocodile conservation in Zimbabwe*''))).

3.2. Document selection

For item and document selection the key word search methods were used in the same search engines above. The search was limited to the title, abstract text and key words. From an initial list of 2 563 articles, the key words in the abstracts as well as the abstract text were screened for relevant items which could be classified or mentioned crocodile conservation and human-crocodile conflicts in Zimbabwe. The aim was to screen the data set to manageable and relevant sizes. After thorough screening, a total of 94 items were used to reflect the breadth of the context citing crocodile conservation in Zimbabwe. An article was included if it met the following criteria: (a). it was published in a reputable journal, international organisation technical report or a book, (b). relevant conference proceedings on the Nile crocodile conservation, and (c) credible human-crocodile reports in citable technical reports of reputable organisations. The review excluded media reports as they are not peer reviewed to be valid and reliable. However, where there was evidence of thorough vetting and in the case of human- crocodile conflicts some credible media reports were cited.

4. Data analysis

4.1. Thematic cluster analysis

The frequency of recurrence of words in the abstract text or keywords was analysed using the Analysis of Qualitative Data (AQUAD 7) to come up with the main topics or thematic clusters which were thoroughly discussed in both quantitative and qualitative terms. The AQUAD can be used in the analyses of pictures and sound files as well as main words in the text and enables linkages, keyword hierarchies and some basic statistical functions (e.g. Chi-square) and is compatible with the free statistics software "R". From the analyses there was a significant frequency of recurrence (Kruskall ANOVA, $p < 0.05$) for the following clusters related to Nile crocodiles in Zimbabwe: Nile crocodile population, Nile crocodile distribution, Conservation status and concerns on Nile crocodiles, Human-crocodile conflicts, Crocodile farming and ranching in Zimbabwe, Crocodile sport hunting, Conservation and research gaps for Nile crocodiles in Zimbabwe. The theoretical clusters were subsequently discussed whilst the geographical influence on the conservation status of the species was discussed as a background issue in the main thematic clusters. The Kruskall ANOVA was used to assess the non-parametric data set, after testing for normality using the Shapiro Wilk test, on crocodile population trends and linear regression in the SPSS 25 version.

5. Results and discussion

5.1. Distribution of Nile crocodile populations in Zimbabwe

Examination of available data showed that most (92%) population surveys were done in the northern parts of the country e.g. Lake Kariba, Zambezi Valley and its associated protected areas (Fergusson, 2010). Some population studies were done in the southern parts or the lowveld area e.g. Zisadza-Gandiwa et al. (2013) in the Gonarezhou National Park, and Chihona (2014) in the Ruti Dam. There were a few crocodile population studies e.g. Hutton (1984, 1987) and Sai et al. (2016) done in the Midlands regions of Zimbabwe. It appears that most of the crocodiles are concentrated in the warmer areas with average temperatures (>26 °C) suitable for breeding purposes (Hutton, 1987). Logically, most of the studies on crocodile populations are also concentrated in the warmer northern and southern parts of the country mainly in the Zambezi Valley (including Lake Kariba) and Lowveld region (ZPWMA, 2015). In fact the crocodile studies (not necessarily the crocodile populations) are divided into two clusters by the watershed with one cluster of studies located in the Zambezi Catchment and the other on the Save-Runde Catchment (ZPWMA, 2015).

It is prudent to indicate that crocodiles are ubiquitous in protected and unprotected wetland systems of Zimbabwe and the division into two main study areas for the crocodile populations is rather artificial and in fact misleading to estimate the actual total population of the species in the country. In the protected areas, neither hunting nor egg collection is permitted unless it is for research purposes (ZPWMA, 2015). The numbers from protected areas form the reserve population which is the basis for the estimation of nationwide crocodile populations (Fergusson, 2006, 2010). The cooler but wetter Manicaland regions, and drier parts of Matebeleland regions are purported to have low numbers of crocodiles ostensibly because of the lower temperatures which do not favour high rates of breeding

(Games and Moreau, 1997), and the low water levels which do not guarantee breeding and nesting sites for the wild crocodiles respectively (Fergusson, 2010).

The Midlands region has a considerable population of crocodiles but there are no concerted studies except for the classic ones by Hutton (1984, 1987) and Hutton and Woolhouse (1989) and Fergusson (2010) which were conducted in the Ngezi Dam and by Sai et al. (2016) in the Sengwa and Kove Rivers. It is imperative to note that there are several protected areas, in the Midlands, Manicaland and Matebeleland South regions, which have suitable habitats and conditions for Nile crocodile proliferation (ZPWMA, 2015). However, population surveys are fewer in such areas for unexplained reasons (ZPWMA, 2015). The situation is more complex for crocodiles outside of protected areas where their distribution and populations are under threat from humans and exogenous factors e.g. siltation, low water flows, pollution and habitat desiccation (Zisadza-Gandiwa et al., 2013). There are no documented studies on the distribution of crocodiles outside of protected areas in Zimbabwe even in the more studied northern and southern areas (ZPWMA, 2015). This scenario implies an underestimation of their population because of lack of crucial information on the distribution of the species, and potential habitats. Thus, it becomes unscientific to just adopt the IUCN conservation classification status for the species without clear information on its distribution (Utete, 2020). It then necessitates for long-term surveys on the distribution of the species inside and outside of protected areas (ZPWMA, 2015).

Moreso, it becomes complex to indicate the hotspots where the crocodiles have the potential to interact with humans if there is a lack of information on its distribution, potential habitats and abundance making conservation impractical (Utete et al., 2017). Mostly, it is only after HCC reports when ZIMPARKS authorities ascertain the distribution of crocodiles in wetland systems outside of protected areas (Dickman, 2010). A pragmatic method to assess the distribution of the species is to identify and map potential and suitable breeding and nesting sites, spoors and denote them as crocodile hotspots in protected and unprotected wetlands (Utete et al., 2017). This is expensive to undertake on the ground (Balaguera-Reina et al., 2018). However, the modern application of GIS and Remote Sensing methods and use of Unmanned Aerial Vehicles (drones) to identify suitable sites is an innovative, cheaper (in the long-term) and efficient technique of mapping crocodile distribution currently implemented in other nations (Balaguera-Reina et al., 2018; Mangewa et al., 2019). Ezat et al. (2018) used UAVs to survey Nile crocodile populations in Lake Nyamithi in the Ndumo Game Reserve in South Africa. Ezat et al. (2018) and Mangewa et al. (2019) indicated that the potential of drones to provide real-time data estimate of crocodile population size, and measure the total length (TL) of individuals accurately and precisely at low costs could improve management actions, enable efficient monitoring of crocodile populations and avoid observer bias besides facilitating improved crocodilian survey techniques. So far examination of the distribution patterns has only concentrated on free-ranging wild stocks and excluded stocks in crocodile ranches in Zimbabwe which is addressed later. Suffice to indicate that ZIMPARKS authorities and Crocodile Farmers Association of Zimbabwe (CFAZ) monitor wild populations as they collect the breeding stocks and eggs (ZPWMA, 2015). This inevitably means that they have knowledge of the nationwide distribution of crocodiles in and outside of protected wetlands. However, this information is kept water tight and not readily available for any meaningful review of the distribution patterns of crocodiles in Zimbabwe (Revol, 1995; Fergusson, 2010). Nonetheless, from the few anecdotal studies examined it shows that most crocodiles are concentrated in the warmer regions of Zimbabwe. Thus, for future researches on crocodile populations the available studies must be adopted as the baseline data sources for the country (ZPWMA, 2015).

6. Nile crocodile populations in Zimbabwe

The Zimbabwe Parks and Wildlife Management Authority (ZPWMA or ZIMPARKS) monitors the free-ranging wild crocodile population in conjunction with the Crocodile Farmers Association of Zimbabwe (CFAZ), and by using spotlight counts, total day counts and nesting effort assessments they are able to provide population estimates (ZPWMA, 2006, 2015). Academic researchers are allowed to assess wild populations in protected areas with permission from ZPWMA (2015). Examination of researches on crocodile populations indicated a fragmented picture with no detailed long-term records. Regardless, several short-term surveys have been conducted over the years which can be scrutinised to estimate the total population. For the purpose of the study available records in the country i.e. in the Zambezi Valley, Gonarezhou and Midlands Regions were assessed to highlight the trends and challenges in estimating wild crocodile populations in Zimbabwe.

6.1. *Case study 1: Nile crocodile estimates in aquatic systems in the Zambezi Valley*

Available data indicated an average of 12 064 Nile crocodiles in the Zambezi Valley (Table 1). For the Kazungula area, the crocodile

Table 1
Crocodile estimates in the Zambezi Valley.

Year	Kazungula	Upper Zambezi	Kariba	Lower Zambezi	Mat North	Total
2000	93	453	7000	3559	12	11,117
2001	25	595	7708	3100	237	11,665
2002	19	509	8626	3014	130	12,298
2003	18	509	9177	2523	56	12,283
2004	18	645	10021	2214	56	12,954
Average	35	542	8506	2882	98	12,064
P value	0.06	0.82	0.06	0.06	0.73	0.08

(Source ZPWMA, 2006).

population declined by 62% from 93 in 2000–18 in 2004. In the Kazungula area, along the Zambezi River course bordering Botswana, parts of Namibia, Zambia and Zimbabwe, the fast flowing stretches support low crocodile numbers (ZPWMA, 2015). Intense anthropogenic infrastructural and technological developments are the main drivers of environmental change as they have destroyed suitable breeding and nesting sites hence the decrease in crocodile populations in the area (ZPWMA, 2015). Despite a marked decrease, the ZIMPARKS area manager indicated that the crocodile population in the Kazungula is rather stable (ZPWMA, 2006). Similarly, the crocodile population for 2000–2004 remained at an average of 98 with extreme outliers recorded in 2001 and 2002 in Matebeleland North (Table 1). The Matebeleland North crocodile population estimates are mostly derived from the artificially pumped water pans in Hwange National Park, and permanent stretches in the Gwayi and Deka Rivers (Msiteli-Shumba et al., 2017). These wetlands are located in arid areas with average rainfalls of 500–900 mm in good rainy seasons (Mugandani et al., 2012). Low water levels in pans and reduced flows in rivers cannot sustain high crocodile populations (Kofron, 1993) accounting for the low figures.

