PROPOSITIONS

1. Food security in the future requires holistic management strategies now, with consideration of trade-offs and co-benefits of land use change.
   (This thesis)

2. Accounting for ecosystem services is fraught with challenges but essential for sustainable management.
   (This thesis)

3. Research does not only seek answers to questions but also questions the answers already out there.

4. For a given dataset, maps can tell very different stories.

5. A PhD is not the end but the start of a journey.

6. As writing a scientific paper can be never-ending, knowing when to stop is a must-have skill.

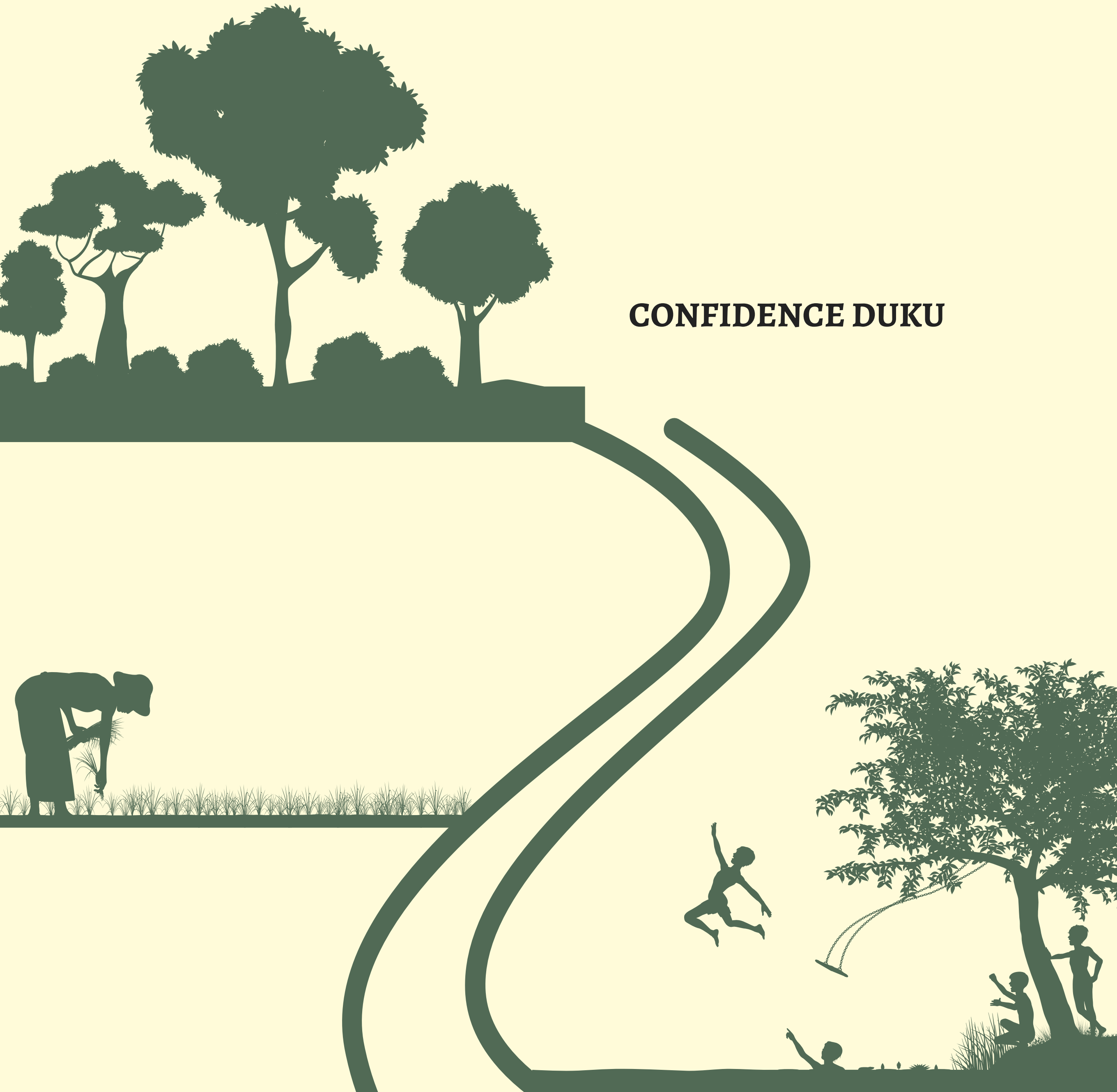
Propositions belonging to the thesis entitled:
“Modelling the dynamic interactions between food production and ecosystem services: a case study in Benin”

Confidence Duku
Wageningen, 5 July 2017

MODELLING THE DYNAMIC INTERACTIONS BETWEEN FOOD PRODUCTION AND ECOSYSTEM SERVICES:

A CASE STUDY IN BENIN

CONFIDENCE DUKU



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MODELLING THE DYNAMIC INTERACTIONS BETWEEN FOOD PRODUCTION AND ECOSYSTEM SERVICES:

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To my beloved parents, Kweku and Hannah
thank you
medaase

CONTENTS

CHAPTER 1

GENERAL INTRODUCTION

1.1 BACKGROUND

FOOD SECURITY IN SUB-SAHARAN AFRICA

There is urgent need to improve food security in sub-Saharan Africa (SSA). The Food and Agriculture Organization (FAO) defines food security to "exist when all people, at all times, have physical, social and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life" (*FAO*, 1996; *FAO et al.*, 2015). In this respect are three central dimensions of food security, i.e. food availability, food access and food utilization. Food availability, which is a consequence of the level of food production, underpins the other dimensions. In SSA, prevalence of undernourishment (i.e. the proportion of the population experiencing chronic hunger), a key index of food insecurity, has been declining over the past twenty-five years (*FAO et al.*, 2015). Eighteen countries in SSA achieved the Millennium Development Goal target of halving prevalence of undernourishment between 1990 and 2015; and seven countries brought prevalence of undernourishment under 5% (*FAO et al.*, 2015). Nevertheless, food insecurity still remains high and one in every four persons (i.e. 220 million people) in this region is undernourished compared to one in eight persons in Asia and one in eighteen in Latin America and the Caribbean (*FAO et al.*, 2015). Food insecurity in SSA is driven by a variety of biophysical, socio-political, and economic factors that contribute to insufficient national food availability and insufficient access to food by households and individuals. These factors include amongst others drought and extreme weather events; biotic stresses; poor agricultural sector development; war and civil strife; population growth; poverty; macroeconomic instability (*Smith et al.*, 2000; *FAO et al.*, 2015). At the household level, food insecurity is a major cause of malnutrition and undernutrition in SSA (*FAO et al.*, 2015). Pregnant women suffering from malnutrition often give birth to underweight babies who are 20% more likely to die before the age of five (*UNICEF.*, 2007). A third of all deaths in children under the age of five in developing countries are linked to undernutrition (*UNICEF.*, 2007).

Agriculture plays a key role towards ensuring food security in SSA by providing: 1) food availability; 2) an important source of income to purchase food; and 3) foods with high nutritional status (*World Bank*, 2008b). Agriculture accounts for 65% of the labour force in Africa, between 25 and 30% of Gross Domestic Product and over half of total export earnings (*IFPRI*, 2004; *World Bank*, 2008a). Currently, much of the effort at improving food security in SSA has been in boosting domestic food production in line with the Sustainable Development Goal 2 i.e. end hunger and achieve food security, in which SSA countries pledge to fully meet domestic food demand by 2030 (*UN*, 2015a). Increasing crop production, especially, can be achieved through four ways: 1) expansion of land under cultivation; 2) intensification on existing farmland by growing two or more crops a year; 3) increasing farmer yields in existing farmland through better nutrient and water management; and 4) raising the yield ceiling by introducing higher-yielding cultivars (*Laborte et al.*, 2012). Between 1961 and 2003, agricultural production in SSA grew at an average rate of about 2.4% per annum (*Kidane et al.*, 2006) of which higher yields, increased cropping intensity (i.e. sequential cropping and intercropping) and cropland expansion accounted for an estimated 38%, 31% and 31%, respectively (*Alexandratos and Bruinsma*, 2012). Despite the improvements, the performance of agriculture in SSA has not kept pace with the rest of the world and productivity remains the lowest in the world (*Kidane et al.*, 2006).

Rapid population growth in SSA has also undercut efforts at improving food security. Whereas food production in SSA grew at an estimated rate of 2.4% per annum between 1961 and 2003, population grew at a higher rate of 2.8% (*Kidane et al.*, 2006). Over this period, production per capita in SSA actually decreased at an average rate of about 0.4% per annum. Hence food imports have increased. In recent decades, cereal import has been increasing and accounted for about 25% of total consumption in SSA in the early 2000s (*Kidane et al.*, 2006; *de Graaff et al.*, 2011). As a consequence, even though the prevalence of undernourishment declined during this period, the actual number of undernourished people has increased (*FAO et al.*, 2015). By 2050, the population of Africa is projected to more than double compared to 2015 estimates i.e. an increase of 1.3 billion people, which represents more than half of projected global population growth between 2015 and 2050 (*UN*, 2015b). As a result, it is projected that current productivity of major food crops will be inadequate to meet food demand by 2050 (*Alexandratos and Bruinsma*, 2012). Increasing productivity on existing predominantly rainfed croplands is also likely to be insufficient to meet projected demand by 2050. For example, the cereal demand for 10 countries in SSA is projected to triple by 2050 compared to 2010 estimates and even without taking into consideration climate change, a recent study showed that it will not be feasible to meet 2050 SSA cereal demand on existing production cropland by yield gap closure alone (*van Ittersum et al.*, 2016).

Climate change also poses a major challenge to food security in SSA. Climate change will affect all three key dimensions of food security (*Schmidhuber and Tubiello*, 2007; *Wheeler and von Braun*, 2013). Temperatures across Africa are projected to increase in all seasons and to rise faster than the global average increase during this century (*Christensen et al.*, 2007; *Joshi et al.*, 2011; *Sanderson et al.*, 2011; *Niang et al.*, 2014). Precipitation projections across SSA are more uncertain than temperature projections (*Christensen et al.*, 2007; *Niang et al.*, 2014), nonetheless climate change is expected to cause changes in rainfall intensity or delay in the onset of rainy season in large areas of SSA (*Thomas et al.*, 2007; *Songok et al.*, 2011; *Sarr*, 2012; *World Bank*, 2013; *Niang et al.*, 2014). Climate change is also likely to increase the incidence of extreme events such as droughts and floods (*New et al.*, 2006; *World Bank*, 2013; *Niang et al.*, 2014). Given that about 97% of cropland area in SSA is used for rainfed cultivation (*You et al.*, 2014), changes in precipitation and temperature associated with climate change will pose serious risks to crop production systems and food security in general. The expected impacts on crop production include: a shortening of the length of growing period (*Jones and Thornton*, 2009; *Thornton et al.*, 2011); reductions in the area suitable for agriculture (*IIASA/FAO*, 2012); increase in epidemics of agricultural pests and diseases (*Schmidhuber and Tubiello*, 2007; *Duku et al.*, 2016a); and reductions in crop yields in many regions of SSA (*Muller et al.*, 2011; *Wheeler and von Braun*, 2013; *World Bank*, 2013; *Connolly-Boutin and Smit*, 2016). Climate change will also affect food access and utilization (*Schmidhuber and Tubiello*, 2007; *Wheeler and von Braun*, 2013). Access to food is dependent on household and individual-level income (*Wheeler and von Braun*, 2013). Water, sanitation and health are key determinants of food utilization and are likely to be affected by climate change as well (*Wheeler and von Braun*, 2013). For example, climate change will cause alterations in certain disease vectors causing changes in the spatial and temporal transmission of infectious diseases especially malaria (*Hay et al.*, 2002).

RELATIONSHIP BETWEEN ECOSYSTEM SERVICES AND FOOD SECURITY

In recent decades, ecosystems have been increasingly recognised and analysed in terms of the services they provide to society (*Daily et al.*, 1997; *MA*, 2005; *TEEB*, 2010). Ecosystem services are broadly defined as the contributions of ecosystems to human well-being (*Daily et al.*, 1997; *TEEB*, 2010; *UN et al.*, 2014a) and this concept is increasingly being used to analyse human-environment interactions. Ecosystem services are generated by a complex interplay of natural cycles involving the interaction of biotic and abiotic processes that operate across a wide range of spatial and temporal scales (*Daily et al.*, 1997; *Hein et al.*, 2006). These services include: 1) provisioning services i.e. material or energy outputs (such as food, water, raw materials) for economic and human activities; 2) regulating services i.e. regulating of earth surface processes, including biochemical, climatic and ecohydrological processes; and 3) cultural services i.e. intellectual and symbolic benefits that people obtain from ecosystems through recreation, knowledge development, relaxation, and spiritual reflection. There has been detailed work on definition of this concept (*MA*, 2005; *TEEB*, 2010; *Haines-Young and Potschin*, 2013; *UN et al.*, 2014a); indicators (*Haines-Young et al.*, 2012; *van Oudenhoven et al.*, 2012); assessment (*MA*, 2005; *Bateman et al.*, 2011; *Bateman et al.*, 2013); operationalization and incorporation into standardized accounting frameworks (*Boyd and Banzhaf*, 2007; *UN et al.*, 2014a; *Hein et al.*, 2015; *Remme et al.*, 2015; *Hein et al.*, 2016). Work is still in progress concerning enhancing accuracy and resolution of ecosystem service modelling and linking to benefits and beneficiaries.

Ecosystems and the services they provide are intricately linked to food security outcomes. Such linkages are complex and multi-directional. Food security is not only dependent on ecosystem services, but pathways to ensuring food security are also some of the greatest drivers of ecosystem degradation and loss of ecosystem services (*Foley et al.*, 2005; *Poppy et al.*, 2014). Ecosystems directly and indirectly support the three key dimensions of food security through the provision of critical ecosystem services that facilitate agricultural production, create income-generating opportunities, and provide energy for cooking (Figure 1.1) (*Richardson*, 2010; *Poppy et al.*, 2014). In terms of food availability, food production depends on the provision of ecosystem services. For example, water availability for irrigation, especially during the critical dry season in the tropics is dependent on the water flow regulation services provided by forests (*Bruijnzeel*, 2004). Globally, agriculture accounts for about 70% of water use (*Alexandratos and Bruinsma*, 2012). In SSA, *Alexandratos and Bruinsma* (2012) estimates that irrigation water withdrawal is likely to increase to 133 km^3 per annum by 2050. Pollination and biological control of pest insects are other important services to crop production provided by ecosystems. Natural and semi-natural ecosystems provide the habitat and diverse food resources required by animal pollinators and natural enemies to agricultural pests (*Tscharntke et al.*, 2005; *Power*, 2010). Globally, approximately 75% of crop species significant for food production rely on animal pollination, primarily by insects (*Klein et al.*, 2007). Ecosystems by providing grazing opportunities also contribute to food availability through the production of livestock for meat and dairy consumption (*Remme et al.*, 2014). There is also the direct consumption of wild foods produced by natural ecosystems for supplementing diets, especially in rural areas of SSA (*McGarry and Shackleton*, 2009; *Bharucha and Pretty*, 2010).

The linkages between ecosystem services, and food access and utilization may not be as evident as that with food availability. Nevertheless, ecosystem services play a vital role in supporting household-level food access and utilization in SSA (*Richardson*, 2010; *Poppy et al.*, 2014). Many households in rural SSA engage in harvesting wood and Non-Timber Forest Products (NTFP) from natural forests and woodlands to sustain household livelihoods and incomes during non-agricultural seasons (*Pimentel et al.*, 1997; *Shackleton and Shackleton*, 2004). *Barrett et al.* (2001) showed that such non-farm income generating activities are positively correlated to household welfare indicators across most of rural Africa. Furthermore, fuelwood harvested from forests and woodlands provide cooking fuel for food utilization in large areas of rural SSA (*WFP*, 2012).

Despite their importance to livelihoods, ecosystems, in particular forests and woodlands, in SSA continue to rapidly decline (*FAO*, 2015b). With 17% of the world's forests, Africa accounts for over half of global deforestation (*Katerere et al.*, 2009). Cropland expansion is the major driver of deforestation in SSA. Between 1990 and 2015, about 90 million ha of natural forests was lost in Africa, largely as a result of cropland expansion (*FAO*, 2015b; *Sloan and Sayer*, 2015). This represents a loss of about 13% of forest cover. Of the about 90 million ha, about 16 million ha was lost between 2010 and 2015 alone. Forest loss as a result of cropland expansion is likely to continue in the coming decades. For example, *Hilderink et al.* (2012) projects a 29% reduction in forest cover in SSA by 2030 as a result of cropland expansion. Croplands are mainly managed to optimize food production at the expense of other ecosystem services yet these benefits depend upon ecosystem services from the wider landscape and the environment for their long-term provision and sustainability (*Foley et al.*, 2005; *Poppy et al.*, 2014). Hence, a key question is what will be the implication of changes in natural and semi-natural ecosystems to crop production and consequently food security.

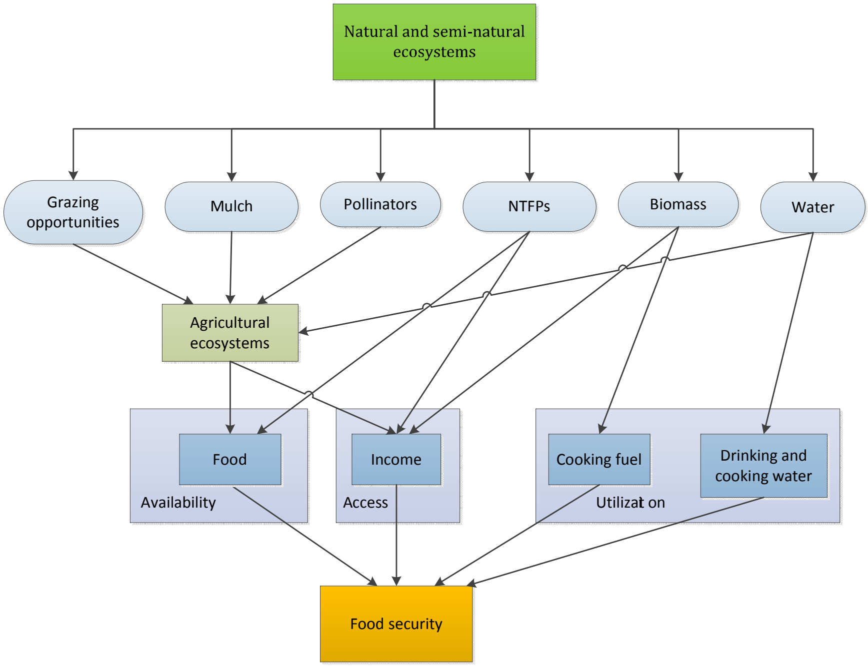


Figure 1.1. Schematic diagram showing the synergies and different routes by which ecosystems contribute to the different dimensions of food security. Adapted from Poppy et al. (2014). NTFP is Non-Timber Forest Products.

1.2 PROBLEM STATEMENT

Given the high levels of food insecurity and the loss of vital ecosystem services associated with deforestation, countries in SSA face a major dilemma. How can they produce enough food in a changing climate to feed an increasing population while protecting natural forests and woodlands that provide a wide array of ecosystem services beneficial to livelihoods? In terms of policy initiatives, this dilemma involves reconciling pledges under Sustainable Development Goal (SDG) 2 i.e. end hunger and achieve food security, with that under SDG 15 i.e. protect, restore and promote sustainable use of terrestrial ecosystems (*UN*, 2015a). To this end, agricultural intensification as a food production pathway is widely considered desirable (*Godfray et al.*, 2010; *Foley et al.*, 2011; *Pretty et al.*, 2011; *Garnett et al.*, 2013; *Hall and Richards*, 2013). In response African countries are investing in or committing to invest in agricultural intensification. For example, one of the targets in the Africa Water Vision 2025, adopted by the African Union member states, is to double the size of irrigated areas in this region by 2025

(*UNECA et al.*, 2003). In addition, *Morris et al.* (2007) calls for an increase and expansion of fertilizer application in Africa in order to achieve the required levels of productivity growth.

Yet, there remain uncertainties regarding the impact of climate change on opportunities for agricultural intensification in SSA. Furthermore, given the rising food demand associated with rapid population growth, the degree of success of agricultural intensification options in meeting this demand needs to be evaluated. The degree of success of agricultural intensification options will be one of the major determining factors of the level of pressure that natural ecosystems are likely to face (*Grassini et al.*, 2013; *van Wart et al.*, 2013a). Trade-offs between competing food security and broader environmental objectives are thus likely. Quantifying and understanding the dynamics and the nature of these trade-offs is important for prioritizing and targeting management interventions. These issues need to be addressed at national and local levels in the unique biophysical and developmental context. In addition, in recent times, much of the research focus on the impact of climate change on agricultural intensification options has been on crop yield responses (e.g. *Muller et al.*, 2011; *IIASA/FAO*, 2012; *Wheeler and von Braun*, 2013; *World Bank*, 2013; *Connolly-Boutin and Smit*, 2016; *Palazzo et al.*, 2017). However, failure to consider the impact of climate change on other intensification options may lead to underestimation of the vulnerability of SSA agriculture as well as misguided policy initiatives. Even without taking into account climate change, *van Ittersum et al.* (2016) showed that in addition to yield gap closure, growing two or more crops per year and irrigation expansion will be needed to achieve cereal self-sufficiency without further cropland expansion in SSA.

Furthermore, there is increasing demand for integrated information that can link analytical and policy frameworks for environmental sustainability, human well-being and economic growth and development. In this regard, the Systems for Environmental-Economic Accounting Experimental Ecosystem Accounting (SEEA EEA) framework (*UN et al.*, 2014a) is being increasingly used. The SEEA EEA is an initiative by the statistical community in collaboration with research groups (*Hein et al.*, 2015). To make this framework operational, scientific innovations are needed, especially regarding biophysical modelling of ecosystem services. Several recent studies have demonstrated the applicability of ecosystem accounting for monitoring a wide range of services (*Remme et al.*, 2014; *Schroter et al.*, 2014). Yet, there is still limited experience in accounting for hydrological ecosystem services in both physical and monetary terms. Of increasing challenge is how to model hydrological ecosystem services with adequate spatiotemporal detail and accuracy at aggregated scales in line with the SEEA EEA and how to conceptualise and empirically distinguish between hydrological ecosystem services flow and capacity. Understanding hydrological ecosystem services is key to understanding the trade-offs between land conversion to croplands and ecosystem services supply from natural and semi-natural ecosystems.

1.3 OBJECTIVES AND RESEARCH QUESTIONS

Considering the dual challenge of food security and ecosystem protection, the objective of this thesis is twofold. First, to further enhance the understanding of the dynamic interactions between food production and forest and woodland conservation goals, with a case study in Benin. Second, to further enhance the understanding of how hydrological ecosystem services can be captured in an accounting framework. To achieve these objectives, four research questions are formulated.

RQ1. How can hydrological ecosystem services be spatially modelled in line with the ecosystem accounting framework?

The ecosystem accounting framework is increasingly being used to link ecosystems and ecosystem services to economic activities (*UN et al.*, 2014a). Biophysical modelling and accounting for ecosystem services requires the empirical distinction of the capacity of ecosystems to provide services and the flow of ecosystem services into human activities. Measuring these components involved in ecosystem service delivery is important for devising sustainable management options. This study demonstrates how a widely used spatially explicit, process-based ecohydrological model can be modified and used to model and map hydrological ecosystem services in line with the ecosystem accounting framework. This research question is addressed in chapter 2.

RQ2. How can the contributions of forests and woodlands to crop production be disentangled and quantified?

In many tropical regions of SSA, crop production decreases substantially in the dry season. Increased application of irrigation, especially in the dry season has often been identified as one of the main pathways to increase crop production. However, water flows, in particular during the dry season, often depend upon the water regulation services provided by forests and woodlands which are increasingly subject to land conversion as well as degradation from the over-exploitation of wood resources. In this part of the thesis, I undertake simulation experiments to quantify the contributions of forests and woodlands to availability of surface-water for irrigation at the watershed level in Benin taking into consideration household water requirements and environmental water requirements. This research question is addressed in Chapter 3.

RQ3. What are the likely impacts of climate change on opportunities for agricultural intensification?

Towards achieving the dual objective of food security and ecosystem protection, agricultural intensification is widely considered the most desirable (*Gornall et al.*, 2010; *Garnett et al.*, 2013). Yet, opportunities for intensification in the predominantly rainfed areas of SSA are threatened by climate change. In this part of the study, I analyse the impact of climate change on three options for agricultural intensification in Benin i.e. increasing yields in rainfed areas; growing two or more crops per year in rainfed areas; and water availability for irrigation expansion. This research question is addressed in chapters 4 and 5.

RQ4. How will climate change and population growth affect trade-offs between food security outcomes and forest and woodland conservation goals?

Given the likely impact of climate change on agricultural intensification options and projected rapid population growth in SSA, it is yet unclear if agricultural intensification options alone will be adequate to even maintain current levels of food availability. In this part of the thesis, I identify and quantify trade-offs between forest and woodland conservation goals in Benin and food availability of three staple crops under different combinations of climate change and population growth scenarios. This research question is addressed in chapter 5.

1.4 STUDY AREA

Two study areas, both in Benin are used to address the objectives of this thesis. The larger area covers the entire Ouémé river basin and southwestern parts of the country that lie outside the river basin. This study area covers 55,000 km^2. Second is the relatively small area, the upstream portion of Ouémé river basin henceforth referred to in this thesis as the Upper Ouémé watershed. This second study area covers an area of approximately 14,500 km^2. The Upper Ouémé watershed located in central Benin is used as a case-study to address research questions 1, 2, and 3. Results and outputs are then upscaled to the entire Ouémé river basin and southwestern parts of the country that lie outside the river basin in order to address research question 4.

The Ouémé river basin and the southwestern parts of the country that lie outside the river basin largely make up the sub-humid tropical area of Benin. The total population in this area is approximately 7 million i.e. 70% of the total population of the country (*Bright et al.*, 2011). Over the period 1991 - 2013, population in Benin increased at a rate of 4% per annum (*FAO*, 2016b). The proportion of the total population of the country that is undernourished is currently 7.5% compared to an average of 9.6% in West Africa (*FAO et al.*, 2015). The average dietary energy supply adequacy (between 2001 and 2011), i.e. the ratio of dietary energy supply and dietary energy requirements, was 114% (*FAO*, 2016b). The southern part of the river basin, which lies outside the Upper Ouémé watershed, is more densely populated Figure 1.2A. This part is relatively more urbanized and land use is dominated by large areas of small-scale rainfed agriculture and oil palm plantations. Most of the forest and woodland areas lie in the Upper Ouémé watershed, which covers the northern part of the Ouémé river basin. Land use in the northern and central part of the study area is dominated by a mosaic of woodland savannah and forest islands with a relatively smaller cropland area. The natural vegetation is a mosaic of savannah woodland and small forest islands. Forest and woodland cover make up approximately 55% (7,500 km^2) of the the Upper Ouémé watershed. The major forest types in this watershed are tropical forest with more than 75% canopy cover; dry forest with a canopy cover of between 50 and 75%; and whereas woodlands have a canopy cover of between 25 and 50% (*Giertz et al.*, 2005). There are also large state-owned protected forests covering an area of about 6,000 km2 mainly in the northern and central part of the larger Ouémé river basin. The irrigation sector is poorly developed and the lack of irrigation water during the dry season is a major problem for many farmers (*Giertz et al.*, 2012). Consequently, crop cultivation mainly takes place during the wet season. Due to land availability in the Upper Ouémé watershed, residents from other parts of the country and neighbouring countries migrate to this region, which has caused an expansion of agricultural areas and led to substantial deforestation (*Giertz et al.*, 2012).

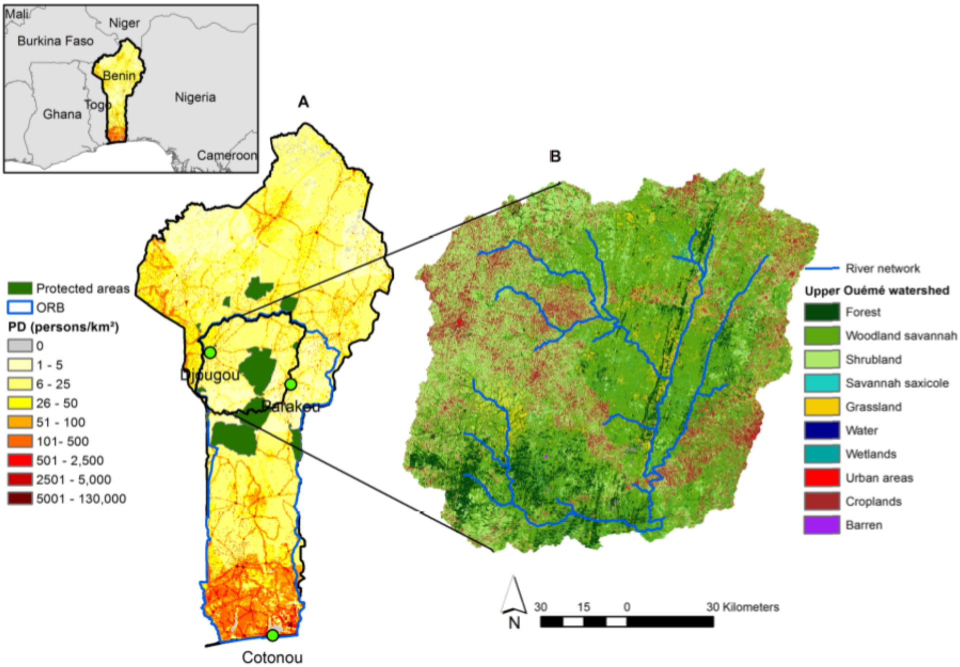


Figure 1.2. Location of the study areas showing: A) ORB is Ouémé river basin and the southwestern parts of Benin that lie outside the basin (ORB) It is the study area for chapter 5; and B) Land cover of Upper Ouémé watershed. It is the study area for chapters 2, 3,and 4. PD is population density.

1.5 THESIS OUTLINE

This thesis consists of six chapters. Chapters 1 and 6 have been written to provide a background, broad overview and synthesis of the research questions in order to address the main objectives of this thesis. Chapters 2 and 3 address research questions 1 and 2 respectively. Chapters 4 and sections of chapter 5 address research question 3, and chapter 5 also addresses research question 4. Chapters 2, 3, 4 and 5 have been written as independent research papers in cooperation with the listed co-authors.

In chapter 2, a spatially explicit ecohydrological model is configured, calibrated, and validated before it is used to model and map key hydrological ecosystem services in line with the ecosystem accounting framework. Two spatial indicators are selected to map and quantify the capacity and flow of each hydrological ecosystem service.

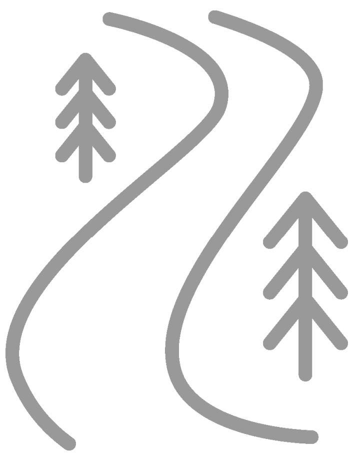
In chapter 3, simulation experiments involving hypothetical deforestation states and irrigation development scenarios are undertaken using the ecohydrological model. Hypothetical deforestation states involved systematic reduction of the forest and woodland extents of the watershed and irrigation development scenarios are defined based on different levels of sustained water flows in the river network. The simulation experiment was carried out using the calibrated and validated model in Chapter 2.

In chapter 4, I analyse opportunities for agricultural intensification under different climate change and socio-economic development trajectories. Croplands and potential croplands that can support the cultivation of two or more crops in a year are mapped and compared under different climate change scenarios. Water availability for irrigation is also quantified under a combination of climate change and socioeconomic development scenarios taking into consideration environmental water requirements and household water requirements.

In chapter 5, I quantify trade-offs between per capita food availability and protecting forests and woodlands at different levels of yield increases taking into account climate change, population growth and land quality of potential arable areas. I carry out these analyses for three major food crops, i.e. maize, cassava and yam, in Benin. The analytical approach combines soil-water balance and crop growth modelling under contrasting climate change and population growth scenarios, and three scenarios of cropland expansion with varying degrees of deforestation.

CHAPTER 2

TOWARDS ECOSYSTEM ACCOUNTING: A COMPREHENSIVE APPROACH TO MODELLING HYDROLOGICAL ECOSYSTEM SERVICES



ABSTRACT

Ecosystem accounting is an emerging field that aims to provide a consistent approach to analysing environment-economy interactions. One of the specific features of ecosystem accounting is the distinction between the capacity and the flow of ecosystem services. Ecohydrological modelling to support ecosystem accounting requires considering among others physical and mathematical representation of ecohydrological processes, spatial heterogeneity of the ecosystem, temporal resolution, and required model accuracy. This study examines how a spatially explicit ecohydrological model can be used to analyse multiple hydrological ecosystem services in line with the ecosystem accounting framework. We use the Upper Ouémé watershed in Benin as a test case to demonstrate our approach. The Soil and Water Assessment Tool (SWAT), which has been configured with a grid-based landscape discretization and further enhanced to simulate water flow across the discretized landscape units, is used to simulate the ecohydrology of the Upper Ouémé watershed. Indicators consistent with the ecosystem accounting framework are used to map and quantify the capacities and the flows of multiple hydrological ecosystem services based on the model outputs. Biophysical ecosystem accounts are subsequently set up based on the spatial estimates of hydrological ecosystem services. In addition, we conduct trend analysis statistical tests on biophysical ecosystem accounts to identify trends in changes in capacity of the watershed ecosystems to provide service flows. We show that the integration of hydrological ecosystem services in an ecosystem accounting framework provides relevant information on ecosystems and hydrological ecosystem services at appropriate scales suitable for decision-making.

This chapter is based on:
Duku, C., Rathjens, H., Zwart, S.J., Hein, L., 2015. Towards ecosystem accounting: a comprehensive approach to modelling multiple hydrological ecosystem services. *Hydrology and Earth System Sciences* 19, 4377-4396.

2.1 INTRODUCTION

Ecosystem accounting provides a systematic framework to link ecosystems to economic activities (*Boyd and Banzhaf*, 2007; *Maler et al.*, 2008; *Edens and Hein*, 2013; *Obst et al.*, 2013; *UN et al.*, 2014a). Specifically, ecosystem accounting aims to integrate the concept of ecosystem services in a national accounting context as described in *UN et al.* (2009). There is increasing interest in ecosystem accounting as a new, comprehensive tool for environmental monitoring and management (*Obst et al.*, 2013). The recently released System of Environmental-Economic Accounting (SEEA)-Experimental Ecosystem Accounting guideline (*UN et al.*, 2014a) provides guidelines for setting up both biophysical and monetary ecosystem accounts. Biophysical accounting for ecosystem services forms the basis for monetary accounting.

Ecosystem services are the contributions of ecosystems to human welfare (*TEEB*, 2010; *UN et al.*, 2014a). Hydrological ecosystem services, specifically, are the contributions to human benefits derived from ecosystems and are produced by the effects of terrestrial ecosystem components on freshwater as it moves through the landscape. Terrestrial ecosystem components directly modify various attributes (such as quantity, quality, location and timing) of various ecohydrological processes, resulting in augmentation or degradation of these processes (*Brauman et al.*, 2007). Factors such as the presence of beneficiaries (*Boyd and Banzhaf*, 2007), spatiotemporal accessibility (*Fisher et al.*, 2009), and management pressure (*Schroter et al.*, 2014) then determine if the ecohydrological processes constitute hydrological ecosystem services. Hydrological ecosystem services are diverse and can be broadly classified into five categories; improvement of extractive water supply, improvement of in-stream water supply, water damage mitigation, provision of water related cultural services, and water-associated supporting services (*Brauman et al.*, 2007). Production of these services underlies water and food security and the protection of human lives and properties.

Biophysical accounting for hydrological ecosystem services allows for the organisation and analysis of biophysical data on these services at different spatial and temporal scales suitable for the development, monitoring and evaluation of public policy (*UN et al.*, 2014a). Biophysical accounting also allows for the distinction between the flow of hydrological ecosystem services and the capacity of watershed ecosystems to provide service flows (*UN et al.*, 2014a). Service flow is the contribution in space and time of an ecosystem to either a utility function (e.g. private household) or a production function (e.g. crop production) that leads to a human benefit, whereas service capacity is a reflection of ecosystem condition and extent at a point in time, and the resulting potential to provide service flows (*Edens and Hein*, 2013; *UN et al.*, 2014a). For hydrological ecosystem services, high service capacity areas and high service flow areas may occur in different points or areas in space (*Fisher et al.*, 2009) making the need for their empirical distinction and separate spatial characterization crucial for land and watershed management.

Many approaches have been used for modelling, mapping and quantifying hydrological ecosystem services (e.g.*Le Maitre et al.*, 2007; *Naidoo et al.*, 2008; *Liquete et al.*, 2011; *Maes et al.*, 2012; *Notter et al.*, 2012; *Willaarts et al.*, 2012; *Leh et al.*, 2013; *Liu et al.*, 2013b; *Terrado et al.*, 2014 for an overview). For ecosystem accounting, however, key aspects requiring further research include the modelling of hydrological ecosystem services with adequate spatiotemporal detail and accuracy at aggregated scales, distinguishing between service capacity and service flow, and linking ecohydrological processes (and ecosystem components) to the supply of dependent hydrological ecosystem services. Addressing these issues requires the consideration of among others physical and mathematical representation of ecohydrological processes, spatial heterogeneity of ecosystems, temporal resolution, and required model accuracy (*Guswa et al.*, 2014). Adequate representation of spatial heterogeneity of the biophysical environment in ecohydrological models is crucial in ecosystem accounting because spatial units form the basic focus of measurement similar to functions of economic units in national accounting (*UN et al.*, 2014a). In addition, if ecosystem accounting is to provide reliable information for the assessment of integrated policy responses at the landscape level, then physical and mathematical representation of model processes should be based on scientific consensus (*Vigerstol and Aukema*, 2011). Furthermore, model results should be accurate and model uncertainties should be understood and reported (*Seppelt et al.*, 2011; *Martínez-Harms and Balvanera*, 2012). Finally, ecohydrological modelling for ecosystem accounting necessitates the use of continuous simulation watershed models that are able to capture short and long-term temporal variability in ecohydrological processes.

