CHAPTER 5

REVIEW OF APPROACHES TO ASSESS THE IMPACTS OF MANAGEMENT AND POLICY SCENARIOS ON ECOSYSTEM FUNCTIONING AND HUMAN WELL-BEING

5.1 Introduction

As highlighted in Chapter 1, one of the limitations to the sustainable management of wetlands in Africa is: the poor understanding of the consequences of alternative policy and management regimes on wetland functioning; and the supply of ecosystem services and human well-being. This chapter reviews different analytical approaches used in the literature for establishing the linkages between ecological and economic systems and evaluating the impacts of alternative management and policy regimes on ecosystem functioning and economic well-being. The review will be used as the basis for choosing an analytical framework to adapt to this study.

5.2 Review of analytical approaches

Three main analytical approaches are used for evaluating the impacts of alternative management and policy regimes on ecosystem functioning and economic well-being in the literature. These are: economic valuation; multi-criteria analysis; and integrated ecological-economic models. These approaches are discussed in detail below.

5.2.1 Economic valuation

Ecosystems provide services that are of value to human welfare. The value of these services depends on the type of functions that are perceived as valuable to society. Only functions that provide services that satisfy a society's demands directly or indirectly have an economic value (Costanza *et al.*, 1989; Turner *et al.*, 2000).

The total economic value framework disaggregates the total economic value into use and non-use values (Figure 5.1). A use value refers to the value of ecosystem services that are used for human and production services. It includes the tangible ecosystems services that can be consumed directly (direct use values) as well as ecosystem services that are intermediate inputs for production of final goods and services for human consumption (indirect use values), such as soil nutrients, water and biological support. A non-use value (also referred to as 'existence value' or 'option value') is the value that humans ascribe to ecosystems for preserving the option to use in future, despite the fact that they may not presently be deriving utility from them.

Economic valuation is an attempt to quantify the direct and indirect benefits from ecosystem services in monetary terms. It is aimed at providing a common metric in which to express the benefits of the diverse services provided by ecosystems (Barbier *et al*., 1997). Valuation can be used in three main ways, according to Pagiola *et al*. (2004). The first is total valuation, which aims at estimating the total value of ecosystem services at a given time (e.g. for national income accounting or to determine its worth as a protected area). This type of valuation can provide useful information on the contribution of ecosystems to human welfare. Most of the wetland valuation studies conducted in southern Africa fall in this category (Seyam *et al*., 2001; Schuyt, 1999).

It is believed that an improved awareness of the contribution of ecosystems to human welfare ensures that the values of ecosystems are better taken into account in decision making and can also be applied at the macroeconomic level for making adjustments to national income accounts. One limitation of this approach is that in most instances it is practically difficult to determine non-market ecosystem services. As a result, most of the valuation studies quantify few selected services.

Secondly, economic valuation can be used as a tool to examine the distribution of costs and benefits of ecosystem services among stakeholders. In this way, economic valuation allows for understanding of how different management interventions affect the poor and other stakeholders (i.e. equity considerations).

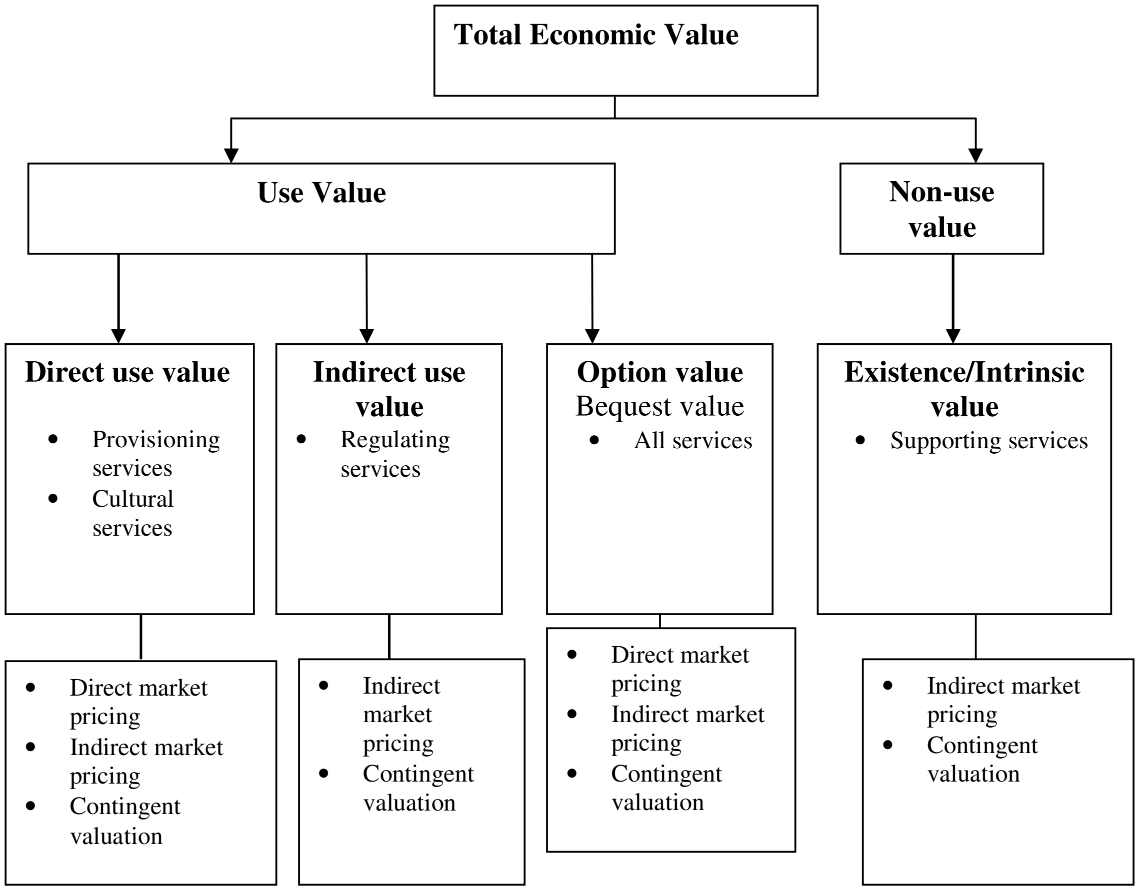


Figure 5.1: The Total Economic Value framework (Adapted from: MEA, 2003)

Thirdly, valuation can be used to evaluate the trade-offs between alternative ecosystem management regimes that alter ecosystems condition and the multiple services they provide. This approach focuses on assessing the impacts of alternative management and policy regimes on ecosystem services. This valuation approach is referred to as partial valuation (Barbier *et al*., 1997). In this approach, the first step is to quantify the biophysical relationships of the impact of management alternatives on ecosystem functioning and how this affects the provision of ecosystem services. The second step is to apply valuation in the narrow sense, which monetarises ecosystem services using prices. This type of valuation is more relevant to policy since it quantifies the trade-offs among alternative uses of an ecosystem.

Economic valuation approaches can also be categorised into those that are static in nature and those that are dynamic. The former quantifies the value of ecosystem services at a single time period. It does not trace the effects of changes in ecosystem condition and ecosystem services over time and thus assumes that ecological processes and ecosystem services are constant over time. Most of the wetland valuation work in Africa falls in this category mainly due to data limitations (e.g. Barbier *et al*., 1991; Schuyt, 1999; Emerton *et al*., 1999). In contrast, the dynamic approach takes into account the fact that changes in ecological functioning play out over time and result in changes in the supply of ecosystem services in the short, medium and long-term. Examples of the application of the dynamic approach to wetland ecosystems are studies by: Chopra and Adhikari (2004); Eppink *et al*. (2004); and Güneralp and Barlas (2003).

The Cost-Benefit Analysis (CBA) is the most widely used framework for valuing ecosystem services. The framework quantifies the costs and benefits of environmental services and enables quantification of trade-offs among ecosystem services. Under the CBA framework, there are several techniques that can be used to value ecosystem services. These can be classified into three broad categories: those that use directly observed market prices for valuation; those that use surrogate market prices for valuation; and those that use survey techniques for valuation[11].

In the first category, valuation is based on direct (observed) market prices of goods and services (revealed preference methods). It includes techniques such as: change in value of direct output; the production function approach; the replacement cost approach; the damage cost avoided approach; and the defensive expenditure method. The second category of methods is based on surrogate markets, that is to say the market value of complementary and substitute goods in cases where the ecosystem service to be valued does not have an observed market price. Examples of valuation techniques which fall in this category include travel cost methods and hedonic pricing. Finally, survey techniques (stated preference methods) can be used to directly ask consumers to state their preferences regarding a non-marketed ecosystem service by presenting to them hypothetical scenarios. Valuation techniques under this category include: contingent valuation methods; conjoint analyses; and choice experiments. The different valuation techniques discussed here have been applied for

[11] See Freeman (1993) for a detailed discussion of the different economic valuation techniques and Barbier *et al.* (1997) for a discussion on the application of valuation techniques to wetland ecosystems.

valuing wetland services in Africa (see Barbier *et al*., 1991; Schuyt, 1999; Turpie *et al*. 1999; Emerton *et al*. 1999).

Although the CBA approach has been applied extensively in valuing ecosystem services, the framework has a number of shortcomings. Apart from its significant data requirements, which affect the accuracy and reliability of results, the framework is primarily based on economic efficiency without considering the distribution of costs and benefits among stakeholders (Acreman, 2001; Gregory and Slovic, 1997). For this reason, other scholars recommend that the CBA needs to be complemented with measures other than economic efficiency to be able to guide decision making (Barbier *et al*. 1997).

5.2.2 Multi-criteria analysis

Considering the limitations of the CBA, some scholars have opted to use multi-criteria analysis (MCA) to evaluate the alternative ecosystem management options based on multiple criteria such as: economic efficiency; environmental security; and equity (Barbier *et al*. 1997; Acreman, 2001; Brouwer and Van Ek, 2004). The MCA approach allows for comparing and ranking different management outcomes using multiple economic, environmental and social indicators. The actual measurement of indicators need not be in monetary terms, but are often based on scoring, ranking and weighting of a wide range of qualitative criteria.

The MCA approach, however, has its own shortcomings. The main shortcoming is related to the subjectivity of the choice of weights that are assigned to each objective. A common technique used to deal with this problem is to undertake a sensitivity analysis of outcomes with varying weights. For this reason, some scholars recommend introducing stakeholders' perceptions, derived from a stakeholder analysis to help in the weighting of different criteria.

The MCA and CBA should not be considered as parallel approaches. In some cases the two approaches complement each other (Brouwer and Van Ek, 2004; Tiwari *et al*. 1999). The MCA can also take the form of integrated disciplinary models, which take into account environmental security, economic value and distributional aspects.

5.2.3 Integrated ecological-economic models

Integrated ecological-economic models are used for evaluating ecological and economic impacts of alternative ecosystem management and policy regimes (Costanza and Ruth, 1998; Cox, 2005; Farber *et al*., 2006). These models integrate various aspects of ecosystem functioning (e.g. hydrology), ecosystem services and their economic value. The models can be analytical or numerical and describe either steady-state or dynamic change. The models are most easily carried out at a local scale, where the interactions between elements in the system can be easily identified.

Turner *et al*. (2000) and Chopra and Adhikari (2004) highlighted that the impacts of management interventions on wetland functioning and human well-being can be better understood through the integrated modelling of ecological and economic processes of wetland systems and scenario analysis. In such models, economic valuation plays an intermediate role of expressing ecosystem services associated with the different management scenarios in monetary terms so that scenarios are comparable.

Two forms of integrated models are used in the literature for evaluating the impacts of alternative management and policy regimes on ecosystem functioning, the supply of ecosystem services and human well-being: modular or heuristic models; and system dynamics models (Turner *et al*. 2000; Ringler and Cai, 2003; Costanza and Ruth, 1998). These forms of models and examples of their applications are discussed in detail below.

5.2.3.1 Heuristic models

In these models, ecological and economic systems are constructed separately with output from one disciplinary model used as an input in another. In other words, the submodels operate independently with loose connections and no feedbacks between models.

A good example of the empirical application of this approach is provided by Van den Bergh *et al*. (2001) who developed spatially integrated economic, hydrological and ecological models to analyse the impacts of alternative land use scenarios (housing, infrastructure, recreation, agriculture and nature conservation) on a wetland system in the Netherlands. Hydrological models were developed to simulate the impacts of these land use scenarios on the ground and surface water quantity and quality in the wetland. The outputs of the hydrological models were fed into an ecological model, which was used to estimate the effect of changes in water quality and quantity on the vegetation species' diversity. The net present value and environmental quality were the two aggregate performance indicators computed for each land use scenario and were later combined to form one welfare index on the basis of which land use options were compared.

The major advantage of heuristic models is that they allow for a detailed analysis of each of the components included in the model. However, by modelling ecological and economic systems separately the approach does not take into account the interactions and feedbacks between elements in the system.

5.2.3.2 System dynamics models

System dynamics models are based on systems theory, which was developed during the mid-1950s as an approach to understand the dynamic behaviour of complex systems (Forrester, 1968). This approach recognises that elements of complex systems are tightly interwoven into one system with direct interactions and feedbacks between them. It is on this premise that the system dynamics approach has also been referred to as the holistic approach (Brouwer and Hofkes, 2008).

