

**AGROECOSYSTEM SUSTAINABILITY:
AN INTEGRATED MODELING APPROACH**

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Graduate Studies and Research in
Partial Fulfillment of the Requirements
for the Degree of Ph. D. in the
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University of Saskatchewan
Saskatoon

By

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of the requirements for the

DEGREE OF DOCTOR OF PHILOSOPHY

by

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Agroecosystem Sustainability: An Integrated Modeling Approach

The purpose of this study was to evaluate the sustainability of agroecosystems. The framework developed within this study is systems-based by making the dynamic linkages between the system components explicit.

The primary objective of the study was to develop a computer model, the Sustainable Agroecosystem Model (SAM), that dynamically integrates the economic and ecological components of an agroecosystem. The model was used to assess the sustainability of agroecosystems, defined by ecodistrict boundaries, in the Brown soil zone of southwestern Saskatchewan. The SAM was comprised of three components: (1) a soils model that simulated soil and crop growth parameters; (2) an economic model that simulated land use and cropping decisions; and (3) a habitat model that calculated habitat and biodiversity parameters. These components were largely self-standing models made up of important processes of the soil, economic and ecological sectors of the agroecosystem respectively. To simulate the co-evolutionary changes of the agroecosystem the component models were dynamically linked, based on a one year time step, through selected input and output parameters.

The output of the component models reflect elements of the natural and man-made capital stock of the target agroecosystems and were used as sustainability indicators. The concept of strong sustainability was adopted in the analysis such that changes in these indicators signal changes in the relative sustainability of the system.

The study focused on two types of simulations: (1) the relative sustainability of four ecodistricts was assessed using baseline simulations. This analysis highlighted the importance of biophysical constraints to the sustainability of an agroecosystem. These

simulations indicated that the development of production technologies and policy initiatives, targeting agroecosystem sustainability, should explicitly consider the regional biophysical constraints faced by farms; and (2) the relative sustainability of a single ecodistrict subjected to economic (carbon credit and carbon tax policies) and environmental (climate change) perturbations was evaluated. These simulations highlighted the difficulty in identifying a single policy that leads to a sustainable agroecosystem. In general, policies that resulted in improvement in some components of the capital stock caused degradation of other components. The identification of preferred policy, in terms of agroecosystem sustainability, requires a weighting of system effects based on societal preferences, ethical responsibilities, degradation thresholds and system co-evolution.

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ABSTRACT

The purpose of this study was to evaluate the sustainability of agroecosystems. The framework developed within this study is systems-based with the dynamic linkages between the system components explicit.

The primary objective of the study was to develop a computer model, the Sustainable Agroecosystem Model (SAM), that dynamically integrates the economic and ecological components of an agroecosystem. The model was used to assess the sustainability of agroecosystems, defined by ecodistrict boundaries, in the Brown soil zone of southwestern Saskatchewan. The SAM was comprised of three components: (1) a soils model that simulated soil and crop growth parameters; (2) an economic model that simulated land use and cropping decisions; and (3) a habitat model that calculated habitat and biodiversity parameters. These components were largely self-standing models comprised of important processes of the soil, economic and ecological sectors of the agroecosystem respectively. To simulate the co-evolutionary changes of the agroecosystem the component models were dynamically linked, based on a one year time step, through selected input and output parameters.

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CHAPTER 1

INTRODUCTION

1.1 BACKGROUND

Human managed social and economic systems are inextricably integrated with local and global ecosystems.¹ Economic systems depend on natural ecosystems for natural capital inputs. Natural capital consists of two separate categories: (1) non-renewable resources such as oil, coal, and minerals; and (2) renewable resources such as those provided by ecosystems. Environmental or ecological services such as the maintenance of the composition of the atmosphere, amelioration of climate, operation of the hydrological cycle including flood control and drinking water supply, waste assimilation, recycling of nutrients, generation of soils, pollination of crops, provision of food from the sea, maintenance of species, the genetic pool, scenery and recreational amenities and landscape aesthetics represent the flows that come out of natural capital stocks.

Agroecosystems represent a relatively unique type of system where the economic and ecological systems are very closely linked and often impossible to differentiate. Agroecosystems are distinguished from natural ecosystems by the dominant role of human management for specific marketable products. An important characteristic of these complex systems is the profound interdependence of the system components.²

When economic systems are small in scale relative to their ecosystem context the natural systems are capable of providing the required resources and services while

¹ In this study environmental systems or ecosystems are defined as systems where energy, matter and information flows are not wholly dependent on human activity. Many ecosystems can be manipulated and managed by humans but depend on natural energy, matter and informational flows to function.

² A discussion of what distinguishes a complex system is provided in Chapter two.

maintaining ecosystem function. The maintenance of ecosystem function ensures that the natural capital demanded by the economic system can be provided for the foreseeable future. However, since 1950 the human population has increased by 3.5 billion (58 percent), with a current growth rate of 88 million per year. This dramatically increasing population has developed economic systems that have reached scales that are now significant with respect to their environmental context. The demands for natural capital have become so great during this time period that the function or stability of ecosystems, and therefore the sustainability of the economic systems, is being compromised.

Few of the earth's ecosystems have been excluded from the pressure of increased demand for natural capital with forest, marine, freshwater, grassland and even arctic ecosystems being extensively degraded. Global species loss is estimated to be at a rate of 50,000 species per year, with 75 percent of bird species populations declining and 25 percent of the mammal species threatened with extinction (Abramovitz, 1997). The conversion, degradation, fragmentation, and simplification of ecosystems has been extensive. In many countries more than 50 percent of the land base has been converted from natural habitat to other, often degradative, uses. As ecosystems become more simplified they decrease in resilience, thereby becoming more vulnerable to collapse in the face of an environmental shock.³

Increases in global economic activity have increased the emission of gases (greenhouse gases) into the atmosphere that can alter global climate patterns.⁴ Atmospheric concentrations of carbon dioxide are at the highest levels in 150,000 years

³ Resilience has been defined as the ability of a system to maintain structure and function after a disturbance or a shock (Holling, 1973). Resilience can be considered a property of any natural or economic system.

⁴ The major greenhouse gases are carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O). The atmospheric levels of all of the greenhouse gases are increasing. Higher atmospheric concentrations of these greenhouse gases can cause changes in climate patterns by either changing the reflection or absorption of solar radiation, or the emission and absorption of terrestrial radiation. It is predicted that climate change will result in increases in average temperatures with greater changes occurring in higher latitudes, changes in precipitation patterns with increased probability of extreme events such as droughts and floods.

and continue to increase. “The world is projected to face a rate of climate change in the next several decades that exceeds natural rates by a factor of 10” (Flavin, 1997).

Climate change in the context of the above discussed ecosystem changes can result in even more serious consequences for the sustainability of ecological and economic systems. Flavin (1997) highlights two aspects of climate change that make it particularly dangerous to ecosystems: (1) the fragmentation and disruption of most natural landscapes will greatly constrain the capacity of the ecosystems to respond to climate change influenced shock; and (2) the overlap with other forms of ecological degradation greatly increases the odds of abrupt and dangerous changes to ecosystem function.

In recent years recognition of global scale environmental degradation has inspired the formalization of the concept of sustainable development. Sustainable development was brought into prominence by The World Commission on Environment and Development report *Our Common Future* (WCED, 1987). The widely cited WCED definition of sustainability “...sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs” has provided a contested but significant foundation for the advancement of the concept. Sustainable development represents an attempt to not only recognize the physical limits to economic activity but also an investigation into how, and if, socio-economic objectives can be attained while maintaining the integrity of ecosystems (Faucheux, 1996).

The ecosystems which serve as the source of natural capital for food production systems have come under particular pressure, with the combination of unprecedented demand for food and decreasing capabilities to meet this demand. Prior to the beginning of this century almost all increases in food production were attained by bringing new land into production. However, global agricultural land area is decreasing due to desertification, urban sprawl and soil degradation and as a result almost all future increases in food production will come from increased output per hectare (Ruttan, 1994). The options for increased agricultural production intensity are limited. There is a growing scarcity of fresh water for irrigation in many countries. For example, 21

percent of U.S. irrigated cropland is being watered by drawing down underground aquifers (Brown, 1997). From 1950 to 1989 fertilizer use increased 10 fold. Since 1989 fertilizer usage has decreased due to the fact that fertilizer is no longer the primary limiting input factor in crop growth in many regions (Brown, 1997). Any increases in agricultural production will likely come about by further drawing down the natural capital stock. In degrading the essential natural capital the agricultural system will at best be limited in its future capacity, and at worst be incapable of being maintained into the future.

The agricultural systems of the Canadian prairies are subject to intense production pressure. The degradation of these agroecosystems is being revealed through changes in the function of these systems. Soil degradation, the pollution of ground and surface water, destruction of wetlands and other wildlife habitat and the loss of biodiversity have prompted questions about the capability of the prairies to sustain current land use patterns (Anderson, 1993). In addition, forced farm foreclosures, decreased rural populations, increases in violence, alcohol and drug abuse are evidence of degradation of the social and institutional components of agroecosystems.(Anderson, 1993).

1.2 PROBLEM

The increased intensity of agricultural production techniques in recent years has resulted in degradation of the natural and man-made capital stock of Canadian agroecosystems. This degradation of the capital stock has brought into question the long-term sustainability of Canadian agroecosystems. The importance of agroecosystems to social, and economic systems at local and global scales has stimulated a demand for analysis into the sustainability of agroecosystems.

Agroecosystems are a complex system and as such are made up of a variety of components and the feedback processes that exist between these interdependent system components. The flows of information energy and matter are what constitutes an agroecosystem and highlights the interdependence of the system components.

Therefore, an analysis of agroecosystem sustainability must explicitly recognize the nature of these linkages and the effect they have on system change over time.

The models developed to address issues of system sustainability have generally not incorporated the linkages between system components, or have included the linkages in a static form. Such models are not capable of capturing the feedback processes that make up an agroecosystem. The problem to be addressed in this study is to develop a framework that can be used to analyze the sustainability of an agroecosystem in a dynamic way.

1.3 OBJECTIVES

The primary objective of this study is to develop a computer model that dynamically integrates the economic and ecological components of an agroecosystem. Using model output, the relative sustainability of landscapes subjected to a variety of policy and climate shocks can be simulated and assessed. The specific study objectives are:

1. to identify important energy, material and informational linkages within and between the economic and ecological components of agroecosystems in the Prairie region of Canada.
2. to create an integrated, dynamic model that will simulate changes in the economic and ecological components of the agroecosystems.
3. to assess the effect of different policy and climate shocks on the relative sustainability of simulated agroecosystems.

1.4 SCOPE OF THE STUDY

This study is a part of the Prairie Ecosystem Study (PECOS) based at the University of Saskatchewan and the University of Regina. The PECOS project is a community-based, interdisciplinary research project focusing on the sustainability of the semi-arid agroecosystems of south western Saskatchewan. The main objectives of the PECOS project are to evaluate the human impacts on the agroecosystem, and to

investigate land use practices that may lead to regional sustainability of the economic, environmental and social systems. The target agroecosystems of this study lie within the PECOS focus area (Anderson, 1993).

The boundaries of the study agroecosystems are defined by biophysical characteristics such as soil texture, relief and vegetation. The focus of the economic component of the model is an average farm with annual crop production as the primary source of revenues. The results produced by the thesis model, hereafter referred to as the Sustainable Agroecosystem Model (SAM), provide insight into the relative sustainability of agroecosystems. Model simulation results should not be considered appropriate for interpreting absolute sustainability of an agroecosystem, or comparing systems with very different climatic and biophysical characteristics.

1.5 ORGANIZATION OF THESIS

This thesis provides an integrated systems-based analysis of the economic decisions and ecological dynamics of an agroecosystem with the objective of assessing relative sustainability. Chapter Two presents an overview of the relevant literature focusing on developing a systems-based modeling approach. Chapter Three develops a conceptual framework for the model with a detailed description of the theoretical foundation of the modeling components. Chapter Four provides the analytical details used to tailor the conceptual framework to the study objectives. Chapter Five discusses the background and empirical details of the policy and climate change shocks imposed on the model for analysis. Chapter Six discusses baseline simulation output targeting four distinct landscapes in the study area. Chapter Seven presents and discusses the policy and climate change scenarios in the context of the sustainability of a single target landscape. Finally, Chapter Eight provides a summary of the insights gained from the model simulations and highlights areas of further research.

CHAPTER 2

LITERATURE REVIEW

2.1 INTRODUCTION

Agriculture, as a human-managed production system depends on the dynamic flow of information, energy and matter between economic, ecological and social sub-systems. There is an extensive literature focusing on the economic, ecological and social components individually, and in various combinations, aimed at addressing agroecosystem sustainability. The purpose of this chapter is to examine the research methodologies and findings of this literature pertaining to sustainability in general, and sustainable agriculture specifically. The insights and shortcomings of this literature, in the context of the objectives of the present study, are highlighted. This chapter is organized around the notion of system sustainability and the utility of a multi or interdisciplinary research approach. The discussion initially focuses on the characterization of complex systems (Section 2.2). Following this the focus is on how the literature has defined the concept of sustainability (Section 2.3) and soil sustainability (Section 2.4), followed by a broad review of studies that have used single discipline models to address issues of land use and environmental quality (Section 2.5). The next section (Section 2.6) reviews models that explicitly incorporate more than one discipline. A summary of the significant findings of the literature review, as relevant to the present study, is presented as the final section of this chapter (Section 2.7).

2.2 SYSTEMS ANALYSIS

A system can be interpreted as almost any subdivision of the universe. Costanza et al. (1993) define systems as groups of interacting, interdependent parts linked together by exchanges of energy, matter and information. In order to better understand systems, models have traditionally been developed using boundaries that minimize the interactions between the target system and the rest of the universe. Although this simplification facilitates the development of tractable models, an assumption of insignificant or linear interactions between the target system and its surroundings is required. Complex systems violate these assumptions and as such can not be interpreted using this traditional modeling approach. Complex systems are characterized by strong, often non-linear, interactions between the components, complex feedback loops that obscure the distinction between cause and effect, and significant time and space lags, discontinuities, thresholds and limits (Costanza et al., 1993). Alternatively, complex systems are defined as those systems that require fine details to be linked to large outcomes (Ahl and Allen, 1996).⁵ Therefore, analysis of complex systems demands a framework that addresses multiple levels of analysis simultaneously.

For any study of complex systems it is worthwhile to note that the target system is usually a research described subset of a larger system. “Complexity does not exist independently of an observer’s questions. Instead, complexity is the product of asking questions in a certain way” (Ahl and Allen, 1996). The present study focuses on a complex system that is a researcher defined subdivision of agroecosystems. Costanza et al., (1993) state that “ecological and economic systems both independently exhibit these characteristics of complex systems. Taken together, linked ecological economic systems are devilishly complex.”. A description of the relationships captured by SAM that facilitate the simulation of the non-linear relationships and long-term feedback effects of a complex system is provided in Section 3.9.

⁵ This characterization of complex systems is consistent with the concept of co-evolution which is discussed in more detail in later sections of this thesis.

2.3 SUSTAINABILITY

In the last 10 to 15 years a sizable literature focusing on sustainability issues has developed. Throughout this literature no definition of sustainable development and its derivative concepts has been universally accepted by each discipline, let alone across disciplines (Hansen, 1996). However, there does seem to be consensus that “sustainable development” is a concept based on intergenerational equity (Batie, 1989; Pearce et al., 1989).⁶ Another characteristic of sustainability is the importance of interactions between components of the system. For example, Lynam and Herdt (1989) state that sustainable agriculture must be defined with respect to systems, rather than inputs or crops, since crop varieties and inputs produce nothing in isolation. Only when combined as components of a system do they produce output. Norgaard (1988) identifies the following fundamental issues concerning sustainable development:

If sustainable development is to be achieved, we will have to devise institutions at all levels of government to reallocate the use of stock resources towards the future, curb the pace and disruption of global climatic changes, reverse the accumulation of toxins in the environment and slow the loss of biological diversity.

Faucheux et al. (1996) explain that sustainable development dictates a reorientation of economic analysis to consider intra and intergenerational equity, the very long term, irreversibility of ecological change, uncertainty and complexity, and technological change. The World Wildlife Fund developed a definition for sustainable development that summarizes the philosophy underlying most definitions:

Sustainable development is people-centered in that it aims to improve the quality of human life, and it is conservation-based in that it is conditioned by the need to respect nature’s ability to provide resources and life-supporting services. In this perspective, sustainable development means improving the quality of human life while living within the carrying capacity of supporting ecosystems. (Reed, 1996)

⁶ Intergenerational equity is defined as the normative principle that current generations must not compromise the ability of future generations to meet their needs and encompasses the question of how much human capital (knowledge), human-made capital (physical capital) and natural capital (environmental resources and services) to pass on to future generations.

This concept of living within the carrying capacity of the supporting ecosystem(s) is particularly germane when considering the sustainability of agricultural systems that are so closely tied to the resource base.

Costanza (1996) argues that the search for an adequate definition of sustainability is misdirected since sustainability is not a definitional problem but a prediction problem and a definition will fail to encompass the many temporal and spatial scales over which sustainability must apply. Costanza states that a sustainable system is simply one that survives for some specified time period. "Thus, what usually pass for definitions of sustainability are actually predictions of what set of conditions will actually lead to a sustainable system" (Costanza, 1996). Sustainability is more appropriately characterized as a long-term goal. The research problem then becomes the prediction of policies and conditions that will lead to this goal.

Within the literature the terms health, integrity and sustainability are often used to identify what appear to be similar concepts. Ecosystem health has been described as an approach to analyze and manage an ecosystem using terms and procedures from medical science. This approach seeks to identify indicators of ecosystem health, to assess health based on these criteria and to provide specific management and policy recommendations (Okey, 1996). Costanza (1992) defines health as a "comprehensive, multiscale, dynamic, hierarchical measure of system resilience, organization and vigor." "These elements are embodied in the term sustainability, which implies the systems' ability to maintain its structure (organization) and function (vigor) over time in the face of stress (resilience)" (Costanza, 1992). The strong relationship between health and sustainability is revealed by Haskell et al. (1992) who state that an ecological system is healthy if it is stable and sustainable. Some authors use the terms health and integrity interchangeably. Okey (1996) states that there is no clear definition of ecosystem integrity. However, a range of functional, structural and aesthetic attributes are attributed to an ecosystem with integrity. Within the literature health is considered a component of integrity by some (Kay, 1993) while others consider integrity an element of health (Rapport, 1992). In general, health, integrity and sustainability describe

desired states and/or goals of the environment using somewhat subjective standards. For the purposes of this thesis the term sustainability will be used.

2.3.1 Weak and Strong Sustainability

Economics has contributed some useful theory to the sustainability literature. Economic sustainability has been described using either a weak sustainability or a strong sustainability criteria. Weak sustainability is based on the assumption that there is substitutability between natural and man-made capital. A system is weakly sustainable when the total capital stock (man-made plus natural capital) is non-decreasing.⁷ This is consistent with the theory of sustainability developed by Hartwick (1977) and Solow (1974) who showed that given sufficient substitutability between reproducible and exhaustible stocks, an investment rule can be developed that will hold consumption constant over time. An economy is weakly sustainable even if it is drawing down its stock of natural capital provided it creates enough man-made capital to compensate for the loss of natural capital (Gowdy and O'Hara, 1997). In order to compensate for natural capital stock degradation with man-made capital stock investment, the value of the degraded natural capital stock must be known. Therefore one of the primary limitations of the weak sustainability criteria is the difficulty or impossibility of assigning a meaningful economic value to many components of natural capital. Ehrlich and Ehrlich (1992) argue that the full contribution of species and processes to the aggregate life-support services provided by ecosystems has not been captured in economic values. In fact these services are likely not measurable in economic value terms. The weak sustainability criteria imposes the risk that as environmental degradation occurs, some of the critical ecosystems services will be systematically eroded thereby decreasing the resilience of this system (Turner et al., 1994).

⁷ A definition for input substitutes can be based on a production process that requires two inputs, x and y, to produce output z. Inputs x and y are considered substitutes if when the quantity of x decreases, z can be maintained by increasing the quantity of y used in the production process.

Weak sustainability has been further criticized for its assumption that natural and man-made capital are substitutes (Common and Perring, 1992). Daly (1994) argues that man-made and natural capital are at most only marginally substitutes:

Man-made capital (along with labor) is an agent of transformation of the resource flow from raw material inputs into product outputs. The natural resource flow (and the natural capital stock that generates it) are the *material cause* of production; the capital stock that transforms the raw material inputs into product outputs is the *efficient cause* of production. One cannot substitute efficient cause for material cause - one cannot build the same wooden house with half the timber no matter how many saws and carpenters one tries to substitute.

Based on this argument a fundamental assumption of the weak sustainability concept is violated.

Strong sustainability is based on the assumption that natural and man-made capital are complements. Strong sustainability requires the maintenance of the total capital stock (man-made and natural) such that each component is maintained intact separately. Daly (1994) states that if one believes that natural and man-made capital are complements, then the complements must be maintained intact (separately or jointly in fixed proportions), because the productivity of one depends on the availability of the other. Daly goes on to state that if natural and man-made capital are complements then it follows that the capital component that is in shortest supply will be the limiting factor. The optimal rate at which the capital is exploited is equivalent to the maximum sustainable yield of that component of the capital stock.⁸ Crabbe (1997) notes that the ability of the environment to assimilate waste products and the long-term integrity of the environmental support system should also be considered within the sustainable yield framework. Based on this strong sustainability framework the sustainability of an agroecosystem can be assessed by examining the state or health of the various components of the capital stock to determine whether the rate of exploitation is within the sustainable yield of that stock.

⁸ Sustainable yield can be defined for renewable resources as keeping the annual removal from a resource equal to the annual growth increment and is equivalent to maintaining the capital stock intact.

The literature in the above discussion presents a range of critical assumptions that underlie the sustainability framework. For the purposes of this study the sustainability framework encompasses: (a) a predictive capacity to facilitate the assessment of intergenerational equity in terms of the availability of the capital stock for future generations; and (b) a multi or inter-disciplinary structure to capture the sustainability of the economic, environmental and social components of the system and how these interactions affect the relevant capital components. Such a framework facilitates an assessment of the condition of the system by monitoring the changes in the capital stock to determine if it is functioning within the carrying capacity of the system components and as such is on the path toward the long term goal of sustainability.

2.4 SOIL SUSTAINABILITY AND LAND USE

The sustainability of an agroecosystem is strongly linked to the sustainable use of the soil component of its natural capital stock. The soil resource of an agroecosystem is comprised of many components and functions. Van Kooten (1993) describes four economic aspects or characteristics of the soil resource:

1. **Perdurable Matrix** - The permanent or indestructible component of the soil that is determined by location, climate, subsoil, drainage, inexhaustible nutrients, macro-relief etc. The non-depletable nature of this component of the soil means that it is not considered in soil conservation questions.
2. **Conservable Flow** - The conservable flow requires some investment to maintain in its original state but is worthwhile to conserve from an economic perspective. The present value of future returns from its conservation is greater than the present value of conservation costs. In addition, degradation of the conservable flow is economically irreversible. An important example of this aspect of the soil is soil organic matter.
3. **Revolving Fund** - "That element of virgin soil fertility that is not economical to conserve but is economical to replace or renew with materials imported from off-site" (van Kooten, 1993). Examples of revolving fund are soil

nutrients, such as nitrogen and phosphorous, that can be replaced with fertilizer.

4. Expendable Surplus - The expendable surplus is often very large, though finite, and as such is economical to deplete but not economical to replace. When the expendable surplus stock becomes very small it will be treated as a conservable flow or revolving fund. Examples of expendable surplus are excess depths of the soil A horizon, or large excess stocks of plant available soil nitrogen.

It is important to note that the soil organic matter stock, when first converted to crop production, has characteristics of the conservable flow, revolving fund and expendable surplus (van Kooten and Furtan, 1987). The soil nitrogen and phosphorous stocks in virgin soil can be categorized as expendable surplus but become revolving fund as the soil degrades. The water holding capacity of soil is partially dependent on the stock of organic matter, therefore a decrease in organic matter that results in a lower water holding capacity represents a loss of conservable flow (van Kooten and Furtan, 1987).

The loss of the conservable flow component of agricultural soils is the main focus of soil sustainability research and within the economics literature is identified primarily as the user cost of soil degradation or soil erosion. "The user cost of soil erosion is the impact of lost soil on future profits via the level of stock; that is, it is the present value of future revenues that are lost if we use a unit of soil today" (van Kooten, 1993). Based on this concept of user cost the farmer's management decisions are influenced by the effect of soil quality on profits. McConnell (1983) shows that soil loss will be incurred until the value of returns obtained from additional soil loss equals the implicit cost of using the soil. The cost of soil loss in foregone future profits is comprised of: (1) the change in soil productivity and concomitant change in profits within the planning horizon; and (2) the change in land price at the end of the planning horizon caused by having lower quality soil. Soil is an asset, as a component of the natural capital stock, and the returns for holding the asset are made up of capital gains and contributions to current profits (McConnell, 1983). This theory indicates that farmers will be willing to adopt management practices that provide smaller contributions

to current profits but are less degrading to the conservable flow and therefore will provide greater capital gains in the long run than some alternative management. Only when the farmer has an infinite planning horizon (dictated by land tenure as discussed later in this section) does user cost exclude the capital gains component and only consider the contributions to current profits.

The model discussed above assumes that farmers are fully aware of the effect of soil quality on current and future production. Van Kooten and Furtan state that farmers are often unable to, or don't distinguish between the conservable flow and revolving fund components of the soil due to: (1) a lack of knowledge regarding the relationship between soil sustainability and management practices; (2) technological advances compensating for soil degradation such that yields are non-decreasing with lower soil quality; (3) fluctuations in interest rates and output prices resulting in an increase in the discount rate such that resources are allocated to the current generation; and (4) a non-linear relationship between soil quality and yield under constant production technology (van Kooten et al., 1989).