Our objective is to present a spatially explicit modelling approach aligned with an ecosystem accounting framework to map and quantify the capacities and the flows of multiple hydrological ecosystem services. We use the Soil Water and Assessment Tool (SWAT), which has been configured with a grid-based landscape discretization and further enhanced to simulate water flow across the discretized landscape units, to simulate the watershed ecohydrology. The model is calibrated and validated. Indicators consistent with the ecosystem accounting framework are used to map and quantify the capacities and the flows of multiple hydrological ecosystem services based on the model outputs. Biophysical ecosystem accounts are subsequently set up based on the spatial estimates of hydrological ecosystem services. We use the Upper Ouémé watershed in Benin as a test case to demonstrate our approach. This case-study area was selected because of a relatively high data availability (*Judex and Thamm*, 2008; *AMMA-CATCH*, 2014). It is also a microcosm of rural sub-Saharan Africa, where large sections of the population depend on smallholder rainfed agriculture for their livelihood, where groundwater is the major source of drinking water, and where rapid population growth and increasing land use change are prevalent. The hydrological ecosystem services we model and account for are crop water supply, household water supply (groundwater supply and surface water supply), water purification, and soil erosion control. We select these four services because they are critical to food and water security for the population. Agriculture is the major source of income and livelihood in the watershed and is predominantly rainfed. Furthermore, groundwater is the major source of household water use (for both drinking and non-drinking purposes).

2.2 STUDY AREA

The Upper Ouémé watershed as depicted in Figure 2.1 is located in central Benin covering an area of approximately 14,500 km². The natural vegetation is a mosaic of savannah woodland and small forest islands. The Upper Ouémé forest reserve is the major protected forest area in the watershed covering about 2 420 km². Smallholder rainfed agriculture is the major economic activity and is supported by climatic conditions that are characterized by a unimodal rainfall season from May to October of about 1,250 mm per annum. Maize, rice, yam, cassava and millet are some of the important food crops cultivated in this area with cotton being the major cash crop in the study area. These crops are predominantly cultivated using rainfed agriculture. The irrigation sector is relatively poorly developed. Fertilizer use is increasing in the region and high fertilizer inputs are associated with crops such as maize, rice and cotton (*Bossa et al*., 2012). An estimated average of 100 – 250 kg ha^{-1} of fertilizer (NPK + Urea) is applied to cotton, rice and maize (*Bossa et al*., 2012). With a population of about 400 000, there is a low demographic density (28 inhabitants km^{-2}) in the watershed (*Judex and Thamm*, 2008). However, the population is growing rapidly (about 4% per annum) due to migrants coming from different parts of the country and other neighbouring countries to farm. Rapid population growth has caused the expansion of agricultural areas and led to both deforestation and increasing scarcity of agricultural land (*Judex and Thamm*, 2008) accompanied by increasing soil degradation due to shortening of the fallow period (*Giertz et al*., 2012). It has been estimated that there will be nearly complete deforestation in a part of the Upper Ouémé watershed assuming a 6% per annum expansion of agricultural areas (*Orekan*, 2007). Conversion of savannah woodland and forests for crop cultivation is mainly through slash and burn techniques. In addition, the population obtains about 90% of their drinking water needs directly from groundwater, with about 5% from small lakes, ponds and rivers collectively referred to in this study as surface water (*Judex and Thamm*, 2008).

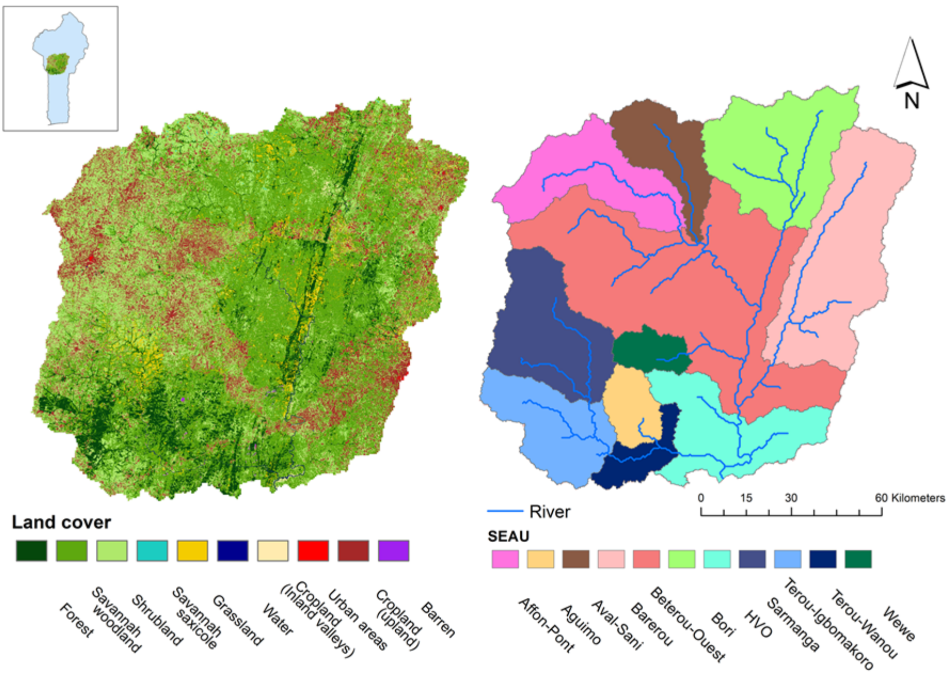


Figure 2.1. Land cover and subwatershed ecosystem accounting units (SEAUs) of the Upper Ouémé watershed. Land cover data adapted from Judex and Thamm (2008).

2.3 METHODS

2.3.1 MODELLING FRAMEWORK

MODEL SELECTION

In order to address modelling challenges regarding model process inclusion, spatial heterogeneity, physical and mathematical representation, temporal resolution, and model accuracy, we considered several watershed models and selected the SWAT model (*Arnold et al.*, 1998) to be most appropriate for this study. The SWAT model has a comparative advantage in integrated assessment modelling of ecohydrological interactions that underpin hydrological ecosystem service provision (*Vigerstol and Aukema*, 2011). The SWAT model is a physically based, ecohydrological model that simulates the impact of land use and land management practices on water, sediments and agricultural chemicals in large complex watersheds (*Neitsch et al.*, 2009). It is a continuous simulation watershed model operated at a daily time-step. In the SWAT model, a watershed can be spatially discretized using three approaches. They are grid cells, representative hillslopes, and hydrologic response units (HRUs) (*Arnold et al.*, 2013). The HRU-based discretization is the most popular and most geographic information system interfaces are set up to use this discretization (e.g. ArcSWAT). Each HRU is a lumped area within a subwatershed that is comprised of unique land cover, soil and management combinations (*Neitsch et al.*, 2009). The hydrological cycle is divided into two phases. The first is the land phase, which controls the amount of water, sediment, nutrient and pesticide loadings to the main channel in each subwatershed. Land phase processes include; weather, hydrology (canopy storage, infiltration, evapotranspiration, surface runoff, lateral subsurface flow, return flow) plant growth, erosion, nutrients and management operations (*Neitsch et al.*, 2009). Surface runoff, lateral flow and return flow from the land phase are then routed through the channel network of the watershed to the outlet in the second phase called the routing phase. This phase also includes processes such as sediment and nutrient routing (*Neitsch et al.*, 2009).

MODEL MODIFICATION

The SWAT model used in this study had two major modifications; the first one was a model process modification whereas the second one was a modification of the spatial discretization scheme. The process modification involved the incorporation of a landscape routing sub-model that simulates surface water, lateral and groundwater flow interactions across discretized landscape units. This sub-model was developed and incorporated into the standard SWAT model by *Volk et al.* (2007) and *Arnold et al.* (2010). The modified model, SWAT Landscape model, addresses an inherent weakness in the standard SWAT model. The standard SWAT model uses an HRU-based discretization and transported water, sediment, nutrient and pesticide loadings from upstream HRUs are routed directly into stream channels bypassing downstream HRUs (*Gassman et al.*, 2007; *Volk et al.*, 2007; *Arnold et al.*, 2010; *Bosch et al.*, 2010). Therefore, the impact of management of upstream HRUs on downstream HRUs cannot be sufficiently assessed. This weakness is a result of the lack of spatial interactions among different HRUs in the land phase of the hydrological cycle (*Neitsch et al.*, 2009). The SWAT Landscape model addresses this weakness by using a constant flow separation ratio to partition landscape and channel flow in each HRU (*Arnold et al.*, 2010). The channel flow portion is routed through the stream network, whereas the landscape flow portion is routed from upstream HRUs to downstream HRUs.

The second modification was a change from the HRU-based spatial discretization scheme of the standard SWAT model to a grid-based landscape discretization scheme. We set up the SWAT Landscape model with this grid-based landscape discretization using SWATgrid (*Rathjens and Oppelt*, 2012). The grid-based setup of the SWAT Landscape model uses a modified topographic index to estimate spatially distributed proportions of landscape and channel flow (*Rathjens et al.*, 2015), unlike the HRU-based setup which uses a constant flow separation ratio. A new parameter called the drainage density factor controls the spatially distributed flow separation in the SWATgrid setup (*Rathjens et al.*, 2015). This parameter can be adjusted during calibration. For this study, the grid-based setup of the SWAT Landscape model was used to delineate the watershed into spatially interacting grid cells. Flow paths were determined from the DEM and the digital landscape analysis tool TOPAZ (*Garbrecht and Martz*, 2000) and runoff from a grid cell flowed to one of eight adjacent cells (*Rathjens et al.*, 2015). A detailed description of the two modifications can be found in *Arnold et al.* (2010) and *Rathjens et al.* (2015).

MODEL INPUT DATA

A combination of spatial and non-spatial input data from a variety of sources were used to set up the model. The spatial input data are described in Table 2.1. A 30m digital elevation model (DEM) was obtained from the National Aeronautics and Space Administration (NASA) ASTER Global Digital Elevation Map to generate stream network, watershed configurations and to estimate topographic parameters. Land cover and soil maps were obtained from the "Integrated Approach to Efficient Management of Scarce Water Resources in West Africa" (IMPETUS) project database (*Judex and Thamm*, 2008). The land cover map had been derived from classification of LANDSAT-7 ETM+ satellite image. Gridded daily precipitation data were obtained from the "African Monsoon and Multidisciplinary Analysis–Coupling the Tropical Atmosphere and the Hydrological Cycle" (AMMA-CATCH) database (*AMMA-CATCH*, 2014) and gridded temperature data were obtained from Climate Research Unit (CRU) TS 3.21 database (*Jones and Harris*, 2013). Data on groundwater and surface water household consumption (including drinking and non-drinking purposes) were obtained from the IMPETUS project database. These had been derived from national census and household surveys in about 200 towns and communities within the watershed (*INSAE*, 2003; *Hadjer et al.*, 2005; *Judex and Thamm*, 2008). In our study area, per capita groundwater consumption was 19 litres per day per person and per capita surface water consumption was 14 litres per day per person (*INSAE*, 2003; *Hadjer et al.*, 2005).

MODEL CONFIGURATION AND PERFORMANCE EVALUATION

The initial model setup was carried out with the ArcSWAT interface, which is based on an HRU configuration. This was essential for generating input data for the grid-based configuration. Simulations of the HRU-based SWAT model were conducted for the period 1999 to 2012. The first two years (1999 and 2000) served as a warm-up period for the model to assume realistic initial conditions. Potential evapotranspiration was modelled with the Hargreaves method

(*Hargreaves et al.*, 1985) and water transfers for households were modelled as constant extraction rates from shallow aquifers (groundwater extractions) and streams (surface water extractions). The Soil Conservation Service curve number approach was used to model surface runoff and daily curve number value was calculated as a function of plant evapotranspiration (*Neitsch et al.*, 2009). The HRU-based SWAT model was first calibrated and validated with streamflow data before calibration and validation of sediment and nitrogen loads. A split-time calibration and validation technique was carried out on the HRU-based model using the Sequential Uncertainty Fitting (SUFI-2) optimization algorithm of the SWAT-Calibration and Uncertainty Program (*Abbaspour et al.*, 2008). For calibration and validation of streamflow, we used daily observed streamflow data from 11 monitoring stations within the watershed. These stations had drainage areas of varying spatial scale to capture watershed-scale and subwatershed-scale ecohydrological processes. Calibration was mostly from 2001 to 2007 and validation was from 2008 to 2011. To evaluate transport of sediments and nutrients, the model was further calibrated with weekly measured sediment and organic nitrogen load data. Two years of data (2008 to 2009) were available from a single monitoring station, Beterou station. Sediment and organic nitrogen load data for 2008 were used for the calibration whereas data for 2009 were used for the validation.

Table 2.1. Description of spatial input data of the Upper Ouémé watershed for the SWAT Landscape model

Data type	**Description**	**Resolution**	**Source**
Topography	ASTER Digital Elevation Model (DEM)	30m	NASA
Land use/ land cover	Classified LANDSAT-7 ETM+ image	28.5m	IMPETUS
Soil types	Soil map and associated parameters derived from geological maps and field surveys	1:200,000	IMPETUS
Precipitation	Gridded daily precipitation data (1999 to 2012)	25km	AMMA-CATCH
Temperature	Gridded monthly average minimum and maximum temperatures (1999 to 2012)	50km	CRU TS 3.21
Household water consumption	Groundwater and surface water extractions	(village level)	IMPETUS

*Table 2.2. Calibrated parameter values of the SWAT Landscape model. Superscript **a** indicates that the fitted values depended on the land cover type. Superscript **b** indicates that this parameter was used only in the calibration of the grid-based SWAT Landscape model. Subscript **v_** indicates that the parameter value is replaced by the fitted value. Subscript **r_** indicates the parameter value is multiplied by (1 + the fitted value).*

Parameter name	Description	Fitted values
$_{r}$_CN2	Initial SCS runoff curve number for moisture condition II	(from -0.2 to -0.05)[a]
$_{v}$_RCHRG_DP	Deep aquifer percolation fraction	0.2
$_{v}$_GW_REVAP	Groundwater re-evaporation coefficient	0.18
$_{v}$_GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur	1000
$_{v}$_REVAPMN	Threshold depth of water in the shallow aquifer for re-evaporation or percolation to the deep aquifer to occur	500
$_{v}$_SURLAG	Surface runoff lag coefficient	0.12
$_{r}$_SOL_AWC	Available water capacity of the soil	0.1
$_{v}$_ESCO	Soil evaporation compensation factor	(from 0.001 to 0.2)[a]
$_{v}$_EPCO	Plant uptake compensation factor	(from 0.1 to 1)[a]
$_{v}$_USLE_P	USLE equation support practice factor	0.13
$_{v}$_USLE_C	Minimum value of USLE C factor for water erosion applicable to the land cover	(from 0.038 to 0.45)[a]
$_{v}$_NPERCO	Nitrate percolation coefficient	0.2
$_{v}$_N_UPDIS	Nitrogen uptake distribution parameter	70
$_{v}$_SDNCO	Denitrification threshold water content	1.1
$_{v}$_CDN	Denitrification exponential rate coefficient	1.4
$_{v}$_DD[b]	Drainage density factor which affects the flow separation ratio	7.5

The calibrated and validated input parameter sets from the HRU-based setup was transferred to the grid-based setup of the SWAT Landscape model using the SWATgrid interface (*Rathjens and Oppelt*, 2012). Given the computational resources and time required to run a grid-based setup of the SWAT Landscape model at a higher spatial resolution (e.g. 1 ha) for a relatively large watershed, such as the Upper Ouémé (*Arnold et al.*, 2010; *Rathjens and Oppelt*, 2012), we resampled the DEM, soil and land cover data to a resolution of 500 m × 500 m. The resampling allowed for a balance between computational efficiency during model simulation and maintenance of accurate spatial representation of landscape patterns. Grid-based simulations of the SWAT Landscape model were conducted for the period 1999 to 2012. The first two years served as a model warm-up period. The grid-based setup of the SWAT Landscape model was then calibrated manually by adjusting only the drainage density factor parameter. The full calibrated parameter values are listed in*Table 2.2*. Three quantitative statistics recommended by *Moriasi et al.* (2007) were selected to evaluate model performance: Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and ratio of the root mean square error to the standard deviation of measured data (RSR). Nash-Sutcliffe efficiency is a normalized statistic that determines the relative magnitude of the residual variance compared to the measured data variance (*Nash and Sutcliffe*, 1970); PBIAS measures the average tendency of the simulated data to be larger or smaller than their observed counterparts (*Gupta et al.*, 1999); RSR standardizes root mean squared error using the observations standard deviation (*Moriasi et al.*, 2007).

2.3.2 SPATIAL ASSESSMENT OF HYDROLOGICAL SERVICES

For each hydrological ecosystem service, two appropriate indicators were selected to model service flow and service capacity. Computations were made for each grid cell enabling the model to reflect spatial differences in service flow and in service capacity. The selected hydrological ecosystem services and their service flow and service capacity indicators are shown in Table 2.3.

Table 2.3. Overview of selected hydrological ecosystem services and associated service flow and service capacity indicators (GP is growing period)

Hydrological ecosystem service	Service flow indicator	Service capacity indicator
1. Crop water supply	Total number of days during the growing period in which there was no water stress (days GP^{-1})	Total number of days in a year in which moisture supply (i.e. the sum of daily precipitation and plant available soil water content) was equal to or exceeded potential evapotranspiration (days yr^{-1})
2. Household water supply		
a. Groundwater supply	Amount of groundwater extracted (m^3 ha^{-1} yr^{-1})	Groundwater recharge (m^3 ha^{-1} yr^{-1})
b. Surface water supply	Amount of surface water extracted (m^3 ha^{-1} yr^{-1})	Water yield (m^3 ha^{-1} yr^{-1})
3. Water purification	Rate of denitrification (kg ha^{-1} yr^{-1})	Denitrification efficiency (% denitrified)
4. Soil erosion control	Reduction in soil loss (metric tons ha^{-1} yr^{-1})	Maximum potential reduction in soil loss (metric tons ha^{-1} yr^{-1})

CROP WATER SUPPLY

An important hydrological ecosystem service input to crop production in rainfed agricultural systems is the provision of plant available water by ecohydrological processes that affect the soil water balance (*Pattanayak and Kramer*, 2001; *IWMI*, 2007; *Zang et al.*, 2012). Crop water stress is a major limitation to crop production in rainfed agricultural systems (*IWMI*, 2007). The land cover input data did not differentiate the types of crops grown in croplands. For our simulations, we assumed that all croplands were used for only maize cultivation (which is the most common crop in our study area in terms of extent of cultivated land area). For maize cultivation, the growing period (GP), i.e. the time-period between crop establishment and harvesting, was 103 days, whereas GP for rice cultivation was 123 days. For both maize and rice, crop establishment was in the month of June. Service flow of crop water supply was modelled as the total number of days during a growing period in which there was no water stress (i.e., days when the total plant water uptake was sufficient to meet maximum plant water demand). Service flow depends on the specific type of crop cultivated. This approach is based on the model output variable, daily water stress, and is a modification of *Notter et al.* (2012). For each day, the model used Eq. (2.1) to compute water stress for a given grid cell, j (*Neitsch et al.*, 2009). After model simulation, service flow was computed using Eq. (2.2).

$$W_{strs,j} = 1 - T_{act,j}/T_{max,j}, \tag{2.1}$$

$$S_{f,j} = N(d_1, d_2, \ldots, d_n \mid W_{strs} = 0)_j, \tag{2.2}$$

where W_{strs} is daily water stress, T_{act} is plant water uptake or actual transpiration (mm), and T_{max} is maximum plant water demand or maximum transpiration (mm). S_f is the service flow (days GP^{-1}), N is the number of days d_1 to d_n, when W_{strs} was zero.

Service capacity on the other hand was modelled as the total number of days in a year in which moisture supply (i.e. the sum of daily precipitation and plant available soil water content) was equal to or exceeded potential evapotranspiration (e.g. *Jones and Thornton*, 2009). For a given spatial unit, this gives an indication of the number of days when potentially there will be no crop water stress irrespective of the crop type to be cultivated. This approach has management relevance.

HOUSEHOLD WATER SUPPLY

This hydrological ecosystem service refers to the amount of water extracted before treatment for household consumption (drinking and non-drinking purposes) (*UN et al.*, 2014a). This measurement boundary excluded other sources of water (e.g. tap water) where economic agents or inputs (e.g. water treatment facilities) were used to modify the state of the water resources before household consumption. We acknowledge that inflows to reservoirs of water distribution and processing facilities that deliver tap water can be considered as a hydrological ecosystem service. However, we excluded this from our study. This is because in our study area, the population obtains about 90% of their drinking water needs from groundwater, with about 5% from small lakes, ponds and rivers collectively referred to in this study as surface water (*Judex and Thamm*, 2008). A distinction was made between service capacity and service flow from groundwater, and service capacity and service flow from surface water.

To model service flow from groundwater and surface water, data on water consumption per capita, village population and water access for about 200 communities within the watershed were used. These data had been extracted from the 2002 national census (*INSAE*, 2003) and from household surveys in the study area (*Hadjer et al.*, 2005). The data represented household water consumption (including drinking and non-drinking purposes) and lacked information on the actual points of extraction. Therefore, in modelling the service flow, we assumed that there is a positive spatial correlation between points of consumption and points of extraction. Furthermore, to estimate village population from 2003 to 2012, we applied a 4% per annum growth rate (*Judex and Thamm*, 2008). Water consumption per capita, however, was kept constant. A population density grid was created using ArcGIS Kernel Density function (*ESRI*, 2012) and multiplied by water consumption per capita to estimate the amount of water consumed per grid cell. The amount consumed per grid cell, then gives an indication of the amount extracted per grid cell.

The ecosystem's capacity to support groundwater extractions was modelled as groundwater recharge, which is the total amount of water entering the aquifers within a specified time-step (e.g. month or year) (*Arnold et al.*, 2013). The ecosystem's capacity to support surface water extractions, however, was modelled as the water yield. Water yield is the net amount of water contributed by a grid cell to the river network within a specified time-step (*Arnold et al.*, 2013). Both groundwater recharge and water yield are model output variables.

WATER PURIFICATION

In the Upper Ouémé watershed, fertilizer application is increasing and high fertilizer inputs are associated with crops such as maize, rice and cotton (*Bossa et al.*, 2012). Increasing fertilizer application can lead to contamination of groundwater and surface water resources through nutrient leaching. This poses serious environmental and health risks to the beneficiaries of these systems (*Tilman et al.*, 2002; *Wolfe and Patz*, 2002). In our study area, groundwater provides over 90% of the total household water consumption. Water purification is, therefore, an essential ecosystem service in the Upper Ouémé watershed that increases the quality of groundwater for human consumption as well as other purposes. One of the challenges in terms of quantifying hydrological ecosystem services is the identification of management relevant indicators that can be enhanced through management interventions to augment the service production. For this study, we used soil denitrification as an indicator of this hydrological ecosystem service. Soil denitrification controls the rate of nitrate leaching by determining the quantities (after plant uptake) of nitrate available for leaching into groundwater systems (*Jahangir et al.*, 2012). For example, *Kramer et al.* (2006) observed that organic farming supports more active and efficient denitrifier communities leading to a considerable reduction in nitrate leaching as compared to conventional farming. In this study, the SWAT Landscape model was used to simulate the complete nitrogen cycle and service flow was estimated directly as the rate of denitrification, a model output variable. We should emphasize that the SWAT Landscape model does not explicitly simulate microbial processes and dynamics, but rather it simulates the ecohydrological conditions suitable for denitrification to occur (*Boyer et al.*, 2006). The model, therefore, computes denitrification as a function of soil moisture content, soil temperature, presence of a carbon source and nitrate availability using Eq. (2.3) and Eq. (2.4) (*Neitsch et al.*, 2009).

$$N_{dn} = \mathrm{NO_3} \cdot \left(1 - \exp\left[-\beta_{dn} \cdot \gamma_{tmp} \cdot C_{org}\right]\right) \qquad \text{if } \gamma_{sw} \geq \gamma_{sw,\ thr}\,, \qquad (2.3)$$

$$N_{dn} = 0 \qquad \text{if } \gamma_{sw} < \gamma_{sw,\ thr}\,, \qquad (2.4)$$

where N_{dn} is the amount of nitrogen lost through denitrification (kg ha^{-1}), NO_3 is the amount of nitrate in the soil (kg ha^{-1}), β_{dn} is the rate coefficient for denitrification, γ_{tmp} is the nutrient cycling temperature factor, γ_{sw} is the nutrient cycling water factor, $\gamma_{sw,thr}$ is the threshold value of nutrient cycling water factor for denitrification to occur, C_{org} is the amount of organic carbon (%). The values of β_{dn} and $\gamma_{sw,thr}$ are user defined values and were adjusted during calibration; β_{dn} was 1.4 and $\gamma_{sw,thr}$ was 1.1.

Service capacity was estimated as the denitrification efficiency, which in this study was computed using Eq. (2.5). When the ecohydrological conditions required for denitrification are present, the rate of denitrification (service flow) is determined by the amount of nitrate available in the soil. Unlike other land cover types (which only receive nitrogen or nitrates from wet deposition or from overland flow), cropland areas receive additional nitrogen or nitrates through fertilizer application. Therefore, for a given grid cell, denitrification efficiency determines the proportion of the total nitrate that is denitrified. As a measure of service capacity, denitrification efficiency gives an indication of the suitability of a spatial unit for denitrification.

$$\mathrm{DN_{eff}} = \left(N_{dn} / N_{total} \right) \cdot 100 \,, \tag{2.5}$$

where DN_{eff} is the denitrification efficiency (%), N_{dn} is the amount of nitrogen lost through denitrification in the time-step (kg ha^{-1}), N_{total} is the total amount of nitrogen available (e.g. through fertilizer application, wet deposition etc.) in the time-step (kg ha^{-1}).

SOIL EROSION CONTROL

Controlling soil erosion in the watershed has numerous benefits including maintaining soil fertility, preventing river sedimentation, and downstream water quality. There are inherent physical soil and landscape properties such as soil erodibility and slope that affect soil erosion (*Williams*, 1975). However, we focussed on the role of vegetation cover in controlling soil erosion. Service flow was modelled as the actual reduction in soil loss produced by the existing vegetation cover and was computed using Eq. (2.6).

$$\mathrm{SD_{rtd}} = S_{yld,\ pot} - S_{yld} \,, \tag{2.6}$$

where SD_{rtd} is the reduction in soil loss produced by the existing vegetation cover (metric tons ha^{-1}), $S_{yld,\ pot}$ is the maximum potential soil loss in the absence of vegetation cover (metric tons ha^{-1}), and S_{yld} is the soil loss under prevailing vegetation cover and land management practices (metric tons ha^{-1}). Both $S_{yld,\ pot}$ and S_{yld} were computed with the Modified Universal Soil Loss Equation (*Williams*, 1975) incorporated in the SWAT Landscape model.

For service capacity of soil erosion control, we used the maximum potential reduction in soil loss produced by the vegetation cover as an indicator. This maximum potential reduction in soil loss (maximum potential soil retention) can be said to be equal to the maximum potential soil loss. For example, for a specified spatial unit, if the maximum potential soil loss in the absence of the vegetation cover is estimated as 2 metric tons ha^{-1} yr^{-1} then it indicates that the potential reduction in soil loss due to the vegetation cover cannot be greater than 2 metric tons ha^{-1} yr^{-1}. The maximum potential soil loss was modelled assuming there was no vegetation cover (e.g. *Leh et al.*, 2013; *Terrado et al.*, 2014).

2.3.3 ACCOUNTING FOR HYDROLOGICAL SERVICES

Biophysical ecosystem accounts are the basis for monetary accounting and were set up in accordance with SEEA-Experimental Ecosystem Accounting guidelines (*UN et al.*, 2014a). We defined 11 Subwatershed Ecosystem Accounting Units (SEAUs) as the spatial scales of aggregation. We set up annual biophysical service capacity and service flow accounts for each SEAU. The 11 SEAUs were defined from a total of 44 subwatersheds based on the drainage areas of streamflow monitoring stations within the watershed. The monitoring stations are listed in Table 2.4. The 44 subwatersheds were delineated from the ASTER Global Digital Elevation Map as part of the initial model setup with ArcSWAT. Some monitoring stations with smaller drainage areas were nested within those with larger drainage areas. In such cases the SEAU was defined as the drainage area of the nested monitoring station because we wanted to set up spatially disaggregated accounts. Large drainage areas of other monitoring stations had nested subwatersheds within them that were ungauged. In these cases also, the SEAU was defined as the nested subwatershed. For each SEAU, the spatial estimates of service capacity-load per grid cell (500m × 500m) and service flow-load per grid cell (500m × 500m) that had been computed in Sect. 2.3.2 were then aggregated.

A key motivation for ecosystem accounting is to provide information for tracking changes in ecosystems and linking those changes to economic and other human activities (*UN et al.*, 2014a). Trend analysis statistical tests were conducted on the total annual values (or total seasonal values for crop water supply) of service capacity accounts in each SEAU. Trend analysis determines if the changes in service capacity over time are due to random variability or statistically significant and consistent changes. This was conducted using the non-parametric Mann-Kendall test for trend. The Mann-Kendall test for trend statistically determines if there is a monotonic upward or downward trend of a variable over time. A trend was detected if temporal variation in service capacity was statistically significant at 5% significance level (P-value < 0.05). If a trend was detected, the Mann-Kendall statistic and Sen's slope estimator were calculated. The Mann-Kendall statistic is a measure of the strength and direction of a trend, whereas Sen's slope estimator is a measure of the magnitude of a trend.

2.4 RESULTS

2.4.1 SWAT LANDSCAPE CALIBRATION AND VALIDATION RESULTS

Table 2.4 shows the statistical results of the model calibration and validation and Figure 2.2 and Figure 2.3 show the graphical results. There are no established absolute criteria for judging model performance. For this study, we used the criteria recommended by *Moriasi et al.* (2007). A watershed model is said to be performing satisfactorily if the NSE > 0.50 and RSR < 0.70, PBIAS within the range -25 to 25 for streamflow, -55 to 55 for sediment, and -70 to 70 for nutrients. At different spatial scales (e.g. Affont-Pont, 1,172 km^2; Igbo, 2 309 km^2; Beterou, 10 046 km^2), the model simulated hydrological processes satisfactorily as shown in Figure 2.2. Seven out of 11 stations recorded NSE values greater than 0.5 during model validation of streamflow. Even though the NSE values for some monitoring stations were less than 0.5, all but one were greater than 0.0, indicating that the simulated streamflow was still a better predictor than the mean of the observed values. Monitoring stations with larger drainage areas recorded higher NSE values than stations with smaller drainage areas. The PBIAS values in Table 2.4 show the level of bias in simulated streamflow. A negative PBIAS value indicates model overestimation whereas a positive PBIAS value indicates model underestimation. The validation results show that the model largely underestimated streamflow at upstream stations and overestimated it downstream. The RSR scores show varying levels of residual variation indicating the level of errors in simulated streamflow as compared to observed streamflow. The closer the RSR value is to zero, the lower the level of residual variation in simulated streamflow. During model validation, five stations recorded RSR values lower than 0.7. The statistical and graphical results of sediment load and organic nitrogen load calibration are shown in Figure 2.3 and Table 2.4. The model performed satisfactorily during validation of sediment and organic nitrogen loads.

Table 2.4. Calibration and validation results for streamflow, sediment and organic nitrogen loads (Prefix $_{H}$__ indicates results for streamflow calibration and validation; prefix $_{S}$__ indicates results for sediment load calibration $_{N}$__ indicates results for organic nitrogen load calibration). NSE is Nash-Sutcliffe efficiency, PBIAS is percent bias, and RSR is ratio of the root mean square error to the standard deviation of measured data.

Monitoring stations	**Drainage area (km²)**	**Calibration**			**Validation**		
		NSE	**PBIAS**	**RSR**	**NSE**	**PBIAS**	**RSR**
Upstream stations							
$_{H}$_Affon-Pont	1 172	0.69	27.0	0.56	0.62	15.9	0.62
$_{H}$_Aval-Sani	760	0.70	12.0	0.55	0.64	7.8	0.60
$_{H}$_Bori	1 608	0.65	-24.7	0.59	-0.49	-121.4	1.22
$_{H}$_Tebou	522	0.47	43.5	0.72	0.58	20.3	0.65
Downstream stations							
$_{H}$_Beterou	10 046	0.85	5.7	0.39	0.78	-17.8	0.47
$_{H}$_Barerou	2 128	0.71	20.8	0.54	0.72	-22.7	0.53
$_{H}$_Cote-238	3 040	0.69	3.5	0.56	0.68	-18.4	0.56
$_{H}$_Igbomakoro	2 309	0.76	11.3	0.49	0.71	-4.0	0.54
$_{H}$_Sarmanga	1 334	0.48	23.2	0.72	0.44	17.2	0.75
$_{H}$_Aguimo	394	0.25	-20.9	0.87	0.12	-.60.1	0.94
$_{H}$_Wewe	297	0.42	21.6	0.76	0.42	-6.5	0.76
$_{S}$_Beterou	10 046	0.45	6.9	0.74	0.83	2.55	0.42

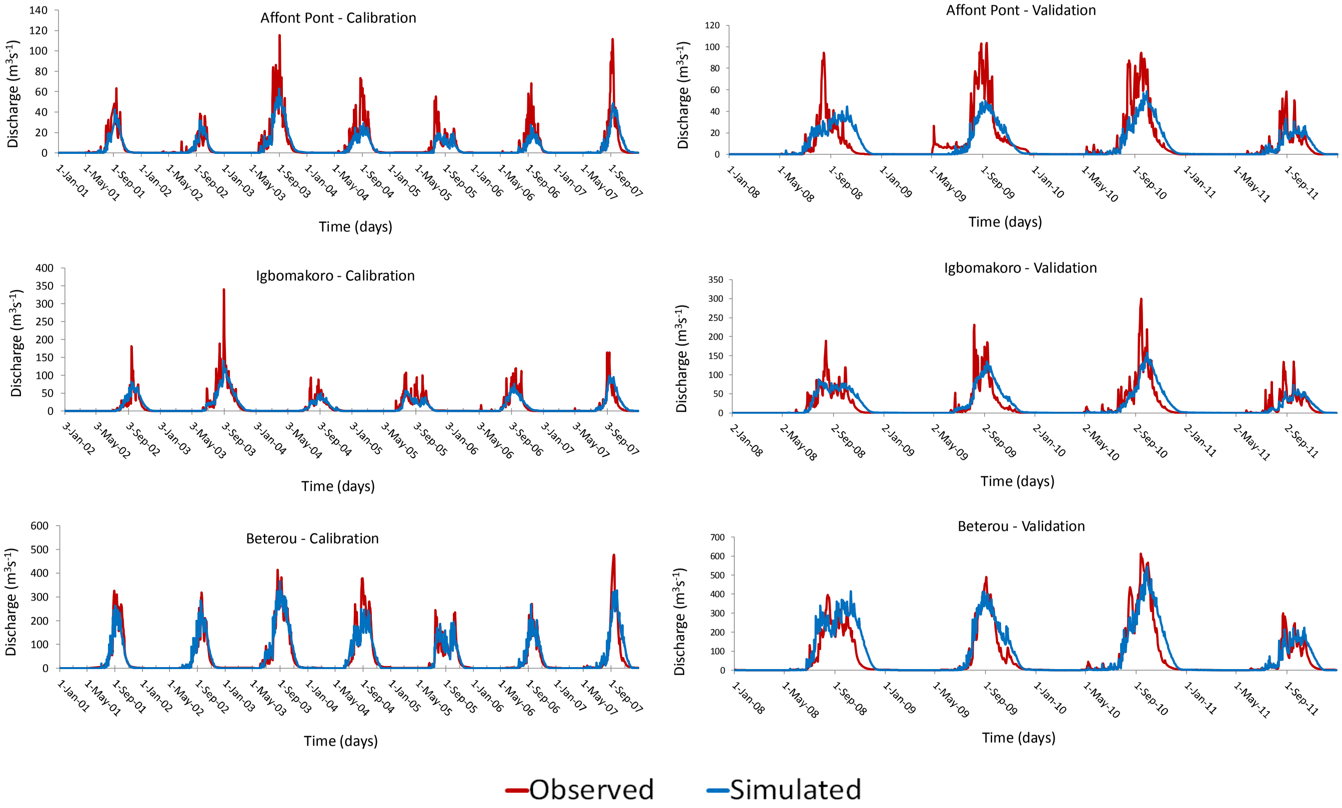


Figure 2.2. Comparing simulated and observed streamflow for three monitoring stations with varying drainage areas; Affont-Pont, 1,172 km^2; Igbomakoro, 2,309 km^2; Beterou, 10,046 km^2

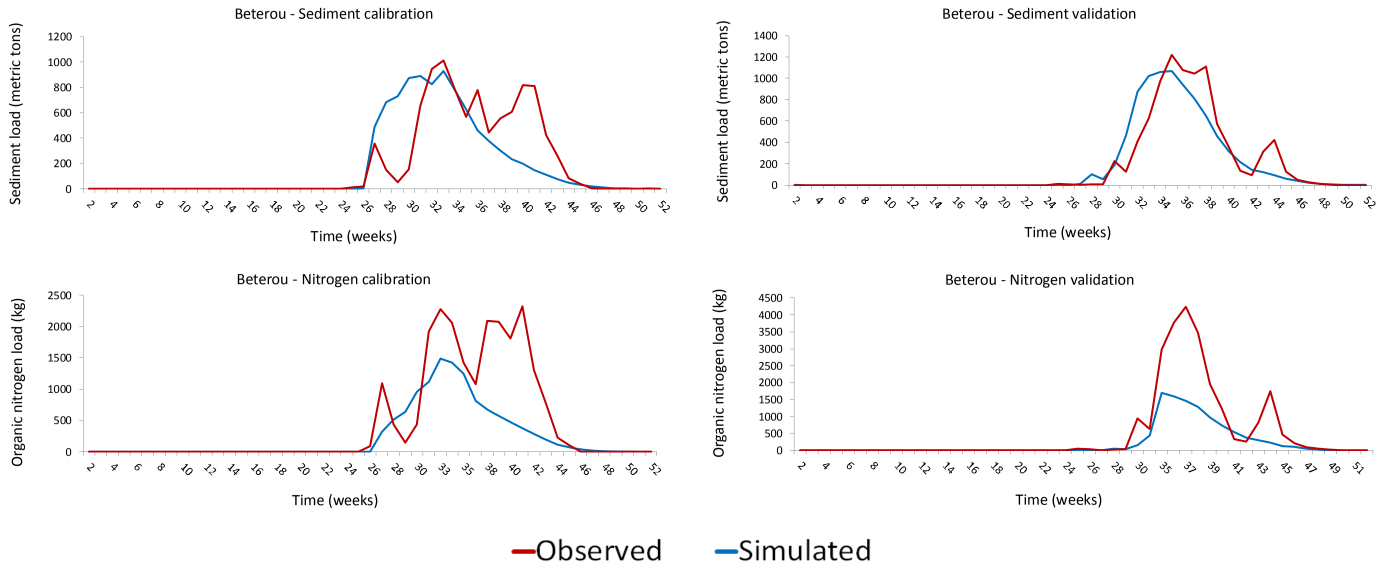


Figure 2.3. Comparing simulated and observed sediment loads and organic nitrogen loads during calibration and validation at Beterou monitoring station for the period 2008 to 2009.