What makes system dynamics models different from other modelling approaches used in studying complex systems is the use of stocks and flows. To take into account the links between the natural system and socio-economic system, the two systems are usually integrated as modules of models (Costanza *et al*. 1993). Difference equations are used to describe the dynamics of stocks in the system together with equations specifying relationships between flows (e.g. human consumption of ecosystem services) and other elements in the system. The totality of the model equations constitutes the structure of the model (or the system). It is essential in the system dynamics methodology that the model structure provides a reasonable representation of the main interactions in the system being modelled.

Although the system dynamics framework was originally developed for understanding the dynamics of industrial processes, it has been widely applied in understanding the dynamic behaviour of ecosystems, particularly in evaluating the impacts of alternative management regimes on ecosystem functioning, ecosystem services' supply and human well-being.

For example, Van Beukering *et al*. (2003) developed and applied a system dynamics model to examine the economic consequences of alternative management options of a national park in Indonesia. They developed ecological and economic modules to predict the impacts of alternative management regimes on ecosystem functioning and ecosystem services provided by the national park. Three management regimes for the national park were considered: deforestation; conservation; and selective use. Selected ecosystem services were considered in the model, which are: water supply; fisheries; flood prevention; agriculture and plantation; hydroelectricity; timber and non-timber products; tourism; biodiversity; fire prevention; and carbon sequestration. The economic valuation module was used as an intermediate step in the modelling process to estimate the economic (monetary) value associated with each management option. The study found that conservation of the national park spreads the benefits of the national park equally among all stakeholders and therefore prevents potential social conflicts while deforestation widened the income gap between the rich and the poor.

In a study in the Brazilian Amazon forests, Portela and Rademacher (2001) used a dynamic simulation model to investigate the value of forest ecosystem services under farming and ranching uses. They developed a model with three modules: i) deforestation drivers module, which considered the socio-economic drivers of forest clearing; ii) the ecosystem services for quantifying the impacts of land use patterns on forest ecosystem services; and iii) ecosystem valuation module for calculating the economic value of changes in forest ecosystem services. The key forest ecosystem services considered in the model are: hydrological regulation; nutrient cycling; carbon sequestration; and species diversity. The losses in the value of ecosystem services due to different land use practices (farming and rangeland management) were compared to the forest reference value, which was based on a global average value of forest ecosystems to find the net welfare impacts of land use practices. Portela and Rademacher (2001) showed that there are significant losses in the value of ecosystem services under farming and rangeland management regimes compared to the forest reference value.

Gambiza *et al*. (2000) examined the ecological and economic impacts of changing stock rates, tree removals, fire regimes and woodland structures for the Miombo woodland ecosystems of Zimbabwe. A dynamic simulation model with the following five interactive modules was developed: rainfall; grass production; fuel load; fire occurrence; and tree dynamics. The economic impacts of alternative woodland management regimes were explored by comparing the net present values accruing to the state authority that manages the forest and communal dwellers dependent on the forest under different management regimes (grazing pressure, high or reduced impact logging, varying proportion of harvestable timber cut). Their study concluded that the net present value to the state authority managing the forest remained constant under the different management regimes despite the marked ecological response.

Higgins *et al*. (1997) developed a dynamic simulation model to examine the value of ecosystem services provided by mountain *fynbos* ecosystems under alternative management regimes in South Africa. Three management regimes were considered: pristine management (uninvaded, no clearing required); present management (invaded, no alien clearing); and proactive management (invaded, intense clearing). Like the other studies discussed above, they divided their model into modules and used economic valuation as an intermediate step in the modelling process. Their model has five modules: hydrological; fire; plant; management; and economic valuation modules. The first three modules were used to quantify the impacts of management regimes on the *fynbos* ecosystem and the supply of selected ecosystem services while the economic valuation module estimated the value of the services under each management regime. By considering key ecosystem services provided by forests they were able to demonstrate that the costs of clearing invasive alien plants were a small proportion of the value of *fynbos* ecosystem services thus justifying an investment in clearing alien plants in *fynbos* ecosystems.

Application of system dynamics models to model dynamic behaviour of wetland ecosystems has recently gained prominence. For example, Chopra and Adhikari (2004) developed and applied an ecological-economic model to simulate effects of alternative regimes on ecological health and incomes derived from a wetland system in Northern India. Their model has three environmental modules which examine changes in three environmental variables that affect the ecological health of the wetland water, biomass and birds modules and a net income module, which sums up the impact of changes in each of the environmental modules on income derived from tourism and resource extraction. Upstream agricultural activities were assumed to cause pressures that affect stock of water and biodiversity (biomass and birds), which in turn determine the ecological health and hence amenity value of the wetland. The number of tourist visits to the wetland was considered to be a function of ecological health fir the wetland. The sensitivity of tourist visits to wetland ecological health indices were derived through simulation of scenarios with respect to future pressures on the wetland. The travel cost method was applied to estimate demand functions and consumer surplus accruing as welfare gain to tourists from amenity values derived from the wetland. They concluded that direct and indirect income obtained from the wetland is positively related to the ecological health of the wetland demonstrating a positive incentive to conserve the wetland.

Eppink *et al*. (2004) presented a general dynamic simulation model for analysing interactions between land use and wetland biodiversity. The model comprises of four modules: a land accounting module, which tracks changes in agricultural and urban land use; a biodiversity module describing the impacts of land use on biodiversity (measured in terms of species richness and evenness); a land use decision module describing the process that leads to decisions on urban expansion; and a social evaluation module in which social welfare is modelled as a function of income per capita, population density and wetland biodiversity was used to assess scenario outcomes. Using different scenarios for population, agricultural and urban growth, simulation experiments were performed to assess the effects of these scenarios on wetland biodiversity and social well-being. The study showed that there may be conflicts between urban growth and the conservation of wetland biodiversity.

Güneralp and Barlas (2003), working on a lake ecosystem in Turkey, developed and applied a system dynamics model to assess the impacts of different scenarios on ecosystem and economic activities. The objective of the model was to find a balance between improving the well-being of inhabitants living around the lake and maintaining ecological integrity of the lake ecosystem. They simulated dynamics of: ecological elements of the lake ecosystem; economic activities such as crop production, industrial activities and fishing; and the demographics of inhabitants in the study area. Their study concluded that there is no threat of a shift in algal dominance in the lake although there is potential for a decline in the welfare of inhabitants due to an increase in population.

In southern Africa, there is limited empirical work on evaluating the impacts of alternative management and policy regimes on wetland functioning, ecosystem services supply and human well-being. Apparently, one study by Turpie *et al*. (1999) attempted to assess the economic and ecological impacts of various management options of wetland systems in the Zambezi basin using a dynamic simulation model. Although the study does not give a detailed description of the model the information available shows that four management scenarios were simulated, which are: the maintenance of the status quo; implementing wise use practices; delimiting protected areas; and commercial agricultural development. The model integrated ecological submodels describing the impacts of management scenarios on wetland functioning and selected ecosystem services (fish, wild animals, palms, reeds and papyrus production, flood plain grazing and crop production) and an economic valuation module for estimating values of ecosystem services under each management scenario. Their results showed that the status quo management practices will result in reduced wetland benefits in future, while wise use practices maximise future wetland benefits to the community.

5.3 Approaches and methods of the study

This study adopts the system dynamics framework to establish the linkages between ecological and economic systems in the Ga-Mampa wetland area. This framework is chosen, because of its ability to take into consideration the feedback effects between ecological and economic systems and also its ability to capture the intertemporal effects of interventions on ecosystem dynamics (Costanza *et al*. 1993; Costanza, 1996).

In developing the system dynamics model one can draw upon earlier studies on the systems modelling interactions between ecological and economic systems in wetland systems presented by Eppink *et al*. (2004); Güneralp and Barlas (2003) and Chopra and Adhikari (2004).

The adapted analytical framework is presented in Figure 5.2. The framework involves three steps: (i) evaluating the impacts of management scenarios on wetland ecosystem functioning; (ii) quantifying the effects of changes in ecosystem functioning on the supply of ecosystem services; and (iii) measuring the effects of the change in ecosystem services on human well-being. The bulk of the work involves quantifying the biophysical relationships along a causality chain. This involves integrating models from different disciplines.

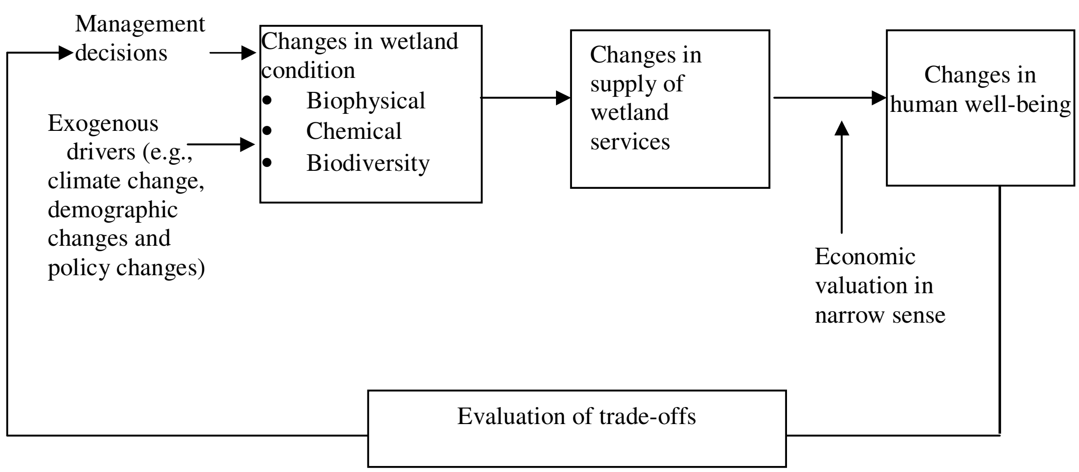


Figure 5.2: Analytical framework for evaluating the impacts of alternative wetland ecosystem management and policy regimes on ecosystem functioning, ecosystem services and human well-being (Adapted from: MEA, 2003)

5.4 Concluding Summary

This chapter reviewed the main analytical approaches used for evaluating the impacts of alternative management and policy regimes on ecosystem functioning, the supply of ecosystem services and human well-being. The review showed that three main analytical approaches are used for this purpose, which are: economic valuation; multi-criteria analysis; and integrated ecological-economic models (heuristic and systems dynamics models). Due to its ability to capture economic and ecological systems as integral components of one system and the feedbacks between them, the system dynamics approach in developing an ecological-economic model was chosen. The model is developed and applied to simulate the impacts of alternative management and policy scenarios in the next chapter.

CHAPTER 6

EMPIRICAL MODEL AND RESULTS FROM ANALYISIS OF THE IMPACTS OF ALTERNATIVE MANAGEMENT REGIMES ON WETLAND FUNCTIONING AND ECONOMIC WELL-BEING

6.1 Introduction

This chapter develops an empirical ecological-economic model for evaluating the impacts of alternative policy and management regimes on the wetland system and economic well-being. The first section of the chapter presents a generalised conceptual framework highlighting the main components in the system and their interactions. Section two discusses in detail the components of the empirical model and the assumptions behind their specification. The section that follows presents the entire system of the empirical model showing the linkages between ecological and economic systems and parameters used in the model. The fourth section validates the model. The model is then used to perform simulations of alternative wetland management and policy regimes the results of which are presented and discussed in the fifth section. A concluding summary of the chapter is presented at the end of the chapter.

6.2 Conceptual framework

This study attempts to develop an ecological-economic model based on the system dynamics framework. As highlighted in the previous chapter, the said framework takes into consideration feedback effects between ecological and economic systems as well as involved tradeoffs in the supply of individual constituents of the bundle of multiple services provided by wetlands. This framework also captures the intertemporal effects of interventions on ecosystem dynamics. In order to understand the ecological-economic interactions in the wetland system under study it is important to first identify the main components of the system and their interactions. The adapted framework consists of five subsystems: socio-economic; wetland hydrology; natural wetland vegetation; crop production; and land use change trade-offs. These subsystems are interlinked and changes in one subsystem impact on others with some feedbacks among them (Figure 6.1). Crop production and livestock production as well as natural wetland vegetation subsystems are linked to the wetland hydrological module through changes in water use. Crop and livestock activities abstract water from the wetland thereby affecting the wetland system's hydrology and water budget. Water use on the other hand influences the productivity of crops, livestock and natural wetland vegetation, which in turn affects the economic welfare component of the socio-economic subsystem. Crops and natural wetland vegetation also influence the wetland water budget as they lose water through evapotranspiration.

Crop and livestock production and natural wetland vegetation subsystems are also interrelated through competition for land and labour resources. For example, conversion of the wetland for crop cultivation reduces the wetland area and consequently the availability of its natural products, including vegetation for livestock grazing. There are therefore trade-offs involved between these activities, which also require the use of labour and other inputs supplied by the communities and hence competition for these inputs.