A second assumption of the model is that land prices reflect the capitalization of future rents, which are dependent on soil quality (McConnell, 1983). Burt (1986) found a strong correlation between the price of Illinois annual crop land and land rents. However, Falk (1988) states that although farmland price and rent movement are highly correlated, price movements are much more volatile than rent movements. In contrast, Clark et al. (1992) showed that land price and land rent are not highly correlated implying that land prices do not reflect the discounted sum of the expected value of future rents. This result indicates that even if the conservation of soil conservable flow could be ascribed to greater production and higher future rents, the investment into soil quality would not be reflected in higher capital gains to the land asset.

A further assumption required for farmers to incorporate the user cost of soil degradation into their production decisions is that the land tenure facilitates the farmer realizing the benefits of maintaining the conservable flow component of the soil. The length of the planning horizon of the farmer, as dictated by land tenure, will influence the user cost of soil degradation. McConnell (1983) identifies three tenure

arrangements: (1) owned family farms; (2) rented family farms; and (3) corporate farms. Corporate farms are assumed to have very long planning horizons and therefore have higher current user cost. The planning horizon of an owned family farm is adopted by the head of the household. If several generations of farm owners are assumed, and/or asset market are assumed to work smoothly, the user cost of soil degradation to the owned family farm will be the same as the corporate farm (McConnell, 1983). In contrast, McConnell (1983) states that “the current user cost is lower for renters because farm resale value is unimportant. The only reason for renters to conserve soil is for its productive capacity. If soil depth does not affect production, the renter will ignore soil loss”.

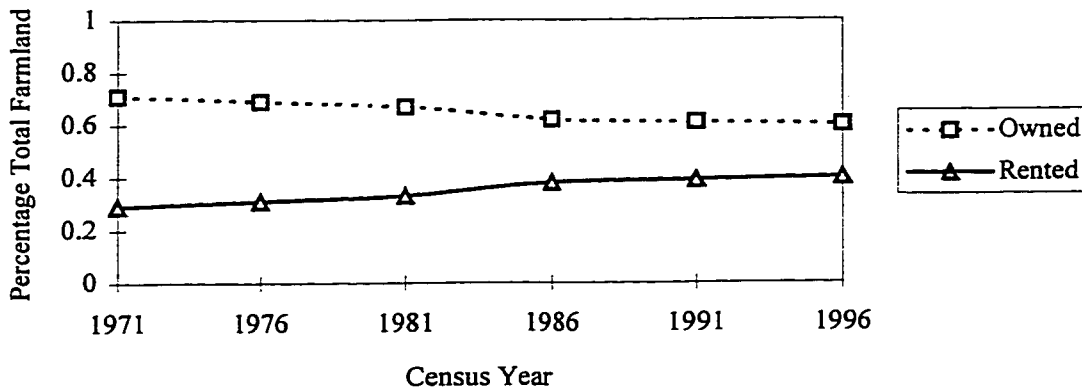
The present study assumes that farmers do not incorporate the user cost of soil degradation in their production decisions. The above discussion indicates that this assumption is not an unreasonable one. Van Kooten and Furtan (1987) state that given the rents that can be captured by exploiting the expendable surplus and conservable flow, the short-term debt obligations of many farmers and the presence of imperfect land markets that make it unlikely for soil depletion to be adequately accounted for in land values, farmers may view it as beneficial to draw down the soil capital. Within Saskatchewan an increasing area of farmland is rented (Table 2.1) which implies a low current user cost on approximately 40 percent of the farmland.

A brief discussion of studies that have attempted to incorporate user cost in farm production decisions will be presented in later sections of this chapter.

2.5 DISCIPLINARY MODELS

Models are developed to decrease the uncertainty of responses or reactions observed in a target system. Models take many forms from conceptual representations to functioning operational models designed to simulate the important relationships within the defined system. Costanza et al. (1993) describe three general criteria that can be used to classify models: (1) realism - simulating system behaviour in a qualitatively realistic way; (2) precision - simulating behaviour in a quantitatively precise way; (3)

Table 2.1. Land tenure in Saskatchewan as percentage of total farmland.



(Statistics Canada, 1996)

generality - representing a broad range of systems behaviours with the same model. No single model can maximize these three criteria. The prioritization of the above criteria depends on the objectives of the research question for which the model has been tailored.

To assess the sustainability of a system models that focus on the relevant aspects of the system are required. The present study identifies the economic system (economic sustainability) and the soil, crop, and ecological systems (environmental sustainability) as the relevant components to evaluate agroecosystem sustainability. This section will review some of the previously developed disciplinary models that focus on either the economic or environmental components of the agricultural system. These models have been developed with the primary objective of analyzing the effect of a change in policy on the allocation of resources, and changes in environmental quality. Whether or not the movement of the system towards sustainability is an objective of the model, these models assist in the selection of policy that is most desirable under the model assumptions. The discussion will be organized according to the agroecosystem components that are the primary focus of the present study. The following sections will focus on: (1) economic/land use models; (2) soil quality/crop production models; and (3) habitat/biodiversity models.

2.5.1 Economic/Land-Use Models

An important component of a sustainable agriculture system is economic sustainability. Gray (1991) states that “Sustainable agriculture is, at least in part, a maintenance of the flow of income from agricultural production”. Farms, or agricultural firm, in this study, are defined as economic units of production. Farms defined in this narrow way are generally assumed to be perfect competitors, and as such are price takers in both inputs and outputs. Another characteristic of most agricultural production is that inputs vary proportionally with the land input. That is, how a firm allocates its land input has a strong effect on the allocation of other production inputs (Shumway et al. 1984). For example, a farm that allocates a large amount of land to wheat production has very different non-land input requirements from a farm that allocates large amounts of land to pea production. Therefore how a farm uses its land input is a critical factor in its production decisions.

Models designed to reflect production decisions and the allocation of the land input are referred to as: (1) supply response; (2) acreage response; or (3) area response models (Clark and Klein, 1992; Schmitz, 1968). Area and acreage response models capture the changes in the quantity of land allocated to a particular output while supply response models capture the changes in the quantity of output (grain) produced. For the purpose of this study the area response models are the most appropriate. The majority of area response models are comprised of some form of regression analysis using time series data. While these econometric models have empirical and statistical validity, the application of these models is limited by an absence of detailed disaggregated regional data. In addition, the econometric models tend to be more descriptive than predictive and as a result may not be appropriate for analysis of the future sustainability of a production system. The consideration of future time periods implies that a predictive area response or simulation modeling framework may be more appropriate.

One form of predictive model that has been used in a number of studies incorporates linear programming to optimize an objective function for a group of homogeneous firms (Hazell and Norton, 1986). This approach has many inherent advantages including minimal data requirements and a constraint structure that is well

suited to characterizing resource, environmental, or policy constraints. In addition, the assumption of Leontief production technology inherent in most programming models has an intrinsic appeal of input determinism when modeling farm production (Howitt, 1995).⁹ This type of predictive model has been used extensively to analyze the effect of changes in existing or new policy on agricultural production systems. However, since few data points are used the data are primarily average or representative values for such parameters as yield and input requirements. Applying the results from this type of model means that marginal behavioural reactions to policy changes are interpreted based on average data (Howitt, 1995; Just, 1993). Only when the policy range is small enough to justify linear technologies can it be assumed that average and marginal conditions are equivalent. These linearities also cause significant calibration problems in traditional mathematical programming models.

The Positive Mathematical Programming (PMP) approach maintains Leontief technology but imposes decreasing returns to size by including a non-linear cost term in the objective function. Decreasing returns to size leads to an upward sloping supply curve that is a consequence of non-homogeneity of land resources, agronomic constraints, technical constraints, and risk aversion (Howitt, 1995). A PMP style model was constructed within the Canadian Regional Agricultural Model (CRAM) that is described in detail by Horner et al (1992).

A model framework that has similar characteristics to the traditional mathematical programming framework is the fixed proportions model (Gardner, 1987). The fixed proportions model has disadvantages similar to the linear programming models and in fact has at its core the characteristics of Leontief production relationships. Although the fixed proportions model may not be capable of providing results that are quantitatively precise, the results can be considered qualitatively realistic. The fixed proportion model will be discussed in detail in Chapter Three.

⁹ The Leontief production relationship leads to a fixed production relationship which implies that the elasticity of substitution between all input pairs is zero.

2.5.2 Soil Quality/Crop Production Models

An important component of the natural capital of an agricultural system is the soil. Agricultural sustainability is strongly linked to the health of the soil.¹⁰ This has been forcefully stressed in some landmark documents. “Sustainable agriculture cannot be based on methods that mine and deplete the soil” (WCED, 1987); “Soil degradation is the most serious threat to the agricultural industry in the long term” (Sparrow, 1995). Therefore, a model that effectively predicts changes in soil quality over time and under different management strategies is an important component in the analysis of agricultural sustainability. In recent years a number of such soil models have been developed.

The CENTURY model was developed to simulate long term (10-1000yr) patterns in soil organic matter dynamics, plant production, and nutrient cycling (nitrogen (N), phosphorous (P), and sulfur (S)) (Parton and Rasmussen, 1994). The CENTURY model uses a monthly time step and monthly average maximum air temperature, monthly precipitation, soil texture, dead plant material nutrient and structural characteristics, and atmospheric and soil inputs of N as driving variables. The model simulates decomposition rates and the associated carbon and N flows as a function of the variables listed above. Structurally the model is composed of three interdependent submodels: (1) soil Carbon model - calculates the dynamics of soil C over time; (2) Soil N model - calculates the dynamics of soil N over time; and (3) crop/plant production model - calculates above ground plant production based on water, carbon and nitrogen availability. A more detailed description of the submodels and the input and output characteristics of the CENTURY model is provided in Parton et al., (1987) and Parton et al., (1988).

Another important model that focuses on soil system dynamics is EPIC (Erosion/Productivity Impact Calculator). EPIC was developed to predict or estimate the long-term relationship between soil erosion and soil productivity, calculated at a daily time step (Williams et al., 1990). The model includes two parts:

¹⁰ Soil health is often substituted for soil quality and is defined as “the soil’s fitness to support crop growth without becoming degraded or otherwise harming the environment” (Acton and Gregorich, 1995)

1. Physical procedures that simulate such characteristics as weather, hydrology, wind and water erosion, nutrient cycling, crop growth, soil temperature, tillage, and plant environmental control.
2. Economic procedures to assess erosion costs and determine optimal management strategies.

These procedures, or submodels, are linked interactively. EPIC uses procedures to calculate the relationship between erosion and productivity that involves plotting the values of an erosion/productivity index on the y axis against corresponding values for erosion on the x axis.¹¹ For a detailed description of the input requirements and output characteristics refer to Williams et al. (1990).

The CENTURY and EPIC models have been the most commonly used soil process models, however there have been a number of other physical process models developed to simulate the soil and crop components of the agricultural system. Table 2.2 provides references for detailed discussion of these models.

EPIC and CENTURY provide detailed and often very robust (quantitatively precise) results, however the input requirements are substantial and the complicated nature of the models decrease the flexibility of the procedures for wide spread application. In addition, it has been shown that these models are poorly validated, particularly for western Canada (Greer and Anderson., 1991; Beckie and Moulin, 1992). In addition, the complexity of the models, and the modeling environment in which CENTURY and EPIC were created make it difficult to incorporate these models into an integrated systems framework that requires model components to be linked in a dynamic way.

A model that is less quantitatively precise and more qualitatively realistic, using similar procedures to the CENTURY model, is the Simulator of Productivity Lost by Erosion (SimPLE) model (Greer and Schoenau, 1992; Greer et al., 1992). The SimPLE model was designed to simulate spring wheat yields incorporating the essential

¹¹ The erosion/productivity index (EPI) is defined in two distinct procedures: (1) the ratio of annual crop yield from an eroded field to the annual yield from a noneroded field; (2) the ratio of mean crop yield for an eroded soil profile to the mean crop yield for the soil profile at the start of an EPIC simulation (Williams et al, 1990).

Table 2.2. Selected physical process environmental models.

Model Name	Focus	Reference
CERES - CEREAL	Crop Growth	Singh et al. (1989)
CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems)	Chemical and sediment fate and transport for surface water applications	Knisel (1980)
GLEAMS (Groundwater Loading Effects of Agricultural Management Systems)	Chemical and sediment fate and transport for groundwater applications	Leonard et al. (1987)
SWRRB (Simulator for Water Resources in Rural Basins)	Crop growth, hydrology, runoff	Arnold et al. (1990)

relationships between topsoil erosion and productivity loss in Chernozemic soils.¹² The key relationships in the SIMPLE model describe: “(1) how plants create yields from water, N and P, (2) how the soil provides these nutrients, and (3) how erosion impacts on the supply of each nutrient (Greer et al. 1992). Details of the SIMPLE model will be presented in Chapter Three.

2.5.3 Habitat/Biodiversity Models

The fields of biology and ecology have increasingly depended on mathematical models to analyze the processes and reactions within ecological systems. Jorgensen (1994) states:

The application of models in ecology is almost obligatory, if we want to understand the function of such a complex system as an ecosystem. It is simply not possible to survey the many components and their reactions in an ecosystem without the use of a model as a synthesis tool. The reactions of the system might not necessarily be the sum of all the individual reactions; this implies that the properties of the ecosystem as a system cannot be revealed without the use of a model of the entire system.

¹² Chernozemic soils are defined as an order of soils developed under cool, subarid-to-subhumid grasslands, characterized by a mineral surface horizon darkened by accumulating organic matter (Acton and Gregorich, 1995).

Ecological models have been developed for a wide range of ecological systems and system functions. At least one published journal, “Ecological Modelling”, is dedicated exclusively to ecological models. This review will focus on the more important models targeting the temporal and spatial dynamics of biodiversity and habitat, with particular focus on the agricultural landscape.

Biodiversity is a component of the natural capital stock of an agroecosystem and as such biodiversity preservation is regarded as an essential component of a sustainability strategy. However, like sustainability, a widely accepted definition of biodiversity has not yet been developed.¹³ A commonly used definition of biodiversity is “the variety and variability among living organisms and the ecological complexes that they occur” (OTA, 1987). This definition highlights three levels of diversity: (1) ecosystem diversity; (2) species diversity; and (3) genetic diversity. However, Noss (1990) argues that most definitions of biodiversity, the OTA definition included, fail to mention processes such as interspecific interactions, natural disturbances and nutrient cycles. Noss suggests that the three primary attributes of biodiversity - composition, structure and function - should be used to develop an understanding that biodiversity is an end in itself. Measurable indicators of these attributes can be used to assess the status of biodiversity over time.

The definition of biodiversity discussed above indicates that any model that addresses biodiversity will, by necessity, include more than a single level of biodiversity and will need to function at a landscape scale. Rosenzweig (1995) states “You will find more species if you sample a larger area. That rule has more evidence to support it than any other about species diversity.” A simple relationship that links habitat with species numbers or biodiversity is the species-area curve:

$$S = cA^z \quad (2.1)$$

where:

S - the number of species
 A - area of the sample

¹³ See DeLong (1996) for an exhaustive review of definitions of biodiversity that includes a categorization of 85 separate definitions gleaned from the literature.

c and z - constants that are determined empirically

The constants c and z describe the rate at which the number of species change with a change in area. The value of z typically falls within the range of 0.15 and 0.35 for most geographical areas (Rosenzweig, 1995). While the species-area curve relationship has received considerable attention in the literature, and has been applied to a broad range of ecosystems, it has also been criticized. Connor and McCoy (1979) argue that the parameters used to describe the species/area curve have no real theoretical significance. These authors show that the restriction of the z constant to a limited range was a statistical artifact and that none of the theories about the cause of the species/area relationship made any unique predictions about the shape of the curve. However, the literature is consistent in assigning a strong relationship between habitat area and biodiversity.

Another theoretical construct that links habitat to plant and animal populations is the 'source-sink' framework. In general, a source habitat is one in which birth rates are greater than death rates. A sink habitat is one in which birth rates are less than death rates. This framework explicitly accounts for the heterogeneity of a landscape by focusing on the differences in birth and death rates that occur in different habitats. As a result, this model seems to be appropriate for agroecosystem applications. Pulliam (1988) states that,

“...for many populations, a large fraction of the individuals may regularly occur in “sink” habitats,...nevertheless, populations may persist in such habitats being locally maintained by continued immigration from more-productive “source” areas nearby.”

If the surplus population from the source is large and the per capita deficit from the sink is small it is possible for only a small fraction of the total population to occur in areas where local reproduction is sufficient to compensate for local mortality. This concept is demonstrated in Figure 2.1. A management implication of this theory is that two strategies are possible and likely necessary to preserve biodiversity within an agroecosystem. One strategy is conservation of adequate source habitat and maximization of the quality of this habitat to maximize the potential reproductive

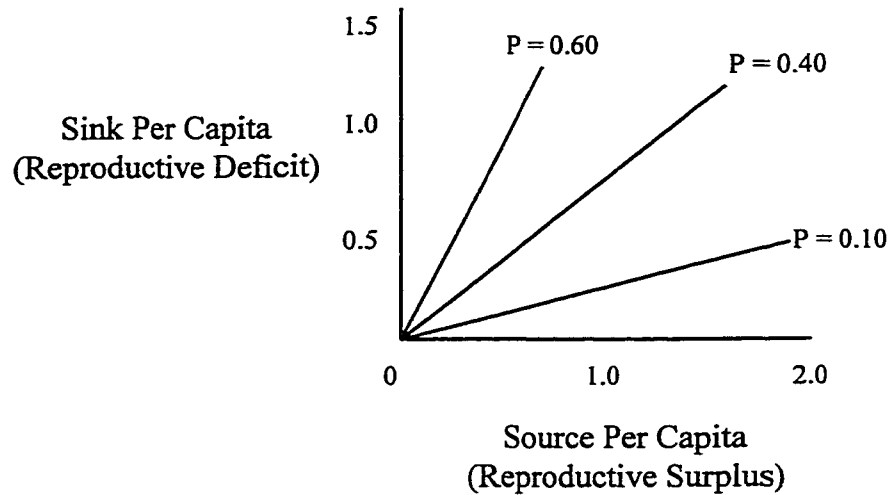


Figure 2.1. Source-Sink. The equilibrium proportion (p) of the population in the source habitat depends on both the per capita surplus in the source and the per capita deficit in the sink. A large proportion of the population may occur in the sink habitat if the source surplus is large and the sink deficit is small (Pulliam, 1988).

surplus within the landscape. The complimentary strategy is to adjust management to minimize the reproductive deficit in the sink habitat.

Although the relationship between habitat area and biodiversity is important, a number of other factors have been identified as explanatory variables for biodiversity.

1. latitudinal gradients - biodiversity generally declines as you move away from the equator, north or south.
2. altitudinal gradients - biodiversity generally declines as altitude increases.
3. productivity - biodiversity has been found to be both positively and negatively correlated to productivity (Huston, 1994).¹⁴
4. spatial heterogeneity - the number of species found in an area is strongly correlated to the spatial heterogeneity of that area (Huston, 1994).

¹⁴ Rosenzweig (1995) defines productivity as the rate at which energy flows in an ecosystem or the amount of solar energy that is captured by plants and converted to carbon compounds.

5. disturbance - biodiversity has been found to be very low in habitats with very high and very low disturbance rates with the highest levels of biodiversity found in areas with a moderate disturbance regime.¹⁵
6. time - biodiversity is strongly correlated with time at all scales. The literature has shown particular interest in the changes in biodiversity over evolutionary time and successional time.

It should be noted that no single explanatory variable will be sufficient to describe a pattern of biodiversity and in most cases a complicated, non-linear, non-deterministic relationship will exist. Any discussion of biodiversity must explicitly consider the relevant spatial and temporal scale.

2.5.3.1 Landscape Ecology

Landscape Ecology has emerged as a distinct discipline concerned with the spatial characteristics of ecological systems. "From an ecological perspective, landscape ecology offers a way to consider environmental heterogeneity or patchiness in spatially explicit terms" (Wiens et al., 1993).¹⁶ By making the spatial heterogeneity of the landscape explicit, landscape ecology models facilitate investigation of the relationships between the landscape composition, structure and function. The landscape ecology literature defines landscape structure in terms of the distribution of resources. Landscape structure is characterized as the number, size and shape of patches and the distances between these patches (With et al., 1997). Table 2.3 describes the primary measurable characteristics of a landscape adopted by the landscape ecology literature. The heterogeneous nature of agricultural landscapes dictate that a landscape ecology informed approach to biodiversity analysis is appropriate. With et al. (1997) state that many models which explicitly deal with spatial habitat characteristics often

¹⁵ Disturbance is any process or condition external to the natural physiology of living organisms that results in the sudden mortality of biomass in a community on a time scale significantly shorter than that of the accumulation of biomass.

¹⁶ Landscape or habitat patchiness refers to the diversity and size of habitat units across a landscape.

Table 2.3. Landscape ecology defined landscape characteristics (Wiens et al., 1993).

Feature	Description
Size distribution	Frequency distribution of sizes of patches of a given type
Boundary form	Boundary thickness, continuity, linearity, length
Perimeter : area ratio	Relates patch area to boundary length; reflects patch shape
Patch orientation	Position relative to a direction process of interest (ie. Water flow)
Context	Immediate mosaic-matrix in which a patch of a given type occurs
Contrast	Magnitude of difference in measures across a given boundary between patches
Connectivity	Degree to which patches of a given type are joined by corridors into a lattice of nodes and links
Richness	Number of different patch types in a given area
Diversity	Equivalence in numbers (or areas) of different patch types in a mosaic (the inverse of the degree of dominance by one or a few patch types)
Dispersion	Distribution pattern of patch types over an area
Predictability	Spatial autocorrelation; the degree to that knowledge about features at a given location reduces uncertainty about variable values at other locations.

portray habitat patches embedded within an inhospitable matrix. Species likely do not have a binary perception of landscapes (presence/absence of resources), but instead respond to a gradient of resource quality. Species respond to habitat in terms of movement behaviour, habitat affinities, assessment of habitat quality, and ultimately the consequences of such habitat relationships for fitness (With et al., 1997).

2.6 CROSS-DISCIPLINARY MODELS

To this point the discussion has focused on models that have been designed primarily to describe processes and relationships of a system that is artificially isolated by discipline defined boundaries. Although many models include recognition of extra disciplinary influences on the target system, these factors are often included only as a constraint or assumption. In response to the inherent shortcomings of the single discipline, or relatively narrow focus studies discussed above, there has been increasing

activity in the development of models that adopt a cross-disciplinary or systems approach.

It is the very nature of systems that they consist of interrelated parts and that they are embedded in larger systems. Conceptually, we separate a system from its surroundings by boundaries in space and time such that what lies outside the system can be regarded as unaffected by, and not affecting, the system of interest. Traditional, reductionist science has chosen this approach for the study of virtually any system..... However, it is the interaction of the system with its surroundings that 'fuels' the processes of any living or nonliving system (Weston and Ruth, 1997).

The models that will be described in this section will somehow explicitly recognize the dynamic influence of extra disciplinary factors on the functioning of the target system in an attempt to have the model describe a more realistic system response.

Economic and ecological analysis needs to shift away from implicit assumptions that eliminate links within and between economic and natural systems because, due to the strength of the real-world interactions between these components, failing to link them can cause severe misperceptions and indeed policy failures (Costanza, 1987).

An analysis and modeling approach that embraces the links between the economic and environmental systems can provide insights into the behaviour of these systems and facilitate an evaluation of system sustainability.

2.6.1 General Conceptual Models

Conceptual models are often developed to represent a system in a general way and thereby promote a better understanding of the critical relationships that make up the target system. The general nature of these conceptual models facilitates their application across disciplines. This cross-discipline application highlights the similarities and differences across systems and indicates areas where disciplinary insights can be employed elsewhere.

An important conceptual model that has proven insightful across disciplines is the game theory model. Game theory was developed for applications in economics by von Neuman and Morgenstern (1944). A game in economics consists of a set of players

(firms or consumers), a set of alternative strategies available to each player and a set of payoffs (profits or utilities) that are ascribed to the players as a function of played strategies. The point in the game where all players have selected their optimal strategies, such that when the other player's choice is revealed, none of the players wishes to change their behaviour, is called a Nash equilibrium. Bishop (1978) describes a game theory approach to public decisions and species extinction using a safe minimum standard that calls for avoidance of extinction unless the social costs are unacceptably large. Game theory has been further adapted to biological applications by introducing evolution such that strategies are identified with genes, and the payoff is high reproductive success (Maynard-Smith, 1982). In this application of game theory a Nash equilibrium would correspond to an evolutionarily stable strategy. Evolutionary game theory has been re-applied to temporally dynamic economics to facilitate understanding of the evolution of economic institutions and of technical processes in production. An example of a conceptual model that was developed with ecological foundations but has a goal of maximum generality is Holling's system flow model (1992) (Figure 2.2). The Holling model contains four elementary functions that are common to all complex systems. System development involves an evolutionary path that passes through these functions: (1) exploitation (e.g. pioneer species, entrepreneurs); (2) conservation (climax systems, rigid bureaucracies); (3) release (political upheavals, fire); and (4) reorganization (abundant natural resources) (Costanza et al, 1993). This general model can be applied to a wide range of economic and ecological system providing insight into the important stocks and flows of a dynamic system.