2.4.2 SPATIAL PATTERNS OF HYDROLOGICAL SERVICES

Water supply by soil moisture is essential to reduce crop water stress in rainfed agricultural systems. If all other factors for crop growth (such as nutrients and temperature) remain constant, then a higher service capacity and higher service flow result in a higher crop yield. Computations of crop water supply were spatially restricted to croplands. High service flow indicates the suitability of a spatial unit under assumed maize cultivation whereas high service capacity indicates the potential suitability for crop cultivation irrespective of the crop type and not maize alone. The results of service capacity are indicative of the least number of days during a year crops would not experience water stress. Figure 2.4 reveals high spatial variability in service capacity and service flow in croplands. Mean annual values of service capacity ranged from 151 to 269 days yr^{-1} with a watershed-wide mean of 195 days yr^{-1} and standard deviation of 24 days yr^{-1}. Mean seasonal values of service flow ranged from 16 to 103 days yr^{-1} with a watershed-wide mean of 61 days yr^{-1} and standard deviation of 29 days yr^{-1}.

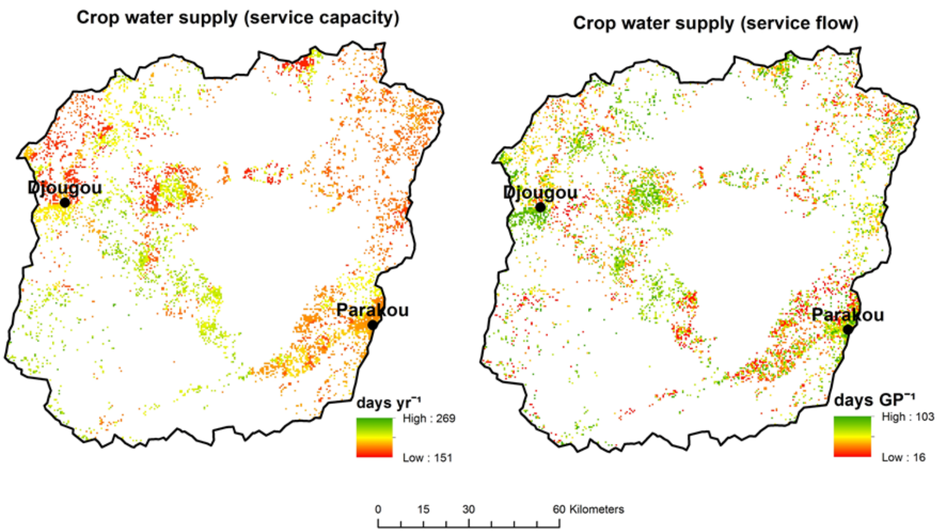


Figure 2.4. Spatial distribution of mean annual values of service capacity and mean seasonal values of service flow of crop water supply in croplands in the Upper Ouémé watershed from the year 2001 to 2012 (GP indicates growing period).

The spatial distribution of mean annual values of service capacity and service flow of groundwater supply and surface water supply are shown in Figure 2.5 and Figure 2.6 respectively. Groundwater is the major source of water for household consumption (drinking and non-drinking purposes) with the service flow (groundwater extraction) significantly higher than service flow of surface water supply (surface water extraction). High service flows of groundwater supply are concentrated in the most populous towns in the watershed. However, service flows in Parakou, which is the most populous city in the watershed, are relatively lower than other areas such as Djougou. This is because the population in Parakou depends mainly on tap water sources. Service capacity of groundwater supply exhibited high spatial variability. High values of service capacity were concentrated in the southwestern part of the watershed. For service capacity of surface water supply, Figure 2.6 shows areas with a high propensity for generating water yield. These areas, referred to as Hydrologically Sensitive Areas (HSAs) (*Agnew et al.*, 2006), were not peculiar to a particular land cover type. They occurred in almost all land cover types. They occurred more frequently in savannah woodland and shrubland because approximately 80% of the total watershed area is either one of this land cover type.

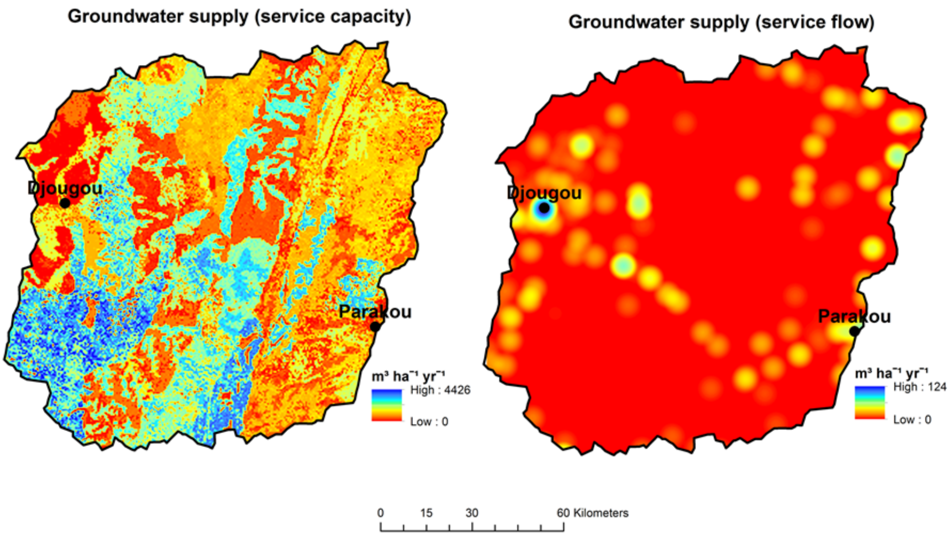


Figure 2.5. Spatial distribution of mean annual values of service capacity and service flow of groundwater supply in the Upper Ouémé watershed from the year 2001 to 2012.

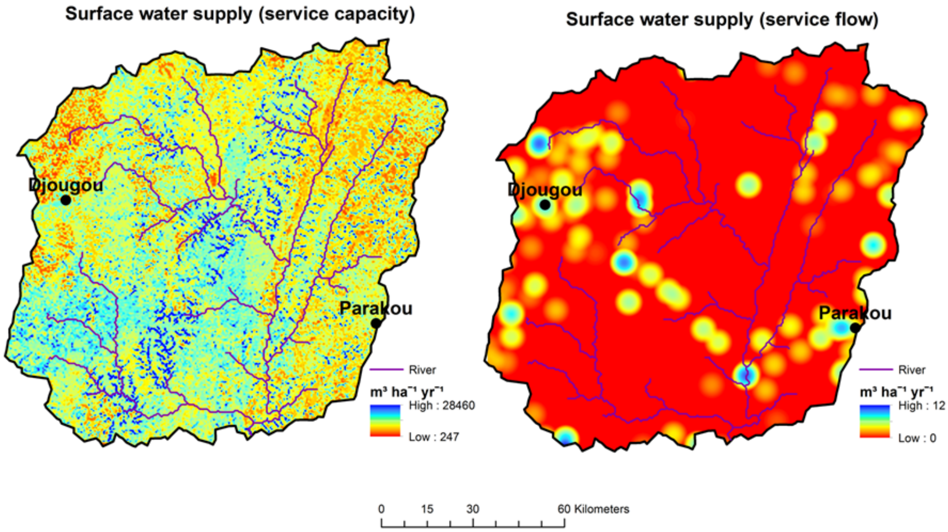


Figure 2.6. Spatial distribution of mean annual values of service capacity and service flow of surface water supply in the Upper Ouémé watershed from the year 2001 to 2012.

Water purification modelled as denitrification is essential to control the quantities of nitrate available for leaching and contaminating groundwater resources (*Jarvis*, 2000; *Jahangir et al.*, 2012). Service capacity was measured as the percentage of nitrate that is denitrified and service flow was the rate of denitrification. The spatial distribution of mean annual values of service capacity and service flow of water purification is distinctly concentrated in the northern and eastern parts of the watershed with the southwestern parts recording zero values (Figure 2.7). All barren land cover types also recorded zero values of service capacity and service flow. The zero values recorded are a result of the lack of soil saturation conditions and not the lack of nitrate availability. Soil saturation induces soil anaerobic conditions required for denitrification to take place. In areas where denitrification was recorded, the highest mean annual values of service flow were recorded in grasslands (7 kg ha^{-1} yr^{-1}). The highest mean annual values of service capacity were also recorded in grasslands (55 % yr^{-1}).

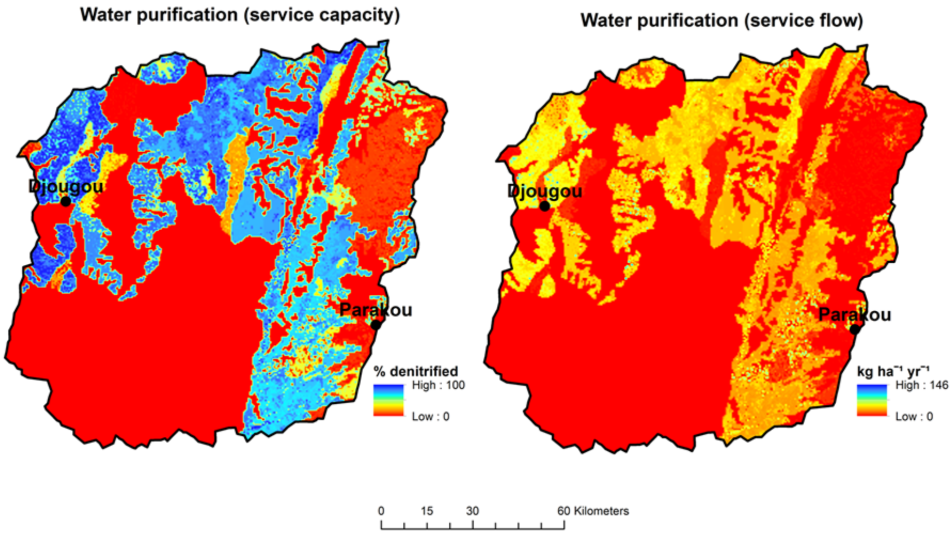


Figure 2.7. Spatial distribution of mean annual values of service capacity and service flow of water purification in the Upper Ouémé watershed from the year 2001 to 2012.

The spatial distributions of mean annual values of service capacity and service flow of soil erosion control are shown in Figure 2.8. High service capacity indicates high potential for reduction in soil loss produced by the vegetation cover. The service flow, however, is a measure of the actual reduction in soil loss under existing vegetation cover. Under existing vegetation cover, mean annual rate of soil loss in the watershed was recorded at 0.01 metric tons ha^{-1} yr^{-1} (standard deviation of 0.02 metric ton ha^{-1} yr^{-1}). The mean annual rate of soil loss in the watershed will increase significantly to 0.05 metric tons ha^{-1} yr^{-1} (standard deviation of 0.07 metric ton ha^{-1} yr^{-1}) should there be complete loss of the existing vegetation cover. This value, 0.05 metric tons ha^{-1} yr^{-1} (standard deviation of 0.07 metric ton ha^{-1} yr^{-1}), can also be interpreted as the maximum potential reduction in soil loss (service capacity) that can be produced by the existing vegetation cover. Under existing vegetation cover and management conditions, however, the actual reduction in soil loss (service flow) was recorded at a watershed-wide mean annual value of 0.04 metric tons ha^{-1} yr^{-1} (standard deviation of 0.07 metric ton ha^{-1} yr^{-1}). For both service capacity and service flow, only about 0.04% of the total area of the watershed recorded mean annual values greater than 1 metric ton ha^{-1} yr^{-1}. These areas had the steepest slopes, indicating the importance of vegetation cover in soil erosion control in these areas. In forested areas, service flow was equal to service capacity, indicating that overall there was no net soil loss from forested areas.

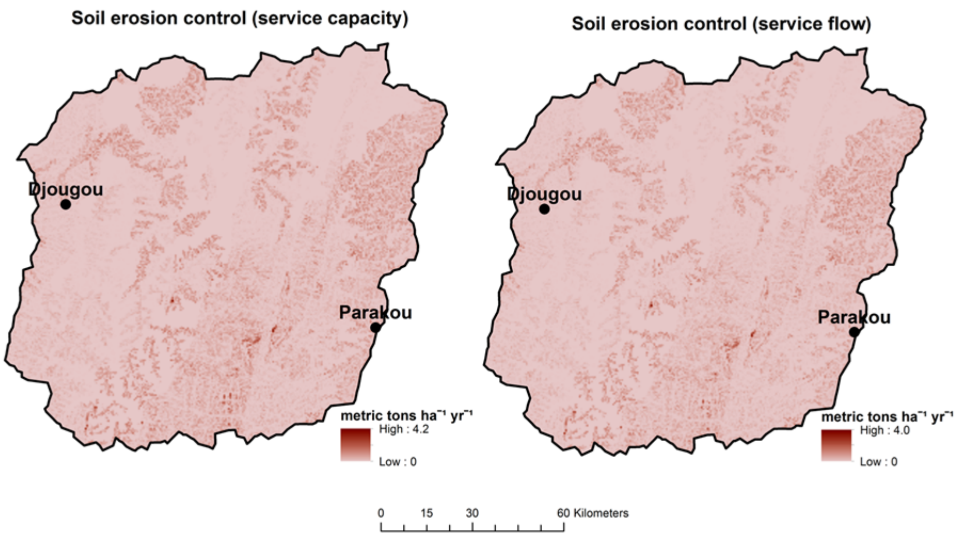


Figure 2.8. Spatial distribution of mean annual values of service capacity and service flow of soil erosion control in the Upper Ouémé watershed from the year 2001 to 2012.

2.4.3 BIOPHYSICAL ECOSYSTEM ACCOUNTS

The service capacity (Table 2.5) and service flow (Table 2.6) ecosystem accounting tables show the distribution of hydrological ecosystem services across the 11 SEAUs for the most current year of simulation, 2012. The total annual values of service capacity correlated with the spatial extent of an SEAU. Larger SEAUs recorded higher values than smaller SEAUs. However, the mean values of service capacity varied depending on the biophysical environment of an SEAU. For example, whereas the Beterou-Ouest SEAU is the largest, the highest mean service capacity of groundwater supply was recorded in Sarmanga and Terou-Igbomakoro SEAUs. This signifies that the rate of groundwater recharge is highest in Sarmanga and Terou-Igbomakoro SEAUs. The service flow table reveals that the ecohydrological conditions required for denitrification (water purification) do not occur in Aguimo, Terou-Igbomakoro, Terou-Wanou, and Wewe SEAUs. However, a total of 77,000 m^3 of groundwater was extracted in Terou-Igbomakoro and Wewe SEAUs in 2012. In Aguimo and Terou-Wanou SEAUs, there is currently no groundwater extraction. For crop water supply, the SEAUs with the largest cropland areas did not necessarily record the highest service flow. For example, the highest service flow was recorded in Sarmanga and Terou-Igbomakoro. This signifies that maize cultivation in these SEAUs is less prone to water stress than in any other SEAU.

Table 2.5. Biophysical ecosystem account for service capacity at the SEAU level in the Upper Ouémé watershed in 2012 (SD is standard deviation)

Subwatershed Ecosystem Accounting Unit (SEAU)	Hydrological ecosystem service									
	Crop water supply		Household water supply				Water purification		Soil erosion control	
			Groundwater supply		Surface water supply					
	Area (10^3 ha)	Mean (SD) (days yr^{-1})	Total (10^6 m^3 yr^{-1} recharge)	Mean (SD) (10^3 m^3 ha^{-1} yr^{-1} recharge)	Total (10^6 m^3 yr^{-1} water yield)	Mean (SD) (10^3 m^3 ha^{-1} yr^{-1} water yield)	Total N added (10^3 kg)	(% N denitrified)	Total (10^3 metric tons yr^{-1})	Mean (SD) (kg ha^{-1} yr^{-1})
Affon-Pont	20.2	189 (24)	121	26 (26)	624	133 (90)	2 719	30	5.2	44 (74)
Aguimo	0.3	217 (15)	58	37 (28)	255	161 (112)	589	0	2.6	65 (102)
Aval-Sani	4.0	206 (22)	114	38 (20)	458	151 (89)	1 370	36	3.5	45 (85)
Barerou	33.4	184 (12)	244	29 (20)	1,328	156 (101)	4 707	11	18.5	87 (101)
Beterou-Ouest	54.3	197 (25)	615	38 (30)	2,526	155 (99)	8 550	19	22.9	56 (102)
Bori	12.0	183 (20)	185	29 (20)	1,082	168 (97)	3 138	26	6.4	40 (79)
HVO	7.0	203 (21)	206	43 (40)	638	133 (80)	1 953	15	8.3	69 (93)
Sarmanga	9.7	212 (19)	304	57 (37)	809	152 (82)	2 382	13	4.5	34 (42)
Terou-Igbomakoro	4.0	221 (22)	222	57 (36)	591	151 (93)	1 561	0	4.9	51 (91)
Terou-Wanou	0.8	220 (9)	73	54 (22)	170	126 (58)	514	0	2.5	74 (90)
Wewe	4.1	211 (20)	48	40 (33)	213	177 (117)	638	0	1.8	61 (182)
Total	149.8		2,190	—	8,694	—	28 121	—	81.1	—

Table 2.6. Biophysical ecosystem account for service flow at the SEAU level in the Upper Ouémé watershed in 2012 (GP is length of growing period between crop establishment and harvest; GP is 103 days.

Subwatershed Ecosystem Accounting Unit (SEAU)	**Hydrological ecosystem service**							
	Crop water supply		**Household water supply**		**Water purification**		**Soil erosion control**	
			Groundwater	**Surface water**				
	Area (10^3 ha)	Mean (SD) (days GP^{-1})	Total (10^3 m^3 yr^{-1} water extracted)	Total (10^3 m^3 yr^{-1} water extracted)	Total (10^3 kg N yr^{-1} denitrified)	Mean (SD) (kg ha^{-1} yr^{-1} denitrified)	Total (10^3 metric tons yr^{-1})	Mean (SD) (kg ha^{-1} yr^{-1})
Affon-Pont	20.2	59 (30)	123	65	810	6.9 (10)	4.4	38 (67)
Aguimo	0.3	52 (35)	0	0	0	0.0 (0)	2.3	58 (92)
Aval-Sani	4.0	64 (29)	8	0.2	498	6.5 (7)	3.2	42 (81)
Barerou	33.4	63 (31)	510	64	503	2.4 (6)	15.9	75 (92)
Beterou-Ouest	54.3	59 (31)	1,124	219	1 613	4.0 (9)	18.9	46 (90)
Bori	12.0	65 (32)	196	30	815	5.1 (7)	5.4	33 (67)
HVO	7.0	56 (32)	71	37	297	2.5 (5)	7.0	59 (79)
Sarmanga	9.7	69 (34)	532	66	317	2.3 (5)	4.0	30 (39)
Terou-Igbomakoro	4.0	69 (34)	95	36	0	0.0 (0)	4.4	45 (85)
Terou-Wanou	0.8	45 (35)	0	0	0	0.0 (0)	2.2	65 (83)
Wewe	4.1	63 (31)	41	41	0	0.0 (0)	1.5	51 (178)
Total	149.8	—	2,700	558.2	4 853	—	69.2	—

Temporal analysis of ecosystem accounts makes it possible to track ecosystem changes and measure the degree of sustainability, degradation or resilience. Decreasing capacity of ecosystems to sustain human welfare over time is a measure of ecosystem degradation (*UN et al.*, 2014a).

Figure 2.9 shows the results of trend analysis statistical tests of service capacities at the SEAU level. Increasing trends were observed in changes in service capacities of water purification, groundwater supply and surface water supply. For ground water supply, increasing trends was observed in all SEAUs. The results in Figure 2.9A are of the five SEAUs with the highest Mann-Kendall statistic. Increasing trend in changes in surface water supply was observed in four SEAUs, whereas increasing trend in changes in water purification was observed in only the Aval-Sani SEAU. No trend was observed in changes in service capacity of crop water supply in croplands in all SEAUs. No trend was also observed in changes in service capacity of soil erosion control in all the SEAUs.

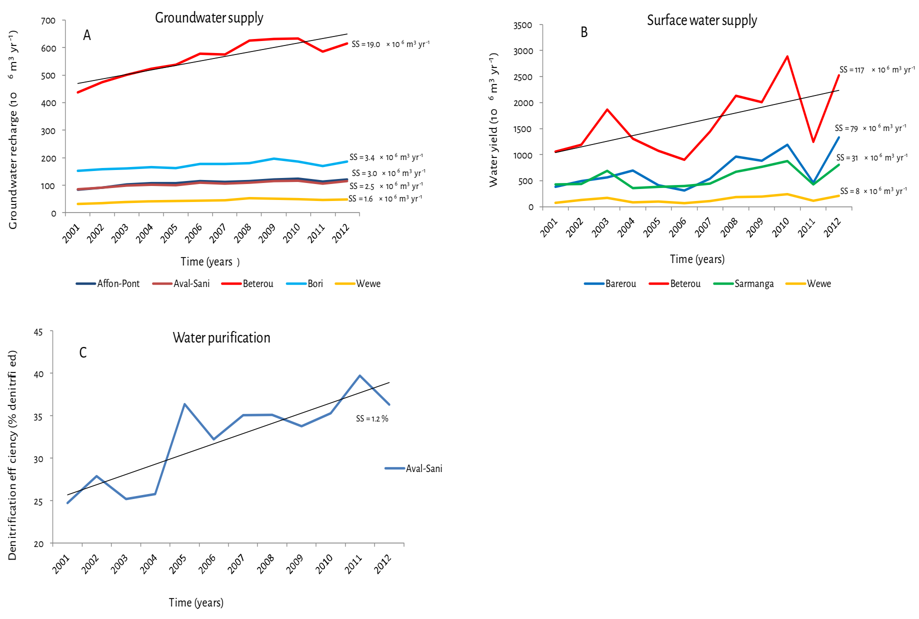


Figure 2.9. Trends in service capacity of hydrological ecosystem services at the SEAU level in the Upper Ouémé watershed (SS is Sen's Slope estimator, which is a measure of the magnitude of change of a trend). For each graph, a single trend line is drawn solely to illustrate the direction of trend.

2.5 DISCUSSION

2.5.1 MODELLING UNCERTAINTIES AND LIMITATIONS

Model results to support decision-making are always associated with a certain degree of uncertainty. Uncertainty in ecohydrological modelling with SWAT may be from input data, model algorithms, model calibration and validation (parameter non-uniqueness) (*Abbaspour et al.*, 2008). The major input uncertainty in our study was a result of resampling of spatial data from fine spatial resolutions to relatively coarse spatial resolutions in order to increase operational feasibility and computational efficiency of the grid-based SWAT Landscape model. We resampled land use/land cover data, DEM and soil map to a spatial resolution of 500 m × 500 m. Even though the spatial rigour of ecosystem accounting requires that modelling approaches that maintain adequate landscape spatial heterogeneity are more suitable, decisions on choice of spatial resolution should be made with model computational efficiency and operational feasibility in mind. For the SWAT model (and SWAT Landscape model), increasing spatial detail results in a considerable increase in computing time irrespective of the spatial discretization scheme employed (e.g. *Arnold et al.*, 2010; *Notter et al.*, 2012). In our case-study area, over 1,400,000 grid cells are generated at 1 ha resolution requiring over two days for each simulated year on 2.6 GHz and 8 GB RAM. Computer storage capacity for the huge data outputs generated may not also be readily available. We acknowledge that in many regions of the world high-resolution spatial input data may not be available at large spatial scales. However, for the grid-based setup of the SWAT Landscape model, when such high-resolution spatial data are available, it may be necessary to compromise spatial explicitness to achieve operational feasibility. This introduces a certain amount of uncertainty with regards to spatial variation in ecohydrological processes, therefore, such decisions should be made taking into consideration the degree of spatial heterogeneity of landscape features. The need to compromise spatial detail for operational feasibility may limit the applicability of this model configuration for larger watersheds.

For larger watersheds, it is also extremely difficult to obtain spatially and temporally correct representations of the underlying ecohydrological processes and interactions. To achieve this, there is the need for multi-site calibration at different spatial scales with a sufficient length of time-series of data to capture high and low flow years, annual, seasonal and monthly variations (*Santhi et al.*, 2008). In our study, the model underestimated streamflow (especially peak flow) at some monitoring stations, whereas at other stations it overestimated streamflow. These biases in streamflow estimation lead to error propagation in the other components of the water balance such as soil moisture and actual evapotranspiration. Whereas the use of 11 years of daily streamflow data from 11 monitoring stations in the Upper Ouémé watershed reduces the uncertainties of modelled results, data for calibration and validation of sediment and nitrogen loads may not have been sufficient to enable the model to more accurately represent sediment and nitrogen transport processes. In evaluating model performance of sediment and nitrogen transport processes, we used only one year of data from a single monitoring station. Without multi-site calibration and validation, there remain large uncertainties in modelled results of sediment and transport processes at different spatial scales. In addition, without long term temporal validation, there remain large uncertainties in the ability of the model to capture annual variability in these transport processes. Even with sufficient length of time series of multi-site data for calibration and validation, the problem of parameter non-uniqueness inherent in complex watershed models such as the SWAT model also introduce a degree of uncertainty in modelled results. Parameter non-uniqueness refers to the reproduction of similar observed ecohydrological signals by different input parameter sets. Therefore, even for so called calibrated and validated SWAT models, there is always a degree of uncertainty introduced by parameter non-uniqueness. To limit this non-uniqueness and consequently reduce parameter uncertainty requires the use of comprehensive data on different fluxes, loads and ecohydrological processes such as crop yield, soil moisture, groundwater level and evapotranspiration (*Abbaspour et al.*, 2008) that are most of the time not readily available.

For this study, we used soil denitrification as an indicator of water purification service. Quantifying denitrification at watershed and subwatershed scales requires the use of models such as SWAT. It involves the simulation of a complex set of processes controlling denitrification that can broadly be classified as the prerequisite environmental/ecohydrological conditions, and microbial processes and dynamics. The SWAT model, however, provides only simplified representations of the complex set of processes controlling denitrification and modelled estimates of denitrification rates remain highly uncertain (*Boyer et al.*, 2006). The model only simulates the environmental/ecohydrological conditions and does not explicitly simulate microbial processes and dynamics. There is, therefore, an inherent assumption of spatial homogeneity with regards to denitrifier community species composition, quantities and activities across all land use types. *Kramer et al.* (2006) reported that specific land use and management types (such as organic, integrated and conventional agriculture) enhance or inhibit soil denitrifier activities affecting the rate of denitrification. In the SWAT model, however, spatial variability in denitrification is determined mainly by spatial variability in ecohydrological and abiotic controlling factors.

2.5.2 LESSONS FOR ECOSYSTEM ACCOUNTING

In ecosystem accounting, detailed and accurate land cover and land use data are important. Apart from their use as inputs in modelling ecosystem services, land cover classes are also used as ecosystem accounting units based on which ecosystem services are aggregated (*Remme et al.*, 2014; *Schroter et al.*, 2014). A single lumped land cover class for agricultural areas or croplands (be it as model input data or ecosystem accounting units) may be suitable when modelling and accounting for other ecosystem services (*Remme et al.*, 2014; *Schroter et al.*, 2014). However, when modelling and accounting for crop water supply, land cover and land use data with detailed and spatially disaggregated information on the types of crops grown in agricultural areas is needed. This is because different crops have different water requirements (*Allen et al.*, 1998). In rainfed agricultural systems, crop water supply is the major limitation to crop production and is the main factor responsible for low yields in the seasonally dry and semiarid tropics and subtropics (*Shaxson and Barber*, 2003). However, in many of these regions, land cover and land use data with this level of detail are currently not available. Obtaining such information is complicated by the small plot sizes and cropping patterns varying from year to year. Our study area was no exception. Despite these constraints, the lack of detailed data reduces the accuracy and reliability of modelled results of service flow of crop water supply. In our study area, this limitation resulted in the simulation of only a single crop type in croplands. Therefore, the results of service flow of crop water supply should be interpreted in the context of the crop simulated. However, because methodologies such as (*Allen et al.*, 1998) have been used extensively to compute the water requirements of various crops, our approach serves as a reference or baseline from which the service flow of crop water supply of a spatial unit could be estimated if a crop other than maize is grown.

A key feature of ecosystem accounting is the distinction between service capacity and service flow. The empirical distinction and separate spatial characterisation of service capacity and service flow is essential in understanding the dynamics of service provision and in planning and devising sustainable management options. The distinction is also important for subsequent monetary valuation. Service capacity and service flow should be based on measurable indicators that have policy and management relevance. Indicators must also be able to represent cause-effect relations. For hydrological ecosystem services, selecting single indicators of service capacity that meet the above requirements and that sufficiently reflect ecosystem condition and their potential to provide service flows is difficult. This is because of the non-linear complex interactions among several ecohydrological processes that each relies on a suite of ecosystem components (*van Oudenhoven et al.*, 2012; *Villamagna et al.*, 2013). In this study, the service capacity indicators of crop water supply and household water supply meet the above requirements. For example, *Ennaanay* (2006) and *Yan et al.* (2013) reported that changes in land use and other ecosystem components alter the hydrological cycle, affecting patterns of evapotranspiration, infiltration, water retention, groundwater recharge and water yield. However, for services such as water purification and soil erosion control, the capacity indicators presented in this study are derived indicators and not actual physical processes. Such indicators do not convey information regarding key physical processes and therefore may not have management relevance. In such cases, a key question that arises is if the underlying ecosystem components and processes should be weighted and aggregated to produce one composite indicator for service capacity (*Edens and Hein*, 2013). For example, soil erosion control is a function of surface runoff, slope, soil erodibility, cover and management factors, and support practice factors. Weighing and aggregation of ecosystem components and processes to establish a composite indicator for service capacity, however, is not straightforward and is challenging (*Weber*, 2007; *Stoneham et al.*, 2012).

2.5.3 IMPLICATIONS FOR WATERSHED AND ECOSYSTEM MANAGEMENT

Three of the key issues critical for watershed management and land use planning in an agricultural watershed, such as the Upper Ouémé are nitrate leaching, non-point source pollution and alteration in streamflow regime. Nitrate leaching contaminates groundwater resources (*Jarvis*, 2000; *Jahangir et al.*, 2012). Agricultural non-point source pollution leads to pollution of river networks (*Agnew et al.*, 2006). Alteration of streamflow regime affects riverine ecological integrity and downstream water availability (*Carlisle et al.*, 2011). Ecosystem accounting and spatial characterization of hydrological ecosystem services capacity and flow provide relevant information to help address these issues in policy-making. Such analyses can reveal high-risk areas (i.e., areas that would be affected from changes or continued trends in watershed ecohydrology) or high service production areas (i.e., areas that are crucial for maintaining water flow downstream). For example, our analyses reveal areas where the ecohydrological conditions required for denitrification do not occur, but where there is currently groundwater extraction. These areas are high-risk areas of groundwater contamination from nitrate leaching. More crucially, there is currently crop cultivation in some of these areas. Agricultural intensification in these areas, therefore, will result in higher nitrate leaching and contamination of groundwater resources

Furthermore, the grid-based setup of the SWAT Landscape model enabled us to identify HSAs at a finer spatial resolution. Characterization of the spatiotemporal dynamics of HSAs is essential in controlling non-point source pollution and in maintaining streamflow regime. Hydrologically Sensitive Areas have significant impact on key ecohydrological processes affecting interaction and transport of water, sediment, nutrients and pollutants. They also provide key landscape controls on riparian ecosystem integrity, including aquatic flora and fauna and downstream water availability and quality. Agricultural intensification in HSAs has a higher potential of generating agricultural non-point source pollution (*Agnew et al.*, 2006). Land use change in these areas can have a more significant impact on the streamflow regime. Such analyses can form the basis for establishing Payment for Ecosystem Services schemes (PES) (*Pagiola and Platais*, 2007; *Turpie et al.*, 2008). Watershed PES provides financial support to ecosystem management in high service production areas that are of particular relevance downstream (*Lopa et al.*, 2012; *Lu and He*, 2014). We acknowledge that detailed ecohydrological modelling is only one of the considerations in establishing a watershed PES. Other considerations include transaction costs and the ability to pay of downstream water users. However, ecohydrological modelling can be used to support watershed PES schemes by providing a tool for upstream water managers to monitor the provision of hydrological ecosystem services or by identifying high service production areas that are potentially relevant for a new PES.

2.6 CONCLUSION

There are various components involved in ecosystem service delivery that need to be measured in order to better understand the full dynamics of service provision and to devise sustainable management options. Key amongst these components is service capacity and service flow. Empirical distinction of service capacity and service flow of ecosystem services is a distinguishing feature of ecosystem accounting and is the basis for monetary accounting. In the case-study area, we have shown that despite the non-linear complex interactions among several ecohydrological processes, it is empirically feasible to distinguish between service capacity and service flow of hydrological ecosystem services. This requires appropriate decisions regarding physical and mathematical representation of ecohydrological processes, spatial heterogeneity of ecosystems, temporal resolution, and required model accuracy. The service flows we modelled are the contributions in time and space of ecosystems to productive and consumptive human activities leading to human benefits, whereas the service capacities we modelled reflect ecosystem condition and extent at a point in time, and the resulting potential to provide service flows. We demonstrated our approach by using a SWAT model, which has been configured with a grid-based landscape discretization and further enhanced to simulate water flow across the discretized landscape units, to map and quantify four hydrological ecosystem services vital to food and water security in the Upper Ouémé watershed in Benin. We set up ecosystem accounting tables for both service capacity and service flow and analysed trends in service capacities. For each hydrological ecosystem service, we were able to identify Subwatershed Ecosystem Accounting Units (SEAUs) where either service capacity or service flow is concentrated. We were also able to identify trends in changes in service capacity of hydrological ecosystem services for some SEAUs. Our approach can be extended and applied to other watersheds because it is based on the SWAT model, which has been tested extensively in different watersheds and landscapes. Our analyses show that integrating hydrological ecosystem services in an ecosystem accounting framework provide relevant information on watershed ecosystems and hydrological ecosystem services at appropriate scales suitable for decision-making.

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CHAPTER 3

MODELLING THE FOREST AND WOODLAND-IRRIGATION NEXUS

ABSTRACT

Major increases in food production are needed to feed the rapidly growing population of sub-Saharan Africa. Increased adoption of irrigated agriculture has often been identified as one of the main pathways to agricultural intensification. However, water flows, in particular during the dry season, often depend upon the water regulation services provided by forests and woodlands which are increasingly subject to land conversion as well as degradation from the over-exploitation of wood resources. Insight in the trade-off between land conversion in sub-Saharan African uplands and sustaining water flows is therefore urgently needed. In this paper, we develop a general modelling approach for analysing the effects of deforestation on the availability of water for irrigation at the watershed level, and we apply the approach to the Upper Ouémé watershed in Benin. We use controlled modelling experiments based on the Soil and Water Assessment Tool (SWAT) in addition to copula functions to quantify surface water availability and irrigation potential under prevailing forest and woodland cover as well as varying forest and woodland extents. We undertake these comparative analyses for two irrigation development scenarios that are defined based on different levels of sustained water flows in the Upper Ouémé river network. Our analyses show that conservation of prevailing forests and woodlands in the Upper Ouémé watershed is needed to allow the development of 80% (15,000 ha) or 71% (20,000 ha) of the irrigation potential in the dry season depending on the scenario. At the prevailing forest and woodland extent, the loss of around 40 ha of forest and woodland area reduces the irrigation potential by 1 ha depending on the scenario. Our irrigation potential calculations are based on the water requirements of rice, which is the most water intensive crop grown in the study area. For other crops, the ratio will be lower (i.e. less forest and woodland area is required to sustain 1 ha of irrigated crop production). The relation between forest and woodland extent and irrigation potential, is, however, not linear, and more hectares of forest and woodland are needed to support 1 ha of irrigated crop production with increasing deforestation. This is relevant for trade-off analysis, where it needs to be noted that the forests and woodlands not only generate water regulation services, but also provide other ecosystem services including fuelwood, timber, opportunities for livestock grazing and carbon sequestration.

This chapter is based on:
Duku, C., Zwart, S.J., Hein, L., 2016. Modelling the forest and woodland-irrigation nexus in tropical Africa: a case study in Benin. *Agriculture, Ecosystems & Environment* 230, 105.

