UNIVERSITY OF CAPE COAST

DIVERSITY AND DISTRIBUTION OF AMPHIBIANS IN THREE LAND USE TYPES IN THE DORMAA-AHENKRO DISTRICT IN THE BRONG-AHAFO REGION OF GHANA

RUTH KWAPONG

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BY

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Thesis Submitted to the Department of Entomology and Wildlife of the School of Biological Sciences, University of Cape Coast in partial fulfilment of the requirements for the award of Master of Philosophy degree in Wildlife Management

JULY, 2014

DECLARATION

Candidate's Declaration

I hereby declare that this thesis is the result of my own original research and that no part of it has been presented for another degree in this university or elsewhere.

Candidate's Signature …………………. Date…………………….

Name: RUTH KWAPONG

Supervisors' Declaration

We hereby declare that the preparation and presentation of the thesis were supervised in accordance with the guidelines on supervision of thesis laid down by the University of Cape Coast.

Principal Supervisor's Signature ……………………… Date……………….

Name: Prof. K.A. Monney

Co-Supervisor's Signature…………………….……… Date…………………

Name: Mr. K.B. Dakwa

ABSTRACT

This study investigated the diversity and distribution of amphibians in the Mpameso Forest Reserve and its surrounding cocoa and teak-acacia plantations in the Dormaa-Ahenkro district, in both rainy and dry seasons. Ninety plots (30 plots per study site) were established along transect lines and searched for amphibians. Specimens were found by visual encounter surveys (VES). A total of 1187 individuals of 16 species belonging to six anuran families (Arthroleptidae, Bufonidae, Hyperoliidae, Ranidae, Ptychadenidae and Petropedetidae), were recorded during the survey. The rainy season survey recorded 786 anurans compared to 401 in the dry season, with increased species richness, 518 anurans were observed in the forest, followed by the cocoa farm, 408 individuals and teak plantation had 261 individuals. There was a significant difference in species diversity between the three land use types in both rainy and dry seasons. The 16 species were irregularly distributed in the three land use types. Twelve of the species documented are classified as Least Concern, one as Near Threaten, two are vulnerable and one as data deficient according to IUCN. Based on the findings of the study, it is recommended that further studies should focus on pesticide use and the effects of constant pesticide application on amphibian species richness and distribution in cocoa growing areas as well as research on specific species to identify the habitat requirements of individual species in all the land use areas. Conservation efforts should consider the status of anurans encountered and also take into consideration the status of amphibians according to the IUCN Red list.

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DEDICATION

This work is dedicated to my lovely parents Mr. and Mrs. Kwapong for their unflinching support and encouragement throughout the program.

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CHAPTER ONE

INTRODUCTION

Background to the study

Herpetology is the branch of zoology concerned with the study of amphibians (frogs, toads, salamanders, newts and gymnophionas) and reptiles (snakes, lizards, amphisbaenids, turtles, terrapins, tortoises, crocodilians and tuataras). Batrachology is a further sub-discipline of herpetology concerned with the study of amphibians alone. Herpetology offers benefits to humanity in the study of the role of amphibians and reptiles in global ecology, especially because amphibians are often very sensitive to even subtle environmental changes, offering a visible warning to humans that significant changes are taking place (Blaustein, Wake & Sousa, 1994). Some toxins and venoms produced by reptiles and amphibians are useful in human medicine (Alder, 1989).

There has been an expression of great concern about the extinction of amphibians globally (Reid & Zippel, 2008), as one in three amphibian species is threatened with extinction (Norris, 2007). This can be justified on the basis that worldwide, amphibians are crucially important in the ecosystem. The general ecological importance of amphibians lies in their being predators, acting as primary and secondary carnivores on insects that may be crop pests or disease vectors (Behangana, 2004). The known important roles amphibians play in the food webs of most biological communities cannot be overemphasized. The possible causes of the decline of amphibians involve various complex combinations of (i) habitat destruction, fragmentation or loss (ii) overharvesting (iii) invasive species and pollution (from industrial, agricultural and pharmaceutical areas) (Beebee & Griffiths, 2005; Moore & Church, 2008). The growing ecological impacts of climate change have also been realized to be a non-traditional cause of amphibian decline (Araύjo, Thuiller & Pearson, 2006). The contributing factors of the decline are even suspected to be acting together in some instances (Blaustein & Kiesecker, 2002; Davidson & Knapp, 2007).

Quite recently, a disease caused by a chytrid fungus (*Batrachochytrium dendrobatidis*)has been found to be a factor most commonly associated with mysterious declines and catastrophic extinctions of amphibian populations and species (Moore & Church, 2008). Even though habitat loss may evidently remain the most significant overall threat (impacting 90% of those species currently considered threatened), it is believed that amphibians are more sensitive than other organisms to environmental deterioration and that the decimation of amphibians is a warning sign of an ever increasing poisoned environment.

It is observed that many small mammals and herpetofaunal species generally have relatively short generation times and quick responses to habitat and microclimatic variations within forest fragments (Cain, Damman, Lue, Yoon & Morel, 2007). Thus, amphibians as bio-monitors can be used to measure this feature (Wasonga, Bekele, Lötters & Balakrishnan, 2006). An alteration of microclimatic conditions is also thought to be of major importance to plants and animals generally in fragmented forests (Harper *et al* 2005).

The global extinction threats to amphibians have generated calls for proactive conservation activities (Stuart *et al*., 2004) and have even prompted a five-year strategic plan by the Amphibian Conservation Action Plan (ACAP) in 11 thematic areas. The strategy has designed a network of conservation sites for amphibians taking into consideration freshwater resources and associated terrestrial landscapes, climate change, biodiversity loss, amphibian declines and captive programmes (Moore & Church, 2008). A detailed study of upland streams in the Central Panamanian Highlands, as an example of the current extinction rate, revealed that ecological effects of amphibian declines could not be taken for granted. The effects of amphibian decline included changes in algal community structure and primary production, altered organic matter dynamics, negative impacts on aquatic insect predators,(e.g. snakes) and reduced energy transfer between streams and associated riparian habitats (Ranvestel, Lips, Pringle, Whiles & Bixby, 2004).

While the host of factors driving wildlife declines in the Guinean rain forest ecosystem involve some of the usual suspects such as logging and land use conversion for agriculture, several studies have demonstrated that hunting of wildlife for human consumption through the bushmeat trade is among the most immediate threats (Milner-Gulland, Bennett & SCB, 2003; Brashares *et al.,* 2004; Cowlishaw, Mendelson & Rowcliffe, 2005). The bushmeat trade refers to the illegal and unsustainable over-hunting of wildlife for meat and income. It is common in many parts of the world where hunting of animals from the wild is undertaken. It involves the sale of any wild species, though western sources tend to focus on the trade specifically involving great apes. The high rate of harvest, combined with habitat loss and alteration, has led to very severe population declines. If this trend is unchecked, extinction of species is likely to occur. Though habitat loss is often cited as the primary threat to wildlife, commercial hunting for the meat of wild animals has become the most significant immediate threat to the future of wildlife in Africa and around the world. It has already resulted in widespread local extinctions in Asia and West Africa especially in Nigeria, Benin and other areas.

With the exception of Chinese and Japanese giant salamanders, *Andrias davidianus* and *japonicas* respectively, the primary part of anurans used by humans for food are the legs, which are extremely popular in Europe, Canada and the United States. In the 1990s, Europe imported 6,000 metric tons of frog legs each year (Jensen & Camp, 2003). Between 1981 and 1984, the United States imported more than 3 million kgs of frog meat per year, the equivalent of approximately 26 million frogs (Jensen & Camp, 2003). Asia is the second largest market for frog leg consumption, where the most common species consumed is the Chinese edible frog, *Hoplobatrachus rugulosus* (Jensen & Camp, 2003). In just one year, over 6 million Chinese edible frogs were imported to Hong Kong from Thailand (Wai-Neng Lau, Ades, Goodyer & Zou, 1999). It is presumed that all these frogs are being collected from the wild since most of the frog farms in Thailand only raise American bullfrogs (Wai-Neng Lau *et al*., 1999). Given the sheer number of frogs collected, *rugulosus* is likely being overharvested and if this practice continues, it could wipe out remaining wild populations. Human pressures with possible adverse effects on reptile and amphibian diversity include the collection of animals for medical uses, the pet trade and killing out of fear or spite.

A recent survey of amphibians of the Togo Hills, in the Akwapim-Togo Ranges, concluded that with 31 amphibian species, the area is more diverse than previously assumed and probably contains at least 41 amphibian species (Rödel & Agyei, 2003). The periphery of Kyabobo National Park located in the Nkwanta district of the northern Volta Region was also found to contain 20 frog species (Rödel & Agyei, 2003). Twelve species of anurans were reported in Kakum National Park and its surroundings (Monney, Darkey & Dakwa, 2011). Again, 24 anuran species of which 11 were forest – dependent species and 13 disturbance- tolerant species, were recorded in Krokosua (Adum, Eichhorn, Oduro, Ofori–Boateng & Rödel, 2013).

Hughes (1988) published a review of the history of herpetological investigations in Ghana and provided a country checklist of 71 amphibian species. However, other researches have reported some of these species to be of uncertain taxonomic status. Examples of these species included *Arthroleptis bivittatus (*Muller, 1885), *A. zimmeri* (Rödel and Bangoura, 2004)] or not occuring in Ghana, examples being *A. variabilis* (Rödel, 2000, Rödel & Bangoura 2002), *Conraua alleni* (Rödel & Agyei 2003) and *Astylosternus occidentalis (*Parker, 1931). The 2002-2004 Global Amphibian Assessment (GAA), by The World Conservation Union /Species Survival Commission, Conservation International Center of Applied Biodiversity Science and Nature Serve, equally lists 71 amphibian species for Ghana, still including doubtful country records(Hughes, 1988).

Proposals of WZACS emphasize the urgent need for more studies of the ecological impacts of amphibian declines and extinctions and there has been a call for accumulation of data on amphibian population biology and habitat viability in the wild. Furthermore, to ensure the appropriate husbandry conditions and management protocols in zoos, it cannot be denied that there also has to be sufficient knowledge of population biology and habitat viability in the wild (Reid & Zippel, 2008). The Guinean rain forest of West Africa is a centre of biological diversity with considerable endemism (Myers, Mittermeier, Mittermeier, Fonseca & Kent, 2000). The percentage of amphibian species endemic to this region far exceeds that of other tetrapod groups. The prevalence of amphibians is 77% compared to mammals and birds with percentages of 8% and 18% respectively (Myers *et al*., 2000). However, concomitant with this impressive diversity is an alarming rate of habitat loss. In Ghana alone, natural forests have diminished to about 11.8–14.5% over some few decades (IUCN, 2006; Poorter, Bongers, Kouame & Hawthorne, 2004). Worldwide, habitat loss and forest fragmentation are recognized as key factors driving the global extinction of genetically – distinct populations and species (Bierregaard, Lovejoy, Kapos, Dossantos & Hutchings, 1992; Hughes, Daily & Ehrlich, 1997; Brooks, Sodhi & Bradshaw, 1999; Stuart *et al*., 2004). Forest and habitat destruction should be checked to prevent the extinction of species in the future.

Statement of the Problem

In Ghana, very little is known about how land use types and seasons influence the diversity and distribution of amphibians. Most of the literature describes the effects of land elevation on the distribution and diversity of amphibians. Farming contributes more than 60% of GDP in Ghana, and agriculture is mostly subsistence involving methods that directly impact the ecosystems which are homes to a lot of amphibians. Since these amphibians are sensitive indicators of ecological change, it is imperative to assess how the different land use types and the changing seasons in Ghana affect their diversity and distribution such that by assessing these we can indirectly infer on the effects of these land use types on the environment.

Purpose of the Study

Information about the natural history and conservation status of a very large number of tropical species is lacking in mainland Africa (Halliday, 2008). Generally in West Africa, the herpetofauna of Ghana remains largely uninvestigated and ecological research on amphibians linked to conservation activities has generally lagged behind. The high biological significance of protected areas, the escalating human exploitation and the negative impact of human exploitations on wildlife suggest that there is the need for a detailed herpetofaunal investigation to document information that can be gathered now before some disappear.

Data on the number of species of amphibians found in any locality in Ghana remain unexplored. Since few detailed studies on the herpetofauna of Ghana have been carried out, this study was undertaken mainly to contribute to the scanty literature on Ghana’s herpetofauna and to provide background data against which analyses such as amphibian declines, amphibian systematics and biogeography, etc, will be measured in future. Such information is vital for conservation priorities, and the study focuses on diversity and distribution of amphibians in three land use types in the Mpameso Forest Reserve as a baseline for determining the species with significant conservation status.

Research objectives

This study was done to investigate whether there is any significant difference between species diversity and distribution of anurans in different land use types, namely: forest, teak plantation and cocoa by finding out whether land under agricultural use influences the distribution and diversity of amphibians by comparing populations in the forest with land under cocoa and teak cultivation. The specific objectives were to determine;

i. The distribution and diversity of anurans in different land use types; forest, teak plantation and cocoa farm.
ii. The diversity and distribution of anurans in both rainy and dry seasons, and find out which is the more favorable to amphibian species.
iii. Species with significant conservation status which inhabit the forest, teak plantation and cocoa farms.

Assumptions

It is assumed that since these amphibians are sensitive to ecological changes, changes in the environment will directly affect their diversity and distribution within a habitat.

Significance of the Study

The results of this study will contribute to a better understanding on the use of anurans as ecological markers of environmental changes.

Delimitations of the Study

i. This study emphasised on anurans (frogs and toads), however, an inclusion of members of a wider class of the Family could have provided further insight into the finer details of environmental change on amphibian diversity.

ii. Brong-Ahafo Region is made up of many districts possibly with different weather patterns and daily minimum and maximum temperatures. The study was however undertaken only in one district, the Dormaa-Ahenkro district. A geographical longitudinal study may have provided data more representative of the Region and of the ecology.

Limitations

i. The use of DNA techniques in acquiring data could have provided more detailed information on the taxonomy and diversity of the anurans sampled.

ii. Inclusion of meteorological data over the study period could also have provided important weather data that might have been influential in contributing to anuran distribution.

Definition of Terms

WZACS: World Zoo and Aquarium Conservation Strategy.

IUCN: International Union for Conservation of Nature.

DOM: Dissolved Organic Matter

Bd: *Batrachochytrium dendrobatidis* - Causes the infectious disease chytridiomycosis of amphibians.

Biphasic life: A type of life cycle which involves two phases.

Vulnerable (VU): A species is considered Vulnerable when it is not critically Endangered or Endangered but is facing a high risk of extinction in the wild in the medium-term future.

Least Concern/Not Threatened (LC): A species is listed as Least Concerned when it has been evaluated against the criteria and does not qualify for Critically Endangered, Endangered, Vulnerable or Near Threatened. Widespread and abundant taxa are included in this category.

Near Threatened (NT): This is where the species is still relatively widely distributed with its Area of Occupancy probably not much greater than 2,000 km^2, and the extent and quality of its habitat is declining, thus making the species close to qualifying for Vulnerable.

Organization of the Study

The project is divided into six (6) major chapters. Chapter One of the study provides general introduction to the research which includes the background of the study, statement of the problem, purpose of the study, the research objectives, assumptions, significance of the study, delimitations, limitations and definition of terms. Chapter Two introduces the theoretical background (literature review) relating to amphibian species density, diversity and distribution in various land use types. Chapter Three consists of the details of the study area, methodology used, research design, data collection procedure and methods of data analysis. Chapter Four of the study is the presentation of the research results. Chapter Five is discusses the analyses of the results obtained. Chapter Six combines the Summary, Conclusion and Recommendations emanating from the findings of the study.

CHAPTER TWO

LITERATURE REVIEW

Systematics and Biogeography of Amphibians

Amphibians represent a unique group of vertebrates containing over 7,140 described species worldwide that demonstrate an intrinsic aspect of evolution, niche segregation and natural history (Frost *et al*., 2006; Amphibia Web, 2013). The evolutionary and phylogenetic history of amphibians go approximately 365 million years back (Carroll, 1992). Amphibians evidently evolved from either the lobe-fin fishes (Crossopterygii) or the lungfishes (Dipnoi) in the early Devonian Period and represent a transition step in the evolution of terrestrial life (Carroll, Kuntz & Albright, 1999; Carroll 2009). Since then, amphibians were shaped and reformed under multiple selective environmental pressures, radiating them into distinct life styles and body forms (Wells, 2007). Multiple extinction events occurred through the evolution of amphibians in the Carboniferous, Permian and early Jurassic Periods, ultimately leaving a handful of evolutionary relics and modern amphibians (Carroll, 2009).