2.6.2 Integrated Models

In general, integrated models are those models that not only encompass more than one scientific discipline, but link these disciplinary models in a meaningful way. Russell (1996) identified two categories of disciplinary integration: (1) strong integration - the disciplines are essentially merged and a new discipline emerges using some combination of the insights of the component disciplines (e.g. environmental economics,

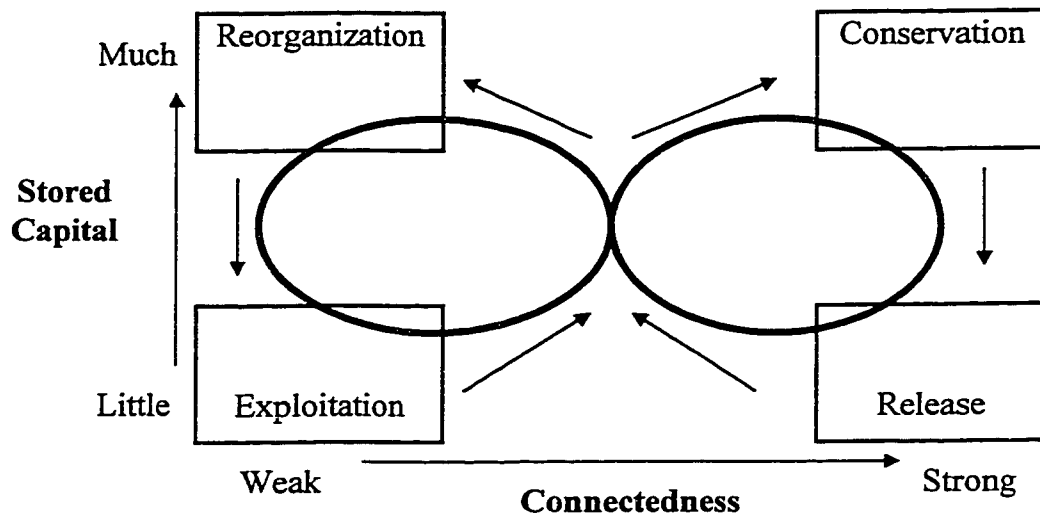


Figure 2.2 The Holling system flow model (Holling,1992).

bioeconomics); (2) weak integration - the disciplines continue to use and refine their own paradigm, appropriate to the target system, but together they create combined models of the interactions between the two systems. The following discussion will first concentrate on strongly integrated models and then examine some relevant weakly integrated models.

2.6.2.1 Strong Integration

Many strongly integrated models that focus on agricultural sustainability include an economic/resource use model integrated with some feature of the physical, or biological environment that constitutes the context for the economic system. Although some of the models that will be discussed do not relate directly to this study's research question, the framework developed by this literature provides valuable insight into the process of developing multidisciplinary models.

2.6.2.1.1 Strongly Integrated Conceptual Models

Conceptual models have been developed that explicitly recognize land use and land use change as the driving force of environmental change. Teague (1996) developed a conceptual framework that highlights the following goals as essential to the sustainable use of range land: (1) account for environmental effects; (2) decrease reliance on depletable and polluting non-renewable resources; and (3) maintain the resilience of ecological systems. This framework identifies land use as the important linkage between economic and ecological processes.

Conceptual models have been developed to address the issue of soil quality, or soil health using a multidisciplinary framework. Doran and Parkin (1994) designed a conceptual model to establish a soil quality index based on: (1) sustainable production - plant production and resistance to erosion; (2) environmental quality - groundwater quality, surface water quality and air quality; (3) human and animal health - food quality that includes food safety and nutritional composition. This model provides a framework for the integration of the environmental and economic components of soil quality.

A conceptual model dealing explicitly with biodiversity was developed by Noss and Cooperrider (1994). This model focuses on human population growth, resource consumption, and the efficiency of resource use as the fundamental agents of biodiversity loss or extinction. However, this model includes population growth, resource consumption, and resource use as constants rather than dynamic processes over time.

These three conceptual models provide examples of frameworks that highlight the importance of the linkages between the economic, ecological, and environmental systems. The conceptual modeling process helps to identify and refine those interactions that are essential to the construction of effective theoretical and applied models.

2.6.2.1.2 Theoretical and Applied Models

A landmark theoretical model that makes explicit the characteristics of the natural environment, within an economic framework, was developed by Arrow and

Fisher (1974). The authors include uncertainty and irreversibility in a commercial development decision concerning an unspoiled natural area that is capable of yielding benefits in its preserved state. This model reveals that “the expected benefits of an irreversible decision should be adjusted to reflect the loss of options it entails.” Albers (1996) developed a model that incorporates irreversibility, uncertainty, and important ecological characteristics into a decision process of optimizing the present value of tropical forest land use. Albers’ model encourages land use decisions that are not permanently disruptive to the ecosystem, a result that is consistent with the findings of the Arrow and Fisher model. Both models provide results that are consistent with the concepts of sustainability, including development within a system’s carrying capacity and the intergenerational transfer of resources.

Common and Perrings (1992) developed a theoretical model of resource allocation that integrates economic and ecological concepts of sustainability. The Common and Perrings model integrates the Solow/Hartwick model of sustainability from economics and the Holling model of the resilience and stability of ecosystems from ecology. The integration of these disciplinary models provide insight into the principal differences between economic and ecological models;

The axiomatic framework of the Solow model is fundamentally blind to the properties of the physical system in which the economic system is embedded. Indeed, it contains a variety of free gifts and free disposals assumptions that insulate the model from its environment, and prevent consideration of the most important dynamic implications of resource use. The axiomatic framework of the Holling model, on the other hand, privileges the system over its component parts. The dynamic economic problem for Holling exist precisely because of the physical feedbacks that characterize the growth and decay of subsystems within the global system (Common and Perrings, 1992).

In general the economic model adopts a micro view of the system while the ecological model adopts a more macro view of the system processes. This integrated model was used to identify some of the basic principles of sustainable resource use and recommends an approach that privileges the requirements of the system above those of the individual. “The system must retain its resilience in order to cope with random shocks, this criteria

is not met if the satisfaction of private preferences are the measure of system performance.” These findings are consistent with the concepts of weak and strong sustainability discussed earlier in this chapter.

Another more general theoretical model examines sustainable development within an endogenous growth model that integrates a dynamic specification of economic and ecological relations and the interactions between the economy and the natural environment (Hofkes, 1996). This model uses a long-term growth model where the growth rate is determined endogenously and may depend on preferences and technology. The analysis indicates that when sustainable development is incorporated both traditional economic variables and the quality of the environment should grow at a constant rate. This is consistent with the Common and Perrings result where the integrity of the natural environment influences the growth rate of economic output, since the quality of the natural environment is also a factor of production.

Orazem and Miranowski (1994) developed a model whereby farmers maximize profit by allocating land to different crop management schemes. The acreage allocation decisions are influenced by current and expected future harvest prices. The magnitude of the harvest prices is determined by producer perceptions of how current crop choices affect future soil productivity, thereby endogenizing the environmental context through a specification of the user cost of soil degradation. Other authors have attempted to model non-myopic farm decision makers by incorporating a representation of the user cost of soil degradation (Goetz, 1997; Hu et al., 1997; Fox et al., 1995; Jones and O'Neill, 1992; Milham, 1994).

Other models have been developed which focus on the economic decision process of agricultural producers while attempting to endogenize such environmental effects as; range land quality (Hu et al., 1997; Huffaker and Cooper, 1995; Torell et al., 1991; Karp and Pope, 1984); and wildlife habitat impacts (Liu, 1993; Powers, 1979). In general these models recommend management strategies that conserve or preserve the relevant natural capital or natural resource base. A shortcoming of these models is that the environmental effects that can be effectively endogenized within an economic framework are relatively limited. There are two main reasons for this shortcoming: (1)

on-site costs of agricultural engendered environmental degradation are small when compared to off-site costs (Ribaud, 1986); (2) many types of damages, on-site or off-site, may have no readily measurable economic value. In addition, environmental resources often have no substitute (technical or economic) and environmental degradation can be irreversible, at least within a relevant time period. As a result, a model that does not rely exclusively on economics processes would be a valuable tool in addressing sustainable agriculture questions.

An important feature of many natural capital components is that there are no representative prices due to the absence of a functioning market. One of the limitations of weak sustainability identified earlier is that the absence of meaningful prices makes it difficult for economic studies to account for changes in resource scarcity and quality. As a result, it is difficult to develop a model of an economic system that explicitly recognizes its environmental context. A number of studies have attempted to prescribe an intertemporally efficient allocation of environmental resources by developing a full set of prices to represent the non-market goods (i.e. environmental goods and resources) in the market. The technique used most often to assign non-market goods with prices is contingent valuation (Bowker and Stoll, 1988; Phillips et al., 1989; Hanemann, 1994). However, the non-market good prices developed through these techniques have been questioned for their applicability to resource allocation problems (Bishop and Heberlein, 1979; Diamond and Hausman, 1994; Stevens et al., 1991). Spash and Hanley (1995) report that biodiversity preservation values elicited through contingent valuation do not represent measures of welfare change due to the prevalence of lexicographic preferences.¹⁷ A significant number of individuals refuse to make a trade-off between biodiversity and market goods, and that knowledge of biodiversity, in terms of its contribution to the health of the ecosystem, is limiting. Common and Perrings (1992) indicate that the optimal value of the stock is the important issue and that prices generated in simulated or surrogate markets are surely to be irrelevant to such a measure.

¹⁷ Lexicographic cases occur when one thing is absolutely preferred over another and therefore individuals are unwilling to make trades.

2.6.2.2 Weak Integration

Models that weakly integrate more than one disciplinary paradigm have received greater attention in recent years with processes being developed for a range of applications. The emerging discipline of ecological economics has embraced these types of models most enthusiastically. In a weak integration modeling framework “....each discipline is called upon to do what it does best, and together they are required to bring their individual predictive models into fruitful communication. One model’s output becomes the other’s input” (Russell, 1996). One of the important concepts that is captured by this style of integrated models is co-evolution. Co-evolution has its origins in biology and has been extended to the interactions taking place between evolving economic and environmental systems (Norgaard, 1988). Co-evolution occurs when there is a change in one of the component systems that initiates a series of changes in other components through feedback mechanisms. While these co-evolutionary processes may be difficult to represent in a strongly integrated model, the weakly integrated framework lends itself to tracking co-evolutionary pathways. This section will review some of the relevant weakly integrated models from the literature.

2.6.2.2.1 Weakly Integrated Conceptual Models

Conceptual models that weakly integrate the economic and environmental components of an agricultural system have been developed to identify the important system linkages, and to highlight areas of future research. Dent et al. (1995) describe a conceptual modeling framework that incorporates existing economic and environmental models in an integrated model. The authors suggest that there is a wide range of adequate models to simulate the crop production, animal production and ecological systems. The integration of these models into a whole farm model is considered to be a relatively simple process. However, the authors state that the socio-economic component of the farm is not well represented by existing models. In particular the assumptions of homogeneous farm units, and financial maximizing decision-makers,

that are often embraced by linear programming procedures, are criticized and considered inappropriate.

Riebsame et al. (1994) proposed a similar conceptual model to Dent et al. (1995) and assert similar criticisms of existing socio-economic models. The authors suggest that the conventional economic criteria of maximized return on investment should be relaxed to include a menu of goals that are not grounded solely in economics, due to evidence suggesting that farmer behaviour is neither economically nor ecologically rational. Riebsame et al. conclude that this type of integrated model will be valuable for the assessment of the impact of such forces as climate change across a range of economic and policy contexts. In addition, integrated models will be valuable in questions of sustainability by changing the focus from purely ecological or social factors.

Sensitivity analyses based on both traditional and innovative measures of sustainability could move researchers beyond the arguments over definitions, while providing a foundation for assessing more encompassing sustainability goals such as ecosystem health and rural community well-being (Riebsame 1994).

2.6.2.2.2 Theoretical and Applied Models

In recent years models that apply the weak integration framework have become more common. Some of these models integrate existing economic and physical process models to assess the environmental implications of agricultural policy. At least two of these weakly integrated models have direct relevance for western Canadian agricultural systems.

Lakshminarayan et al. (1996) developed a framework that integrates the previously discussed EPIC and CREAM models. These authors develop a metamodeling approach to evaluate the impact of agriculture policy on soil degradation in western Canada. Metamodels are statistical summaries of data obtained from complex simulation models, reducing the simulation models to input-output relationships. Metamodeling relies on point averages to represent non-linear environmental process thereby removing the dynamics of the interacting systems. The sub-models do not

interact but run parallel and therefore are not appropriate for analysis of systems in which the temporal dynamics are critical to the functioning of the system.

Gheidi (1997) linked CENTURY and CRAM for a Saskatchewan agriculture application. The two models were linked in a dynamic way to capture the effect of alternative tillage practices on producers' net return and changes in soil organic matter. CENTURY and CRAM were linked manually in this study by independently running each for five year intervals and transferring the output parameters between the two models. Gheidi reported that there is a lack of economic incentive for producers to adopt soil conserving production practices. Those practices with the greatest on-site and off-site costs of soil organic matter losses also provided the highest financial returns to the producer. This result suggests that degradation of the agriculture system's natural capital stock is insufficient, or inadequately recognized by economic signals to inspire a change in resource use.

Foltz et al. (1995) linked EPIC, GLEAMS and a multi-attribute utility function to assess the environmental impact and economic returns from alternative cropping systems. The cropping system choices were then evaluated from a farmer's perspective, and the perspective of an individual with environmental concerns. The authors reported that groups with very different preference rankings could find alternatives that simultaneously satisfy their goals.

Bernardo et al. (1993^a; 1993^b) linked EPIC, GLEAMS and a regional economic mathematical programming model to assess the environmental and economic consequences of groundwater quality protection policies, at a regional level. The evaluated policies targeted: (1) limiting the total quantity of nitrogen applied; (2) limiting the unit-area nitrogen applications; and (3) restricting selected pesticides. The model allowed producers to employ a range of management responses including crop substitution, land retiring, changing irrigation patterns, decrease fertilizer application, chemical substitution, and investment in more efficient irrigation technology. This model provides information on the tradeoffs between economic and environmental consequences of water quality protection policies and highlights the different effects of different policies.

Integrated models have been developed to address a range of other agricultural/environmental issues including groundwater pollution (Johnson et al., 1991), animal manure (Hoogervorst, 1995), and soil erosion (Mallawaarchchi et al. 1996).

A framework developed by Bouzaher et al. (1995) integrated a diverse collection of simulation models. The CEEPES (Comprehensive Environmental Economic Policy Evaluation System) model provides a flexible framework that can be modified by integrating economic and physical models as new policy questions arise. This model was built around four major components; policy space, agricultural decision, fate and transport, and environmental/health risk. The procedures within CEEPES linked the fate of pesticides and nutrients to cultivation practices, application rates, soils and climatic conditions, and agricultural income maintenance policies. The intent of this model was to decrease the uncertainty in policy evaluation, to identify gaps in information, and to highlight areas requiring more research. The modular nature of the CEEPES model is instructive to the present research. CEEPES is constructed to facilitate the incorporation of modeling components that focus on different aspects of the system. This flexible framework ensures that a range of policy questions can be addressed with minimal additional model development.

A weakly integrated model developed by Wu et al. (1996) took a broad, systems based perspective, by including annual crops, perennial forage and range land in the land use menu. The integrated model incorporates four sub-models that individually simulate: (1) soil quality and moisture conservation; (2) wheat production as influenced by rainfall, temperature, soil fertility, nitrogen and management; (3) alfalfa forage production as influenced by rainfall, temperature, soil fertility, phosphorous and management; and (4) animal biomass production. These components were dynamically linked within the model to analyze the effects of long term nitrogen fertilizer application, and the impact of population pressure on agricultural systems. What is significant in this model is the explicit integration of a series of ecological processes, and the attempt to simulate the ecological functions of a regional agroecosystem.

Bockstael et al. (1995) also developed a theoretical model that weakly integrates the ecological and economic systems at a regional scale. This model simulates the

interactions between the ecosystem and economic development “...illustrating how humans intervene in the ecosystem and how different ecosystem configurations contribute to human welfare” (Bockstael et al., 1995). The model includes: (1) an ecosystem component that is a detailed simulation model focusing on the ecological processes of the relevant ecosystem using hydrology as the driving physical process; and (2) an economic land use model based on a spatial econometric framework. These components are run in parallel but communicate ecological and economic outputs thereby linking the subsystems. This model simulates how human decisions change the structure and function of the ecosystem that over time can result in a new arrangement of values for market and non-market goods within the landscape. This style of integrated dynamic models may give insight into the co-evolutionary effect of a range of human activities (agriculture, commercial and residential development, recreation) on the ecosystem and the effect of the ecosystem on the quality and value of goods and services (soil quality, water quality, recreation, environmental aesthetics) and therefore, on human decisions (Bockstael et al. 1995).

2.7 CONCLUSION

Sustainability, as discussed in this chapter implies that:

1. natural and man-made capital are used and degraded at a rate that ensures that capital stocks are maintained for future generations (intergenerational equity).
2. the interactions and flows of energy, matter, and information between ecosystem components are maintained.

A research framework that makes explicit the cross-system flows and feedbacks, and can facilitate analysis into future time periods is appropriate for investigating the sustainability of a complex system. A weakly integrated simulation modeling framework is a suitable modeling structure for an analysis of the sustainability of agroecosystems. In Chapter Three, a conceptual framework with the above characteristics will be described.

CHAPTER 3

CONCEPTUAL FRAMEWORK

3.1 INTRODUCTION

The literature review has provided an overview of agriculture sustainability by bringing the methodologies and insights gained from a range of studies into focus for the present study. Based on this foundation, this chapter focuses on developing a conceptual framework with which to examine the sustainability of agroecosystems at a regional scale. The discussion concentrates on the conceptual foundations of the overall modeling process, and presents details of the component parts selected to simulate agroecosystems. The chapter begins with a synthesis of the insights gained from the literature review, with an emphasis on the systems level research approach (Section 3.2). The following section will break down the conceptual structure and provide a brief overview of the components of the model (Section 3.3). The next sections present an outline of the conceptual model (Section 3.4), and a brief overview of the issue of scale (Section 3.5). The following three sections focus on developing the conceptual framework for the economic model (Section 3.6), the soils model (Section 3.7), and the ecological model (Section 3.8). In Section 3.9 the inter-model linkages, which tie the system together, are identified. The final section (3.10) highlights the important issues presented in this chapter and provides a relevant link to Chapter Four.

3.2 SUSTAINABILITY AND SYSTEMS LEVEL ANALYSIS

The Literature Review provided that questions of sustainability often encompass two main characteristics: (1) multi-disciplinary, or system perspective. The decomposition of sustainability into economic, environmental, and social sustainability emphasizes its system based foundation; and (2) sustainability questions involve the allocation of capital (natural and man-made) to current and future generations. These two characteristics are strongly tied to the spatial and temporal scale at which the sustainability question is posed.

The concept of sustainability has at its core the issue of temporal scale, as highlighted by the issue of intergenerational equity. A question that requires analysis over a very short period of time may be accommodated by a single disciplinary framework. Over longer periods of time, however, exogenous factors do not remain constant and non-linear dynamics develop which limit the efficacy of the single discipline to capture important system changes. The majority of issues related to complex system sustainability involve temporal scales that require a multi-disciplinary framework.

The interdependence of natural and human systems is a defining characteristic of a complex system such as an agroecosystem. For example, precipitation and soil fertility impose limitations on the types of agricultural management that are technically, agronomically, and economically feasible within a region. In turn, the management strategies that are adopted affect the fertility of the soil and the climatic conditions, from micro-climatic conditions to global weather patterns. Using a single disciplinary perspective to examine agroecosystem sustainability would presume that these feedback processes either do not exist, or are sufficiently insignificant to be ignored in a system model. A single discipline economic model would necessarily assume that the natural resource base does not dynamically constrain economic decisions and the quality of the services provided are unaffected by production management. Therefore, an agroecosystem implies a spatial scale where the constraints imposed by the natural ecosystem processes must be considered.

3.2.1 Co-evolutionary Relationships

The changes over time in a system, such as an agroecosystem, are largely the result of components of the system imposing different forces on other components.

Norgaard (1988) describes these interdependent system changes as co-evolution.

Not only is each subsystem related to all the others, but each is changing and affecting the evolution of the others. Deliberate innovations, chance discoveries and random changes occur in each subsystem that affect the distribution and qualities of components and relations in the subsystems.....With each subsystem putting selective pressure on each of the other subsystems, they co-evolve in a manner whereby each reflects the others.

This theory is depicted in a rudimentary way in Figure 3.1, using three components of an agroecosystem: (1) soil system; (2) economic system; and (3) ecological system.

Management decisions made in the economic component affect the soil and ecological components. The soil and ecological components will undergo internal changes as a result of the economic forces. The evolved soil and ecological components then exert altered constraints on the other components. For example, management decisions made at time t in the economic system will result in a changed context (soil system and ecological system) in period $t+n$, necessitating a change in management at this time. The co-evolutionary process continues through time.

The foundation of these co-evolutionary processes is the positive and negative feedback mechanisms that tie complex systems together. Negative feedback is when a change in a component leads to a response in another component that counteracts the original change (Hannon and Ruth, 1994). Negative feedback tends to counteract disturbances, driving the system toward a steady state, and as such are considered characteristic of a resilient system. Positive feedback is when the change in a system component leads to changes in other components that then reinforce the original force. Positive feedback tends to amplify disturbance, leading systems away from a steady state. An agroecosystem can exhibit both positive and negative feedback.

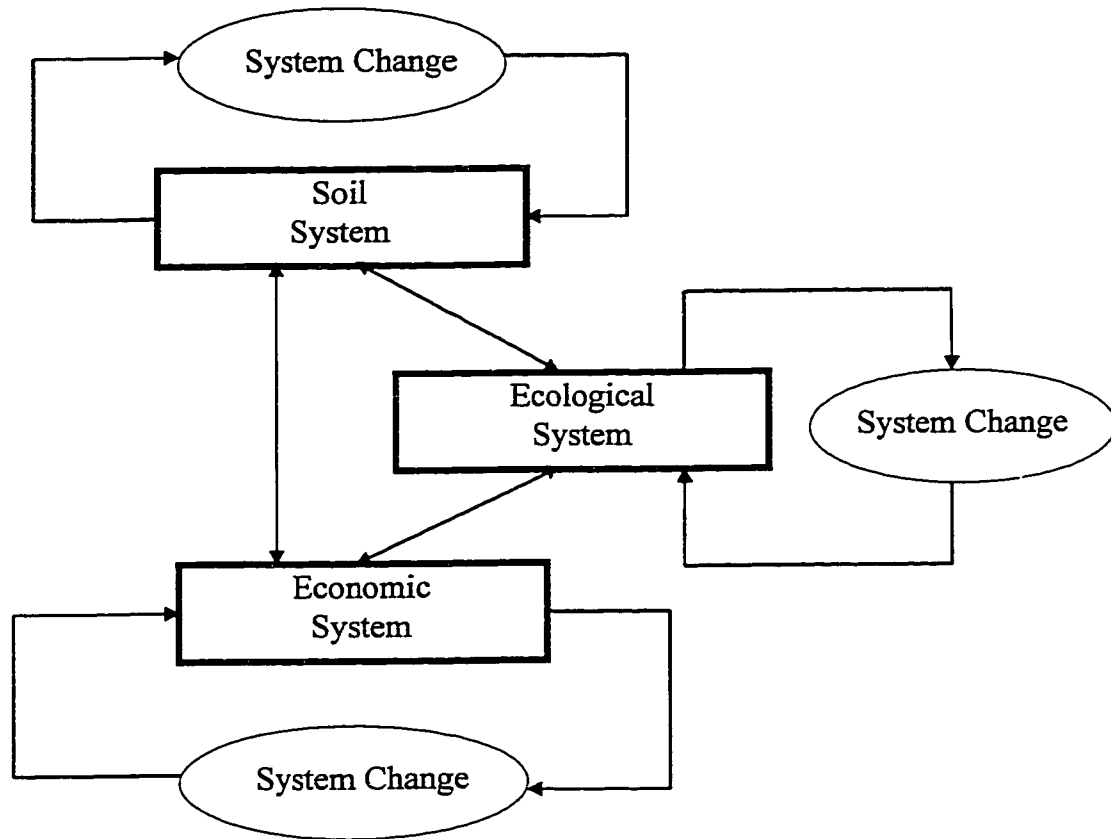


Figure 3.1 Schematic representation of co-evolutionary relationships in an agroecosystem.

3.2.2 Sustainability Indicators

Sustainability indicators, reflecting appropriate elements of the economic and environmental component systems can be evaluated individually, and in combination, to determine system sustainability. Monitoring sets of indicators to assess sustainability is consistent with the idea that sustainability is the ability of a system to meet a diverse set of goals, and that no single indicator of sustainability can exist (Norgaard, 1991). Indicators are measurable system attributes and functions that are linked to the sustainable processes of the system. For example, decreasing profitability of a production process, or increasing profit variability, may indicate that an economic system is on an unsustainable pathway.

3.3 REPRESENTING AN AGROECOSYSTEM

For a model to calculate meaningful indicators it must incorporate the components of the agroecosystem that can: (1) be modeled in a representative way; and (2) represent important elements of the system. Within the weakly integrated modeling framework relevant linkages are established between component models that represent appropriate elements of the overall system. These component models provide, as output, parameters that are meaningful indicators of system function. The indicators can be observed over the course of the modeling simulation to monitor the development of the systems with respect to sustainability.

The process of selecting relevant components for the SAM model is informed by the decomposition of sustainable agroecosystems into distinct components:

- economic sustainability - agricultural production must be economically viable.
- ecological sustainability - agricultural production must prevent environmental degradation or loss of environmental function.
- social sustainability - agricultural production must be acceptable to farmers and society within the cultural, religious, and ethical context.

A model focused on sustainable agriculture should provide indicators relevant to these elements of sustainability. SAM will be comprised of component models representing economic and ecological processes and will not explicitly include a social component as defined above.

3.3.1 The Economic Model

The economic sustainability of an agroecosystem is evaluated in terms of profits and financial risk. A sustainable production system is one that can continue to be economically viable under changing environmental, social and economic forces.

Although there are important natural forces that alter the sustainability of an agroecosystem, of equal or greater importance are the influence of land management decisions. Land management has a profound impact on the economic sustainability of farms and on the integrity of the environment. Although the literature review presented

some criticisms of modeling land use decisions based exclusively on the assumptions of homogeneous farm units and financial maximizing decision makers, there is little doubt that economic signals are an important component in land use decision making.