3.1 INTRODUCTION

Food insecurity is a major problem in sub-Saharan Africa. Still, one in every four persons in this region is undernourished and agricultural productivity remains the lowest in the world (*FAO et al.*, 2015). With the population of sub-Saharan Africa estimated to double by 2050 compared to 2015 estimates (an increase of 1.2 billion people), there is an urgent need to increase food production (*UN*, 2015b). Current productivity levels for major food crops are inadequate to meet projected demand (*Alexandratos and Bruinsma*, 2012). Improving food security requires the sustainable management of many production factors such as water availability, soils, nutrients, crop health etc. The limited use of irrigation in particular has been consistently cited as a major factor for the low productivity levels (*Burney et al.*, 2010; *You et al.*, 2011; *Xie et al.*, 2014). Currently, about 97% of cropland area in sub-Saharan Africa is rainfed with less than 4% irrigated (*You et al.*, 2011). Investments in irrigation are regarded as a potential means to improve food security and as a strategy for poverty reduction in sub-Saharan Africa (*Dillon*, 2008; *FAO*, 2008; *You et al.*, 2011; *Nkhata*, 2014; *Xie et al.*, 2014). For example, in northern Benin irrigated land dedicated to vegetable production significantly increased local vegetable availability resulting in increased consumption and household income especially in the dry season (*Burney et al.*, 2010). Hence, there has been a renewed interest in both small-scale and large-scale irrigation developments in recent years. For example, one of the targets in the Africa Water Vision 2025, adopted by the African Union member states, is to double the size of irrigated area in this region by 2025 (*UNECA et al.*, 2003). An FAO study forecasted that, as a consequence of the ongoing intensification of the agricultural sector, irrigation water withdrawal in sub-Saharan Africa would increase from 96 km^3 to 133 km^3 per annum between 2005 and 2050 (*Alexandratos and Bruinsma*, 2012).

Planning for new irrigation schemes requires an assessment of long-term water availability especially during the critical dry seasons (*Gustard et al.*, 2004). Such an assessment is used to define the potential area that can be irrigated by the supply source. Availability of water for irrigation is influenced by climatic factors; however, anthropogenic influences such as deforestation that alter the water flow regulation service provided by forests and woodlands can also be a major factor. For example, deforestation can reduce soil infiltration capacity and alter the partitioning of precipitation between overland flow and soil infiltration (and subsequently groundwater recharge) (*Costa et al.*, 2003; *Giertz and Diekkruger*, 2003; *Giertz et al.*, 2005; *Hess et al.*, 2010; *Schilling et al.*, 2014). In addition, there is considerable evidence that seasonal availability of water (especially in the tropics where there is a distinct wet and dry season) can be affected by deforestation leading to increased peak flows in the wet season that can lead to flooding (*Bradshaw et al.*, 2007; *McCartney et al.*, 2013; *Tan-Soo et al.*, 2014); and reduced base-flow in the dry season that can lead to streamflow droughts (*Guo et al.*, 2000; *Pattanayak and Kramer*, 2001; *Tallaksen and van Lanen*, 2004; *Simonit and Perrings*, 2013). In general, in tropical regions with seasonal rainfall, the distribution of streamflow throughout the year is of greater importance to food and water security than total annual water yield (*Bruijnzeel*, 2004). Currently, the rate of tropical deforestation especially in lower income countries remains significant (*Sloan and Sayer*, 2015) despite the reduction in the rate of net global deforestation over the past 25 years (1990 to 2015) (*FAO*, 2015b). For example, between 2010 and 2015 there was a 1.1% (50,000 ha per annum) decrease in forest area in Benin (*FAO*, 2015b). The understanding of linkages and interrelationships among forests and woodlands, water and food production is, therefore, critical to support strategies for food and water security while ensuring provision of multiple forest ecosystem services. Although there is a number of studies that connect deforestation to water flows (e.g. *TEEB*, 2010; *McCartney et al.*, 2013; *Simonit and Perrings*, 2013; *Brookhuis and Hein*, 2016) to date there are very few studies that explicitly link deforestation to irrigation potential (e.g. *Tiwari*, 2000; *UNEP*, 2012).

In this study, we conduct a quantitative analysis of the linkages and relationship between forest and woodland conservation, surface water availability for irrigation and irrigation potential. We use the Upper Ouémé watershed in Benin as a case study. We use controlled modelling experiments based on the Soil and Water Assessment Tool (SWAT) to quantify surface water availability and irrigation potential under prevailing forest and woodland cover as well as a series of hypothetical deforestation states. We undertake these comparative analyses for two irrigation development scenarios that are defined based on different levels of riverine ecological water requirements. The novelty of this study lies first in its explicit quantification of the linkages between the water flow regulation service of forests and woodlands, and irrigation potential. This approach is policy relevant and connects forest and woodland conservation to agricultural production. Another novelty is that this study applies copula functions to characterize and quantify the water flow regulation services of forests and woodlands that underpin water availability for irrigation in the dry season. By using copula functions, we are able to quantify the risk deforestation may pose on this ecosystem service.

3.2 METHODOLOGY

3.2.1 CASE STUDY AREA

The Upper Ouémé watershed (Figure 3.1) is located in central Benin covering an area of approximately 14,500 km². The natural vegetation is a mosaic of woodland savannah and forest islands. Forest and woodland cover make up approximately 55% (7,500 km²) of the Upper Ouémé watershed. The major forest types in the area are tropical forest with more than 75% canopy cover (200 km²) and dry forest with canopy cover of between 50 and 75% (1,200 km²) whereas woodlands have a canopy cover of between 25 and 50% (6,100 km²) (*Giertz et al.*, 2005). Protected forest and woodland areas make up 2,500 km² of the total forest and woodland area in the watershed. The main river channel of the Upper Ouémé river network passes through these protected areas. The study area belongs to the tropical savannah zone with an average annual precipitation (2000 to 2012) of 1200 mm per annum. The area has a unimodal wet season and the driest months (precipitation less than 60mm) are from November to March (Table 3.1).

Smallholder rainfed agriculture is the main economic activity. Maize, rice, yam, cassava and millet are some of the important food crops cultivated in this area with cotton being the major cash crop. For maize, planting is normally in April and harvesting in July whereas the planting window of rice is from June to July and the harvesting window from November to December

(*FAO*, 2015a). The planting window for yam, which has a cropping cycle that varies between 150 to 270 days, is from January to March and the harvesting window is from July to October (*FAO*, 2015a). Cassava has a cropping cycle of over a year and is planted in from April to August and harvested in May to November of the next year (*FAO*, 2015a). Millet is generally planted in May or June window and harvested in the August or September (*FAO*, 2015a). Finally, cotton, which is the major cash crop in this area, is planted in the period May to July and harvested in the period October to November (*Judex and Thamm*, 2008). The irrigation sector is poorly developed and the lack of irrigation water during the dry season is a major problem for many farmers (*Giertz et al.*, 2012). Consequently, crop cultivation mainly takes place in the wet season. Rapid population growth including migrants coming from different parts of the country and other neighbouring countries to farm has caused the expansion of agricultural areas and led to both deforestation and increasing scarcity of agricultural land (*Judex and Thamm*, 2008). This has led to increasing soil degradation due to shortening of the fallow period (*Giertz et al.*, 2012). Conversion of forest and woodland areas for crop cultivation is mainly through slash and burn techniques (*Giertz et al.*, 2005). Despite the expansion of agricultural areas, productivity is consistently low.

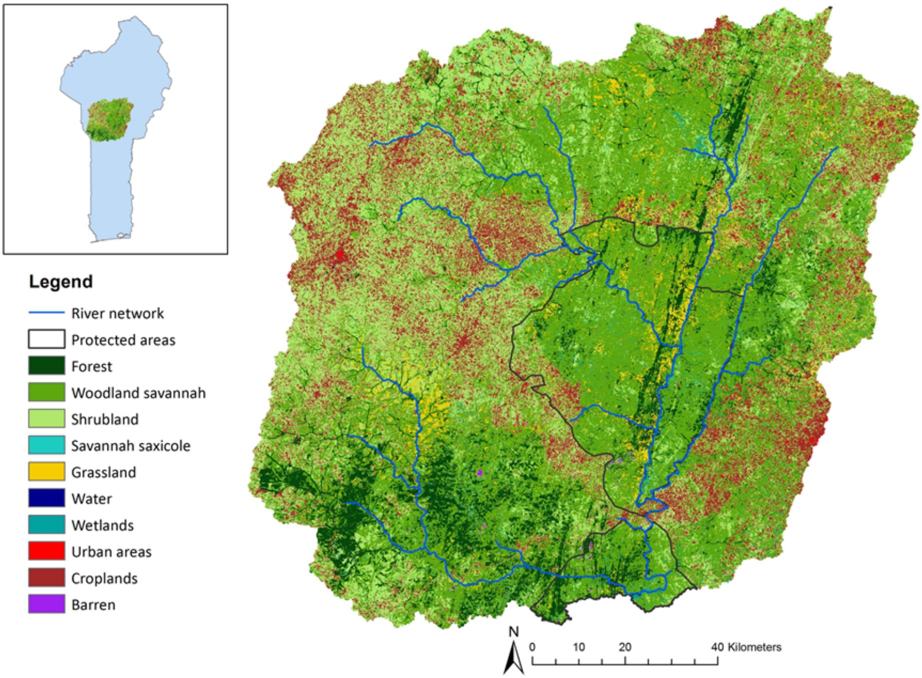


Figure 3.1. Land cover, protected forest and woodland areas and river network of the Upper Ouémé watershed. Land cover data from Judex and Thamm (2008).

Table 3.1 Average monthly total precipitation and reference evapotranspiration (2000 to 2012) in the Upper Ouémé watershed. Reference evapotranspiration was computed with the Hargreaves method (Hargreaves et al., 1985). Identification of dry season was based on Köppen-Geiger climate classification (Peel et al., 2007).

Dry season months	**Precipitation (mm month^{-1})**	**Reference evapotranspiration (mm month^{-1})**
January	4	163
February	10	174
March	17	196
April	69	178
May	121	167
June	157	143
July	223	133
August	262	131
September	247	137
October	100	155
November	10	170
December	3	166

3.2.2 ANALYTICAL FRAMEWORK

MODEL DESCRIPTION

This study builds on the SWAT model which was set up, calibrated and validated in *Duku et al.* (2015). The SWAT model is a physically based, ecohydrological model that simulates the impact of land use and land management practices on water, sediments and agricultural chemicals in large complex watersheds with varying soils, land use and management conditions over a long period of time (*Neitsch et al.*, 2009). In SWAT, a watershed is divided into multiple subwatersheds. Subwatersheds are further subdivided into homogenized units called Hydrologic Response Units (HRU). Each HRU is a lumped area within a subwatershed that is comprised of unique land cover, soil and management combination (*Neitsch et al.*, 2009). The hydrological cycle is divided into two phases. The first division is the land phase which controls the amount of water (and sediment, nutrient and pesticide loadings) to the main channel in each subwatershed. Land phase processes include: weather, hydrology (canopy storage, infiltration, evapotranspiration, surface runoff, lateral subsurface flow, return flow) plant growth, erosion, nutrients and management operations (*Neitsch et al.*, 2009). Surface runoff, lateral flow and return flow from the land phase are then routed through the channel network of the watershed to the outlet in the second phase called the routing phase. This phase also includes processes such as sediment and nutrient routing (*Neitsch et al.*, 2009).

MODEL CONFIGURATION

The SWAT model through its ArcGIS interface, ArcSWAT, was used to simulate watershed ecohydrology including hydrology and plant growth on a daily time-step from 1999 to 2012. The first year served as a warm-up period for the model to assume realistic initial conditions. Input data included a 30m digital elevation model (DEM) obtained from the National Aeronautics and Space Administration (NASA) ASTER Global Digital Elevation Map, which was used to generate stream network, watershed configurations and to estimate topographic parameters. Land cover and soil maps were obtained from the "Integrated Approach to Efficient Management of Scarce Water Resources in West Africa" (IMPETUS) project database (*Judex and Thamm*, 2008). The land cover map had been derived from classification of LANDSAT-7 ETM+ satellite image. The land cover dataset represented the prevailing forest and woodland cover extent as well as other land use and land cover types. Data on protected forest and woodland areas were also obtained from IMPETUS database. For climate data, gridded daily precipitation datasets were obtained from the "African Monsoon and Multidisciplinary Analysis–Coupling the Tropical Atmosphere and the Hydrological Cycle" (AMMA-CATCH) database (*AMMA-CATCH*, 2014) and gridded temperature datasets were obtained from Climate Research Unit (CRU) TS 3.21 database (*Jones and Harris*, 2013). Potential evapotranspiration was computed with the Hargreaves method (*Hargreaves et al.*, 1985) and water transfers for households were modelled as constant extraction rates from shallow aquifers and streams. The Soil Conservation Service curve number approach was used to model surface runoff and daily curve number value was calculated as a function of plant evapotranspiration (*Neitsch et al.*, 2009). A total of 44 river reaches and corresponding subwatersheds of the Upper Ouémé watershed were delineated with ArcSWAT. The SWAT model was calibrated and validated with observed daily streamflow data from 11 monitoring stations within the watershed. These stations had drainage areas of varying spatial scale to capture watershed-scale and subwatershed-scale ecohydrological processes. A split-time calibration and validation technique was carried out using the Sequential Uncertainty Fitting (SUFI-2) optimization algorithm of the SWAT-Calibration and Uncertainty Program (*Abbaspour et al.*, 2008). Calibration was performed using daily discharge measurements from 2001 to 2007 and validation was performed using daily discharge measurements from 2008 to 2011. Detailed information about model set up, calibration and validation results can be found in *Duku et al.* (2015).

HYPOTHETICAL DEFORESTATION STATES

The importance of an ecosystem to water provision and regulation can be assessed by considering the services a modified or a replacement ecosystem would provide (*Brauman et al.*, 2007; *UN et al.*, 2014a). We conducted controlled modelling experiments using the land use update module of the validated SWAT model to analyse the relationship between forest and woodland extent in the Upper Ouémé watershed and dry season irrigation potential. This module allows for spatially explicit (at the HRU level) and dynamic land use change modelling at the watershed scale. The SWAT model setup of the Upper Ouémé watershed included 44 subwatersheds and over 5000 HRUs. Within each HRU the locations and timing of land conversions are determined by SWAT algorithms in line with the deforestation scenario. Deforestation involved the conversion of forest and woodland areas for dryland crop cultivation during the entire simulation period. We used maize as a proxy for crop cultivation because it is the most commonly cultivated crop in the study area. Cleared forest and woodland areas were replaced with this temporary crop during the wet season and followed by a bare soil period that lasted until the next wet season. All other land cover types remained the same as in the prevailing land cover state. In addition to the SWAT model simulation conducted with the prevailing land cover of the Upper Ouémé watershed (*Duku et al.*, 2015), we defined a series of hypothetical deforestation states involving systematic reduction in the forest and woodland extent to 90%, 80%, 70%, 60%, 50%, 40% 30%, 20% 10% and 0% (complete deforestation) at the start of simulation. We also defined another deforestation state where we assumed the conservation of only the existing protected forest and woodland areas with deforestation outside these areas. Each hypothetical deforestation state assumed the defined deforestation state existed at the start of simulation and lasted till the end. We discuss the implications of this assumption in Section 3.4.1. For example, the hypothetical deforestation state involving the conservation of only protected areas represented the conversion of about 5000 km^2 of forests and woodlands outside of protected forest and woodland areas for crop cultivation whereas complete deforestation represented the conversion of all the 7,500 km^2 of forest and woodlands for crop cultivation. For each hypothetical deforestation state, a time-series of daily streamflow for a 13-year period (2000 to 2012) that would have been recorded if that state had existed was generated. To generate the time-series of streamflow in each deforestation state, the parameter sets of the calibrated and validated SWAT model (*Duku et al.*, 2015) remained unchanged and simulation runs were carried out. Our approach allowed for the isolation of the effects of forest and woodland cover on streamflow regime from the effects of climate, and other land cover types.

IRRIGATION DEVELOPMENT SCENARIOS

Maintaining a minimal or target streamflow regime is essential in any assessment of renewable surface water resources for potential irrigation development. The minimal streamflow regime is critical for ensuring adequate water supply for other consumptive purposes (such as household water consumption). In our study area, over 90% of the population depend on groundwater for household consumption (*Judex and Thamm*, 2008). The relatively smaller percentages that depend on surface water obtain them directly from the rivers, small lakes and ponds. A minimal streamflow regime is also critical for sustaining the natural functioning of riverine ecosystems during periods of low flow. This is referred to as riverine ecological water requirements. Riverine ecological water requirement is made up of ecologically relevant low-flow and high-flow requirements (*Smakhtin et al.*, 2004). In this study, we estimated only the low-flow requirement (LFR) because we were interested in changes in low flow associated with changes in forest and woodland cover. During periods of low flow in the dry season, riverine ecosystems are vulnerable to a reduction in the availability of habitat, extremes of water temperature, reduction in dissolved oxygen, deterioration in water quality and habitat fragmentation (*Tallaksen and van Lanen*, 2004).

We defined two irrigation development scenarios i.e., the Q50 and the Q75 scenarios, based on different conditions set for the LFR. The LFR values for the Q50 and Q75 scenarios were determined from flow duration curves of simulated daily streamflow from the year 2000 to 2012. Flow duration curves are cumulative distributions of daily streamflow values that show the relationship between any given streamflow value and the frequency or percentage of time that streamflow value is equalled or exceeded. In the Q50 scenario, the LFR was defined as the streamflow exceeded 50% of the time based on flow duration curves. At the watershed outlet, the LFR in the Q50 scenario was 31.6m^{3}s^{-1}. The Q50 scenario represents a level of irrigation development that allows for adequate residual streamflow needed for acceptable levels of riverine ecological activity as well as water use downstream (*Smakhtin et al.*, 2004). In contrast, the LFR in the Q75 scenario was defined as the streamflow exceeded 75% of the time based on flow duration curves. At the watershed outlet, this value was estimated to be 5.5m^{3}s^{-1}. This scenario prioritises irrigation water withdrawal over environmental sustainability (*Smakhtin et al.*, 2004). We computed the LFR for all 44 subwatersheds of our study area. We used daily simulated streamflow (2000 to 2012) from the validated SWAT model for the computation. In both irrigation development scenarios, the volume of water extracted for household water consumption was the same. This was because it was assumed that extraction of water for household use is prioritised over irrigation water use. Note that the computation of household water consumption was based on observed data (as will be described in Section 3.2.3).

3.2.3 CHARACTERIZING WATER AVAILABILITY FOR IRRIGATION

We characterized surface water availability for irrigation in the dry season in terms of volume, duration and frequency. The volume of water available for irrigation was defined as the amount of streamflow in excess of LFR and household consumption (Figure 3.2); duration was defined as uninterrupted periods (consecutive days) in the dry season with streamflow in excess of LFR (Figure 3.2); and frequency was defined as the recurrence interval for any given duration and corresponding volume of water. In estimating the volume of water available for irrigation, water transfers for household consumption were computed as constant extraction rates from river reaches (and shallow aquifers) in each subwatershed during SWAT model simulation. We used data on water consumption per capita, village population and water access for about 200 communities within the watershed. These data had been extracted from the 2002 national census (*INSAE*, 2003) and from household surveys in the study area (*Hadjer et al.*, 2005).

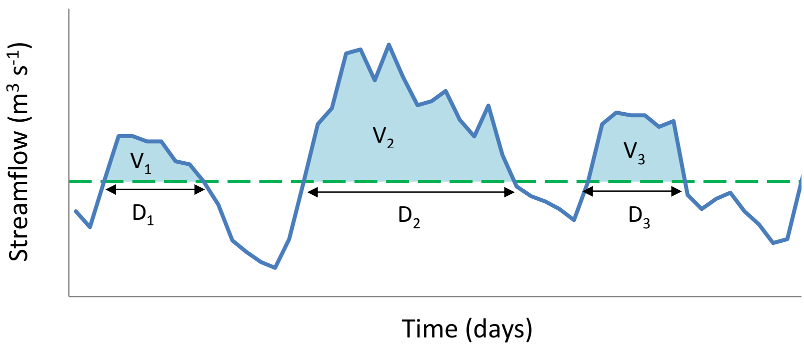


Figure 3.2. An Illustration of the definitions of volume and duration of streamflow available for irrigation; V_1, V_2 and V_3 are the volumes of streamflow in excess of a low-flow requirement (LFR) for the durations D_1, D_2 and D_3 respectively.

To characterize water availability in terms of these three attributes, we first defined the hydrological year as the period from 1 November to 31 October (i.e. the onset of dry season to the end of the wet season). Dry season streamflow (1 November to 31 March) were then extracted from each of the time-series of daily streamflow simulated with the prevailing forest and woodland cover as well as the series of hypothetical deforestation states. From the dry season streamflow, sequences of duration (i.e. D_1, D_2, D_3 etc. as shown in Fig. 2) and associated volume of water available (i.e. V_1, V_2, V_3 etc.) were identified. For downstream river reaches, the volume of water available for irrigation was computed based on residual streamflow after upstream river abstractions for irrigation. The sequences of duration and associated volume were each fitted to a range of marginal probability distributions (e.g. logistic, lognormal, normal, exponential etc.). The parameters of the distributions were estimated using the maximum likelihood method and the goodness of fit of each marginal distribution was evaluated based on Kolmogorov-Smirnov one sample test at the 5% significance level ($P < 0.05$). The results are shown in Table A1 in Appendix I. The best fitting distributions were used for further analysis. To model the bivariate joint cumulative distribution of duration and volume of water available for irrigation, we used copulas (*Sklar*, 1959). Copulas are functions that join univariate distribution functions to form multivariate distribution functions. Copulas are able to model the dependence structure among random variables independently of the marginal distributions. We fitted the marginal distributions to a series of copula functions namely Gumbel, Frank and Clayton. The Akaike Information Criteria (AIC) (*Akaike*, 1974; *Fang et al.*, 2014) were used to evaluate the goodness of fit of each copula. The AIC is a measure of the relative quality of statistical models for a given dataset. The results of the copula fitting are presented in Table A2 in the Appendix I. Based on this test, the Clayton copula was the most suitable and was used to model the joint cumulative distribution function of duration and volume of water availability. Random events were generated with the fitted copula function and Eq. (3.1) was used to compute the frequency for any given duration and associated volume of water availability.

$$\begin{aligned} T_r &= \frac{1}{P\,(V \geq v, and\ D \geq d)} \\ &= \frac{1}{1 - F_V(v) - F_D(d) + C[F_V(v),\ F_D(d)]} \end{aligned} \tag{3.1}$$

where T_r is frequency (years); *P* is probability; *V* is the random variable volume of water available for irrigation, *v* is any value of *V*; *D* is the random variable duration of uninterrupted streamflow in excess of LFR; *d* is any value of *D*; $F_V(v)$ and $F_D(d)$ are the univariate cumulative distribution functions (CDFs) of volume and duration respectively; $C[F_V(v), F_D(d)]$ is the copula function for computing the joint CDF.

3.2.4 ESTIMATING IRRIGATION POTENTIAL

Irrigation potential is a function of a wide variety of physical and socioeconomic factors including water availability, topography, soil suitability, proximity to water source, market access, financial capital, population density etc. (*You et al.*, 2011; *Xie et al.*, 2014). In this study, however, we estimated the irrigation potential from a water resources perspective, i.e., assuming surface water availability was the major controlling factor and all the other factors were non-limiting for implementation (e.g. *Altchenko and Villholth*, 2015). We discuss the implications of this assumption on the results in Section 3.4.1. The irrigation potential (in hectare) was estimated for each subwatershed (including both upstream and downstream subwatersheds) as the quotient between the volume of water available over the entire dry season and the per-hectare irrigation water requirements. In each subwatershed, streamflow available for irrigation could only be obtained from the river reach of that subwatershed and no other river reach. For downstream subwatersheds, calculation of irrigation potential was based on residual streamflow after irrigation potential had been satisfied in upstream subwatersheds. Irrigation water requirement, which is a function of crop water demand and irrigation efficiency, was estimated using rice as a proxy crop. This allowed for the estimation of the maximum crop water demand in the study area. In addition to vegetables, rice is one of the most irrigated crops in the study area and has the highest crop water demand amongst the cultivated crops (*Allen et al.*, 1998). To obtain the total irrigation water requirement, we applied an efficiency factor to account for delivery and field losses associated with irrigation water supply and application based on Eq. (3.2). We excluded residual soil moisture in our calculations of irrigation water requirement because an analysis of soil moisture content in the dry season (as simulated by the SWAT model) revealed that for a significant portion of the time soil moisture content was below the wilting point.

$$IR_{wr} = (K_c \times ET_p/E_{ff}) \times 10 \tag{3.2}$$

IR_{wr} is the total amount of irrigation water required in the dry season (m^3 ha^{-1} dry season^{-1}); K_c is the average crop coefficient value of rice (1.05) (*Allen et al.*, 1998); ET_p is reference evapotranspiration (mm). E_{ff} is irrigation efficiency factor (assumed to be 0.45) (*FAO*, 1997).

$$\mathrm{SWIP} = V_t/IR_{wr} \tag{3.3}$$

where SWIP is surface water irrigation potential (ha dry season^{-1}), V_t is the volume of water available for irrigation during the dry season (m^3 dry season^{-1}), IR_{wr} is the irrigation water requirement (m^3 ha^{-1}).

3.3 RESULTS

3.3.1 FOREST AND WOODLAND CONSERVATION AND WATER AVAILABILITY FOR IRRIGATION

The contour plots of volume, duration and frequency of water available for irrigation in the dry season for different forest and woodland extents are shown in Figure 3.3 and Figure 3.4. They show the correlation between the extent of forest and woodland cover and all the three attributes of water availability for irrigation in the dry season i.e. duration, volume and frequency. The results are from computations using daily simulated streamflow at the watershed outlet. At the start of simulation, the prevailing forest and woodland cover had an extent of 7,500 km^2 representing 55% of the watershed area whereas the protected forest and woodland areas had an extent of 2,500 km² representing 17% of the watershed area. The complete deforestation state had no standing forest and woodland cover. The effect of deforestation is observed in all three attributes of water availability. The duration and volume of water available for irrigation in the dry season decrease in response to decrease in forest and woodland extent. In addition, deforestation reduces the frequency of occurrence or reliability of water flows for irrigation in the dry season. For example, Figure 3.3 (Q50 scenario) shows that under the prevailing forest and woodland cover, a volume of 3 million m^3 day^{-1} of water available for irrigation for a duration of 60 days is likely to occur approximately once every 3 years. However, under conservation of only protected areas and complete deforestation, the frequency for the same volume and duration of water available for irrigation reduces to approximately once every 13 and 15 years respectively. Figure 3.4 shows that the effect of deforestation on water availability is less severe in the Q75 scenario. In this scenario, the frequency under prevailing forest and woodland cover for the same volume and duration of water is less than 2 years whereas under conservation of only protected areas and complete deforestation, it is 4 and 6 years respectively. This, however, represents a trade-off in terms of residual downstream flow and riverine ecological water requirements on one hand, and water availability for irrigation on the other hand. Whereas in the Q50 scenario the estimated LFR at the watershed outlet was 31.6m^3s^{-1}, in the Q75 scenario the estimated LFR was 5.5m^3s^{-1}. In general, the greater the forest and woodland loss the more unreliable water flows for irrigation in the dry season.

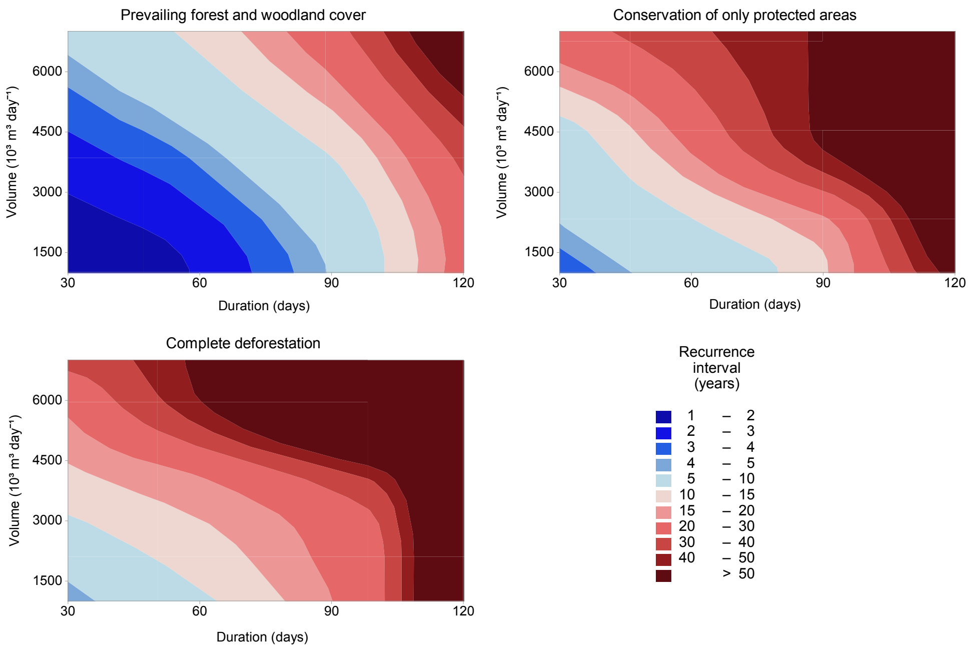


Figure 3.3. Contour plots of volume, duration and frequency of surface water available for irrigation in the dry season under the Q50 scenario. The contour plots were estimated using dry season daily streamflow (2000 to 2012) simulated at the watershed outlet.

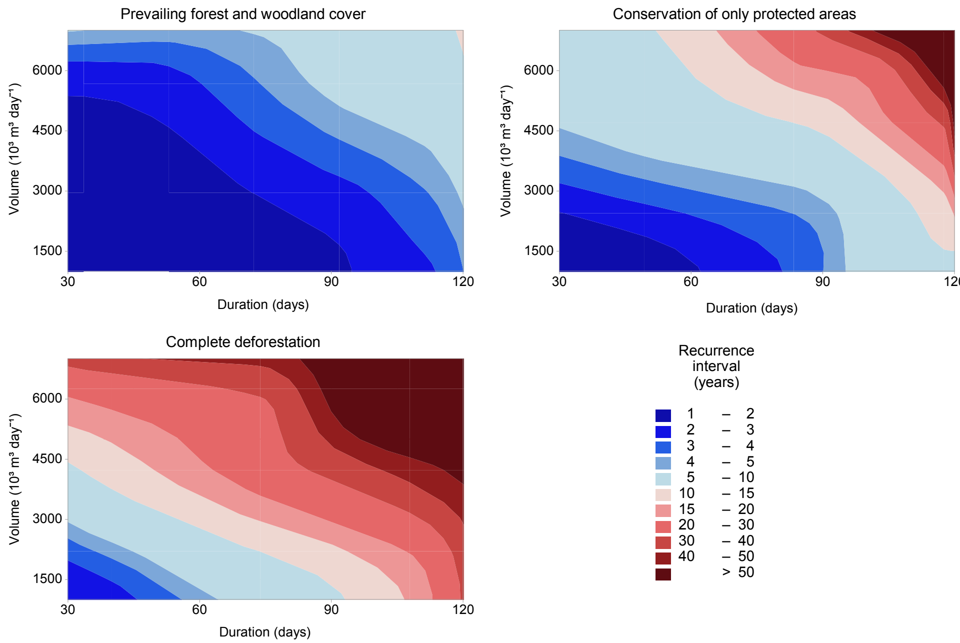


Figure 3.4. Contour plots of volume, duration and frequency of surface water available for irrigation in the dry season under the Q75 scenario. The contour plots were estimated using dry season daily streamflow (2000 to 2012) simulated at the watershed outlet.

3.3.2 IRRIGATION POTENTIAL IN THE DRY SEASON

Table 3.2 shows the total irrigation potential of the Upper Ouémé watershed under prevailing forest and woodland cover as well as under conservation of only protected areas and complete deforestation for different irrigation development scenarios. The results show that conservation of the prevailing extent of forests and woodlands in the watershed is needed for the development of 80% (approximately 15,000 ha) and 71% (approximately 20,000 ha) of the dry season irrigation potential in the Q50 and Q75 scenarios respectively. Conservation of only the existing protected forest and woodland areas in the watershed with deforestation outside of these areas will generate surface water in the dry season for the development of only 18% of the irrigation potential in both scenarios. This represents approximately 3,300 ha and 5,000 ha in the Q50 and Q75 scenarios respectively. Our analyses also show that with conservation of 90% or more of the prevailing forest and woodland cover, every 42 ha and 36 ha of standing forest and woodland area can support 1ha of irrigated rice agriculture in the Q50 and Q75 scenarios respectively (Figure 3.5). Deforestation, however, alters this dynamic significantly. If there should be a loss of 50% or more of the prevailing forest and woodland cover, the forest and woodland area per hectare of irrigation potential increases significantly to a range of 51 to 75 ha in the Q50 scenario and 37 to 45 ha in the Q75 scenario (Figure 3.5).

Table 3.2. Total irrigation potential of the Upper Ouémé watershed in the dry season under different scenarios computed from streamflow at the watershed outlet

Forest and woodland extent	**Irrigation development scenarios**	
	Q50	Q75
Prevailing forest and woodland cover	18,452	28,138
Conservation of only protected forest and woodland areas	6,932	13,234
Complete deforestation	3,672	8,225

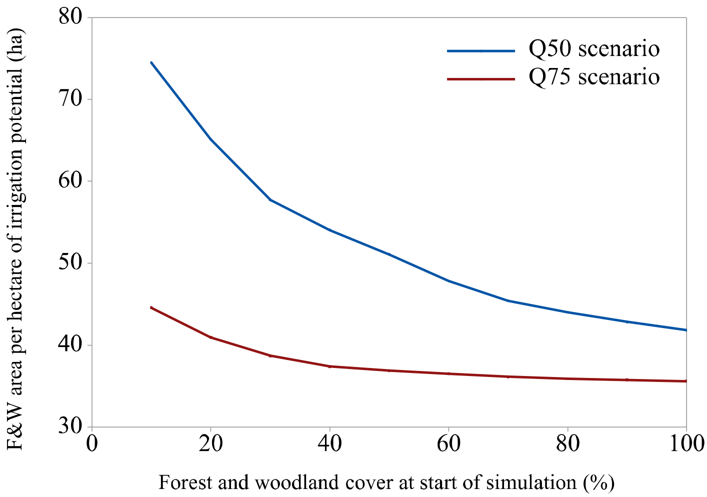


Figure 3.5. *Graph showing the forest and woodland (F&W) area per hectare of dry season irrigation potential in the Upper Ouémé watershed at varying levels of total watershed forest and woodland extent. The graphs are for two irrigation development scenarios, i.e. Q50 and Q75, defined based on different levels of riverine ecological water requirements. Each point on the curve represents the maximum forest and woodland cover (% of the prevailing forest and woodland cover) on the horizontal axis that can produce the specific response on the vertical axis.*

The disaggregated irrigation potential for the 44 subwatersheds under the two irrigation development scenarios is shown in Figure 3.6 and Figure 3.7. For downstream subwatersheds, the irrigation potentials are the residual potential after upstream abstraction of water for irrigation. Generally, first order subwatersheds recorded higher irrigation potentials than second and third order subwatersheds. In all the subwatersheds, irrigation potential was higher with prevailing forest and woodland cover than with partial and complete deforestation. Even in some subwatersheds where the forest and woodland extent under prevailing cover and conservation of only protected areas were the same, the irrigation potential was higher under prevailing forest and woodland cover. This could be attributed to the fact that most of the protected forest and woodland areas are in second and third order subwatersheds. Therefore upstream deforestation affected the amount of inflows into these subwatersheds. In addition, in some subwatersheds, the irrigation potential was the same under conservation of only protected areas and complete deforestation states. This also could be attributed to the fact that these subwatersheds were first order subwatersheds with no protected forest and woodland areas.

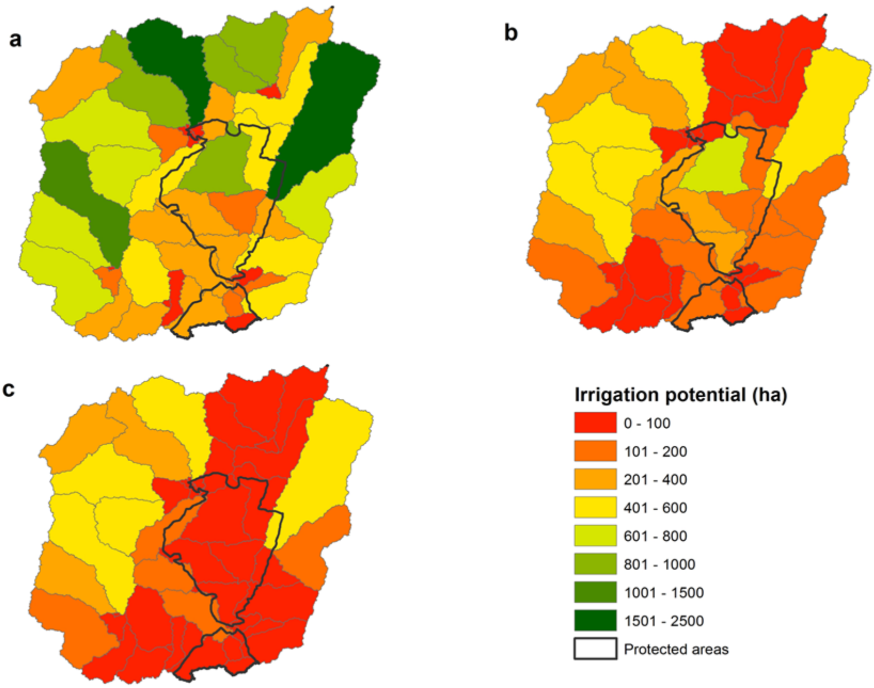


Figure 3.6. Maps of dry season irrigation potential of the 44 subwatersheds of the Upper Ouémé under the Q50 scenario. Map a is irrigation potential under prevailing forest and woodland cover i.e. both protected forest and woodland areas and outside these designated areas; b is irrigation potential under conservation of only protected forest and woodland areas; and c is irrigation potential under complete deforestation. For downstream subwatersheds, the irrigation potentials presented are based on residual streamflow after upstream irrigation potential had been satisfied.