A positive relationship between growth in biomass of natural wetland vegetation and wetland groundwater level links the natural wetland vegetation to the underlying hydrological system and captures the trade-offs between crop and wetland biomass production due to competition for water. For instance, as groundwater levels are lowered through wetland conversion to agriculture, natural wetland vegetation is adversely affected by competition with non-wetland plant species (Eppink *et al*. 2004). As biomass increases the actual growth rate is expected to decrease due to competition for limited resources (e.g. light, water, nutrients and space). This is also true the other way around, when biomass is removed from the wetland (e.g. through biomass harvesting) the actual growth rate will increase.

The economic welfare component of the system is influenced by benefits derived from exploiting the wetland ecosystem (i.e. crop, livestock and natural products as well as domestic water supply) and income derived from other sources (i.e. off-farm employment and social transfers). This socio-economic subsystem on the other hand supplies labour and other inputs for which various crop, livestock, natural product harvesting and off-farm activities compete.

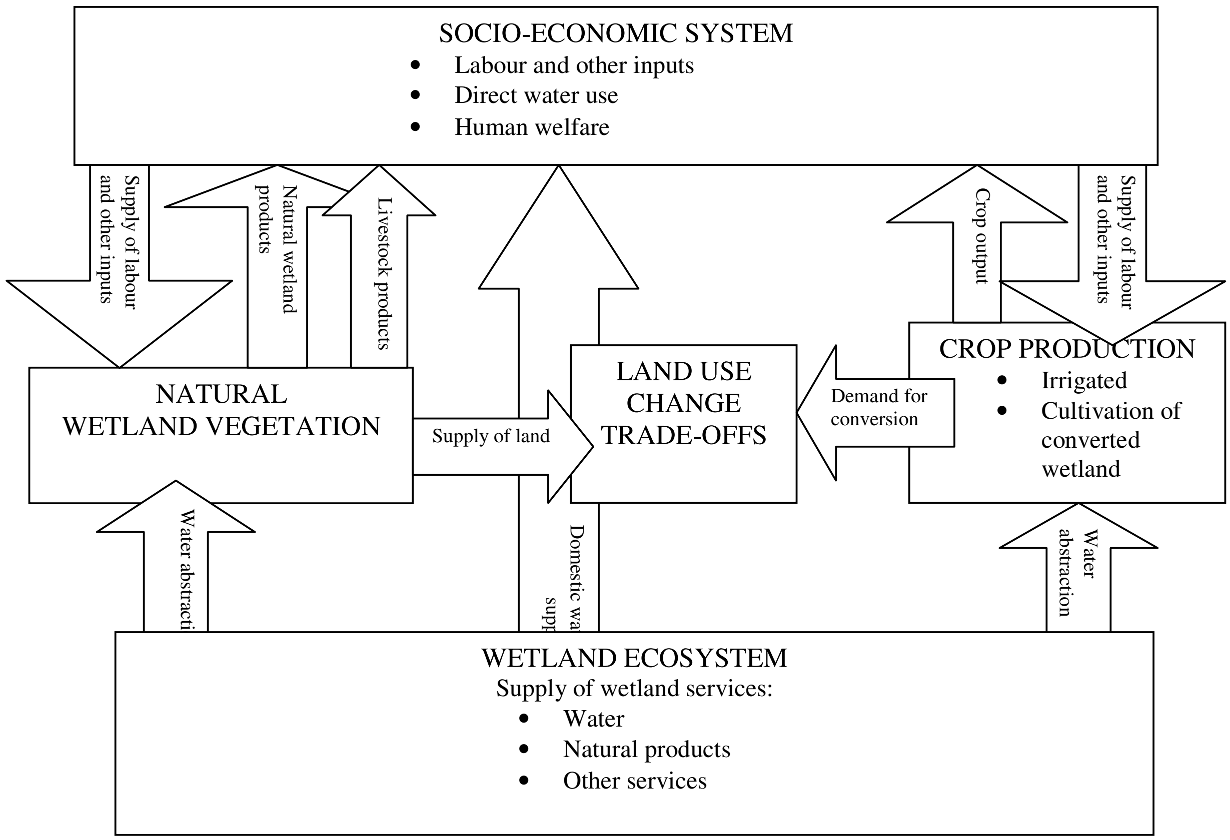


Figure 6.1: Conceptual framework showing the interactions between components of the system (Adapted from: Güneralp and Barlas, 2003)

6.3 The empirical model components and assumptions

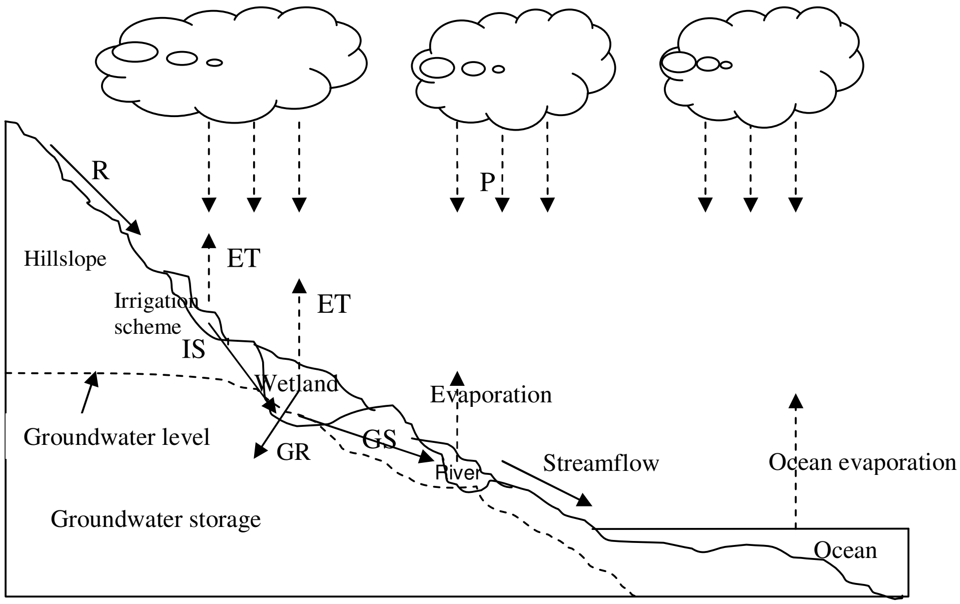
Although the wetland system under study provides several direct services, crop production and natural products harvesting[12] are the most important services supporting the well-being of the population in the study area (Adekola, 2007). Therefore, this study's empirical model focuses on these two services. The model integrates five modules which are discussed in detail below.

[12] Livestock production and domestic water supply have been excluded from the empirical model due to lack of data for estimating livestock products and domestic water supply and input demand system.

6.3.1 Hydrology module

Wetland hydrology is the primary driver of wetland ecosystem dynamics and many important functions of wetlands are directly linked to wetland hydrological processes (Eppink *et al*., 2004; Zhang and Mitsch, 2005; Mitsch and Gosselink, 2000). The objective of this module is to assess the impacts of wetland uses (crop production and natural wetland vegetation products) on the wetland water budget. The module is modelled in just enough detail to reflect the fundamental system dynamics and have input-output exchanges with the other modules. Standard stock-flow equations are used to relate the different wetland water budget components including inflows, and outflows from the wetland, which are mainly groundwater recharge and discharge processes and their link to soil water.

This study's wetland hydrological system comprises of five linked sub-systems: the upper catchment; the hillslopes; the irrigation scheme; the wetland aquifer; and the river system. The wetland is fed primarily by recharge from precipitation and irrigation schemes and losses through the evapotranspiration of crops and natural vegetation and seepage from the wetland to the river (Masiyandima *et al*., 2006).



Where: P is precipitation; IS refers to irrigation scheme seepage; GS is groundwater seepage from the wetland to the river; GR is recharge from wetland soil to groundwater; R is surface runoff; and ET refers to evapotranspiration losses.

Figure 6.2: Schematic representation of the main hydrological fluxes of the wetland (Adapted from: Bullock and Acreman, 2003)

The main surface flow through the wetland is the river, which passes through the edge of the wetland. The river inflow is influenced mainly by runoff generated in the catchment upstream. Although the bare soils in the wetland can generate significant runoff this is assumed to be minimal, as water infiltrates into the wetland due to the high permeability of peat soils in the wetland (McCartney, 2005).

Although it is believed that the lateral flow of groundwater from the hillslopes contributes to wetland recharge through seepage, this has not been validated through an empirical analysis. Therefore, this component is not included in the model. Discharge of wetland groundwater occurs through outflow of groundwater from the wetland to the river through seepage. The discharge of groundwater from the wetland is also influenced by artificial drainage activities as farmers drain groundwater from wetland plots to lower the water table and increase crop yields. In the studied wetland, 5% of the wetland represents the open drain area and much of the drained water is lost through evaporation before reaching the river system. However, to simplify the model and also due to lack of data, artificial drainage was not considered in the model.

In addition to groundwater seepage from the wetland to the river, wetland groundwater level is also influenced by groundwater recharge from saturated wetland soils (GR) and recharge from irrigation (IS). Recharge from wetland soils is influenced by soil moisture dynamics, which are in turn influenced by rainfall and evapotranspiration. Upstream of the wetland is a water diversion for the irrigation scheme on the perimeter of the wetland. The diversion from the river is channelled to the irrigated fields via a primary canal and several secondary canals, all of which leak severely. A principal canal transports water to the primary and secondary canals, which then feeds into the fields.

It is assumed that some water seepage from the irrigation area into the wetland groundwater storage occurs, recharging the wetland. The volume of diverted irrigated water for irrigation depends on the geometry of the canal as well as the water level in the weir. The canal's capacity is 130 litres per second (l/s). An estimated 94% of the diverted water is lost through seepage in the network of canals from the primary to field canals leaving only 6% available for crops (Chiron, 2005). It was assumed that the seepage losses from irrigated area recharge wetland groundwater.

Crop and natural vegetation evapotranspiration is the major component of water loss from the studied wetland system[13] (McCartney, 2005). Evapotranspiration consists of actual evapotranspiration from natural vegetation (ETv_t^i) and actual crop evapotranspiration from cultivated area (ETc_t^i). Therefore, the total evapotranspiration (ET_t^i) is given by the following equation:

$$ET_t^i = ETc_t^i + ETv_t^i \qquad (6.1)$$

$$ETc_t^i = ET_{a,t}^i * AC_t^i$$

[13] Abstraction of water for domestic uses and watering of livestock is limited.

Where i refers to the two systems: irrigation; or wetland system that is to say i(r,w). $ET^{i}_{a,t}$ is actual evapotranspiration per hectare of cultivated area in system i at the time period t (mm/ha) and AC^{i}_{t} is the area cultivated in system i (ha) at the time period t.

Equation 6.1 is only true for the wetland since there is no natural vegetation in the irrigation system. Also, since our primary interest is to model the hydrological dynamics in the wetland system, equations 6.2-6.4 focus on the wetland system (i.e. i = w = wetland system). For the wetland system, the rate of evapotranspiration from natural vegetation varies with every season and is as high as 5mm per day during the rainy season and is approximately 1mm to 2mm per day during the winter season (Dye *et al*. 2008; Von der Heyden and New, 2003; Kleynhans, 2004). Using these values, it is assumed that actual evapotranspiration from natural wetland vegetation (ETv^{w}) is approximately 1100mm per unit area of wetland per year. Thus, evapotranspiration from natural vegetation in the wetland system is given by the following equation:

$$ETv^{w}_{t} = \eta * TA^{w}_{t} \qquad w = \text{wetland system} \tag{6.2}$$

Where η is a parameter showing the rate of evapotranspiration from natural vegetation per hectare per year and TA^{w}_{t} refers to the total wetland area.

For the area cultivated in the wetland system, we considered that recharge to groundwater occurs when the water content of the root zone is above field capacity. The water holding capacity for the type of soil texture found in the study area ranges from 140mm to 170mm per metre of soil depth (Saxton and Rawls, 2006). Therefore, we assume that the field capacity of the soil is 140mm.

Given the earlier description of the wetland hydrological system and the fact that run-off is limited in the wetland, the soil moisture content in the root zone can be expressed as a water balance equation as follows:

$$MC_{t+1} - MC_{t} = P_{t} + CR_{t} - GR_{t} - ET^{w}_{t} \tag{6.3}$$

w = wetland system

Where MC_t and CR_t refer to soil moisture content and capillary rise from the shallow groundwater, respectively.

The wetland hydrological fluxes discussed above impact on the wetland groundwater level through recharge and discharge processes. The equation for the change in the wetland groundwater level is given by[14]:

$$GWL_{t+1} = GWL_t + [GR_t + IS_t - GS_t - CR_t]/10^3 \qquad (6.4a)$$

Where GWL_t wetland is groundwater level (in metres) and the other variables are as defined earlier.

Since recharge to groundwater from saturated wetland soils is assumed to occur only if the soil water content is above field capacity, groundwater recharge from wetland soils is modelled using a logical if-then-else statement as follows[15]:

$$GR = \text{If } (MC_t + P_t + CR_t - ET_t^w \succ WHC) \text{ then } (MC_t + P_t + CR_t - ET_t^w - WHC) * (AC_t^w/120) \text{ else } 0 \qquad (6.4b)$$

Where WHC is a parameter for the water holding capacity of the wetland soil.

The hydrological components GS_t, IS_t, and CR_t were also modelled using: if-then-else logical statements; the information known about these processes at the study sites; and reasonable assumptions where necessary and these are presented in Appendix A2.