Therefore, an agroecosystem model should contain a component that focuses on land use decisions within an economic framework. This component should be capable of calculating a range of indicators that can be used to evaluate the economic sustainability of the agroecosystem through changes in the man-made capital stock.

3.3.2 The Soils Model

Environmental sustainability is essential to the sustainability of the agroecosystem. The principle component of the bio-physical environment that the producers interact with is the soil resource. Management decisions are influenced by the quality of the soil resource, and in turn the quality of the soil resource is affected by management decisions. This relationship is temporally dynamic, encompassing positive and negative feedbacks. For example, a management decision that degrades the soil over time alters management decisions in some future time period due to changing soil productivity, changing input requirements and changing price signals. The sustainability of an agroecosystem is therefore closely linked to the health of the soil system. For this reason the model should contain a component which can provide indicators of soil health over time. The indicators in SAM reflect changes in the natural capital stock associated with the soil resource.

3.3.3 The Ecological Model

It is a relatively easy argument to make that any model developed to address agroecosystem sustainability questions should include explicit recognition of the economic and soil systems. In contrast, changes in the biodiversity of an agroecosystem will likely not have an immediate and/or apparent influence on system function, economic signals, or management decisions. The degradation of an ecological system must be evaluated using two important criteria: (1) uncertainty; and (2) irreversibility.

As a complex system, an ecological system is comprised of many dynamic flows of energy, information and matter. The relationships and flows within the ecological system are largely unknown. As a result, any management decision that changes the biodiversity of a system imposes unknown costs on current and future generations.

Perrings (1994) states that decisions that change biodiversity are

...characterized by fundamental uncertainty about the long-run ecological implications of biodiversity change. They also raise questions about the ethics of depleting a common resource that contains the genetic blueprint for the stock of natural capital available to all future generations.

The uncertainty associated with biodiversity changes is strongly linked to the irreversibility of such changes.

Extinction is forever. If we purposely or innocently extirpate a species directly through over-harvest or indirectly by reducing its habitat below a critical minimum threshold, that is the end at present. We will have destroyed an asset of unknown value (Brown, 1996).

Not only will the change in the integrity of a system, as a result of a loss of biodiversity, be unknown, but the change will be permanent. Applying the co-evolutionary theory to this issue suggests that the system continues to change and possibly degrade as components adapt to dynamic changes in the co-evolving system.

Wildlife habitat and biodiversity have been significantly modified in the agricultural landscape of Saskatchewan. The changes in biodiversity are due to such factors as habitat destruction and modification, the introduction of exotic invader species and poisoning with agricultural chemicals. Habitat destruction and modification is a primary force in decreasing agroecosystem biodiversity.

3.4 THE CONCEPTUAL MODEL

A schematic representation of the conceptual model shows how it explicitly captures the important interdependencies and linkages between the economic, soil and ecological components (Figure 3.2). The black shaded boxes represent the exogenous

variables to the model, and the gray shaded boxes represent the economic, soil, and biodiversity components of the model.

Management decisions are driven by economic signals on the input (input costs) and output (gross revenue) sides. The economic signals are partially dependent on exogenous input and output prices and endogenous yield and input requirements. An important driver of yield and input requirements is soil quality, which is simulated dynamically by the soils model. Soil quality is theoretically affected by climatic forces, management decisions (i.e., zero tillage versus conventional tillage), yield and biodiversity (biodiversity of soil micro and macro fauna). The yield - soil quality feedback process is an important one that should be highlighted. Changing soil quality affects the ability of the soil to provide nutrients and water for plant growth. Plant growth determines the quantity of organic residues added to the soil which influence soil organic matter content and soil function. Changes in plant growth and yield also influences land use decisions, through input costs and revenues, which influence soil quality. Yield is also influenced by inputs (i.e., fertilizer). Nutrient availability, which is a function of soil quality, determines the need for fertilizers. Higher quality soil provides more of the nutrient requirements of the plant resulting in increased yields relative to the level of purchased inputs.

Land use decisions have a significant impact on habitat availability and quality on the landscape. Habitat availability and quality are altered primarily by changing plant communities. Examples of these changes include clearing of native areas, changing disturbance frequency and intensity (i.e., fallow rotations versus continuous zero-till cropping versus perennial tame forage) and the indirect influence of surrounding land use on a habitat unit (context). Bio-physical factors, such as climate and soil quality, also affect habitat variables, which influence biodiversity, since biodiversity and habitat variables are strongly correlated and interdependent. Biodiversity, in turn, can influence land use decisions directly through the land use ethic of the manager (changing management regimes in response to a decrease in populations of a desirable wildlife species), and indirectly through soil quality, and ecological services affecting input requirements (i.e., pollination services, natural pest control etc.).

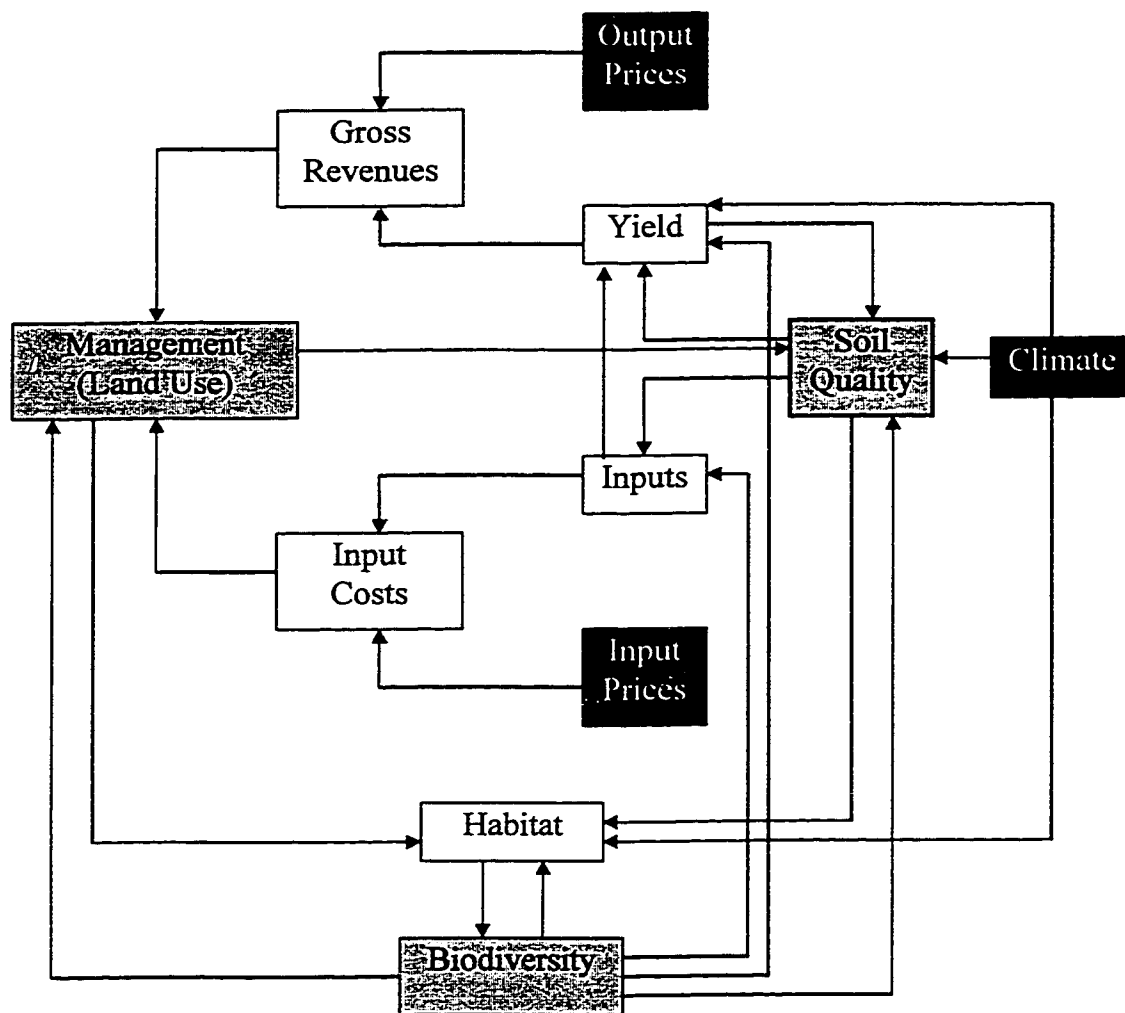


Figure 3.2 Conceptual model of an agroecosystem highlighting information, matter and energy linkages.

This conceptual framework highlights some of the relationships and feedback processes that are essential to a functioning agroecosystem. As discussed in Chapter Two, the long-term feed back effects, non-linear relationships and thresholds that are characteristic of a complex system can be captured by the framework described.

3.5 MODELING SCALE

The appropriate modeling scale is one that is large enough to enable a meaningful analysis, but small enough to focus on a relatively homogeneous unit that can technically be modeled with minimum interactions between the target system and its surroundings (Costanza et al., 1993). Economics and ecology traditionally analyze problems from very different temporal and spatial scales. Therefore, the selection of an appropriate scale is difficult when the modeling framework encompasses economic and ecological components of a system. “The dynamics of ecosystems and markets are extremely different, and these differences are reflected in the time steps and time horizons commonly employed in models of ecological and economic systems” (Costanza et al., 1993). The data available for the analysis of economic decisions are often based on time steps no less than one to three months, and more commonly on an annual basis. In contrast, ecological research has data sets for some system processes at a high level of temporal resolution. For example, the EPIC and CENTURY soil models are based on a daily and monthly time step, respectively. Further, ecological models are frequently designed to capture processes that occur over long-time horizons, often encompassing 25 year modeling horizons, and when considering ecosystem changes will calculate values beyond 100 years. Economic theory often assumes that future shocks and adjustments are impossible to predict and tend to ignore modeling horizons greater than 25 years, and frequently restrict analysis to five to ten years.

A one year time step was selected for the model to be consistent with the temporal characteristics of the decision making scope of agricultural firms. Although it is acknowledged that such factors as intra-year precipitation variability can be important determinants of crop growth, profitability, soil erosion etc., the extra complexity that greater temporal precision would add to the model was unnecessary to meet the model objectives. With respect to simulation time horizons the inconsistency between the long-term ecological system analysis and short-term economics system analysis is an important issue and will be considered when interpreting simulation results.

Inconsistency in spatial scale is also a problem when integrating economic and ecological models. Ecological models generally focus on physical flows such as water,

energy, or biomass, which have distinct spatial characteristics. In contrast, economic models usually focus on informational flows such as money and prices which are not as spatially bounded (Bockstael et al., 1995). As a result, economic models are rarely explicitly spatially oriented, and most often use the boundaries of the market to define the modeling scale. Economic data is available at the firm level, and most often collected and aggregated at a regional level, where the region is defined by market or political boundaries. In contrast, ecological models are often explicitly spatial with boundaries defined by physical features such as biomes, soil zones or watershed. Ecological data is therefore tied to a precisely defined spatial scale.

The spatial foundation of the model is an ecodistrict. An ecodistrict is defined as “subdivisions of ecoregions, characterized by distinctive assemblages of land form, relief, surficial geological material, soil, water bodies, vegetation and land uses” (Acton et al., 1998). The spatial scale selected for this model is informed by the desire for a qualitatively precise simulation model.

1. The soils model calculates soil parameters on a per hectare scale based on the dominant soil characteristics of the target ecodistrict.
2. The economic model calculates economic parameters based on input and output statistics representing an average hectare of land in the ecodistrict. The soils and economic parameters can then be aggregated to an ecodistrict scale.
3. The ecological model calculates habitat parameters at an ecodistrict scale.

This framework enables the disciplinary models to communicate at a common spatial and temporal scale.

3.6 ECONOMIC LAND ALLOCATION

A mathematical programming style area response model was used for this research application. A simple variation of the common mathematical programming model, the fixed proportion framework, was adapted for the present study. The fixed proportion framework is easily constructed and as such is amenable to assembly within a

modeling environment that is also appropriate for constructing models representing other components of the system.

Although the fixed proportions framework is quite restrictive in its assumptions, integration of the economic and soils model in SAM relaxes some of these restrictions. The economic model allocates only chemicals and other inputs in fixed proportions with the land input. Mathematically, the economic component of the model incorporates the following production function:

$$x = f(g, \min\{y, c, o\}) \quad (3.1)$$

where:

x - output
g - substitutable inputs (nitrogen fertilizer, soil nutrients, water etc.)
y,c,o - inputs in fixed proportions (land, chemicals, “other inputs”)

Later sections of this chapter and chapter four will provide details on how such inputs as synthetic fertilizer, soil nutrients and water are provided through sufficiency relationships and not as fixed proportions.

3.6.1 Fixed Proportions Model

The fixed proportions framework is a very simple policy analysis tool driven by the derived demand for land (Gardner, 1987). The foundation of the fixed proportions framework is a system of vertically linked supply and demand relationships representing the input and output markets associated with the production of a particular commodity. Two primary assumptions are necessary to construct the vertically linked system.

The first assumption is that output is produced in fixed proportions with the required inputs. This assumption can be summarized by the following relationship:

$$x = y = z \quad (3.2)$$

where:

x = units of output
y = units of land input
z = units of non-land input

Equation 3.2 demonstrates that for each additional unit of output produced (x), one more unit of each of the inputs (y, z) is required. The inputs and outputs in this framework are tied to the land input such that one unit of land is one hectare and one unit of output is the quantity of output produced on one hectare, and one unit of non-land input is the quantity of input required to produce one unit of output on one hectare of land. The fixed proportion assumption dictates that the elasticity of substitution between all input pairs must be zero.

The second assumption is that at equilibrium, profits are zero:

$$\Pi = P_x(x) - P_y(y) - P_z(z) = 0 \quad (3.3)$$

or

$$P_x(x) = P_y(y) + P_z(z) \quad (3.4)$$

where:

Π = Profit (\$/unit)

P_x = Price of output (\$/unit)

P_y = Price of land input (\$/unit)

P_z = Price of non-land input (\$/unit)

Equation 3.4 illustrates that at equilibrium total revenue equals total cost. Within this perfectly competitive industry the agricultural firms are assumed to be price takers such that individual firms have no effect on market prices, and as a result take market prices as given when making production decisions. It is also assumed that entry into, and exit from the industry is relatively easy such that when profits are positive or negative firms will enter or exit the industry thereby describing output supply. Combining the fixed proportions and perfect competition assumptions gives the following relationship:

$$P_x = P_y + P_z \quad (3.5)$$

Equation 3.5 indicates that at equilibrium the price per unit of output is equal to the sum of the per unit input prices. Based on these relationships the fixed proportions model is constructed using a set of input and output markets that are vertically linked and

horizontally tied through the land input. This structure is demonstrated graphically in Figure 3.3.

Figure 3.3 represents an industry with two sectors (A and B) each producing an output. Each sector faces a perfectly elastic output demand function. The product of the output price (\$/tonne) and output yield per land unit (tonnes/hectare) is the gross revenue received for each output (\$/hectare). The gross revenues are reflected by the P_{xj} lines at the top of Figure 3.3.

The supply curve for non-land inputs is upward sloping indicating that these inputs are in some way specific to the relevant output and as such can not be re-allocated to other outputs at zero cost. Assuming a linear supply curve for non-land inputs results in the following inverse supply curve for non land inputs:

$$P_{zj} = \delta_{zj} + \gamma_{zj} Q_{zj} \quad (3.6)$$

where:

$$\begin{aligned} P_{zj} & \text{ - price of non-land input for output } j \\ Q_{zj} & \text{ - quantity of non-land input for output } j \end{aligned}$$

The demand for land input for each of the outputs is a derived demand based on the gross revenue received from each unit of land, and the non-land inputs supply function. The price of land is determined by rewriting equation 3.5 as, $P_y = P_{xj} - P_{zj}$. Substituting equation 3.6 into this expression gives the derived inverse demand for land:

$$P_y = \eta_{yj} + \lambda_{yj} Q_{yj} \quad (3.7)$$

where:

$$\eta_{yj} = P_{xj} - \delta_{zj}$$

$$\lambda_{yj} = -\gamma_{zj}$$

$$Q_{yj} \text{ - quantity of land input for output } j$$

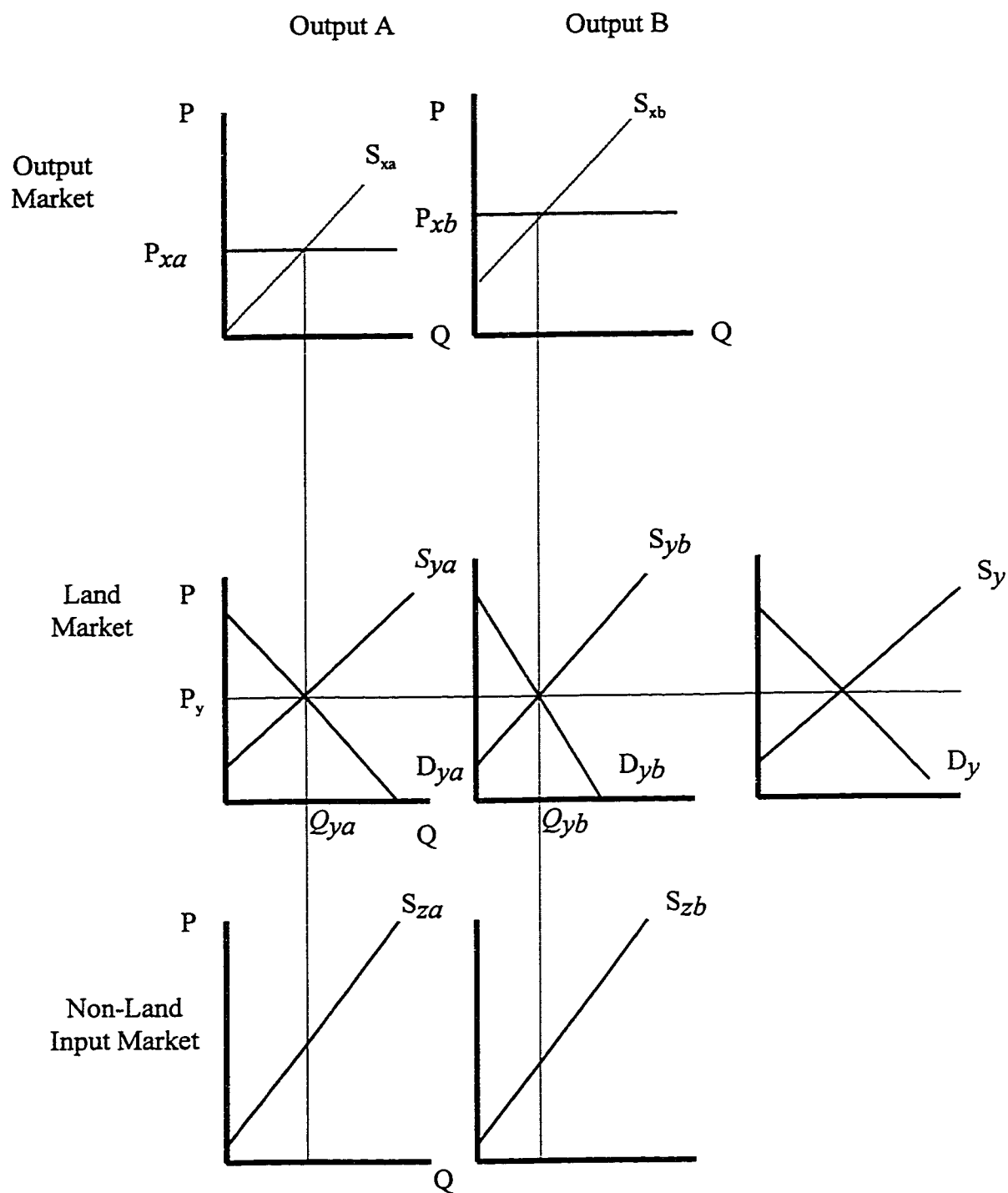


Figure 3.3. Graphical representation of fixed proportions model vertical system.

The demand for land within each sector can be derived by solving equation 3.7 for Q_{yj} :

$$Q_{yj} = \rho_{yj} - \theta_{yj} P_y \quad (3.8)$$

where:

$$\rho_{yj} = \frac{\eta_{yj}}{\lambda_{yj}}$$

$$\theta_{yj} = \frac{1}{\lambda_{yj}}$$

The overall demand curve for land is the summation of the derived land demand curves in equation 3.8:

$$Q_y = \rho_y + \theta_y P_y \quad (3.9)$$

where

$$\rho_y = \sum_{j=a}^b \rho_{yj}$$

$$\theta_y = \sum_{j=a}^b \theta_{yj}$$

The supply curve for the overall land market is upward sloping reflecting the technical possibilities for increasing or decreasing the cultivated land stock. Assuming a linear supply relationship for the overall land market results in the following overall supply curve for land:

$$Q_y = \alpha_y + \beta_y P_y \quad (3.10)$$

where:

Q_y - quantity of land

The market clearing land price can be formed by equating the supply and demand equations for land:

$$P_y = \frac{\rho_y - \alpha_y}{\beta_y - \theta_y} \quad (3.11)$$

P_y represents the opportunity cost of the land input and is measured in terms of dollars per unit (\$/hectare). Within SAM the market clearing land price is directly related to land rent. Entering P_y into the derived land demand curves for each output (D_{yj}) the

model calculates the equilibrium quantity of land dedicated to each of the outputs (Q_{yj}). Through the fixed proportions assumption the quantity of non-land inputs required (Q_{zj}) and the quantity of output (Q_{xj}) at equilibrium are known.

The fixed proportions framework is a very simple modeling structure that can be used to capture land use decisions under changing conditions. The endogenous nature of the derived land demand functions ensures that changes in the output demand or input supply functions will result in a new equilibrium land allocation and associated input and output quantities. The example in Figure 3.3 and the previous discussion is based on a two input, two output system. However, the structure can be expanded to include more inputs and outputs without greatly increasing the complexity of the calculations. Within SAM the economic model output is the quantity of land allocated to four crop rotations such that one unit of output is the quantity of grain and/or oilseeds produced on an average hectare of land in the rotation. The inputs in SAM are separated into four separate categories: (1) land; (2) fertilizer; (3) chemicals; and (4) “other inputs”. A full description of the input and output characteristics of the SAM model is provided in Chapter Four.

3.6.2 Other Land Stock

The economic framework described in section 3.6.1 encompasses land allocated to annual crop production. The management of land that is allocated to non annual crop use is an important consideration in assessing agroecosystem sustainability. The conceptual framework recognizes two perennially vegetated categories of other land: (1) tame forage land (tame hay and improved pasture); and (2) native grass (native pasture, shrub land, and wetlands).

The economic framework discussed in section 3.6.1 is appropriate for modeling land use changes that have very low or no conversion cost. Land allocated to tame forage or native grass is maintained in that use for periods of time greater than one year due to higher conversion costs. Conversion costs for tame forage encompass two or three year establishment periods and include breaking costs when converting from tame

forage to annual cultivation and fencing costs in the case of improved pasture. Native grass conversion costs are primarily comprised of breaking costs. Since these conversion costs are not included in the fixed proportions model, the economic framework discussed up to this point is inappropriate for the simulation of land use decisions on these perennially vegetated lands.

The market clearing land price calculated by the economic model represents the opportunity cost of all land in the agroecosystem and as such can be used to drive land use decisions for all types of land. Within SAM the net revenues (gross revenues minus costs, including conversion costs) for forage lands are evaluated relative to the current land rent value at each time step. Land is allocated to annual crop or forage production based on the highest potential economic returns to each land use. The conversion costs discussed above result in inertia in the land use switching dynamics between annual crop and tame forage production. As a result the economic threshold for converting land from forage to annual crop (break-up threshold (BUT)) is distinct from the economic threshold for converting annual crop to forage (set-aside threshold (SAT)).

The dynamics of land switching between tame forage and annual crop is depicted in Figure 3.4. The D_t line represents the per hectare difference between grain (G_t) and forage (H_t) net revenue. The horizontal line labeled 0 represents the point where G_t and H_t are equal ($D_t=0$). If it is assumed that at $t = 0$ land is allocated to forage production, when D_t crosses the BUT ($t=X$) the land will be converted to grain production. Land will remain in grain production until D_t crosses the SAT ($t=Y$), at which point it is converted to forage production. Conversion costs tempers the conversion decision such that land remains in grain production for a number of time periods where $H_t > G_t$.

For native land the cost of conversion is evaluated relative to land rent at each time step to determine if it is economically attractive to convert native grass to annual crop production. The framework assumes that land converted from native grass to annual crop production can be used for annual crop or tame forage production only in future time periods. Therefore the stock of native grass will never increase over the course of a simulation. Using these processes the economic model provides output on the area of land allocated to annual crop rotations, tame forage and native grass.