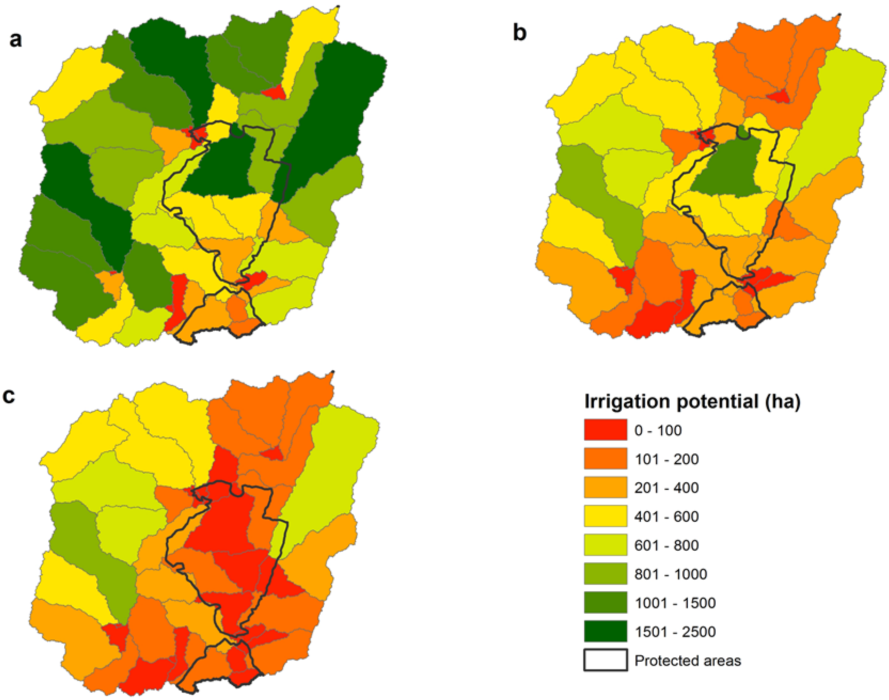


Figure 3.7. Maps of dry season irrigation potential of the 44 subwatersheds of the Upper Ouémé watershed under the Q75 scenario. Map a is irrigation potential with prevailing forest and woodland cover i.e. both protected forest and woodland areas and outside these designated areas; b is irrigation potential under conservation of only protected forest and woodland areas and; c is irrigation potential under complete deforestation. For downstream subwatersheds, the irrigation potentials presented are based on residual streamflow after upstream irrigation potential had been satisfied.

3.4 DISCUSSION

3.4.1 METHODOLOGICAL UNCERTAINTIES AND LIMITATIONS

The aim of this study was to quantify the contribution of forest and woodland areas to the irrigation potential in the dry season through streamflow augmentation. We quantified this as the difference between the hydrological response of the watershed in the dry season in the presence and absence of forest and woodland cover. The sole purpose of the deforestation states in this study were to serve as reference states based on which contribution of forest and woodland cover to dry season water availability and irrigation potential could be elicited. Therefore, we assumed that by the start of simulation in 1999, all forest and woodland areas had been deforested under the complete deforestation scenarios; and all forest and woodland areas outside of protected forest areas had been deforested under the partial deforestation scenario. We acknowledge that deforestation has a temporal dimension and deforestation rates may change over time. In addition, in practice the mode of deforestation employed in this study represents the worst-case scenario. However, though simplified, our approach is instructive and allowed for the isolation and quantification of the contribution of forest and woodland conservation to dry season streamflow augmentation and subsequent irrigation potential. The effects of other land cover types and climate are thus eliminated. The effects of residual forest and woodland cover that would have arisen by simulating deforestation as a dynamic process over the period of simulation are also eliminated.

Irrigation potential is a function of several physical and socioeconomic factors including water availability, water accessibility (i.e. distance to surface water), soil and land suitability (including topography and soil types), market access etc. In this study, we assumed that water availability was the only limiting factor and all the other factors were non-limiting. We estimated irrigation potential based only on water availability. This approach has two implications on the results reported in this study. First, the irrigation potentials reported in this study are the maximum attainable in the dry season. In practice, the other factors listed above will constrain the irrigation potentials to relatively lower values. The second implication is that the dynamics of subwatershed irrigation potential as shown in Figure 3.6 and Figure 3.7 may change if the other factors are taken into consideration. Subwatersheds with higher irrigation potentials may not necessarily have higher suitability in terms of soil, topographic and other socioeconomic factors suitable for irrigation development. Another source of uncertainty in this study is the irrigation efficiency. Irrigation efficiency varies with soil types and water management techniques. However, we used a single irrigation efficiency factor across the whole study area.

3.4.2 POLICY AND MANAGEMENT IMPLICATIONS

First and foremost, our study shows that agricultural intensification in sub-Saharan Africa based on expanding irrigated crop production requires the protection of forest and woodland cover in the headwaters of the rivers feeding the irrigation schemes. If the forests and woodlands in Upper Ouémé watershed were to completely disappear, the irrigation potential is reduced by 70 to 80% depending upon the scenario (Table 3.2). In addition, our analysis quantitatively shows that water flows become less reliable if the forests and woodlands disappear, and that periods of drought (in terms of low availability of irrigation water) are more frequent when headwaters become deforested.

We believe our approach involving the estimation of volume, duration and frequency of surface water available for irrigation using copula functions offers a suitable approach to connect the water regulating service of forests and woodlands to irrigated agricultural potential. Our study is, however, not able to specify which land use option or scenario is to be preferred, given the trade-offs involved between converting forests and woodlands to (temporary) cropland and the establishment of irrigated agriculture. This would require an analysis of the marginal returns on land, water, labour and capital investments for the different types of irrigated and non-irrigated crop production, which we intend to do in a follow-up paper. Among others, such an analysis needs to consider the different types of irrigated crops. We would like to emphasize that we analysed irrigated water demand for rice, which is the most water-intensive irrigated crop in

West Africa (*Schmitter et al.*, 2015). One of the considerations in a follow-up analysis needs to be what crops are of importance for food security from an economic perspective. Note also that the selection of the irrigated crop influences the ratio between irrigated crop and forest and woodland cover acreage. For crops requiring less water than rice a lower number of hectares of forest and woodland is required on a per hectare basis. Another agronomical consideration is that deforestation in the Upper Ouémé watershed is caused to a large extent by shifting cultivation (*Giertz et al.*, 2005), which is a low-input-low-output type of agriculture with very limited returns to land and labour compared to other forms of agriculture. In addition, decreasing fallow periods in the watershed over time have led to concerns about soil degradation (*Giertz et al.*, 2012) which means that future returns on shifting cultivation will be even lower than present day returns.

A full cost benefit analysis of land use options will also have to include other ecosystem services provided by forest and woodlands, as well as by the land that is being replaced by irrigated agriculture. Other ecosystem services provided by forests and woodlands include soil erosion control important for soil fertility and water quality; groundwater recharge important for household water consumption (*Duku et al.*, 2015), supply of timber, fuelwood and non-timber forest products, carbon sequestration and providing opportunities for livestock grazing. For example, in our study area, over 90 % of the population depend on groundwater for drinking and general household use, and the large majority of households depend on fuelwood as the main source of energy for cooking. In addition, forests are important for maintaining evaporation and rainfall patterns in inland parts of the continent hence deforestation may also evoke lower rainfall in more inland parts of Benin and neighbouring countries. Finally, climate change poses serious questions on the already limited productivity of rainfed agriculture/slash and burn. In sub-Saharan Africa, yields from rainfed agriculture are projected to decrease as a result of climate change (*World Bank*, 2013). Adaptation measures such as irrigation will be needed to reduce these impacts.

We believe that our general modelling approach – and the link between deforestation and reduced opportunities for irrigation - will apply to a substantial part of sub-Saharan Africa, in particular the sub-humid maize growing belt with seasonal water shortages, soils with generally low inherent fertility, and undulating terrain. Across sub-Saharan Africa, the maize belt covers around 2.5 million km^2 (*Dixon et al.*, 2001) and dry forests and woodlands such as those that characterize the Upper Ouémé watershed cover an approximate area of 17.3 million km^2 (*Chidumayo and Gumbo*, 2010). However, the specific relation between deforestation and irrigation water availability will vary with the watershed as a function of size and shape of the watershed, specific rainfall patterns, etc. A particular concern is the non-linearity of the system, where the availability of water responds differently to an additional unit of deforestation as a function of remaining forest and woodland cover. Hence, past experience may not provide good insights on future hydrological changes that are bound to occur with ongoing deforestation in sub-Saharan Africa.

3.5 CONCLUSION

In this study, we analysed the relation and linkages between forest and woodland cover and dry season water availability for irrigation development. We quantified the contribution of forest and woodland cover to the irrigation potential of the watershed in the dry season. We used controlled modelling experiments based on the SWAT model and quantified the irrigation potential under prevailing forest and woodland cover and a series of hypothetical deforestation states. We did this for two irrigation development scenarios defined based on riverine ecological water requirements. Our analyses show that conservation of forests and woodlands in the Upper Ouémé watershed is needed to allow the development of 80% (15,000 ha) or 71% (20,000 ha) of the irrigation potential in the dry season depending on the scenario. Conservation of only the existing protected forest and woodland areas with deforestation outside these zones will only generate surface water in the dry season for the development of 18% of the current irrigation potential. At the current forest and woodland cover, the loss of between 36 and 42 ha of forest and woodland area reduces the irrigation potential by 1 ha of irrigated rice production depending on the scenario. For other crops (crops with less water demand), the ratio will be lower (i.e. less forest and woodland area is required to sustain 1 ha of irrigated crop production). It is generally acknowledged that sub-Saharan Africa has sufficient renewable water resources with opportunities for substantial irrigation expansion. However, our study has shown that deforestation can significantly limit these opportunities, in particular the sub-humid maize growing belt with seasonal water shortages. Across these areas, the specific relation between deforestation and irrigation water availability will vary as a function of size and shape of the watershed, specific rainfall patterns, etc. Hence, past experience may not provide good insights on future hydrological changes that are bound to occur with ongoing deforestation in sub-Saharan Africa. Our findings also show non-linearity in the relation between forest and woodland extent and per hectare irrigation potential i.e., more hectares of forest and woodland are needed to support 1 ha of irrigated rice production at lower forest and woodland extents. In general, the findings of this study point to an important trade-off in agricultural intensification, with deforestation freeing up land for rainfed crop production but leading to lower water availability for irrigated agriculture needed to increase productivity. This trade-off needs to be considered in planning for agricultural intensification across the sub-humid zone of sub-Saharan Africa.

ACKNOWLEDGEMENTS: This research was conducted at Wageningen University as part of the "Realizing the agricultural potential of inland valley lowlands in sub-Saharan Africa while maintaining their environmental services" project (RAP-IV). The project is implemented by the Africa Rice Center and its national partners and is funded by the European Commission through the International Fund for Agricultural Development (IFAD). We thank the IMPETUS project and AMMA-CATCH regional observing system in Benin for making data available for this research through their public geoportals. We also thank anonymous reviewers whose in-depth comments have improved this chapter.

CHAPTER 4

DIMINISHING AGRICULTURAL POTENTIAL IN A CHANGING CLIMATE

ABSTRACT

In recent decades, there have been substantial increases in crop production in sub-Saharan Africa (SSA) as a result of higher yields, increased cropping intensity, expansion of irrigated cropping systems, and rainfed cropland expansion. Yet, to date much of the research focus of the impact of climate change on crop production in the coming decades has been on crop yield responses. In this study, we analyse the impact of climate change on the potential for increasing rainfed cropping intensity through sequential cropping and irrigation expansion in central Benin. Our approach combines hydrological modelling and scenario analysis involving two Representative Concentration Pathways (RCPs), two water-use scenarios for the watershed based on the Shared Socioeconomic Pathways (SSPs), and environmental water requirements leading to sustained water flows in the river network. Our analyses show that in Benin, one of the effects of climate change will be that increasing crop production through expansion of sequential cropping in the future will be severely limited. Depending on the climate change scenario between 50% and 95% of cultivated areas that are used for sequential cropping or can support it will revert to single cropping. The results also show that even under a combination of RCP2.6 and SSP1 scenarios, the irrigation potential of the watershed will at least be halved by mid-century. Given the urgent need to increase crop production to meet the demands of a growing population in SSA, our study outlines challenges and trade-offs -and the need for planned development- that need to be overcome to improve food security in the coming decades.

This chapter is based on:
Duku, C., Zwart, S.J., Hein, L., 2017. Diminishing agricultural potential in a changing climate: the case of a sub-humid tropical watershed in Benin. *PLoS ONE* (Submitted)

4.1 INTRODUCTION

Increasing crop production in sub-Saharan Africa (SSA) is urgently needed. The population of the region is projected to double by 2050 compared to 2015 (*UN*, 2015b). About 97% of current cropland area is under rainfed cultivation (*You et al.*, 2011) and current productivity levels for major food crops, which are the lowest in the world, are inadequate to meet projected demand (*Alexandratos and Bruinsma*, 2012). To meet the food demand of a growing population several options for increasing crop production must be harnessed. These include amongst others crop intensification in rainfed systems from higher yields and/or increased cropping intensity, expansion of irrigated area and cropland expansion. Over the past decades, higher yields, increased cropping intensity (i.e. sequential cropping and intercropping) and cropland expansion have accounted for an estimated 38%, 31% and 31% respectively of the recorded increases in crop production in SSA (*Alexandratos and Bruinsma*, 2012).

In the coming decades, these various options for increasing crop production will be affected by climate change. Changes in precipitation and temperature will pose serious risks to crop production systems and food security in general. West Africa in particular has been identified as a regional hotspot of climate change with climate departure from historical variability projected to occur faster than the global average (*Diffenbaugh and Giorgi*, 2012; *IPCC*, 2013; *Mora et al.*, 2013). Several studies have highlighted the vulnerabilities and risks to crop production systems. However, much of the research focus has been on crop yield responses to climate change (e.g. *Sonneveld et al.*, 2011; *Waha et al.*, 2013; *World Bank*, 2013). To date information on the likely impacts of climate change on rainfed cropping intensity in cultivated areas or potential arable land as well as on the irrigation potential in SSA are limited. Focussing on crop yield responses alone result in underestimation of the impact of climate change on crop production (*Cohn et al.*, 2016). Sequential cropping has been one of the ways of increasing cropping intensity (in addition to intercropping) and involves cultivation of two or more crops on the same field after each other or with overlapping growing periods (relay cropping) (*Francis*, 1986). In SSA, the length of growing period in addition to high labour intensity, lack of knowledge and lack of market access are the key constraints to increasing cropping intensity in rainfed systems (*Waha et al.*, 2013). Increased application of irrigation is also regarded as reliable means to guarantee year-round crop production. Investments in irrigation are therefore increasing in SSA and irrigation water withdrawal is expected to increase from 96km^3 (2005 estimate) to 133km^3 per annum by 2050 (*Alexandratos and Bruinsma*, 2012).

To provide a better understanding of the varied impacts of climate change on opportunities for increasing crop production, we examine three potential options for increasing crop production in a large sub-humid tropical watershed in central Benin, the Upper Ouémé watershed. First, we analyse the impact of climate change on the potential for increasing rainfed cropping intensity through sequential cropping in cultivated areas. Second, we analyse the impact of climate change on the suitability of potential arable land areas for rainfed sequential cropping. Finally, we analyse the combined impacts of climate change and socioeconomic development on the potential for irrigation expansion in the watershed taking into account household water demand and riverine environmental water requirements which are the major non-agricultural water uses in the watershed. Our approach combines hydrological modelling and scenario analysis involving two Representative Concentration Pathways (RCPs), two water-use scenarios for the watershed based on the Shared Socioeconomic Pathways (SSPs), and environmental water requirements leading to sustained water flows in the Upper Ouémé river network.

4.2 METHODS

4.2.1 STUDY AREA

The Upper Ouémé watershed in central Benin covers an area of approximately 14,500 km² with an estimated population of about 510,000 people (Figure 4.1) (*Bright et al.*, 2011). It is located in the sub-humid tropical zone and is characterized by unimodal rainfall season from May to October of about 1250mm per annum. In general, Benin is affected by an alteration of cool and humid monsoon air mass (originating from the Gulf of Guinea), and hot, dry and dusty Saharan air mass (*Fink*, 2010). Rainfall anomalies in Benin and West Africa in general have been associated with the northward or southward position of the Inter-Tropical Convergence Zone and the associated low and upper level jet streams (*Fink*, 2010). The natural vegetation of the watershed is a mosaic of savannah woodland and small forest islands. Smallholder rainfed agriculture is the major economic activity. Maize, rice, yam, cassava, sorghum and millet are the most important food crops cultivated in this area, with cotton being the major cash crop. The cropping intensity of these staple crops is 1.5 (*You et al.*, 2014) indicating that a substantial portion of land either devoted to these crops or other crops is harvested twice per annum. The irrigation sector is poorly developed and the lack of irrigation water during the dry season is a major problem for many farmers (*Giertz et al.*, 2012). Pastoral communities from neighbouring countries such as Nigeria often migrate to this study area and for grazing in particular in the dry season when water and food resources are scarce in the less humid zones to the North of the Upper Ouémé (*Judex and Thamm*, 2008).

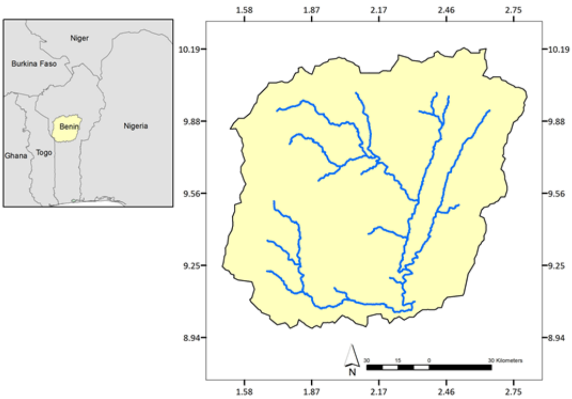


Figure 4.1. The Upper Ouémé river network and watershed in central Benin

4.2.2 RCP AND SSP SCENARIOS

This study builds on the grid-based Soil and Water Assessment Tool (SWAT) landscape model which was set up, calibrated and validated in Duku et al (*Duku et al.*, 2015). Detailed information about the grid-based SWAT landscape model description can be found in Rathjens et al (*Rathjens et al.*, 2015) and Arnold et al (*Arnold et al.*, 2010). The SWAT model in general is a spatially explicit, physical, ecohydrological model that simulates the impact of land use and land management practices on water, sediments and agricultural chemicals in large complex watersheds with varying soils, land use and management conditions over a long period of time (*Gassman et al.*, 2007; *Neitsch et al.*, 2009). For the present study, simulations involving daily time-steps were undertaken from 2003 to 2012. This represented the current climatic conditions henceforth referred to as the baseline conditions.

For simulations under climate-change scenarios, we obtained downscaled multi-model mean climate data of 17 General Circulation Models (GCMs) from the MarkSimGCM geoportal (http://gisweb.ciat.cgiar.org/MarkSimGCM/) (*Jones and Thornton*, 2013). Appendix I provides the list of 17 GCM models used in this study. The MarkSimGCM geoportal is part of the Consultative Group for International Agricultural Research (CGIAR) research program on Climate Change Agriculture and Food Security (CCAFS). A detailed description of the downscaling and bias correction approach can be found in *Jones and Thornton* (2013). The multi-model mean monthly precipitation totals, and maximum and minimum temperature data were obtained for two Representative Concentration Pathways, i.e. RCP2.6 and RCP8.5; and for two time-periods i.e. 2041 – 2050 (2040s) and 2091 – 2100 (2090s). The RCP2.6 scenario is an emission pathway that leads to the lowest concentration levels of atmospheric greenhouse gases (*Moss et al.*, 2010; *IPCC*, 2013). It represents a peak in greenhouse gas emissions before 2100 followed by a consistent decline throughout the rest of this century. It is the pathway needed to realize the targets set during the twenty-first Conference of Parties of the UN Framework Convention on Climate Change, i.e. keep mean global warming to within 2^{o}C above pre-industrial levels. The RCP8.5 scenario, on the other hand, is characterized by increasing greenhouse gas emissions leading to the highest concentration of atmospheric greenhouse gases by the end of this century (*Moss et al.*, 2010; *IPCC*, 2013). It is representative of the business-as-usual scenario i.e. continued increase in greenhouse gas emissions.

In addition to the RCPs, we developed water-use scenarios in line with the qualitative narratives of the Shared Socioeconomic Pathway (SSP) (*Kriegler et al.*, 2012; *O'Neill et al.*, 2013) for the calculation of irrigation potentials. The SSPs are a set of alternative reference assumptions about future socioeconomic development in the absence of climate policies. The SSP1 scenario depicts a development pathway characterised by rapid economic development especially in low-income countries leading to rapid technological development, increased resource use efficiency and low population growth whereas SSP3 represents a scenario with slow economic growth, slow technological development, low resource use efficiency and rapid population growth. In this study, each SSP was characterized by three variables; population growth, irrigation efficiency, and per capita domestic water use (Table 4.1). Estimates of population projections in the watershed under each SSP for different time-periods were computed from (*Jones and O'Neill*, 2016). Data on irrigation efficiency was obtained from Hanasaki et al (*Hanasaki et al.*, 2013) and had been derived from the qualitative narrative of each SSP. Finally, we derived estimates of per capita domestic water use in our study area based on the qualitative narratives of each SSP and data from *FAO* (2011).

To generate the daily time-series of climate data needed to run the SWAT model, we used the change factor approach (e.g. *Diaz-Nieto and Wilby*, 2005; *Anandhi et al.*, 2011; *Luo et al.*, 2013). The change factor approach involves the calculation of the relative changes in monthly precipitation and the absolute changes in temperature between the multi-model GCM data and the baseline climate data for each time period (i.e. 2040s and 2090s) under each RCP scenario (*Neitsch et al.*, 2009). The change factors were calculated separately for each of the 44 subwatersheds of the Upper Ouémé watershed (the SWAT model assigns one climate station to each subwatershed based on proximity analysis) (*Neitsch et al.*, 2009). The calculated changes were used to perturb the baseline climate observations in the calibrated and validated grid-based SWAT landscape model. In simulating watershed hydrology under each RCP scenario, the effect of increased atmospheric carbon dioxide concentration was not taken into account. We discuss the implications of excluding increasing atmospheric carbon dioxide in Section 4.4.1.

Table 4.1. Characterization of Shared Socioeconomic Pathways (SSPs). Current population in the watershed is about 510,000 (Bright et al., 2011). Irrigation efficiency in the baseline was 0.45 (FAO, 1997) and per capita domestic water use was 19 L day^{-1} (INSAE, 2003; Hadjer et al., 2005).

Variable	**2040s**		**2090s**	
	SSP1	**SSP3**	**SSP1**	**SSP3**
Irrigation efficiency[a]	0.52	0.45	0.61	0.45
Population growth (%)[b]	101	156	114	285
Per capita domestic[c] water use (L day^{-1})[c]	60	20	100	30

a Values of irrigation efficiency are based on *Hanasaki et al.* (2013).
b Percentage increase in population with 2010 as the base year. For the 2040s and 2090s decades, the projected population of the years 2050 and 2100 respectively were used.
c Data derived from *FAO* (2011) and the qualitative storyline of each SSP scenario.

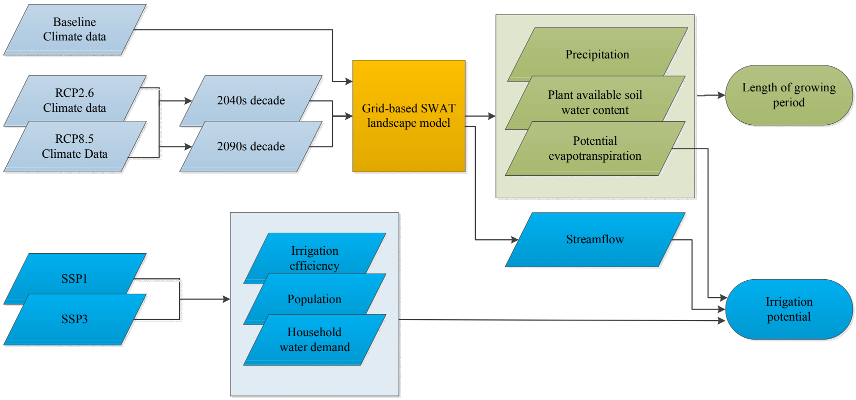


Figure 4.2. Schematic diagram of the approach

4.2.3 MODELLING METEOROLOGICAL DROUGHT

We first characterised meteorological drought conditions under each RCP scenario using the Standardized Precipitation Evapotranspiration Index (SPEI) (*Vicente-Serrano et al.*, 2010). The SPEI methodology estimates the severity and frequency of meteorological drought by accounting for changes in evapotranspiration demand caused by changes in temperature in addition to precipitation. It is based on a probability distribution fitted to a time-series of the difference between monthly precipitation and potential evapotranspiration (i.e. climatic water balance) aggregated over different time-scales using a moving window (e.g. 3-month, 6-month etc.). This probability distribution is transformed to the cumulative distribution function of the standard normal distribution (with a mean of 0 and standard deviation of 1). In this study, the regional (i.e. average over the entire watershed) monthly precipitation and potential evapotranspiration time-series were used. Potential evapotranspiration was computed using the Hargreaves method (*Hargreaves et al.*, 1985). The sequences of climatic water balance in the baseline and for the 2040s and 2090s in each RCP scenario were used to compute the 6-month (seasonal drought) and 12-month (annual drought) SPEIs.

4.2.4 MODELLING LENGTH OF GROWING PERIOD

The length of growing period (LGP) of a spatial unit is indicative of its potential for rainfed crop production. We modelled the LGP as the number of days in a year in which moisture supply (i.e. the sum of daily precipitation and plant available soil water content) was equal to or exceeded potential evapotranspiration and temperature was above 5°C (e.g. *Jones and Thornton*, 2009). Water balance and soil water content were simulated using the SWAT model. Single and sequential cropping zones under each climate scenario were then delineated based on the Agro-Ecological Zone methodology (*IIASA/FAO*, 2012) shown in Table 4.2. Sequential cropping zones delineated in this study do not involve the cultivation of wetland rice.

Table 4.2. Delineation of potential rainfed sequential cropping zones under rainfed conditions (IIASA/FAO, 2012). Sequential cropping zones are disaggregated into relay and double cropping zones

	Length of growing period (days)	Accumulated temperature above 5°C over the growing period	Accumulated temperature above 10°C over the growing period
Single cropping	≥ 120	-	-
Relay cropping	≥ 200	≥ 3200	≥ 2700
Double cropping	≥ 240	≥ 4000	≥ 3200

4.2.5 MODELLING STREAMFLOW DROUGHT AND IRRIGATION POTENTIAL

STREAMFLOW DROUGHT

Streamflow droughts affect the availability of water for irrigation and other consumptive purposes. We used two approaches to compute streamflow droughts; the Standardised Streamflow Index (SSI) (*Hao et al.*, 2014; *Farahmand and AghaKouchak*, 2015) and Severity-Duration-Frequency curves (SDF) (*Shiau et al.*, 2007). The SSI approach is similar to the SPEI methodology, however, in this case a non-parametric probability distribution was fitted to the time-series of monthly streamflow totals simulated at the watershed outlet (*Hao et al.*, 2014; *Farahmand and AghaKouchak*, 2015). Unlike the SSI approach, the SDF approach involved daily streamflow simulated at the watershed outlet and streamflow droughts occurred when streamflow levels were below a specified threshold level for at least a specific period of time (e.g. *Shiau et al.*, 2007; *Van Loon et al.*, 2015). We used the environmental water requirements of the river as the threshold level. The environmental water requirement is critical for sustaining the natural functioning of riverine ecosystems during periods of high flow in the wet season and low flow in the dry season. We defined the environmental water requirements separately for each month as the streamflow value with a 75% exceedance frequency (Q75) (*Smakhtin et al.*, 2004). We estimated these threshold levels based on flow duration curves under baseline climatic conditions. For example, the Q75 value of the month of August was 138m^3s^{-1} whereas the value in November was 22 m^3s^{-1}. Using the monthly variable threshold approach, the seasonality of streamflow is preserved and there is an inherent recognition that streamflow droughts (or negative streamflow anomalies) could occur even during wet seasons. In other words, below the environmental water requirement streamflow was not available for potential consumptive uses. Using the daily streamflow time-series simulated under baseline and RCP scenarios, streamflow drought events were identified based on two criteria. First, a streamflow drought event must be made up of at least seven days of streamflow below a threshold level (*Tallaksen and van Lanen*, 2004). Second, two separate drought events should occur at least five days apart or they were considered as a single drought event. The sequences of identified drought duration and drought severity were each fitted to a range of marginal probability distributions (e.g. Logistic, Normal, Generalized Extreme Value, Gamma etc.). The goodness of fit of each marginal distribution was evaluated based on the negative log likelihood test. Based on this test, the time-series of duration were fitted to the Generalized Extreme Value distribution whereas the associated severity series were fitted to a lognormal distribution. To model the bivariate joint cumulative distribution of duration and severity of streamflow drought events, we used copulas (*Sklar*, 1959). Copulas are a type of functions that join univariate distribution functions to form multivariate distribution functions. We fitted the marginal distributions to a series of copula functions namely Gumbel, Frank, Clayton and Gaussian. The Akaike Information Criteria (AIC) (*Akaike*, 1974; *Fang et al.*, 2014) were used to evaluate the goodness of fit of each copula. The AIC is a measure of the relative quality of statistical models for a given dataset. Based on this test, the Gaussian copula was the most suitable and was used to model the joint cumulative distribution function of duration and severity of streamflow droughts. Random streamflow drought events characterised by severity and duration were generated with the fitted copula function and Eq. (4.1) was used to compute the joint return periods.

$$T_r = \frac{1}{P\ (S \geq s, and\ D \geq d)}$$

$$= \frac{1}{1 - F_S(s) - F_D(d) + C[F_S(s),\ F_D(d)]} \quad (4.1)$$

where T_r is frequency (years); *P* is probability; *S* is the random variable severity of steamflow drought event, *s* is any value of *S*; *D* is the random variable duration of streamflow drought event; *d* is any value of *D*; $F_V(v)$ and $F_D(d)$ are the univariate cumulative distribution functions (CDFs) severity and duration respectively; $C[F_S(s), F_D(d)]$ is the copula function for computing the joint CDF.

IRRIGATION POTENTIAL

We estimated the irrigation potential from a water resources perspective i.e. assuming surface water availability was the major limiting factor (e.g. *Altchenko and Villholth*, 2015; *Duku et al.*, 2016b). We created a matrix of RCPs and SSPs, where each RCP was characterized by the simulated total streamflow in excess of the Q75 threshold level and potential evapotranspiration (crop water demand); and each SSP was characterized by irrigation efficiency, population growth and per capita domestic water use (Table 4.1). Irrigation potential (in hectares) was computed as the quotient between total volume of streamflow available and irrigation water requirement (*Duku et al.*, 2016b). The volume of streamflow available for irrigation was computed using Eq. (4.2). The total irrigation water requirement was computed as the quotient between crop water demand (using rice as a proxy crop) and irrigation efficiency.

$$W_a = [\sum_i (V_t - Q75)] - (D_p \times Pop_t) \quad (4.2)$$

where W_a is total annual volume of streamflow available for irrigation ($m^3\ yr^{-1}$); V_t is total monthly volume of streamflow at month i (m^3); Q75 is environmental flow requirement of month i (m^3); D_p is per capita domestic use ($m^3\ person^{-1}\ yr^{-1}$); Pop_t is total watershed population.

4.3 RESULTS

4.3.1 PRESENT AND PROJECTED PATTERNS OF METEOROLOGICAL DROUGHT

Our SPEI analyses clearly show a substantial shift from relatively wetter climatic conditions under the baseline to increasingly drier climatic conditions under both RCP2.6 and RCP8.5 scenarios (Figure 4.3). The SPEI values are the number of standard deviations by which the anomaly in climatic water balance deviates from the long-term mean. Not only will the probability of seasonal drought (6-month SPEI) increase under both RCP2.6 and RCP8.5 but annual droughts (12-month SPEI) will increase as well.

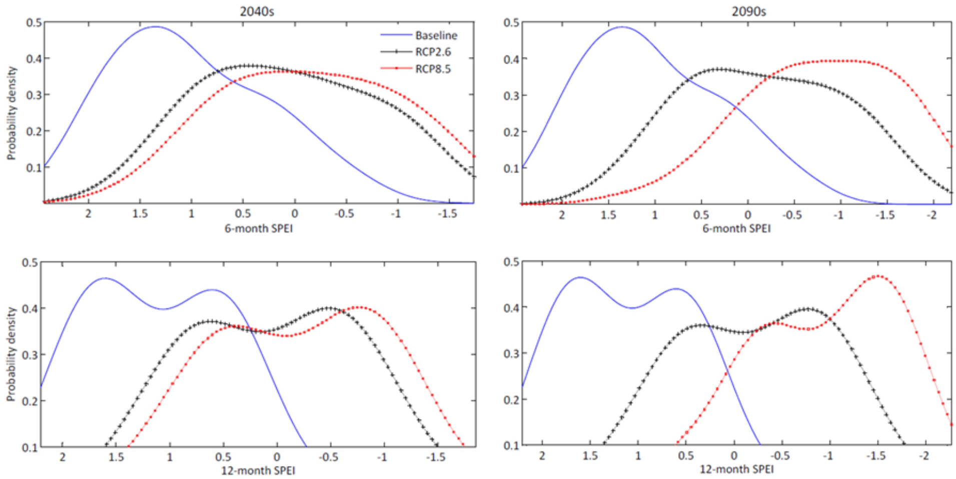


Figure 4.3. Probability density plots of 6-month and 12-month Standardized Precipitation Evapotranspiration Index (SPEI) derived under baseline climate conditions (2003 – 2012) and under two future climate scenarios. SPEI values < 0 represent meteorological drought and the greater the absolute value the higher the severity. SPEI values > 0 represent wet conditions.

4.3.2 IMPACT OF CLIMATE CHANGE ON RAINFED PRODUCTION POTENTIAL

Our analyses show that a watershed-wide average of between 15 and 30 growing days will be lost depending on the climate change scenario. Despite the loss of growing days, cultivated and uncultivated areas that currently are used for single cropping or can support it will still be suitable depending on the type of crop cultivated. However, substantial areas of hitherto rainfed sequential areas will only be suitable for single cropping (Figure 4.4 and Figure 4.5). Between 50% (30,000ha) and 95% (57,000ha) of cultivated areas that are currently used for rainfed sequential cropping or can support it will only be suitable for single cropping depending on the climate change scenario (Figure 4.4 and Figure 4.5). In uncultivated areas, between 10% and 60% of hitherto rainfed sequential areas will revert to single cropping. The effect of the loss of growing days is less in uncultivated areas than in cultivated areas because soils in uncultivated areas (mainly forested) have relatively higher water-holding capacity. Currently over 90% of areas that can support sequential cropping in the watershed, i.e. over 570,000ha, are not under cultivation and lie mainly in the south-western part of the watershed (Figure 4.4). In the coming decades and especially under RCP2.6, a large part of these areas will still be able to support sequential cropping albeit either loss or shortening of the fallow period (Figure 4.5). Adequate soil and nutrient management will then be required to increase productivity.

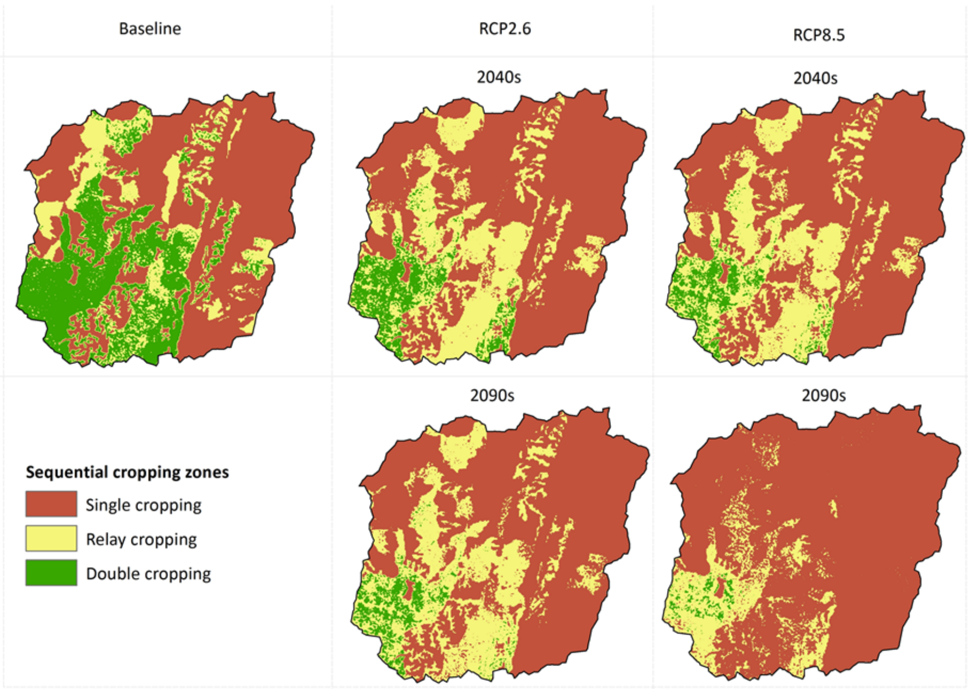


Figure 4.4. The cropping zones in the watershed under baseline climatic conditions and RCP scenarios. Sequential cropping zones have been disaggregated into full double cropping and relay cropping zones. These zones indicate the areas where water availability is sufficient to permit different cropping systems.