Modern amphibians have diverged into three orders with distinct anatomical features: Urodela (salamanders), Anura (frogs and toads) and Gymnophiona (caecilians, limbless amphibians). Among all amphibians, anurans have the widest distribution across many biogeographical regions with the highest diversity in the Oriental, Neotropical and Afrotropical regions. Diversity of urodelans is prominent in the Neartic and Neotropical regions, while caecilians are restricted to tropical wet biomes are mostly diverse in the Oriental and Neotropical regions (Duellman, 1999; Duellman & Sweet, 1999).

General Characteristics of Amphibians

Amphibians are cold-blooded animals, characterized by a unique feature (i.e. survival both in water and on land). Amphibians lay eggs which hatch into larvae. These larvae usually look entirely different from their adult form. Frogs are a classic example of amphibians but they are not the sole members of this class. The name "frog" is commonly applied to those forms with long legs and smooth, mucus -covered skins while "toad" is used for a variety of robust, short-legged anurans, especially those with rough skins. The name "toad" is applied so unevenly that one member of a family may be called a toad and a closely related member a frog. There are other amphibians too but some of them, like the salamanders, may appear as reptiles because of their close resemblance to lizards.

Amphibians show great variations in temperature sensitivity. Some species are active only in higher temperature whereas, others can survive lower temperatures without hibernating. Amphibians generally shed their skin periodically and like birds and mammals that shed their skin in flakes, amphibians shed theirs as a single piece. They generally feed on exuviated skin. Amphibians are distributed in water and land throughout the world, except in Antarctica and Greenland. Amphibians hibernate for long time, often for several months (Plough, 2013).

Morphological Characteristics of Amphibians

Anthropogenic land use of areas surrounding wetlands may affect larval and post-metamorphic amphibians (Hecnar & M'Closkey,1996; Bonin, Desgranges, Rodrigue & Ouellet, 1997; Dodd 1997; Alford & Richards, 1999; Semlitsch, 2000) by influencing many ecological mechanisms that regulate the growth and mortality rates of individuals in aquatic and terrestrial environments (Werner,1986). Agricultural cultivation (i.e., arable cropland) may confine amphibians to wetlands, resulting in species associations and population densities different from those that are found in similar undisturbed ecosystems (Knutson *et al*., 1999; Kolozsvary & Swihart, 1999; Gray, Smith & Brenes, 2004). Consequently, landscape cultivation may influence post-metamorphic body size of amphibians by affecting density of conspecifics (Oldham, 1985). Cultivation of the watershed can also increase sedimentation in wetlands, which decreases hydroperiods (Martin & Hartman, 1987; Corn & Bury, 1989) and may reduce the duration of development (Brady & Griffiths, 2000).

Agricultural chemicals (nitrates, ammonia, organophosphates) can bio-accumulate and reduce food densities, foraging activity, and growth of larval amphibians (Hall & Kolbe, 1980; Baker & Waights, 1993, 1994; Freemark & Boutin, 1995; Hecnar, 1995), with possible negative effects on postmetamorphic body size. Understanding this possible relationship is important because body size positively influences survival, reproduction, and recruitment of amphibians (Wilbur, 1984). Research has shown that larger amphibians within a species are better at acquiring food resources, escaping predators, withstanding dehydration and attracting mates than smaller individuals (Berven, 1982). Also, larger female amphibians have greater fecundity than smaller females (Berven, 1982; Krupa, 1986). Thus, amphibian populations composed of larger individuals may be less likely to experience demographic declines than those composed of smaller individuals. As anthropogenic disturbance negatively affects wildlife populations (Primack, 2000), there is also the probability that cultivation of terrestrial landscapes surrounding wetlands would negatively influence body size of amphibians.

Reproductive Characteristics of Amphibians

Amphibians generally reproduce in fresh water, but some species reproduce in brackish water in mangrove swamps. Most amphibian eggs have a gelatinous coat that swell up when they come into contact with water. Most amphibians have indirect development; their eggs hatch into larval forms which are quite dissimilar to the adult form. These larvae undergo metamorphosis and become miniature adults. Once this stage is reached, the growth process happens and the young animal grows to adulthood. Some species undergo internal fertilization whereas others, like frogs, undergo external fertilization where the females lay unfertilized eggs in water and males deposit their sperm over the eggs and fertilize them (Plough, 2013).

For amphibians with biphasic life histories, loss of either aquatic or terrestrial habitat can diminish population persistence (Semlitsch, 1998; Semlitsch & Bodie, 2003). Persistence of amphibian populations depends largely on processes that occur on at least two spatial scales: local and landscape. Post-metamorphic amphibians live, forage, and overwinter in terrestrial uplands near aquatic reproduction sites (Semlitsch, 2008). Consequently, amphibians require suitable terrestrial habitat for growth and survival duringnon-breeding portions of the year (Semlitsch, 1998; Semlitsch & Bodie, 2003). Pond-breeding amphibians require appropriate nearby habitat that links their terrestrial activity centres to aquatic reproduction sites in order to successfully move between them. Loss of local connectivity between terrestrial and aquatic environments can negatively affect amphibians and has been shown to lead to population declines in biphasic amphibians (Becker, Fonseca, Haddad, Batista & Prado, 2007; Harper, Rittenhouse & Semlitsch, 2008).

At the landscape level, patches of amphibian populations can experience reproductive failures due to pond drying, predator establishment (e.g. odonate larvae, fish), or other factors (Semlitsch, Scott, Pechmann & Gibbons, 1996). Since as many as 82% of amphibian species are forest-dependent (Stuart *et al*., 2004), activities such as forest clearing and land conversion have great potential to affect amphibian populations. Indeed, past studies have shown that forest clearing can reduce amphibian richness and abundance by reducing survival and/or promoting evacuation of harvested habitats (deMaynadier & Hunter 1995; Todd & Rothermel, 2006; Semlitsch, Corner, Hocking, Rittenhouse & Harper, 2008).

Natural History of Amphibians

Amphibians are dependent on moist conditions and high relative humidity. Amphibian diversity is therefore highest in regions with high precipitation and/or lower evaporative water loss (Duellman & Trueb, 1994). Many require freshwater habitats to breed and develop into adulthood. A few amphibian species have independently evolved to breed in foam nests constructed outside aquatic habitats. Some species have completely lost their larval stages and lead a completely terrestrial mode of life (Beebee, 1996; Wells, 2007). Amphibians have radiated into terrestrial, aquatic (streams, cascades, and wetlands), scansorial (arboreal, phytotelms, rock outcrops) and fossorial (leaf litter, organic top soil) niches in both the Old and the New Worlds. They are also found throughout elevation gradients in tropical, subtropical and temperate biomes with considerable niche diversification at different ranges of altitude (Duellman, 1999; Wells, 2007). Thirty-nine modes of reproduction and development have been recorded among amphibians, including parental care, viviparity, and terrestrial direct development (Wells, 2007). Most amphibians are generalist insectivores although a few species are known to be specialist predators of gastropods, earthworms, ants and termites. For most nontropical amphibians, prey selection is season dependent (Duellman & Trueb, 1994). Being poikilotherms and having a metamorphic lifecycle with an aquatic larval stage, they encounter a wide range of environments and habitats, each with different physiological constraints. Environmental and climatic parameters such as temperature, access to water, availability of microhabitat refugia, humidity, vegetation cover, and insect prey distribution affect their biological activities such as reproduction, foraging, local migration, and distribution (Gibbs, 1998; Beebee & Griffiths, 2005). The optimal conditions of the above environmental parameters preferred by amphibians mostly prevail in relatively undisturbed forested habitats and aquatic habitats with substantial forested buffer zones. However, there are amphibians that can tolerate long, cold winters and hot, dry summers (Duellman, 1999).

Amphibian Species Diversity and Distribution

Species diversity is an important property of communities because it is often related to their functioning and potential for change (Stachowicz, Bruno & Duffy, 2007). Diversity is a measure of how likely two randomly – selected individuals in a community belong to different species. Thus, diversity is affected by two other properties of communities: (i) richness, which is the total number of species, and (ii) evenness, which is the degree of similarity in abundance among the species (Krebs, 1999). Environmental stress is the negative force exerted by the abiotic environment on the performance of organisms and has been identified as an important factor which plays a major role in determining local species diversity mediated by interspecific interactions (Whittaker, Willis & Field, 2001).

As an example, Smith, Weldon, Conradie & du Preez (2007) suggested that mid-altitude distribution ranges and early colonization of such areas enhance diversification of tree frogs; whereas Wiens, (2007) found that early colonization of mid-elevation habitats explain species richness patterns in salamanders. Kozak & Wiens (2007) found latitudinal differences in the altitudinal and climatic overlap of sister species, suggesting that climatic divergence along elevational gradients may increase opportunities for speciation and promote diversification in amphibians.

Moreover, present levels of extinction risk for tropical species (Stuart *et al*., 2004) give a sense of urgency to studies aiming to expand current understanding of factors influencing speciation in these areas. Identifying mechanisms that have shaped species richness within highly diverse tropical environments like Ghana will therefore increase our understanding of worldwide patterns of species diversity (Wiens, 2007).

Agricultural Effects on Amphibians

Agricultural practices have a potentially large influence on amphibian populations because of the problems of habitat loss, isolation, chemical and nutrient contamination (Bishop, Mahony, Stuger & Pettit, 1999; Kolozsvary & Swihart, 1999; Zampella & Bunnell, 2000; Joly, Miaud, Lehmann & Grolet, 2001). In recent studies on the effects of agricultural intensification on biodiversity, amphibians have usually been neglected. This is surprising, as among amphibian researchers, various processes related to agricultural intensification are regarded as major threats to amphibians (Joly *et al*., 2001) and are partly responsible for global amphibian population declines. Land clearance for agriculture will occur mainly in the tropical developing countries (Tilman *et al*., 2001) and the resulting loss of biodiversity (Brooks *et al.*, 2008) will be paralleled by a decline in associated ecosystem functions and services and a weakened resilience against other threats such as climate change (Hooper *et al.*, 2005). Understanding the value of the agricultural landscapes for native biodiversity will not only assist sustainable management, but also poverty alleviation through changing crop yields (Steffan-Dewenter *et al*., 2007).

Most studies that determine the effects of agriculture on tropical faunal diversity focus on birds or invertebrates (Rice & Greenberg, 2000; Adu-Pakoh, Oppong & Aduse-Poku, 2008), and highlight the importance of rainforest trees or the nearby presence of pristine habitats to sustain high diversity (Schroth and Harvey, 2007). This makes it difficult to develop sensible evidence – based management recommendations which a cause for concern is given that anurans are part of the most threatened vertebrate taxa on the planet and are particularly susceptible to habitat destruction and climate change (Whitfield *et al*., 2007).

Anurans are a diverse vertebrate group with large variation in physiological, behavioral, morphological and ecological characteristics (Feder

& Burggren, 1992). They employ a general life history strategy entailing the use of aquatic habitats for reproduction and larval development.

Factors that influence the structure and composition of amphibian communities

Habitat availability

The maintenance and survival of amphibian populations in any landscape requires the availability of suitable aquatic habitats, such as a waterbody (pond, dam or lake), wetland (swamp, marsh) or stream, and terrestrial habitats (Wells, 2007). The amount and type of amphibian habitat available is affected by several processes that occur in urban and suburban environments including: (1) habitat loss; (2) habitat fragmentation and isolation; and (3) habitat creation and restoration.

Habitat loss, fragmentation and isolation

Human impact on natural vegetation eliminates large portions of habitat from the landscape; the remaining patches are often fragmented and isolated, and the remaining animal populations are smaller (Radeloff, 2005). The importance of habitat loss and fragmentation in the decline of local populations of amphibians has been outlined in recent reviews (Cushman, 2006; Gardner, Barlow & Peres, 2007). Gardner *et al.* (2007) identified a gradient of increasing severity of impact on amphibian species richness with decreasing structural and habitat complexity arising from habitat loss.

Many amphibian populations are naturally patchy across the landscape at local scales, which may comprise larger networks of meta-populations towards regional scales (Marsh & Trenham, 2001; Smith & Green, 2005). Moreover, many amphibian species depend on the linking of complementary habitats at multiple spatial scales to successfully fulfill their complex life cycle requirements, and their populations are thus structured as patchy networks or meta-populations (Pope, Fahrig & Merriam, 2000; Marsh & Trenham, 2001). For instance, urbanization reduces the ability of these networks of populations to function due to the construction of roads and urban infrastructure such as buildings, fences and open areas that inhibit or discourage amphibian dispersal (Vos & Chardon, 1998).

Nearly all studies reviewed reported a negative relationship between habitat destruction and amphibian species richness, presence/absence, abundance or community structure. Overall amphibian decline in an area is directly associated with changes in landscape structure due to urbanization that results in decreased wetland area and density, and increased wetland isolation, decreased wetland vegetation, forest cover and other upland terrestrial habitat (Lehtinen, Galatowitsch & Tester, 1999; Rubbo & Kiesecker, 2005; Parris, 2006; Gagne´ & Fahrig, 2007).

Urbanization

Studies into changes in amphibian habitat over time have reported an inverse relationship between urbanization and extant habitat. Gibbs (2000) conducted an analysis of wetland mosaics along an urban–rural gradient in the New York city region, USA, and reported reductions in wetland density and an increase in nearest-neighbour habitat distances associated with the shift in human settlement patterns from rural to urban. Wood, Greenwood & Agnew (2003) attributed the decline of the great crested newt (*Triturus cristatus*) in the UK to the loss of pond habitat caused by urban development. They proposed that these critical temporary pond habitats are at greater threat in the

UK than any other small water body because they are typically shallow, vulnerable to soil drainage, and highly susceptible to pollution. Similarly, vernal pools, which constitute habitat for many amphibian species across the northeastern USA, are also at risk of destruction from urbanization due in part to their diminutive sizes and short hydroperiods (Grant, 2005), and because they are rarely afforded protection (Dodd & Smith, 2003; Semlitsch, 2003; Windmiller & Calhoun, 2007). Small temporary wetlands (<4.0 ha) are critically important for amphibian breeding success and may function as stepping-stones to reduce inter-wetland distances (Gibbs, 1993, 2000; Semlitsch & Bodie, 1998), and thus every effort should be made to preserve and even enhance these habitats in urban and suburban landscapes in order to maintain local, regional and global amphibian biodiversity. Habitat loss, fragmentation and isolation may also affect population genetic structure. For example, the landscape genetics of *Physalaemus cuvieri* in the Brazilian Cerrado show a signature of effects of human occupation and habitat loss on genetic differentiation at the regional scale, with discontinuities to gene flow in two particular regions with more intense habitat loss and older human settlement (Telles *et al.,* 2007).

Habitat Creation and Restoration

Amphibians with broad habitat requirements may be able to persist within urban landscapes because they are able to use artificial habitats such as garden ponds, ornamental lakes and dams, retention ponds and drains. Indeed, there is evidence that some species have benefited from the construction of ponds and wetlands, particularly during the early phase of urbanisation when colonisation by amphibians is less impeded, because they may replace the function of rural or natural ponds destroyed during the process. For example, the common frog (*Rana temporaria*) in Britain persists in urban and suburban areas more so than in rural areas, which is most likely due to the abundance of garden ponds (Carrier & Beebee, 2003).

However, water bodies, wetlands and streams in urban and suburban areas are often limited in their suitability for amphibian species with more specific habitat requirements because many are artificially stocked with exotic fish, have inappropriate hydrological regimes, receive contaminated runoff (fertilizers, sediment, pesticides, road surface grease and oil, heavy metals), and have high human visitation rates and artificial lighting, which disrupts breeding activity (Baker & Richardson, 2006).