[14] We divide the expression by 10^3 to convert it from millimetres to metres since the wetland groundwater level (GWL) is measured in metres.
[15] To take into account the relative area of wetland and cultivated wetland, groundwater recharge is weighted by the proportion of wetland under cultivation $AC_t^w/120 = (1 - TA_t^w/120)$

6.3.2 Crop production module

This module assesses grain dynamics and their link to the other modules. Based on the grain supply function specified in equation 4.1, grain supply is a function of socio-economic variables. The parameter estimates for the grain supply function are presented in Chapter 4 (Table 4.5).

The crop production module is linked to the hydrology module through crop water use. Crops abstract water from the wetland thereby affecting the hydrology of the wetland, and in turn crop water use influences crop yields. To estimate crop water use we employ a linear crop yield-water response function based on the CROPWAT model developed by FAO (Doorenbos and Kassam, 1979) and widely applied in estimating crop water use (e.g. Igbadun *et al*., 2007; Raes *et al*., 2006; Ringler and Cai, 2003).

The model is specified as:

$$Y^{i}_{a,t} = Y^{i}_{m}\left[1 - k_{y} * \left(1 - ET^{i}_{a,t} / ET^{i}_{m,t}\right)\right] \qquad (6.5)$$

Where: i, represents a wetland or irrigation system; $Y_{a,t}$ = actual yield (tonnes/ha) at the time period t; Y_{m} = maximum yield (tonnes/ha); $ET_{a,t}$ = actual crop evapotranspiration per hectare over the cropping season (mm/ha); $ET_{m,t}$ = maximum crop evapotranspiration over the cropping season (mm); and k_{y} = crop yield response factor

To link the crop yield-water response function (equation 6.5) and the grain supply function specified in the agricultural household model of Chapter 4 with the slope parameter adjusted with the average values of the variables of household characteristics (equation 4.1) a two-step process is followed. First, the grain supply and the area cultivated are aggregated across all households in irrigated and wetland systems to get a total grain supply (TG^{i}_{t}) and total area cultivated in system i (AC^{i}_{t}) as follows:

$$TG_t^i = \left(\sum_{q=1}^{Q_i} G_{q,t}^i \Big/ 10^3 \right) \tag{6.6a}$$

$$AC_t^i = \sum_{q=1}^{Q_i} A_{q,t}^i$$

Where: $G_{q,t}^i$ is grain supply per household in system (in kgs); $A_{q,t}^i$ is the area cultivated per household in system i at the time period t (in ha); and q is the number of households in system i where total households in that system ranges from 1 to Q.

The number of households in the irrigation system (Q_r) is constant (see Table 6.4) while the number of households cultivating in the wetland system (Q_w) is computed by dividing the total wetland area under cultivation by the cultivated wetland area per household. Therefore, the equation for Q_w is given as:

$$Q_w = AC_t^w \big/ wc_0 \tag{6.6b}$$

Where AC_t^w is the total wetland area under cultivation and wc_0 is the cultivated wetland area per household.

The second step computes average yield in system i (Y_a^i) as:

$$Y_{a,t}^i = TG_t^i \big/ AC_t^i \tag{6.7}$$

The average yield is substituted for actual yield (Y_a) in equation 6.5 to solve for ET_a^i (this corresponds to crop water use per hectare in system i).

The parameters used to solve $ET_{a,t}^i$ using equation 6.5 are given in Table 6.1. Values for parameters k_y and ET_m were taken from the work of Durand (2008) on crop water use for the 19 water management areas in South Africa based on the CROPWAT model. Values of Y_m in the irrigated and wetland area were obtained from the work done in the study area by Chiron (2005).

Table 6.1 Parameters used in the CROPWAT model for maize grain

	Wetland	Irrigation
k_y (a)	1.25	1.25
Y_m (b)	3	2.5
ET_m (a)	490	490

Sources: (a) Durand (2008); (b) Chiron (2005)

It is assumed that the demand levels for local production and agricultural input are too small to influence market prices, therefore crop output and input prices are considered exogenous. The producer price series of grain, derived from national statistics (Department of Agriculture, 2009) and local observations in 2006 were used for valuing maize output[16].

Two inputs are considered in the specification of the grain supply system: water and labour. Crop water use (which corresponds to ET_a calculated from equation 6.5) is used as the proxy for quantity of water used in wetland grain production. As the actual quantity of water used for irrigated maize production is difficult to determine since the irrigation system in the study area uses gravity to convey water directly from the river into the fields through canals the ET_a for maize grain under irrigation is used as an alternative.

Since rainwater is not supplied by an economic agent at a cost, the price of water used in wetland maize grain production does not exist. We accordingly used water tariff figures for agricultural water in South Africa for 2009 to attach a cost to water. Although, there are other costs related to labour for canalisation for irrigated crops, these were not included due to data limitations. In addition, water losses due to the low efficiency of water distribution systems from river to irrigation plots was not accounted for in the model.

[16] All production is valued irrespective of whether it is self-consumed, sold or retained.

The labour costs associated with grain production are calculated based on the labour demand for grain production given in Chapter 4 (equation 4.2 of the agricultural household model with the slope parameter adjusted with the average values of the variables of household characteristics) the parameters of which are presented in Table 4.5. The net value of grain (R_t), which links this module to the human well-being module is calculated using the following equation:

$$R_t = P_{G,t} \sum_i \left(Y_{a,t}^i AC_t^i\right) - P_{W,t} \sum_i \left(ET_a^i AC_t^i\right) - W_t * L_{G,t} * \sum_i Q_i \tag{6.8}$$

Where: $P_{G,t}$ is the price of grain at the time period t (Rand/tonne); $P_{W,t}$ is the price of water at the time period t (Rand/mm); W_t is the wage rate (Rands/hour); $L_{G,t}$ is the labour time used in grain production per household in the time period t (hours/household/year) and Q_i is the total number of households in system i.

6.3.3 Land use change module

This module captures the dynamics in the area cultivated with grains in the wetland and under irrigation. Three land use systems are present in the area under study: irrigated area; natural wetland area; and area of wetland converted to crop production. Based on information from key informant interviews in the study area, the irrigated area is assumed to be constant over time and is estimated to be equal to 170 ha (Chiron, 2005). However, the area of wetland converted to cultivation grows over time whereas the natural wetland area is cleared for crop production causing the total wetland area to decline.

Therefore, the dynamics of the total wetland area are modelled using the following equation:

$$TA_{t+1}^w = TA_t^w - WCA_t \tag{6.9}$$

Where TA_t^w represents the total wetland area and WCA_t is the area of wetland converted to cultivation in time period t.

Based on focus group discussions conducted in the study area, it is assumed that changes in the total area of wetland under cultivation are a function of three sets of factors: (i) the changes in population, which increase consumption, demand for food grain; (ii) crop output prices and input prices, which provide incentives (or disincentive) to convert the wetland to crop production; and (iii) a decline in annual precipitation, which results in new farmers moving into the wetland to cultivate because of its ability to retain soil moisture throughout the year. To predict the effect of these factors (precipitation, agricultural prices for output and inputs and population) on the total wetland area under cultivation (AC_t^w) we fitted historical annual time series data for these variables on area of wetland cultivated in the past, using a multiple regression analysis[17]. As the population in the study area is only known for the year 2006, we used the district average annual population growth of 1.7% (Statistics South Africa, 2004) to extrapolate the population for additional years corresponding to the periods for which historical data on the area of wetland under cultivation is available and use that for the regression estimation.

The regression equation for area of wetland under cultivation in period t is given by:

$$AC_t^w = a_1 P_t + a_{2,} P_{G,t} + a_3 P_{Y,t} + a_4 Pop_t \qquad (6.10a)$$

Where: a_1, a_2, a_3 and a_4 are parameters; and P_t, P_G, P_Y and Pop_t represent precipitation, price of grain, price of agricultural input and population at time period t, respectively.

Thus the area of wetland converted to cultivation in period t (WCA_t) is given by the following equation:

$$AC_{t+1}^w - AC_t = WCA_t \qquad (6.10b)$$

[17] The Consumer Price Index, which we use as the proxy for the price of market goods as will be explained later, was excluded from the regression due to its high collinearity with the price of agricultural inputs. As discussed in Chapter 4, the price of maize seed is used as the proxy for the price of agricultural inputs as this is the main input cost in grain production.

The initial value of the total area of wetland under cultivation starting in year 1990 was set up in such a way as to reach the levels in 2006 that were estimated to be 66 ha (Adekola, 2007).

6.3.4 Natural wetland vegetation module

This module describes the dynamics of wetland natural biomass. Due to limited data on the study site, the formulation of this module relied mainly on literature. Reeds (*Phragmites australis* and *Phragmites mauritanus*) are the major constituents of biomass in the studied wetland system (Kotze, 2005). Following Hellden (2008), a simplified S-shaped growth curve (logistic growth function) is employed to model biomass growth dynamics. Biomass per hectare of wetland area is specified by the following equation:

$$B_{t+1} = B_t(1 + r_t) \tag{6.11}$$

Where B_t is biomass per hectare at time period t (tons/ha) and r_t is the actual growth rate of biomass stock at time period t.

Wetland biomass per hectare was set to a maximum of 70 tons per annum, which is the maximum annual productivity of reeds or carrying capacity (Finlayson and Moser, 1991 cited in Turpie *et al*. 1999). One can expect that as biomass increases the actual growth rate decreases due to competition for limited resources (e.g. light, water, nutrients and space). This is also true the other way around, when biomass is removed from the wetland (e.g. through biomass harvesting) the actual growth rate will increase. To capture the changes in actual growth rate as biomass stock changes we multiply the intrinsic growth rate by a density dependent factor (or growth rate multiplier) in computing the actual growth rate. Thenya (2006) estimated that the annual intrinsic growth rate (s_0) of wetland *phragmites* species (common reeds) can be as high as 300% after harvest during the rainy season. We assume a very moderate estimate for the intrinsic growth rate of 0.3. This rate applies when there are no limitations to biomass growth.

However, to capture the limitations caused by competition for resources as biomass stock grows, the intrinsic rate is adjusted by the growth rate multiplier. The growth rate multiplier is equal to 1 (100%) when biomass stock is close to zero and the rate decreases to close to zero when biomass stock is in full growth and is reaching carrying capacity. Thus, the growth rate multiplier is negatively related to the ratio of biomass stock in each time period to the carrying capacity (which is set at the maximum biomass per hectare). This is modelled as a graphical relationship based on the work of Hellden (2008). Following this work, the growth rate multiplier is a graphical function of the following form:

$$\sigma_t = \mathrm{GRAPH}\,(B_t/k_B)\,;\ 0<\sigma_t<1 \qquad (6.12)$$

Where σ_t is the growth rate multiplier, k_B is carrying capacity and B_t is biomass per hectare at time period t (tons/ha), as defined earlier.

Although little is known on the effects of water regimes or the productivity of wetland plant species, changes in wetland groundwater are bound to affect wetland biomass production. For instance, as the groundwater level is lowered through the wetland's conversion to agriculture, wetland vegetation is adversely affected and loses the competitive struggle with non-wetland plant species (Eppink *et al*. 2004). Therefore, the actual growth rate of biomass is linked to changes in wetland groundwater level in a linear form. This relationship links this module to the hydrology module and captures the trade-offs between crop production and wetland natural resources production due to competition for water. Given that there is very limited literature on the relationship between the below ground groundwater level and biomass growth, the above ground water depth-reeds growth correlations done by Tarr *et al*. (2004) is relied upon to obtain a gross parameter estimate on the wetland groundwater level effects on biomass growth.

Therefore, the actual growth rate is given by:

$$r_t = s_0 * \sigma_t + \mu_1 \mathrm{GWL}_t \qquad (6.13)$$

Where s_0 is the intrinsic growth rate, μ_1 is a parameter, σ_t is the growth rate multiplier and GWL is the wetland groundwater level as defined earlier.

Total biomass stock (TB_t) (measured in tons) is calculated as a product of biomass per hectare (B_t) and wetland area (TA_t^w) minus quantity of biomass harvested (h_t):

$$TB_t = TA_t^w * B_t - h_t \qquad w = \text{wetland system} \qquad (6.14)$$

The quantity of biomass harvested (h_t) is a product of the reduced form household biomass supply function (X_H^H) (measured in tons per household per year) which is derived from an agricultural household model in equation 4.1 in Chapter 4 with the slope parameter adjusted with the average values of the variables of household characteristics and the number of biomass harvesting households (NH_t):

$$h_t = NH_t * X_H^H \qquad (6.15)$$

The number of biomass harvesting households varies over time and is influenced by the total biomass stock. It is assumed that the number of households that harvest households is positively related to the total biomass stock. For as the total biomass stock declines, so does the number of households that harvest biomass and the efforts required to meet the required biomass needs, increases. As time series data on the total biomass stock for the study area does not exist, the author resorted to fitting historical annual time series data on the natural wetland area (which is used as a proxy for total biomass stock) and the number of wetland harvesting households using a simple linear regression in order to estimate the parameter (c). The relationship between the number of biomass harvesting households and the total wetland area is given by the following equation:

$$NH_t = c * TA_t^w \qquad (6.16)$$

Where c is a parameter.