The crocodile population of the Upper Zambezi area has remained stable at an average of 542 (Table 1). Cross examination indicated that the crocodile population in the Upper Zambezi was robust for 2000–2004. This is partly because of the flat and undulating nature of the Zambezi River before it reaches Victoria Falls, which allows for the formation of suitable nesting and breeding habitats in the riparian zones (Wallace et al., 2011). Coupled with the territorial nature of the species where it chooses a suitable breeding and nesting habitat and dwells there for years, it partly explains the stable and uniform population recorded (Hutton, 1987; Kofron, 1993). For the middle Zambezi area represented by Lake Kariba, the average crocodile population over 2000–2004 was 8 506 (Table 1). Analysis indicated that there is a high crocodile population in Lake Kariba, with highest figure of 10 021 recorded in 2004, relative to the rest of the Zambezi Valley (Table 1). High crocodile populations in Lake Kariba relates to the potential suitable habitats offered by the large reservoir for basking, breeding, nesting and hunting and the high temperatures in the area (Hutton and Woolhouse, 1989). Moreso re-introduction of the species in the late 1990s as part of conservation efforts increased its population in Lake Kariba (Anderson and Pariela, 2005). Nile crocodile population is increasing at a rate of almost 1000 crocodiles per year in the lake and further conservative extrapolation implies that currently there could be about 18,000–21,000 crocodiles in the lake. Increased crocodile population in Lake Kariba is also related to the concerted efforts by ZIMPARKS authority and a majority (80%) of crocodile farmers who closely monitor collection of eggs and breeding stock and the mandatory release of some hatchlings back into the lake (Revol, 1995; Fergusson, 2010; ZPWMA, 2015). The lucrative nature of the crocodile ranching industry necessitated for joint efforts by ZIMPARKS and crocodile farmers and locals in monitoring crocodile egg poaching and persecution of the species especially in lake shores with intense human activity (Revol, 1995).

In the Lower Zambezi Valley, the average crocodile population was 2 882 (Table 1). Examination of available numbers reflected that the crocodile population has declined by 37% from 3 559 in 2000–2 214 in 2004 within Lower Zambezi Valley. The ZPWMA (2015) attributed the decreases to shifts in channel topography and morphometry because of intermittent releases of water through the Lake Kariba floodgates in times of heavy rains. However, the low flow and channel modification in the lower plains of the Zambezi River is causing a decline in suitable nesting and breeding habitats especially in low rainfall periods (Wallace et al., 2011). In an aerial survey Dunham et al. (2010) sighted 57 crocodiles against an estimated number of 510 in the Lower Zambezi area comprising Mana Pools National Park, Hurungwe, Sapi, Chewore, Charara and Doma Safari Areas. In this case, the low numbers can be attributed to the use of the aerial survey method and day counts which tend to underestimate the number of crocodiles which are more nocturnal (Dunham et al., 2015; Sai et al., 2016). An aspect of HCC cannot be down played as studies e.g. Musiwa and Mhlanga (2020) have

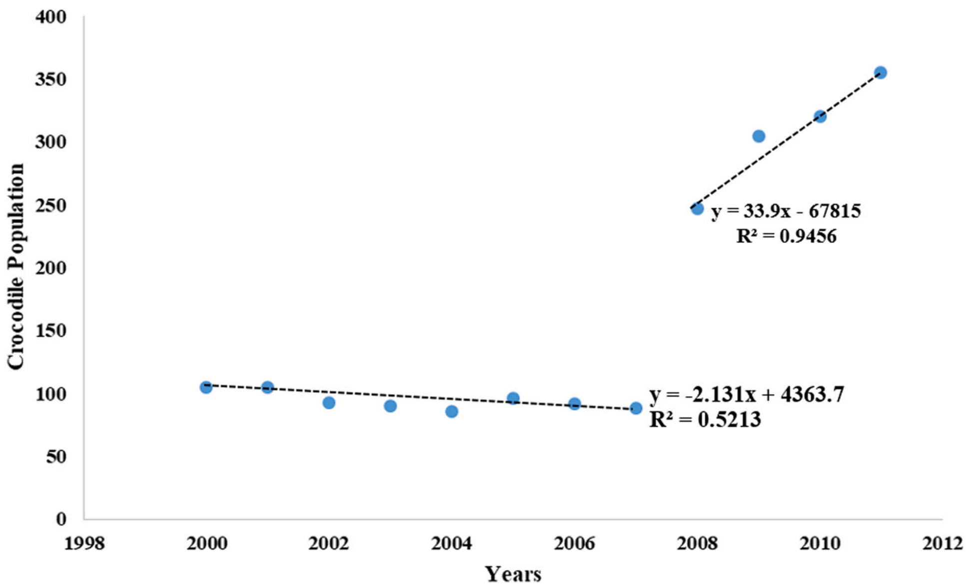

Fig. 2. Nile crocodile populations in Gonarezhou National Park in Zimbabwe over 2000–2011.
Source Zisadza-Gandiwa et al. (2013).

shown that humans encroach into wetland systems in the Lower Zambezi Valley and destroy crocodile eggs and suitable habitats. The key aspect is to review and assess the drivers of HCC and other contextual challenges causing crocodile population declines in the Lower Zambezi Valley which has a proximate riparian human community (Dunham et al., 2010, 2015).

Total estimates for the Zambezi Valley suggested slight increases in crocodile populations in the wetland systems (Table 1). Regardless, there were no significant statistical (Kruskal ANOVA, $p > 0.05$) differences in population fluctuations in the Zambezi Valley indicating that the crocodile population has remained relatively stable. Nevertheless, for future crocodile conservation efforts, there is a need to first establish the carrying capacity of the wetlands which can only be done through exploration, identification and mapping of suitable habitats (Fergusson, 2010). Afterwards, contextual challenges peculiar to the area need to be assessed and their impacts on the crocodile population evaluated. Thus, continuous collection of crocodile eggs and breeding stocks in Lake Kariba without clear distribution maps and accurate population records will be detrimental for crocodile conservation in the future (Revol, 1995; Fergusson, 2010; IUCN, 2017).

6.2. *Case study 2: Nile crocodile estimates in the Gonarezhou National Park*

For the Gonarezhou National Park (GNP), some crocodile surveys were carried out by ZPWMA in conjunction with private organisations from 2000 to 2007, 2013, and in 2008–2011 by Zisadza-Gandiwa et al. (2013). The first set of surveys by ZPWMA and private organisations concentrated on Save and Runde Rivers and amalgamated the overall crocodile population. The second data set by Zisadza-Gandiwa et al. (2013) added Mwenezi River to the survey. The first data set indicated that crocodile populations decreased from 2000 to 2007 in the wetlands of GNP (Fig. 2). The second data set showed that the crocodile populations increased in the wetlands of GNP (Fig. 2). The importance of the first data set is that it exposed several points: Firstly, the crocodile populations in the major rivers i.e. Save, Runde and Mwenezi decreased at a marginal rate of almost 5% per year from an estimated record of 105 in 2000–88 in 2007. Secondly, there was no significant statistical change in the crocodile population in GNP from 2000 to 2007. The first point suggested the influence of exogenous factors on crocodile populations inside and outside the GNP. These factors included increased silt load from surrounding communities, pollution loads, habitat degradation in the river stretches outside the park, and retributive persecution by humans outside the park (Zisadza-Gandiwa et al., 2013, 2016; Matseketsa et al., 2019). Moreso, crocodile egg poaching and juvenile predation is prevalent in and outside the GNP (Gandiwa, 2011). However, the key driver of crocodile population declines in GNP is the human settlement encroachment into the riparian zones of the Mwenezi, Runde and Save Rivers which has increased HCC and siltation in the rivers degrading habitats and reducing recruitment (Gandiwa, 2011, 2012).

Statistically, there were no significant differences in the decrease of the crocodile population over 2000–2007 as the average figure hovered at 94 (Appendix 1). However, for conservation of small numbers of keystone species, the statistical connotations proffer intangible solutions in the case of decreases (Fergusson, 2010; Utete, 2020). Pragmatically, it entails for an assessment of suitable and potential nesting and breeding habitats in wetlands inside and immediately outside the GNP and an estimation of the carrying capacity of the sites (Utete, 2020). Examination of available crocodile survey data indicated complete exclusion of permanent and even temporary pans and pools which are potential crocodile habitats inside the GNP (Kofron, 1993) and this could have led to an underestimation of the actual population (Zisadza-Gandiwa et al., 2013). Afterwards, the focus should shift towards the actual observed population modelled against the carrying capacity for implementation of corrective conservation efforts (Craig et al., 1992; Loveridge and Hutton, 1992; Taylor et al., 1992; Bishop et al., 2009).

The second data set by Zisadza-Gandiwa et al. (2013) showed crocodile population increases from 2008 to 2011 in the main three rivers of GNP. Zisadza-Gandiwa et al. (2013) further indicated that the Runde River, a tributary of Save River, had the highest crocodile population, as it provided 78.2% ($n = 960$) of all of the crocodile sightings from 2008 to 2011. Mwenezi River constituted 14.6% ($n = 180$) of the crocodile sightings during the study period whereas Save River had the lowest proportion of sightings, only 7.2% ($n = 88$). Runde River has suitable habitats for crocodile proliferation whereas the upstream Manyuchi Dam alters the flow regime and results in low water volumes in Mwenezi River dissuading crocodile habitation whilst only a small portion of the Save River passes through the GNP to make a meaningful contribution to the total crocodile population (Zisadza-Gandiwa et al., 2013). Save River is marginally bigger relative to Runde and Mwenezi Rivers yet it contributes the least numbers of crocodiles to the GNP owing to the rocky edges prevalent along its course which offer less sites for basking, courtship, mating and egg laying (Kofron, 1989, 1993). There is high silt and sediment deposition in the three rivers because of increased (legal and illegal) settlements in riparian zones which degrades suitable habitats (Gandiwa, 2011; Matseketsa et al., 2019).