Moreover, the physical structure of urban ponds may exclude some species. For example, a vertical pond wall may mean that a pond is suitable only for tree frogs because they are able to climb out when emigrating (Parris, 2006). Urban wetlands are also often surrounded by roads and urban related infrastructure that can form barriers to amphibian dispersal, potentially rendering them inaccessible to species with moderate to high dispersal requirements (Rubbo & Kiesecker, 2005). Therefore, species with specific habitat or life-history requirements may be attracted to constructed habitat of inferior habitat quality, and thus created ponds may function as habitat traps or sinks (Battin, 2004). Restoration activities may improve the ecological function of urban ponds and wetlands, despite their limitations. For example, wetlands in an urban area of Minnesota, USA, were successfully restored by destroying portions of drainage tile or filling ditch systems and allowing water to re-flood the basins, and were subsequently colonised by amphibians conditional on distance to source ponds (Lehtinen & Galatowitsch, 2001). Restoration of wetlands on the Danube Island, Austria, was successful in attracting a suite of amphibian species where fish were absent (Chovanec *et al.,* 2000). The ability of restored wetlands in urban landscapes to provide suitable habitat for amphibians requires the creation and maintenance of appropriate levels of habitat succession, suitable fluctuations in hydroperiod, availability of upland terrestrial habitat, good water quality, connectivity to surrounding populations, and the absence of native and exotic predatory fish (Beebee, 1996; Porej & Hetherington, 2005; Petranka, Harp, Holbrook & Hamel, 2007). The restoration of meta-populations of amphibians at the landscape scale is critical for larger- scale and long-term recovery of amphibians (Semlitsch, 2002), although this poses a serious challenge in highly modified urban landscapes. There is also the possibility of re-introducing amphibians into restored water bodies, wetlands and streams in urban and suburban areas via translocated stock, although this action raises ethical issues and concerns with transport of diseases, in addition to whether restoration fully satisfies the ecological requirements of the target species and provides adequate connectivity in the landscape (Marsh & Trenham, 2001; Seigel & Dodd, 2002; Calhoun & Hunter, 2003).

Habitat Quality

The quality of amphibian habitat is influenced by the amount and type of vegetation in the water body, wetland or stream and surrounding terrestrial habitat, the hydroperiod, water quality, the presence of predators and competitors, the prevalence of diseases and the nature and frequency of human disturbances. Amphibian habitat provides resources for breeding and non-breeding activities, such as foraging and dispersal, and shelter and overwintering sites (Wells, 2007). Species with complex life cycles, such as pond-breeding amphibians, may depend on landscape complementation, where different breeding and non-breeding habitats are linked through movement, to complete their life cycles (Pope *et al.*, 2000). Amphibians with simple life cycles, such as terrestrial salamanders with direct development, may require specialized habitat types (Wyman, 2003). Poor quality habitats may not support viable populations and these marginal habitats could potentially become species sinks depleting the larger- scale meta-population (McKinney, 2002).

Type of Vegetation

Human activities may result in the loss of aquatic vegetation within ponds, wetlands and streams, or the loss of forest and other upland terrestrial plant communities from the landscape. Aquatic vegetation provides shelter for larval and adult amphibians, and oviposition sites (Egan & Paton, 2004; Skidds, Golet, Paton & Mitchell, 2007), whereas terrestrial vegetation fringing ponds and wetlands, and upland plant communities, provide opportunities for dispersal, food, shelter and overwintering sites once individuals have metamorphosed (deMaynadier & Hunter, 1999). Forested wetlands also provide habitat for wetland- dependent amphibians (Baldwin, Calhoun & deMaynadier, 2006a). Along streams, changes in bed sediments, nutrient enrichment and turbidity contribute to a reduction in the diversity of stream macrophytes (Suren, 2000), and large woody debris is also reduced in urban streams (Paul & Meyer, 2001). In addition to vegetation removal, modifications to the structure and composition of vegetation in and around water bodies, wetlands and streams have implications for the ability of amphibian populations to persist. For example, over storey vegetation composed of exotic species of planted trees may encroach on urban ponds and result in increased pond shading, whereas weeds may smother the surface area of ponds, out-compete native aquatic species and reduce foraging success (Maerz, Blossey & Nuzzo, 2005).

Pond shading can lower water temperatures, reduce the concentration of dissolved oxygen, and decrease the abundance of periphyton, a common food source for larval amphibians, thereby depressing larval growth rates and activity levels (Skelly, Yurewicz, Werner & Relyea, 2002; Thurgate & Pechmann, 2007). Many amphibian species in North America that favour open, early successional habitats are usually absent from ponds where forest canopies have closed over the pond basin (Skelly, Werner & Cortwright, 1999; Werner, Yurewicz, Skelly & Relyea, 2007), and there is also a negative relationship between canopy cover and similar species in urban and suburban areas. In urban areas, water bodies, wetlands and streams may also be shaded by buildings, bridges and other urban-related infrastructure. Conversely, some forest-dependent amphibians (e.g. plethodontid salamanders) require mature forests with a closed canopy that provide cool, moist terrestrial microhabitats to complete their life cycle, and so are impacted by the removal of shady forest. Moreover, there are species that can inhabit ponds along the entire gradient of vegetation succession, such as the wood frog (*Rana sylvatica*), which is a canopy generalist (Skelly *et al.,* 2002).

Hydroperiod

Hydroperiod, the length of time a water body, wetland or stream continuously holds water, is known to strongly influence the structure and composition of amphibian communities (Wellborn, Skelly & Werner, 1996; Werner *et al.*, 2007). Hydroperiod is likely to invoke the strongest and most contrasting responses across amphibian communities in urban and suburban areas. For example, some species require ephemeral ponds for breeding that hold water briefly (e.g. one or two months), whereas others require permanent aquatic habitats that never dry out. Rubbo & Kiesecker (2005) suggested that hydroperiod may play a significant role in determining amphibian distributions across various human – disturbed environments owing to the complex life histories of individual species and the relationship between stream hydrology is a common outcome of urbanization, involving changes in the extent, duration, frequency and timing of inundation, and quantity and flow of water, respectively. Stream hydrology can be greatly modified in urban and suburban catchments; increased surface runoff often results in rapid flood peaks, thereby increasing flood magnitude and frequency (Paul & Meyer, 2001; Allan, 2004). Miller, Hess & Moorman (2007) posit that a combination of increased peak flows and sedimentation, reduced base flow and chemical changes likely reduce the abundance of salamanders in urban and suburban streams. Increased flood frequency and magnitude can result in scour of the stream banks, which removes coarse woody debris and disturbs in-stream vegetation (Ehrenfeld, 2000). Impacts of altered stream flow regimes on stream dwelling amphibians may include loss of shelter and breeding sites, reduced prey abundance .Adults and larvae may also be flushed downstream by high flow rates following heavy rains (Willson & Dorcas, 2003). The encroachment of urbanization into riparian zones has the potential to reduce the quality of habitat for amphibians and lead to population declines. For example, Price, Dorcas, Gallant, Klaver & Wilson (2006) suggested that the increased rate of urbanization from 1972 to 2000 near Davidson, North Carolina, USA, may be responsible for the significant and rapid decline in stream salamander populations reported from this region.

Predators and Competitors

The presence of predatory fish, particularly non-native species, in water bodies, wetlands and streams often results in a decrease in the presence and diversity of amphibians. The aquatic larvae of many amphibians are vulnerable to predation by exotic species of fish (Knapp & Matthews, 2000; Gillespie, 2001; Kats & Ferrer, 2003). Predatory fish are often absent from water bodies and wetlands with short hydroperiods because they are frequently dry, whereas predatory fish tend to persist in more permanent water bodies, which are often dominant in urban and suburban areas (Kentula, Gwin & Pierson, 2004). For example, Rubbo & Kiesecker (2005) reported that fish were more common in permanent wetlands in urban and suburban areas than in less permanent rural wetlands in central Pennsylvania, USA. Accordingly, they found that heavily disturbed wetlands in the area had lower larval amphibian species richness than rural wetlands. Many urban and suburban water bodies, wetlands and streams are also actively and accidentally stocked with exotic fish (Paul & Meyer, 2001; Ficetola & De Bernardi, 2004; Rubbo & Kiesecker, 2005) which reduces their suitability as habitat for amphibians that cannot co-exist with fish (Kiesecker, 2003). For example, the construction of permanent ponds and the introduction of non-native fish into wetlands in the Willamette Valley, Oregon, USA, which has promoted the spread of non-native bullfrogs, have been implicated in the decline of the Oregon spotted frog (*Rana pretiosa*) (Pearl, Adams, Leuthold & Bury, 2005).

Introduced invertebrates may also impact amphibian populations in natural ecosystems through predation. For example, Riley, Busteed, Kats, Vandergon, Lee, Dagit, Kerby, Fisher & Sauvajot, (2005) found that the presence of exotic crayfish (*Procambarus clarkia*) reduced the abundance of California tree frogs (*Hyla cadaverina*) in streams in southern California. They also suggested that habitat destruction and fragmentation had increased water depth and flow, resulting in more permanent streams, which allowed crayfish to persist, even in dry years.

In addition to the negative impacts of exotic fish and invertebrates on amphibian habitat quality, domestic pets, especially those that have become feral, may invoke high mortality on local amphibian populations. For example, Woods, McDonald & Harris (2003) estimated that a British population of approximately 9 million domestic cats killed 4–6 million reptiles and amphibians during a five-month survey period. Introduced amphibians may also compete with native amphibian species for limited resources in urban areas and ultimately displace local populations (Kiesecker, 2003). For example, modification of wetlands in western North America frequently benefits introduced bullfrogs because large, shallow, ephemeral wetlands are commonly converted to smaller permanent ponds (e.g. retention ponds), which provides the conditions required for successful bullfrog breeding (Adams, 1999).

Reduced vegetation and the spatial clumping of edge vegetation in the permanent ponds appear to intensify competition between larvae of the introduced bullfrog and the native red-legged frogs (*Rana aurora*). This more open habitat may also intensify predation by adult bullfrogs on larval and juvenile red-legged frogs (*Rana draytonii*).

Terrestrial Habitat

Many amphibians require terrestrial non-breeding habitat to access essential resources such as shelter and food as well as overwintering sites, and upland habitat may be a critical element of the habitat mosaic of pond-breeding amphibians (Semlitsch, 2000). These non-aquatic habitats (e.g. forests, grasslands) can be located adjacent to water bodies, wetlands and streams, or they can occur over hundreds of metres to kilometres from aquatic-breeding sites depending upon the species (Semlitsch, 1998; Trenham & Shaffer, 2005; Rittenhouse & Semlitsch, 2007). For example, Baldwin *et al.* (2006a) reported that the wood frog (*Rana sylvatia*) selected forested wetlands as summer refugia following use of breeding pools in spring; post breeding movements ranged from 102 to 340 m and included stopovers in upland forest floors. Terrestrial habitats also provide the necessary resources (rocks, woody debris and rotten logs) for amphibians with direct development (does not involve an aquatic larval stage) (e.g. plethodontid salamanders; Wyman, 2003). Thus, maintaining amphibian populations in any landscapes requires the conservation of not only aquatic habitats but the associated terrestrial habitats as well.

The quality of terrestrial habitat also determines whether amphibians can successfully disperse from breeding sites to upland forests and other wetlands in the surrounding landscape. The movement and survival of amphibians in the terrestrial environment are the critical components that ensure successful dispersal and re-colonization within regional meta-populations (Semlitsch, 2003), however, maintaining connectivity over terrestrial habitats is extremely challenging in urban and suburban landscapes (Gibbs, 2000). Disturbed areas through agriculture contain a suite of formidable barriers to amphibian movement. Dense networks of roads, buildings, fences and other physical barriers prevent many amphibians from successfully dispersing among the multiple habitat patches they need to access in order to fulfill critical life cycle processes (Knutson *et al.,* 1999; Dodd & Smith, 2003). Juvenile amphibians are often the most highly dispersive life stage of many species and are therefore at greatest risk of mortality in the upland habitat matrix, and many species avoid crossing open areas while emigrating (Rothermel & Semlitsch, 2002; Mazerolle & Desrochers, 2005).

Amphibians are susceptible to being killed while crossing roads, which may have significant impacts on amphibian populations in urban and suburban areas, especially close to breeding sites (Carr & Fahrig, 2001; Hels & Buchwald, 2001; Eigenbrod, Hecnar & Fahrig, 2008). For example, using population projections based on spotted salamander (*Ambystoma maculatum*) life tables, Gibbs & Shriver (2005) showed that an annual risk of road mortality for adults of >10% can lead to local population extirpation.

Water Quality and Pollution

Amphibians are generally regarded as being highly sensitive to environmental pollutants due to their biphasic life cycle and physiological requirements (Phillips, 1990; Blaustein, Wake & Sousa, 1994). Many water bodies, wetlands and streams receive storm water runoff from large areas of impervious surfaces such as roads, parking lots, buildings and open space composed of asphalt and concrete, which may contain a wide range of pollutants including heavy metals, phosphorus, fertilizers, pesticides, suspended solids, hydrocarbons and salts (Paul & Meyer, 2001).

Apart from direct application, pesticides may be deposited in farmland areas due to atmospheric transport from surrounding agricultural land (Boone & Bridges, 2003). Larvae of aquatic-breeding amphibians are most at risk of potential contamination because they are confined to the aquatic environment (Semlitsch, 2000). However, it has been suggested that terrestrial salamanders with direct development may also be sensitive to environmental contaminants, such as soil acidification arising from the deposition of airborne pollutants (Wyman, 2003).

Previous studies have documented the effect of sediments, nitrogen pollution and heavy metals on amphibians in disturbed environments, which have been shown to lower survivorship, growth and development rates (Boone & Bridges, 2003; Massal, Snodgrass & Casey, 2007). For example, Snodgrass, Komoroski, Bryan & Burger (2008) exposed embryonic and larval amphibians to sediments collected from storm water retention ponds, which had elevated levels of metals (e.g., zinc, lead and copper). They recorded 100% mortality in a species that is sensitive to urbanization (*Rana sylvatica*), whereas *Bufo americanus*, which is relatively insensitive to human activities, suffered relatively minor lethal effects and metamorphosed at a smaller size. However, a smaller size at metamorphosis can reduce survival to maturity and reproductive fitness, and therefore, impact on population dynamics (Smith, 1987; Berven, 1990). Snodgrass, Komoroski, Bryan & Burger (2008) suggested that storm water retention ponds could act as ecological traps for pond-breeding amphibians such as *R. sylvatica* because storm water ponds present cues that might be attractive (i.e. they contain vegetation and surface waters) and accumulate pollutants that may prove toxic. Differential sensitivity to water quality and pollutants may therefore occur within amphibian communities where some species are more sensitive than others (Marco, Quilchano & Blaustein 1999; Hammer *et al.*, 2004; Griffis-Kyle and Ritchie, 2007). The presence of dissolved metals and salts in water (i.e. high conductivity) and high nutrient loads negatively affect amphibian populations in urban and suburban areas. Finally, the increasing proportion of urban land use in a catchment generally decreases algal species diversity due to a reduction in water quality (Paul and Meyer, 2001), which would potentially decrease the amount of food for larval amphibians.

Disease

The most enigmatic pathogen to emerge as a potential agent in global amphibian declines since the 1970s is the chytrid fungus *Batrachochytrium dendrobatidis* (Bd), which causes the infectious disease chytridiomycosis in amphibians. This pathogen has been implicated in the mass mortalities in several amphibian species around the world (Daszak *et al.,* 2003; Muths *et al.,* 2003). In suburban landscapes, Bd may be transported by humans to areas supporting naïve amphibian populations through inadvertent or deliberate introduction of amphibians to new regions via releases of pet species (Carey *et al.,* 2003). For example, Daszak *et al.* (2004) demonstrated that bullfrogs can be infected by Bd, but are relatively resistant to chytridiomycosis, which is lethal to many other amphibian species. By demonstrating that bullfrogs are likely to be efficient carriers of this pathogen, their results showed that this host species is important in the spread of chytridiomycosis, particularly by commercial activities.