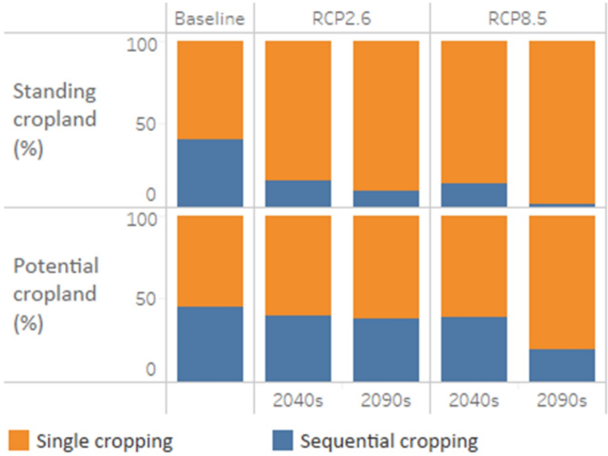


Figure 4.5. Proportion of cultivated and potential arable areas suitable for sequential and single cropping under different RCP scenarios in the watershed. The total cultivated areas in the watershed are 150,000ha and potential arable land areas are about 1.3 million ha.

4.3.3 IMPACT OF CLIMATE CHANGE ON STREAMFLOW DROUGHT AND IRRIGATION POTENTIAL

Our SDF and SSI analyses show that climate change will increase the severity, duration and frequency of streamflow droughts (Figure 4.6 and Figure 4.7). For example, a streamflow drought event with 120 days duration and a total deficit volume of about 50 million m^3 water in the watershed is estimated to occur once every 50 years (return period) under baseline climatic conditions. However, in the 2040s, the return period is projected to be 18 years and 13 years under RCP2.6 and RCP8.5 scenarios respectively (Figure 4.6). Increasing severity, duration and frequency of streamflow drought affects the availability of water for household consumption, riverine ecosystem requirements and irrigation. In addition to population increase, streamflow droughts will increase competition for water resources (Figure 4.8). In this study, because we prioritized household water demand and riverine ecosystem requirements over irrigation, streamflow droughts result in a substantial reduction in the irrigation potential of the watershed. Figure 4.8 shows that even under a combination of RCP2.6 and SSP1 scenarios, the irrigation potential will be halved.

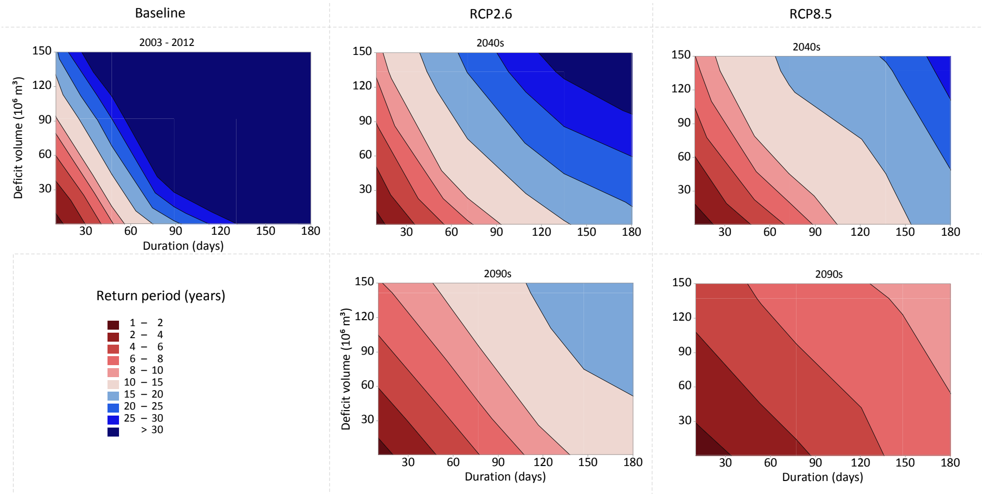


Figure 4.6. Contour plots showing severity-duration-frequency relationships of streamflow droughts in the Upper Ouémé watershed under the baseline climatic conditions (2003-2012) and two RCP scenarios. Contour plots were derived using daily streamflow simulated at the watershed outlet.

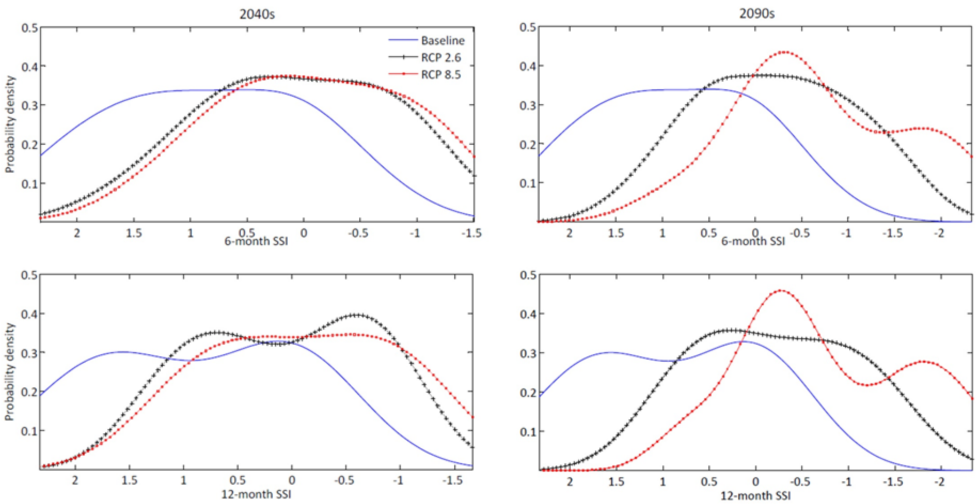


Figure 4.7. Probability density plots of 6-month and 12-month Standardized Streamflow Index (SSI) derived under baseline climatic conditions and under RCP scenarios. SSI values < 0 represent streamflow drought and the greater the absolute value the higher the severity. SSI values > 0 represent higher streamflow.

	Baseline	RCP2.6		RCP8.5	
	2003 - 2012	2040s	2090s	2040s	2090s
Baseline	27,089				
SSP1		13,272	12,025	11,505	5,843
SSP3		11,838	9,392	10,300	4,804

Figure 4.8. Irrigation potential in the Upper Ouémé watershed under a combination of RCP and SSP scenarios.

4.4 DISCUSSION

4.4.1 METHODOLOGICAL UNCERTAINTIES

Three main sources of uncertainties occur in this study. First is the use of the same land-cover and soil parameters for SWAT simulations under current and projected climatic conditions. Land-cover dynamics affect actual evapotranspiration and together with soil properties influence the partitioning of rainfall into overland flow and soil infiltration. By maintaining these watershed characteristics constant we were able to isolate the impacts of climate change on watershed hydrology from the impacts of plausible land-use changes. Nevertheless, incorporating plausible land-use changes could impact on our analyses in a variety of ways and the net effect on streamflow depends on the types and extent of land-use changes. For example, conversion of forests and woodlands for crop cultivation reduces dry season streamflow and consequently water available for irrigation (*Duku et al.*, 2016b). Second, elevated atmospheric carbon dioxide under projected climatic conditions was not included in our simulations. The effect of changes in atmospheric carbon dioxide on watershed hydrology is incorporated in the SWAT model through a modification of the Penman-Monteith equation for computing potential evapotranspiration (*Neitsch et al.*, 2009). In this study, however, we used the Hargreaves equation (*Hargreaves et al.*, 1985) to compute potential evapotranspiration because of inadequate data to apply the Penman-Monteith equation. Elevated atmospheric carbon dioxide has competing effects on the transpiration rate of vegetation cover. On one hand, it reduces transpiration rate by reducing leaf stomatal conductance (*Wand et al.*, 1999; *Medlyn et al.*, 2001; *Luo et al.*, 2013). On the other hand, it increases transpiration rate by stimulating plant growth (*Pritchard et al.*, 1999; *Wand et al.*, 1999; *Luo et al.*, 2013). The net effect on the magnitude and seasonality of the components of the hydrological cycle depends on local weather conditions and vegetation characteristics. For example, in forested watersheds of the northern Coastal Ranges and Sierra Nevada mountain range in California, research has shown that elevated carbon dioxide concentrations reduced evapotranspiration by around 3% and consequently increased streamflow (*Luo et al.*, 2013). More research, however, is needed on the net effect in the sub-humid tropics of West Africa dominated by woodland savannah. Third, we assumed that water availability was the only limiting factor for irrigation and all the other factors were non-limiting. The irrigation potential reported in this study therefore is the maximum irrigable land area because in practice other physical and socioeconomic factors will be constraining. However, current limitations such as water access, access to fertilizer, irrigation infrastructure and financial capital may be overcome in the future by socioeconomic development and technological advancement. Therefore, analysing the impact of climate change on maximum irrigable land area is instructive and important for long-term irrigation development planning.

4.4.2 IMPLICATIONS ON CROP PRODUCTION

In rainfed production systems that are characteristic of the watershed and SSA in general, crop growth and yield are closely related to the LGP. Across SSA inadequate LGP is one of the major constraints to establishing rainfed sequential cropping systems. In this study we have demonstrated that increasing drought risk due to climate change will considerably reduce the LGP across the study area. Due to the reduction in LGP, substantial areas in both cultivated and uncultivated areas hitherto suitable for rainfed sequential cropping will revert to single cropping. Cultivated and uncultivated areas that currently are used for single cropping or can support single cropping will still be suitable in the coming decades. However, the number of crops that can be cultivated (especially long-cycle crops such as cassava and yam) will be limited as a result of the shortening of the LGP. Our analyses show that currently about 60,000ha (40%) of cultivated areas in the study area are suitable for sequential cropping (including relay cropping). It is difficult to ascertain if all of these areas are actually used for sequential cropping due to inadequate data. However, the average cropping intensity of staple crops such as maize, yam, cassava and sorghum in the study area is 1.5 i.e. the harvested area is one and half times greater than the physical area devoted to the cultivation of each crop (*You et al.*, 2014). This can be attributed to both sequential cropping and intercropping. If all 60,000ha of suitable sequential cropping areas in cultivated areas are currently used for sequential cropping, then in the coming decades farmers will have to shift to either single cropping systems or adopt crop cultivars with shorter growing periods. To a degree, the impact on crop production may be mitigated by agronomy including breeding of drought resistant varieties. If, however, a substantial portion is only used for single cropping, then climate change will severely limit such opportunities for increasing crop production. In both situations the substantial reductions in sequential cropping areas may result in relatively greater rainfed cropland expansion in order to make up for lost opportunities to increase crop production. This will affect current land uses in these areas.

Over 90% of the total suitable sequential cropping areas are currently not under cultivation and lie in the forested south-western part of the watershed. In these areas the higher soil water-holding capacity allows for relatively longer LGPs and the impact of climate change is relatively less. However, most of these potentially suitable sequential cropping areas are presently either forested or are woodland savannahs. The forested areas are essential for biodiversity conservation, wood resources, water flow regulation, carbon sequestration etc. and the woodland savannah areas provide grazing opportunities for livestock. Particularly during the dry season, pastoral communities from other parts of Benin and neighbouring countries such as Nigeria often migrate to this study area for grazing (*Judex and Thamm*, 2008). Nevertheless, it is likely that forested and woodland savannah areas will be increasingly under pressure from land use change in the coming decades. Among others, this may cause tensions between pastoralists traditionally using the areas for grazing and new settlers.

In Benin, the total actual land area equipped for irrigation is only 23,000ha (*FAO*, 2016a). In our study area, irrigation is almost non-existent. The lack of irrigation water during the dry season has been a major problem for many farmers (*Burney et al.*, 2010; *Giertz et al.*, 2012). To sufficiently increase crop production in the coming decades, irrigation will have to play a crucial role and is a pathway that has been proposed for other parts of SSA (*Dillon*, 2008; *Nkhata*, 2014). Under current climatic conditions there is considerable scope for irrigation expansion. A maximum land area of 27,000ha can be irrigated even after household water demands and environmental water requirements have been implemented. However, our study has shown that future opportunities for irrigation expansion will be severely constrained by increased severity, frequency and duration of streamflow droughts. Streamflow droughts coupled with increased household water-use due to population growth and socioeconomic development will result in increased competition for surface-water resources. Where household and environmental water requirements are prioritized over irrigation as in this study, streamflow droughts will substantially reduce the irrigation potential of the watershed. Potential cropland expansion into currently forested areas will also affect seasonal distribution of streamflow further reducing irrigation opportunities especially in the critical dry season (*Duku et al.*, 2016b). Deforestation leads to in increased peak flow in the wet season and reduced baseflow in the dry season.

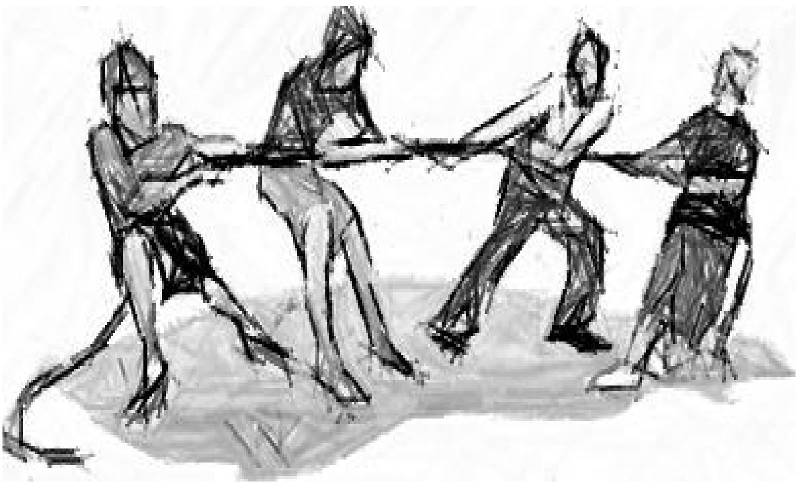
4.5 CONCLUSIONS

In this study, we have shown that in addition to crop yield responses climate change will affect other options that have been used to increase crop production in recent decades in SSA i.e. rainfed sequential cropping, rainfed cropland expansion and irrigation expansion. Currently, about 41% of cultivated areas in the Upper Ouémé watershed are either used for rainfed sequential cropping or can support it. However, by 2050 this will decrease to between 2% and 16% depending on the climate change scenario. Farmers will therefore have to shift to single cropping systems or adopt improved agronomic practices including drought-resistant and short-cycle cultivars. Farmers may also be driven to expand to hitherto uncultivated areas to make up for lost opportunities to increase crop production. In the Upper Ouémé watershed, over 90% of land areas that can support rainfed sequential cropping are not currently under cultivation and are under different land uses such as forest and woodland savannah. This situation is unlike other parts of Benin where the availability of currently uncultivated land is much lower. A large part of these potential arable lands will still be able to support rainfed sequential cropping in the coming decades despite the loss of between 15 and 30 growing days due to the relatively higher soil water holding capacities. If these areas are to be used for rainfed sequential cropping, then fallow periods will have to be shortened or lost completely and improved soil and nutrient management will be needed to increase productivity. However, in Chapter 3 of this thesis, I showed that conversion of forested and woodland savannah areas to crop cultivation will have negative feedbacks on water availability for irrigation. In the present study, we have shown that even if there is no change in forest cover at least 50% of irrigation potential will be lost in the coming decades. Forest and woodland areas, therefore, will be needed to regulate water flows and increase dry season streamflow in addition to the provision of other ecosystem services. Given the urgent need to increase crop production to meet the demands of a growing population in SSA, a better understanding of the varied impacts of climate change on cropping intensity, suitability for cropland expansion and irrigation expansion in addition to crop yields is needed for agricultural development planning.

ACKNOWLEDGEMENTS: This research was conducted at Wageningen University as part of the "Realizing the agricultural potential of inland valley lowlands in sub-Saharan Africa while maintaining their environmental services" project (RAP-IV). The project is implemented by the Africa Rice Center and its national partners and is funded by the European Commission through the International Fund for Agricultural Development (IFAD). We thank the IMPETUS project and AMMA-CATCH regional observing system in Benin for making data available for this research through their public geoportals.

CHAPTER 5

QUANTIFYING TRADE-OFFS BETWEEN FUTURE FOOD AVAILABILITY AND FOREST AND WOODLAND CONSERVATION



ABSTRACT

Meeting the dual objectives of increasing crop production and consequently improving food security on one hand and protecting natural ecosystems by limiting cropland expansion on the other hand will be a major challenge in the coming decades in sub-Saharan Africa (SSA). To this end agricultural intensification is considered desirable, yet, several important questions need to be addressed for major food crops at national and sub-national levels in SSA. Will intensification measures such as yield gap closure and increasing cropping intensity be adequate to meet food security objectives? How will climate change affect the degree of success of intensification measures and how much additional land may be required? To address these questions, we quantify trade-offs between per capita food availability and protecting forests and woodlands at different levels of yield increases taking into account climate change, population growth and land quality of potential arable areas. We carry out these analyses for three major food crops, i.e. maize, cassava and yam, in Benin. Our analytical approach combines soil-water balance and crop growth modelling under two scenarios of Representative Concentration Pathways (RCPs) with two scenarios of population growth based on Shared Socioeconomic Pathways (SSPs) and three scenarios of cropland expansion with varying degrees of deforestation. Our analyses show that depending on the scenarios 1) maize and yam yields will have to increase at a faster rate than has been recorded over the past two and half decades in order to maintain current levels of food availability in case of no expansion of croplands; 2) major investments in higher-yielding cultivars (especially for cassava) and irrigation will be required to raise the yield ceiling in order to overcome the biophysical limitations on yields imposed by climate change; 3) substantial areas of forests and woodlands may have to be converted to croplands to make up for lost opportunities to increase production. However, forest and woodland loss will affect water availability for irrigation. Our study shows that food security outcomes and forest and woodland conservation goals in Benin and the larger SSA region are inextricably linked together and require holistic management strategies that considers trade-offs and co-benefits.

This chapter is based on:

Duku, C., Zwart, S.J., van Bussel L.G., Hein, L., Quantifying trade-offs between future food availability and forests and woodland conservation in Benin. *Science of The Total Environment* (Submitted)

5.1 INTRODUCTION

Substantial increases in crop production are needed to meet growing food demand of the population of sub-Saharan Africa (SSA), which is projected to double by 2050 compared to 2015 estimates (*UN*, 2015b). For example, the cereal demand for 10 countries in SSA is projected to triple by 2050 compared to 2010 estimates, assuming an average population growth of 1.5% per annum and modest dietary changes in these countries (*van Ittersum et al.*, 2016). Currently in SSA, about 97% of cropland area is under rainfed cultivation (*You et al.*, 2011) and productivity levels for major food crops, which are the lowest in the world, are inadequate to meet projected demand (*Alexandratos and Bruinsma*, 2012; *van Ittersum et al.*, 2016).

In recent decades, the slow rate of increase in productivity levels of major crops has led to conversion of vast areas of forests and woodland to croplands (*Pretty et al.*, 2011). Over the past 25 years (1990–2015), forest and woodland extent in SSA has been declining (*Sloan and Sayer*, 2015) despite the reduction in the rate of net global deforestation (*FAO*, 2015b). For example, between 2010 and 2015 there was a 1.1% (50,000 ha per annum) decrease in forest area in Benin (*FAO*, 2015b). Forests and woodlands in SSA provide, however, essential ecosystem services of local, national and global benefits. Forests and woodland areas of SSA are essential for amongst others biodiversity conservation (*Chidumayo and Gumbo*, 2010), water flow regulation (*Duku et al.*, 2016b), wood and non-wood forest products, desertification control and soil amelioration, carbon sequestration (*Lal*, 2004; *UNFCCC*, 2006), and grazing opportunities for livestock (*Judex and Thamm*, 2008).

To meet the dual objectives of increasing crop production, and consequently ensuring food security, on one hand and protecting natural ecosystems by limiting cropland expansion on the other hand, agricultural intensification is considered most desirable (*Palm et al.*, 2010; *Foley et al.*, 2011; *Pretty et al.*, 2011; *Garnett et al.*, 2013; *Hall and Richards*, 2013). Yet, several important questions need to be addressed for major food crops at national and sub-national levels in SSA. Will intensification measures alone be adequate to meet growing food demand? Moreover, how will climate change affect the degree of success of intensification measures and how much additional land may be required? Even without climate change, a recent study showed that it will not be feasible to meet 2050 SSA cereal demand on existing production cropland by yield gap closure alone (*van Ittersum et al.*, 2016). Taking into account climate change especially in such analysis is important because there are biophysical limitations to yield increases imposed by temperature and water supply (*Gornall et al.*, 2010; *Hall and Richards*, 2013; *van Ittersum et al.*, 2013). West Africa in particular has been identified as a regional hotspot of climate change with climate departure from historical variability projected to occur faster than the global average (*Diffenbaugh and Giorgi*, 2012; *IPCC*, 2013; *Mora et al.*, 2013).

Therefore, in this study we quantify trade-offs between per capita food availability and forests and woodland conservation at different levels of yield increases, taking into account climate change, population growth and land quality of potential arable areas. We undertake these analyses for three major food crops in Benin i.e. maize, cassava and yam. Specifically, we analyse levels of yield increases and/or cropland expansion required to at least maintain per capita food availability for the decades 2041 – 2050 (2040s) and 2091 – 2100 (2090s) at baseline levels (i.e. 2001 – 2011). We are in no way positing that per capita availability of these crops was optimal for the period 2001 – 2011 even though for this period the average dietary energy supply adequacy, i.e. the ratio of dietary energy supply and dietary energy requirements, was 114% (*FAO*, 2016b). Cereals, roots and tubers accounted for over 70% of the dietary energy supply in Benin over this period (*FAO*, 2016b). These crops all are produced locally (*FAO*, 2016b). Maize is the most widely cultivated cereal, and cassava and yam are the main root and tuber crops. Our analytical approach combines soil-water balance and crop growth modelling under two scenarios of Representative Concentration Pathways (RCPs) with two scenarios of population growth based on Shared Socioeconomic Pathways (SSPs) and three scenarios of cropland expansion with varying degrees of deforestation, to investigate the trade-offs between capita food availability and forests and woodland conservation.

5.2 METHODOLOGY

5.2.1 STUDY AREA

Our study area (Figure 5.1) covers the Ouémé river basin in Benin and the south-western parts of the country that lie outside the river basin. It covers an area of approximately 55,000 km^2 which is about half of the total area of Benin. Benin is divided into eight agricultural zones based largely on climate, soil type, elevation and different crops grown (*MDR*, 1998; *FAO*, 2015a). Six of these zones lie within our study area (Table 5.1). The total population in the study area is approximately 7 million which is 70% of the total population of the country (*Bright et al.*, 2011). The proportion of the total population of the country that is undernourished is currently 7.5% (*FAO et al.*, 2015). The study area is located in the sub-humid tropical region with aridity indices ranging between 0.50 and 0.65. Land use in the northern and central part of the study area is dominated by a mosaic of woodland savannah and forest islands with relatively smaller cropland area. The southern part of the study area is relatively more urbanized and land use is dominated by large areas of small-scale rainfed agriculture and oil palm plantations. There are also large state-owned protected forests covering an area of about 6,000 km^2 mainly in the northern and central part of the study area. The irrigation sector is poorly developed and the lack of irrigation water during the dry season is a major problem for many farmers (*Giertz et al.*, 2012). Consequently, crop cultivation mainly takes place during the wet season. Due to land availability in the northern and central parts of the study area, residents from other parts of the country and neighbouring countries migrate to this region, which has caused expansion of agricultural areas and led to substantial deforestation (*Judex and Thamm*, 2008).

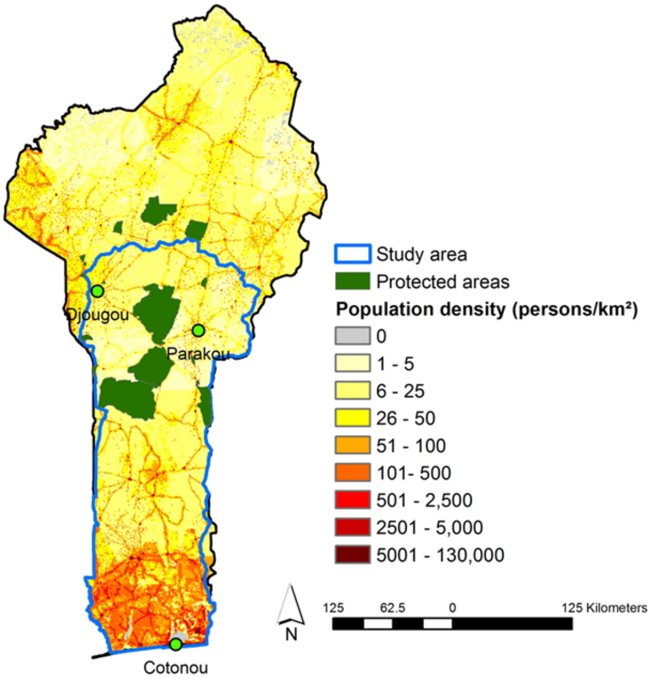


Figure 5.1. Map showing current population density of Benin (Bright et al., 2011), our study area and protected areas.

Table 5.1. Shares of cropland area in agricultural zones used for cultivation of maize, cassava and yam (You et al., 2014)

Agricultural zones	**Total cropland area in study area (ha)**	**Share of cropland area for each crop (%)**		
		Maize	Cassava	Yam
Zone 3	58,100	20	3	9
Zone 4	48,927	7	11	5
Zone 5	321,449	25	8	14
Zone 6	140,965	39	25	< 1
Zone 7	52,992	33	19	< 1
Zone 8	73,571	57	16	2
Total	696,004			

5.2.2 CLIMATE CHANGE AND POPULATION GROWTH SCENARIOS

We used two contrasting climate change scenarios, Representative Concentration Pathways (RCPs) 2.6 and RCP8.5. For each scenario, we obtained precipitation, maximum and minimum temperature and solar radiation data projected for the decades 2041 – 2050 (2040s) and 2091 – 2100 (2090s). The climate data were downscaled multi-model mean climate data of 17 General Circulation Models (Appendix II) obtained from the MarkSimGCM geoportal (*Jones and Thornton*, 2013). The RCP2.6 scenario is an emission pathway that leads to the lowest concentration levels of atmospheric greenhouse gases (*Moss et al.*, 2010; *IPCC*, 2013). It represents a peak in greenhouse gas emissions before 2100 followed by a consistent decline throughout the rest of this century. The RCP8.5 scenario, on the other hand, is characterized by increasing greenhouse gas emissions throughout this century leading to the highest concentration of atmospheric greenhouse gases by the end of this century (*Moss et al.*, 2010; *IPCC*, 2013).

Future population estimates were based on two Shared Socioeconomic Pathways (SSPs) for 2050 and 2100 (Table 5.2) (*O'Neill et al.*, 2013). The SSPs are a set of alternative reference assumptions about future socioeconomic development in the absence of climate policies. The SSP1 scenario depicts a development pathway characterised by rapid economic development especially in low-income countries leading to rapid technological development, increased resource use efficiency and low population growth; SSP3 represents a scenario with slow economic growth, slow technological development, low resource use efficiency and rapid population growth.

Table 5.2. Future population estimates in the study area (O'Neill et al., 2013). For the 2040s and 2090s decades, the population represents projections for 2050 and 2100 respectively. Current population is approximately 7 million (Bright et al., 2011)

	2040s	2090s
SSP1	8,200,000	14,700,000
SSP3	10,100,000	23,700,000

5.2.3 CROPLAND EXPANSION SCENARIOS

The total area of land currently under cultivation in the study area is about 700,000 ha of which 30% is used for maize cultivation, 13% for cassava cultivation and 8% for yam cultivation (*You et al.*, 2014). Table 1 shows the disaggregated share of current cropland area in each of the eight agricultural zones. Three cropland expansion scenarios involving varying degrees of conversion of forest and woodland savannah areas for crop cultivation by 2050 and 2100 were defined. The first scenario was the 'No cropland expansion scenario', representing prioritization of forest conservation over cropland expansion. Therefore, the current cropland extent in the study area would be maintained, without further conversion of forest and woodland savannah area for crop cultivation. The second scenario represents the situation where only the conservation of protected forests would be prioritized and forest clearing for crop cultivation could occur outside protected areas ('Conservation of only protected areas'). The third scenario represents the prioritization of cropland expansion over forest conservation ('Unlimited cropland expansion'). Therefore all forested areas including protected areas would be available for crop cultivation.

In each cropland expansion scenario, we assumed no change in shares of areas of the different crops in each agricultural zone. We acknowledge that for example trends over the past two decades indicate increase in maize area at the expense of sorghum in SSA, however, it is not clear how the share of each crop area in Benin will develop in the future. We also assumed that the best lands in terms of soil moisture availability in current and new croplands would be set aside for cultivation of maize, yam and cassava because these crops were the major crops in the study area (in addition to rice and cotton). Our assumptions were necessary because land use and land cover data available for the study do not have sufficient detail to allow for the differentiation of specific crops in the cropland areas.

5.2.4 WATER-LIMITED YIELD POTENTIAL

Water-limited yield potential (Yw) is the maximum yield of a crop under conditions where only water supply is limiting, nutrients are assumed to be non-limiting and there is effective biotic stress control (*Lobell et al.*, 2009; *van Ittersum et al.*, 2013). Yw is influenced by soil properties especially rooting depth and root-zone water holding capacity (*Lobell et al.*, 2009; *van Ittersum et al.*, 2013). Water-limited yield represents the yield potential in rainfed cropping systems. Our approach for simulating Yw for the study area involved three steps: soil-water balance modelling to simulate spatio-temporal dynamics of soil moisture, crop growth simulation, and Yw upscaling. Because of inadequate data, we carried out the soil-water balance and crop growth simulation for only a part of the study area (*Duku et al.*, 2015). Climatic and biophysical features in the study area in *Duku et al.* (2015) are largely representative for the whole study area of the present paper (*Licker et al.*, 2010; *HarvestChoice*, 2011; *van Wart et al.*, 2013b; *Claessens and van Wart*, 2014). All simulations were carried out on gridded data at 1 ha spatial resolution.

We used the Soil and Water Assessment Tool (SWAT) to simulate the spatio-temporal dynamics of soil moisture content, required for Yw simulation. The SWAT model is a physically based, ecohydrological model that simulates the impact of land use and land management practices on the complete hydrological cycle in large complex watersheds with varying soils, land use and management conditions over a long period of time (*Neitsch et al.*, 2009). The SWAT model has been calibrated and validated for the northern part of the study area in a previous study *Duku et al.* (2015).

Simulated time-series of soil moisture content were used as input for the simulation of Yw, in addition to precipitation, temperature and solar radiation data. We utilized the AEZ crop growth model to simulate Yw. The AEZ crop growth model is based on *Kassam* (1977), *Doorenbos and Kassam* (1979) and *Smith* (1992). The AEZ crop modelling framework uses agronomic-based information in addition to spatially explicit biophysical datasets to simulate crop production potentials (*IIASA/FAO*, 2012). The model firsts calculates maximum attainable biomass and yield as determined by temperature and radiation regimes followed by computation of Yw based on precipitation and soil moisture dynamics (*IIASA/FAO*, 2012). In the AEZ crop growth model, the effect of increasing atmospheric carbon dioxide concentration on crop yield is accounted for by yield-adjustment factors (*IIASA/FAO*, 2012). For each crop, an iterative simulation of Yw with varying starts of growing season was undertaken and the maximum Yw in each grid cell (1 ha) was selected.

Simulated Yw were then upscaled to the entire study area based on delineation of Hydro-Climatic Response Units (HCRU). Given data constraints, such upscaling approaches are useful for extending coverage of estimates of Yw. In this study, each HCRU is a discretized landscape unit with homogenous land cover, soil type, slope (*Neitsch et al.*, 2009) and in which the aridity index varies by not more than one standard deviation. The aridity index is the ratio of annual precipitation and potential evapotranspiration totals and represents the degree to which precipitation can satisfy vegetation water requirements (*UNEP*, 1993). The study area is located in the sub-humid tropical region with aridity indices ranging between 0.50 and 0.65 (mean of 0.59 and a standard deviation of 0.02). Grid cells in each HCRU, therefore, have similar climatic water balance and exhibit similar hydrological response in terms of soil moisture content. HCRUs identified in the northern part of the study area were matched to HCRUs identified in the rest of the study area and Yw were assigned to matching HCRUs.

5.2.5 FOOD AVAILABILITY

Food availability is one of the four key dimensions of food security (*FAO*, 1996). We computed per capita food availability for each studied crop under different combinations of RCPs, SSPs and cropland expansion scenarios. We first estimated potential cropping intensity based on the length of growing period. Under each combination of RCP and cropland expansion scenario, the potential area under cultivation was delineated into sequential double and single cropping zones based on the length of growing period (See Chapter 1). The cropping intensity was then computed based on Eq (5.1). A cropping intensity of 1.5 implies that half of the total land area under cultivation can be used to grow two or more crops per year. For baseline conditions, we estimated cropping intensity based on harvested and physical areas (*You et al.*, 2014).

$$C_i = 1 + (D_{cz}/A_c) \tag{5.1}$$

where C_i is cropping intensity; D_{CZ} is total area that can grow two or more crops per year (ha); A_c is total area under cultivation (ha).

Part of the crop production is used as feed for livestock and poultry, seeds for cultivation, and other uses such as starch in the case of cassava. In addition, there are also postharvest and other forms of losses. We used the mean food component of total production over the baseline period for the computations of future food availability. Per capita food availability of each crop was then computed based on Eqs (5.2) and (5.3).

$$T_p = Y \times P_a \times C_i \tag{5.2}$$

$$F_a = (k \times T_p)/Pop \tag{5.3}$$

where T_p is total production of each crop in the study area (kg); Y is average yield of each crop (kg/ha); P_a is physical area for cultivation of each crop (ha); C_i is the cropping intensity; F_a is per capita food availability (kg/person/year); k is the proportion of total production that is used as food;; Pop is total population of the study area [persons];

For each crop, we also calculated the average rate of yield increase over the period 1991 - 2014 and extrapolated the results into 2050 and 2100. We calculated per capita food availability based on these on these trends in yield increases in existing cropland areas.

5.3 RESULTS

Figure 5.2 shows that Yw of maize is likely to decrease from 7.2 t/ha under baseline climatic conditions to 4.5 t/ha under RCP8.5 by 2100. Water-limited yield potential of cassava is also likely to decrease from 26.3 t/ha to 19.7 t/ha under RCP8.5 by 2100 whereas Yw of yam will decrease from 30.4 t/ha to 20.4 t/ha.

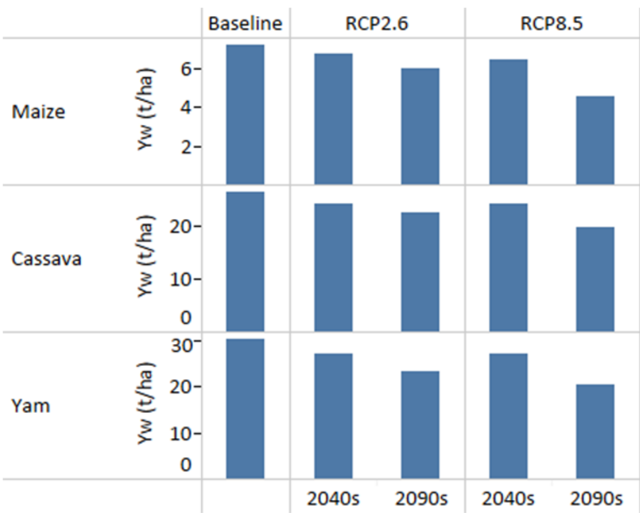


Figure 5.2. Water-limited yield potentials (Yw) under different climate change scenarios.

5.3.1 MAIZE

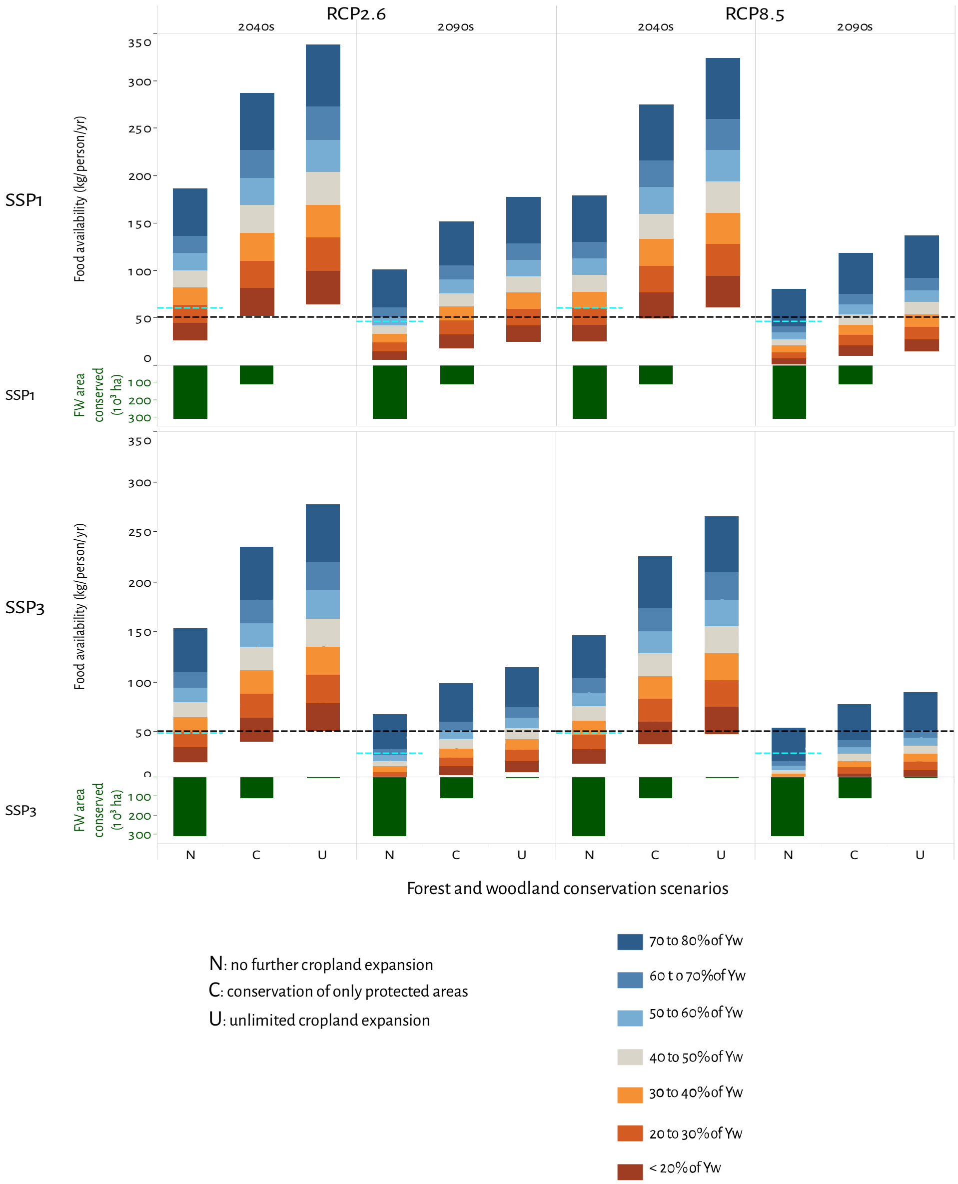


Figure 5.3. Trade-offs between per capita maize availability, and forest and woodland conservation under different scenario combinations of climate change, population growth and yield increases. FW is forest and woodland area conserved. Black dashed line indicates availability of food per capita for the baseline period (2001 – 2011). Blue dashed lines indicate per capita food availability based on continuation of trends in maize yields in existing cropland areas (i.e. an increase of 14 kg/ha/yr). The current yield is about 20% of Yw.