In as much as both researches used the aerial survey method, the first data set was obtained from studies specifically targeting other species i.e. elephant populations in the GNP and crocodile sightings were secondary (ZPWMA, 2015). Contrastingly, the second data set was obtained from concerted efforts towards estimating crocodile abundances, distribution and population trends (Zisadza-Gandiwa et al., 2013). For the first data set, there was additional use of nesting sites to estimate crocodile populations which inevitably led to different results relative to the second data set obtained using aerial surveys of whole animal counts (Bourquin and Leslie, 2012). Differences in the crocodile estimates for the two data sets could also be due to the surveying of only two rivers for the first data set. However, if conservation efforts were to be implemented based on the first data set it would imply that the crocodile population in GNP has decreased necessitating for more protective measures (Fergusson, 2010). The second data set would imply that the crocodile population in the GNP increased and would ultimately stabilise at a threshold carrying capacity (IUCN, 2017). Excluding intrinsic and exogenous factors, the two diametrically opposed data sets subtly reflected that either there were no more suitable habitats in GNP hence the decrease in the crocodile numbers from 2000 to 2008, or there were actually suitable habitats waiting to be utilised by the species which could account for the sharp increases in the numbers from 88 in 2008–355 in 2011 (Appendix 1).

The average number of crocodiles in the wetlands in the GNP from 2008 to 2011 was 307 which is higher than the average figure of

94 in the 2000–2007 period. This study does not rule out the positive impacts of increased anti-poaching patrols and enforcement of stringent protection of wetland systems in and outside the GNP on the crocodile population especially juveniles after the coming in on board of private organisations such the Germany based Frankfurt Zoological Society which helped conservation efforts particularly from 2008 onwards (Gandiwa, 2012; Zisadza-Gandiwa et al., 2013; Musakwa et al., 2020). For conservation practitioners and policy makers it means that conservation efforts towards a targeted species must only be implemented using rigorous long term species specific data sets obtained using similar or comparable data collection methods (Utete, 2020).

6.3. Case study 3: Nile crocodile estimates in the Midlands Region

In the Midlands region, crocodile surveys were implemented in Ngezi Dam by Hutton (1984, 1987); Hutton and Woolhouse (1989); Fergusson (2010) and Sai et al. (2016) in Sengwa and Kove Rivers in Sengwa Wildlife Research Area. Hutton (1984) indicated that there were over 300 crocodiles in Ngezi Dam (Appendix 1). In a follow up study Hutton (1987) suggested the crocodile population to be almost 228 in the reservoir. Furthermore, in follow up surveys crocodile populations were estimated to be between 121 and 230 in the dam (Fergusson, 2010; ZPWMA, 2015. This reflected a steady decline in the crocodile population in the reservoir over 1984–2018. Fergusson (2010) indicated that there was a once-off release of crocodiles into the Mvuma and Ngezi Rivers, by a disgruntled evicted white farmer who reared the reptiles, at the peak of the land reform program in 2001 causing an increase in Ngezi Dam. Nevertheless, a breakdown in some sections of the protective fence around Ngezi Recreational Park has led to humans and livestock encroaching into the reservoir with consequent increases in crocodile-livestock conflicts (CLC) and HCC (ZPWMA, 2015). The case of Ngezi Dam is a classic case of humans perceiving protective fences as barriers to accessing natural resources (Matseketsa et al., 2019; Musakwa et al., 2020). Nonetheless, the same fences serve to protect (and minimise antagonistic contact among) humans, livestock, crocodiles, and wildlife, which leads to HWC, HCC and CLC (Gandiwa, 2011). Examination of the studies indicated that low recruitment rates of crocodiles and decreased populations are partly because of reduced habitat sizes, food availability and reproductive activity caused by high drawdowns in the reservoir, and increased HCC attributed to vandalism of protective fences (Fergusson, 2010). Regardless, the studies have not been exhaustive to pinpoint the exact (and interactive effects of the) drivers of declines in crocodile populations in the reservoir.

In a once-off study Sai et al. (2016) estimated the crocodile populations in Kove and Sengwa Rivers in Sengwa Wildlife Research Area. This study recorded a total of 82 crocodiles in both rivers within and outside the protected area. The populations in the two rivers exhibited a predominantly juvenile based structure which indicated a healthy growing population especially inside the protected area (Hutton, 1987; Kofron, 1993). The study coincided with the breeding period which is normally towards winter i.e. May-August in Zimbabwe hence juveniles were more prevalent relative to hatchlings and adults (Kofron, 1989). Regardless, these two studies in the Midlands region aptly indicated that there are potential suitable habitats and viable crocodile populations inside the country apart from the northern and southern parts where most of the surveys are concentrated. What has been lacking are long-term crocodile population records even for well protected large and small national and recreational parks nationwide (Fergusson, 2010). Examination of literature revealed fewer surveys on wild crocodile stocks especially for those in localities outside protected areas relative to crocodile population surveys in the more established national parks mainly Lake Kariba and Gonarezhou National Park (ZPWMA, 2015). This implies gross underestimation or even overestimation of crocodile populations in wetland systems without solid evidence from long term scientific surveys. The key question is what data are being used to categorise the conservation status of the crocodile population in the country? As far as the wild stock is concerned there is a lack of clear data on the distribution and location of suitable habitats and reliable records on population trends in the country. Basing national population estimates on few anecdotal researches is a dangerous proposition for the conservation of any species moreso for the Nile crocodile which is a keystone species of ecologic and economic importance in developing countries like Zimbabwe (IUCN, 2017).

From the three case studies it is easily discernible that estimating crocodile abundances in both lotic and lentic systems is challenging. Like all surveys, crocodile population estimates suffer from biases and survey errors (Ferreira and Pienaar, 2011). The three case studies are based on short-term data thus, it is very difficult to ascertain the accuracy of trends and patterns inherent. The first issue inducing uncertainty in crocodile population estimation is simply availability bias i.e. the proportion of time a crocodile is available to be sampled (Redfern et al., 2002; Da Silveira et al., 2008). Crocodiles are predominantly nocturnal but are also very active and rather shy during the day and thus, for more accurate results there is a need for both day and night surveys (Ferreira and Pienaar, 2011). Moreso, the use of a combination of techniques ranging from ground and aerial surveys (that are hazardous at night) may reduce availability bias. The three cases studies, besides the GNP by Zisadza-Gandiwa et al. (2013), were mostly based on ground surveys and thus, the accuracy of the results and trend detection may not be truly reflective of the actual crocodile populations in the sampled areas. It is imperative to indicate that some perceptions of trends in populations is a key source of information for determining the conservation status of species under the IUCN Red List assessment (IUCN, 2017). However, biases and errors induce uncertainty in estimates which in turn is a key element in determining trends (Gerrodette, 1987). On a severity scale, this carry risks of making a Type 1 error of assuming and concluding that there is a trend when there is none. For the three case studies, statistical analysis indicated that there were no significant differences (Kruskall ANOVA, $p > 0.05$) in the crocodile populations sampled in consecutive years. However, the data is based on few sampling points such that merely glossing over the figures can grossly mislead one to conclude that the populations are declining, stable or increasing. Nonetheless, if one considers that the data are based on mean annual averages that mask the monthly, weekly or daily variations then there is also a risk of committing Type 11 error i.e. concluding there is no trend when there is one (Gerrodette, 1987; Ferreira and Pienaar, 2011). The implications of committing both types of errors are dire to the conservation of iconic species such as Nile crocodiles where the statistical significance of population size trends is immaterial but rather where every individual must be accounted for in and outside of protected areas. This becomes even more important in the case where the actual figures show sharp declines in the population sizes (Fergusson, 2010; Utete, 2020).

The key then is to instigate long-term national surveys using a combination of modern ground and aerial technology to avoid biases and errors. This must be buttressed by the use of generalised models to derive correction factors. However, knowing trends is meaningless if the reasons for those are not understood (Ferreira and Pienaar, 2011). In most cases, population estimates need to be complemented by additional information such as the existence of a benchmark estimate i.e. either a valid and reliable documentation of what the population estimates were or what the current crocodile management policy planned to achieve in Zimbabwe. In either case, there is no clear current or even dated data on the crocodile (both free-ranging and farmed) population estimates for the country or is the Crocodile Management Plan of Zimbabwe cogent on the actual optimal estimates for the carrying capacity target. Ideally, modelling of contextual environmental drivers of change and the current estimate of crocodiles is important to predict future trends in populations that will enable crafting of proactive and responsive conservation measures for Zimbabwe.