The virulence and density of pathogens such as Rana viruses and trematode parasites in amphibian populations has been shown to become intensified in urban and suburban areas supporting disturbed or degraded habitats (Johnson *et al.,* 1999; Carey *et al.,* 2003). Many of these pathogens are distributed among amphibian populations via the introduction of invasive species such as fish (e.g. trout and aquarium fish) and infected amphibians (e.g. exotic bullfrogs; Kiesecker, 2003). King *et al.* (2007), however, reported that urbanisation may hinder parasite transmission to frogs by limiting access of other vertebrate hosts of their parasites to wetlands.

Human Disturbance

Amphibians are known to respond to physical disturbance by humans (Rodrı´guez-Prieto & Ferna´ndez-Juricic, 2005), artificial light (Baker & Richardson, 2006) and noise pollution (Sun & Narins, 2005; Bee & Swanson, 2007), all of which may disrupt breeding behaviour, thereby potentially reducing recruitment rates and thus affecting population dynamics. For example, Baker & Richardson (2006) demonstrated that male green frogs (*Rana clamitans melanota*) produced fewer advertisement calls and moved more frequently when exposed to artificial light compared to ambient light conditions.

In a study of a mixed-species anuran calling assemblage in central Thailand, Sun & Narins (2005) showed that man-made acoustic interference (e.g. road traffic, airplanes) may directly affect anuran chorus behaviour. Urban streams and wetlands can experience high human visitation rates, either because of active recreational viewing or incidental visits. For example, Rodrı´guez-Prieto & Ferna´ndez-Juricic (2005) assessed the effects of recreational activities on Iberian frogs (*Rana iberica*) in the Guadarrama Mountains of central Spain. By simulating different levels of human visitation to stream banks, they found 80% and 100% decrease in stream bank use with a fivefold and a 12-fold increase in direct disturbance rate, respectively. Amphibians may also be collected by humans for food, fishing bait or as pets in urban and suburban areas, which may reduce population size or introduce species into previously uninhabited regions (Jensen and Camp, 2003).

The Conservation Status of Amphibians

The rampant depletion of biodiversity worldwide in recent decades is exemplified by the current status of the world's amphibian species. A recent study reports that nearly one-third of global amphibian population are threatened with extinction, many of which have not been seen in decades (Stuart *et al.,* 2004).While habitat destruction and over-exploitation have been observed to be the primary threats to much of the world's fauna, many amphibian declines and disappearances have taken place in protected wilderness areas where no obvious cause can be identified (Bradford, 1991), and unidentified processes threaten 48% of rapidly declining species (Stuart *et al*., 2004).

Amphibians have been declining worldwide for many years (Stuart *et al.,* 2004). Yet we lack even a basic understanding of the drivers of amphibian distributions and diversity at broad spatial scales. In Central America, declines have generally occurred above 500 m altitude, and in the Andes above 1,000 m (Young *et al.,* 2001). Population declines of montane harlequin frogs (*Atelopus* spp) have been particularly severe: all 28 upland species with sufficient population trend data have suffered declines, and 21 (75%) of these species are presumed extinct. In Australia, 41% of upland species (predominantly distributed above 400 m) are threatened, versus only 8% of lowland species (Hero & Morrison, 2004) and there are at least four tropical species (*Litoria nannotis, L. rheocola, Nyctimystes dayi* and *Taudactylus eungellensis*) whose upland populations have declined precipitously, while lowland populations have remained stable. Though the cooler temperatures associated with montane areas have been shown to increase the sensitivity of larval and embryonic amphibians to UV-B radiation, it is unlikely that the detrimental effects of UV-B would be a significant factor in the decline of montane rainforest amphibians worldwide (Richards *et al.,* 1993; Lips *et al.,* 2001), as the thick rainforest canopy provides substantial protection from harmful UV-B radiation. Conservation programmes urgently require the accurate identification of the causal agent(s) responsible for these high-altitude amphibian declines and extinctions.

Causes of Amphibians Species Declines and Actions to Remedy the Situation

Declines in amphibian populations, including population crashes and mass localized extinctions, have been noted since the 1980s from locations all over the world. These declines are perceived as one of the most critical threats to global biodiversity, and several causes have been suggested, including disease, habitat destruction and modification, exploitation, pollution, pesticide use, introduced species, and increased ultraviolet-B radiation (UV-B). The causative factors are even suspected to be acting in synergy in some instances (Blaustein & Kiesecker, 2002). However, many of the causes of amphibian declines are still poorly understood, and the topic is currently a subject of much ongoing research (McCallum, 2007).

Even though habitat loss may evidently remain the most significant threat overall (impacting 90% of those species currently considered threatened), it is believed that amphibians are more sensitive than other organisms to environmental deterioration and that the decimation of amphibians is a warning sign of an ever increasing poisoned environment. It is observed that many small mammals and herpetofauna (species generally have relatively short generation times and a quick response to habitat and microclimatic variations within forest fragments, Cain *et al.,* 2007). Thus, amphibians as bio-monitors can be used to measure this feature (Wasonga *et al.*, 2006).

The global extinction threats to amphibians have generated calls for proactive conservation activities (Stuart *et al.,* 2004) and have even prompted a five-year strategic plan by the Amphibian Conservation Action Plan (ACAP) in 11-pronged thematic areas. The strategy has designed a network of conservation sites for amphibians taking into consideration freshwater resources and associated terrestrial landscapes, climate change, biodiversity loss, amphibian declines and captive programmes (Moore and Church, 2008).

UV-B Radiation as a Possible Cause of Amphibian Declines

Anthropogenic ozone depletion has significantly increased UV-B (280–315 nm wavelengths) radiation at ground level in high latitudes. UV-B has multiple effects on ecosystems, some harmful and some beneficial to individual organisms. Experiments in natural ponds, mostly in mountainous regions of North America, implicated increases in UVB radiation as a possible cause of amphibian declines.

Embryos of some species survived much better when shielded from UV-B than when exposed to current ambient levels (Blaustein *et al.*, 1994). Similar damaging effects of elevated UV-B have also been observed in the laboratory, including with European species. Furthermore, satellite-based measures of UV radiation levels at 20 sites in Central and South America recorded increases between 1979 and 1998 that were greatest in areas where amphibian declines have been most severe (Middleton *et al*., 2001).

However, embryos and larvae of many declining species in tropical rainforests are not exposed to UV-B in the same way as those reproducing in the open mountain pools and lakes of temperate countries. Another problem with attributing declines to increases in UV-B radiation is that moderate concentrations of dissolved organic matter (DOM) ameliorate the effects of UV-B below the water surface. One recent study suggests that eggs and larvae in the majority of breeding sites used by amphibians in North American mountain regions are well protected by the DOM (Heyer, 2003). Nevertheless, the significance of increased UV-B radiation in amphibian declines remains uncertain, and ongoing conflicts over methodologies, analysis and interpretation of available data show no sign of waning (Cummins, 2002; Blaustein et al., 2003; Heyer, 2003). As is often the case with amphibian responses to other anthropogenic factors, it seems likely that responses to UV-

B – as well as interactions between UV-B and other agents of decline – vary considerably among species, regions and microclimates.

Impacts of Climate Change on Amphibians

Recent changes in the global climate might impact adversely on amphibian populations. Global mean temperature rose by about 0.6 °C over the past 100 years with an accelerating trend since the 1970s, and there is increasing evidence for multiple effects of climate change on wildlife and ecosystems (Walther & Gosler, 2001).

There is currently no evidence that climate change has led to tolerance limits in temperature or moisture being exceeded in amphibians. There have, however, been detectable effects of climate change on breeding phenology although the extent of this varies between studies. These differences may represent real variation between species and regions, or may be a function of different methodologies or study timeframes. Asynchronous changes in phenology might alter predation rates and thus disadvantage particular species within communities. The golden toad (*Bufo periglenes*) of the Costa Rican rainforest disappeared completely at the end of the 1980s and has not been seen since (Pounds *et al.,* 1999).

Many species of this rainforest biota declined over the past 20 years and several taxa previously restricted to lower altitudes have ascended higher into the mountains. The forest has consequently become drier, and amphibian (Beebe, 2005) breeding less successful (Pounds *et al*., 1999). However, the recent climate patterns are not unprecedented and there is no evidence that similar conditions within the past 50 years led to amphibian declines

(Alexander & Eischeid, 2001). It is therefore uncertain as to whether recent climate change is a significant cause of amphibian declines.

Use of Amphibians as Ecosystem Indicators

Ecological indicators can have many purposes, including being used to assess the condition of the environment or monitor trends in condition over time (Cairns *et al*., 1993). Some species suitable for monitoring trends in condition over time may be useful as indicators of restoration success in ecosystems in which restoration activities are occurring. Amphibians are widely considered to be useful as indicator species (Welsh & Ollivier, 1998; Sheridan & Olson, 2003),

Amphibian species or communities have been touted as useful indicators in many situations recently (Welsh & Ollivier, 1998; Hammer *et al.,* 2004). Some studies use amphibians as indicators of environmental contamination or pollution (Hammer *et al.,* 2004). Others attempt to use the species assemblage (Sheridan & Olson, 2003) or the abundance of populations (Campbell *et al.,* 2005) as indicators of ecosystem health or habitat quality.

Characteristics that Render Amphibians Useful Indicators of Healthy Ecosystems

Amphibians have several characteristics that make them useful as indicator species and these include their permeable skin and biphasic life cycle that are likely sensitive to environmental stress (Blaustein *et al*., 1994), but there is some debate about whether this sensitivity is consistent and predictable. Amphibian distribution, population size, abundance, site occurrence, diversity, and their growth rate have considerably affected environmental conditions. Because amphibians have intimate contact with many components of the environment due to their highly permeable skins, they are considered to be valuable gauges of environmental health or stress (Blaustein*et al.,* 1994). Amphibians are also functionally important for nutrient cycling and ecosystem energy-flow in most freshwater and terrestrial habitats (Beebe, 2005).

Since amphibians are primary consumers as larvae and primary predators as adults (Blaustein *et al.,* 1994), as well as being prey for other invertebrates and vertebrates (Duellman & Trueb, 1986). It is likely that amphibians will be good indicators of changes to the whole ecosystem because they are sensitive to changes in the aquatic and terrestrial environments. The aquatic environment is required for reproduction in most species (Duellman & Trueb, 1986) and the permeable skin of amphibians makes them sensitive to water quality and UV radiation in the egg and larval as well as adult life stages (Taylor, Fahrig & With, 2005).

Terrestrial Habitats and Amphibian Conservation

Many amphibians require terrestrial non-breeding habitat to access essential resources such as shelter and food as well as over-wintering sites, and upland habitat may be a critical element of the habitat mosaic of pond-breeding amphibians (Semlitsch, 2000). These non-aquatic habitats such as forests and grasslands can be located adjacent to water bodies, wetlands and streams, highlands or they can occur over hundreds of meters to kilometres from aquatic-breeding sites depending upon the species (Semlitsch, 2000).

Terrestrial habitats also provide the necessary resources (rocks, woody debris and rotten logs) for amphibians with direct development that do not involve an aquatic larval stage (e.g. plethodontid salamanders). The quality of terrestrial habitat also determines whether amphibians can successfully disperse from breeding sites to upland forests and other wetlands in the surrounding landscape. The movement and survival of amphibians in the terrestrial environment is the critical component that ensures successful dispersal and recolonisation within regional meta-populations (Rothermel & Semlitsch, 2002).

Juvenile amphibians are often the most highly dispersive life stage of many species and are therefore at greatest risk of mortality in the upland habitat matrix, and many species avoid crossing open areas while emigrating (Rothermel & Semlitsch, 2002). Relatively few landscape-level studies of amphibian density and movement have been conducted. Most existing studies have focused on relationships between forest cover and species occurrence. These have shown positive relationships between amphibian populations and area of forest in the surrounding landscape and negative relationships with urban development and roads.

Studies of landscape composition effects have found relationships between forest cover and amphibian presence at spatial scales ranging from 100 m to over 3000 m radii. Several studies also note a general pattern of increased species richness with increasing forest cover (Gibbs, 1998). Despite these generalizations, reliable inferences about habitat area effects require attention to species-specific ecological characteristics and their interactions with environmental conditions at a range of spatial scales. Species-specific characterization of habitat is essential if scientists are to evaluate the effects of habitat loss on populations. For example, the suggestion that forest cover in the landscape benefits amphibians may not apply to species that are fully aquatic or that depend on non-forested upland habitat.

In addition, the location and slope of critical thresholds in habitat amount are species specific, and related to reproductive potential, dispersal ability, home range size, habitat specificity, and other characteristics. Thus, it is essential to explicitly link the habitat tolerances of a species to the extent and pattern of those habitats in the landscape if one is to produce reliable inferences about relationships between habitat area and species distributions.

Habitat Destruction and Amphibian Species Decline

Humans currently appropriate more than one third of the production of terrestrial ecosystems and about half of the usable fresh water on earth (Tilman, Reich, Knops, Wedin & Mielke, 2001) and the rapid growth of human population shows no signs of slowing. It is surprising then that habitat loss is one of the most significant treats to terrestrial biodiversity (Brooks, Pimm & Oyugi, 1999). Humans alter and destroy habitats by logging forests, draining swamps, paving grasslands, damming rivers introducing weeds and through other actions and pose dangers to wildlife especially amphibians.

In Ghana alone, natural forests have diminished to about 11.8–14.5% of their former cover over a few decades (IUCN, 1996). Worldwide, habitat loss and forest fragmentation are recognized as key factors driving the global extinction of genetically distinct populations and species (Hughes, 1988; Brooks et al., 1999; Stuart *et al.*, 2004).Habitats loss, alteration and fragmentation are likely the primary cause of amphibian species decline and extinction worldwide. For example in Australia, habitat fragmentation is associated with decline in lowland frogs, negatively impacting on the 11 of the

12 threatened lowland species (Hughes *et al.,* 1988). Habitats alteration can directly remove amphibians breeding and feeding areas or block access to them. Deforestation alters amphibian species assemblages and reduces species diversity on the landscape scale (Hughes *et al.,* 1988).

Another major concern is the loss of important wetlands which contain unique amphibian assemblages yet often receive little legal protection (Adams, 1999; Gibbs, 2000). Conservation of amphibians has focused on protecting breeding habitat (streams and pond) used by all amphibians in all stages of life cycle (egg, larval, juvenile and adult stages) (Taylor *et al.,* 2006) including uplands is of crucial concern to ecologist because of their ecological contributions to the environment.

The Herpetofauna of Ghana

Rain forests currently cover about 7% of the African continent, and represent slightly more than one fifth of the total remaining tropical forest worldwide. While rain forests everywhere are under severe and increasing pressures, a recent survey indicates that African forests, relative to forest of Asia and Latin America, are the most depleted, representing only about one-third of their historical extent (Collins, 1992).

West African rain forests are one of the 25 most important biodiversity hotspots of the world (Myers *et al.* 2000). They are highly threatened by logging, agriculture and increasing human populations (Bakarr, Bailey, Byler, Hams, Olivieri & Omland, 2001). In Ghana even less of the original forest cover is still present. West Africa was the target of herpetological investigations for more than 100 years, but the present knowledge is still rather scanty. Hughes (1988) provides an overview on the history of herpetological investigations in Ghana. Most of the described West African amphibian and reptile biological data are still completely lacking certain information. According to an unpublished checklist, only 36 amphibian species have been recorded so far from Guinea to Ghana and most of the species still remains undiscovered (Rödel *et al*., 2005)

A recent survey of amphibians of the Togo Hills concluded that, with 31 amphibian species, the area is more diverse than previously assumed and probably contains at least 41 amphibian species (Rödel & Agyei, 2003). The periphery of Kyabobo National Park was also surveyed and found to contain a total of 20 frog species (Rödel & Agyei, 2003). More investigations in Ghana were not focused on forest habitats or were not undertaken with special emphasis on amphibians (Hughes, 1988). Judging from a similar study in Côte d'Ivoire, the results show that the herpetofauna of Ghana is probably only very incompletely known. This especially concerns the almost neglected Togo highlands in the east of the country. For this reason the eastern Ghanaian forests has been defined as an area with an exceptionally high priority level for rapid assessment during the Conservation Priority Setting Workshop in Ghana (Bakarr *et al.,* 2001). This study is therefore undertaken to contribute to the scanty literature or investigations on herpetofauna in Ghana with emphasis on the forest habitat.