Given that the actual number of wetland biomass harvesting households is known for the survey year, the other years we extapolated by assuming that it is 24% of the total number of households (which is the proportion of households engaged in harvesting obtained from the survey). As the wetland area is used as a proxy for wetland biomass, the parameter c was adjusted to take into account the average biomass per hectare.

The biomass supply function is influenced by several exogenous factors as shown in equation 4.1 in chapter 4. The labour used in biomass harvesting (equation 4.2 with the slope parameter adjusted with the average values of the variables of household characteristics) is used to compute the labour costs incurred in biomass harvesting. The labour cost function for biomass harvesting (b_t) is given as:

$$b_t = W_t * L_H * NH_t \qquad (6.17)$$

Where L_H is the labour used in biomass harvesting, measured in hours per harvesting household per year.

Therefore, the net value of biomass harvested (V_t) is given by:

$$V_t = h_t * P_{H,t} - b_t \qquad (6.18)$$

$P_{H,t}$ is the market price of harvested biomass at time period t.

6.3.5 The economic well-being module

This module deals with the welfare of the human population in the study area, which influences the demand for grain and wetland natural products for their own consumption and sales for cash income. Communities living in the area also supply labour for these activities. Following Woodwell (1998) and Hellden (2008) this study used an exponential population growth function where population growth is assumed to vary with natural growth rate, g (birth and death rate) and out-migration (EM_t). Although both death rate and birth rate are dependent on a number of factors (e.g.

family policies, access to markets and health services,) these are not considered in the model. However, it is assumed that emigration rates (e_t) vary over time and are influenced by the availability of off-farm employment opportunities (the proxy for this is GDP per capita) and rainfall. Low rainfall reduces agricultural productivity, which results in more people migrating to urban areas to seek off-farm income opportunities to cushion themselves from income shocks. Therefore, the population in the study area is linked to GDP per capita and rainfall through the emigration rate equation (equation 6.21).

The initial population was set in such a way as to reach the population levels of the study area in 2006 that were estimated to be approximately 2700 people (Adekola, 2007). The average annual population growth rate for the area is set at the district average which is estimated at 1.7% (Statistics South Africa, 2004). Focus group discussions conducted in the study area showed that immigration (in-migration) is minimal and therefore we assume that there is no in-migration in the area so the immigration rate is set at zero.

Therefore, the population dynamics are given by:

$$Pop_{t+1} = Pop_t(1+g) - EM_t \qquad (6.19)$$

Where Pop_t is as defined earlier, g is the natural population growth rate and EM_t is the number of emigrants at time t

The number of emigrants is estimated using the following equation:

$$EM_t = e_t * Pop_t \qquad (6.20)$$

Where e_t is the emigration rate.

The equation for emigration rate is specified as follows:

$$e_t = f_0 + f_1 GDP_{k,t} + f_2 P_t \qquad (6.21)$$

Where f_0, f_1 and f_2 are parameters; GDP_k and P_t represent GDP per capita and precipitation, respectively.

The parameters for equation 6.21 were derived from estimating a regression of historical emigration rates. These rates were for a typical rural area in South Africa for the midpoint years of a five-year period given by Kok and Collison (2006) with national the GDP per capita figures and the annual rainfall data corresponding to these years for the area.

In each given period, population determines the total labour supply and hence the total available labour (LS_t) (measured in hours per year) is specified as follows:

$$LS_t = (\kappa_1 m_1 + \kappa_2 m_2) * Pop_t \tag{6.22}$$

Where κ_1 and κ_2 are parameters representing the proportion of adults and children in the population, respectively; m_1 and m_2 are also parameters representing the total labour supply per adult and child, respectively (measured in hours/person/year).

Labour demand (LD_t) is given by summing the labour demand for each of the livelihood activities taking into account the number of households involved in each of the activities:

$$LD_t = L_{o,t} * NO_t + L_{H,t} * NH_t + L_{G,t} * \sum_i Q_i \tag{6.23}$$

Where L_o, L_H and L_G represent the labour used in off-farm work, biomass harvesting and grain production (measured in hours/household/year), respectively; NO_t and NH_t represent the number of households engaged in off-farm work and biomass harvesting, respectively.

It is assumed that labour is free to move between the different livelihood activities. Thus, the market clearing condition was imposed to solve the equilibrium wage rate as follows:

$$LS_t = LD_t + L_{z,t} * TH_t \qquad (6.24a)$$

Where $L_{z,t}$ represents the labour time used in leisure (hours per household per year) and TH_t the total number of households.

The total number of households (TH_t) is equal to the population divided by the average household size (hs_0):

$$TH_t = Pop_t / hs_0 \qquad (6.24b)$$

This module also derives the value of services of the wetland ecosystem and income from different sources. Four main forms of income are considered in the model: the net value of grain production; the net value of biomass harvested; off-farm wage income; and exogenous income (income from government social grants)[18]. The net values of grain and biomass harvested are shown by equations 6.8 and 6.18, respectively.

Off-farm wage income is a function of labour time used in off-farm work and the wage rate. The labour time used in off-farm work per household (L_o) (in hours/household/year) is also a function of exogenous factors as shown in equation 4.2 in chapter 4.

Therefore, the off-farm income function (O_t) is specified as:

$$O_t = NO_t * W_t * L_{o,t} \qquad (6.25)$$

Where NO_t is the number of households engaged in off-farm work.

[18] Crop income and natural resource income gives the net value of all productions and harvested biomass at market prices including productions or harvests sold, consumed and retained.

The number of households engaged in off-farm work assumedly changes over time and is influenced by national economic performances measured in terms of GDP per capita (which is used as a proxy for availability of off-farm employment opportunities). Historical employment figures from ward level census data in the study area and the national GDP per capita figures were used for a regression analysis to establish the relationship between the number of households engaged in off-farm work and the GDP per capita. The equation for NO_t is given by:

$$NO_t = d_0 + d_1 GDP_{k,t} \tag{6.26}$$

Where d_0 and d_1 are parameters and GDP_k is GDP per capita.

The main form of exogenous income in the study area is government transfers through child grants given for children under the age of 14. The equation for exogenous income (social transfers) (E_t) is specified as:

$$E_t = z_t * NS_t \tag{6.27}$$

Where z_t is the social grant rate (Rand/beneficiary/year) and NS_t is the number of households that benefit from social grants.

The National Treasury of South Africa (2008) highlighted that the social grant rate has been increasing over the years in line with inflation, mainly to protect its purchasing power. Based on this observation, the author assumes that the social grant rate is a function of the consumer price index (CPI) and used historical social grants rates and CPI values to regress these two variables and find parameters for their relationship. The social grant rate can be expressed as:

$$z_t = k_0 + k_1 CPI_t \tag{6.28a}$$

Where k_0 and k_1 are parameters and CPI is the consumer price index.

The number of households benefiting from social grants (NS_t) is assumed to be a proportion of the total number of households at each time period and is given by the following relationship:

$$NS_t = n_0 * TH_t \tag{6.28b}$$

Where n_0 is a parameter.

The total net income for the population in time period t, NI_t, is the summation of income derived from off-farm wage work, exogenous sources (social grants) and net value of maize production and biomass harvested:

$$NI_t = R_t + V_t + O_t + E_t \tag{6.29}$$

It is assumed that the economic well-being of the targeted population in time t measured as net income per capita (SW_t) is a function of total net income such that the economic well-being function is given by:

$$SW_t = \frac{NI_t}{Pop_t} \tag{6.30}$$

The net income per capita is the measure (index) of economic well-being that is used to assess scenario outcomes.

6.4 The full system of equations showing the linkages between modelled ecological-economic systems

In order to clearly show the linkages between economic and ecological processes in the system being modeled we present the full system of equations and the model variables to solve for endogenously are defined in Table 6.2. The model is specified and solved in STELLA[19], a simulation software which is well suited for simulating dynamics of ecological-economic systems (Costanza and Gottlieb, 1998). The model is run on an annual time step.

(A) Hydrology module

Total evapotranspiration (mm): (i = r, w) $ET_t^i = ETc_t^i + ETv_t^i$ (6.1)

Actual crop evapotranspiration from cultivated area (mm): $ETc_t^i = ET_{a,t}^i * AC_t^i$

Actual evapotranspiration from natural vegetation (mm): $ETv_t^w = \eta * TA_t^w$

(6.2)

Soil moisture content (w = wetland) (mm): $MC_{t+1} - MC_t = P_t + CR_t - GR_t - ET_t^w$

(6.3)

Wetland groundwater level (m): $GWL_{t+1} = GWL_t + [GR_t + IS_t - GS_t - CR_t]/10^3$

(6.4a)

Groundwater recharge from wetland soils (mm):

$$GR = \text{If } (MC_t + P_t + CR_t - ET_t^w \succ WHC) \text{ then } (MC_t + P_t + CR_t - ET_t^w - WHC) * (AC_t^w/120) \text{ else } 0$$
(6.4b)

(B) Crop production module

Actual crop yield (tons/ha): $Y_a^i = Y_m^i \left[1 - k_y * \left(1 - ET_a^i / ET_m^i\right)\right]$ (6.5)

Household grain supply function (kg/household/year):

$$G_{q,t} = \alpha_0 + \alpha_1 E_t + \alpha_2 W_t + \alpha_3 P_{G,t} + \alpha_4 P_{H,t} + \alpha_5 P_{M,t} + \alpha_6 P_{Y,t} \quad (4.1)$$

Household labour used in grain production (hours/household/year):

$$L_{G,t} = \beta_0 + \beta_1 E_t + \beta_2 W_t + \beta_3 P_{G,t} + \beta_4 P_{H,t} + \beta_5 P_{M,t} + \beta_6 P_{Y,t} \quad (4.2)$$

[19] The software requires that the variables in the system are categorised into stocks (state variables), flows (rate of change of stock variables) and converters (intermediate variables used for miscellaneous calculations). The linkages between these through difference equations represent the links between the ecological and economic components in the integrated model. The model state variables are presented in Table 6.3.

Total grain supply (tons):

$$TG_t^i = \left(\sum_{q=1}^{Q_i} G_{q,t}^i \Big/ 10^3 \right) \tag{6.6a}$$

Total area cultivated (ha):

$$AC_t^i = \sum_{q=1}^{Q_i} A_{q,t}^i$$

Households cultivating the wetland system: $Q_w = AC_t^w / wc_0$ (6.6b)

Average yield (tons/ha): $Y_{a,t}^i = TG_t^i / AC_t^i$ (6.7)

Net value of grain (Rands):

$$R_t = P_{G,t} \sum_i \left(Y_{a,t}^i AC_t^i \right) - P_{W,t} \sum_i \left(ET_a^i AC_t^i \right) - W_t * L_{G,t} * \sum_i Q_i \tag{6.8}$$

(C) Land use change module

Total wetland area (ha): $TA_{t+1}^w = TA_t^w - WCA_t$ (6.9)

Total area of wetland under cultivation (ha): $AC_t^w = a_1 P_t + a_{2,} P_{G,t} + a_3 P_{Y,t} + a_4 Pop_t$

(6.10a)

Area of wetland converted to cultivation (ha):

$$AC_{t+1}^w - AC_t = WCA_t \tag{6.10b}$$

(D) Natural wetland vegetation module

Biomass per hectare (tons/ha): $B_{t+1} = B_t (1 + r_t)$ (6.11)

Growth rate multiplier: $\sigma_t = GRAPH\,(B_t / k_B)\,;\, 0 < \sigma_t < 1$ (6.12)

Actual growth rate: $r_t = s_0 * \sigma_t + \mu_1 GWL_t$ (6.13)

Total biomass stock (tons): $TB_t = TA_t^w * B_t - h_t$ (6.14)

Total biomass harvested (tons): $h_t = NH_t * X_{H,t}^H$ (6.15)

Number of biomass harvesting households: $NH_t = c * TA_t^w$ (6.16)

Household biomass supply function (tons/household/year):

$$X^{H}_{H,t} = \theta_0 + \theta_1 E_t + \theta_2 W_t + \theta_3 P_{G,t} + \theta_4 P_{H,t} + \theta_5 P_{M,t} + \theta_6 P_{Y,t} \quad (4.1)$$

Labour cost for biomass harvesting (Rands):

$$b_t = W_t * L_{H,t} * NH_t \quad (6.17)$$

Household labour used in biomass harvesting (hours/household/ year):

$$L_{H,t} = \rho_0 + \rho_1 E_t + \rho_2 W_t + \rho_3 P_{G,t} + \rho_4 P_{H,t} + \rho_5 P_{M,t} + \rho_6 P_{Y,t} \quad (4.2)$$

Net value of harvested biomass (Rands):

$$V_t = h_t * P_{H,t} - b_t \quad (6.18)$$

(E) Economic well-being module

Population (No. of people): $Pop_{t+1} = Pop_t(1+g) - EM_t$ (6.19)

Number of Emigrants (No. of people): $EM_t = e_t * Pop_t$ (6.20)

Emigration rate: $e_t = f_0 + f_1 GDP_{k,t} + f_2 P_t$ (6.21)

Total labour supply (hours/year): $LS_t = (\kappa_1 m_1 + \kappa_2 m_2) * Pop_t$ (6.22)