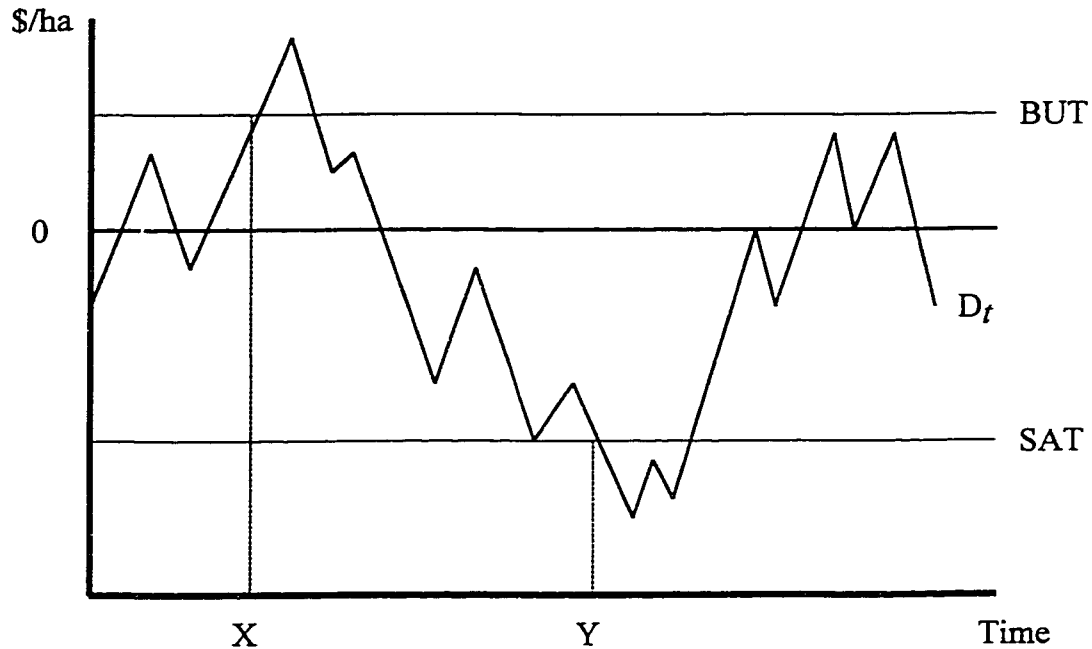


Figure 3.4. Economic thresholds for converting land from annual crop to tame forage (SAT) and from tame forage to annual crop (BUT).

3.7 SOIL HEALTH AND PRODUCTIVITY

The soils model within SAM is based partially on the theoretical foundation provided by the Simulated Productivity Lost by Erosion (SimPLE) model developed by Greer and Schoenau (1992). SimPLE was developed with the objective of creating a “simple spring wheat model which captured the essential relationships between topsoil erosion and productivity loss in Chernozemic soils” (Greer et al., 1992). SimPLE incorporates three important relationships that describe: (1) how wheat crops generate yield from available water, nitrogen, and phosphorous; (2) how these three inputs are provided by the soil; and (3) how the supply of these inputs is affected by soil erosion. The model was validated based on a continuous spring wheat rotation.

The underlying relationships of the SimPLE model are based on the theory of boundary line yield prediction, such that when a number of non-substitutable resources control the production of a good, a limitation of one of these resources will limit output. Specifically, within the SimPLE model wheat yields are limited by available water,

available N and available P such that the combined limitations will determine yield (Greer et al., 1992).

To construct the soils component for the SAM, curves were empirically fitted around the boundary of scatter-plot data relating crop yields to available water (Figure 3.5), P and N for wheat, canola and peas. These boundary line plots were developed to represent sufficiency curves for the crop and the three inputs. The maximum potential yield (Y_{max}) for a given crop is the yield that would result if none of the inputs are limiting. The sufficiency curves are interpreted such that some level of inputs will be sufficient to produce some fraction of maximum potential yield (Y/Y_{max}). These fractional sufficiencies for each of the three inputs are then multiplied. This multiplicative relationship assumes that the yield resulting from a single input is further limited by the other inputs, unless those other inputs are completely sufficient ($Y/Y_{max}=1$). It should be noted that within the SAM peas are not limited by N due to the ability of this crop to produce N. Further, canola is limited by heat such that high temperatures during flowering will decrease yields. .

Crop yield is assumed to be dependent on the three inputs, therefore the soil model predicts, at a yearly time-step, soil moisture, available N and available P given the specific climatic and biophysical parameters of the target landscape. The model is constructed of a series of dynamically linked components that simulate the important processes of the soil system (Figure 3.6). Each of the components: (1) soil water; (2) soil organic matter carbon; (3) soil N; (4) soil P; (5) soil erosion; (6) crop production will be introduced briefly here. Further information on the theoretical foundation of these components is available in Greer et al. (1992) and CSALE (1997^a; 1997^b).

3.7.1 Soil Water Component

The availability of water for plant growth is an important crop productivity parameter in this model due to there being a growing season moisture deficit within Canadian prairie agroecosystems. The water input is divided into rainfall during the growing season, and precipitation accumulated since the last growing season (snow

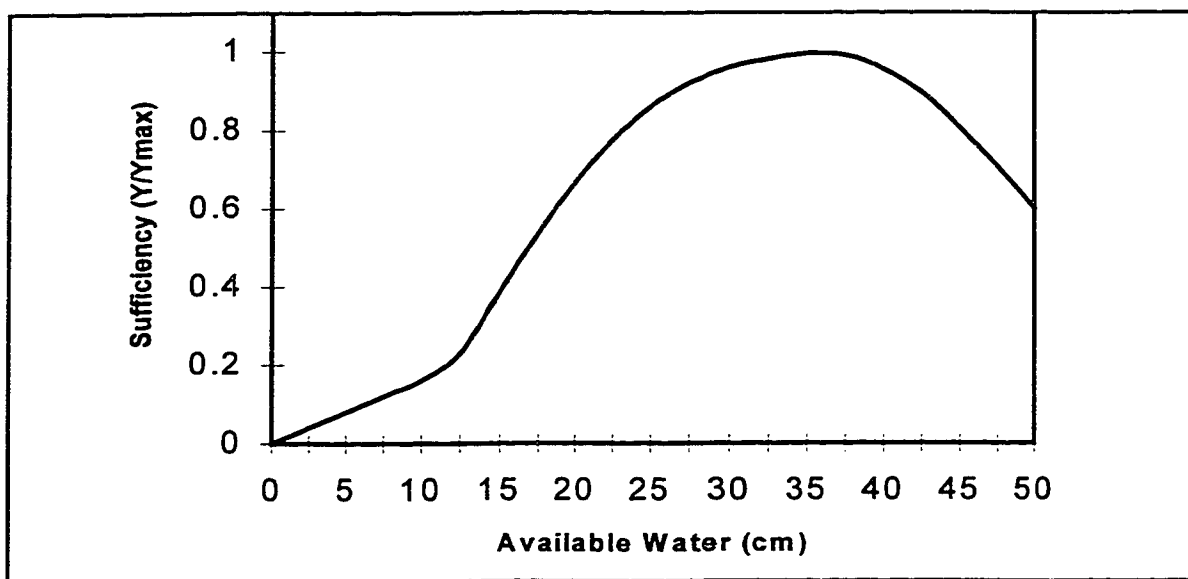


Figure 3.5. Boundary conditions (sufficiencies) for available water and wheat (from CSALE, 1997^b).

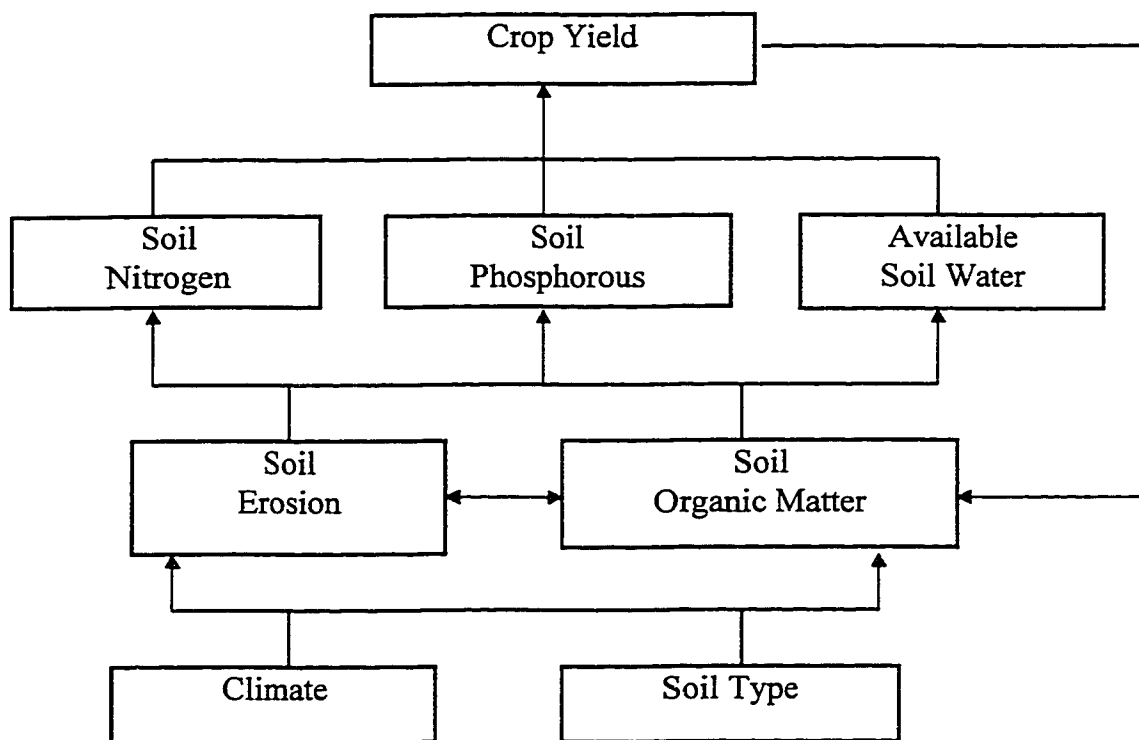


Figure 3.6. Schematic representation of relationships within the soils component of the SAM.

water equivalent). The proportion of precipitation that becomes available for crop growth is influenced by the infiltration rate, the recharge rate, a storage factor and crop growth in previous periods. These processes are influenced by soil texture and, for growing season precipitation, surface trash cover. The storage factor is based on the available water storage capacity of the soil, and is dependent on the clay and soil organic matter carbon content of the soil. Available water is used to calculate the water sufficiency parameter.

3.7.2 Soil Organic Matter Carbon Component

The soil organic matter carbon (SOMC) component of the model simulates the conversion of crop growth to crop residues and crop residues to either SOMC or CO_2 . Crop residues enter the soil as surface trash. The amount of carbon in the residue is determined by the type of residue (crop type), and grain yield, which reflects biomass production (CSALE, 1997^a). Plant residues are partitioned between surface trash, SOMC or CO_2 in the next time step. The rate of flow to these stocks is dependent on the rate of SOMC formation and the rate of decomposition, which in turn are dependent on such factors as soil N and soil water. The size of the SOMC stock is determined by the balance between the rate of loss of organic C, through residue decomposition and soil erosion, and the rate of gain of organic C from surface trash (CSALE, 1997^a).

3.7.3 Soil Nitrogen Component

“Farmers on Chernozemic soils know that, next to water, added nitrogen will give the largest yield response” (Greer et al., 1992). The quantity of soil N is controlled by SOMC levels and the rate of N turnover from the SOMC. The N turnover rate is a function of soil water content, soil temperature and soil thickness. The model predicts the quantity of N available to the crops as the sum of the internally derived soil N and the producer provided fertilizer N. Available N is used to determine the N sufficiency value for the model.

3.7.4 Soil Phosphorous Component

Phosphorous, with water and N, is the third of the inputs that are limiting to crop production in the soils model. Soil P is derived from inorganic (mineral) P, organic P, and fertilizer P sources. The turnover rate of inorganic P is determined by the soil clay content and organic P is supplied through the SOMC turnover rate, based on the assumption that SOMC contains a ratio of 1:10:100 for C:N:P. Approximately 25% of the fertilizer P is considered to be available for plant growth (Greer et al., 1992). Available P is used in the model to determine the P sufficiency value.

3.7.5 Soil Erosion Component

Within the soils model, erosion is set exogenously. The erosion parameter simply strips away an amount of the A and B horizon of the soil at each time step. The erosion process affects a number of the relationships within the model. Erosion causes a loss of SOMC and a concomitant decrease in available N and available P. That combined with reduced water holding capacity, and reduced surface trash result in lower water infiltration rates and less available water.

3.7.6 Crop Production Component

The soils component of SAM dynamically calculates crop yield. Yield is determined through a multiplicative relationship between the maximum yield for a given area and crop variety (Y_{max}), and the water sufficiency, N sufficiency and P sufficiency parameters discussed above. In turn, these sufficiency values are a function of crop yields in previous time periods.

3.8 HABITAT DYNAMICS AND BIODIVERSITY

A single parameter can not effectively serve as an indicator of biodiversity changes within an agroecosystem. The conceptual framework described here encompasses a series of parameters that indicate changes in different aspects of the

habitat complement of the simulated agroecosystem. These changes in habitat can be used to imply changes in biodiversity. Each of the parameters calculated by the ecological component of the model will be described separately.

3.8.1 Habitat Area Characteristics

Within an agroecosystem an important driver in the loss of biodiversity is the degradation and destruction of habitat. The agricultural landscape is heterogeneous with a range of habitat types, and habitat qualities being represented. Landscape ecology has developed models and processes that consider environmental heterogeneity or patchiness in spatially explicit terms. The absence of spatially explicit data in the present model preclude application of most of these indicators. However three of these indicators, or at least a reasonable proxy, can be calculated with the available land use output of the economic component of the SAM:

1. richness - refers to the number of different habitat types in a given area. It is generally assumed that landscapes with a greater number of habitat types available will have a greater level of biodiversity.
2. diversity (evenness) - the proportion of a given habitat type relative to the entire landscape. Diversity measures the extent to which a landscape is dominated by a few or many habitat types. A landscape that has a high level of richness but with one or two habitats dominating will be less attractive than a landscape with equivalent or slightly lower level of richness but with relatively equal quantities of each habitat.
3. context - refers to the probability that a given habitat type is adjacent to, or falls within another habitat type. For example, a wetland will provide less attractive habitat for many species if it is adjacent to, or falls within a fallow field, than a similar wetland that is adjacent to, or falls within a continuous crop field or a native pasture.

It should be noted that in isolation these indices provide only limited information on the habitat mosaic of a landscape. However, evaluating these indices together, and in

combination with the other parameters calculated by the ecological model, can provide insight into the habitat quality and quantity afforded by an agroecosystem.

3.8.2 Indicator Species Habitat

The biodiversity and habitat models discussed in the literature review highlighted the importance of characterizing landscapes not as a presence or absence of resources but as a continuum of resource quality (With et al., 1997). This characterization was echoed by the source - sink model (Pulliam, 1988). The primary difficulty with the source - sink framework is the large data requirements. For most Saskatchewan species this data does not exist. However it is possible to develop a very simple qualitative assessment of habitat patterned after the source - sink theoretical framework.

Within the SAM a series of indicator species are identified. Based on data availability the indicator species used in this framework are avian. The indicator species are selected partially on the basis of their preferred habitat, such that all habitats in the agroecosystem are represented, and partially based on the amount of information available on their habitat requirements and preferences. For each species the potential habitats available are ranked based on the attractiveness for breeding purposes under average conditions. This qualitative assessment simply ranks the top six breeding habitats for each species in terms of preference. It is assumed that under average conditions the top ranked habitat for each species is the most productive, and the bottom ranked habitat is the least productive of the preferred habitats. Relative habitat specialists prefer only two or three of the available habitats, with the remaining habitats providing no breeding value. The species that are habitat generalists prefer six or more habitats with varying levels of productivity. The habitat mosaic within a simulated agroecosystem is evaluated using this ranking to assess habitat value and changes.

3.8.3 Wetland Index

Wetlands provide a greater range of habitat types to more species than any other cover type in the prairie ecosystem. These habitat units come in a range of size and

permanence classes that are vitally important to many species at some point in their life cycle. The habitat and biodiversity component of the SAM includes a very simple process to account for changes in wetland conditions.

Within an agroecosystem the primary forces changing wetlands are: (1) conversion to agricultural production through draining and/or clearing; and (2) climatic variability. Within SAM land use data from the economic component of the model can be used to evaluate the pressures of agricultural production and land use change on wetlands in the landscape. The proportion of precipitation that serves as runoff to wetlands is partly determined by land use and cropping strategies. The rainfall and soil water infiltration parameters from the soils model can be used to imply changes in the water conditions of wetlands. Bethke and Nudds (1995) showed that precipitation over the previous two years is strongly correlated with current wetland conditions. Combining land use, precipitation and soil water simulation output, a rudimentary index of wetland conditions on the landscape can be developed. This wetland index indicates the wetland conditions, relative to average conditions, and thereby indicates whether wetland conditions are improving or degrading over a SAM simulation.

3.9 CONCEPTUAL MODEL LINKAGES

The conceptual framework described in this chapter encompasses a weakly integrated, dynamic model. The economic, soil and ecological component models have been identified as the critical components in this agroecosystem model. In order to represent the agroecosystem in a meaningful way the critical linkages and feedbacks that exist between the component models must be established (Figure 3.7).

The soil component produces dynamic yield data as a function of climate parameters and soil parameters that are in turn a function of the short term cropping history. Yield output from the soils component is an input into the economic component to calculate revenues relevant to the modeled annual crop rotations. The soil component calculates the ability of the soil to supply N which determines the N input requirement. Based on revenues, soil water, and soil N parameters the model will determine the

Based on revenues, soil water, and soil N parameters the model will determine the quantity of N fertilizer to add to the system. The quantity of N fertilizer is used in the economic model to calculate input costs. The soil model also calculates soil water infiltration parameters that are used by the ecological model to calculate runoff values and the wetland index.

The economic component calculates land use parameters representing the quantity of land allocated to annual crop rotations, tame forage and native grass in the agroecosystem. This information is used by the ecological model to calculate the habitat area indices and the indicator species habitat indices.

In summary the framework described can capture the “large outcomes” that occur as a consequence of “fine details” within a complex system (Ahl and Allen, 1996). For example, a small change in soil productivity can result in large shifts in land use patterns through changes in yield, revenue and future soil productivity. Within the SAM framework these changes are temporally dynamic and can be redirected by small shifts in system components within a simulation.

3.10 CONCLUSION

The conceptual framework described in this chapter ensures that the critical aspects of agroecosystem sustainability are explicitly captured. The weakly integrated structure enables the agroecosystem model to be constructed from the economic, soils and ecological models in a manner that can reflect co-evolutionary changes. The structure of the component models ensures that indicators of changes in natural and man-made capital stock are calculated. In addition, the linkages between these component parts enable the SAM to provide a qualitatively realistic representation of the agroecosystem. This framework can provide important insight into the interactions of a complex system and can highlight changes that occur relative to other simulation output of the same model. The next chapter will focus on the empirical details of tailoring this conceptual framework to a specific landscape application.

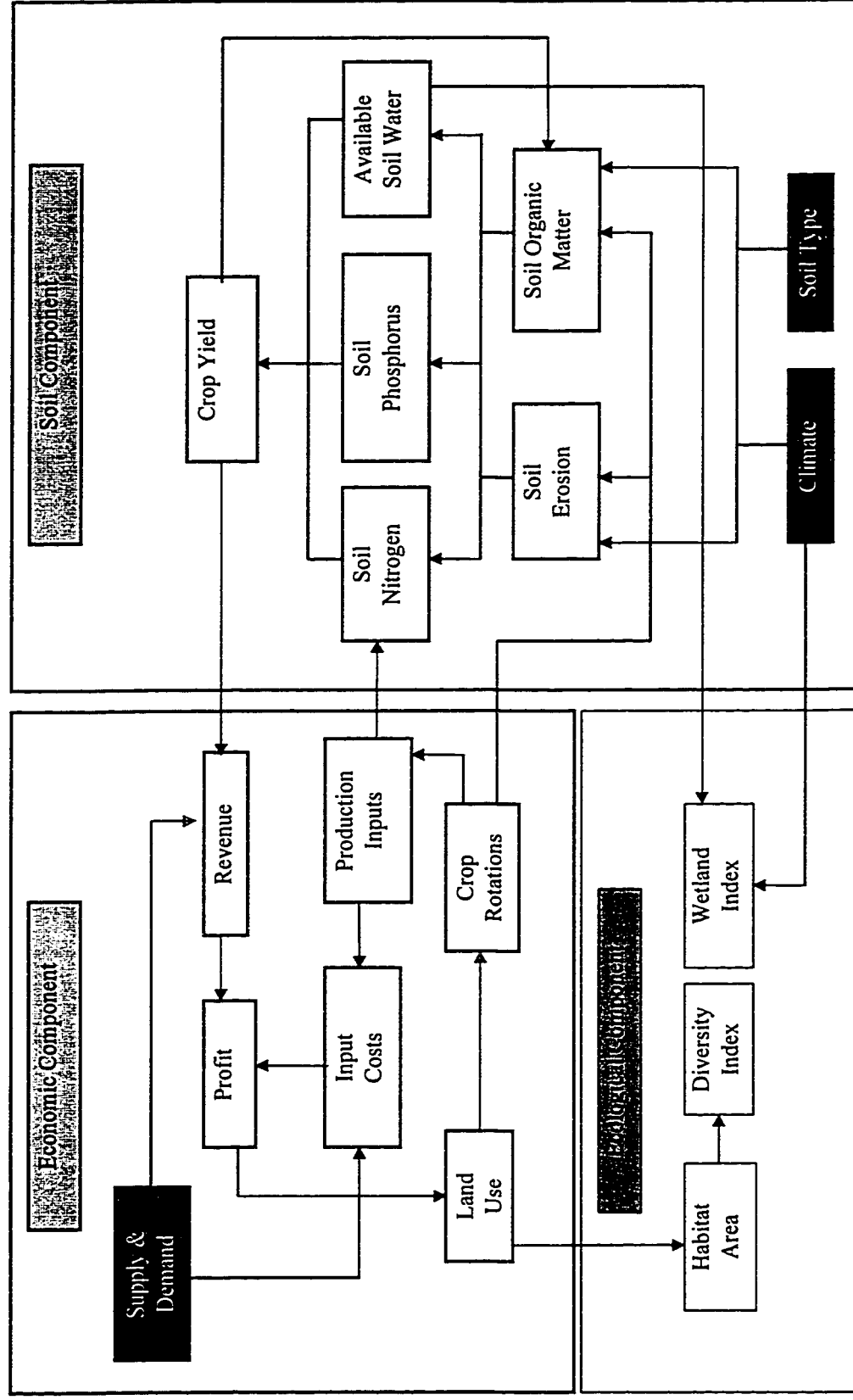


Figure 3.7. Schematic of the Agroecosystem Model structure showing the economic, soils and ecological components and the inter component linkages. Black cells identify exogenous parameters.

CHAPTER 4

ANALYTICAL FRAMEWORK

4.1 INTRODUCTION

The previous two chapters have provided the theoretical and conceptual foundation to evaluate agroecosystem sustainability. This framework incorporates dynamic linkages between soils, economic, and ecological modeling components, to simulate the important feedback relationships that exist within a complex system. The integrated model simulates changes in the quality and quantity of the natural and man-made capital stock caused by economic or environmental forces. This chapter provides detailed description of the data and processes incorporated to adapt the conceptual framework to specific ecodistricts. Section 4.2 describes the study area giving details of its economic and biophysical characteristics. Section 4.3 briefly discusses the modeling software used. The next sections describe the processes and data used to tailor the soils component (Section 4.4), the economic component (Section 4.5) and the ecological component (Section 4.6) of the conceptual framework to the study area. Section 4.7 provides a summary of the sustainability indicators to be used in the thesis analysis. Finally, a summary of the discussion in this chapter and the relevance of this discussion to the following chapters is presented in Section 4.8.

4.2 STUDY AREA

This study is part of the Prairie Ecosystem Study (PECOS). The study area for the present study is defined by the focus area of the PECOS project. The PECOS study

area is crop district 3BN, a 15,700 square kilometer area within the Mixed Grassland Ecoregion of south western Saskatchewan. The region is at the fringe of cultivated agriculture due to a moisture limitation with an evapotranspiration rate double the annual precipitation. The area contains a variety of landscapes and soils, with level productive clay soils in the north west to hilly, stony or sandy soils in pockets throughout the region. An important physical land form is the South Saskatchewan river which runs through the study area from west to east.

Four ecodistricts are the focus of the thesis: (1) Eston Plain - ecodistrict 808; (2) Beechy Hills - ecodistrict 813; (3) Antelope Creek Plain - ecodistrict 820; and (4) Gull Lake Plain - ecodistrict 824 (Figure 4.1). The wheat-summerfallow production system is dominant in this area with spring and durum wheat being produced on approximately 45 percent of the annual cropland, and summer fallow comprising another 40 percent of this land (Statistics Canada, 1996). Other significant annual crops include barley, oats, canola, lentils, peas, flax and canary seed. Other land use relevant to the present study are tame hay, improved pasture, native pasture, and other land.¹⁸ The land use categories that fall within the scope of farm land use decisions, and included in SAM are presented in Table 4.1. Table 4.2 contains area statistics for land use types that will not be directly affected by private land use decisions due to land ownership (community pastures), or physical characteristics (large wetlands and saline wetlands), but are significant land use categories within the ecodistrict. These lands are not included in the SAM model.

Soils in the study area range from a predominantly Brown Clay in the Eston Plain ecodistrict, to a mixture of Brown loam and sandy loam soils in the Gull Lake Plain ecodistrict. For a general description of the physical characteristics of each of the ecodistricts see Acton et al. (1998).

¹⁸ The census category other land, sometimes identified as unimproved land, captures privately owned non-farmland and can include farm yards including farm houses and outbuildings, shelterbelts, fencerows, ditches, wetlands, shrub land, and wood land.