In the period, 1991 to 2014, maize yields in Benin increased at an average rate of 14 kg/ha/yr (*FAO*, 2016b). Our analyses show that at this rate maize production in existing croplands would barely be adequate to maintain current levels of maize availability by 2050 in all climate change and population growth scenario combinations (Figure 5.3). However, continued yield increases at the current rate will be inadequate to maintain the current level of availability by 2100 in all climate change and population growth scenario combinations. If population grows at 1.3% per annum under SSP1, maize yields in existing production areas will have to increase at a rate of at least 17 kg/ha/yr in all climate change scenarios. However, if population grows at 3% per annum under SSP3, maize yields will have to increase at 37 kg/ha/yr in all climate change scenarios. If maize yields continue to increase at the current rate and if population grows at 3% per annum under SSP3 regardless of the climate change scenario then a physical area of about 200,000 ha outside protected areas (at a cropping intensity of 1.5) in addition to the current harvested area of 350,000 ha will be adequate to maintain current levels of per capita availability.

5.3.2 CASSAVA

Unlike maize, cassava yields have been increasing at a much faster rate in Benin. In the period 1991 – 2014, cassava yields increased at a rate of 333 kg/ha/yr (*FAO*, 2016b). At this rate, cassava production in existing croplands would substantially increase availability by 2050 in all combinations of climate change and population growth scenarios (Figure 5.4). By 2100, however, achieving 80% of Yw will be inadequate to maintain current levels of availability in all combinations of climate change and population growth scenarios without cropland expansion. Currently, the harvested area of cassava in the study area is about 131,000 ha. If population grows at 1.3% per annum, an additional physical area of 92,000 ha at a cropping intensity of 1.5 in addition to continued yield increases at the current rate will be required to maintain current availability. However, if population grows at 3% per annum even achieving 80% of Yw on a harvested area of 223,000 ha (which is the total available area for cassava cultivation based on current shares of crops and cropping intensity) will be inadequate to maintain current levels of availability by 2100.

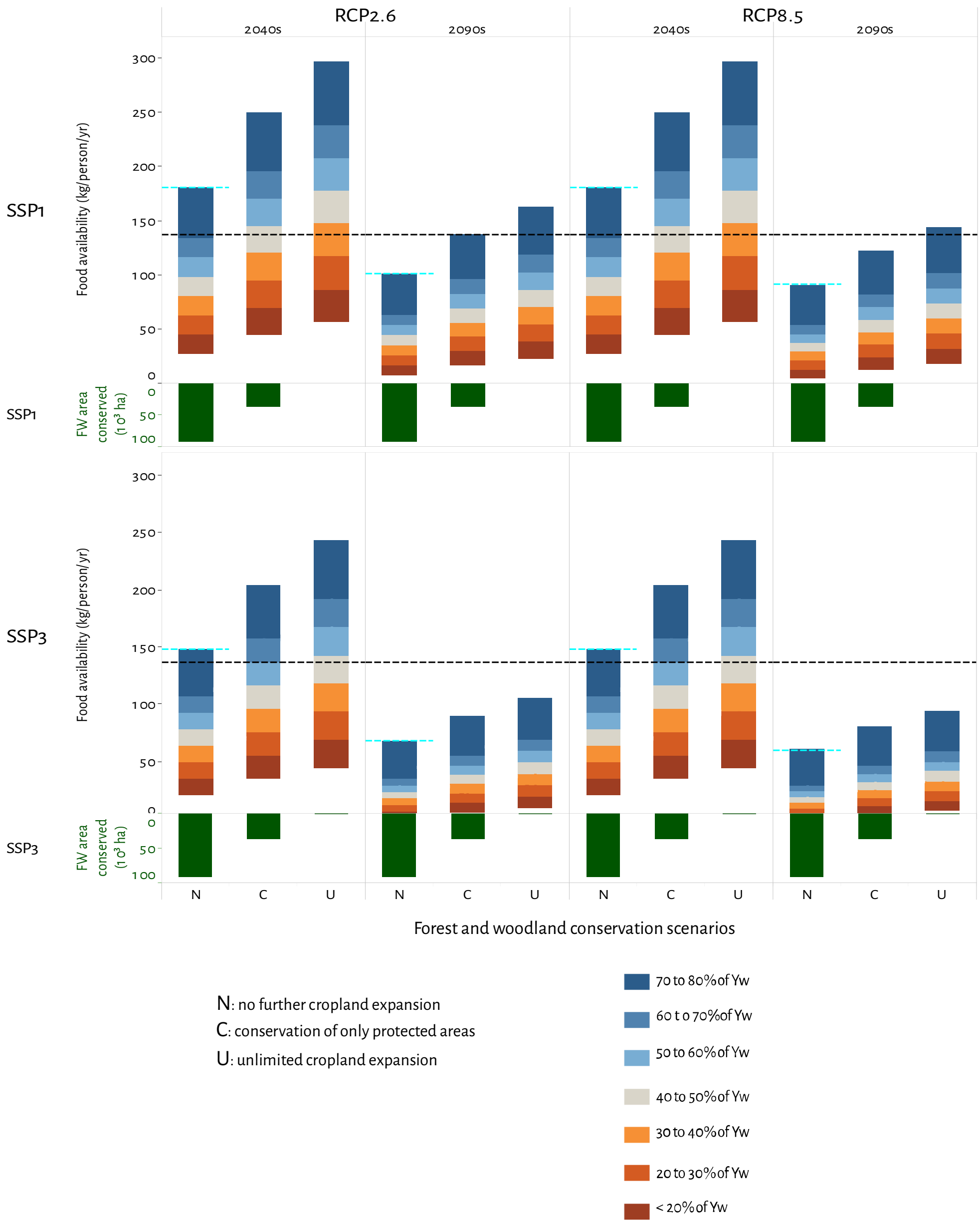


Figure 5.4. Trade-offs between per capita cassava availability, and forest and woodland conservation under different scenario combinations of climate change, population growth and yield increases. FW is forest and woodland area conserved. Black dashed line indicates availability for the baseline period (2001 – 2011). Blue dashed lines indicate per capita food availability based on continuation of trends in cassava yields in existing cropland areas (i.e. an increase of 333 kg/ha/yr). The current yield is about 50% of Yw. For all time-periods, extrapolated yields were higher than Yw and hence Yw were used.

5.3.3 YAM

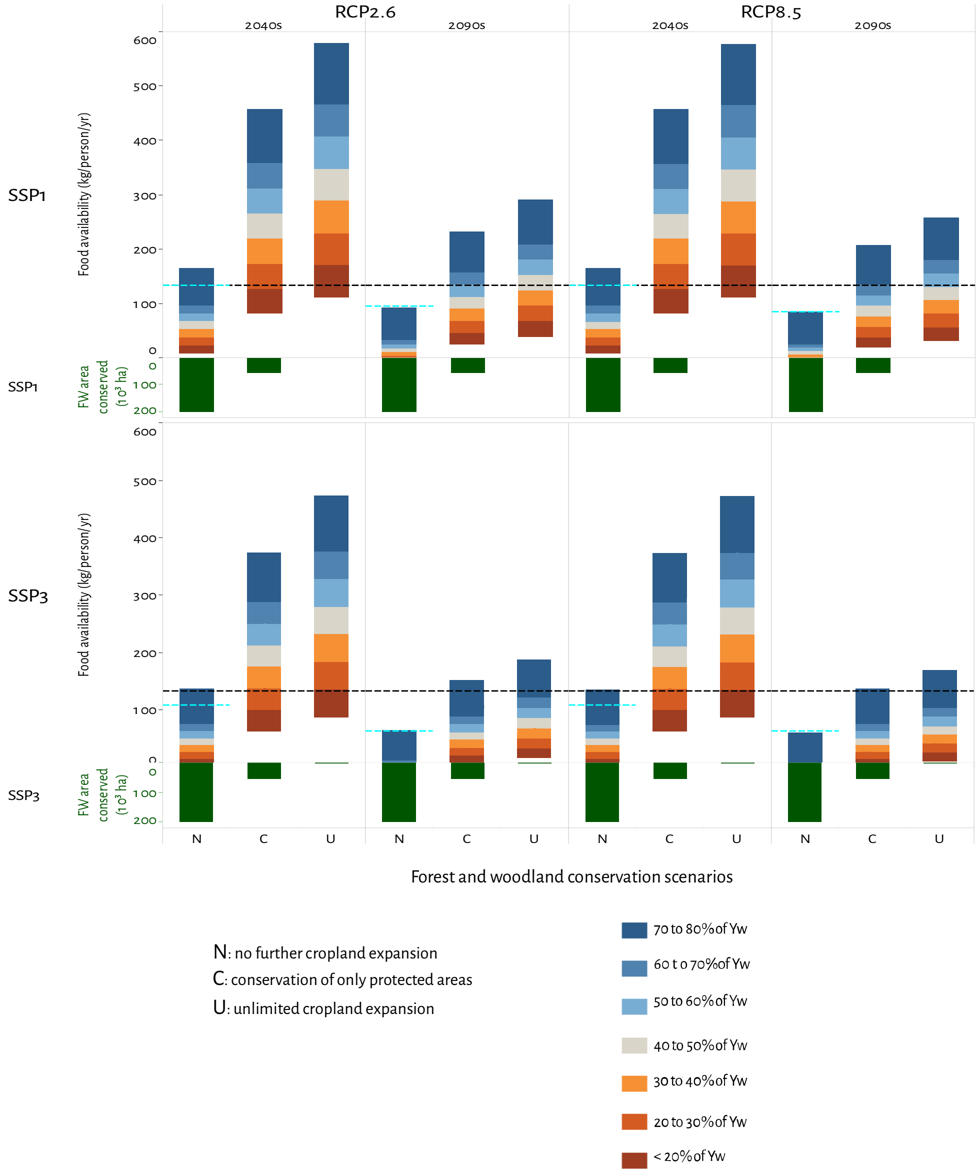


Figure 5.5. Trade-offs between per capita yam availability, and forest and woodland conservation under different scenario combinations of climate change, population growth and yield increases. FW is forest and woodland area conserved. Black dashed line indicates availability for the baseline period (2001 – 2011). Blue dashed lines indicate per capita food availability based on continuation of trends in yam yields in existing cropland areas (i.e. an increase of 160 kg/ha/yr). The current yield is about 50% of Yw. For the 2090s, extrapolated yields were higher than Yw and hence Yw were used.

Yam yields in Benin increased at an average rate of 160 kg/ha/yr in the period 1991 – 2014 (*FAO*, 2016b). If yam yields continue to increase at this rate and if population grows at 1.3% per annum, production in existing croplands will be barely sufficient to maintain current levels of availability by 2050 (Figure 5.5). However, if population grows at 3% per annum, yam production in existing croplands will be inadequate to maintain current levels of availability by 2050. Under such a population growth scenario, yam yield will have to increase at 305 kg/ha/yr in existing croplands by 2050 in order to maintain current levels of availability. If yam yields continue to increase at the current rate then cropland expansion may be required. Based on the current share of areas of crops, a maximum physical area of 140,000 ha outside protected areas can be converted for yam cultivation. In addition to the current harvested area of 80,000 ha, this will be adequate to maintain current per capita yam availability by 2050 and 2100 in all climate change and population growth scenarios.

5.4 DISCUSSION

The degree of success of intensification measures (such as yield gap closure and increasing cropping intensity) and consequently the level of cropland expansion required to meet food security objectives in Benin will depend on population growth and climate change. Our study shows that if current food availability is to be maintained without further cropland expansion, yields will have to increase at a faster rate than has been recorded over the past two and half decades because of rapid population growth. Particularly for yam and maize, the average rate of yield increases recorded over the past two and half decades will have to be doubled in the next three decades and eight decades respectively if population grows at 3% per annum as projected under SSP3 scenario. Population growth rate of 3% per annum is not unlikely and is line with the United Nations population projections (*UN*, 2015b). Over the period 1991 - 2013, population in Benin increased at a rate of 4% per annum (*FAO*, 2016b). To increase yields at the required rates will require major investment in sustainable intensification such as: improvement of farmer knowledge and capacity through the use of farmer field schools and modern information and communication technologies; science and farmer inputs into technologies and practices that combine crops–animals with agro-ecological and agronomic management; engagement with the private sector for supply of goods and services (*Pretty et al.*, 2011).

Despite these efforts aimed at increasing yields, the maximum yield that can be attained in rainfed production systems will be affected by climate change. Climate change will impose biophysical limitations on yield increases by reducing Yw. Reduction of Yw will result in lost opportunities to increase crop production through yield increases in rainfed systems. The implication is that regardless of optimal nutrient management, biotic control and the aforementioned measures, maize, cassava and yam yields in rainfed systems cannot exceed 4.5 t/ha, 19.7 t/ha and 20.4 t/ha respectively under RCP8.5 by 2100. In some climate change and population growth scenario combinations, even closing the yield gap in the coming decades will be inadequate to maintain current levels of food availability. To a degree, good agronomic practices and irrigation expansion can provide opportunities to overcome biophysical yield limitations imposed by climate change. Water-limited yield potentials can be raised by introducing higher-yielding cultivars that may be either drought resistant or have relatively shorter growth duration (*Gornall et al.*, 2010; *Hall and Richards*, 2013). Increased application of irrigation can also ensure that full yield potentials are realized instead of water-limited yield potentials (*van Ittersum et al.*, 2013).

Increasing cropping intensity as an option to increasing production in existing croplands will also be affected by climate change. Currently, the cropping intensity of maize, cassava and yam in the study area is 1.5, i.e., the harvested area of each crop is 1.5 times larger than physical area set aside for cultivation (*You et al.*, 2014). In Chapter 4 of this thesis, I showed that between 50% (30,000 ha) and 95% (57,000 ha) of existing croplands in the northern part of the study area (with an area of 14,500 km^2) that can support the cultivation of more than one crop in a year will revert to single cropping as a result of climate change. In the present study, because we assumed that the best lands in terms of soil moisture availability would be used for the cultivation of maize, cassava and yam, the computed cropping intensity of these crops was the same under all climate change scenarios. The implication is that in the coming decades maintaining or increasing this cropping intensity in existing croplands will be essential for maintaining current levels of food availability. The consequence of a lower cropping intensity is that either yields will have to increase at a much faster rate and/or more cropland expansion will be required than estimated in this study. Increasing cropping intensity in SSA has been challenging in the past and has been constrained by factors such as high labour intensity, lack of knowledge and lack of market access in addition to length of growing period (*Waha et al.*, 2013). In the coming decades these factors must be addressed and the aforementioned lessons drawn from successful intensification projects in SSA will be useful (*Pretty et al.*, 2011).

Our results show that, as a result of population growth and the impact of climate change, it is likely that forested and woodland savannah areas will be increasingly under pressure in the coming decades in Benin. This may cause tensions between pastoralists traditionally using the areas for grazing and new settlers. Particularly during the dry season, pastoral communities from other parts of Benin and neighbouring countries such as Nigeria often migrate to this study area for grazing (*Judex and Thamm*, 2008). Furthermore, if there is no change in shares of areas of different crops then our analyses show that in some scenarios the maximum area for cultivation of maize and cassava excluding protected forests will be inadequate to maintain current levels of food availability. State-owned protected forests in the study area are therefore likely to come under increasing pressure from surrounding villages and migrants from the southern part of Benin looking for land to farm. These protected forests have remained largely intact in recent decades. Apart from the loss of biodiversity and ecosystem services, deforestation and forest degradation also has direct negative impacts on irrigation opportunities (*Duku et al.*, 2016b). In Chapter 4 of this thesis, I showed that even if there is no change in forest and woodland cover in the northern part of the study area, at least 50% of irrigation potential will be lost in the coming decades as a result of climate change. Changes in forest and woodland cover are likely to reduce irrigation potential by as much as 20,000 ha i.e. 80% of the total irrigation potential in the northern part of the study area (*Duku et al.*, 2016b). If irrigation expansion will play a crucial role in this study area, then a certain threshold of forest and woodland extent especially in the northern and central parts will be needed to regulate water flows and increase dry season streamflow.

5.5 CONCLUSION

Food security in the coming decades will be a major challenge in SSA and our study has shown that efforts to address this will be further challenged by climate change, rapid population growth and the need to protect natural ecosystems such as forests and woodlands. Whereas crop production on existing croplands will have to increase in order to maintain current levels of per capita food availability due to rapid population growth, climate change will reduce the maximum yields that can be attained in existing rainfed production systems. The combined effect is that depending upon the scenarios 1) maize and yam yields will have to increase at a faster rate than has been recorded over the past two and half decades in order to maintain current levels of food availability; 2) major investments in higher-yielding cultivars (especially for cassava) and irrigation will be required to raise the yield ceiling in order to overcome the biophysical limitations on yields imposed by climate change; 3) substantial areas of forests and woodlands may have to be converted to croplands to make up for lost opportunities to increase production – however such land conversion should consider the feedback effect of forest and woodland loss on water availability for irrigation (see also (*Duku et al.*, 2016b)). Our study shows that food security outcomes and forest and woodland conservation goals in Benin and likely the larger SSA region are inextricably linked together and require holistic management strategies that considers trade-offs and co-benefits.

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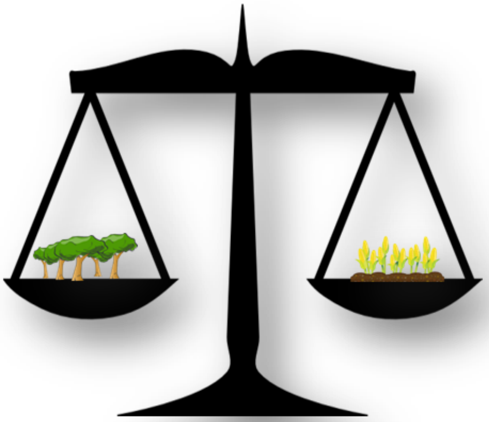
CHAPTER 6

SYNTHESIS



6.1 INTRODUCTION

The main objectives of this thesis were twofold. First, to further enhance the understanding of the dynamic interactions between food production and forest and woodland conservation goals, with a case study in Benin. Second, to further enhance the understanding of how hydrological ecosystem services can be captured in an accounting framework. To achieve this objective, I examined the impacts of climate change on several opportunities for agricultural intensification in Benin; identified and quantified synergies and trade-offs between food security pathways, and forest and woodland conservation; and demonstrated how ecosystem accounting as a standardized accounting framework can be used to monitor hydrological ecosystem services. The research questions were:

1. How can hydrological ecosystem services be spatially modelled in line with the ecosystem accounting framework?
2. How can the contributions of forests and woodlands to crop production be disentangled and quantified?
3. What are the likely impacts of climate change on opportunities for agricultural intensification?
4. How will climate change and population growth affect trade-offs between food availability and forest and woodland conservation?

This final chapter consists of five sections. In Sections 6.2 to 6.5, I synthesize the results of the previous chapters, reflect on how the research questions have been addressed, and discuss the broader implications for the larger sub-Saharan African (SSA) region as well as the methodological underpinnings. In Section 6.6, I present overall conclusions and recommendations arising from this thesis.

6.2 ECOSYSTEM ACCOUNTING FOR MONITORING ECOSYSTEMS AND ECOSYSTEM SERVICES

There is increasing interest in ecosystem accounting as a comprehensive tool for environmental monitoring and management (*Obst et al.*, 2013). The System of Environmental-Economic Accounting Experimental Ecosystem Accounting guideline (SEEA EEA) provides guidelines for setting up both biophysical and monetary ecosystem accounts (*UN et al.*, 2014a). The SEEA EEA is suited to monitoring the multi-directional linkages between ecosystems and economic activities because it inherently involves (1) tracking the contributions of ecosystems to economic activities through the measurement of ecosystem service flows; (2) tracking the changes in ecosystem assets through the measurement of ecosystem capacity which is a reflection of ecosystem condition and extent. The SEEA EEA is also particularly suited for monitoring hydrological ecosystem services. The generation of hydrological ecosystem services involves the dynamic interactions between individual environmental assets (e.g. land cover, water, and soil resources). Whereas accounting for these individual environmental assets is adequately captured in the Systems for Environmental-Economic Accounting - Central Framework (*UN et* *al.*, 2014b), the benefits accruing from the interactions are not fully addressed, leaving a critical gap in the monitoring of these services that underpin food and water security. The SEEA EEA by using a systemic view of relationships between the individual environmental assets provides a suitable framework for quantifying hydrological ecosystem services and linking them to human and economic activities. Capacity and flow of ecosystem services in general are key concepts in the SEEA EEA. Service flows are conceptualised as the contributions in time and space of ecosystems to productive and consumptive human activities leading to human benefits, whereas the capacity reflects ecosystem condition and extent at a point in time, and the resulting potential to provide service flows. Hence, service flows must be linked to identified beneficiaries whereas the concept of service capacity does not involve the identification of beneficiaries. Service capacity and service flow should be based on measurable indicators that have policy and management relevance. Indicators must also be able to represent cause–effect relations.

To make the SEEA EEA operational, scientific innovation in biophysical quantification of ecosystem services involving the empirical distinction between capacity and flow of services is required. For hydrological ecosystem services, particularly, selecting single indicators of service capacity that meet the above requirements and that sufficiently reflect ecosystem condition and their potential to provide service flows is difficult. This is because of the non-linear complex interactions among several ecohydrological processes that each relies on a suite of ecosystem components (*van Oudenhoven et al.*, 2012; *Villamagna et al.*, 2013). Nevertheless, empirical distinction and separate spatial characterization of hydrological ecosystem services is crucial for land and watershed management because high service capacity areas and high service flow areas may occur at different points in space (*Fisher et al.*, 2009). This thesis has shown that regardless of the challenges, it is feasible to empirically distinguish between capacity and flow of hydrological ecosystem services in line with the SEEA EEA. Modelling hydrological ecosystem services in line with the SEEA EEA requires a dynamic systems approach and must especially consider the following factors: (1) spatial heterogeneity of the ecosystem/watershed; (2) representation of relevant physical processes; (3) temporal resolution; and (4) required model accuracy. Even though these are factors that are also considered in hydrological modelling for other purposes, several unique challenges crop up within the context of ecosystem accounting. For example, in terms of spatial rigour, one of the challenges is hydrological modelling to support the development of ecosystem accounts at national scales while preserving landscape spatial heterogeneity. Another challenge is temporal resolution, especially intra-annual or seasonal considerations. Temporal variability in ecohydrological processes causes variations in hydrological ecosystem service provision over time within the same spatial unit (*Santhi et al.*, 2008; *Fisher et al.*, 2009). This can have significant impacts on the type of stakeholders and the values attached to hydrological ecosystem services (*Zhang et al.*, 2013). For example, seasonal or intra-annual temporal variability is needed in modelling water flow regulation service, which includes different aspects such as flood control and dry season streamflow augmentation. Furthermore, in order to link hydrology to land use change, hydrological models that allows modelling and integration of overland processes such as run-off and run-on and the deposition of soil particles in streams and waterways are more suitable (*Bosch et al.*, 2010; *Tetzlaff et al.*, 2010). These specific considerations and the need for timely information for monitoring purposes may lead to trade-offs between overall model accuracy and operational feasibility. I demonstrated the applicability of a widely used process-based spatially-explicit hydrological model, SWAT, which addresses the aforementioned factors, for supporting ecosystem accounting. The SWAT model is reconfigured with a grid-based landscape discretization and further enhanced to simulate water flow across the discretized landscape units. This provided increased flexibility in terms of setting up spatially disaggregated accounts to support decision-making at various levels. Overall, I have shown that consideration of the four factors allows for the empirical distinction and separate spatial characterization of capacity and service flow of hydrological ecosystem services. The dynamic systems modelling approach also allowed for identification and attribution of priority areas i.e. 1) high risk areas that would be affected by continued trends in watershed ecohydrology and 2) high production areas that are crucial for generation of services. Second, constructing ecosystem accounts for multiple years also allowed for monitoring trends in the capacity of ecosystems to provide services.

6.3 SYNERGIES AND TRADE-OFFS BETWEEN FOOD SECURITY PATHWAYS AND FOREST AND WOODLAND CONSERVATION

The relationship between food security and natural and semi-natural habitats is complex and multi-directional. For example, both land clearing and more intensive use of existing croplands could contribute to the increased crop production needed to meet growing food demand. Both land clearing and more intensive use of existing croplands could also lead to varied negative environmental feedbacks to crop production which are difficult to quantify because of the complexity (*Tilman et al.*, 2011). In this thesis, I identified and quantified some of the synergies and trade-offs between pathways to improving food security and forest and woodland conservation goals.

6.3.1 SYNERGIES

I examined the contributions of forests and woodlands to food availability through increased opportunities for irrigation expansion. Water flow regulation by forests and woodlands underpin increased opportunities for irrigation expansion. Water flow regulation leads to streamflow augmentation in the dry season and hence can provide critical resource input for crop production at a time when it is needed the most. However, this service is usually poorly understood and overlooked in land management decisions because of complexity of quantification (*Fisher et al.*, 2009; *Johnston and Russell*, 2011). *Fisher et al.* (2009) classified these types of ecosystem services as intermediate ecosystem services. Failure to consider the contributions of water flow regulation and intermediate ecosystem services in general in decision-making can lead to negative externalities and misguided policy (*Brown et al.*, 2007; *Fisher et al.*, 2008). This thesis offers a suitable approach to quantify the water flow regulation service of forests and woodlands and connect it to crop production.

First, I used a general modelling approach based on simulation experiments and I focussed specifically on the effects of forest and woodland conservation on dry season streamflow and not total annual streamflow. This disaggregation of streamflow into seasonal flow is very important to capture seasonal redistribution of streamflow, a key effect of forest conservation/deforestation in the tropics. Using this approach, I was able to address certain inherent biases in several studies that focussed on the effects of afforestation and reforestation on total annual streamflow and not dry season streamflow. These studies concluded that there are substantial losses in streamflow following afforestation and reforestation whereas forest clearing results in increased streamflow (*Andreassian*, 2004; *Farley et al.*, 2005). These conclusions may be true at the annual temporal scale, however, these studies fail to show the key effect of forests on seasonal redistribution of streamflow especially in the tropics. In the tropics, seasonal distribution of streamflow, especially dry season streamflow, are more relevant to food and water security than measures of total annual streamflow (*Bruijnzeel*, 2004; *Ellison et al.*, 2017).

Second, I have shown the applicability of copula functions to characterize and quantify the link between forest and woodland conservation and surface water availability for irrigation. Copulas are functions that join univariate distribution functions to form multivariate distribution functions and hence are able to model the dependence structure among random variables independent of the marginal distributions (*Akaike*, 1974; *Fang et al.*, 2014). The advantage of copula functions is that they allow for the quantification and relation of various attributes of water availability for irrigation. I decomposed surface water availability for irrigation into three attributes i.e. volume, duration and frequency. This is important because flow quantity, timing, and frequency are intrinsic attributes of water flow regulation that are affected by ecosystem properties as water moves through a landscape (*Brauman et al.*, 2007). I also quantified the risk deforestation poses on this ecosystem service in terms of recurrence interval of specific volume and duration of surface water available for irrigation. My computations took into consideration environmental water requirements of the river network and household water needs. These other sectoral water needs were prioritized over irrigation water requirements.

Forests and woodlands in SSA provide essential ecosystem services of local, national and global benefits. Forests and woodland areas of SSA are essential for amongst others biodiversity conservation (*Chidumayo and Gumbo*, 2010), water flow regulation (*Duku et al.*, 2016b), wood and non-wood forest products, desertification control and soil amelioration, carbon sequestration (*Lal*, 2004; *UNFCCC*, 2006), and grazing opportunities for livestock (*Judex and Thamm*, 2008). I have shown that agricultural intensification based on expanding irrigated crop production requires the protection of forests and woodlands in the headwaters of the rivers feeding the irrigation schemes. The results – the link between deforestation and reduced opportunities for irrigation - will apply to a substantial part of sub-Saharan Africa, where dry forests and woodlands cover an area of 17.3 million km^2 in SSA and are the most dominant land-use types in the headwaters of all the major river basins in SSA (Senegal river basin, Niger, Volta, Chad, Congo, Nile, Zambezi, Limpopo and Orange) (*Chidumayo and Gumbo*, 2010). However, the specific relation between deforestation and irrigation water availability will vary with the watershed as a function of the size and shape of the watershed, specific rainfall patterns etc. For example, in Kenya it is estimated that deforestation at a rate of 5,000 ha per annum in the period 2000 to 2010 reduced the total availability of water (for all purposes, including irrigation) by 62 million m^3 per annum by 2010 and resulted in a loss of approximately 25 million US dollars in present value that would have accrued from agricultural output alone if the forests had been conserved (*UNEP*, 2012). Forest management institutions in many SSA countries currently remain weak, leaving forests highly vulnerable to clearance and degradation (*Keenan et al.*, 2015). As a result of this and other factors e.g. population growth, the rate of deforestation in SSA countries still remains high (*FAO*, 2015b). Hence, there is the need for a broader and concerted approach at forest and woodland conservation in SSA. It is quite apparent that irrigation opportunities obtained from water flow regulation by forests and woodlands is not the primary factor to be considered in forest and woodland protection policies in most if not all countries in SSA. Current paradigms in forest and woodland conservation are driven mainly by carbon sequestration and storage benefits (*Ellison et al.*, 2017). Nevertheless, this study provides further incentive for a broader and comprehensive approach for forest and woodland protection aimed at securing benefits including irrigation opportunities. Indeed, *Ellison et al.* (2017) call for a reversal of policy paradigms, from a carbon-centric approach to one that treats the hydrologic, climate-cooling, and rainfall cycle maintenance effects of trees and forests as the first order of priority. Such regional effects of forest and woodland conservation have been overlooked in management and even science because of inherent complexity in quantification. I have also shown in Chapter 4 that even if the current extent of forests and woodlands are maintained in the watershed, the irrigation potential of the watershed will be reduced by at least 50% in the coming decades due to climate change. Thus the combined effects of climate change and deforestation on irrigation opportunities in the watershed is likely to be more substantial than computed in Chapters 3 and 4 separately. Forest and woodland conservation and re-afforestation programmes can rather be used as climate adaptation and mitigation measures to reduce agricultural vulnerability to climate change.

6.3.2 TRADE-OFFS

There is increasing concern and awareness of the loss of vital ecosystem services as a result of deforestation, which in SSA is driven mainly by cropland expansion. In the coming decades, increasing food demand of a rapidly growing population and climate change are likely to increase the pressure on forests and woodlands for conversion to croplands. For example, a recent study showed that without cropland expansion it will not be feasible to meet 2050 SSA cereal demand (*van Ittersum et al.*, 2016). In Chapters 4 and 5 of this thesis, I have shown that opportunities to increase crop production on existing cropland areas are likely to diminish with climate change in Benin. Furthermore, most SSA countries are committed to achieving food self-sufficiency and reduce reliance on food imports. As a result of the interplay of these factors, trade-offs are bound to occur between competing food security and broader environmental objectives – with the specific notion that these environmental considerations also have repercussions for food production. Quantifying and understanding the dynamics and the nature of these trade-offs is important for prioritizing and targeting management interventions. Trade-off analysis has become an increasingly important approach for evaluating system level outcomes of agricultural production (*Klapwijk et al.*, 2014) and for identifying a set of biophysical options to keep production within safe and acceptable operating space.

In Chapter 5, I quantified trade-offs between future per capita food availability and forest and woodland conservation at different levels of yield increases. The impact of cropland expansion on ecosystem services is not explicitly quantified, but rather forest and woodland extents are used as proxies. First, I examined if a continuation of current yield increases in existing production areas will be adequate to maintain current levels of food availability in the coming decades. This was based on extrapolation of linear trends in crop yields over the past two and half decades. The yield projections are corrected using simulated water-limited yield potential in order to keep projections within what is biophysically possible (*van Ittersum et al.*, 2013). This analysis was carried out under different combinations of climate change and population growth scenarios. If current yield increases are adequate, this thesis shows potential areas of forest and woodland that would have been conserved. For scenarios where continuation of current yield increases is insufficient to maintain current levels of food availability, this thesis quantifies (1) the rate of yield increases in existing production areas required taking into consideration biophysical limitations of yield increases as a result of climate change and/or; (2) the land area required to maintain current levels of per capita food availability for each crop. The results show that food security outcomes and forest and woodland conservation goals in Benin are inextricably linked together and require holistic management strategies that consider trade-offs and co-benefits.

Driven largely by rapid population growth, cropland expansion is increasingly exerting enormous pressure on natural ecosystems all across SSA, replacing and fragmenting savannahs, woodlands and forests. For example, between 1975 and 2013, the area covered by crops doubled in West Africa, reaching a total of 1,100,000 km^2, or 22.4%, of the land surface (*CILSS*, 2016). Benin for instance recorded one of the highest rates of cropland expansion over this period i.e. about 6% per annum (*CILSS*, 2016). The implications of these land-use changes are complex and cover broad spatial and temporal scales and are likely to pose major risks to livelihoods in SSA including food security. Deforestation affects all three key dimensions of food security, i.e. availability, access and utilization. In this thesis, I examined the effects of deforestation on food availability through its feedback on irrigated crop production. In Chapter 3, I have shown that at the prevailing forest and woodland extent in the Upper Ouémé watershed in Benin, the loss of about 40 ha of forest and woodland area reduces the irrigation potential by 1 ha. And that increasing deforestation will mean that more forest and woodland areas are needed to support 1 ha of irrigated crop production. Deforestation in general increases the risk of streamflow droughts in the dry season. This will also affect water availability for livestock, people and ecosystems. There are other negative feedbacks of deforestation on food access and utilization. Deforestation erodes the livelihood of the vast rural population in SSA and hence affects food access. Food access is determined mainly by household and individual-level incomes. During non-agricultural seasons, many households in rural SSA engage in harvesting wood and Non-Timber Forest Products (NTFP) from natural forests and woodlands to sustain household livelihoods (*Pimentel et al.*, 1997; *Shackleton and Shackleton*, 2004). *Barrett et al.* (2001) showed that such non-farm income generating activities are positively correlated to household welfare indicators across most of rural Africa. In rural SSA, food utilization depends on availability of fuelwood in addition to clean water and good health (*Wheeler and von Braun*, 2013). Deforestation from cropland expansion reduces availability of fuelwood for cooking even though the collection of fuelwood in itself leads to deforestation (*WFP*, 2012). Deforestation in SSA has also been recorded to result in an increase in incidence of malaria and/or its vectors (*Patz et al.*, 2004).

6.4 THE IMPACT OF CLIMATE CHANGE ON OPPORTUNITIES FOR AGRICULTURAL INTENSIFICATION

Given the need to protect natural ecosystems, agricultural intensification as a production pathway is considered most desirable (*Foley et al.*, 2011; *Garnett et al.*, 2013). The degree of success of agricultural intensification options will be one of the major determining factors of the level of pressure that natural ecosystems, especially forested and woodland areas in SSA are likely to come under. *van Wart et al.* (2013a) and *Grassini et al.* (2013) noted that the extent to which increased food production requires expansion of cultivated area will be determined largely by crop yield potential. In addition to socioeconomic and technical challenges facing agricultural intensification in SSA, climate change is likely to affect the degree of success of intensification options. In this thesis, I examined the impacts of climate change on three intensification options 1) increasing crop yields in rainfed areas; 2) Increasing cropping intensity in rainfed areas and 3) irrigation expansion.

6.4.1 INCREASING CROP YIELDS IN RAINFED AREAS

Water-limited yield potentials and yield gaps provide a measure of biophysical opportunities available for further yield increases in rainfed areas. Given that rainfed cultivation is predominant in Benin and SSA in general (*You et al.*, 2014), quantifying water-limited yield potentials are important for estimating the food production capacity and to help formulate policies and research priorities for improving food security. They are also important indicators of the need for genetic improvements to raise yield ceilings and the need for irrigation to overcome water-limitations on yield in order to ensure food security. Changes in temperature and precipitation, especially over a growing season are likely to affect the water-limited yield potentials (*Godfray et al.*, 2010). Reductions in water-limited yield potentials erode substantial opportunities to increase yields under current agronomic management conditions. In Chapter 5, I quantified water-limited yield potentials of maize, cassava and yam in Benin under current climatic conditions and two contrasting climate change scenarios. In this section, I discuss the implications of first, my approach and second results for the larger SSA region.