7. Ranching as a Nile crocodile population changer

Crocodile ranching or farming is a lucrative activity worldwide which provides meat for household consumption, employment, income and skins for the luxury leather industry (Revol, 1995). Ranching confers economic value to crocodiles (Fergusson, 2010), helps in dispelling crocodile "myths" and "fears", thus, changing the perceptions of people towards the species with positive conservation benefits in some cases (IUCN, 2017). The first crocodile ranching initiative involving artificially incubating wild collected eggs and releasing a proportion of the ranched stock into the wild was licenced in 1965 in the then Rhodesia (Revol, 1995; ZPWMA, 2015). From 1985 there were at least 46 registered crocodile farms in the country (CSG, 2004). Currently, there are about 18–26 crocodile farms licenced to operate in Zimbabwe with the bulk of the businesses located in hot regions such as Kariba, Binga, Victoria Falls and the Lowveld, with smaller concerns also scattered in Deka (Hwange), Mhangura and Chinhoyi (CFAZ, 2016). More than five enterprises are at 80% operational capacity with the rest operating below 30% capacity utilisation (CFAZ, 2016). The fact that crocodile farming is a high end industry requiring expensive capital input for feed, infrastructure, water availability, disease prevention, meat production and leather curing and expert management implies that few locals can afford to penetrate the industry (Revol, 1995; CFAZ, 2016). In fact crocodile ranching is considered as a preserve of the elite few (FAO, 2013) with low entry opportunities for impoverished locals. Although some community based crocodile farming initiatives have been mooted these have not been as successful as small-scale fishing cooperatives with low capital input requirements (FAO, 2013). Zimbabwe sold 76,567 skins in 2000, and there has been a steady decline in the range 73,000–64,000 of skins traded over the years (ZPWMA, 2015). Marketing of crocodile products is hampered by a growing anti-crocodile product trade crusade spearheaded by animal rights movements and depressed global markets (CSG, 2009; IUCN, 2017). The net result has been a reduction in crocodile ranching activities not only in Zimbabwe but globally (IUCN, 2017).

The crocodile producer association, the Crocodile Farmers' Association of Zimbabwe (CFAZ), is responsible for compiling detailed production data which is forwarded to the Management Authority, the ZIMPARKS, which also acts as an impetus for monitoring both wild and farmed crocodile populations (CSG, 2009; Fergusson, 2010; IUCN, 2017). From 1985–2002 crocodile production increased sharply and standards improved although there have been problems in recent years associated with the chaotic land reform program and a hyperinflationary economy (CSG, 2004). The number of CFAZ members holding stock and producing crocodiles has decreased from 47 in 1992–18 at present (CSG, 2004). In 2014, crocodile farms held a total of 115 608 hatchlings, 207,206 'rearings', 3,890 'growers' and 6 871 'breeders. In the same year, 78% of eggs used in the farms had originated from farms themselves, with the remainder originating from the wild (ZPWMA, 2015). Normally, permits for egg collection are issued on the condition that stations submit monthly stock returns (Hutton and Child, 1986). The idea is to ensure that only eggs and no live crocodiles are collected for ranching purposes which would lead to depletion of the wild stocks by ranchers who may want to cut costs (ZPWMA, 2015). To buttress this statute, it is subtly mandatory for all crocodile farms to be part of the CFAZ and their members must be accompanied by rangers from the ZPWMA when collecting eggs from the wild, and tagging of breeding stocks taken from ranches is also compulsory (ZPWMA, 2015). In retrospect, these requirements are very strategic conservation initiatives as they ensure that data on nests recorded during egg collection is used to assess the wild crocodile populations in Zimbabwe. The catch is that all permit holders who are crocodile farmers are required to remit 5% of the eggs collected back into the wild as juvenile crocodiles (Fergusson, 1992) in the event that the wild stocks begin to decline (ZPWMA, 2015).

Further examination of some of the requirements by ZIMPARKS indicated flaws in their effectiveness for assessing and conserving national crocodile populations. Firstly, the crocodile farmers are in business and would rather collect the eggs (60 000–80 000 per year) from established sources in most cases (60–80%) either from Lake Kariba shoreline egg collectors or re-use eggs from the breeding stocks already established at their farms (CFAZ, 2016). Consequently, there is gross underestimation of the national crocodile population which may be detrimental to the species if the potential suitable habitats are not adequate in the wild. However, this bodes well if there are adequate numbers of crocodiles close to the threshold carrying capacity of the country. Literature search indicated that the carrying capacity for crocodiles has never been explicitly revealed in Zimbabwe simply because of lack of data on the distribution of suitable habitats for the species (Zisadza-Gandiwa et al., 2013). Moreso, lack of knowledge of the actual national carrying capacity for the crocodiles is detrimental in cases where the juveniles are released into the wild habitats as it leads to increases of wild stocks in and outside of protected areas with negative ramifications for the survival of the species, and has the potential to increase HCC in surrounding communities (Dickman, 2010). In some cases, high mortalities of released juvenile crocodiles defeats the conservation purpose (ZPWMA, 2015).

The collection of *C. niloticus* eggs for ranching led to economic benefits for local communities providing some motivation for improving the conservation status of the species (ZPWMA, 2015). What may be clear from this review is that there is good record keeping and documentation among crocodile farmers with indications that the local population (juveniles, sub adults and adults) may be close to 300,000–400,000 (Revol, 1995; FAO, 2013). Regardless, there are some weaknesses in the crocodile ranching industry in Zimbabwe. Loveridge (1996), CSG (2004, 2009), and Fergusson (2006, 2010) indicated a plethora of complexities comprising: a) the lack of adequate and current management plans; b) poor communication between CFAZ and the MA; and, c) inadequate monitoring of the wild population. It must be pointed out that some of the concerns were addressed by a workshop held in 1996 during which The revised Policy and Plan for Crocodile Management in Zimbabwe was developed and subsequently approved by the Minister of Environment and Tourism in May 1997 (CSG, 2004). The policy fundamentally aimed to zone crocodile habitats geographically and contextually cognisant of the unequal socioeconomic developmental rates in Zimbabwe in order to provide appropriate levels of protection and utilisation (CSG, 2004).

However, the implementation of the policy has not been without glitches as the CFAZ no longer has a significant role in the marketing of crocodile products (externally and locally) which is highly regulated and monitored at local levels by ZIMPARKS and internationally by CITES and responsible watchdog organisations such as the Crocodile Specialist Group among others (CSG, 2009; IUCN, 2017). Moreso, the management authority (ZIMPARKS) has been transformed into a self-funding authority with a resultant reduction in investment in crocodile management. Most of the crocodile issues are now dealt with by the Scientific Services Division through the Aquatic Ecology Section. Regardless, most of the individuals within ZIMPARKS staff do not have specialist knowledge of crocodiles and the crocodile ranching or farming industry (CSG, 2004; Fergusson, 2010). The net effect has been a breakdown in communication and monitoring in the crocodile ranches, with loss of institutional memory, lax record keeping and some mistrust between the industry and the regulators (SCG 2004). However, what is commendable is that the 1996 crocodile management plan actually remains in effect hirtherto with flexible adjustments effected along the way (ZPWMA, 2015), and has been adopted as a viable model in other countries such as Namibia, South Africa and Zambia (CSG, 2004).

The crucial aspect is that crocodiles on ranches are for business and must never be misconstrued to be part of the wild stocks but rather as a reserve for depleting wild stocks. A more tenable situation is to assess the wild stocks as a stand-alone unit and keep the populations of farmed crocodiles as a reserve gene pool in case of wild population crashes. Currently, conflation of the wild population, and the farmed population (which is more or less stable and well taken care of) is misrepresenting the natural situation in and outside of protected areas (Fergusson, 2010; Marais, 2014). Some countries e.g. Namibia, Botswana and South Africa keep separate records of the wild and farmed crocodile stocks. As a result they have revised the conservation status of the species based on the population of wild stocks (CSG, 2004; IUCN, 2017). The true population of crocodiles (wild plus farmed) in Zimbabwe has never been clearly stated with conflicting numbers in the public domain. Thus, it is not clear how the crocodile ranching industry has signficantly influenced the wild crocodile populations for Zimbabwe This review advocates for a clear separation in records for the wild and farmed crocodile stocks in Zimbabwe and for all purposes strive to use the population statistics for the wild stocks in classifying the conservation status of the species.

The study used the Non-Detrimental Finding Process developed by the IUCN (see Rosser and Haywood, 2002) to contextualise the threats of egg harvesting on crocodile populations. Taking collection of eggs from the wild for crocodile ranching for export as a threat for crocodile populations, and using the average number of 70,000–73,000 skins sold per year as a representation of the average number of eggs that were harvested from the wild and successfully hatched (CFAZ, 2016) against an estimated maximum range stocking capacity in the wild and the crocodile farms of 250,000–300,000 adult crocodiles (ZPWMA, 2015) then it implies that per year at best there is a wild stock yield range of 24.3–29.2% that is harvested and traded. Using the NDFPs sensitivity criteria; firstly, in captivity and even in the wild the crocodiles (and eggs) are fairly adaptable to the environment though they are sensitive to diseases and water temperature (which must be >26 °C) for optimal reproductive purposes. Secondly, considering ecological biomass requirements, the harvested eggs per se do not occupy large spaces. Thus, egg collection or harvesting does not imply a significant reduction in free-ranging wild crocodile population over a given space. Thirdly, on average an adult female, with an average lifespan of 40–50 years, lays 25–30 eggs per clutch, and collection of 70,000–73,000 eggs per year for ranching implies that 2700–3000 breeding free ranging female crocodiles can satisfy the needs of the crocodiles farmers every year. Factoring the strict control, protection and monitoring of egg collection and the regular enforcement of the no access zone limits imposed on breeding sites and nests by ZPWMA authorities and the mandatory release of 5% of the juveniles back into the wild by crocodile ranches it means that egg harvesting does not constitute a major threat to the wild stocks of Nile crocodiles at the current yield rate of 24.3–29.2% for Zimbabwe. Though it must be noted that the NDFP process can better be used to contextualise threats when there is adequate current long-term baseline data and information on the Nile crocodile population estimates and intensity of exploitation (Rosser and Haywood, 2002).