Total labour used in livelihood activities (hours/year):

$$LD_t = L_{o,t} * NO_t + L_{H,t} * NH_t + L_{G,t} * \sum_i Q_i \quad (6.23)$$

Labour market equilibrium: $LS_t = LD_t + L_{z,t} * TH_t$ (6.24a)

Total number of households (hhlds): $TH_t = Pop_t / hs_0$ (6.24b)

Off-farm income (Rands/year): $O_t = NO_t * W_t * L_{o,t}$ (6.25)

Household labour used in off-farm work (hours/household/year):

$$L_{o,t} = \delta_0 + \delta_1 E_t + \delta_2 W_t + \delta_3 P_{G,t} + \delta_4 P_{H,t} + \delta_5 P_{M,t} + \delta_6 P_{Y,t} \quad (4.2)$$

Number of households engaged in off-farm work (households):

$$NO_t = d_0 + d_1 GDP_{k,t} \quad (6.26)$$

Exogenous income (Rands/year): $E_t = z_t * NS_t$ (6.27)

Social grant rate (Rand/beneficiary/year): $z_t = k_0 + k_1 CPI_t$ (6.28a)

Number of social grants beneficiaries (hhlds): $NS_t = n_0 * TH_t$ (6.28b)

Total net income (Rands/year): $NI_t = R_t + V_t + O_t + E_t$ (6.29)

Economic well-being (Rands/capita): $SW_t = \dfrac{NI_t}{Pop_t}$ (6.30)

Table 6.2: Definition of endogenous model variables

Variable	Definition	Units
ET^i_t	Total evapotranspiration for system i (i = wetland or irrigation system) at the time period t	Millimetres
ETc^i_t	Actual total crop evapotranspiration from cultivated area from system i (i = wetland or irrigation system) at the time period t	Millimetres
ET^i_a	Actual crop evapotranspiration per hectare of cultivated area in system i (i = wetland or irrigation system)	Millimetres/ha
ET^w_v	Actual evapotranspiration from natural wetland vegetation	Millimetres
GWL_t	Wetland groundwater level at the time period t	Metres
GS_t	Groundwater discharge from wetland at the time period t	Millimetres
CR_t	Capillary rise at the time period t	Millimetres
GR_t	Groundwater recharge from wetland soils	Millimetres
MC_t	Wetland soil water content	Millimetres
TA^w_t	Total wetland area at the time period t	Hectares
AC^w_t	Total area of wetland under cultivation	Hectares
WCA_t	Area of wetland converted to cultivation	Hectares
Q_w	Number of households in the wetland system	Households
Y^i_a	Actual crop yield (i= wetland or irrigation system)	Tons/ha
$G_{q,t}$	Household grain supply at the time period t	kg/household/year
$L_{G,t}$	Household labour used in grain production at the time period t	Hours/household/year
TG^i_t	Total grain supply from system i (i= wetland or irrigation system) at the time period t	Tons
R_t	Net value of grain at the time period t	Rands/year
B_t	Biomass per ha at the time period t	Tons/ha
TB_t	Total biomass stock	Tons
V_t	Net value of harvested biomass at the time period t	Rands/year
r_t	Actual growth rate at the time period t	Non-dimensional
σ_t	Growth rate multiplier at the time period t	Non-dimensional
TB_t	Total biomass stock at the time period t	Tons
h_t	Total biomass harvested at the time period t	Tons
NH_t	Number of biomass harvesters at the time period t	Households
$X^H_{H,t}$	Household biomass supply at the time period t	Tons/household/year
b_t	Labour costs for biomass harvesting at the time period t	Rands/year

Table 6.2 (continued): Definition of endogenous model variables

Variable	Definition	Units
$L_{H,t}$	Household labour used in biomass harvesting at the time period t	Hours/household/year
Pop_t	Population at the time period t	People
EM_t	Number of emigrants at the time period t	People
e_t	Emigration rate at the time period t	Non-dimensional
TH_t	Total number of households at the time period t	Households
NS_t	Number of social grants beneficiaries	Households
LS_t	Total labour supply at the time period t	Hours/year
O_t	Off-farm income at the time period t	Rands/year
NO_t	Number of households engaged in off-farm work at the time period t	Households
$L_{o,t}$	Household labour time used in off-farm work at the time period t	Hours/household/year
LD_t	Total labour demand by livelihood activities at the time period t	Hours/year
E_t	Exogenous income at the time period t	Rands/year
z_t	Social grant rate at the time period t	Rands/person/year
NI_t	Total net income at the time period t	Rands/year
SW_t	Human well-being at the time period t	Rands/capita

Table 6.3: State variables (stocks) in the model

Module variable	Definition	Units
GWL_t	Wetland groundwater level	Metres
MC_t	Wetland soil water content	Millimetres
AC_t^w	Total area of wetland under cultivation	Hectares
TA_t^w	Total wetland area	Hectares
B_t	Biomass per hectare	Tons/ha
Pop_t	Population at the time period t	People

6.5 Specification of model parameters and validation

Data for model parameters were obtained from a wide range of sources. Table 6.4 presents the full model parameter values and their sources.

Table 6.4: Parameter values and sources

Parameter label	Symbol	Value	Source
Crop yield response to water factor for maize	k_y	1.25	Durand (2008)
Constant in the grain supply function	α_0	6.47	Agricultural household model grain supply function estimates from Chapter 4; adjusted by the average values of household size and education.
Coefficient for exogenous income in the grain supply function	α_1	0.01	Agricultural household model grain supply function estimates from Chapter 4
Coefficient for wage rate in the grain supply function	α_2	-0.013	Agricultural household model grain supply function estimates from Chapter 4
Coefficient for the price of grain in the grain supply function	α_3	0.06	Agricultural household model grain supply function estimates from Chapter 4
Coefficient for the price of wetland biomass in the grain supply function	α_4	-0.01	Agricultural household model grain supply function estimates from Chapter 4
Coefficient for the price of market goods in the grain supply function	α_5	-0.08	Agricultural household model grain supply function estimates from Chapter 4
Coefficient for the price of agricultural inputs in the grain supply function	α_6	-0.08	Agricultural household model grain supply function estimates from Chapter 4
Constant in the labour use equation for grain production	β_0	8.38	Agricultural household model estimates of labour use in grain production from Chapter 4 adjusted by the average values of household size and education.
Coefficient for exogenous income in the labour use for grain production	β_1	-0.016	Agricultural household model estimates of labour use in grain production from Chapter 4.
Coefficient for wage rate in the labour use equation for grain production	β_2	-0.039	Agricultural household model estimates of labour use in grain production from Chapter 4.
Coefficient for the price of grain in the labour use equation for grain production	β_3	0.054	Agricultural household model estimates of labour use in grain production from Chapter 4.
Coefficient for the price of wetland biomass in the labour use equation for grain production	β_4	-0.01	Agricultural household model estimates of labour use in grain production from Chapter 4.
Coefficient for the price of market goods in the labour use equation for grain production	β_5	-0.001	Agricultural household model estimates of labour use in grain production from Chapter 4.
Coefficient for the price of agricultural inputs in the labour use equation for grain production	β_6	-0.01	Agricultural household model estimates of labour use in grain production from Chapter 4.
Constant in the biomass supply function	θ_0	82.81	Agricultural household model estimates of the wetland biomass supply function from Chapter 4; adjusted by the average values of household size and education.
Coefficient for exogenous income in the biomass supply function	θ_1	-0.09	Agricultural household model estimates of the wetland biomass supply function from Chapter 4
Coefficient for wage rate in the biomass supply function	θ_2	-0.036	Agricultural household model estimates of the wetland biomass supply function from Chapter 4
Coefficient for the price of grain in the wetland biomass supply function	θ_3	-0.13	Agricultural household model estimates of the wetland biomass supply function from Chapter 4
Coefficient for the price of wetland biomass in the wetland biomass supply function	θ_4	0.01	Agricultural household model estimates of the wetland biomass supply function from Chapter 4
Coefficient for the price of market goods in the wetland biomass supply function	θ_5	-0.37	Agricultural household model estimates of the wetland biomass supply function from Chapter 4

Table 6.4 (Continued): Parameter values and sources

Parameter label	Symbol	Value	Source
Coefficient for the price of agricultural inputs in the wetland biomass supply function	θ_6	0.11	Agricultural household model estimates of the wetland biomass supply function from Chapter 4
Constant in the labour use equation for wetland biomass harvesting	ρ_0	13.41	Agricultural household model estimates of labour use in wetland biomass collection from Chapter 4; adjusted by the average values of household size and education.
Coefficient for exogenous income in the labour use equation for biomass collection	ρ_1	-0.02	Agricultural household model estimates of labour use in wetland biomass collection from Chapter 4
Coefficient for wage rate in the labour use equation for biomass collection	ρ_2	-0.086	Agricultural household model estimates of labour use in wetland biomass collection from Chapter 4
Coefficient for the price of grain in the labour use equation for biomass collection	ρ_3	-0.45	Agricultural household model estimates of labour use in wetland biomass collection from Chapter 4
Coefficient for the price of wetland biomass in the labour use equation for biomass collection	ρ_4	0.02	Agricultural household model estimates of labour use in wetland biomass collection from Chapter 4
Coefficient for the price of market goods in the labour use equation for biomass collection	ρ_5	-0.12	Agricultural household model estimates from Chapter 4
Coefficient for the price of agricultural inputs in the labour use equation for biomass collection	ρ_6	0.34	Agricultural household model estimates of labour use in wetland biomass collection from Chapter 4
Constant in off-farm labour use equation	δ_0	-6.60	Agricultural household model estimates of labour use in off-farm work from Chapter 4; adjusted by the average values of household size and education.
Coefficient for exogenous income in off-farm labour use equation	δ_1	-0.74	Agricultural household model estimates of labour use in off-farm work from Chapter 4
Coefficient for wage rate in off-farm labour use equation	δ_2	0.014	Agricultural household model estimates of labour use in off-farm work from Chapter 4
Coefficient for the price of grain in off-farm labour use equation	δ_3	-0.12	Agricultural household model estimates of labour use in off-farm work from Chapter 4
Coefficient for the price of wetland biomass in off-farm labour use equation	δ_4	-0.01	Agricultural household model estimates of labour use in off-farm work from Chapter 4
Coefficient for the price of market goods in off-farm labour use equation	δ_5	-0.93	Agricultural household model estimates of labour use in off-farm work from Chapter 4
Coefficient for the price of agricultural inputs in off-farm labour use equation	δ_6	0.64	Agricultural household model estimates of labour use in off-farm work from Chapter 4
Natural population growth rate	g	0.017	Statistics South Africa (2004)
Constant in the number of people employed off-farm-GDP per capita regression	d_0	-3.62	Regression analysis of the number of people employed in off-farm work and the GDP per capita
Coefficient for GDP per capita effect on number of people employed in off-farm work	d_1	0.01	Regression analysis of the number of people employed in off-farm work and the GDP per capita
Constant in the emigration rate equation	f_0	-4.17 e(-03)	Multiple regression analysis of the emigration rate, GDP per capita and rainfall
Coefficient for GDP per capita effect on emigration rate	f_1	2.70e(-07)	Multiple regression analysis of the emigration rate, GDP per capita and rainfall
Coefficient for rainfall effect on emigration rate	f_2	-6.9e(-07)	Multiple regression analysis of the emigration rate, GDP per capita and rainfall

Table 6.4 (Continued): Parameter values and sources

Parameter label	Symbol	Value	Source
Coefficient for the effect of CPI on social grant rate	k_1	1.58	Social grant rate-consumer price index regression analysis
Biomass carrying capacity	k_B	70tons/ha/year	Finlayson and Moser, 1991 cited in Turpie *et al.*1999
Constant for the CPI effect on social grant rate	k_0	-48.35	Social grant rate-consumer price index regression analysis
Coefficient of rainfall in cultivated wetland area regression	a_1	-0.042	Multiple regression estimates of wetland cultivated area and rainfall, grain price, agricultural input price and population
Coefficient of grain price in cultivated wetland area regression	a_2	0.021	Multiple regression estimates of wetland cultivated area and rainfall, grain price, agricultural input price and population
Coefficient of the price of agricultural input (seed maize) in the cultivated wetland area regression	a_3	-0.041	Multiple regression estimates of wetland cultivated area and rainfall, grain price, agricultural input price and population
Coefficient of the population in the cultivated wetland area regression	a_4	0.032	Multiple regression estimates of wetland cultivated area and rainfall, grain price, agricultural input price and population
Proportion of working adults (aged 15-64years) in the population	κ_1	0.5	Statistics South Africa (2004)
Proportion of children (aged 4-15years) in the population	κ_2	0.3	Statistics South Africa (2004)
Total labour supplied per adult per year (hours)	m_1	1600	Stephenne and Lambin (2001)
Total labour supplied per child per year (hours)	m_2	400	Stephenne and Lambin (2001); adjusted to take into account the fact that most of children go to school
Intrinsic growth rate for wetland biomass	s_0	0.3	Thenya (2006)
Coefficient for biomass stock in the regression for the number of biomass harvesters	c	0.0042	Regression analysis of the number of biomass harvested and natural wetland area historical time series data
Field capacity of the soil	WHC	140mm/m	Saxton and Rawls (2006)
Area under irrigation	AC^r	170ha	Chiron (2005)
Total number of households under irrigation	Q_r	283 households	Computed by dividing the area under irrigation (from Chiron, 2005) by the irrigated area per household, which is 0.6 ha per household (from household survey data)
Cultivated wetland area per household	wc_0	0.66	Household survey
Proportion of households that obtain social grants	n_0	0.64	Household survey
Coefficient of the effect of the groundwater level on wetland biomass growth rate	μ_1	0.0001	Estimate based on the reeds yield-water depth correlations by Tarr *et al.* (2004)
Average household size	hs_0	7.3 people	Household survey
Actual evapotranspiration from natural wetland vegetation per year	η	1100mm/ha/year	Dye *et al.* (2008); Von der Heyden and New (2003); Kleynhans (2004).
Precipitation	P_t	500mm/year	McCartney (2005)
Maximum grain yield in wetland	Y_m^w	3tons/ha	Chiron (2005)