CHAPTER THREE

MATERIALS AND METHODS

Study Area

The study was carried out at the Mpameso Forest Reserve and surrounding cocoa farm and teak plantation (Fig 1and 2). The three study areas shared boundaries at the east of the Mpameso Forest Reserve and serve as a protected and conservation area, commercial and subsistence farming area or as an afforestation area. They are located within the Dormaa-Ahenkro District of the Brong-Ahafo Region of Ghana. The mean annual rainfall for the area is 1120mm and the mean daily temperature ranges from 25° C in the wet season (March-October) to 27°C during the dry season (November - February) (Dormaa East District Assembly, 2010). The topography is generally undulating and rises between 180 metres and 375 metres above sea level.

Mpameso Forest Reserve (MFR)

Mpameso Forest Reserve is a protected area under the jurisdiction of the Forestry Services Division of the Ghana Forestry Commission. It covers a land area of about 189km² approximately 3.5%of the total land area of the Brong-Ahafo Region of Ghana and about 0.6 % of the total country surface. The area is categorised as a dry moist semi-deciduous forest ecological zone (Hall & Swaine, 1981) and it is mostly a degraded secondary forest. The major types of flora found in these forests include shrubs, climbers and giant silk cotton trees. Timber species including wawa (*Triplochiton scleroxylon*), odum (*Milicia excelsa*), sapele (*Guthagrophrama* sp), African teak (*Pericopsis elata*) and mahogany (*Swietenia mahogoni*) are found in the reserve.

Cocoa Farm

The research was also carried out in a cocoa farm, which covers an area of about 1.8km². The farm lies adjacent to the east of the Mpameso Forest Reserve. It was established about six years ago by certain individuals in the nearby villages purposely as a cash crop from which they could earn some income to support their families. Within the cocoa farm were recently grown plantain and cocoyam to feed the farmers and their families. The farm is visited frequently by the farmers to perform the necessary cultural practices needed to ensure healthy growth of the plant and quality of its yield. The farmers clear the weeds to allow free circulation of air, as well as preventing rodents from destroying the produce of the farm. They also remove parasitic plants from the cocoa to prevent competition for nutrients, sunlight and avoid the formation of canopies as well. Chemicals such as confidor, ridomin gold, sungikil 50 and cocide 2000 are applied on the cocoa to prevent the browning, blackening and rotting of the cocoa pods and beans. The farmers also apply dichloro-diphenyl-trichloroethane (DDT), aceta star and arimon star to prevent the defoliation, yield reduction and death of trees. A chemical known as Roundup is also used to prevent weeds in the farm.

Teak Plantation

The teak plantation is found adjacent to the east of the MFR. It covers a land area of about 37.8 km². This plantation consists of a mixture of acacia (*Acacia polyacantha*) and teak (*Tectona grandis*), but there were more teak than acacia. The acacia is located at the edges of the plantation to act as a fire belt as well as along the riverine areas of the forest. The plantation was established in 2004, about 10 years ago. It used to be a bare land covered with

Acheampong weed (*Chromolaena odorata*). The teak plantation serves as an afforestation plantation to restore tree cover that was destroyed through logging, fire and other anthropogenic activities. There has not been any planting previously, as this is the first plantation in the history of the area around the forest reserve. This plantation was also established under the jurisdiction of the Forest Services Division of Forestry Commission and is visited occasionally by the forest guards to prune off branches of the trees. They also weed the edges to demarcate the boundary of the forest reserve and log some of the teak to prevent overcrowding. The logging has been done about three times within the 10 years of planting.

Research Design

The data for this study were collected from three specific areas identified to represent three different land use types. Mpameso Forest Reserve represented a protected and conservation area; a teak-acacia plantation represented an afforestation plantation area and; cocoa farm, a commercial and subsistence farmland. The selection of these areas was necessary to determine the impact of land use types on the diversity, distribution and density of amphibian species. Three transect lines were thus established in each study site and a total of 10 sample plots established along each transect. Ninety plots (30 plots per study site) were established and searched for amphibians. Each plot measured 25m×25m and the distance between transects in each study area was at least 100m. The distance between the plots was at least 25 m.

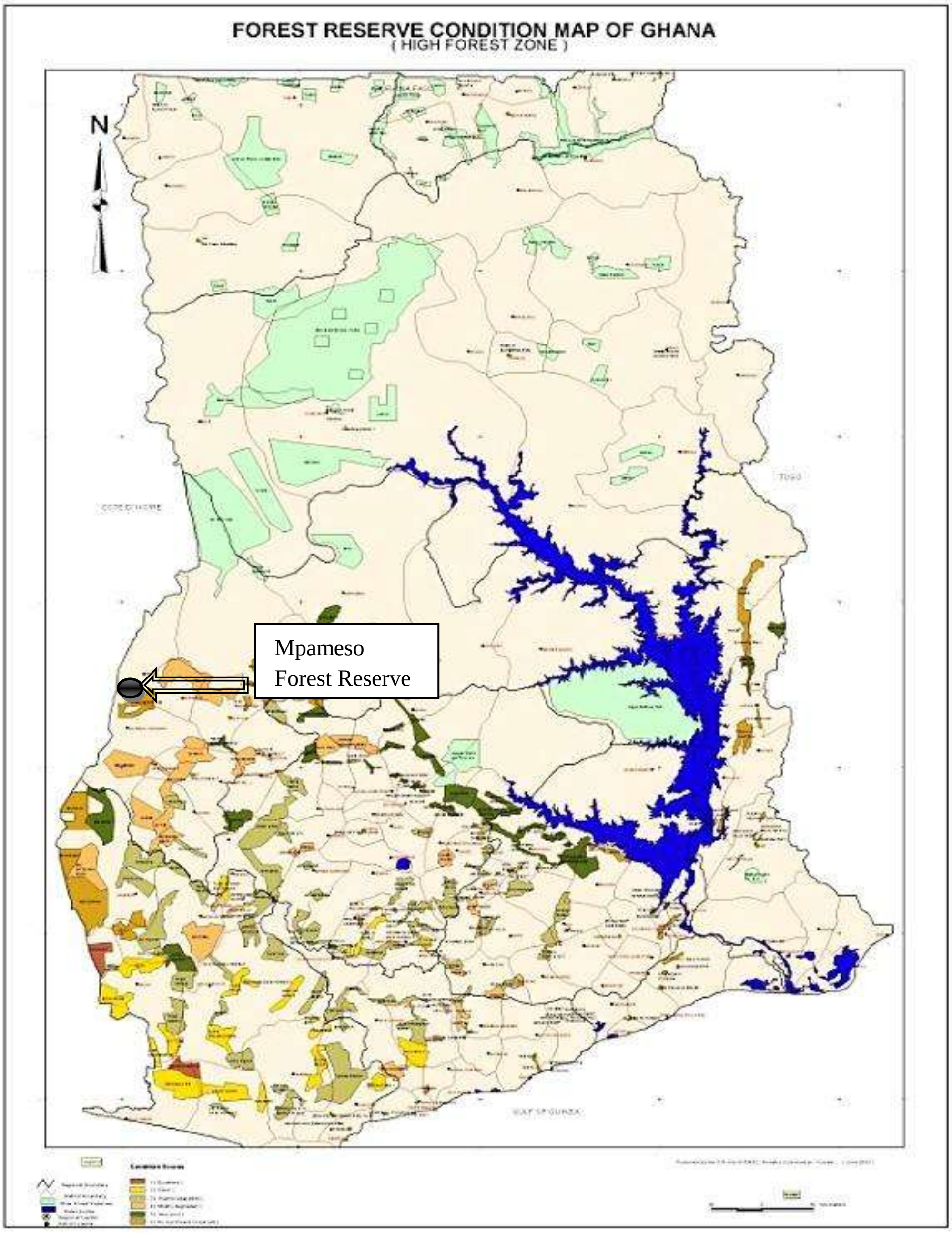

Figure 1: Map of Ghana showing the study area

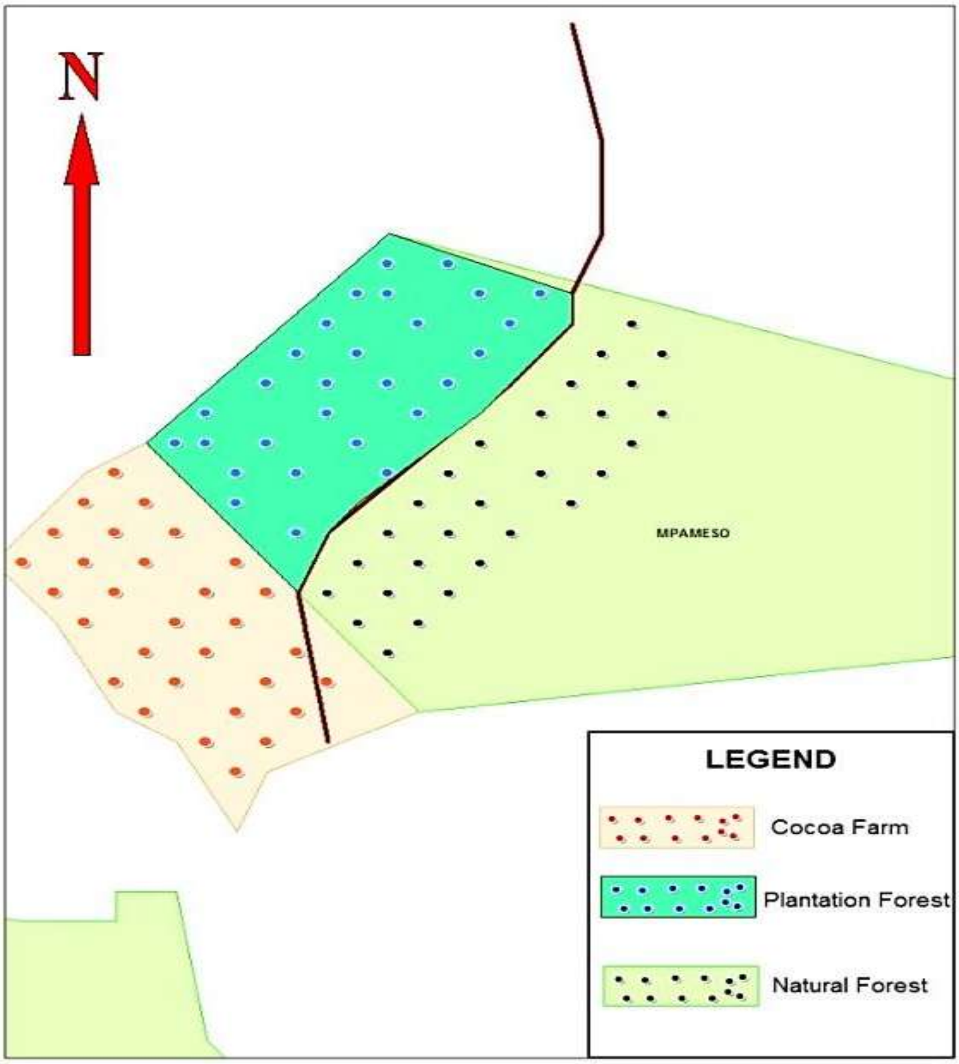

Figure 2: Layout of the three land use types

Data Collection

Specimens were found by visual encounter surveys (VES) (Heyer *et al*., 1994; Rödel & Ernst 2004; Monney *et al.,* 2011), a method widely used for sampling reptiles and amphibians (Crump & Scott, 1994; Doan, 2003). A plot is searched for amphibians for a specified period of time. Usually the number of individuals of a species counted is standardized by the time used for the search or area searched (i.e. effort) to determine the relative abundance of the species. Each amphibian was then released in the same habitat after measuring the snout-vent length and taking a photograph of the specimen. Surveys were conducted during the day only and were intensive to increase the effort to detect all available species. This was necessary to avoid nocturnal surveys because of lack of escorts by the forest guards due to security reasons and also to follow the examples of other studies in Ghana (Monney *et al.,* 2011; Adum *et al.,*2013) in which exclusive diurnal surveys were effective. Data were collected in both the wet season (October – November) and the dry season (January – February).

Data Analysis

The data were analysed using Microsoft Excel software 2007 and PAST (Hammer *et al*., 2001). The mean and standard deviation (SD) for each species was calculated. The species richness and diversity were obtained for the forest, plantation and the farmland. The Shannon indices were used to determine species diversity in each habitat type. The Kruskal-Wallis test was used to determine the significance at ($p=0.5$) among the three land use types, while Mann-Whitney test assessed the significance level between two land use types. Furthermore, correlation analysis was used to establish the relationship between amphibian species richness and distribution at ($p=0.5$) these indices made use of t-test at 95% confidence level. Sörensen analysis was also conducted to identify species that share common microhabitat.

CHAPTER FOUR

RESULTS

Seasonal distribution of anurans

A total of 1,187 individuals of anurans, comprising 16 species belonging to six families were encountered during the survey. The highest number of individuals was encountered in the forest (518), followed by the cocoa farm (408 individuals) and teak plantation (261 individuals) in both the rainy and dry seasons.

A total of 786 anurans were recorded during the rainy season, comprising 16 species belonging to six families (Fig 3). Out of these Petropedetidae, representing 35.9% were the most frequently encountered family in the rainy season, followed by Athroleptidae (16.2%), Bufonidae (15.3%), Hyperoliidae (15.1%), Ptychadenidae (12.8%) and the least Ranidae (4.7%). In the dry season, a total of 401 anurans comprising 10 species belonging to six families were encountered during the survey (Fig. 4). The family Petropedetidae was again most frequently encountered with 33.1%, followed by family Athroleptidae (21.7%), Hyperoliidae with 18.7%, Bufonidae (17.0%), and Ptychadenidae (7.0%) whilst anurans belonging to the Ranidae (2.5%) recorded the least number of family distributed across the various land use areas.

In terms of the significant differences between the two seasons, the rainy season recorded a larger number of anurans (786) representing 66.2% of the total individuals encountered for both seasons as compared to the dry season(401) which also represents 33.8% of anurans encountered.

Again, the rainy season recorded a higher number of species (16) compared to the dry season (10). Some of the species were encountered only in the rainy season but not the dry season (Fig. 3). The rainy season species included *Amnirana galamensis, Ptychadena oxyrhynchus, Phrynobatrachus alleni, Phrynobatrachus plicatus, Leptopelis viridis and Hyperolius cf. vindigulosus.* Most of the species were more abundant in the rainy season than the dry season, *Arthroleptis* spp recorded 127 individuals in the rainy season as compared to 87 in the dry season, *Bufo maculatus* recorded 54 in the rainy season and 27 in the dry season, *Bufo regularis* recorded 66 in the rainy season compared to 41 individuals in the dry season. There were 156 individuals of *Phrynobatrachus calcaratus* species in the rainy season as compared to 108 in the dry season. On the other hand, species such as *Ptychadena bibroni,, Hyperolius cf. torrentis* and *Leptopelis hyloides* recorded 108,35 and 43 individuals respectively and were more abundant in the dry season than *Ptychadena bibroni,* (25), *Hyperolius cf. torrentis,* (31) and *Leptopelis hyloides*, (30)in the rainy season.