8. Nile crocodile sport hunting in wetland systems of Zimbabwe

In Zimbabwe, legal crocodile (sport or trophy) hunting is limited and mainly concentrated in the safari, communal, forestry, and recreational parks, and on private land such as conservancies (Fergusson, 2010; IUCN, 2017). Crocodile hunting is controlled through annual quotas allocated by the CITES (Convention on International Trade in Endangered Species) to which Zimbabwe is a signatory (CSG, 2009; ZPWMA, 2015). The current Nile crocodile quota for Zimbabwe ranges from 150 to 250 with strict and clear specifications on the length (minimum hunted length is 3.2 m), age (> 3 years) and sex (preferably dominant males and strict avoidance of breeding females) of the crocodiles which can be hunted (CSG, 2009). Strict enforcement of the quota is done by ZIMPARKS which is the CITES management authority (MA) in conjunction with the police force and legal institutions where and when the quota and its provisions are violated. This ensures strict adherence to the CITES provisions (ZPWMA, 2015). Coupled with the fact that the quotas are allocated in agreement with national wildlife authorities this guarantees that the conservation aspects of crocodiles are prioritised (IUCN, 2017). The assumption is that there is adequate data on the distribution, abundance, population trends and conservation challenges for the species, and the allocated quota guarantees species sustainability (IUCN, 2017). What should be clear is that CITES provisions only indicate a certain number, and some of the key biological attributes for a specific species to be removed (IUCN, 2017). Critical analysis of the CITES provisions indicated that they are not always very clear for each species (CSG, 2004).

For crocodile hunting there is intense obsession with the morphometric parameters yet the genetic aspects of the hunted animals are never considered (Fergusson, 2010). Recently, crocodile sport and trophy hunters have become sophiscated as they insist, in addition to other morphometric parameters, on the high quality of the skin, rate of tail wagging (a sign of aggression and virility), wider head frontal view (an indication of genetic superiority) and gait of the crocodiles (which indicates sturdiness) before settling for a kill (IUCN, 2017). This means that genetically superior males are being harvested more often leaving the inferior males for breeding in the wild. The traits aimed for by skilled hunters are not even explicitly outlined in the appropriate sections of the Appendices of the CITES (CSG, 2004, 2009; Fergusson, 2010; IUCN, 2017). Regardless, the skilled hunters and clients alike are familiar with the morphometric traits which infer genetic superiority in crocodiles and hence there is a disjuncture between the provisions on paper and the actual crocodile trophy harvested in the wild (IUCN, 2017). The assumption that targeting dominant males has no impact on the breeding females is rendered void as long as the superior males are targeted by hunters leaving less dominant (but not always genetically inferior) wild stocks which may be prone to diseases, parasites and changes in water quality and climate vagaries (Fergusson, 2010; IUCN, 2017). However, the less inferior or non-dominant males also need to breed for the widening of the gene pool and strengthening of the species' resilience and adaptability to environmental variability (van Asch et al., 2019,).

Furthermore, it is imperative to indicate that within Zimbabwe, the crocodile quotas allocated to different land categories differ and follow established crocodile distribution patterns (ZPWMA, 2015). Areas deemed to have higher distribution and abundance of crocodile populations are allocated higher crocodile sport hunting quotas. Considering the paucity of solid data on the country wide distribution and abundance of crocodile populations this is misleading as the more studied areas (which does not always imply the more crocodile populations) in this case the northern regions e.g. Lake Kariba and the Zambezi Valley, and the southern parts e.g. Gonarezhou National Park and its surrounding areas tend to get the highest quotas (CSG, 2004). Although this may be appropriate to a certain extent given the available baseline data on crocodile populations and distribution, the onus is to ascertain the current distribution, abundance, population trends and challenges in the other less studied (which does not always mean less crocodile populations) areas and in wetland systems outside protected areas (Zisadza-Gandiwa et al., 2013; Utete, 2020). In areas with intense crocodile hunting there is a need to model the responses of the remaining crocodile populations to prolonged sustainable harvesting or hunting. The available models of Nile crocodile population growth and responses in Zimbabwe to prolonged sustainable harvest provided by Craig et al. (1992) and Loveridge and Hutton (1992) and Taylor et al. (1992) indicated robust recruitment and replacement in exploited areas. However, more recently, Bishop et al. (2009) discussed the reduced effective recruitment and disproportionate population structures in the overexploited Okavango Delta indicating a classic case of allowing legal hunting without clear evidence on the abundance and distribution of crocodiles in an area.

Using the NDFP (Rosser and Haywood, 2002) to assess the effects of trophy hunting as a threat to the Nile crocodile population it is important to assess the quota levels and the recovering and recruitment levels of free ranging crocodile populations for Zimbabwe. The Nile crocodile populations are in CITES Appendix II for Zimbabwe and the country has been allocated a quota of a minimum 1 600 wild specimens for sale and commercial purposes (CITES, 2021). Taking a maximum of 50,000–70,000 free ranging wild stocks (excluding farmed crocodiles) this implies that Zimbabwe has a leeway to harvest at least 3.2% of the free ranging wild crocodile and strictly only males for commercial purposes mainly through trophy hunting. This exploitation rate as indicated by Loveridge and Hutton (1992) and Taylor et al. (1992) is adequately offset by the annual recruitment of wild crocodiles where an average population of females, which are strictly forbidden from harvesting by ZIMPARKS regulations, can easily replenish the stocks. Though it must be noted that the females start reproducing at 12–16 years old and it nests 2–3 times per year from the onset of sexual maturity in optimal environmental conditions. Applying the Non-Detrimental Finding Process (NDFP) basic technological requirements and tenets ranging from; an adaptable life history, resilient sexual strategy, minimal (though fragmented and less reliable) documented wild stock data, strict enforcement and control and astute management of allocated CITES quota of 150-250 and a minimum threshold allowed>1600 wild stocks, and no harvesting of breeding or non-breeding females regulations imposed by wildlife authorities (ZPWMA, 2015) appear to be key to assessing the threats of trophy hunting and harvesting on wild crocodile stocks in Zimbabwe. Overall, the allocated CITES 2021 Nile crocodile quota for Zimbabwe (150-250 with a minimum threshold allowed> 1600 crocodiles per year) is rather sustainable and well within the carrying capacity of the assessed wetland systems in the country. It seems the exploitation intensities of both crocodile egg harvesting and trophy hunting for sale and commercial purposes is not detrimental to the free ranging wild Nile crocodile populations in the country. However, for a more robust assessment of the non-detrimental exploitation effects the data on export levels, capture effort and the existence of safe populations of crocodile in and outside of protected areas needs to be availed in future studies.

9. Human-crocodile conflicts in wetland systems of Zimbabwe

The Nile crocodile (*C. niloticus*) is the only true crocodilian species extant in wetland systems of Zimbabwe (Kofron, 1993). Apart from wetlands located in protected areas most aquatic systems are surrounded by permanent rural, urban and peri-urban communities in Zimbabwe (Utete, 2020). This normally leads to competition for water and fisheries resources among a growing human population, slow moving livestock e.g. cattle, goats, chickens, and predatory crocodiles (Musambachime, 1987; Anderson and Pariela, 2005). Activities such as bathing, water abstraction for domestic and agricultural uses, reed collection, and religious rituals like water based baptism, risky fishing behaviour e.g. use of seine nets, gill nets, spears and baskets, laundry, recreational activities such as canoeing, close edge angling, and swimming among others expose humans to crocodile attacks in wetland systems (Musambachime, 1987; Chihona, 2014; Pooley et al., 2019). Crocodiles attack humans on an opportunistic basis even when there is adequate natural food in the aquatic system and this is related to diet changes as they grow from juveniles where they prefer insects, arthropods and small invertebrates into the adult stage where they prefer large mammals, fish and birds (Cott, 1961). Regardless, crocodiles are prone to retributive persecution by humans when they encroach into communities and attack humans and their livestock (CSG, 2004; McGregor, 2005; Chihona, 2014). Most of the threats to crocodile populations are attributed to destruction of crocodiles, their nests, eggs and killing of juveniles by humans (ZPWMA, 2006). Thus, there is a two-way interaction that inevitably leads to antagonism with injury, casualties, and loss of property, crops, livestock and livelihoods (Musambachime, 1987; Fergusson, 2004; Pooley et al., 2019).

Analysis of literature on HCC in Zimbabwe indicated an increasing trend with frequent attacks in shallow wetland systems especially those in close proximity to protected areas and in less developed rural areas where people directly access water for daily use (Gandiwa, 2011; Zisadza-Gandiwa et al., 2016). The nation has been recording an average of 25–30 human deaths attributable to crocodiles from 1965 onwards (ZPWMA, 2015). Nonetheless, records of HCC in the country are fragmented and anecdotal with media reports dominating information dissemination on HCC in unprotected wetland systems (Pooley et al., 2019). It appears that decent records of HCC coincide with the period when the first crocodile ranches were set up as the initial stages involved exploitation of wild breeding stocks with inevitable conflicts in the process (Revol 1995; Anderson and Pariela 2005; McGregor 2005). This implies that wildlife authorities became more seriously involved in crocodile conservation after the setting up of crocodile farms and subtly reflected that they are partly reliant on data from the CFAZ moreso as it pertains to monitoring abundance and distribution of wild stocks (ZPWMA, 2015). However, what is baffling is that even though crocodiles account for at least 42–50% of HWC reported to the wildlife authorities they are not placed in the six member List of the Most Dangerous Animals (ZPWMA, 2015). As alluded to earlier in this review this aberration requires urgent redress.

Furthermore, there are reports of sporadic crocodile attacks on humans and livestock and home invasion in communities proximal to wetlands mainly during the rainy season when the water levels rise and force outward migration of the species with a potential for HCC (Fergusson, 2004, 2010; McGregor, 2005; Gandiwa, 2012; Chihona, 2014). In a curious case reported by almost all national tabloids, a crocodile escaped from the flooded Chiredzi River and tracked 16 km in a storm drain leading into an overcrowded Chiredzi District Hospital full of COVID-19 patients in Chiredzi District in the Lowveld area of Zimbabwe (The Herald, 2021). Though there were no casualties such a case highlights the varied and peculiar nature of HCC which assumes several forms (Anderson and Pariela, 2005; Dickman, 2010). There is no particular pattern in HCC which assumes varied typologies (Le Bel et al., 2011), is circumstantial, contextual, and has different consequences besides death and injuries and destruction of properties (Dickman, 2010; Pooley et al., 2019). This intrinsically complicates HCC management especially in wetlands located just outside of protected areas (Lamarque et al., 2009; Gandiwa, 2011; Matseketsa et al., 2019).