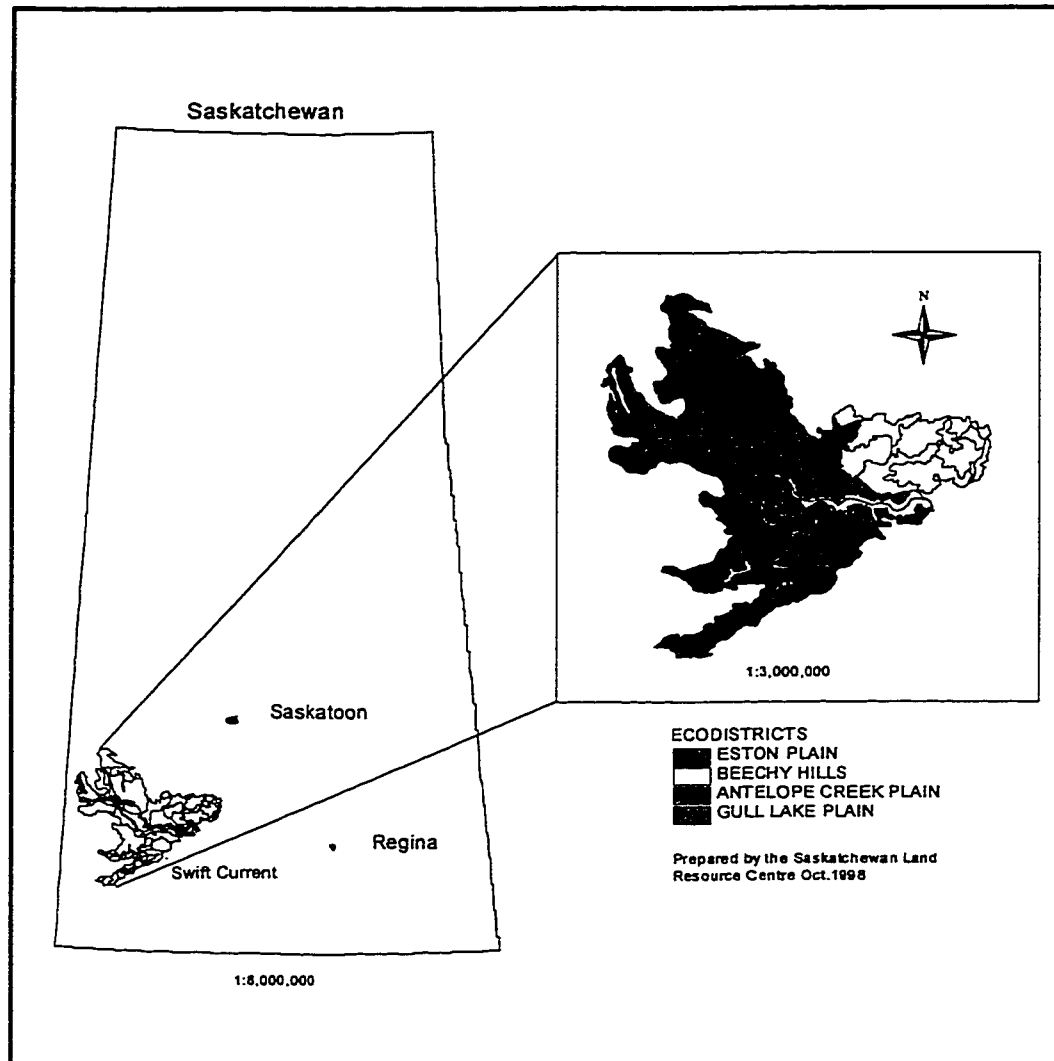


Figure 4.1. Thesis study area highlighting focus ecodistricts.

Table 4.1. Land use statistics for the four target ecodistricts.

Land Use (hectares)	Eston Plain	Beechy Hills	Antelope Creek Plain	Gull Lake Plain
Total Farmland ^a	783,571	218,000	230,560	131,912
Annual Crop ^a	370,604	91,104	101,481	42,291
Summer fallow ^a	272,442	46,038	70,168	26,095
Tame Hay ^a	4,379	4,295	6,496	5,005
Improved Pasture ^a	11,714	13,383	11,998	13,339
Native Pasture ^a	108,822	58,453	35,444	41,181
Other Land ^a	15,614	4,726	4,973	4,002
Ephemeral Wetland ^b	32,025	13,533	7,010	2,900
Seasonal Wetland ^b	588	219	999	378
Semi-perm. Wetland ^b	993	518	616	346
Permanent Wetland ^b	86	1555	1549	972

(^a Statistics Canada, 1996; ^b Saskatchewan Soil Survey, 1998)

Table 4.2. Ecodistrict land use not included in Agroecosystem Model land stock.

Land Use (hectares)	Eston Plain	Beechy Hills	Antelope Creek Plain	Gull Lake Plain
Community Pastures ^c	29,235	51,930	0	4,144
Lakes ^d	2,476	3,458	647	297
Saline Wetlands ^d	3,891	15	4,490	3,131

(^cSaskatchewan Institute of Pedology, 1979; ^dSaskatchewan Soil Survey, 1998)

4.3 MODELING ENVIRONMENT

The SAM was built entirely within a single modeling environment to facilitate the dynamic exchange of information between the component models. All of the model components were built using STELLA® Research, version 5.0, graphical programming language, on a Windows95® platform. The STELLA modeling sectors and equations for the SAM are included in this thesis as Appendix A.

4.4 EMPIRICAL SOIL MODELING

The soils model in the SAM contains procedures to simulate wheat, canola and pea crops. In addition, the effect of summer fallow on soil water, N and P was also developed in the model. These procedures were assembled to simulate changes in soil

health parameters, and the yield effect of different cropping rotations. The following rotations were identified as agronomically appropriate for the study area:

1. a cereal-based crop-fallow system with a relatively high frequency of tillage, and low input requirements (wheat-fallow (WF)).
2. a cereal-based crop-fallow system with a lower frequency of tillage, and low input requirements (wheat-wheat-fallow (WWF)).
3. a longer rotation system of cereal and pulse crops with fallow every three years and a combination of chemical and tillage fallow (wheat-fallow-peas (WFP)).
4. a continuous cropping system based on a cereal, oilseed and pulse crop rotation with minimum or zero tillage (wheat-canola-peas (WCP)).

The procedures for canola, peas and summer fallow were developed by Dr. M. Boehm (CSALE, 1997^a; CSALE, 1997^b).

The soils model incorporates landscape relevant baseline data to ensure that simulations are consistent with local environmental constraints (Table 4.3). The data is specific to the target ecodistrict (soil texture, climatic parameters) or the Brown soil zone (initial stocks of SOMC and surface trash, initial depth of the soil A and B horizons). Within the model a growing season and non-growing season precipitation value is randomly selected, at each time step, from a gamma distribution for precipitation, with ecodistrict specific mean and standard deviation.¹⁹

The soil model output includes soil quality parameters and crop yields within the relevant rotation and environmental context. For example, the model calculates separate wheat yields for wheat grown in each of the WF, WWF, WFP and WCP rotations. Rotation relevant output ensures that soil model parameters will provide meaningful links to the other components of the model.

¹⁹ A gamma distribution is a skewed, continuous distribution. The level of skewness can be manipulated by changing two parameters. For a description of the gamma distribution see Mood et al.(1974)

Table 4.3. Biophysical and climatic characteristics of target ecodistricts.

Parameter	Eston Plain	Beechy Hills	Antelope Creek Plain	Gull Lake Plain
Soil Texture	Brown clay	Brown loam	Brown loam	Brown loam/Brown sandy loam
Growing Season	19	18	19	19
Precipitation (cm) ^a				
Non-Growing Season	15	14	17	18
Precipitation (cm) ^b				
Growing Degree Days	1,479	1,507	1,459	1,397
Mean Daily	1	2	3	4
Temperature (days) ^c				
Initial Solum (cm) ^d	60	60	60	60
Initial SOMC (t/ha) ^e	50,000	50,000	50,000	50,000
Initial Surface Trash (kg/ha) ^f	4,200	4,200	4,200	4,200

^a Growing season precipitation represents the precipitation received between May and July.

^b Non-growing season precipitation represents the snow-melt equivalent precipitation received between August and April.

^c Mean daily temperature represents the number of days within a growing season where the mean daily temperature exceeds 24 degrees celcius.

^d Solum represents the depth, in centimetres, of the A and B horizons of the soil, a value specific to the soil zone.

^e SOMC content of the soil to the depth of solum, in tonnes per hectare. This value is specific to the soil zone.

^f Surface trash is the amount of residue carbon found on the surface of the soil, in kilograms per hectare. This value is specific to the soil zone.

4.5 EMPIRICAL ECONOMIC MODELING

The economic component was adapted for agroecosystem simulation by incorporating production cost data specific to the target ecodistricts. This adaptation was necessary for the model to incorporate soils component output, and generate output that is consistent with the other model components. The following sections present the analytical details of adapting the fixed proportions framework to simulate the target agroecosystems. The objective function within the economic component of the SAM assumes that agricultural firms are myopic decision makers with respect to soil quality in future time periods, as influenced by present management strategies. In other words farmers assume that the user cost of soil degradation is zero.

4.5.1 Output Relationships

The economic model in SAM is structured to provide output based on the four crop rotations (WF, WWF, WFP and WCP). The average gross revenue for each rotation (\$/hectare) is calculated based on market commodity prices and yield output from the soil component. For example, the gross revenue for the WFP rotation is calculated:

$$\eta_{wfp} = \frac{[(p_w * t_w) + (p_p * t_p)]}{3} \quad (4.1)$$

where:

- η_{wfp} -gross revenue for WFP (\$/hectare)
- p_w - market price for wheat (\$/tonne)
- p_p - market price for peas (\$/tonne)
- t_w - wheat yield in WFP rotation (tonne/hectare)
- t_p - pea yield in WFP rotation (tonne/hectare)

The market price for wheat (\$167.00/tonne), canola (\$351.00/tonne), and peas (\$181.00/tonne) are three year averages based on crop years 94-95, 95-96 and 96-97. Wheat prices are based on the Canadian Wheat Board pool return, basis Saskatoon (Saskatchewan Agriculture and Food,1997^a). Canola prices are weighted average prices for the province (Saskatchewan Agriculture and Food,1997^a). Pea prices were obtained from the “1997 Specialty Crop Report” (Saskatchewan Agriculture and Food, 1997^b). Crop prices are fixed for the duration of the simulation. Wheat, canola and pea yields, in rotation context, are generated at each time step by the soil component of the SAM.

4.5.2 Input Relationships

Within the economic component crop production inputs are partitioned into land, fertilizer, chemicals, and “other inputs” categories. The analytical details of deriving the supply relationship for each of these inputs will be discussed separately.

4.5.2.1 Land Input

A linear upward sloping land supply function, as described in the previous chapter, implies that the marginal cost of bringing additional land into annual crop production increases in a linear fashion. However, the total stock of land may more appropriately be characterized as having distinct uses and distinct conversion costs. To reflect these conversion cost characteristics it was assumed that the entire annually tilled land stock in the landscape is allocated to the four annual crop rotations. Further, it was assumed that the annual crop land stock can only be increased by converting tame forage or native grass lands, thereby decreasing the land stock in these categories. Based on these assumptions the land supply curve in the model is described as perfectly inelastic with a quantity intercept equal to the annual crop land stock in the ecodistrict at that point in time. When land is allocated from some other use to annual crop production, or converted from annual crop to perennial forage, there will be a parallel shift in the land supply curve. The procedures developed for modeling land conversion are discussed in detail in later sections of this chapter.

The initial stock of annually cultivated land is the sum of the “annual crop” and “summer fallow” census categories (Table 4.1). The proportion of the annually cultivated land stock allocated to each of the rotations, in each of the ecodistricts, is presented in Table 4.4. The land allocation in Eston Plain is consistent with actual land use, based on 1996 census statistics (Statistics Canada, 1996). However, initial runs of SAM revealed that low and variable canola and pea yields in the other three ecodistricts resulted in low revenues for the WCP and WFP rotations in many years. Within the fixed proportions economic framework the initial “other inputs” supply relationships are derived based on gross revenues, and fertilizer, chemical and land input costs. The combination of low revenues for WCP and WFP and fixed initial fertilizer, chemical, and land costs resulted in the model deriving “other input” supply relationships that imposed unrealistically low “other inputs” costs for these two rotations. The very low “other input” costs resulted in the model calculating a large derived demand for land for the WCP and WFP rotations and an allocation of large quantities of land to these two rotations.

Table 4.4. Proportion of initial annually cultivated area dedicated to each rotation in simulation year zero.

Rotation	Eston Plain	Beechy Hills	Antelope Creek Plain	Gull Lake Plain
WF	0.45	0.55	0.55	0.57
WWF	0.35	0.40	0.40	0.40
WCP	0.05	0.02	0.02	0.01
WFP	0.15	0.03	0.03	0.02

In order to compensate for the modeling difficulties caused by very low WCP and WFP revenues a minimum cost of “other inputs” was set for the two rotations in the three ecodistricts (see section 4.6.2.4). In addition, small initial quantities of land were allocated to the WCP and WFP rotations in the Beechy Hills, Antelope Creek Plain and Gull Lake Plain ecodistricts (Table 4.4). Maintaining the WCP and WFP rotations in these ecodistricts, albeit on a reduced land base, ensured that these rotations are production options under favourable economic and/or environmental conditions.

The initial market clearing land price used for all ecodistricts in the model (\$49.42/ha) is based on the “land investment cost” for the Brown soil zone (Saskatchewan Agriculture and Food, 1998) (Table 4.5).

4.5.2.2 Fertilizer Input

It is assumed that producers are price takers in the fertilizer market resulting in a perfectly elastic supply function in all rotations. The intercept value represents the cost per unit (\$/ha) of fertilizer input for that rotation (Table 4.5). Fertilizer price and application rate data, relevant to the Brown soil zone (Saskatchewan Agriculture and Food, 1998), were used to calculate the fertilizer input cost. Nitrogen requirements were assumed to be met by 46-0-0 at a price of \$247.00 per tonne (\$537.00 per tonne actual N). Phosphorous requirements were met by 12-51-0 at \$375.00 per tonne (\$735.00 per tonne actual P). Saskatchewan Agriculture and Food recommended fertilizer application rates in the Brown soil zone are: P on all crops - 22.4 kg/ha; N on stubble seeded pulse

Table 4.5. Initial input cost statistics for the four rotations

Input	WF	WWF	WFP	WCP
Land (\$/ha)	49.42	49.42	49.42	49.42
Fertilizer (\$/ha)	14.27	24.08	20.89	36.62
Chemicals (\$/ha)	34.65	37.60	60.37	65.68

(Saskatchewan Agriculture and Food,1998).

crops - 11.2 kg/ha; N on fallow seeded pulse crops - 4.5 kg/ha; N on stubble seeded crops - 50.7 kg/ha; N on fallow seeded crops - 22.5 kg/ha. The calculation of fertilizer input cost for the WFP rotation is shown in equation 4.2.

$$\delta_{wfp}^F = \frac{\left[(N_w + N_p) * p_N \right] + \left[(Ph_w + Ph_p) * p_{Ph} \right]}{3} \quad (4.2)$$

where:

- δ_{wfp}^F - fertilizer input cost for WFP (\$/ha)
- N_w - nitrogen input requirement of wheat (tonnes/ha)
- N_p - nitrogen input requirement of peas (tonnes/ha)
- Ph_w - phosphorous input requirement of wheat (tonnes/ha)
- Ph_p - phosphorous input requirement of peas (tonnes/ha)
- p_N - market price of nitrogen (\$/tonne)
- p_{Ph} - market price of phosphorous (\$/tonne)

The fertilizer input requirement within SAM is determined by how much of the nitrogen and phosphorous requirement of the crops is met by available soil nitrogen and phosphorous. The amount of these nutrients available for the crop is equal to the amount available in the soil, plus the amount added as fertilizer. Soil nutrient quantities are a function of soil quality, which is calculated dynamically by the soils model. Within SAM phosphorous is added at a constant rate in each time step. The nitrogen fertilizer rate is calculated within the model. At each time step the model sets nitrogen fertilizer rates such that nitrogen is non-limiting to crop growth, in other words soil nitrogen plus fertilizer nitrogen equals 100% of the nitrogen sufficiency. The nitrogen fertilizer rate is then adjusted according to:

1. previous year net revenue - if net revenue in the previous year was below average the rate of nitrogen fertilizer used in that rotation is reduced. This is based on the assumption that low economic returns will limit investment in production inputs.
2. spring soil moisture - it is assumed that soil moisture is the most limiting factor to crop production on the prairies and with poor soil moisture nitrogen fertilizer inputs will decrease. Poor soil moisture conditions in the beginning of the growing season will result in agricultural firms reducing their financial risk by decreasing input costs. Very good soil moisture conditions will result in increased soil nitrogen additions (Boehm and Belcher, 1998).

The dynamic nitrogen fertilizer application rates result in annual changes in nitrogen fertilizer input cost as a function of dynamic climatic and soil quality parameters.

4.5.2.3 Chemical input

Within SAM chemical inputs include herbicide, insecticide, fungicide, and other chemical input requirements for each of the crops in the rotation. It was assumed that producers are price takers in the chemical market. The cost per unit of chemical input (\$/ha) for each rotation is reported in Table 4.5. The calculation of the chemical input cost for WFP is shown in equation 4.3.

$$\delta_{wfp}^{Ch} = \frac{(C_{sw} + C_f + C_{pf})}{3} \quad (4.3)$$

where:

- δ_{wfp}^{Ch} - chemical input cost for WFP (\$/ha)
- C_{sw} - chemical cost for wheat on stubble (\$/ha)
- C_f - chemical cost for chem-fallow (\$/ha)
- C_{pf} - chemical cost for peas on fallow (\$/ha)

The chemical input costs used for these cost calculations are as follows: wheat on stubble (\$43.51/ha); wheat on fallow (\$32.74/ha); canola on stubble (\$52.46/ha); peas on stubble (\$101.06/ha); peas on fallow (\$101.06/ha); chem-fallow (\$36.55/ha) (Saskatchewan Agriculture and Food, 1998).

4.5.2.4 Other Inputs

The “other input” category captures all those inputs that are not represented by the land, fertilizer, and chemical input markets. “Other inputs” costs include machinery, fuel, buildings, management skill, agronomic constraints, risk etc. These inputs are assumed to be somewhat specific to each rotation, and as a result cannot be switched between rotations at zero cost. This relationship is ensured by industry scale independent “other input” supply relationships for each rotation. This assumption is critical to the model for if all inputs were freely transferable between rotations, all inputs would be used in the production of only the most profitable rotation. It is assumed that the “other input” supply relationship is upward sloping which imposes an economic constraint on the quantity of “other inputs” that can be purchased for each rotation, and thereby constrains the quantity of land allocated to each rotation.

To derive the “other input” supply function, initial price and quantity values for “other inputs”, and an “other input” supply elasticity are required. Initial price and quantity values are calculated based on the assumption that initial land use conditions are at equilibrium, the fixed proportions model assumptions of perfect competition (equation 3.3) and fixed input and output proportions (equation 3.2).

To calculate “other input” supply elasticity values the fixed proportions framework was built into a Microsoft® Excel spreadsheet as a series of closed form equations that solve for the equilibrium quantity of output and inputs dedicated to each rotation. This structure facilitates the calculation of own price and cross price area response elasticities for wheat, canola and peas. Using the “solver” function in Excel the system was set up to find “other input” supply elasticities for the four rotations that provide area response elasticities consistent with published values. However, published area response elasticities are: (1) based on western Canadian, or Saskatchewan data, and thereby reflect a much larger spatial scale than the present study; and (2) are based on historical data and as such do not reflect current technology.²⁰ In addition, crops within

²⁰ Between 1980-81 and 1996-97 crop years the area dedicated to pea production in Saskatchewan has increased by 3000 percent (12,140 hectares in 1980-1981 to 364,225 hectares in 1996-97) which implies that area response elasticities for these crops will also have changed over this time period.

SAM are tied to rotations which constrains the ability of the model to provide area response elasticities equivalent to literature values. The wheat own price elasticity of 0.5 was considered appropriate since similar values have been reported in studies using data sets from different time periods, and different spatial scales (Watson, 1995; Clark and Klein, 1992; Schmitz, 1968). The “solver” procedure was set up to provide a wheat own price elasticity of 0.5, and other crops own price elasticities were constrained to be positive. The cross price elasticities were not constrained to be negative due to the limitations imposed by crops being tied to rotations. Using this procedure the “other input” supply elasticities were derived (Table 4.6).

A minimum price of “other inputs” was imposed in SAM to correct for the cost calculation problem outlined in section 4.5.2.1. Within the model a minimum price of “other inputs” was imposed such that “other input” supply relationships were calculated only if rotation revenues were sufficient to cover total input costs, including the imposed minimum “other input values. The values selected (WCP = \$135.00/ha; WFP = \$120.00/ha) are based on Saskatchewan Agriculture and Food cost of production statistics (1998).

Table 4.6. "Other Input" supply elasticities for the Brown soil zone.

Crop Rotation	Other Input Supply Elasticity
WF	1.5
WWF	3.0
WFP	5.0
WCP	17.0

4.5.3 Other Land

A procedure was developed, as an extension of the annual crop economic framework, to simulate the economic relationships driving land use decisions on land that is not annually cultivated. This procedure is comprised of two parts focusing on: (1) tame forage land (tame hay and improved pasture); and (2) native land (native pasture, shrub land, and wetlands).

4.5.3.1 Tame Forage Land

To simulate land use switching the relevant BUT and SAT values, as discussed in section 3.6.2, were empirically derived. A Microsoft® Excel spreadsheet model was employed to calculate the BUT and SAT threshold values for Saskatchewan. The calculations employed 1971 to 1996 time series data (Saskatchewan Agriculture and Food, 1996). All prices are adjusted to 1995 dollars using a GDP price index. The steps used in the calculation closely follow the procedure described by Gray et al. (1993).

1. Calculate a net income stream (per hectare) for grain production. Since production expenditure data does not differentiate between expenses in livestock and crop production a proxy total expenditure on crops (TEC_t) at time t was calculated:

$$TEC_t = \left(\frac{\text{Total} \cdot \text{Crop} \cdot \text{Receipts}_t}{\text{Total} \cdot \text{Cash} \cdot \text{Receipts}_t} \right) * \text{Gross} \cdot \text{Operating} \cdot \text{Expenses}_t \quad (4.4)$$

Net income per hectare for grain (G_t) could then be calculated:

$$G_t = \frac{(\text{Total} \cdot \text{Crop} \cdot \text{Receipts}_t - TEC_t)}{\text{Total} \cdot \text{Annual} \cdot \text{Crop} \cdot \text{ha}_t} \quad (4.5)$$

Where total annual crop ha is the sum of census categories cropland and summer fallow hectares, netting out tame hay and improved pasture hectares.

2. Calculate a per hectare net income stream for hay (H_t). It was assumed that production costs comprised two thirds of gross hay revenue:

$$H_t = \frac{(\text{Average} \cdot \text{Hay} \cdot \text{Yield}_t * \text{Average} \cdot \text{Hay} \cdot \text{Price}_t)}{3} \quad (4.6)$$

3. Calculate future income streams for grain and hay. To generate a stream of future income, the statistical characteristics of past income must be considered. It was assumed that hay and grain incomes do not move independently. The income regressions were linked such that a large separation between G_t and H_t will result in the incomes tending to move together to decrease separation. By including this

characteristic in the estimation process the two income values will not move in complete independence. Based on the income streams calculated in steps one and two, the following regressions were performed:

$$G_{t+1} = \psi + \xi(G_t - \bar{G}) + e_t \quad (4.7)$$

$$D_{t+1} = \nu + \tau(D_t) + \varepsilon_t \quad (4.8)$$

where:

$$D_t = G_t - H_t \quad (4.9)$$

\bar{G} - mean of the grain net income stream.

ϕ, ξ, ν, τ - regression coefficients.

The mean and standard deviation of the two normally distributed error terms (Table 4.7) were used to simulate 50 year net income streams for grain and hay crops. A total of 500 such sets were simulated based on these statistics.

4. Calculate SAT and BUT values that optimize the total net income over the 500 sets of 50 year income streams. The Excel spreadsheet model calculates D_t each year. The D_t value is evaluated in terms of the SAT or BUT, depending on the current land use, netting out appropriate establishment, re-establishment and conversion costs.

The framework described here is depicted in flow chart form in Figure 4.2. The figure indicates that at each time step the model calculates the optimum land use, based on the SAT and BUT values, and the management costs associated with each action. The land use at each time step contributes appropriate net income values to total income for the 50 year simulation. In establishment years it is assumed that hay land returns only 50 percent of the non-establishment year hay income. This assumes that a grain cover crop is used in the hay establishment year.²¹ The optimal BUT and SAT levels are selected based on the greatest aggregate net income for the 500 data sets.

²¹ A cover or companion crop is an annual crop, that is seeded with the forage crop to provide wind erosion protection and income in forage establishment years (University of Saskatchewan, 1987)

Table 4.7. Statistical characteristics of income regression error terms.

Error Term	Mean	Standard Deviation
e_t	1.54 E -14	22.7904
ε_t	1.48 E -15	24.63521

The actual SAT and BUT values calculated using the above process are based on provincial average data. As a result, these values are inappropriate for a specific application to ecodistricts in the southwest part of Saskatchewan. The calculated SAT and BUT interval described above was maintained, while the actual threshold levels used in SAM were derived based on model simulation output for each of the target ecodistricts. The threshold levels were selected based on the values that maintained simulated tame forage land area in the ecodistrict at levels approximately consistent with census statistics (Table 4.8). The input and conversion costs used in the threshold calculations were based on values published by Saskatchewan Agriculture and Food (1995) for the Brown soil zone.(Table 4.8).

Within SAM land rent, is used as a proxy for G_t . The H_t value is calculated using a fixed hay price (\$60.00 / tonne) and dynamic hay yield. Hay yield (tonnes/ha) is calculated by the model, at each time step, using a linear regression with precipitation (generated by the model) as the independent variables.²² Hay yield regressions based on simulated yields (1962 - 1996) on Brown clay, Brown loam, and Sandy loam soils using Swift Current precipitation data (1962 - 1996). The linear hay yield relationship is

²² Hay yield estimates based on rainfall were calculated using the GRASSGRO® pasture growth model. GRASSGRO uses mathematical models to assess how weather, soils and management factors combine to affect pastoral productivity (Cohen et al., 1995). The model, developed in Australia, has been validated for Saskatchewan conditions and contains site specific soil texture and historical precipitation data for a number of sites in Saskatchewan including the Swift Current area.