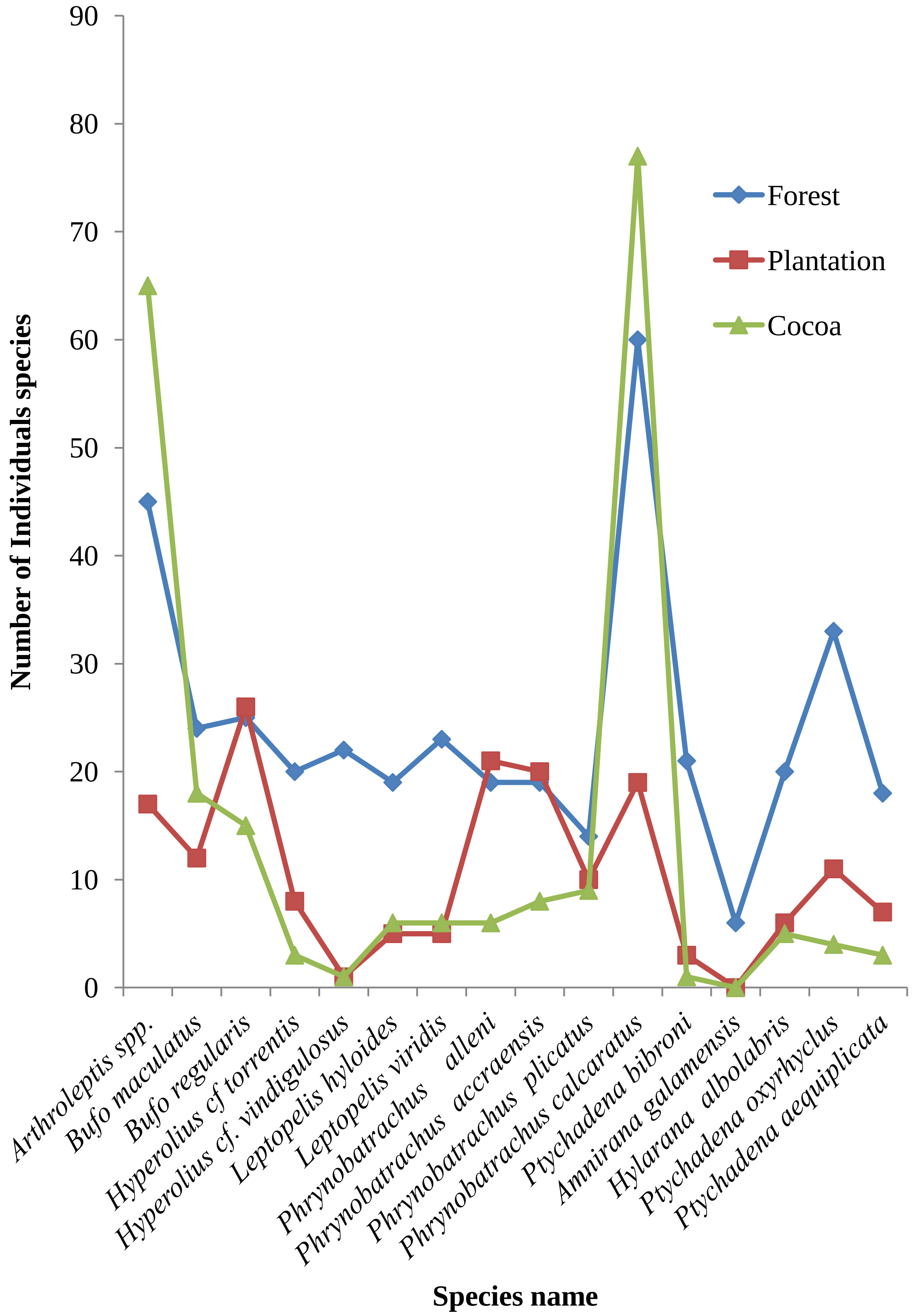

Figure 3: Distribution of Anuran Species in the Rainy Season

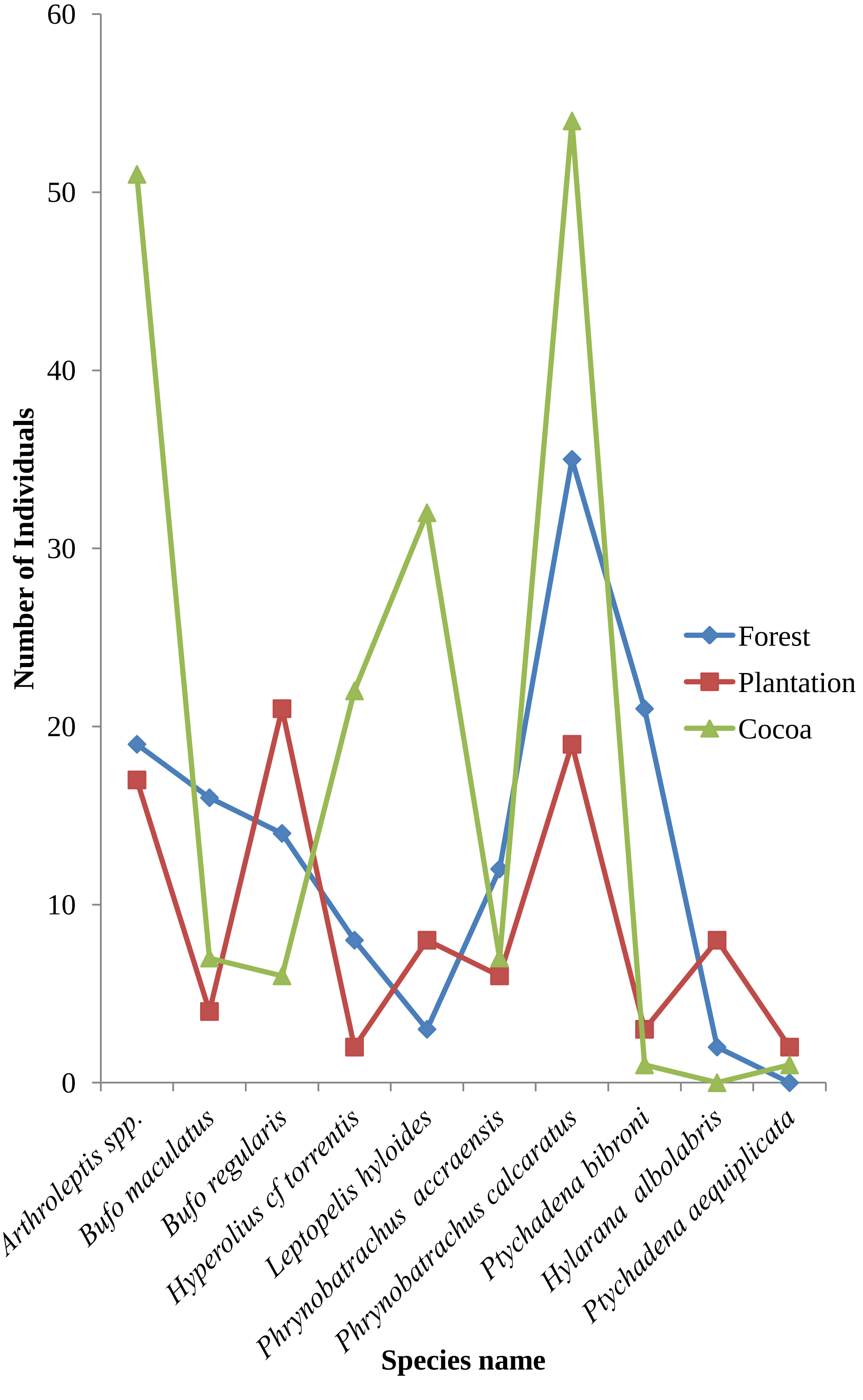

Figure 4: Distribution of Anuran species in the Dry Season

Distribution in the land use types

A total of 1,187 individuals were encountered during the survey in all the three habitat types, being the most abundant in the forest area (518), representing 43.6%, followed by cocoa(408) representing 34.4% and teak plantation(261) representing 22%.

In the rainy season, the highest number of anurans was recorded in the forest area with 388 individuals (49.4%) belonging to 16 species and 6 families compared to the cocoa growing area with 227 individuals (28.9%) belonging to 15 species and six families while the teak plantation recorded the least number of individuals 171, 21.7%) belonging to 15 species comprising six families. The same species were common among all the three habitat types except *Amnirana galamensis* which was not encountered in the cocoa and teak plantation and this may imply that the common species could resist environmental disturbance from either natural or human systems.

The distributions of individual anuran species of the land use areas were not the same. For instance in the forest area *Arthroleptis* spp., *Phrynobatrachus calcaratus* and *Ptychadena oxyrhynchus* were dominating, recording 11.6%, 15.5% and 8.5% respectively as compared to the teak plantation where *Phrynobatrachus alleni (*12%), *Bufo regularis* (14.9%), *Bufo maculatus* (6.9%), *Arthroleptis* spp. (9.7%), *Phrynobatrachus accraensis (*11.6%),were identified as the most dominant species whereas in the cocoa farm, the species which were found to be dominating were *Phrynobatrachus calcaratus*(33.9%), *Arthroleptis* spp (28.6%), *Bufomaculatus* (7.9%).

In the dry season, out of the 401 anurans, the cocoa farm recorded the highest number of individuals; 181individuals (45.1%) belonging to nine species and five families, followed by the forest; 130 individuals (32.4%) also belonging to 9 species and six families, while the teak plantation recorded the least number of individuals (90) representing 22.4% which belong to 10 species and six families. The distribution of anurans across the three land use types during the dry season shows that, in the forest area the most dominating species include *Phrynobatrachus calcaratus* (26.9%), *Arthroleptis* spp (14.6%),*Bufo maculatus*(12.3%), *Bufo regularis* (10.8%),*Phrynobatrachus accraensis* (9.2%), *Ptychadena bibroni* (16.2%), and *Hyperolius cf. torrentis* (6.2%). It was also revealed by the study that in the plantation area, *Bufo regularis* was recorded as the most dominant species (23.3%), *Arthroleptis* spp (18.9%), *Phrynobatrachus calcaratus* (21.8%),whereas *Phrynobatrachus accraensis* (6.7%), *Leptopelis hyloides* (8.9%)and *Hylarana albolabris* (8.9%) moderately dominating in the plantation area.

The survey also revealed that in the cocoa farm where anuran numbers were higher, *Arthroleptis* spp (28.2%), *Phrynobatrachus calcaratus* (29.8%), *Leptopelis hyloides* (17.7%) and *Hyperolius cf. torrentis* (12.2%) were still found to be dominating despite human disturbance.

Density of Anurans

In all, a total of 786 anurans were identified during the survey in the rainy season covering a land area of 5.6 hectares. The distribution patterns of anurans with regard to density in the three land use types indicated that the forest recorded the highest number of anurans in terms of density, with the mean number of 207 (SD=12.64, N=30) individuals per hectare, the density of the teak plantation was 93 (SD=7.80, N=30) anurans per hectare while the cocoa farm also recorded a total of 148 (SD=22.80, N=30) anurans per hectare (Fig. 5).

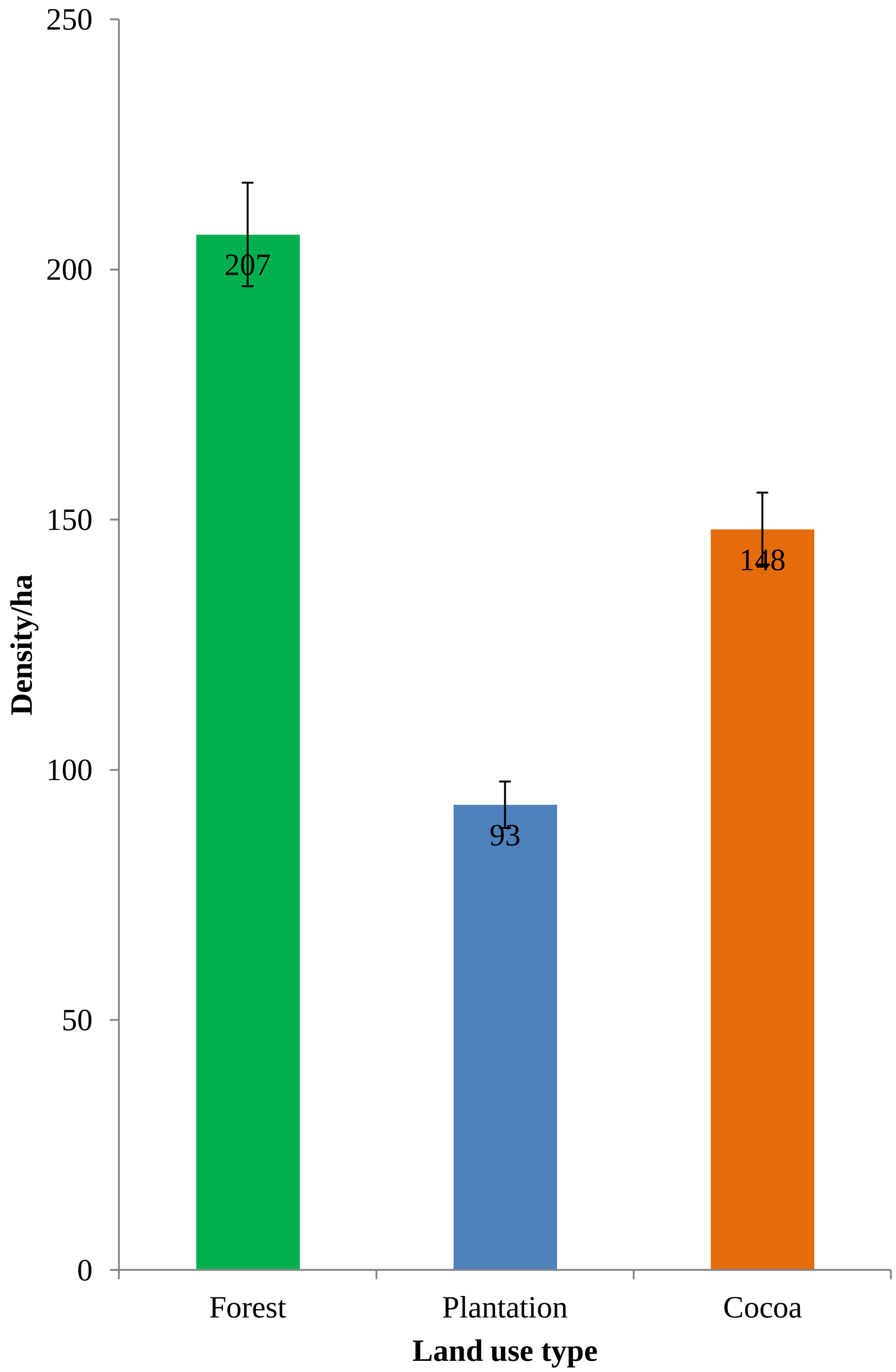

Figure 5: Density of anurans in the three land use types during the Rainy Season. (Error bars represent Kruskal-Wallis tests at 0.05% confidence interval)

Comparing the three land use types, Kruskall Wallis test revealed significant differences in anuran distribution among the forest, teak plantation and cocoa farm. The forest area recorded the highest number of individuals than the cocoa farm whilst the teak plantation recorded the least number (H=15.16, p=0.005). Further tests using the Mann-Whitney U – test however showed a significant difference in the abundance of anurans population between the forest and teak plantation (*U*=40.5, *p*=0.001), and forest and cocoa farm (*U*=40, *p*=0.001) but there was no variation found in the abundance of anurans between the teak plantation and the cocoa farm (*U*=102, *p*=3.461); this might be due to the fact that, teak plantation and cocoa farm are both disturbed habitats with similar micro-climatic conditions which support common anuran species.