In a bid to increase crocodile populations conservationists implemented a successful re-introduction program in Lake Kariba in Zimbabwe (Anderson and Pariela, 2005). However, this resulted in an unprecedented increase in crocodiles leading to high prevalence of HCC (McGregor, 2005). This case indicated that appropriate and well-meaning and at times successful conservation measures on a species may later on induce other complexities. It exposes the need to involve local communities at the inception phase of conservation initiatives as neglecting human concerns leads to negative experiences and perceptions and hurt-rage towards the species which derails the conservation process in the long term (IUCN, 2017). Careful analysis of media reports on HCC indicated a lopsided view where the crocodiles are labelled as the instigators, and humans or livestock as the victims yet in most cases humans and livestock would have invaded wetland systems which are the natural habitats of crocodiles (Utete, 2020). This invariably creates negative perceptions among communities and leads to persecution of crocodiles threatening their conservation status (Pooley, 2016a, 2016b). It is important to indicate that HCC is rather a generalisation of the interaction between humans and crocodiles which is mostly portrayed as antagonistic as in some cases humans positively interact and co-exist with crocodiles e.g. through cultural and religious uses and ecotourism purposes with mutual benefits (Pooley, 2016b).

Underreporting, late reporting and overreporting of HCC is prevalent in communities close to wetland systems and limits the inferential use of existing HCC data in conserving the Nile crocodile population (Pooley, 2016a). Thus, rather than giving a mathematically precise output on the exact numbers and statistics of HCC this review aimed to stimulate qualitative insights and contextual overviews of HCC in wetland systems across the country. However, the few cases cited above indicated the need for mapping social, environmental, cultural, economic and ecological dimensions of human-crocodile encounters in space and time (Anderson and Pariela, 2005; Chihona, 2014); testing theories on the influence of biophysical factors on the seasonality of crocodile bites on humans and livestock (Wallace et al., 2011, 2013); and infographics as a means of exploring and communicating about conflictual human-crocodile relations in Zimbabwe (Pooley, 2016b).

The HCC management and resolution mechanisms are compounded by the fact that past crocodile conservation programs were developed by ecological experts without reference to local perceptions and attitudes in Zimbabwe (Taylor et al., 1992). Contemporary adaptive HCC management strategies and crocodile conservation efforts integrate and provide tangible economic benefits to local communities through initiatives such as CAMPFIRE i.e. the Communal Areas Management Programme for Indigenous Resources (Fergusson, 2004; Dzoma et al., 2008; CSG, 2009; Gandiwa, 2011; Tchakatumba et al., 2019). The weaknesses of adaptive management in crocodile conservation stems from multiplicity and confused and vague role allocation by wildlife authorities and low or limited input by locals (Anderson and Pariela, 2005; Fergusson 2009). Moreso, the fact that in Zimbabwe, conservation authorities maintain strict regulations on wildlife conservation skewed towards conserving and preserving the crocodiles at the expense of humans leads to negative perceptions and low community participation in such crocodile conservation efforts (Gandiwa, 2011; Sai et al., 2016; Matseketsa et al., 2019). The onus is to promote mutually beneficial co-existence between humans and crocodiles, though it is prudent to indicate that each wetland system may require contextual policies and management programs to mitigate the negative effects of HCC (Zisadza-Gandiwa et al., 2016; Musiwa and Mhlanga, 2020). Good management programs focusing on public education, prompt removal of problem animals, and the sustainable use of crocodiles have resulted in mostly peaceful co-existence of humans with potentially dangerous crocodiles in developed nations e.g. South Africa, Australia and United States of America (Fergusson, 2004). However, these approaches require resources that are not typically available in developing nations such as Zimbabwe (ZPWMA, 2015).

A raft of removal and non-removal management strategies and conservation initiatives have been implemented with less success for crocodiles (and hippos) relative to terrestrial mammals e.g. elephants, buffaloes, lions, monkeys, baboons and leopards among others (ZPWMA, 2015; Matseketsa et al., 2019; Musakwa et al., 2020). Conservation authorities have largely implemented lethal control where the actual problem crocodiles are identified and then eliminated after considering the reporting protocols, humane methods to be used, and the impacts on the remaining populations (Fergusson, 2004). The advantage is that it creates positive perceptions on the affected communities and guarantees peace in the short-term although the remaining crocodiles may turn out to be problematic in future even after elimination of the problem crocodile (Fukuda et al., 2014). Translocations have been done in Zimbabwe for re-introduction in areas where the crocodile numbers were decreasing (Anderson and Pariela, 2005). Translocation helps contain the problem crocodiles, does not alter overall national populations, and helps crocodile research, education and awareness and promotes sustainable ecotourism through safari concerns (CSG, 2009). However, it is expensive and has been less favoured by experts because of the distinct and strong territorial instincts of the crocodiles where their feeding habits and reproductive ecology are disturbed in new habitats (Kofron, 1993; Fergusson, 2004; Combrink, 2014; Fukuda et al., 2019). Trophy hunting is also one major crocodile removal strategy which helps control the populations simultaneously providing economic benefits to the surrounding communities and the country (Gandiwa, 2011, 2012; Fukuda et al., 2014).

Non-removal strategies such as: education, research, planning and awareness campaigns and outreaches, risk-based approaches or problem area zonation; barriers and enclosures; enhanced and improved separation infrastructure to keep humans, livestock and crocodiles apart among others have all been tried and implemented with success in the more affluent and developed parts of the world (CSG, 2004). More successful strategies used in Zimbabwe and other developing nations e.g. Zambia included training and funding rapid response teams or Problem Crocodile Control units which remove problem crocodiles and aid victims (Nyirenda, 2015; ZPWMA, 2015). This stemmed from the realisation that access to health care after attacks is important as in most HCC cases the victims die because of late, slow or no medical response to the wounds or injuries (mostly inflicted on limbs) which are often grievous and infectious (Fergusson, 2004). To diffuse tensions and alter negative perceptions towards the crocodiles and foster good relations with communities, compensation schemes are available under the CAMPFIRE initiative although the net benefits are far outweighed by the damages inflicted by the crocodiles (Gandiwa, 2012; Matseketsa et al., 2019; Musakwa et al., 2020). For success in non-removal strategies the key is to involve citizen science through local community integration and co-option in crocodile conservation efforts (Musakwa et al., 2020).

Although commendable, most non-removal strategies are expensive, time consuming and not easy to administer because of a raft of socioeconomic, cultural, political and environmental factors. In some cases the CAMPFIRE initiative has failed totally whilst in other areas it has thrived especially with regards to managing HCC (Tchakatumba et al., 2019; Musakwa et al., 2020). The decentralisation reforms of CAMPFIRE were intended to introduce new institutional arrangements with a shift in power loci and devolved

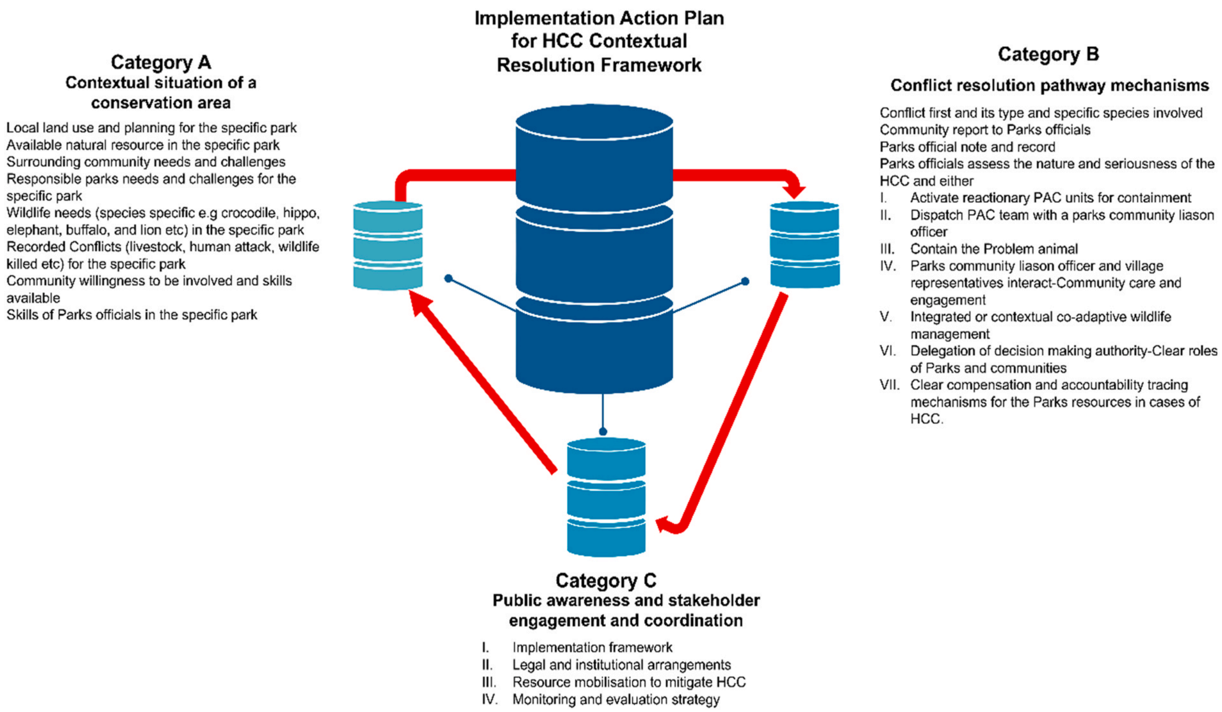

Fig. 3. Proposed Human Crocodile Conflict Contextual Resolution Framework (HCCCR) for Zimbabwe.

responsibilities, however, for the crocodiles (and other wetland species e.g. hippos) there has been no meaningful reduction in conflicts with humans attributable to these reforms. Fergusson (2004) and CSG (2009) indicated that pleonasmic emphasis on the success of CAMPFIRE in a few areas such as Mbire and Chiredzi Districts masks the failure of different non-removal management strategies in stemming HCC which to the contrary has been on the increase in Zimbabwe. There is a need for a revision of the co-existence, adaptive co-management and crocodile conservation concepts and measures used in the country in order to explore and establish the mainstreaming points to stem the ever increasing HCC incidences (ZPWMA, 2015).