First, this thesis has shown the applicability of a loose-coupling approach between a process-based, spatially explicit, ecohydrological model, SWAT, (*Gassman et al.*, 2007; *Arnold et al.*, 2010) and a crop growth model for quantifying water-limited yield potentials. Generally, there are three approaches for quantifying water-limited yield potentials 1) model simulations; 2) field experiments; and 3) maximum farmer yields (*Lobell et al.*, 2009). Simulation models are considered to provide a more accurate estimate of water-limited yield potentials at large scales and are useful for analysing the impact of climate change on crop yield potentials in general (*Lobell et al.*, 2009). Water-limited yield potential is influenced by soil properties and dependent hydrological processes and these must be considered for a more accurate quantification (*Lobell et al.*, 2009; *van Ittersum et al.*, 2013). This makes quantifying water-limited yield potential using simulation models, particularly more difficult compared with quantifying yield potential, for which it is assumed that non-water limiting conditions exist (*Lobell et al.*, 2009). Ideally, spatially and temporally continuous measurements of soil moisture are needed for accurate simulation of water-limited yield potential. In many regions of SSA, such data are not available at scales beyond relatively small experimental plots. This thesis has shown how ecohydrological models such as SWAT can be used to provide needed soil moisture data in order to improve the accuracy of simulated water-limited yield potentials. I simulated the spatiotemporal dynamics of soil moisture content using detailed data on soil properties and landscape features including bulk density, soil texture, rooting depth, root-zone water holding capacity, topography, and land cover. The SWAT model was calibrated and validated at different spatial scales using long-term daily streamflow data and not soil moisture data because of inadequate data. Even though model validation using soil moisture data would have been more suitable, streamflow represent system level hydrological responses and accurate simulation of streamflow dynamics is generally indicative of satisfactory representation of underpinning processes such as soil moisture.

Second, the results are instructive for a large part of SSA. In this thesis, I have shown that water-limited yield potentials of major food crops in Benin i.e. maize, cassava and yam, are likely to decrease substantially as a result of the shortening of the length of growing period in the coming decades. Just like Benin, *Jones and Thornton* (2009) showed that in a large part of SSA, the length of growing period is likely to be shortened as a result of climate change by 2050. *Thornton et al.* (2011) also showed that by the end of this century there is likely to be substantial reductions in the length of growing period all across SSA with the exception of highland areas in Kenya and Tanzania. The shortening of the length of growing period is likely to result in reductions in water-limited yield potentials in these areas. They are also likely to lead to increased crop failure rates, i.e. the probability of complete crop loss due to drought in rainfed systems. The effect on crop production is that the exploitable yield gaps in rainfed areas that currently exist are likely to diminish i.e. even without growth limitations of nutrients, pests and diseases, crop yields are likely to plateau earlier. Diminishing exploitable yield gaps are likely to affect strategies to meet the dual objectives of improving food security and protecting natural ecosystems such as forests and woodlands in the coming decades. Currently, even though crop yields in Benin and the larger SSA region are considerably lower, it is generally acknowledged that there is considerable scope for further yield increases. For example, maize yields in Benin and most of SSA are currently about 20% of water-limited yield potentials (*van Ittersum et al.*, 2016). In Benin and most of SSA, these low farm yields are generally as a result of nutrient deficiencies more than any other biophysical factor (*Mueller et al.*, 2012). However, this thesis has shown that in the coming decades, better nutrient management and biotic control in rainfed areas is unlikely to be adequate to increase yields substantially in SSA. Despite the effect of climate change, yield ceilings in rainfed production systems can be raised with the introduction of higher-yielding cultivars and irrigation expansion (*Gornall et al.*, 2010; *Hall and Richards*, 2013). Increasing water-limited yield potentials is complex and challenging nonetheless there remains unexploited opportunities (*Hall and Richards*, 2013).

6.4.2 INCREASING CROPPING INTENSITY IN RAINFED AREAS

Increasing cropping intensity, i.e. growing two or more crops per year, can boost crop production in existing croplands and reduce pressure to expand cultivated areas. Increased cropping intensity in many areas of the world has been achieved through irrigation expansion (*Alexandratos and Bruinsma*, 2012; *Liu et al.*, 2013a). However, given that more than 95% of cropland areas in Benin and SSA are under rainfed cultivation (*You et al.*, 2011), understanding spatial patterns in cropping intensity in these areas is important for identifying target regions for intensification, where trade-offs between crop production and ecosystem protection can be minimized (*Stephan et al.*, 2016). Currently, much of the research focus on the impact of climate change in SSA has been on crop yield responses (e.g. *Muller et al.*, 2011; *IIASA/FAO*, 2012; *Wheeler and von Braun*, 2013; *World Bank*, 2013; *Connolly-Boutin and Smit*, 2016; *Palazzo et al.*, 2017). This may lead to an underestimation of the impact of climate change on crop production potential because climate change is also likely to affect other agricultural intensification options such as cropping intensity and opportunities for irrigation expansion. Hence analysing the impact of climate change on the potential to increase cropping intensity in rainfed areas is important for agricultural development planning and for targeted management interventions.

In this thesis, I used a generic framework based on the AEZ methodology (*IIASA/FAO*, 2012) to spatially delineate and analyse the impact of climate change on existing cropland areas and potential cropland areas that can support the cultivation of two crops per year in Benin. The delineation was based on the length of growing period and did not differentiate between specific cropping systems that can be supported. A consideration of specific cropping systems across different agro-ecological zones would have been more suitable. However, in Benin and many parts of SSA land use data with this level of detail are currently not available. Obtaining such information is complicated by the small plot sizes and cropping patterns varying from year to year. Moreover, in SSA, the length of the growing period is a key biophysical constraint to increasing cropping intensity in rainfed systems. This thesis has demonstrated how the impact of climate change on opportunities to increase cropping intensity can be analysed in order to support agricultural development planning.

In Benin, the average cropping intensity of staple crops such as maize, yam, cassava and sorghum is 1.5 i.e. the harvested area is one and half times greater than the physical area devoted to the cultivation of each crop (*You et al.*, 2014). This can be attributed to both sequential cropping and intercropping. The results in chapter 4 show that whereas there is currently substantial scope to increase cropping intensity in existing production areas in Benin, in the coming decades, climate change will substantially erode these opportunities due to reductions in the length of growing period. Therefore, strategies to reduce impacts of climate change on crop production should also seek to limit production losses from changes in cropping intensity. The results are instructive for a large part of the SSA region where growing two or more crops per year is a common indigenous management practice (*Waha et al.*, 2013). In these areas sequential cropping systems include maize-based systems, cassava-based systems, groundnut-based systems and millet based systems (*Waha et al.*, 2013). *Deepak and Jonathan* (2013) showed that in these areas there is currently substantial scope to increase cropping intensity provided that soil fertility can be maintained or enhanced through nutrient inputs. However, just like Benin, *Jones and Thornton* (2009) and *Thornton et al.* (2011) showed that in a large part of SSA, there is likely to be substantial reductions in the length of growing period as a result of climate change. Even though the degree of reductions in the length of growing period may vary in different parts of SSA, generally they are likely to either lead to decrease in areas that can support cultivation of two or more crops or a shortening of the fallow period, which will require improved agronomic management. The results underlie the need for identification of practicable adaptation strategies for cropping systems to climate change, especially through the choice of adequate cropping system and crop cultivar (*Lobell et al.*, 2008; *Waha et al.*, 2013).

6.4.3 IRRIGATION EXPANSION

Irrigation expansion is an important contributor to crop yield increases and increased cropping intensity. Crop yields under irrigated conditions are generally much higher than that under rainfed conditions, hence *Alexandratos and Bruinsma* (2012) noted that even if irrigated yields would remain unchanged in the future, a shift from rainfed to irrigated production systems would lead to an increase in average yields. The need for irrigation expansion in order to ensure food security is even more critical given the impact of climate change on water-limited yields (Chapter 5) and cropping intensity in rainfed areas (Chapter 4). *van Ittersum et al.* (2016) also concluded that even without considering climate change, irrigation expansion may be required in order to meet 2050 SSA cereal demand on existing production cropland. Renewable water resources that underpin irrigation expansion are also affected by climate change. Climate change affects the water cycle through changes in temperature, the timing and magnitude of precipitation, soil moisture, run-off, the magnitude and frequency of extreme events, and a number of secondary effects (*Strzepek and Boehlert*, 2010).

In Chapter 4, I used copula functions and the concept of irrigation potential to quantify the impact of climate change on renewable water resources that underpin irrigation expansion (Chapter 4). First, I showed the applicability of copula functions for quantifying the impact of climate change on water availability for irrigation in terms of frequency of streamflow droughts. Changing temperature and precipitation as a result of climate change can affect water availability for food production through episodes of drought propagated through the water cycle i.e. from meteorological drought to streamflow drought. Droughts are complex, stochastic phenomena with multiple attributes including duration, severity and frequency and probability theory as well as stochastic process methods are more suitable approaches for characterizing them (*Shiau*, 2006; *Shiau and Modarres*, 2009). There exists a correlation among drought attributes and multivariate analysis provides a more complete characterization. However, applying traditional multivariate frequency analysis methods is unsuitable because drought attributes may require different distribution functions (*Shiau*, 2006; *Song and Singh*, 2010). Copula functions, hence provide a suitable approach for characterizing drought characteristics and to derive the joint distribution of drought attributes (*Mishra and Singh*, 2010). Over the last decade, copulas have emerged as a method for addressing multivariate problems in several disciplines including hydro-climatology (*Salvadori et al.*, 2007). The use of copula functions in this thesis allowed for the quantification and the relation of severity and duration of streamflow droughts to the frequency of occurrence under different climate change scenarios. The approach used in this thesis can be applied in large areas of SSA where water resource planning and allocation, including irrigation planning, design and management requires a detailed knowledge of reliability of water flows.

Second, I used the concept of irrigation potential to assess the impact of climate change on renewable water resources that underpin irrigation expansion. The irrigation potential calculations were mainly from a water resources perspective and reflect the fact that renewable water resources that are adequate for irrigating any given amount of land today may not be so in the future as a result of climate change. The analyses took into consideration competing demands of surface water resources for household use and for supporting the riverine ecosystem functioning. Water allocation for household use and for supporting riverine ecosystem functioning were prioritized over allocation for irrigation purposes. Overall, this thesis has illustrated how opportunities for irrigation expansion diminish as a result of climate change. In practice, given that there is very little irrigated area in Benin and large parts of SSA, the impact of climate change on renewable water resources may not directly translate into production losses in the short-term. In the long-term, however, water resource allocation for irrigation needs may be affected as a result of the combination of increased frequency of episodes of streamflow droughts and increased water demand by other sectors.

In this thesis, I focussed on the impacts of climate change on surface water availability for irrigation. However, unlike surface water resources, groundwater resources are more resilient to moderate levels of climate change and hence can provide a buffer against the impacts of climate change on water availability for irrigation (*Kundzewicz and DÖLl*, 2009). Currently, groundwater provides enormous potential for irrigation expansion across the African continent. *Altchenko and Villholth* (2015) estimated that total groundwater availability for irrigation across Africa ranges from 692 to 1644 km^3 depending on the scenario of environmental water requirement. This translates into a total irrigation potential ranging from 45 million to 105 million ha corresponding to between 21% and 48% of the total cropland area. However, there exists substantial spatial heterogeneity in the potential of groundwater resources to support irrigation even at the national and sub-national scales as a result of climate, land cover, soil and other biophysical factors (*Altchenko and Villholth*, 2015). Hence, for agricultural development planning, groundwater resources must be evaluated within the unique national and sub-national developmental, climatic and biophysical contexts.

6.5 THE IMPACT OF POPULATION GROWTH ON FOOD SECURITY AND FOREST AND WOODLAND CONSERVATION

Rapid population growth has undercut gains made in food production in SSA and has led to cropland expansion at the expense of forests and woodlands. The population in many SSA countries has been growing at a faster rate than crop yield increases (*FAO*, 2016b). As a result, per capita availability of domestically produced food has not changed substantially at the continental scale over the last five decades (*Pretty et al.*, 2011). In fact, *Adesina* (2010) and *Liu et al.* (2008) noted that per capita agricultural output in Africa has declined in recent decades, especially for staple crops such as maize, rice, cassava and sorghum. The total population of SSA grew by 2.7% per annum between 1990 and 2015. As a result, even though the prevalence of undernourishment declined during this period, the actual number of undernourished people increased by 44 million (*FAO et al.*, 2015). In Benin, the population has been growing at a faster rate than yield increases in maize, cassava and yam. Whereas the population grew at a rate of about 4% per annum between 1990 and 2014, yields of maize, cassava and yam increased at a rate of 2.2%, 2.8% and 1.5% per annum respectively (*FAO*, 2016b). At the same time, rates of cropland expansion, which is mostly at the expense of forests and woodlands, increased substantially. For example, in West Africa, croplands have more than doubled since 1975 to cover about 23% of total land area (*CILSS*, 2016). Since 1975, annual rates of cropland expansion, mostly at the expense of forests and woodlands, has been more than 5%, in countries such as Benin, Togo, Mauritania, Burkina Faso (*CILSS*, 2016).

In this thesis, I used a scenario-matrix approach (*van Vuuren et al.*, 2013) to analyse the effect of future population growth on trade-offs between per capita food availability and forest and woodland conservation in Benin under different climate change scenarios. The scenario-matrix approach involved analysing per capita food availability under combinations of two contrasting population growth scenarios as described under the Shared Socio-economic Pathways (SSP)(*O'Neill et al.*, 2013) and two future climate change scenarios corresponding to Representative Concentration Pathways (RCP) (*Moss et al.*, 2010). The population growth scenarios represented continued increases at 3% per annum and 1.3% per annum. I showed that population growth more than climate change is likely to determine the level of trade-offs between food availability and forest and woodland conservation. I also showed that without further cropland expansion, yield increases at the current rate are unlikely to even maintain current levels of per capita food availability if population also continues to grow at the current rate regardless of the RCP scenario. To maintain current levels of per capita food availability without further cropland expansion, crop yields will have to increase at a faster rate than has been recorded over the past two and half decades. However, in most cases the rate of yield increases required is beyond what is biophysically possible under rainfed conditions increasing the likelihood of cropland expansion at the expense of forests and woodlands. In the coming decades, the population of SSA is expected to continue to grow rapidly and it is projected to double by 2050 compared to 2015 estimates (*UN*, 2015b) leading to rising food demand. For example, the cereal demand for ten countries in SSA is projected to triple by 2050 compared to 2010 estimates, assuming an average population growth of 1.5% per annum and modest dietary changes in these countries (*van Ittersum et al.*, 2016). Rising food demand in SSA means that expansion of croplands is likely to increase at the expense of forests and woodlands as has been witnessed in the past. *Hilderink et al.* (2012) projects a 29% reduction in forest cover in SSA by 2030 as a result of agricultural expansion.

6.6 CONCLUSIONS AND RECOMMENDATIONS

This thesis provides novel and detailed insights on the dynamic interactions between food production and natural and semi-natural ecosystems. It also enhances the understanding of how these interactions will be affected by climate change and population growth. In addition, this thesis demonstrates how hydrological ecosystem services can be captured in an accounting framework. In particular, I show that:

1. the integration of hydrological ecosystem services into an accounting framework can provide relevant information at appropriate scales suitable for decision-making. It is empirically feasible to distinguish between service capacity and service flow of hydrological ecosystem services. This requires appropriate decisions regarding physical and mathematical representation of ecohydrological processes, spatial heterogeneity of ecosystems, temporal resolution, and required model accuracy. The challenge is dealing with computational power and modelling to support the development of ecosystem accounts at national scales while preserving landscape

spatial heterogeneity. This may lead to trade-offs between operational feasibility and model accuracy.

2. opportunities for irrigation expansion in SSA depend on conservation of forests and woodlands in the headwaters of the rivers feeding the irrigation scheme. The relation between forest and woodland extent and irrigation potential is, however, not linear, and more hectares of forest and woodland are needed to support 1 ha of irrigated crop production with increasing deforestation. The specific relation between deforestation and irrigation water availability across SSA will vary with the watershed as a function of size and shape of the watershed, specific rainfall patterns, etc.
   In the Upper Ouémé watershed in Benin, conservation of current forest and woodland cover is needed to allow the development of 80% (15,000 ha) or 71% (20,000 ha) of the irrigation potential in the dry season depending on the desired level of water flows to support riparian ecosystem functioning;
3. opportunities for agricultural intensification in SSA are likely to diminish with climate change, hence increasing pressure to expand cultivated areas in order to meet increasing food demand. Climate change will lead to substantial reductions in; exploitable yield gaps for major food crops, rainfed cropland areas that can support the cultivation of two or more crops per year, and water availability for irrigation expansion.
4. (i) maize and yam yields will have to increase at a faster rate than has been recorded over the past two and half decades in order to maintain current levels of per capita food availability;
   (ii) major investments in irrigation and higher-yielding cultivars, especially for cassava, will be required to raise the yield ceiling in order to overcome the biophysical limitations on yields imposed by climate change;
   (iii) substantial areas of forests and woodlands may have to be converted to croplands to make up for lost opportunities to increase production – however such land conversion should consider the feedback effect of forest and woodland loss on water availability for irrigation.

Based on the results of this thesis, four main recommendations to help address the dual challenge of food security and ecosystem protection in Benin and the larger SSA region are made.

(I) PROMOTE A PRECAUTIONARY APPROACH TO FOREST AND WOODLAND CONSERVATION

In this thesis, I have shown how forests and woodlands contribute to food security through increased opportunities for irrigation expansion (Chapter 3). I have also shown how rapid population growth and climate change are likely to exert increasing pressure on forests and woodlands for conversion to croplands (Chapter 5). However, the dynamic interaction between food security outcomes and forest and woodland goals are complex and go beyond what has been shown in this thesis. Some of them were highlighted in Chapter 1 and discussed in a broader context in Chapter 6. The effects of deforestation on biodiversity, ecosystem services and livelihoods in general at local, national and global scales have been documented in several studies (*Bruijnzeel*, 2004; *Lal*, 2004; *Foley et al.*, 2005; *UNFCCC*, 2006; *Chidumayo and Gumbo*, 2010; *Ellison et al.*, 2017). However, there still remain major scientific uncertainties about the effects of the loss of forest biodiversity and ecosystem services as a result of the complexity of the ecological system. Hence, there is the need for adoption and tight integration of the precautionary approach to forest and woodland management in SSA. The precautionary approach involves identifying and protecting critical habitats, even though there is currently uncertainty in the specific services they deliver. This approach inherently recognises scientific and technical limitations and promotes regulatory action in the absence of full evidence of a cause-effect relationship.

(II) PROMOTE CROSS-SECTORAL POLICY COHERENCE AND CONSULTATIONS

Given the complexity and multi-directional nature of the interactions between food production and natural and semi-natural ecosystems, there is the need for active and greater policy coherence and consultation across different intergovernmental agencies and ministries in most SSA countries. Currently, ministries and intergovernmental agencies responsible for setting policy agenda for natural resources management are fragmented based on the specific natural resources (e.g. lands, forestry, water resources etc.) and also are disconnected from agricultural ministries and agencies. To promote policy coherence and consultations across these different sectors, multi-stakeholder platforms can be created. Multi-stakeholder platforms help institutionalize cross-sectoral policy consultations where planning and implementation of holistic management strategies that consider trade-offs and co-benefits between food production and forest and woodland conservation can be considered. Multi-stakeholder platforms can be established at different levels of policy and decision-making, from local to national levels.

(III) PROMOTE THE DEVELOPMENT OF SATELLITE ECOSYSTEM ACCOUNTS CONSISTENT WITH NATIONAL ACCOUNTS

Several countries in SSA have pledged to support the development of natural capital accounts that include ecosystem accounts. But very few countries have actually committed to the development of these accounts. Developing satellite ecosystem accounts consistent with national accounts help to elucidate the contributions of ecosystems to economic activities recorded in the national accounts, as well as for capturing exchange values of some ecosystem services that are not included in these accounts. Ecosystem accounts are not developed based on specific policy goals, but rather they are information systems that can be used to support policy formulation, monitoring and evaluation.

(IV) IDENTIFY, EVALUATE AND IMPLEMENT ADAPTATION AND RESILIENCE MEASURES TO REDUCE AGRICULTURAL VULNERABILITY TO CLIMATE CHANGE

My thesis has shown that climate change will affect all the major pathways for increasing crop production on existing croplands. Adaptation and resilience measures that can help reduce vulnerability of agricultural systems to climate change should therefore be identified, evaluated and implemented. This will require detailed spatial analysis at local and regional scales to identify the most vulnerable areas to climate change in order to prioritize the allocation of limited resources. Adaptation and resilience measures can be in capacity building and technology transfer, research, value chains, policy and governance. In this thesis, I have shown that one resilience measure is to conserve forests and woodlands in order to maintain opportunities for irrigation expansion especially during the dry season.

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APPENDIX

APPENDIX I: ADDITIONAL INFORMATION FOR CHAPTER 3

Table A1. P-values of Kolmogorov-Smirnov goodness of fit tests for marginal distributions

Forest and woodland extent	Duration	Volume
	Q50 scenario	
Prevailing forest and woodland cover	Logistic (0.751)	Exponential (0.716)
Conservation of only protected forest and woodland areas	Lognormal (0.703)	Lognormal (0.984)
Complete deforestation	Lognormal (0.865)	Lognormal (0.623)
	Q75 scenario	
Prevailing forest and woodland cover	Logistic (0.856)	Lognormal (0.785)
Conservation of only protected forest and woodland areas	Lognormal (0.868)	Lognormal (0.437)
Complete deforestation	Lognormal (0.714)	Lognormal (0.851)

Table A2. Akaike information criteria (AIC) goodness of fit results for different copula functions. The lower the AIC value the better the fit.

Copula functions		Q50 scenario	
	Prevailing forest and woodland areas	Conservation of only protected forest and woodland areas	Complete deforestation
Clayton	67.5	37.4	34.2
Frank	70.6	40.3	37.9
Gumbel	69.3	41.6	39.1
		Q75 scenario	
Clayton	43.6	34.8	39.0
Frank	47.9	39.1	41.2
Gumbel	50.3	45.8	46.7

APPENDIX II: ADDITIONAL INFORMATION FOR CHAPTERS 4 AND 5

Table A3. General Circulation Models

General Circulation Model	Institution
BCC-CSM 1.1	Beijing Climate Center, China Meteorological Administration
BCC-CSM 1.1(m)	Beijing Climate Center, China Meteorological Administration
CSIRO-Mk3.6.0	Commonwealth Scientific and Industrial Research Organisation and the Queensland Climate Change Centre of Excellence
FIO-ESM	The First Institute of Oceanography, SOA, China
GFDL-CM3	Geophysical Fluid Dynamics Laboratory
GFDL-ESM2G	Geophysical Fluid Dynamics Laboratory
GFDL-ESM2M	Geophysical Fluid Dynamics Laboratory
GISS-E2-H	NASA Goddard Institute for Space Studies
GISS-E2-R	NASA Goddard Institute for Space Studies
HadGEM2-ES	Met Office Hadley Centre
IPSL-CM5A-LR	Institut Pierre-Simon Laplace
IPSL-CM5A-MR	Institut Pierre-Simon Laplace
MIROC-ESM	Atmosphere and Ocean Research Institute (The University of Tokyo), National Institute for Environmental Studies, and Japan Agency for Marine-Earth Science and Technology
MIROC-ESM-CHEM	Atmosphere and Ocean Research Institute (The University of Tokyo), National Institute for Environmental Studies, and Japan Agency for Marine-Earth Science and Technology
MIROC5	Japan Agency for Marine-Earth Science and Technology, Atmosphere and Ocean Research Institute (The University of Tokyo), and National Institute for Environmental Studies
MRI-CGCM3	Meteorological Research Institute
NorESM1-M	Norwegian Climate Centre

SUMMARY

Countries in sub-Saharan Africa (SSA) face a major dilemma. How can they produce enough food in a changing climate to feed an increasing population while protecting natural forests and woodlands that provide a wide array of ecosystem services beneficial to livelihoods? In terms of policy initiatives, this dilemma involves reconciling pledges under Sustainable Development Goal (SDG) 2 i.e. end hunger and achieve food security, with that under SDG 15 i.e. protect, restore and promote sustainable use of terrestrial ecosystems. To this end, agricultural intensification as a food production pathway is widely considered desirable. In response African countries are investing in or committing to invest in agricultural intensification. For example, one of the targets in the Africa Water Vision 2025, adopted by the African Union member states, is to double the size of irrigated areas in this region by 2025.

Yet, there remain uncertainties regarding the impact of climate change and land use change on opportunities for agricultural intensification in SSA. Furthermore, given rising food demand associated with rapid population growth, the degree of success of agricultural intensification options in meeting this demand needs to be evaluated. The degree of success of agricultural intensification options will be one of the major determining factors of the level of pressure that natural ecosystems are likely to face. Trade-offs between competing food security and broader environmental objectives are thus likely. Quantifying and understanding the dynamics and the nature of these trade-offs is important for prioritizing and targeting management interventions. These issues need to be addressed at national and local levels in the unique biophysical and developmental context. In addition, in recent times, much of the research focus on the impact of climate change on agricultural intensification options has been on crop yield responses. However, failure to consider the impact of climate change on other intensification options may lead to underestimation of the vulnerability of SSA agriculture as well as misguided policy initiatives.

Furthermore, there is increasing demand for integrated information that can link analytical and policy frameworks for environmental sustainability, human well-being and economic growth and development. In this regard, the Systems for Environmental-Economic Accounting Experimental Ecosystem Accounting (SEEA EEA) framework is being increasingly used. The SEEA EEA is an initiative by the statistical community in collaboration with research groups. To make this framework operational, scientific innovations are needed, especially regarding biophysical modelling of ecosystem services. Several recent studies have demonstrated the applicability of ecosystem accounting for monitoring a wide range of services. Yet, there is still limited experience in accounting for hydrological ecosystem services in both physical and monetary terms. Of increasing challenge is how to model hydrological ecosystem services with adequate spatiotemporal detail and accuracy at aggregated scales in line with the SEEA EEA and how to conceptualise and empirically distinguish between hydrological ecosystem services flow and capacity.

Understanding hydrological ecosystem services is key to understanding the trade-offs between land conversion to croplands and ecosystem services supply from natural and semi-natural ecosystems.

Therefore, the objectives of this thesis are twofold. First, is to further enhance the understanding of the dynamic interactions between food production, and natural and semi-natural ecosystems with a case study in Benin. Second, is to further enhance the understanding of how hydrological ecosystem services can be captured in an accounting framework. These objectives are addressed through a critical reflection of both the conceptual and methodological frameworks required. These include the ecosystem services and ecosystem accounting frameworks and the development and application of spatially explicit ecohydrological and agro-ecological models. To address these objectives four research questions were formulated:

1. How can hydrological ecosystem services be spatially modelled in line with the ecosystem accounting framework?

2. How can the contributions of forests and woodlands to crop production be disentangled and quantified?

3. What are the likely impacts of climate change on opportunities for agricultural intensification?

4. How will climate change and population growth affect trade-offs between food availability and forest and woodland conservation?

Two study areas, both in Benin are used to address the objectives of this thesis. The larger area covers the entire Ouémé river basin and southwestern parts of the country that lie outside the river basin. This study area covers 55,000 km^2. Second is the relatively small area, the upstream portion of the Ouémé river basin henceforth referred to in this thesis as the Upper Ouémé watershed. This second study area covers an area of approximately 14,500 km^2. The Upper Ouémé watershed located in central Benin is used as a case study to address research questions 1, 2, and 3. Results and outputs are then upscaled to the entire Ouémé river basin and southwestern parts of the country that lie outside the river basin in order to address research question 4. My thesis consists of six chapters. Chapters 1 and 6 have been written to provide a background, broad overview and synthesis of the research questions in order to address the main objectives of this thesis. Chapters 2 and 3 address research questions 1 and 2 respectively. Chapters 4 and sections of chapter 5 address research question 3, and chapter 5 also addresses research question 4. Chapters 2, 3, 4 and 5 have been written as independent research papers in cooperation with the listed co-authors.

In chapter 2, I examine how a spatially explicit ecohydrological model can be used to analyse multiple hydrological ecosystem services in line with the ecosystem accounting framework. The hydrological ecosystem services include crop water supply for rainfed agriculture, household water supply (both groundwater supply and surface water supply), water purification, and soil erosion control. The Soil Water and Assessment Tool (SWAT), which has been configured with a grid-based landscape discretization and further enhanced to simulate water flow across the discretized landscape units, is used to simulate the ecohydrology of the Upper Ouémé watershed. Indicators consistent with the ecosystem accounting framework are used to map and quantify the capacities and the flows of multiple hydrological ecosystem services based on the model outputs. Service flows are conceptualised as the contributions in time and space of ecosystems to productive and consumptive human activities leading to human benefits whereas service capacities reflect ecosystem condition and extent at a point in time and the resulting potential to provide service flows. Biophysical ecosystem accounts are subsequently set up based on the spatial estimates of hydrological ecosystem services. In addition, I conduct trend analysis statistical tests on biophysical ecosystem accounts to identify trends in changes in capacity of the watershed ecosystems to provide service flows. I show that the integration of hydrological ecosystem services in an ecosystem accounting framework provides relevant information on ecosystems and hydrological ecosystem services at appropriate scales suitable for decision-making.

In chapter 3, I develop a general modelling approach for analysing the effects of deforestation on the availability of water for irrigation at the watershed level, and I apply the approach to the Upper Ouémé watershed in Benin. I use controlled modelling experiments based on the SWAT model from chapter 2 in addition to copula functions to quantify surface water availability and irrigation potential under prevailing forest and woodland cover as well as varying forest and woodland extents. I undertake these comparative analyses for two irrigation development scenarios that are defined based on different levels of sustained water flows in the Upper Ouémé river network. The analyses show that conservation of prevailing forests and woodlands in the Upper Ouémé watershed is needed to allow the development of 80% (15,000 ha) or 71% (20,000 ha) of the irrigation potential in the dry season depending on the scenario. At the prevailing forest and woodland extent, the loss of around 40 ha of forest and woodland area reduces the irrigation potential by 1 ha depending on the scenario. Our irrigation potential calculations are based on the water requirements of rice which is the most water intensive crop grown in the study area. For other crops, the ratio will be lower (i.e. less forest and woodland area is required to sustain 1 ha of irrigated crop production). The relation between forest and woodland extent and irrigation potential, is, however, not linear, and more hectares of forest and woodland are needed to support 1 ha of irrigated crop production with increasing deforestation. This is relevant for trade-off analysis, where it needs to be noted that the forests and woodlands not only generate water regulation services but also provide other ecosystem services including fuelwood, timber, opportunities for livestock grazing and carbon sequestration .

In chapter 4, I analyse the impact of climate change on the potential for increasing cropping intensity in rainfed cropland areas (sequential cropping) i.e. growing two or more crops per year. I also analyse the impact of climate change on water availability for irrigation expansion in central Benin. My approach combines hydrological modelling and scenario analysis involving two Representative Concentration Pathways (RCPs), two water-use scenarios for the watershed based on the Shared Socioeconomic Pathways (SSPs), and environmental water requirements leading to sustained water flows in the river network. The analyses show that in Benin, one of the effects of climate change will be that increasing crop production through expansion of sequential cropping in the future will be severely limited. Depending on the climate change scenario, between 50% and 95% of cultivated areas that are used for sequential cropping or can support it will revert to single cropping. The results also show that even under a combination of RCP2.6 and SSP1 scenarios, the irrigation potential of the watershed will at least be halved by mid-century. Given the urgent need to increase crop production to meet the demands of a growing population in SSA, our study outlines challenges and trade-offs -and the need for planned development- that need to be overcome to improve food security in the coming decades.

Finally in chapter 5, I quantify trade-offs between per capita food availability and protecting forests and woodlands at different levels of yield increases taking into account climate change, population growth and land quality of potential arable areas. I carry out these analyses for three major food crops, i.e. maize, cassava and yam, in Benin. The analytical approach combines soil-water balance and crop growth modelling under two RCP and SSP scenarios and three scenarios of cropland expansion with varying degrees of deforestation. The analyses show that depending on the scenarios 1) maize and yam yields will have to increase at a faster rate than has been recorded over the past two and half decades in order to maintain current levels of food availability in case of no expansion of croplands; 2) major investments in higher-yielding cultivars (especially for cassava) and irrigation will be required to raise the yield ceiling in order to overcome the biophysical limitations on yields imposed by climate change; 3) substantial areas of forests and woodlands may have to be converted to croplands to make up for lost opportunities to increase production. However, forest and woodland loss will affect water availability for irrigation. Our study shows that food security outcomes and forest and woodland conservation goals in Benin and the larger SSA region are inextricably linked together and require holistic management strategies that considers trade-offs and co-benefits.

My thesis provides novel and detailed insights on the dynamic interaction between food production and forest and woodland conservation goals and demonstrates how hydrological ecosystem services can be captured in an accounting framework. Specifically, this thesis has demonstrated how to quantify hydrological ecosystem services in line with the SEEA EEA. The generation of hydrological ecosystem services involves the dynamic interactions between individual environmental assets (e.g. land cover, water, and soil resources). Whereas accounting for these individual environmental assets is adequately captured in the Systems for Environmental-Economic Accounting - Central Framework, the benefits accruing from the interactions are not fully addressed, leaving a critical gap in the monitoring of these services that underpin food and water security. The SEEA EEA by using a systemic view of relationships between the individual environmental assets provides a suitable framework for quantifying hydrological ecosystem services and linking them to human and economic activities. In this thesis, I show that the empirical distinction and separate spatial characterization of capacity and flow of hydrological ecosystem services is feasible though not without challenges including computational efficiency, data requirements, and temporal resolution.

This thesis also offers a suitable approach to quantify the water flow regulation service of forests and woodlands and connect it to crop production. Water flow regulation leads to streamflow augmentation in the dry season and hence can provide critical resource input for crop production at a time when it is needed the most. However, this service is usually poorly understood and overlooked in land management decisions because of complexity of quantification. Failure to consider the contributions of water flow regulation in decision-making can lead to negative externalities and misguided policy.

This thesis fills critical knowledge gaps regarding the impact of climate change on opportunities for agricultural intensification by analysing the impact of climate change on the 1) potential to grow two or more crops per year, and 2) water availability for irrigation expansion. Such detailed analyses are required to support agricultural development planning and for targeted management interventions. Currently, much of the research focus on the impact of climate change in SSA has been on crop yield responses. This may lead to an underestimation of the impact of climate change and consequently misguided policy.

Based on the results of this thesis, four main recommendations to help address the dual challenge of food security and ecosystem protection in Benin and the larger SSA region are made. (1) To promote a precautionary approach to forest and woodland conservation in Benin and the larger SSA region. A precautionary approach to forest and woodland conservation involves the identification and protection of critical habitats even though there is currently uncertainty in the ecosystem services they deliver. (2) To promote cross-sectoral policy dialogues and consultations for planning and implementation of holistic management strategies involving forest management and agricultural production. These can be achieved through the creation of multi-stakeholder platforms involving intergovernmental agencies and ministries. (3) To develop satellite ecosystem accounts at national levels that are consistent with national accounts. Developing satellite ecosystem accounts consistent with national accounts help to elucidate the contributions of ecosystems to economic activities recorded in the national accounts, as well as for capturing exchange values of some ecosystem services that are not included in these accounts. (4) To identify and evaluate adaptation and resilience measures to reduce agricultural vulnerability to climate change.

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ABOUT THE AUTHOR

Confidence Duku was born on 26 February 1984 in the historic coastal town, Elmina, in Ghana. Confidence had his early education in Elmina and Cape Coast, before proceeding to Kwame Nkrumah University of Science and Technology, for his undergraduate studies. There, he graduated with a First Class Honours in BSc Biological Sciences. He went on to first work in the same University as a Teaching Assistant for a year, supervising practical sessions in Basic Microbiology for undergraduates, before enrolling in the MSc Environmental Sciences programme. It was during this programme that Confidence developed an interest geospatial and environmental modelling. The MSc programme after opening his eyes to this exciting field, however, could not sate his appetite. He wanted more. Hence, he decided to change courses opting for a more specialised and technical programme in geospatial and environmental modelling. In this respect, he won a Commonwealth Scholarship by the Department for International Development, UK to study Geographic Information Systems (GIS) and Environmental Modelling for his master's degree at the University of Hull, UK. This programme was a real eye-opener for him. He was thrilled by the power of GIS to help address different kinds of problems including environmental, food security and health. After completing his studies, he joined Africa Rice Center, an international agricultural research institute in Cotonou, Benin as a GIS Specialist. There, he built on the work of Crop health specialists and developed a spatial model for identifying hotspots of rice yield losses due to biotic stresses in East Africa. After almost two years at Africa Rice Center, Confidence left to start his PhD with the Environmental Systems Analysis group of Wageningen University. His PhD research focussed on exploring the dynamic interactions between food production and ecosystem services with a case study in Benin. His PhD research was part of the interdisciplinary research project "Realizing the Agricultural Potential of Inland Valleys while Maintaining their Environmental Services" led by the Africa Rice Center in collaboration with Wageningen University. During his PhD research, he also supported educational activities and gave a guest lecture to MSc students on spatial modelling in support of environmental systems analysis. Confidence is now looking forward to the next challenge in his life and career.

LIST OF PUBLICATIONS

Duku, C., Rathjens, H., Zwart, S.J., Hein, L., 2015. Towards ecosystem accounting: a comprehensive approach to modelling multiple hydrological ecosystem services. Hydrology and Earth System Sciences 19, 4377-4396.

Duku, C., Sparks, A.H., Zwart, S.J., 2016. Spatial modelling of rice yield losses in Tanzania due to bacterial leaf blight and leaf blast in a changing climate. Climatic Change 135, 569-583.

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Duku, C., Zwart, S.J., van Bussel, L.V.G., Hein, L., 2017. Quantifying trade-offs between future food availability and forest and woodland conservation in Benin. Science of the Total Environment (Accepted).

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The SENSE Research School declares that **Mr Confidence Duku** has successfully fulfilled all requirements of the Educational PhD Programme of SENSE with a work load of 45 EC, including the following activities:

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- Cost benefit analysis and environmental valuation, Wageningen University (2013)
- Techniques for writing and presenting a scientific paper, Wageningen University (2014)
- Writing grant proposals, Wageningen University (2016)
- International Spring University on Ecosystem Services Modeling - Artificial Intelligence for Ecosystem Services Modeling, Basque Centre for Climate Change (2016)

Management and Didactic Skills Training

- Supervising MSc students in writing scientific essays as part of the course 'Introduction to global change' (2014)
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Oral Presentation

- *Modelling hydrological services for ecosystem accounting.* International Society of Ecological Economics (ISEE) 2016 Conference, 26-29 June 2016, Washington DC, USA

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