Investigation of anuran density in the dry season indicates that the cocoa farm recorded the highest density of amphibians 233(SD=20.78, N=30) per hectare, followed by the undisturbed area; forest 184(SD=10.59, N=30) per hectare while the density of the teak plantation was recorded as the area with least number of individuals 176(SD=7.29, N=30) (Fig. 6). Using Kruskal Wallis test however indicate no difference among the density of the three land use types (*H*=0.01, *p*=0.94). Further analysis was carried out in determining similarities or difference between the various land use types sampled. The result revealed no significant differences between the density of individual species found in the forest and teak plantation (*u*=67.5, *p*=81), teak plantation and cocoa farm (u=67)

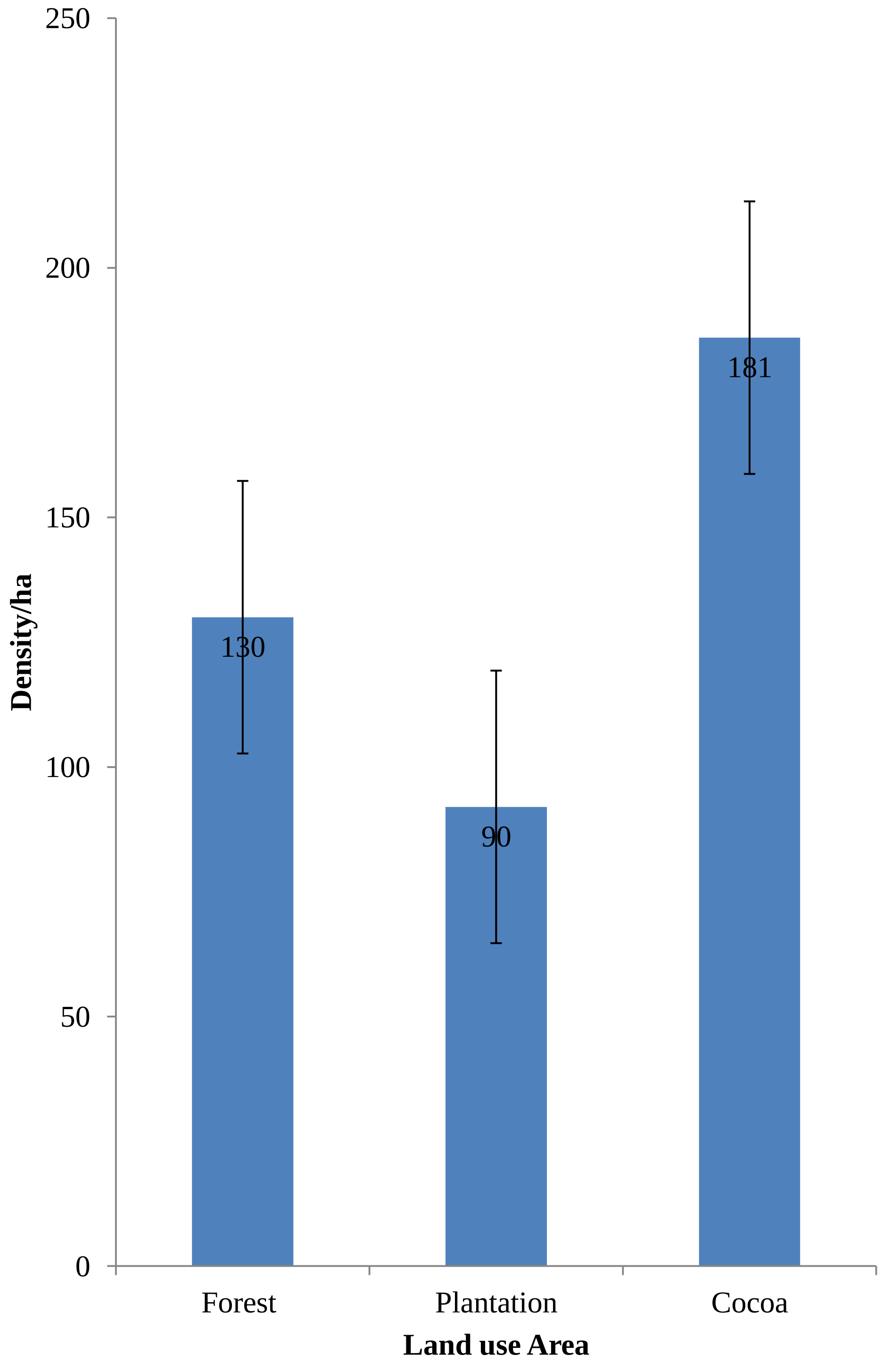

Figure 6: Density of anurans in the three Land use types during the Dry Season. (Error bars represent Kruskal-Wallis tests at 0.05% confidence interval)

Species diversity

The diversity of anurans as determined by Shannon index in the forest during the rainy season was 2.661(evenness=0.9596), teak plantation= 2.497(Evenness= 0.992) and cocoa farm= 0.7862(evenness=0.7225) (Fig 7). Furthermore, diversity *t*-test revealed a significant difference in anuran diversity between the forest and the teak plantation (*t*=3.5301, *p*=0.0004).Similarly, a significant difference was found between the teak plantation and cocoa farm (*t*=6.0467, *p*=3.6705E^{-9}) as well as between forest and cocoa farm (*t*=-9.0984, *p*=1.7706E^{-17}).

Similarly, the diversity of anurans as determined by Shannon index in the forest during the dry season indicates that, 1.969(evenness=0.8963), teak plantation 2.014(Evenness= 0.8749) and cocoa farm 1.702(evenness=0.7747) (Fig.4.6). Further statistics however found differences in the diversity of anurans in the forest and teak plantation during the dry season (*t*=-0.28057, *p*=0.77938E^{-17}), between teak plantation and cocoa farm (*t*=-3.3223, *p*=0.001) and also between forest and the cocoa (t=-3.3223, P=0.0001), respectively.

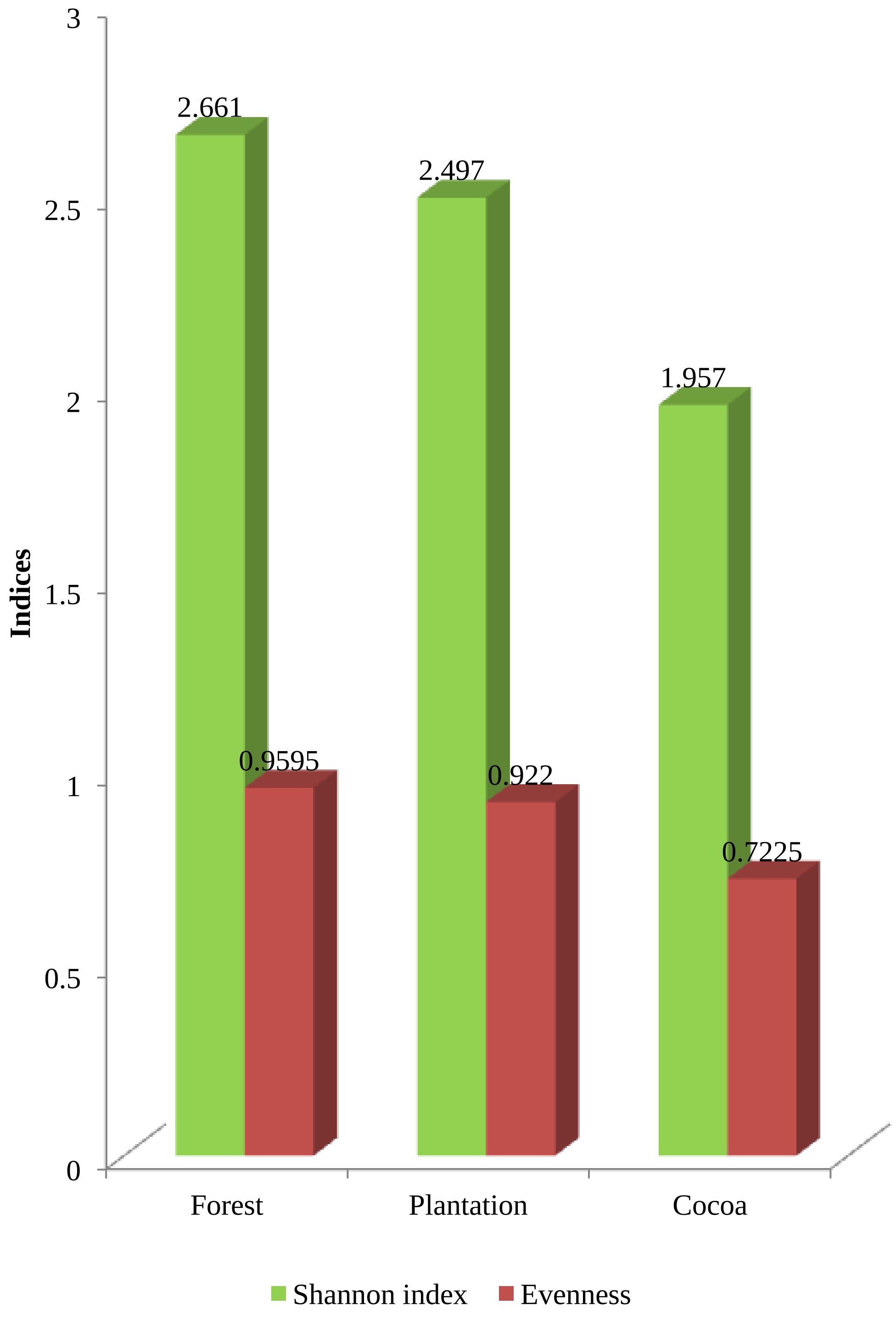

Figure 7: Diversity indices of the anurans in the Rainy Season

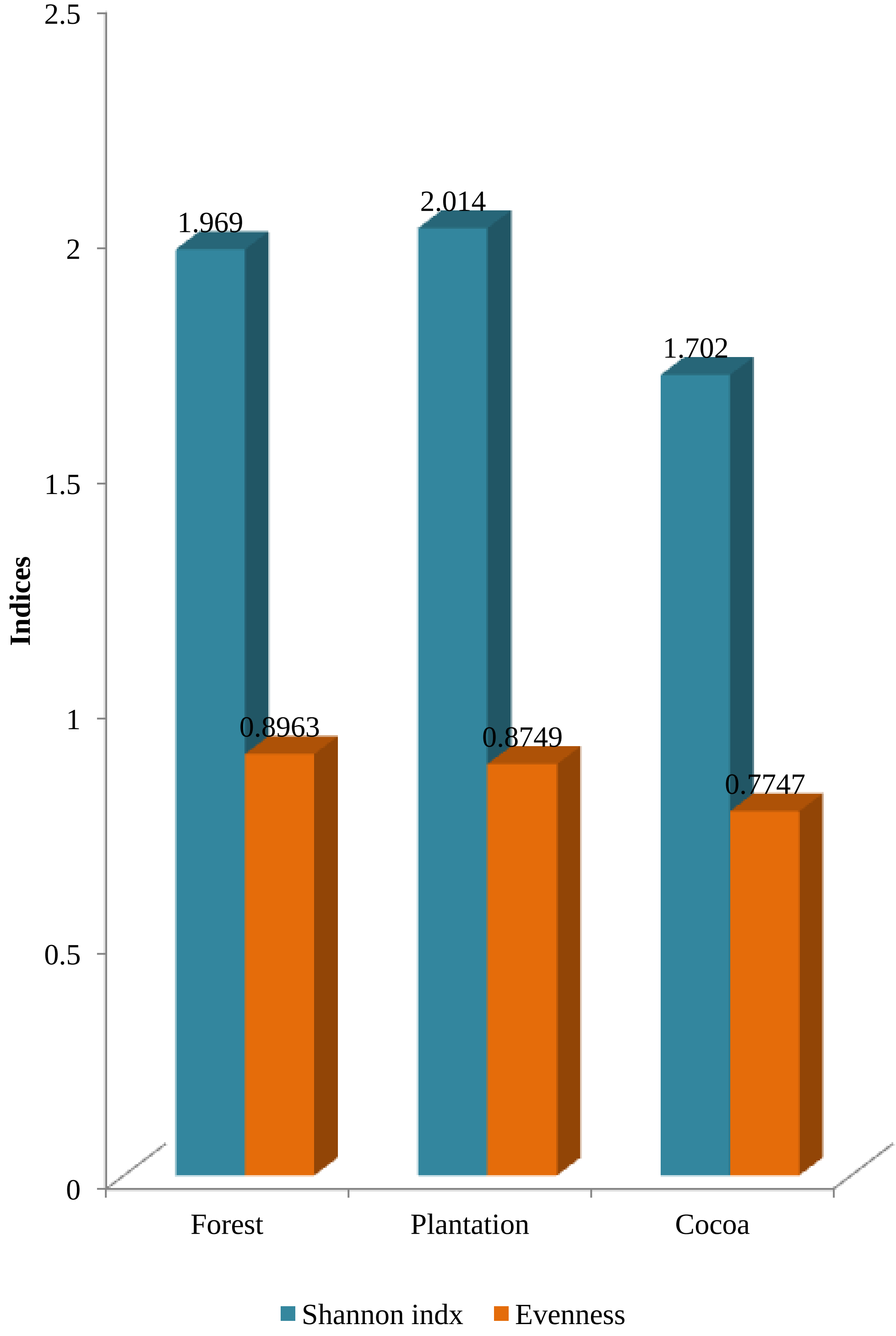

Figure 8: Diversity indices of the anurans in the Dry Season

Seasonal Variability of Anuran Species Diversity

The study compared the diversity of anurans in the same study area in both rainy and dry seasons and found no differences in the diversity of anurans between the forest in both rainy and dry seasons (t=-11.959, p=5.4902E^{-25}), the teak plantation in the rainy season and the dry season (t=5.5551, p=1.491^{E-7}). The reason for no difference in species diversity could be due to the occurrences of similar species such as *Arthroleptis* spp., *Phrynobatrachus calcaratus*, *Ptychadena oxyrhynchus*, *Bufo regularis*, *Bufo maculatus, Arthroleptis* spp., *Phrynobatrachus accraensis* found for both the forest and teak plantations of the study area. Unlike the forest and teak plantations, the study found significant differences between the diversity of anurans in the cocoa farm during both the rainy and dry seasons (t=-2.6215, p=0.001). This could also be due to the presence of species such as *Bufo regularis, Bufo maculatus, Arthroleptis* spp., *Phrynobatrachus accraensis, Leptopelis hyloides* and *Hyperolius cf torrentis* which were recorded as the most dominant species in the cocoa farm in both rainy and dry seasons. It can therefore be deduced from the results that the diversity of amphibians changes with seasonal variations.

Common Anuran Species Shared Between Any Two Land Use Types

Species turn over analysis revealed that about 64% of species in the forest during the rainy season were found in the teak plantation and cocoa farm. On the contrary, all species recorded in the teak plantation in the rainy seasons were also found in the cocoa farm (100%). This percentage indicates no significant variation in the habitat of both the cocoa and the teak plantation. It also signifies that, the disturbance level in both areas may be the same.

During the rainy season, 99% of the species were found to share common habitat between the forest and teak plantation and also between teak plantation and cocoa farm. Between the forest and the cocoa farm, it was found that all the species shared same habitats (100%). The Sörensen indices of all the land use types are presented in Table 1.

Table 1: Sörensen Indices of Species

Combinations	**Rainy season**	**Dry season**
Forest and Plantation	0.64	0.1
Plantation and Cocoa	0.0	0.1
Forest and Cocoa	0.64	0.0

Plate 1: ***Bufo maculatus:***

Very common and widespread African toad, that inhabits all habitat types from degraded forests to moist savannas. Has a flat back with dorsum tan with darker blotches and a light patch just below the eye. Has light mid-dorsal stripe present.

Plate 2: ***Bufo regularis:***

Very common West and Central African toad that is especially abundant around human settlements. Large and compact toad with warty skin. Has numerous dark patches on the dorsal parts of the extremities and eyelids.

Plate 3: *Arthroleptis* spp.

Very common frog in degraded forest. Has a bright eyelid. Well-developed single subarticular tubercles on the digits. Have distinct tympanum with expended digit tips.

Plate 4: *Phrynobatrachus accraensis*

New DNA analyses proved this species to be conspecific with *Phrynobatrachus latifrons,* and also with the Volta "endemic" *Phrynobatrachus latifrons togolensis.* Has two pairs of comma shaped scapular glands with no ridges. Has more developed webbings.

Plate 5: *Phrynobatrachus alleni:*

Has distinct ridges. They are shorter. These species are near threatened.

Plate 6: *Phrynobatrachus calcaratus*

Widespread West and central African forest frog. Presence of an eyelid cornicle. Has face mask with a warty skin.

Plate 7: ***Phrynobatrachus plicatus***

Widespread West African forest frog. Snout is more rounded in lateral and more pointed in dorsal view. Has a distinct long and X- shaped dorsal ridges that clearly exceed the middle of the back. Less broad head and more developed webbing.

Plate 8: ***Leptopelis viridis***

Widespread West African savanna frog. Brown with darker patterns. Has an N- shaped dorsal marking. Small dorsum with distinct tympanum.

Plate 9: *Ptychadena oxyrhychus*

Widespread and common African savanna frog. Has pointed snout with very long legs. Has a short longitudinal ridge, the two median ones emerge on a level with the posterior border of the eyes. Has clearly visible tympanum.

Plate 10: *Ptychadena aequiplicata*

Widespread but never abundant West and Central African forest frog. Has a pointed snout with distinct tympanum. Dorsum is olive -brown with small dark spots. Head is moderate.

Plate 11: ***Ptychadena bibroni***

Very common West African inhabitant of degraded forests and moist savannas. Has four dorsal ridges which usually run from the level of the posterior eye border to the end of the body. Gray to pale brown dorsum with numerous tiny dark brown to black patches.