Table 6.4 (Continued): Parameter values and sources

Parameter label	Symbol	Value	Source
Maximum grain yield in irrigation	Y_m^w	2.5 tons/ha	Chiron (2005)
Maximum crop evapotranspiration per season	ET_m	490mm/year	Durand (2008)
Wage rate	W_t	R8/hour	Adekola (2007)
Price of grain	$P_{G,t}$	R1.58/kg	Household survey
Price of wetland biomass	$P_{H,t}$	R2/kg	Adekola (2007)
Price of agricultural inputs	$P_{Y,t}$	R5.29/kg	Household survey
Price of water	$P_{W,t}$	R0.13/mm	Department of Water Affairs
Price of market goods	$P_{M,t}$	R345	Household survey
Consumer Price Index	CPI_t	138	SARB (2009)
GDP per capita	$GDP_{k,t}$	R34234/capita/annum	SARB (2009)

In system dynamic modelling, the ultimate objective of the validation process is to establish the structural validity of the model with respect to the modelling purpose. Confidence in the model simulation results is high only if the model has robust predictive ability in reproducing historical trends. Dynamic simulation models are validated by comparing model predicted versus observed past trends for selected variables. However, the validity tests should place emphasis on pattern prediction of key variables rather than point predictions, mainly because of the long-term orientation of these models (Güneralp and Barlas, 2003). Because of the limited availability of observed time series data for most of the variables in the model, the validation exercise was done for a few variables for which past trend data could be obtained. The period used for the validation is 1990 to 2006. After validation the model will be used to conduct policy simulations for a 14-year post validation period, (i.e. 2006 to 2020).

Figures 6.3 and 6.4 compare the observed versus the model predicted values for the wetland area converted to agriculture and social grant rates, respectively. Figure 6.3 shows that the wetland area converted to agriculture has been increasing with a corresponding decrease in the wetland area. This has been primarily driven by the increasing frequency of droughts, which increases wetland conversion rates due to its fertile soils and ability to retain soil moisture.

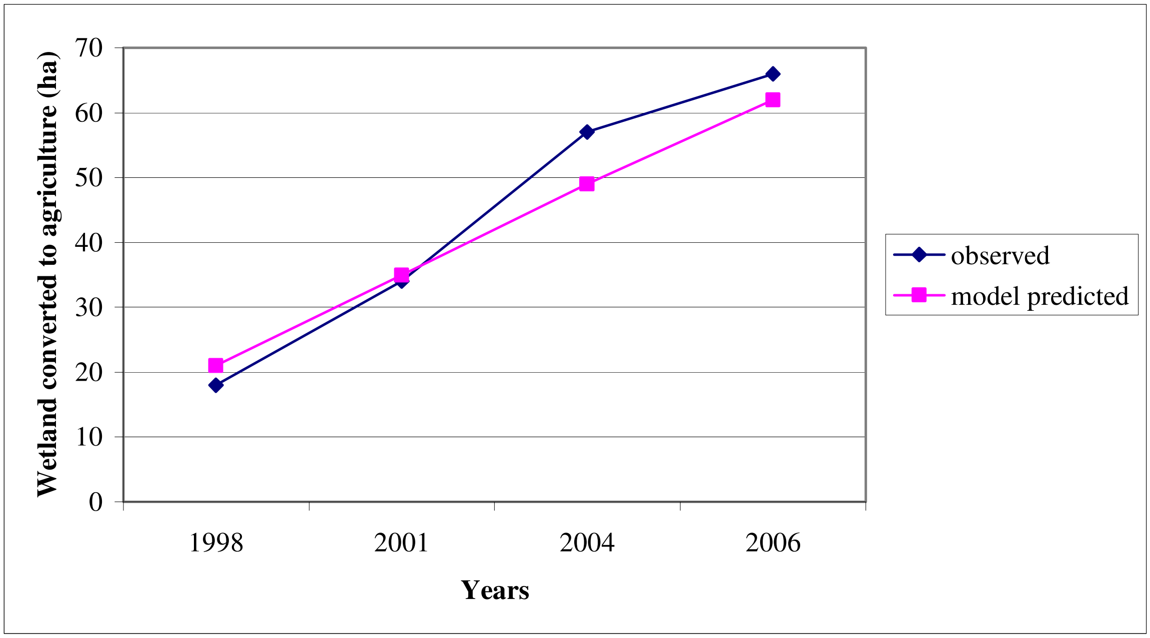


Figure 6.3: Comparison of model predicted and actual wetland area converted to agriculture (Observed data obtained from: Sarron, 2005; Adekola, 2007)

The predicted social grant rate follows an increasing trend in line with the observed trend due to an increase in inflation (Figure 6.4). Whilst the model predicted values are not exactly equal to the observed values in both cases, the model does well in predicting the observed pattern of these two variables. The correlation between the model predicted and the observed values is more than 0.9 in both cases, suggesting that the model can be used with confidence.

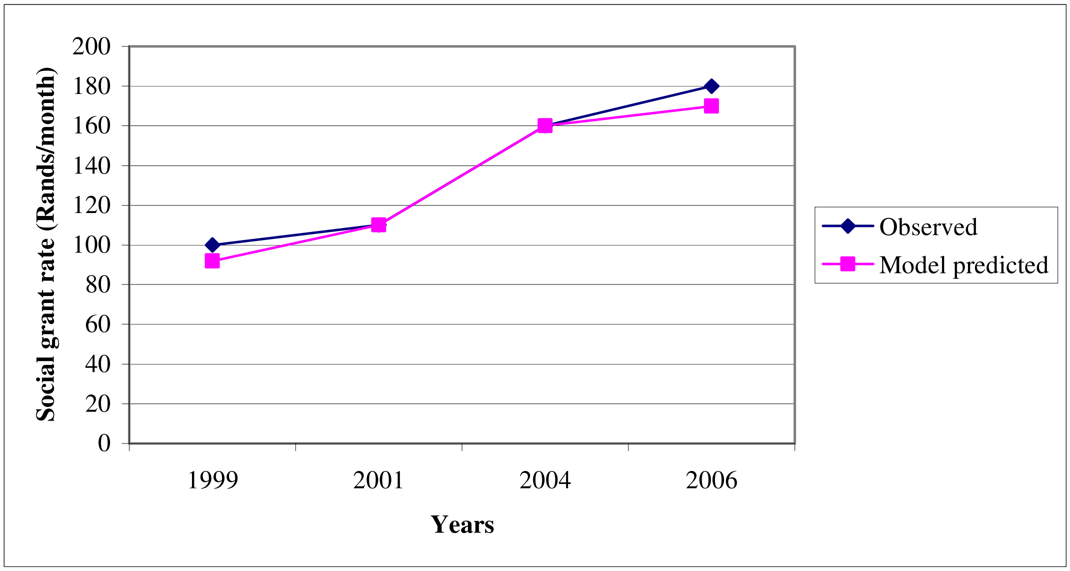


Figure 6.4: Comparison of model predicted versus actual social grant rate (Observed data obtained from: National Treasury, 2008)

Clearly it would be possible to establish a much stronger case if more numerical time series data were available for more variables in the model. Lack of past trend data on most variables severely restricted the study's validation options and collecting new dynamic data necessitates long time periods. However, it should be kept in mind that the main purpose of this model is to capture broad dynamic behaviour patterns of the real system, not provide point predictions.

6.6 Simulation of impacts of alternative wetland management and policy regimes

The first step in performing a simulation experiment is to run the baseline scenario, which becomes the benchmark against which simulated scenarios are compared. Scenario simulations are performed by changing values of exogenous variables in the model and comparing the outcomes with the base scenario. Policy scenarios considered for simulations are selected on the basis of possible government policy interventions. The policy scenarios simulated include: tax and subsidy policy regimes that work through changing effective prices of agricultural outputs, inputs, and market goods; as well as government policy instruments such as direct income transfers and changes in availability of off-farm work which are driven by changes in social policy and economic growth trends.

In order to maintain a functional wetland ecosystem in which biodiversity protection is maximal it is necessary to put part of the wetland area under protection. However, total protection is not always necessary in order to maintain high levels of diversity, but would be necessary if the goal is to maintain an ecosystem intact in its natural state, which in most cases is done for promoting ecotourism. In this study's simulation experiments the author considered a scenario of partial protection through placing some percentage of the wetland under conservation.

Although climate change predictions for precipitation are less consistent, most simulations for southern Africa indicate that rainfall will decline in the next 100 years. Predictions for 2050 show that rainfall in southern Africa could be 10% to 20% lower than the 1950 to 2000 averages (IPCC, 2001). Based on these predictions, a scenario of a 10% reduction in annual precipitation is considered in the simulation experiments.

To evaluate the social desirability of simulated intervention scenarios, final outcome values are compared (values at the end of the simulation period, which is the year 2020) for selected indicators with the baseline scenario as done in other studies (Eppink *et al*., 2004; Saysel *et al*. 2002). As the primary purpose of this study is to investigate the impacts of alternative policy regimes on wetland functioning, ecosystem services and human well-being, the key variables considered in the evaluations are: (1) wetland crop (grain) production and harvested biomass and their values (the two wetland services considered in the model); (2) the total wetland area and the total biomass stock (indicators of wetland conservation status), (3) wetland soil water content and groundwater level (indicators of wetland hydrological regulation services) and (4) net income per capita (a proxy for human well-being). The specific policy scenarios evaluated and results of the simulation experiments are given in Table 6.5.

Table 6.5: Changes in value of selected indicator variables, expressed as percentages of baseline values

Percentage change in indicator variables compared to their baseline levels									
Policy scenarios	Total biomass harvested (tons)	Total biomass stock (tons)	Total wetland grain supply (tons)	Wetland ground water level (m)	Soil water content (mm)	Net value of wetland grain (Rands)	Net value of harvested biomass (Rands)	Total wetland area (ha)	Net income per capita (Rands/capita/year)
(1) Taxing grain production (30% on price)	0.11	0.04	-0.01	0.26	0.01	-10.06	4.29	0.43	-0.21
(2) Taxing biomass products (30% on price)	-0.11	0.06	0.01	-0.01	-0.01	0.04	-17.79	-0.12	0.02
(3) Combined tax on grain and biomass (30% each)	-0.85	0.15	-0.01	0.45	0.07	-10.07	-10.15	0.46	-0.23
(4) 30% increase in agricultural input prices[a]	0.12	0.14	-0.21	1.76	5.1	-0.19	3.6	1.15	-0.28
(5) 30% increase in the off-farm wage rate	-0.55	0.01	-0.01	0.04	0.01	-39.58	-26.65	0.14	6.59
(6) Increased availability of off-farm opportunities (5% increase in GDP per capita)	0.10	0.29	-0.81	0.27	1.42	-0.01	0.03	2.72	6.40
(7) Putting 30% of wetland area under protection	-0.22	38.63	-22.45	0.06	43.77	-22.45	-0.45	92.98	-0.46
(8) 10% reduction in precipitation	-0.89	-33.01	1.60	-0.91	-13.6	-4.10	-2.24	-76.58	-0.13

[a]Price of maize seed is used as this is the key variable input used in wetland grain production.

A total of eight policy experiments have been simulated. Simulation results show that taxing wetland conversion to agriculture through reduced grain output prices (scenario 1) weakens the incentive for expanding the cultivated area in the wetland, leading to decreases in wetland crop production. This leads to an increase in the total wetland area and thus lowers evapotranspiration from cultivated land (crop water use), reducing the total evapotranspiration from the system. As a result, soil water content in the wetland increases lifting the wetland groundwater level as the recharge to groundwater is increased. In response, the actual growth rate of wetland biomass increases (equation 6.13) causing an increase in wetland biomass per hectare (equation 6.11).

The total biomass stock is consequently higher due to increases in the actual growth rate of biomass and total wetland area, and the number of biomass harvesters increases as a result. The net income per capita decreases due to the substantial reduction in the net value of grain production, which by far exceeds the increase in net value of biomass harvested. In a nutshell, taxing grain output production discourages wetland conversion to agriculture, which negatively impacts human well-being to the advantage of maintaining wetland ecological integrity.