The concept of co-existence, adaptive co-management or crocodile conservation per se is muddled and confused with regards to its effects on HCC in Zimbabwe (Fergusson, 2004). A critical appraisal of the Policy and Plan for Crocodile Management in Zimbabwe indicated no clear references to HCC as it is mainly aimed to guide and enhance crocodile management in ranches and farms not in the wilderness. The assumption was that CAMPFIRE and other initiatives were adequate to reduce HCC in communal areas outside of protected areas (Gandiwa, 2012; Tchakatumba et al., 2019). What should have been done was to develop a HCC resolution policy or framework which is totally absent from the ZIMPARKS Act 20:14. A functional, flexible and contextual interactive relationship between the wildlife authorities, and local communities towards HCC is tenable. This review advocated for the crafting of a Contextual Wildlife-Conflict Interactive Resolution Policy which must be inserted as a stand-alone management tool in the proposed 2021 Wildlife Act and must be divorced from the conflicted and vague 2013 Environmental Management Act [Chapter 20:27] jointly used with the 1975 based Parks and Wildlife Management Act [20:14]. This policy must be species specific and comprehensive enough not the vague 'injure one injure all' wildlife policies being used which lack transparency and accountability enough to resolve HCC (Musakwa et al., 2020).

After examination of HCC literature in Zimbabwe this study suggested for development and application of a Human Crocodile Conflict Contextual Resolution Framework (HCCCR) indicated in Fig. 3 for HCC resolution. The HCCCR conjoins three multidimensional categories comprising: Category A-contextualisation of a conservation area. This category A fully documents the contextual situation of the specific conservation area e.g. location, wildlife resources prevalent in an area, surrounding land-use patterns, livelihoods, challenges and skills databases of local communities and parks officials for a proper understanding of the underlying causes of HCC. Category B- conflict resolution mechanisms and pathways. This category B seeks to craft a simple non-bureaucratic conflict resolution pathway with a cogent role and decision making allocation for involved parties, and more importantly this aims to foster community care and engagement and transparent compensation mechanisms in order to ameliorate adaptive co-management. Category C-public awareness and stakeholder engagement and coordination. Category C seeks to create public awareness and stakeholder engagement and coordination for mitigating HCC especially with regards to solid legal and institutional back up for contextual adaptive co-management driven by resource mobilisation to support the implementation of the HCCCR. It is important to indicate that the whole HCCCR plan is interlinked with no stand-alone stage and is adaptable and flexible to be adopted for any species which is involved in HWC. However, for a proper refinement and adoption at regional and international scale more area specific studies including long term sustained surveys or censuses and local community awareness campaigns are needed (Fergusson, 2004; Zisadza-Gandiwa et al., 2016). Generally, it is cogent that HCC poses dire consequences on the crocodile population and threatens its conservation status in Zimbabwe (Fergusson, 2004).

Apart from the proposed HCCCR, it is prudent to cast insights on plausible resolution frameworks for reducing and mitigating and managing HCC in Zimbabwe. The whole idea is to instigate a reduction in the frequency of HCC statistics mainly through incalculating behavioural change among concerned stakeholders in the conflicts i.e. human communities. Thus, concepts from the theory of change, conflict theory and relevant human-wildlife conflict resolution frameworks apply where there is a need to achieve a desired change in a particular context (Chen, 1990; Weiss, 1995). The issue is then how to fill in the activities, interventions and strategies to achieve the desired goal or outcome of reducing and mitigating HCC in and outside of protected wetlands in Zimbabwe. The second step involves the key actors or stakeholders, their roles, interests and points of conflict in resolving the HCC. This involves role allocation with each stakeholder framing a pathway of activities and interventions that is integrated with other stakeholders and this is mapped in an outcomes framework. The outcome framework also lays out the preconditions under which the different stakeholders coordinate to achieve the outcome of reducing HCC. In Zimbabwe, the stakeholders include local communities, wildlife authorities, policy makers, and academia and conservation practitioners. Each stakeholder has a role to play through different activities and interventions e.g. through proper education and conservation awareness of HCC by wildlife authorities for local communities. Government may craft compensation policies for victims of HCC and institute programs that benefit local communities e.g. crocodile egg collection incentives, and photographic safari tourism (ZPWMA, 2015). Academia and conservation practitioners need to institute research activities on crocodile ecology and behavioural aspects, and to inform wildlife authorities and government on conservation implications of trade, overexploitation and socioeconomic covariates on crocodiles.

The onus, then is to craft an interlinked pathway connecting all outputs to achieve the desired goal of reducing HCC in Zimbabwe. This was the premise of the workshop that developed the Plan and Policy on Crocodile Management of Zimbabwe in 1996. Nonetheless, the desired outcome was skewed towards increasing productivity in crocodile ranches and minimising impacts of egg harvesting on wild crocodile stocks (CSG, 2004). Lack of a shared goal inevitably led to conflicts of interest among stakeholders, and local communities were excluded in the exploitation and conservation of crocodiles in Zimbabwe (Gandiwa, 2011). The theory of change concepts works when all the stakeholders share a common desired outcome which in this case must be reduction of HCC (Anderson, 2005). Moreso, in accordance with the theory of conflict different stages or levels of conflict exist and these need to be addressed with interventions appropriate to not only the context but the sources of conflict in order to achieve the desired outcome of reducing HCC in wetlands of Zimbabwe.

At best local communities want to access water and fisheries resources with minimal competition from crocodiles, whereas the crocodiles need and have a right to utilise the resources in their aquatic habitats. The wildlife authorities have a mandate to protect wildlife and minimise risks to locals whilst the government has to provide an enabling environment and mechanism for harmonization of the desires of stakeholders. Consequently, in the proposed HCCCR, theory of change and theory of conflict like in any HCC and HWC mitigation frameworks applied globally for conservation, there is conflict among the expectations of the stakeholders to be considered first before any consideration of the injuries, fatalities, casualties and damage inflicted by the crocodiles (Pooley, 2016a, 2016b). As such the key aspect is then to develop a locally relevant and contextual proactive crocodile management framework involving all concerned stakeholders with clear and shared desired outcomes and a cogent output and intervention pathway and relevant monitoring and evaluation feedback channels and conflict resolution mechanism (Pooley, 2016a, 2016b).

10. Conservation status of Nile crocodiles in Zimbabwe

Analysis of literature on the estimated global populations of wild Nile crocodiles revealed wide disparities depending on the data sources (Kyalo, 2008; Fergusson, 2010). Some estimates indicated that the global crocodile population ranged between 50 000–70 000, and this range is frequently emblazoned on the websites of most donor funded conservation organisations (IUCN, 2017). Relatively, more credible and trustworthy websites indicated that there were no reliable global data on wild crocodile population estimates (Fergusson, 2010). This thread is actually discernible across all credible websites and even from more authoritative sources on crocodile conservation such as the CSG, CITES and IUCN (Fergusson, 2010). For instance, the CSG (2009) indicated a more plausible global population of 250 000–500 000 wild crocodiles. The CITES organisation indicated that each of the twenty six member countries where the species occurs had a range of 60 000–90 000 wild crocodiles (CSG, 2009; IUCN, 2017). Using the average value this translates to at least 1,950,000 wild crocodiles in Africa. From this review point the figure is nearer to the actual population which may be slightly more or less depending on the abundance and distribution of suitable wetland habitats (Fergusson, 2010). Adding the total estimates of farmed crocodiles which ranges between 120,000–150,000 in major crocodile producing countries e.g. South Africa, Zimbabwe, Namibia and Botswana (Fergusson, 2004, 2010) to the CITES estimates then there are well over two million Nile crocodiles globally.

The confusion on the factual global statistical estimates emanates from, and is prevalent in all African countries where there is paucity of solid scientific census data on wild and farmed crocodiles (IUCN, 2017). The crocodile ranching industry is elitist and tends to be closed off to public and media and even scientific scrutiny (Revol, 1995). This is the case in Zimbabwe where the CFAZ (2016) does not always supply verifiable figures to the management authority because of differences in the survey methods used to collect the data (Bourquin and Leslie, 2012). Moreso some crocodile ranches were not even part of the CFAZ which implies that there is a disjuncture in the data on farmed crocodile populations (Revol, 1995). Furthermore, it is a tedious and bureaucratic process to obtain data on the estimates of wild and farmed crocodile populations from the wildlife authorities in Zimbabwe (Utete, 2020). This distorts statistics on the crocodile populations in the country. Available data for the northern and southern parts of the country showed that the wild population ranges from 150,000–300,000 crocodiles, whereas the farmed populations ranges from 180,000–250,000 in the country (CFAZ, 2016) giving a total upper limit of around 600,000–700,000 crocodiles. Nevertheless, this data does not include estimates in wetlands inside and outside of protected areas in the less studied regions e.g. Manicaland, Midlands and parts of Matebeleland and Mashonaland Provinces and could actually be an underestimate of the total crocodile population in Zimbabwe.