Plate 12: ***Amnirana albolabris***

Has a triangular head. Snout is slightly rounded with prominent eyes. Has a circular tympanum, base of toe much swollen. Has white stripes extending from the extremity to the snout.

Plate 13: ***Leptopelis hyloides***

Widespread West African forest species**.** Dorsal skin is finely areolate and coarsely granulates on the venter. More webbing and a uniform or dark symmetrical spots of streaks with a triangular or T- shaped marking between the eyes.

Plate 14: ***Hylarana albolabris***

It is highly coloured with no pigmentation of eyelids. It also has horizontal irises.

Plate 15: ***Hyperolius cf vindigulosus***

It has a smooth skin, moderate foot webbing and presence of hand webbing.

Abundance, Relative Density and Conservation Status of Anurans in both Rainy Season and Dry seasons

Tables 2 and 3 provide results on species abundance, relative density, and the conservation status of each anuran encountered in both rainy and dry seasons. The conservation status was determined by identifying the status of each individual species according to the International Union of Conservation of Nature Amphibian Assessment Report (Red List) (2006). The result showed that during the rainy season surveys the relative densities of species such as *Arthroleptis* spp and *Phrynobatrachus calcaratus* were relatively higher in the cocoa farms than the forest and teak plantations. However, the relative density of *Bufo maculatus* and *Bufo regularis* appeared to be the same in all the land use types despite variations in their relative abundance (Table 2). Additionally, the relative density of species such as *Hylarana albolabris*, *Hyperolius cf torrentis* and *Hyeperolius cf vindigulosus* were marginally high in the forest as compared with the relative density of same species recorded in both teak plantations and cocoa farm.

With regard to the conservation status of species found during the rainy seasons, it was found that 81.25% of all species identified were categorized as "Least Concern", 12.5% were listed as Vulnerable and therefore should be under strict conservation while conservation status of the remaining 6.25% has not yet been categorized. Even though the conservation status of most anuran species recorded in the rainy season was mostly Least Concern, uncontrolled human actions may result in their population decline which ultimately can change their conservation status.

In the dry season also, it was found that species such as *Phrynobatrachus calcaratus*, *Athroleptis* spp and *Hyperolius cf torrentis* were higher in the cocoa farm than the other areas as shown in Table 3. The conservation status of species found during the dry season survey also revealed about 90% of total species found to be of Least Concern species according to the IUCN red list except for one species; *Ptychadena aequiplicata* which was found to be highly Vulnerable yet found in the highly disturbed cocoa farm and the teak plantation respectively.

Table 2: Abundance, Relative Density and Conservation of Anurans in the Rainy Season

	FOREST		PLANTATION		COCOA		
Species name	Abundance	RD	Abundance	RD	Abundance	RD	Conservation Status(IUCN)
Arthroleptis spp.	45	0.1	17	0.1	65	0.2	Least Concern
Bufo maculatus	24	0.1	12	0.1	18	01	Least Concern
Bufo regularis	25	0.1	26	0.1	15	0.1	Least Concern
Hyperolius cf torrentis	20	0.1	8	0.0	3	0.0	Least Concern
Hyperolius cf. vindigulosus	22	0.1	1	0.0	1	0.0	Vulnerable
Leptopelis hyloides	19	0.0	5	0.0	6	0.0	Least Concern
Leptopelis viridis	23	0.1	5	0.0	6	0.0	Least Concern
Phrynobatrachus alleni	19	0.0	21	0.1	6	0.0	Near Threatened
Phrynobatrachus accraensis	19	0.0	20	0.1	8	0.0	Least Concern

Phrynobatrachus plicatus	14	0.0	10	0.1	9	0.0	Least Concern
Phrynobatrachus calcaratus	60	0.2	19	0.1	77	0.3	Least Concern
Ptychadena bibroni	21	0.1	3	0.0	1	0.0	Least concern
Amnirana galamensis	6	0.0	0	0.0	0	0.0	Status not certain
Hylarana albolabris	20	0.1	6	0.0	5	0.0	Least Concern
Ptychadena oxyrhynchus	33	0.1	11	0.1	4	0.0	Least Concern
Ptychadena aequiplicata	18	0.0	7	0.0	3	0.0	Vulnerable

Table 3: Abundance, Relative Density and Conservation of Anurans in the Dry Season

	FOREST		PLANTATION		COCOA		
Species name	Abundance	RD	Abundance	RD	Abundance	RD	Conservation Status (IUCN)
Arthroleptis spp.	19	0.1	17	0.2	51	0.3	Least Concern
Bufo maculatus	16	0.1	4	0.0	7	0.0	Least Concern
Bufo regularis	14	0.1	21	0.2	6	0.0	Least Concern
Hyperolius cf torrentis	8	0.1	2	0.0	22	0.1	Least Concern
Leptopelis hyloides	3	0.0	8	0.1	32	0.2	Least Concern
Phrynobatrachus accraensis	12	0.1	6	0.1	7	0.0	Least Concern
Phrynobatrachus calcaratus	35	0.3	19	0.2	54	0.3	Least Concern
Ptychadena bibroni	21	0.2	3	0.0	1	0.0	Least Concern
Hylarana albolabris	2	0.0	8	0.1	0	0.0	Least Concern
Ptychadena aequiplicata	0	0.0	2	0.0	1	0.0	Vulnerable

CHAPTER FIVE

DISCUSSION

Diversity, Density and Distribution of Anurans

This survey probably represents the most current description of the natural diversity, density and distribution of anurans in the Mpameso Forest Reserve since there are no known available data relating to work on anurans in the study area. The results indicate that conservation efforts are needed to protect anurans in the Mpameso Forest Reserve and its surrounding cocoa and teak plantations since the area is less diverse in anuran species as compared to results of similar studies which have been reported in Ghana and other parts of West Africa (Monney et al., 2011). Conservation efforts could include increasing tree cover or forest reclamation and preservation of water bodies where they can breed. Conservation efforts may also target protection of the food web by reducing pesticide use especially in land use types such as cocoa farms.

The differences in anuran numbers observed among the land use types may have resulted from the large habitat differences. In particular, canopy cover varies greatly among the land use types; from a dense or closed canopy in the forest, to a relatively sparse canopy in cocoa farm, to teak plantation which has a sparser and lower canopy. These canopy cover differences in turn cause habitat differences among the forest, teak plantation and cocoa. Within the forest, the dense canopy results in relatively low atmospheric and water temperatures, reduced growth of algal and herbaceous vegetation and low dissolved oxygen concentrations from the decomposition of leaf litter (Battle and Golladay, 2001) which may favour the forest–dependent species of anurans such as *Phrynobatrachus alleni* and *Phrynobatrachus plicatus*. Moreover, the recording of the disturbance- tolerant species such as *Bufo maculatus* in all three land use types may be an indication of their resilience to microhabitat changes. The greater number of individuals of anurans in the forest (518) during the survey may also be attributed to the fact that the anurans may have concentrated in the forest to avoid extreme or increased exposure to extreme weather conditions as a result of human land clearing activities in the cocoa (408) and 261 for teak plantations (Mitchell *et al.*, 1997). In the present study, all the species listed were found in all the three land use types except for one, *Amnirana galamensis,* which was recorded only in the forest but not in the teak plantation or cocoa farm. This may suggest that *Amnirana galamensis* may be a forest species.

Little or virtually nothing is known about the effects of pesticides or other agrochemicals, climate change or invasive species on amphibians in the study area, even though there is uncontrolled use of agrochemicals especially on cocoa farms. However, the results in the present study may suggest that in some ways such chemicals may have little or no effect on the anuran species, considering that the number of species of anurans in the forest during the rainy season was16 as compared to cocoa and teak plantation which recorded 15 species each. Again in the dry season, the teak plantation recorded 10 species, while the forest and cocoa recorded nine species. The significantly higher number of individuals of a species among the three land use types suggests that more food may be available in the forest compared to cocoa and teak plantation since pesticide use may negatively impact on the food webs in the cocoa and teak plantations that these anurans depend on. The number of individuals in a species observed in the cocoa farm (408) compared to the teak plantation (261) could have resulted from the cocoa farm providing a buffer habitat that would promote faunal population stability in the optimal forest habitat and may also provide corridors of acceptable habitat for animals dispersing between small forest patches (Rice and Greenberg, 2000). Though agricultural or forestry practices can negatively affect local habitat quality for selected herpetofaunal species, at least in the short-term (deMaynadier and Hunter, 1995), or for a longer term (Petranka *et al.*, 1993) the results of this study confirm the level of diversity in the anurans as reported in earlier studies.

This study revealed that forest habitats are more favourable than cocoa and teak plantation for anurans in Mpameso Forest Reserve. Again, the rainy season recorded more species than the dry season. It is plausible that some of these species may aestivate or move out to find more suitable habitats during the dry season hence the fewer number of species recorded. It can therefore be deduced that habitat type and season are major factors responsible for the distribution and diversity of anurans in the study area. In the dry season, the cocoa farm recorded the highest number of anurans which may be attributed to the fact that some amphibians could still survive in even heavily disturbed areas and at harsh environmental conditions.

Sixteen species of anurans were reported in the present study in the Mpameso Forest Reserve in the Brong-Ahafo Region. An earlier study by Hughes (1988) nationwide recorded 71 amphibian species including those listed in the present study. Even though some of the species identified by Hughes (1988) were reported to be of uncertain taxonomic status by Muller

(1985) and Rödel and Bangoura, (2004) or not found in Ghana by Barbour and Loveridge, 1927. All the species identified in the present study are among the list of known amphibian species in Ghana. With a record of 16 species, the Mpameso Forest Reserve seems to be less diverse in anuran species compared to the 31 species recorded in the Togo Hills and the 20 species recorded in the periphery of the Kyabobo National Park (Rödel and Agyei, 2003). However, the Mpameso Forest Reserve may be more diverse in anuran species than the Kakum National Park from where 12 species were recorded by Monney *et al.* (2011). Data from the current study also list forest dependent species such as *Phrynobatrachus alleni* and *Phrynobatrachus plicatus,* and disturbance-tolerant species such as *Bufo maculatus*, *Bufo regularis* and *Phrynobatrachus calcaratus* which were also reported by Adum, *et al.* (2013).

This study emphasizes the importance of land use types in supporting amphibian diversity and distribution. If future habitat conservation efforts in this area are to succeed, they must recognize the important roles that all land use types play in supporting biological diversity.

Comparison of Disturbed Land Use Area and Undisturbed Forests

Several studies have examined how changes in land use through logging and conversion into tree plantations may influence native herpetofauna (Lieberman, 1986; Heinen, 1992; Aukland *et al.,* 1997). Miyata (1980) found increased herpetofaunal densities in cocoa and rubber plantations in Ecuador relative to primary forest, and Lieberman (1986) and Heinen (1992) found a similar pattern between primary forest and cocoa plantations in Costa Rica. Heinen (1992) further suggested that the abundance of anurans in recently disturbed sites decreases with time. In contrast to density trends, Heinen (1992) found that undisturbed forests in Costa Rica had higher diversity of species and greater equitability in protected areas than older cocoa plantations. This result from the Mpameso forest area also showed a similar trend where the number of species recorded in the forest was relatively higher than cocoa and teak plantations. This shows the resilience of forest areas in providing suitable micro-climatic condition for herpetofaunal species.

Aukland *et al.* (1997) found a different situation in their studies in western Uganda, with higher litter frog densities and lower diversity in unlogged forest than forest that had been selectively logged in the last five years. The selectively logged forest studied at Kibale was logged nearly 30 years ago, and the tree plantations are of a similar age with a dense, predominately native flora regenerating underneath the pines. The results from Mpameso indicate that herpetofaunal communities may respond differently to different forest management strategies and therefore can affect the population of amphibians at a particular point in time.

Teak as an exotic species may also have negative impacts on wildlife species even though it is regarded as a fast management mechanism for forest regeneration. Some studies (Lugo, 1992; Zanne, 1997) suggest that exotic tree plantations can be used to facilitate forest regeneration in cases where natural succession is very slow or arrested. However, the ability of exotic tree plantations to aid in the restoration of native herpetofaunal communities is unclear.

CHAPTER SIX

SUMMARY, CONCLUSION AND RECOMMENDATIONS

Summary and Conclusions

The results revealed variations in the distribution of anurans among the three land use areas. A total of 786 individual anurans comprising of 16 species belonging to six families were encountered during the rainy season. The highest number of individuals was encountered in the forest area (388); individual anurans encountered were greater in the forest than the cocoa and plantation areas. Again, in the dry season, 401 anurans comprising 10 species and belonging to six families were encountered. Unlike the anurans encountered during the rainy season, it was found that in the dry season, anuran distribution was higher in the cocoa growing areas as compared to forest and plantation areas respectively.

The distribution of anurans in the rainy season shows that *Arthroleptis* spp. (11.6%), *Phrynobatrachus calcaratus* (15.5%) and *Ptychadena oxyrhynchus* (8.5%) were the dominant species in the forest. This was not the same in the teak plantation where species such as *Phrynobatrachus alleni (*12%), *Bufo regularis (*14.9%), *Bufo maculatus (*6.9%), *Arthroleptis* spp. (9.7%), *Phrynobatrachus accraensis* (11.4%), *Phrynobatrachus calcaratus (*10.9%) were identified as the dominant species. Similarly, in the cocoa farm, species such as *Phrynobatrachus calcaratus (*27.8%), *Arthroleptis* spp. (23.5%), *Bufo regularis (*5.4%), *Bufo maculatus* (6.5%) *and Phrynobatrachus plicatus* (3.5%) were found to be relatively high in abundance as compared to the other land use areas.

The distribution of anurans in the dry season indicates that *Phrynobatrachus calcaratus* (26.9%), *Arthroleptis* spp (14.6%),*Bufo maculatus* (12.3%), *Bufo regularis* (10.8%), *Phrynobatrachus accraensis* (9.2%), *Ptychadena bibroni* (16.2%), *and Hyperolius cf. torrentis* (6.2%) were high in the forest. It was also revealed by the study that in the teak plantation, *Bufo regularis* was recorded as the most dominant species (23.3%), *Arthroleptis* spp (18.9%), *Phrynobatrachus calcaratus* (21.8%),whereas *Phrynobatrachus accraensis* (6.7%), *Leptopelis hyloides* (8.9%)and *Hylarana albolabris* (8.9%) were moderately dominant in the teak plantation. In the cocoa farm, species found in the forest andteak plantations such as *Arthroleptis* spp (28.2%), *Phrynobatrachus calcaratus* (29.8%), *Leptopelis hyloides* (17.7%) and *Hyperolius cf. torrentis* (12.2%) were still found to be dominant despite human disturbance.

The diversity of anurans in the three land use types during the rainy season showed significant difference in amphibians diversity between the forest and the teak plantation (t=3.5301, p=0.0004), in the same way a significant difference was found between teak plantation and cocoa farm (t=6.0467, p=3.6705E^{-9}) as well as between forest and cocoa farm (t=-9.0984, p=1.7706E^{-17}). Similarly in the dry season, there was a difference in diversity of anurans in the forest and teak plantation (t=-0.28057, p=0.77938E^{-17}), between teak plantation and cocoa farm (t=-3.3223, p=0.001) and also between forest and the cocoa farm (t=-3.3223, P=0.0001). The difference in amphibian species in the three land use types showed that human activities can have negative impacts on the micro-habitat of amphibians which subsequently affects their distribution.

Recommendations

Based on the findings of the study, the following were recommended for further studies

i. Research on pesticide use and the effects of constant pesticide application on amphibian species richness and distribution in cocoa growing areas.
ii. Further studies should be conducted on specific species to identify the habitat requirements of individual species in all the land use areas.
iii. A study should be conducted to identify plantation plant species that provide environmental conditions suitable for amphibians
iv. Further studies should also involve both nocturnal and diurnal surveys to compare species richness and distribution to determine the actual changes in amphibian species distribution.
v. Studies focusing on comparing environmental variables such as soil chemistry, depth of litter, humidity, and temperature and their influence on amphibian species richness and distribution in different land use types. Complete description of these variables would assist in developing conservation strategies for the various land use types.

Conservation efforts should consider the status of anurans encountered and also take into consideration the status of amphibians according to the IUCN Red list.

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