Taxing the excessive harvesting of biomass products (scenario 2) through lowering the product prices, reduces the total biomass harvested and increase biomass stock. The wetland grain supply increases (equation 4.1) causing an increase in crop water use (ET_a) with consequent reductions in soil water content and wetland groundwater level. Although the reduction in the groundwater level reduces the natural wetland biomass growth (equation 6.13), the total biomass stock increases due to a reduction in the total of harvested biomass. The net value of harvested biomass decreases substantially due to a reduction in the total of harvested biomass. On the one hand, the incentive for grain production improves leading to a higher conversion of wetland area for agriculture, which in turn causes the net income per capita to increase. On the other hand, the result of this tax scenario also shows that increasing the price of harvested biomass increases returns to biomass products relative to that of wetland grain and therefore reduces conversion of wetland to agriculture.

These results demonstrate the trade-offs that need to be managed between improving human well-being in the short-run and conserving the wetland ecosystem (long-term sustainability goals), and between supply of the two wetland services (crop production and biomass harvesting) competing for water, labour and land resources.

A combined tax on both grain and biomass products (scenario 3) is found to be more effective in conserving the wetland and maintaining hydrological integrity than levying separate taxes on biomass and grain production. This of course comes at a higher welfare cost.

An alternative way of taxing wetland conversion is through increasing agricultural input prices (scenario 4), which has similar but stronger effects compared to increasing grain prices. It increases agricultural production costs and reduces returns to agricultural production and therefore reduces the rate of conversion of the wetland area to cultivated agriculture. As can be seen from Table 6.5, a much higher growth in total wetland area is obtained under the input price policy interventions than with the grain price tax policy (scenario 1). Also a much larger impact on water levels and wetland hydrology are realised. This, however, comes at a higher loss in the economic welfare measured in net income per capita. The above results suggest that, while policy interventions such as agricultural prices, support policies (e.g. subsidies) have the potential to improve the welfare of poor rural farmers they can also lead to agricultural intensification and environmental degradation.

Like taxing prices of other inputs, intervention through the urban wage rate policy instrument (scenario 5) reduces wetland grain supply (equation 4.1) and its value. Improving off-farm wages, however, results in substantial decreases in production and the net value of harvested biomass since labour is the main input in biomass harvesting and hence the high sensitivity to movements in wages. Despite this, the net income per capita increases due to a substantial increase in the off-farm income (equation 6.25) component of total net income (equation 6.29). At the same time the wage rate option achieves conservation objectives, but at lower levels compared to commodity price (tax/subsidy) regimes. This makes clear the importance of understanding the important distinctions carefully weighing the potential net impacts of alternative policy intervention choices and instruments.

The wetland area and net income per capita grow with the highest percentage through an increase in off-farm income opportunities (scenario 6). This result derives from the fact that an increase in off-farm income opportunities (through increasing GDP per capita) causes an increase in the emigration rate (equation 6.21). This leads to a reduction in the population, which in turn reduces the rate of wetland conversion to agriculture as demands for land and food is reduced. Accordingly, wetland grain supply and the net value of grain decline. Income from off-farm employment opportunities increases as the number of households engaged in off-farm work increases. The increase in off-farm income totally offsets reductions in net value of harvested biomass and grain resulting in a significant increase in net income per capita. Like improved off-farm wages, this scenario has a double dividend effect as it simultaneously improves economic well-being and conserves the wetland ecosystem.

This result demonstrates the potential for indirect economic incentive measures such as improving off-farm employment and income opportunities to contribute towards improving both human well-being and wetland conservation. However, as demonstrated by Brandon and Wells (1992) and Ferraro and Kramer (1997) such measures do not automatically lead to sustainable resource management and in some cases the availability of alternative income sources leads to the intensification of resource use activities. For alternative livelihood and income sources to spur conservation of wetland resources, it is important to emphasise the overall economic development in the area to increase the availability of off-farm employment opportunities outside of the natural resources or agriculture-based economy. Promoting livelihood diversification out of agriculture becomes an important strategy for enhancing sustainable wetland management.

The results of the wetland conservation strategy (scenario 7) show that the economic well-being of the local population declines considerably due to substantial reductions in the value of biomass harvested and grain produced in the wetland, as harvesting of natural products and the conversion of the wetland to cropland are restricted. However, the reduction in the economic welfare to the local community only takes into account direct use benefits of the wetland without considering its non-use values and indirect benefits of maintaining biodiversity intactness and hydrological regulation services.

The predicted reduction in precipitation (scenario 8) produces by far the worst results in terms of conserving the wetland. The wetland area declines by close to 90% due to an increased rate of conversion of the wetland to cultivation and total cultivated wetland area as rainfall declines (equations 6.10a and 6.10b). The rate of wetland conversion to cultivation increases as more households move into the wetland due to its ability to retain soil moisture throughout the year. As a consequence, the total area of wetland under cultivation expands and, accordingly, the total wetland area declines.

A reduction in precipitation adversely affects wetland soil moisture content and the groundwater level, which in conjunction with the recession of the total wetland area leads to a reduction in total biomass.

6.7 Concluding Summary

This chapter developed and applied a dynamic ecological-economic model to analyse the linkages between the economic and ecological elements in the wetland system under study. The model was used to analyse the impacts of various policy and management regimes on wetland functioning and economic well-being.

The model showed that economic and ecological systems are intricately linked with important feedback effects. Changes in the socio-economic system influence wetland ecosystem processes while changes in ecosystem processes influences the economic system through provision of services, which influence economic well-being.

The results of the policy simulations suggest that wetland ecosystem services (crop production and natural resource harvesting) are interlinked with subtle trade-offs involved through their competition for labour, water and land resources. Some policy interventions such as improving profitability of cultivation through supporting agricultural output prices and/or subsidizing input prices may improve economic well-being, but at the expense of wetland conservation.

Results also suggest that increasing off-farm income and employment opportunities has a double dividend effect, because it simultaneously improves economic well-being and enhances wetland conservation. Therefore, promoting livelihood diversification out of agriculture becomes an important strategy for enhancing sustainable wetland management.

A pure conservation strategy that aims at protecting the wetland leads to substantial reductions in economic welfare of the local population unless their livelihood sources are diversified into alternative non-farm employment and income options. This study also confirms that the predicted reduction in rainfall in southern Africa is likely to accelerate wetland conversion to agriculture and undermine wetland conservation efforts.

CHAPTER 7

SUMMARY, CONCLUSIONS AND IMPLICATIONS FOR POLICY AND RESEARCH

7.1 Introduction

This chapter summarises the key findings of this study and draw conclusions and policy insights from the research results. The first section of the chapter summarises the main findings of the study and draws policy implications. The section that follows articulates the limitations of the study and suggests possible areas for further research.

7.2 Summary of key findings and policy implications

This study developed an empirical model to analyse the determinants of rural household labour allocation and supply decisions for competing livelihood activities including the production of agricultural and wetland products. The study also developed a dynamic ecological-economic model based on the system dynamics framework and applied it to evaluate the trade-offs between provisions of various components of a bundle of multiple wetland services through simulation of the impacts of alternative management and policy regimes on wetland functioning, the services they provide and economic well-being. This aspect is largely ignored in the literature on wetlands in Africa. Most studies on wetlands in Africa have dwelled much on static economic valuation approaches aimed at valuing the contribution of wetland services to human welfare at a given time and do not consider the intertemporal nature of the interaction between ecological and economic systems. The results of the study are useful for designing effective policies to enhance sustainable management of wetland resources in developing countries.

Results of the study showed that improved household education level enhances diversification into off-farm work. The policy implication of this result is that the government needs to promote investments in education and skills development for the rural population to enhance diversification of their livelihoods out of agriculture and reduce pressure on wetlands.

The results also indicate that household exogenous income and wealth status (asset endowment) enhance farm production and reduce dependence on harvesting wetland products for livelihood. This result implies that government should pursue policy measures that reduce rural household liquidity constraints and enhance investment in productive assets (e.g. improving rural household access to credit and off-farm income opportunities) to boost farm production and provide positive incentives for the rural population to conserve wetlands.

Findings also suggest that asset-poor households with limited non-farm incomes, most of whom are female-headed, rely heavily on wetland products for their livelihood. This finding is consistent with the hypothesis that poorer households are more reliant on local environmental resources than wealthy households are. This also suggests that wetlands play an important role as livelihood safety nets for rural poor households by reducing their vulnerability to shocks such as droughts and other income shocks.

Two main policy implications can be drawn from this result. The first one is that the government, policy-makers and natural resource managers need to acknowledge the livelihood safety net role wetlands play in rural livelihoods and recognise that environmental protection policies limiting or banning access and use of wetland resources can deepen rural poverty, as the poor suffer more from the deprivation of these resources. Therefore, instead of adopting strict wetland protection policies, there is need to invest in the development and promotion of use of sustainable wetland management practices (in particular crop, livestock and natural products management practices) that allow the poor to utilise wetlands to enhance their economic well-being with minimum adverse effects on the wetland ecological condition. The second policy implication that can also be drawn from this result is the importance of the provision of safety nets for the poor through the promotion of government programmes and policies that support diversification into off-farm livelihood and income sources to provide positive incentives for wetland conservation and sustainable use. This suggests that sustainable wetland management has to be integrated within the broader rural development programmes aimed at reducing poverty in order to provide the necessary incentives for the poor to adopt sustainable wetland management options.

The dynamic ecological-economic model developed in this study demonstrated the importance of considering feedback effects between ecological and economic systems. Due to its modularity, the model developed in this study can easily be adapted to similar small-scale wetlands in southern Africa.

Policy scenario simulations using the model showed that policy interventions such as improving the profitability of cultivation through supporting agricultural output prices and/or subsidising input prices may improve economic well-being, but at the expense of wetland conservation.

Simulation results also suggest that increasing off-farm income opportunities has a double dividend effect because it simultaneously improves economic well-being and enhances wetland conservation. Therefore, promoting livelihood diversification out of agriculture becomes an important strategy for enhancing sustainable wetland management as also suggested earlier. Livelihood diversification can be supported through increased government investment in rural infrastructure, downstream value chains, health and education.

The simulation results further suggest that increasing returns to the collection of wetland natural products reduces wetland conversion. This implies that the development of a competitive marketing system for harvested biomass products, which increases returns to wetland biomass products relative to that of wetland grain, has the potential to reduce the conversion of wetlands to agriculture, which poses a major threat to the ecological integrity of the wetland than the harvesting of natural products.

The results also showed that a pure conservation strategy that aims at protecting the wetland leads to substantial loss in the economic welfare of the local population unless their livelihood sources are diversified into alternative non-farm employment and income options. This again emphasises the need to diversify the livelihood options for rural populations and also identify and promote local level sustainable wetland management strategies rather than putting wetlands under strict protection.

This study also confirms that the predicted reduction in annual rainfall in southern Africa is likely to accelerate wetland conversion to agriculture and undermine wetland conservation efforts. The implication of this result is that improving the capacity of rural farmers to adapt to climate change, especially droughts, is important to reduce pressure on wetland resources. Strategies that reduce the dependence on wetlands for agriculture should be promoted, such as: investments in water harvesting and storage; efficient irrigation methods; and promoting the use of drought tolerant crops and diversifying out of agriculture.

7.3 Limitations of the study and areas for further research

The agricultural household model presented in this study does not consider risk and uncertainty, which is a common feature in the environment under which rural households make decisions. Therefore, a possible extension of the present study is the development of a household model based on expected utility theory taking into account risk and uncertainty. In addition, the agricultural household model can also be improved by including institutional (property rights) and social factors that influence access and use of wetland resources.

Although the dynamic ecological-economic model that was developed generated useful results and policy insights for wetland management it has a number of limitations, which could be the basis for further research. The main challenges in the development of the model were the limited availability of data to validate the model and insufficient understanding of several feedback mechanisms in the modelled system. Possible improvements in the model include:

- including groundwater flow from hillslope to wetland, which is a key component of the hydrology of the wetland and artificial drainage activities which affect groundwater levels;
- modelling the hydrological processes at a monthly or seasonal time step instead of an annual time step to capture the seasonal variations of wetland water;

- adding a module sector on wetland soil organic matter, which is linked to wetland soil moisture; and
- including feedbacks from well-being to population dynamics and also capture the feedbacks of emigration on total net income through remittances.

As some of the components of the wetland hydrology were not included in the model due to data limitations the results of the hydrological effects of the simulated scenarios have to be considered with caution. There is also scope to extend the model by going beyond the two wetland services considered in this study (crop production and biomass harvesting) and include other provisioning and regulating services provided by the wetland.

Because of the limitations imposed by the structure of the ecological-economic model, it was not possible to consider some important wetland management strategies in the simulation analysis. In light of the evidence shown by the given results that wetlands are a key resource for the livelihood of the poor especially in managing the effects of climate variability on agriculture, it is important to identify local level sustainable wetland management practices that farmers can use with minimum effects on wetland ecosystem conditions. Therefore, instead of focusing on external drivers and macro policies, there is need to improve the ecological-economic model presented here and expand the simulation analysis to include local level wetland management scenarios. The scenarios would then include alternative wetland crop and livestock management practices, which enables the identification of management practices that the rural people can use to enhance their economic well-being with minimum impacts on wetland ecological conditions.

The ecological-economic model can also be further improved by integrating social and institutional aspects with the presently modelled environmental and economic systems. Last but not least, future research can also consider the spatial aspects into the dynamic analysis presented here by looking at the ecological and economic effects of the alternative management regimes beyond the local level, to be able to understand the full consequences (off-site effects) of these regimes at a broader scale.