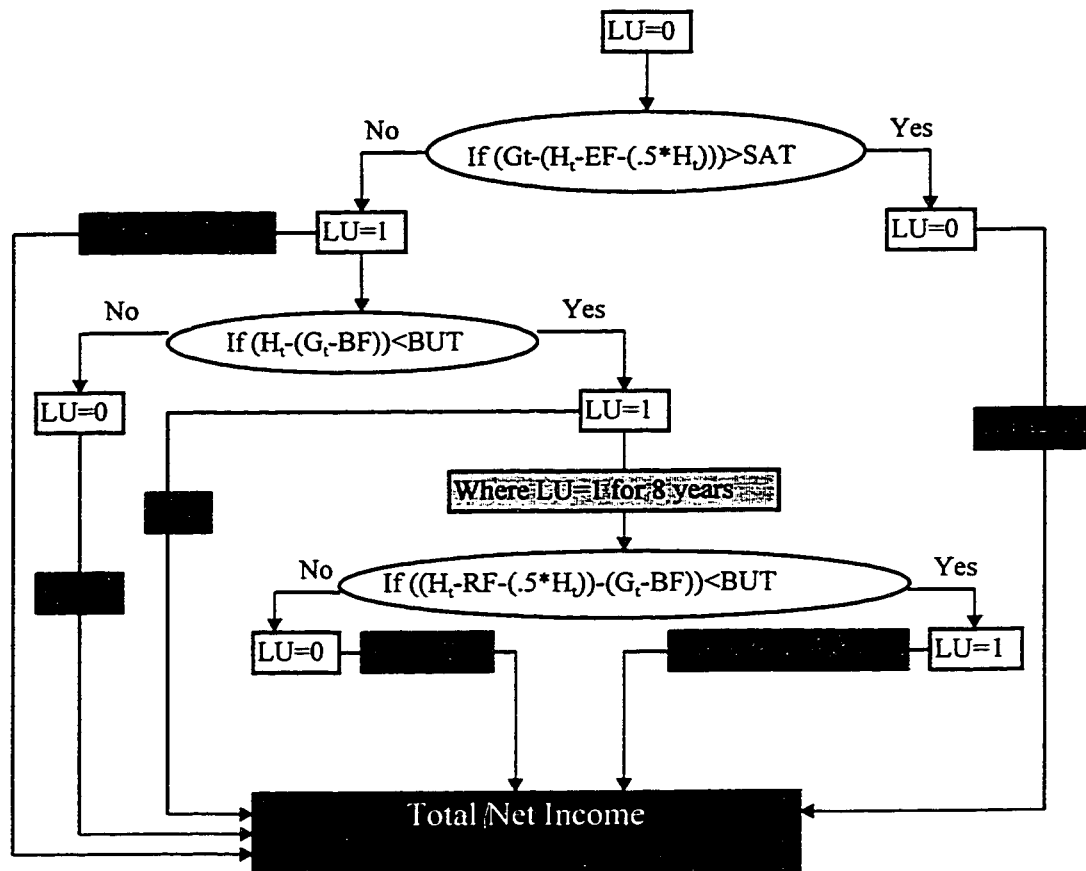


Figure 4.3. Flow chart depicting the procedure used to calculate total net income from grain and hay production, based on set SAT and BUT levels.

Where:

- LU=0 - land use is grain production
- LU=1 - land use is hay production
- G_t - net grain income in year t (\$/ha)
- H_t - net hay income in year t (\$/ha)
- EF - hay establishment cost (\$/ha)
- BF - hay breaking cost (\$/ha)
- RF - hay re-establishment cost (\$/ha)

Table 4.8. Hay and grain management costs used in the calculation of net income and the empirically derived provincial average, and actual modeled SAT and BUT levels.

Forage Establishment Cost	\$36 / ha
Forage Re-establishment Cost	\$50 / ha
Forage Breaking Cost	\$17 / ha
Average Set-Aside Threshold (SAT)	-\$77 / ha
Average Break-Up Threshold (BUT)	-\$7 / ha
SAT ecodistrict (EP)	-\$50/ha
BUT ecodistrict (EP)	+\$20/ha
SAT ecodistrict (BH, ACP, GLP)	-\$40/ha
BUT ecodistrict (BH, ACP, GLP)	+\$30/ha

shown in equation 4.10, and the calculated coefficients are presented in Table 4.9.

$$Y_t^h = c + \mu(gs_t) + \nu(ws_t) \quad (4.10)$$

where:

- Y_t^h - tame hay yield in time period t .
- gs_t - growing season precipitation in time period t .
- ws_t - non-growing season precipitation in time period t .
- c - empirically derived constant.
- μ, ν - empirically derived coefficients.

At each time step in SAM, H_t is calculated as one third of the product of hay yield and hay price. To calculate D_t the model uses a moving two year average of net hay income to soften the effect of one year revenue spikes. The D_t value is then evaluated in terms of SAT and BUT to determine if land will be converted between hay production and grain production based on the highest net economic returns to the land.

4.5.3.2 Native Land

Native lands include grass, shrubs, bush and wetlands of different depths and permanence. As a result, conversion of these lands to cultivation will not impose a constant, or linearly increasing marginal conversion cost on the farmer. Van Kooten,

Table 4.9. Empirically derived hay yield coefficients for Brown clay, loam and sandy loam soil textures in the study area.

Coefficients	Brown Clay Soil	Brown Loam Soil	Brown Sandy Loam Soil
c	1.16 (t-stat - 2.71)	1.31 (t-stat - 2.38)	0.48 (t-stat - 1.20)
μ	0.04 (t-stat - 2.50)	0.03 (t-stat - 1.53)	0.08 (t-stat - 5.05)
ν	0.05 (t-stat - 2.94)	0.06 (t-stat - 3.01)	0.04 (t-stat - 2.40)
R-squared	0.29	0.24	0.46

(1993) developed a cost relationship for the conversion of native land, on a single farm, that is a function of the amount of native land remaining on the farm and the quantity of land converted in that year. The marginal cost function that is derived from van Kooten's cost function is elastic when stocks of native land are large but becomes increasingly inelastic as stocks of native land approach zero. When the stock of marginal land is large, the marginal unit of land to be converted to cultivation will likely be of similar quality and impose similar conversion costs, as the average hectare of land in the cultivated land stock. However, as the stock of native land decreases the quality of the marginal unit of native land decreases, and/or the cost of conversion will increase. For example, in most agricultural regions there is a portion of the stock of native land that is of relatively good quality for annual crop production. A small increase in returns to cultivated land will make this native land economically attractive to convert to annual crop production. In contrast, most agricultural regions contain native land that would never be converted to cultivation, or converted only when the economic returns to cultivated land increase drastically. Examples of this latter category are permanent wetlands, heavily treed land, hilly land and soils of very poor quality.

The native land supply curve developed for the present study resembles the function in the van Kooten model. However, since the spatial scale of SAM is regional, rather than a single farm, the quantity of land converted in a given time period is less constrained. The supply curve developed for the SAM is a function of the returns to cultivated land (land rent) and the remaining stock of native land in the target ecodistrict.

The mathematical form used to describe the native land supply relationship is shown in equation 4.11.

$$Q_t = \left(\frac{\chi}{\Delta P^\omega} \right) * Q_{t-1}^\circ \quad (4.11)$$

where:

Q_t - quantity of native land converted to cultivation in time period t

Q_{t-1}° - stock of native land in time period $t-1$

ΔP - change in land price between period $t-1$ and period t

χ - empirically derived constant (0.025)

ω - empirically derived constant (-1.0)

Equation 4.11 describes a land supply function that is very elastic when Q_{t-1}° is large, and becomes increasingly inelastic as the stock of native land approaches zero. An example of this supply function, representing a landscape with 10,000 ha of native land at $t = 0$, and an initial equilibrium land price of \$50.00/hectare, is shown in Figure 4.3. In this example, when $Q_{t-1}^\circ = 9,500$, the supply elasticity = 25, when $Q_{t-1}^\circ = 2000$, the supply elasticity = 0.72.

The native land supply function described above is used within SAM to simulate the conversion of native land to cultivated land. When land rent (as calculated by the model) exceeds the previous maximum land rent there is a conversion event and some quantity of native land (as calculated using equation 4.11) becomes a permanent part of the cultivated land stock (annual crop or tame forage). Due to the assumption of permanent loss of native land stock with each conversion event the demand for native land can only move upwards along the native land supply curve and can not move back towards the origin. It is assumed that conversion of native land to cultivation is a significant management decision. Therefore, the conversion procedure is constrained in two ways to ensure that a large conversion event does not occur as a result of a short-term land rent spike:

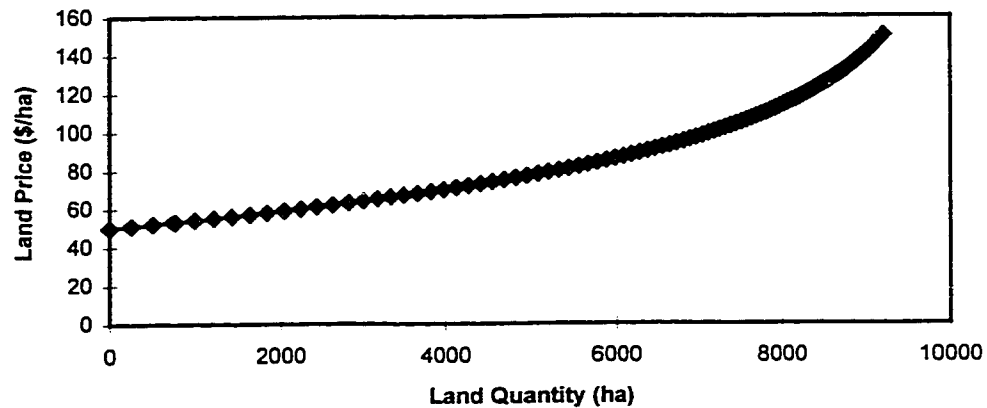


Figure 4.3. Native land supply function representing an hypothetical landscape containing 10,000 ha of native land at $t=0$.

1. The land price value (P) used in equation 4.11 is a moving three year average of land rent. This also ensures that the land use response to an increase in land rent will first convert tame perennial forage to annual cultivation before converting native.
2. the quantity of land converted in one period is constrained to be less than 10 percent of the initial native land stock.

The output from this component of the model is the quantity of native land remaining in the target landscape.

4.6 EMPIRICAL ECOLOGICAL MODELING

The three land use categories within the economic component of SAM are: (1) annual cropland; (2) tame forage; and 3) native land. Within the ecological component these land use categories are divided into nine separate habitat categories: (1) fallow; (2) cropland; (3) tame hay; (4) improved pasture; (5) native grass; (6) shrub land; (7) seasonal wetland; (8) semi-permanent wetland; and (9) permanent wetland. Fallow and cropland habitat area are derived directly from the annual cropland stock based on the area of each of the annual crop rotations. Tame hay and improved pasture habitat area are derived from the tame forage stock based on the proportion of each of these habitat

categories in the initial tame forage stock. Native grass (native pasture), shrub, seasonal wetland, semi-permanent wetland and permanent wetland habitat areas are equivalent to the proportion of each reported in the initial native land stock. These procedures assume that forces changing land use are equivalent on all habitats within the land use category. For example, if native land was initially comprised of 50 percent native grass, 20 percent shrub, and 10 percent wetland, this proportion would remain constant under all changes in native land area. This is not a realistic assumption since certain categories of native land are much more expensive to convert (i.e. permanent wetlands) than others (i.e. native pasture), and would not be as affected by land use change forces. However, the SAM framework did not facilitate a more detailed procedure and the output provided can provide insight into the potential forces being exerted on the habitat types.

4.6.1 Quantitative Habitat Indices

Habitat richness is simply the number of habitat types in the ecodistrict. The habitat richness algorithm counts the number of habitats present in the ecodistrict at each time step.

Relative habitat abundance is the proportion of a given habitat area relative to the total landscape area. Based on the land use stock output from the economic model relative habitat abundance is calculated at each time step as shown in equation 4.12.

$$A_i = \left(\frac{h_i}{\sum_{i=1}^k h_i} \right) \quad (4.12)$$

where:

- A_i - relative habitat abundance for habitat i
- h_i - total hectares of habitat i
- k - the total number of habitat types observed

In equation 4.12 the denominator represents the total relevant area of the target ecodistrict, such that $\sum_{i=1}^k A_i = 1$. As discussed in the conceptual framework chapter, relative habitat abundance can be used to assign a habitat context.

Habitat diversity measures the degree to which an ecodistrict is dominated by a few or many land uses. This index is calculated at each time step by the ecological model using equation 4.13.

$$D = -\sum_{i=1}^k (A_i) \log(A_i) \quad (4.13)$$

where:

D - habitat diversity index

It should be noted that these three indices provide only quantitative habitat information for the agroecosystem. An increase in diversity resulting from an increase in fallow area and a decrease in native lands may actually be detrimental to biodiversity. Therefore, a meaningful evaluation of these indicators with respect to agroecosystem sustainability should be couched in a qualitative interpretation of the habitat changes.

4.6.2 Indicator Species Habitat

Avian indicator species were selected based on their primary breeding habitat requirements such that most of the habitat categories are represented by at least one species. Wetland and upland habitat specialists are Canvasback (*Aythya valisineria*), and Sprague's Pipit (*Anthus spragueii*) respectively. The remaining indicator species represent relative habitat generalists with a recognized habitat structure preference: (1) sparse cover (Horned Lark (*Eremophila alpestris*), Northern Pintail (*Anas acuta*)); (2) moderate cover (Western Meadowlark (*Sturnella neglecta*), Killdeer (*Charadrius vociferus*), Eastern Kingbird (*Tyrannus tyrannus*), Blue-Winged Teal (*Anas discors*)); and (3) dense cover (Bobolink (*Dolichonyx oryzivorus*), Clay-Colored Sparrow (*Spizella pallida*), Mallard (*Anas platyrhynchos*), Gadwall (*Anas strepera*)). The indicator species habitat ranking was based primarily on species occurrence data due to the lack of

productivity data for most species. It is assumed that species occurrence and abundance in a given habitat implies a relative preference for the habitat by that species. The ranking was developed based on a literature review and consultation with prairie bird habitat experts.

The habitat ranking matrix (Table 4.10) is linked with habitat area output of the economic component to compute the total area of preferred habitat for each indicator species at each time step. For example, if a SAM simulation reported 1,000 ha of fallow, 1,000 ha of cropland, 300 ha of improved pasture and 200 ha of native grass, based on Table 4.10 the ecological component of SAM would calculate 2,500 ha of habitat for the Horned Lark.

4.6.3 Wetland Index

Wetlands within an agroecosystem are affected by a combination of factors. Land management can impact wetlands indirectly by changing water infiltration rates of the soil and thereby influencing the quantity of non-growing season precipitation runoff reaching the wetland basin. It is assumed in the soils component that there is a growing season water deficit on all annual crop land such that all summer precipitation is used by crops and none contributes directly to wetland conditions through runoff. However, growing season precipitation can contribute to fall wetland conditions. A wetland index was developed using a simple algorithm to weight the effects of precipitation over a three year period (equation 4.14).

$$W_t = [0.3 * (R_{t-2} + GS_{t-2})] + [0.6 * (R_{t-1} + GS_{t-1})] + (R_t) \quad (4.14)$$

where:

W_t - wetland conditions for period t
 R_t - total runoff from non-growing season precipitation in period t
 GS_t - growing season precipitation in period t

The variable R_t reflects the non-growing season precipitation and the recharge rate. Recharge rate is calculated by the soil model based on soil texture and the ability of the

Table 4.10. Habitat ranking for avian indicator species.

Species	Fallow	Crop	Hay	Impr. Pasture	Native Grass	Shrub	Seas.		Semi.		Perm.	
							Wet.		Wet.		Wet.	
Mallard ^a	-	6	3	5	4	2	1		-		-	
Northern Pintail ^b	-	2	3	4	1	-	-		-		-	
Gadwall ^c	-	5	1	2	3	-	4		-		-	
Blue-Winged Teal ^d	-	5	1	4	3	-	2		-		-	
Canvasback ^e	-	-	-	-	-	-	3		1		2	
Killdeer ^f	4	3	5	2	1	-	-		-		-	
Eastern Kingbird ^g	-	-	-	-	1	2	-		-		-	
Horned Lark ^h	1	2	-	4	3	-	-		-		-	
Sprague's Pipit ⁱ	-	-	3	2	1	-	-		-		-	
Bobolink ^j	-	-	1	3	2	-	-		-		-	
Western Meadowlark ^k	-	4	3	2	1	-	-		-		-	
Clay-Colored Sparrow ^l	-	5	2	4	1	3	-		-		-	

^a Higgins (1977); Kirsch et al. (1978); Johnson et al. (1987); Klett et al. (1988); Greenwood et al. (1995); Maxson and Riggs (1996); Macfarlane (1998); ^b Higgins (1977); Kirsch et al. (1978); Klett et al. (1988); Macfarlane (1998); ^c Higgins (1977); Kirsch et al. (1978); Duebbert and Frank (1984); Klett et al. (1988); Macfarlane (1998); ^d Higgins (1977); Kirsch et al. (1978); Klett et al. (1988); Macfarlane (1998); ^e Maxson and Riggs (1996); Davis (1998); ^f Kantrud (1981); Basore et al. (1986); Davis (1998); ^g Hartley (1994); Davis (1998); ^h Owens and Myres (1973); Kantrud (1981); Hartley (1994); Davis et al. (1998); Sutter (1996); Davis (1998); ⁱ Owens and Myres (1973); Kantrud (1981); Davis and Sealy (1997); Davis et al. (1998); Sutter (1996); Davis (1998); ^j Kantrud (1981); Basore et al. (1986); Bollinger et al. (1990); Hartley (1994); Davis (1998); ^k Owens and Myres (1973); Kantrud (1981); Basore et al. (1986); Hartley (1994); Sutter (1998); Davis (1998); ^l Owens and Myres (1973); Kantrud (1981); Basore et al. (1986); Hartley (1994); Davis et al. (1998); Sutter (1998); Davis (1998).

soil to take up and hold water, as a function of soil organic matter. Since soil organic matter is controlled by cropping practices runoff is also impacted by cropping practices. Growing season precipitation is included in the wetland condition calculation to account for the effect of summer water conditions on the wetland. For example, following a very dry summer the wetland condition, going into the winter and the following spring, will be poor and require greater levels of spring runoff to improve conditions. Two wetland condition values are calculated based on: (1) simulated land use and precipitation; and (2) average precipitation values and fixed land use equivalent to initial land allocation. The wetland index is calculated as a ratio of these two wetland condition values. When this ratio is greater than one wetland water conditions can be considered better than average. Wetland conditions are worse than average when the ratio is less than one.

It should be noted that this wetland index provides a very limited indicator of wetland conditions. Factors not captured by the SAM such as quantity of snow caught by vegetation, speed of spring thaw and degree of frost in ground during spring runoff can have a much greater influence on wetland conditions than levels of SOMC in the soil.

4.7 Sustainability Indicators

As discussed in Chapter Three, the selection of appropriate indicators is of critical importance to assess the changes in an agroecosystem relative to a goal of sustainability. The indicators are used to monitor the change in components of the natural and man-made capital stock. Based on the discussion in the previous chapters changes in components of the capital stock can indicate whether the system meets the criteria for strong sustainability. Evaluation of agroecosystem sustainability requires the integration of these indicators to monitor the changes in the system due to economic or environmental forces. Most of the indicators selected for this study have been highlighted in the course of the discussion of this chapter. A summary of the indicators is provided below.

1. Soil Organic Matter Carbon - an important indicator of soil quality. Changes in SOMC affect nutrient availability, water storage capacity, soil structure and soil biology. SOMC is affected by tillage and residue management, crop varieties, cropping intensity, and levels of fertilizer inputs (Boehm and Belcher, 1998). In general, increases in SOMC stocks indicate an improvement in the capital stock. SOMC is interpreted as: (1) a stock averaged over total annual crop hectares; and (2) as a cumulative stock for each rotation on a per hectare basis in year 50 of the simulation.
2. Soil Nitrogen - soil N is closely linked to SOMC. Soil N is a flow resource and is simulated as a quantity available per hectare per year. Soil N is influenced by soil nutrient cycling dynamics and N fertilizer additions. Soil N increases with decreased tillage. Increases in soil N indicate an improvement in the soil capital. Soil N is interpreted at the same scales as SOMC.
3. Net Nitrogen - represents the nitrogen balance of the soil. Large positive net N values indicate large surpluses of nitrogen in the soil which can lead to leaching and pollution of ground and surface water. Large negative net N values indicate a mining of the nitrogen capital of the soil and thereby a decrease in the health of the soil. Net N is interpreted at the same scales as SOMC.
4. Soil CO₂ Emissions - plant residues are converted to SOMC or decomposed to form CO₂ emissions. Decomposition rates increase with increasing tillage. Soil CO₂ is one of the dominant contributors to atmospheric carbon in an agroecosystem. Soil CO₂ is interpreted at the same scales as SOMC.
5. Crop Yield - as an indicator crop yield integrates the ability of the soil to supply nutrients and water, and the ability of the economic system to provide energy and nutrient subsidies in the form of fossil fuels, equipment and synthetic fertilizers. Crop yield also reflects environmental change and stress. Crop yield is interpreted as an average yield (kg/ha) for each crop within its rotation and ecodistrict context.
6. Land Rent - represents the economic returns to land (the fixed factor of production) and as such is the best indicator, in this model, of the economic health of the agroecosystem. Changes in land rent reflect changes in the profitability of the annual

crop production system. Further, the standard deviation of land rent indicates the economic riskiness of the system. Increases in land rent and/or decreases in the variability of land rent indicate greater probability of maintaining components of the man-made capital stock. Land rent is interpreted as an average for the ecodistrict over the simulation period.

7. Net Revenue - represents the relative profitability and economic risk (standard deviation of net revenue) associated with each rotation. This indicator facilitates a comparative evaluation of the annual crop rotations. Net revenues are interpreted as simple averages for each rotation at an ecodistrict scale over the simulation period.
8. Land Use - represents the allocation of the land resource to the different land use categories at an ecodistrict scale. Increases in certain land uses (i.e. fallow) and/or decreases in other land uses (i.e. native land), or dominance of a landscape by a single land use may indicate decreasing sustainability at a number of levels.
9. Habitat Richness - reflects the number of habitat types in an ecodistrict. An agroecosystem with a greater number of habitats may have greater levels of biodiversity thereby maintaining that aspect of the natural capital. Habitat richness is interpreted at an ecodistrict scale.
10. Habitat Abundance - reflects the relative proportion of an ecodistrict that is allocated to a particular habitat type. Habitat abundance can be used to assess potential habitat quality through a context interpretation of habitat quality thereby reflecting biodiversity. Habitat abundance is interpreted at the ecodistrict scale.
11. Habitat Diversity - reflects the dominance of a given ecodistrict by one or a few habitat types. More diverse landscapes can generally provide greater habitat and therefore potentially greater levels of biodiversity. Habitat diversity is interpreted at the ecodistrict scale.
12. Indicator Species Habitat - reflects the quantity of preferred habitat, for a given species, in an ecodistrict. The range of habitats present in the ecodistrict are represented by the selected indicator species. Therefore, decreases in habitat area or habitat quality (changes in highly preferred habitat) will result in decreased biodiversity. Indicator species habitat is interpreted at the ecodistrict scale.

13. Wetland Index - reflects the relative quality of wetland habitat in terms of water conditions. Decreases in wetland quality reflect decreases in habitat and decreases in biodiversity. Wetland index is interpreted at the ecodistrict scale.

4.8 Conclusion

Chapters Two, Three and Four have described the background and the conceptual and analytical details associated with developing a model to evaluate the sustainability of an agroecosystem. In this framework sustainability is evaluated through indicators reflecting parts of the economic and biophysical components of the system. The remaining chapters in this thesis will present results and discussion from simulations performed with this model.

CHAPTER 5

SIMULATION SCENARIOS

5.1 INTRODUCTION

The preceeding chapters have focused on detailing the background and the theoretical and analytical development of the Sustainable Agroecosystem Model. One of the primary objectives of this study is to use the model to simulate and evaluate agroecosystem changes in response to economic and biophysical perturbations. This chapter will present empirical details of the policy and climate change scenarios selected to meet this objective. The chapter begins with a brief background on climate change (Section 5.2). The next two sections focus on the empirical details of simulating carbon credit (Section 5.3) and carbon tax (Section 5.4) policies within the model. Section 5.5 discusses the empirical details of simulating climate change within the model. The last section (5.6) summarizes the importance of the selected scenarios and provides a link to the following results oriented chapters.

5.2 AGROECOSYSTEMS AND CLIMATE CHANGE

Climate change is emerging as possibly the most important environmental problem at both a global and local scale. Climate change is caused by increasing atmospheric concentrations of primary greenhouse gases: (1) carbon dioxide (CO_2); (2) methane (CH_4); (3) nitrous oxide (N_2O); and (4) chloroflourocarbons (CFC). The increase in atmospheric concentrations of these gases is directly or indirectly a function of increased human activities and growing economic systems. Liu (1995) reports that

Canadian agriculture contributes over 20,000 kilo-tonnes (kt) of CO₂ emissions per year.²³ Agricultural emissions of CO₂ are caused by:

1. Mobilization of soil carbon which occurs through the decomposition of soil organic matter. Rates of decomposition of soil organic matter increase with conversion of perennial tame or native vegetation, or intensifying annual crop management.²⁴
2. Burning of fossil fuels in field operations, transportation, grain drying etc.
3. Carbon emission created during the manufacturing of production inputs. Fertilizer and pesticides manufacturing processes consume a variety of CO₂ emitting energy sources. The CO₂ emissions associated with inputs in this way will be referred to as embodied carbon.

The increase in greenhouse gas concentrations has resulted in proposals to decrease emissions and/or increase the quantity of carbon removed from the atmosphere. This process was formalized by the United Nations Framework Convention on Climate Change, in 1992. The long-term goal of the framework is to “stabilize atmospheric concentrations of greenhouse gases at a level that will prevent dangerous anthropogenic interference with the climate system.” (Bruce et al., 1998). The Kyoto Protocol, adopted in December 1997, is the most recent step towards this goal. Under the Kyoto Protocol Canada has accepted a target of reducing CO₂ emission levels to six percent less than 1990 levels by the year 2010. However, since few emission reduction schemes are currently in place, and Canada will significantly increase emissions by 2010, the reduction below the 2010 emissions levels are projected to be 20 to 25 percent (Bruce et al., 1998).