A brief outline of the IUCN Red Listing process shows that the Red List Authority (RLA) in this case the ZIMPARKS and other concerned experts, consultants, reviewers and partner organisations are the key actors in collating data on the crocodiles in Zimbabwe (IUCN, 2021). The first stage being the identification and coordination of the RLA, the second step being the collection and review (Pre-Assessment) of all relevant published and unpublished grey literature on the Nile crocodiles of interest (i.e. ecology, distribution, population sizes, trade and conservation threats among others) by the RLA who then agree on timelines and workshop dates for assessment (IUCN, 2021). The third step involves the actual assessment carried out in the IUCN Species in System platform or any other agreed system. The assessment draft is prepared in the SIS and reviewed at workshops convened by the IUCN in tandem with the RLA. At every stage there are monitoring and evaluation and feedback mechanisms that can result in either rejection or acceptance of the draft (IUCN, 2021). After review and checking for consistency if accepted the draft assessment is submitted for consideration for publication. After submission to IUCN there is assessment by identified global experts who are part of the Red List Unit (RLU) to review the submission and provide feedback. If accepted for publication, the assessment is further checked by another team from the RLU for consistency and accuracy and eventually published in the SIS data base and is published in the appropriate IUCN Red List website (IUCN, 2017, 2021). From this brief outline it can be seen that the IUCN Red Listing process is rather water tight in some respects although the main stage of weakness inevitably is the quality, amount, volume, accuracy and the time lines (current or dated) of the available data for the taxon of interest that is the Nile crocodile in this case. It is the view of this study that there was lack of country wide data collection, and an overreliance on available data which is predominantly from the northern and southern parts (Zambezi Valley, Lake Kariba and Lowveld Region) with neglect on data from other regions e.g. Midlands and Manicaland where there is grey data in the form of Parks records, e.g. PAC and HCC and police reports on Nile crocodiles (Utete, 2020). Moreso, the last tangible and concrete workshop on crocodiles was in 1996 where the revised Policy and Plan for Crocodile Management in Zimbabwe was developed and subsequently approved by the Minister of Environment and Tourism in May 1997 (CSG, 2004). Surprisingly, the policy fundamentally aimed to zone crocodile habitats geographically and contextually cognisant of the unequal socioeconomic developmental rates in Zimbabwe in order to provide appropriate levels of protection and utilisation (CSG, 2004). This has never been implemented to fulfilment by responsible wildlife authorities. Since then the RLAs for assessment, mostly for updated CITES permit requirements, where mostly academics and practitioners and consultants who did not thoroughly understand the need to assess data from the other regions inside and outside of protected areas (ZPWMA, 2015).

Thus, the main question is how are the authorities in tandem with CITES arriving at the annual quota of crocodiles for trade export and trophy hunting? Moreso, if there are available suitable habitats as the data suggests where are they located? Are there more crocodiles in wetlands outside of protected areas relative to those in protected areas as queried by Zisadza-Gandiwa et al. (2013)? These questions revealed a need for country wide long-term monitoring of the crocodile population. Nonetheless, some authors opined that there was no point in undertaking expensive long-term censuses especially for resource constrained developing nations like Zimbabwe where the crocodile populations are stable anyway (Fergusson, 2010; IUCN, 2017). This view is plausible in the short-term. However, it is easy for the authorities to institute monitoring surveys in the less studied regions and combine the data with the already well-studied regions for more accurate estimates of the crocodile populations. Overall, this review indicated that the crocodile population is rather stable and may actually be increasing in some parts of the country. However, unless there is solid evidence and accurate up to date data on the countrywide abundances, distribution, and hotspot maps of suitable habitats for both farmed and wild crocodile populations it is safe to ascribe the conservation status of the species in Zimbabwe as Vulnerable or Near Threatened rather than the current Least Concern or Low Risk status in CITES Appendix II and IUCN Red List Data (IUCN, 2017).

11. Conclusions, missing gaps and recommendations

This scoping review is not comprehensive because of missing data in the less studied regions where even the few available records are fragmented to be reliable and valid for enhanced crocodile conservation (ZPWMA, 2015). Still, it indicated that most studies are concentrated in the northern and southern parts of the country based on the assumption that the higher temperatures prevalent in these areas are favourable for reproduction and thus, support more crocodiles (Cott, 1961; Hutton, 1987; Kofron, 1993). This inevitably conceals the contribution of the cooler (western, eastern and midlands) regions on the total national crocodile population possibly underestimating the figures with a potential to undermine conservation efforts and HCC in the future (Fergusson, 2004). Regardless, the few available data indicated that the wild crocodile population is stable and actually increasing (ZPWMA, 2015). However, what is need are comprehensive long-term censuses before the trends can be clearly stated with validity.

Operational capacity of crocodile ranches has declined over the years because of a myriad of socioeconomic and political factors in the country (IUCN, 2017) whereas the total crocodile product outputs e.g. skins, and meat has marginally increased from 1965 onwards (Revol, 1995). Coordinated conservation efforts, stochastic market prices and strict monitoring by authorities have led to increased crocodile production mainly to offset the ever increasing capital input requirements (CFAZ, 2016). To the contrary, the CITES quota allocation for crocodile trophy hunting has increased from 150 to 250 per year on the backdrop of recorded increases in the crocodile populations in the more studied northern and southern parts (CSG, 2009; IUCN, 2017). Still, there is a need to establish the abundance and distribution of the national crocodile population and correct data capturing before remitting returns to CITES so that the country is allocated accurate and sustainable quotas. Using the estimated national crocodile population suggested in this paper, the CITES quota allocated for crocodile hunting is sustainable for Zimbabwe (CSG, 2004; Fergusson, 2004). This review suggested that what is important is to model and assess population recovery, and the drivers and barriers e.g. genetic and morphological variations, and climate dynamics over a long term (>15 years) to determine the impacts of trophy hunting and commercial harvesting on crocodile populations in wetland systems of Zimbabwe (Loveridge and Hutton, 1992; van Asch et al., 2019).

Incidences of HCC are increasing in Zimbabwe with human fatalities in the region of 25–30 per year (ZPWMA, 2015). Human encroachment into wetlands and subsequent destruction and dwindling of suitable habitats has led to increased HCC particularly in wetlands outside of protected areas (Zisadza-Gandiwa et al., 2013, 2016). Although various removal and non-removal strategies are being implemented the scourge of HCC is actually increasing (Fergusson, 2004). This review reflected that more HCC incidences occur in fringe or areas outside but proximal to protected areas although it requires further remote sensing based hotspot mapping for validation. Critically, it implies that current HCC mitigation strategies have not been totally effective and need revision. This review identified a missing gap in HCC mitigation where there is a lack of contextual HCC resolution mechanisms and clear pathways even in the widely adopted CAMPFIRE model which appears to be functional for terrestrial mammals rather than aquatic species (Tchakatumba et al., 2019; Musakwa et al., 2020). An adaptable Human Crocodile Conflict Contextual Resolution Framework (HCCCR) which conjoins the local context of a wetland and its tenets, conflict resolution mechanisms among stakeholders and pathways and the involvement of local communities and relevant institutions forms part of the solution to HCC mitigation in Zimbabwe. The proposed HCCCR can be refined in local contexts as different regions have different land-use and climatic patterns which affect wetland systems dynamics and the magnitude and severity of impacts on the crocodile populations and HCC (CSG, 2004). Involvement of multiple agencies although very strategic on paper actually leads to duplication of roles and multiplicity of responsibilities and complexities and detrimental conflicts which hampers HCC resolution in Zimbabwe (IUCN, 2017). Fixation about the exact HCC statistics tends to overshadow the qualitative analysis of its context and limits development of sound mitigation strategies in the country (Musiwa and Mhlanga, 2020).

Overaly, this review indicated that the national crocodile population, i.e. both wild and farmed, ranges around 600,000–700,000 although this may be an underestimate if the remaining understudied areas are monitored in the medium-long term. The fact that there are several methods to estimate crocodile populations means authorities and researchers can chose to standardise and use similar methods depending on the resources and expertise available for comparability, replicability and reliability of results (Bourquin and Leslie, 2012). This has not been the case with different researchers using different methods in similar wetlands compromising the validity of the results in Zimbabwe (CGS 2009). For now the few available anecdotal data on crocodile populations may be taken as the baseline reference data subject to refinement after long-term reliable records are available. Generally, individual signatory member countries to the CITES need to assess the conservation status of the Nile crocodile species considering their own unique socioeconomic and environmental context rather than blindly following the IUCN classification (Utete, 2020). The point here is that country-specific assessments need to be even more water tight because they form the basis for the global assessments and it is at national and regional level where conservation policy is often implemented (IUCN, 2021). It is imperative to note that for a permissible addition to the appropriate CITES Appendix classification list there is a prerequisite for scientific and solid evidence of the species' abundance, population and interaction with humans in and around the wetland systems in the country (IUCN, 2017). On the contemporary and for posterity it may be safe to ascribe the conservation status of Nile crocodiles in Zimbabwe as Vulnerable or Near Threatened rather than the current Least Concern or Low Risk status in CITES and IUCN Red List Data.

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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.

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Appendix A. Raw data of Nile crocodile populations in the surveys carried out in Zimbabwe

Region			Northern Part		
Year	Kazungula	Upper Zambezi	Kariba	Lower Zambezi	Mat North
2000	93	453	7000	3559	12
2001	25	595	7708	3100	237
2002	19	509	8626	3014	130
2003	18	509	9177	2523	56
2004	18	645	10021	2214	56
2005	34.6	542.2	8506.4	2882	98.2
Region	Southern		Midlands		Midlands
Year	Lowveld	Year	Ngezi	Year	Sengwa
2008	105	1984	337	2016	82
2009	105	1987	228		
2010	93	2010	122		
2011	90	2018	58		

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