Canadian agriculture, while an important source of atmospheric CO₂, may be called upon to remove CO₂ from the atmosphere. The process of sequestering atmospheric carbon in soils or plant material is referred to as a sink in the Kyoto protocol. Sinks are defined as “...a process or activity which removes a greenhouse gas

²³ 1 kilo tonne (kt) = 1,000 tonnes

²⁴ Carbon is converted from CO₂ in the atmosphere to organic compounds in plants through photosynthesis, and to organic matter in the soil as plant and animal residues. This soil carbon is in two forms, as stable organic matter (humus) or as active or labile organic matter. The soil carbon that is in the humus form is “sequestered” from the atmosphere. (Gregorich et al., 1995)

from the atmosphere” (United Nations, 1997). Although the only sinks currently recognized in the protocol are changes in soil carbon stocks as a result of forestry, agricultural soils are currently being negotiated. Article 3.4 of the Kyoto Protocol requires that the Conference of Parties “shall at its first session or as soon as practicable thereafter decide upon modalities, rules and guidelines as to how (carbon) removals in agricultural soils and land use changes...shall be taken into account” (United Nations, 1997)).²⁵ Therefore, it has been acknowledged that carbon sequestration in agricultural soils may be a means by which Canada can meet its CO₂ emission reduction commitments. In addition, increasing soil organic matter carbon stocks result in increased soil quality and function and can increase future agricultural production and conserve a component of the natural capital stock.

Agricultural land use and management practices can have an influence on atmospheric carbon concentrations by changing the quantity of carbon sequestered in the soils, or the amount of CO₂ emitted in production activities. As a result, policy tools such as economic incentives or taxes tied to carbon emissions may prove useful in decreasing atmospheric CO₂ concentrations.

5.3 CARBON CREDIT

A policy instrument that can potentially increase soil carbon stocks is an economic incentive for carbon sequestration or carbon credit. A carbon credit could be paid to farms to adopt long-term management strategies that increase the levels of carbon sequestered in their soils. This credit could be paid to farmers by the government as a component of a national climate change strategy. Alternatively, a carbon market could be established such that private industry firms that wish to continue or increase CO₂ emissions could purchase carbon sequestration services from farms.

Within SAM a carbon credit policy is modeled by linking changes in soil carbon stock to the revenues associated with that land use. The soils component of the model

²⁵ Conference of the Parties refers to the Conference of Parties at the United Nations Framework Convention on Climate Change.

calculates a soil organic matter carbon (SOMC) stock for each annual crop rotation, at each time step. Tame forage land is not included within the soils model and does not have an internal mechanism for calculating soil carbon levels. Therefore, a carbon sequestration rate of 0.7 tonnes/ha/year is assumed for hay land soils (Bruce et al., 1998).

With the imposition of a carbon credit the annual change in the per hectare SOMC stock results in a proportional increase in the per hectare revenue for that rotation. At the industry level the carbon credit results in a parallel upward shift in the output demand relationship for each rotation, proportional to the rate of change of the SOMC stock. For example, a 0.25 tonne increase in SOMC in the WF rotation, given a \$75/tonne carbon credit, would result in a \$18.75/ha ($0.25\text{t/ha} * \$75/\text{tonne} = \$18.75/\text{ha}$) increase in WF revenue. Within the SAM simulations carbon credit rates are set at \$25.00, \$75.00 and \$125.00 per tonne of soil sequestered carbon to test the model response.

5.4 CARBON TAX

A carbon tax increases the cost of production inputs proportional to the level of embodied carbon in that input. Increasing the cost of inputs in this way creates an economic disincentive to using inputs with high levels of embodied carbon, or an economic incentive to shift to inputs that have lower levels of embodied carbon. The two categories of annual crop inputs targeted by the carbon tax in the model are fertilizer and chemicals. Within the model the carbon tax results in an upward shift of the fertilizer and chemical input supply functions which is realized at the farm level as an increase in inputs costs for each of the rotations. Carbon tax levels were set at \$25.00, \$75.00, and \$125.00 per tonne of embodied carbon.

Embodied carbon in nitrogen, phosphorous and chemical inputs are reported in Table 5.1. Within the model phosphorous fertilizer application rates are fixed at 0.0224 t/ha for all crops in the Brown soil zone (Saskatchewan Agriculture and Food, 1998). Nitrogen fertilizer rates are calculated within the model as described in Chapter Four.

Chemical input levels are fixed throughout the simulation (Saskatchewan Agriculture and Food, 1998). The estimated embodied carbon levels for fertilizer and chemical inputs, are listed in Table 5.2. The nitrogen fertilizer embodied carbon levels listed in table 5.2 are based on recommended fertilizer application rates (Saskatchewan Agriculture and Food, 1998).

Table 5.1. Embodied carbon levels of selected agricultural inputs.

Input	Embodied Carbon
Nitrogen Fertilizer	1.225 t of C/t fertilizer ^a
Phosphorous Fertilizer	0.225 t of C/t of fertilizer ^a
Chemicals	1 t of C/\$230.00 chemical cost ^b

(^aCoxworth et al., 1995; ^bCoxworth, 1997)

Table 5.2. CO₂ emission levels associated with input embodied carbon for the four rotations.

Rotation	Nitrogen Fertilizer	Phosphorous Fertilizer	Chemicals
WF	0.0138 t/ha	0.0025 t/ha	0.1506 t/ha
WWF	0.0299 t/ha	0.0034 t/ha	0.1635 t/ha
WCP	0.0435 t/ha	0.0050 t/ha	0.2856 t/ha
WFP	0.0225 t/ha	0.0034 t/ha	0.2625 t/ha

5.5 CLIMATE CHANGE

A series of simulations were performed under simulated climate change conditions to examine how agroecosystem functions and sustainability will be affected by climate change. The effect that global climate change will have on the weather patterns in southwestern Saskatchewan is unknown. For this study it was assumed that climate change will cause: (1) an increase in mean temperatures; (2) a decrease in mean precipitation; and (3) an increase in the uncertainty of precipitation patterns, or increase in the probability of extreme events (droughts and floods). The model simulates non-climate change precipitation by randomly selecting growing season and non-growing season precipitation values from a gamma precipitation distribution with a mean and standard deviation equal to historical ecodistrict precipitation data. Climate change was

simulated by decreasing mean precipitation and increasing standard deviation of precipitation by two centimetres. This creates a precipitation distribution that is more skewed toward the y axis with a longer tail, thereby decreasing mean rainfall and increasing the probability of extreme events (Figure 5.1). The model randomly selects growing season and non-growing season precipitation values from this distribution at each time step. To simulate increasing mean temperatures the growing degree day statistic was increased by five percent. Simulated climate change was imposed on the target ecodistrict, with no change in policy, creating baseline climate change simulations. In addition, the carbon credit and carbon tax policies discussed above were imposed on the landscapes under climate change.

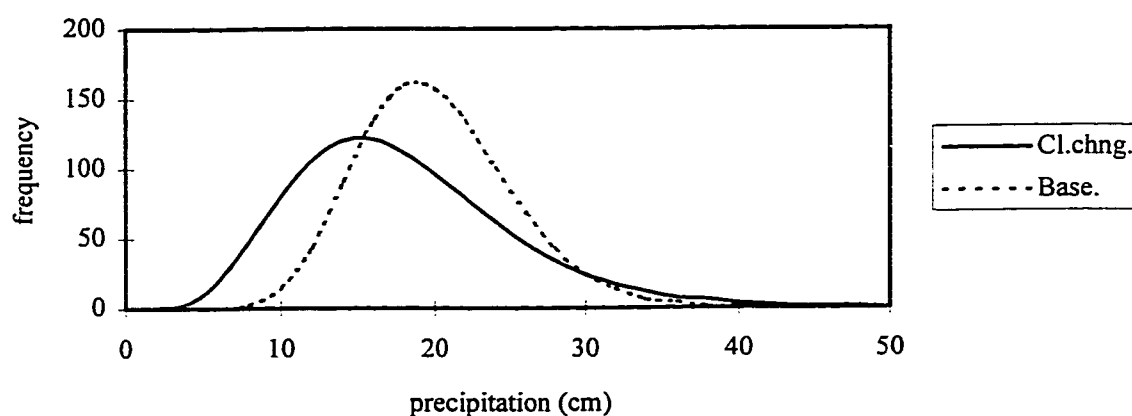


Figure 5.1. Simulated climate change rainfall frequency distribution for the Antelope Creek Plain ecodistrict; Climate change - mean = 17, standard deviation = 7; Baseline climate - mean = 19, standard deviation = 5, n = 1000.

5.6 CONCLUSION

This chapter has provided background to, and described the policy and climatic scenarios that have been imposed within SAM. The output of the model, as a consequence of these economic and environmental shocks, provides insight into the system processes and the relative sustainability of the target agroecosystem. The following two chapters present simulation results and discussion focusing on system sustainability in the context of these scenarios.

CHAPTER 6

BASELINE SIMULATIONS

6.1 INTRODUCTION

This chapter and the chapter following focus on output from Sustainable Agroecosystem Model simulations. The focus of this chapter is baseline runs for the four target ecodistricts. Simulation results and discussion concentrate on changes in selected indicators, and a description of the system dynamics that lead to these changes. This discussion highlights the responsiveness of the model to climatic (precipitation, mean daily temperature, growing degree days) and physical (soil texture) characteristics of the different ecodistricts. The next section (6.2) presents a short preamble to the simulations discussed in this chapter. A detailed discussion of the results, focusing on indicator changes and system feedbacks processes, follows (Section 6.3). Section 6.4 provides a synthesis discussion on the agroecosystem sustainability insights provided by the baseline simulations. Finally Section 6.5 summarizes this chapter and provides a link to the following chapter.

6.2 BASELINE ECODISTRICTS

Baseline simulations were run for the four target ecodistricts: (1) Eston Plain; (2) Beechy Hills; (3) Antelope Creek Plain; and (4) Gull Lake Plain. No policy or climatic shocks were introduced into SAM in these simulations. Input and output prices are fixed throughout the 50 year simulations, and climate parameters are based on weather statistics for the appropriate ecodistricts. The simulations presented in this chapter provide insight into the changes in sustainability indicators in two different contexts: (1)

within each simulated ecodistrict calibrated with the current biological, physical and social parameters; and (2) between ecodistricts, each with different biological, physical, and social parameters. The output data are aggregated from 50 simulations, each of which is 50 years in length.

The following sections present a synthesis discussion on the agroecosystem function insights provided by SAM output. A simple schematic of the model (Figure 6.1) serves as a useful map of the linkages established within the model. The discussion in this chapter focuses on the system effects that are facilitated by the linkages shown in this map.

6.3 AGROECOSYSTEM CHANGES

The SAM simulates land use and cropping decisions of a series of homogeneous, profit-maximizing farms in a given ecodistrict. These farms are faced with fixed input and output prices, and a range ecodistrict specific biophysical and climatic constraints. The environmental characteristics of each of the ecodistricts are presented in Table 6.1. Given these economic and physical constraints the farmer makes land management decisions that maximize profits. Changes in agricultural productivity, due to changing soil quality, which is affected by cropping practices and precipitation fluctuations, result in changing economic signals. The farms respond to these economic signals by altering land allocation among annual crop rotations, tame forage and native land, and changing nitrogen fertilizer input levels. The effect of land allocation on the integrity of the natural and man-made capital associated with the soil, ecological and economic components of the agroecosystem are revealed through a series of indicators.

Land use and agricultural management decisions are important drivers of agroecosystem change at a range of scales. However, the baseline simulations provided by SAM indicate that these management decisions are set in a context of the biophysical and climatic constraints specific to the ecodistrict. For example, the texture of the soil, through its influence on yields, had a profound influence on the economic signals received by the farm. The annual crop yields reported for the Eston Plain ecodistrict,

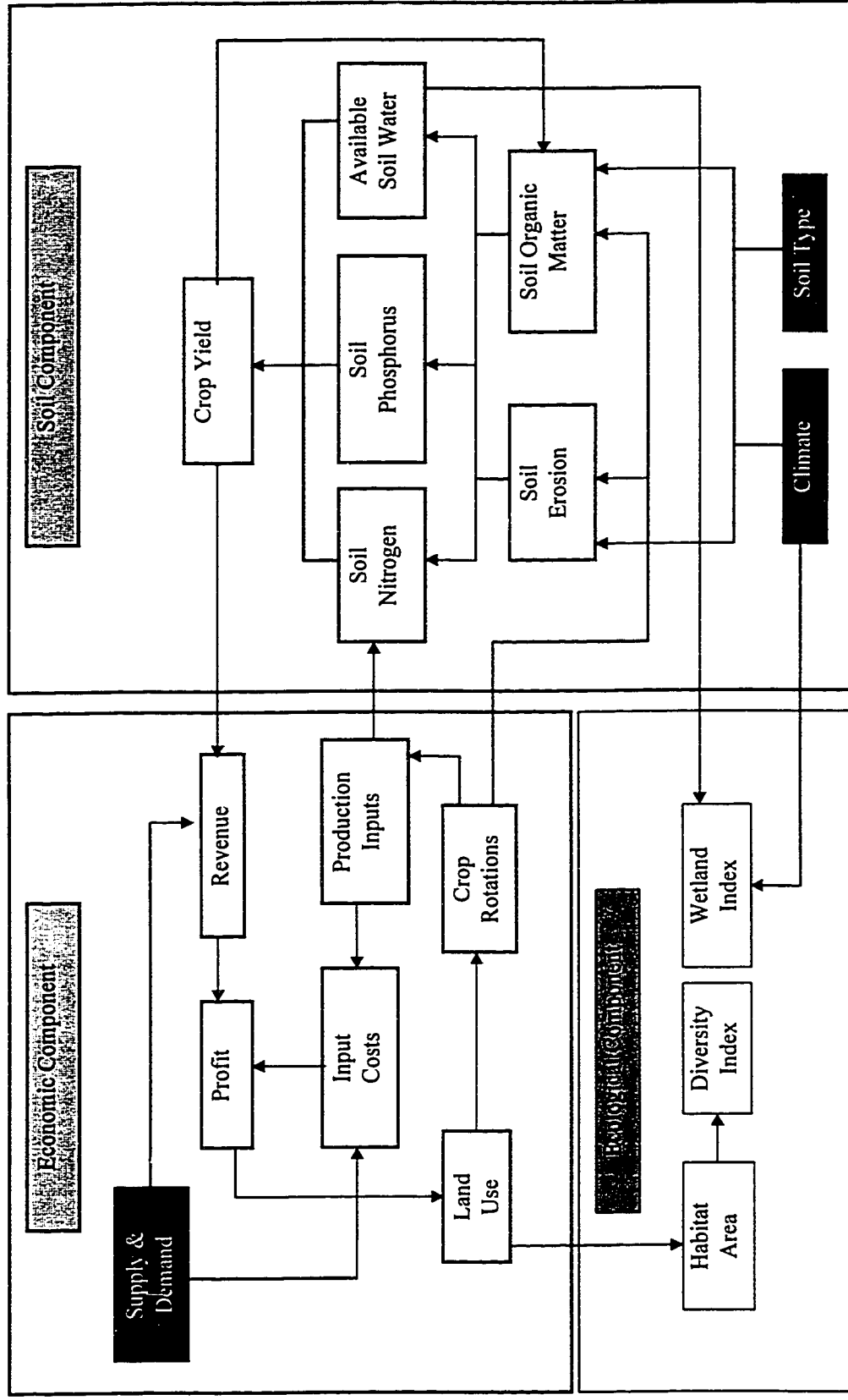


Figure 6.1. Schematic of the Agroecosystem Model structure showing the economic, soils and ecological components and the inter-component linkages. Black cells identify parameters exogenous to SAM..

Table 6.1. Biophysical and climatic characteristics of target ecodistricts.

Parameter	Eston Plain	Beechy Hills	Antelope Creek Plain	Gull Lake Plain
Soil Texture	Brown clay	Brown loam	Brown loam	Brown loam/Brown sandy loam
Growing Season	19	18	19	19
Precipitation (cm)				
Non-Growing Season	15	14	17	18
Precipitation (cm)				
Growing Degree Days	1,479	1,507	1,459	1,397
Mean Daily	1	2	3	4
Temperature (days)				
Initial Solum (cm)	60	60	60	60
Initial SOMC (t/ha)	50,000	50,000	50,000	50,000
Initial Surface Trash (kg/ha)	4,200	4,200	4,200	4,200

which has clay textured soils (Table 6.1), were significantly higher for wheat, peas and canola in all rotations (Table 6.2) and were less variable (Table 6.2) than in the other ecodistricts, which have courser textured soils. Higher and less variable yields resulted in higher and less variable net revenues for each rotation (Figure 6.2; Figure 6.3), and at the ecodistrict scale, the total economic returns to land were higher and less variable (Table 6.3).

The farms in the other ecodistricts faced more severe biophysical and climatic constraints (Table 6.1). The Beechy Hills and Gull Lake Plain ecodistricts had similar yields for all annual crops but canola. The greater frequency of warm days in Gull Lake Plain resulted in very low canola yields due to heat stress. The combination of sandy soils but greater rainfall in Gull Lake Plain imposed similar constraints to those experienced in the Beechy hills, which has loamy soils in combination with less rainfall. Therefore, the economic returns to land (Table 6.3) and the rotation revenues were very similar (Figure 6.2; Figure 6.3) in these two ecodistricts (excluding WCP).

The biophysical characteristics of the Antelope Creek Plain fall between those for the Eston Plain in one extreme, and the Beechy Hills and Gull Lake Plain in the other. The soils of Antelope Creek Plain contain more clay than the Gull Lake Plain but

Table 6.2. Yield and standard deviation of yield for crops in rotation and ecodistrict context (t/ha).

		Ecodistrict			
Rotation and Crop (t/ha)		Eston	Beechy	Antelope	Gull Lake
WF wheat	mean	3.18	2.30	2.60	2.30
	st.dev.	0.34	0.59	0.49	0.54
WWF st. wheat	mean	2.96	1.57	1.87	1.57
	st. dev.	0.58	0.68	0.67	0.62
WWF f. wheat	mean	3.27	2.35	2.66	2.36
	st.dev.	0.36	0.63	0.54	0.58
WCP wheat	mean	2.93	1.39	1.70	1.32
	st.dev.	0.63	0.66	0.67	0.56
WCP canola	mean	1.98	0.57	0.56	0.18
	st.dev.	0.55	0.53	0.51	0.28
WCP peas	mean	2.12	0.96	1.16	0.96
	st.dev.	0.51	0.46	0.49	0.41
WFP wheat	mean	2.95	1.47	1.85	1.48
	st.dev.	0.58	0.65	0.66	0.60
WFP peas	mean	2.48	1.62	1.90	1.63
	st.dev.	0.28	0.52	0.46	0.47
Hay	mean	2.65	2.75	2.98	2.62
	st. dev.	0.31	0.34	0.34	0.41

Table 6.3. Mean and standard deviation of land rent and hay net revenue for the four ecodistricts (\$/ha) average over 50 years.

		Ecodistrict			
		Eston	Beechy	Antelope	Gull Lake
Land Rent (\$/ha)	mean	30.96	24.13	30.60	23.39
	st. dev.	51.15	65.73	61.38	56.01
Net Revenue (\$/ha).	mean	55.57	57.65	62.61	55.12
Hay	st. dev.	6.51	7.13	7.04	8.59

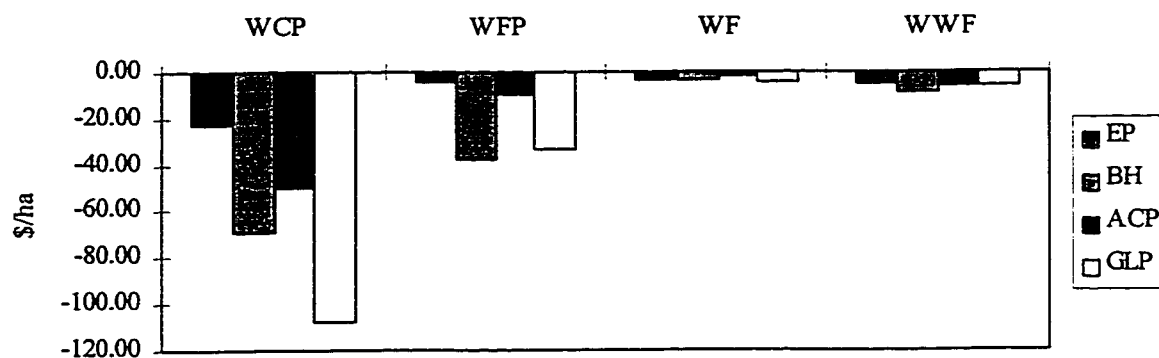


Figure 6.2. Simple average net revenue for annual crop rotations in the four ecodistricts (\$/ha).

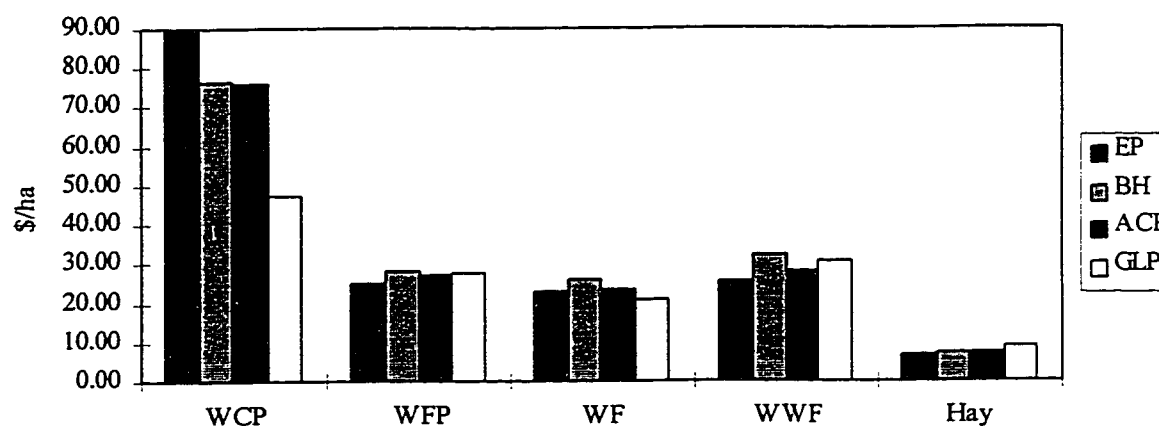


Figure 6.3 Average standard deviation of net revenue for annual crop rotations and hay in the four ecodistricts (\$/ha).

receives more precipitation than the Beechy Hills (Table 6.1). The annual crop yield and yield variability (Table 6.2) and the associated economic indicators (Table 6.3; Figure 6.2; Figure 6.3) followed the same pattern simulated in the other ecodistricts.

The economic signals in each of the four ecodistricts resulted in different land use simulations among the ecodistricts (Table 6.4). In the Eston Plain ecodistrict crop

revenues were sufficient in all rotations to approximately maintain initial land use patterns through the simulation. In contrast, WCP and WFP rotations were eliminated and native land area decreased to be replaced by WF and WWF in the Beechy Hills and Gull Lake Plain (Table 6.4). The economic signals in the Antelope Creek Plain resulted in more moderate changes in land use with minor increases in WF and WWF area. It should be noted that the changes in land use over the course of a simulation reflected the variability of the revenues returned by the rotations. For example, the revenue variability of the WCP rotation was mirrored by the variability of the quantity of land allocated to this rotation over the simulation (Figure 6.4)

In the Beechy Hills, Antelope Creek Plain and Gull Lake Plain ecodistricts hay and native land was converted to WF and WWF production (Table 6.4). In contrast, hay land increased and native land decreased only slightly in the Eston Plain Ecodistrict. The relatively large stock of native land in Beechy Hills, Antelope Creek Plain and Gull Lake Plain (Table 4.1) meant the simulations faced a relatively elastic segment of the native land supply functions. The Eston Plain ecodistrict, in comparison, had small initial stocks of native land and therefore faced a relatively inelastic segment of the native land supply function.

Annual crop yields within SAM are a function of soil quality, which is determined by the regional physical and climatic conditions and the agricultural management practices applied. As a result, the changes in soil quality over the simulations was a very important factor determining land use. An important positive feedback relationship involved yield and SOMC. Higher yields result in larger additions of plant residue to the soil. Plant residues are allocated to the SOMC stock or to the CO₂ pool through decomposition. Greater stocks of SOMC result in more available soil water, greater soil nitrogen, through nitrogen turnover, and greater soil phosphorous, providing conditions for greater yields.

The Eston Plain Ecodistrict, which has the greatest inherent yields due to the greater water storage capability of the clay textured soils, benefited most from the SOMC - yield feedback process. The higher initial yields simulated in the Eston Plain resulted in greater SOMC stocks (Table 6.5) and therefore greater future yields and

Table 6.4. Initial and final land use (percent) of the four ecodistricts.

Land Use	Ecodistrict							
	Eston		Beechy		Antelope		Gull Lake	
	yr. 0	yr. 50	yr. 0	yr. 50	yr. 0	yr. 50	yr. 0	yr. 50
WF	35.41	33.94	32.25	36.10	39.22	40.30	28.55	34.09
WWF	27.54	27.42	23.46	33.03	28.52	33.93	20.04	29.28
WCP	3.93	6.29	1.17	0.69	1.43	1.51	0.50	0.00
WFP	11.80	10.34	1.76	0.00	2.14	0.47	1.00	0.00
Hay	1.97	6.01	7.56	3.81	7.68	6.87	13.44	7.10
Native	19.34	15.99	33.78	26.37	21.02	16.93	36.47	29.52

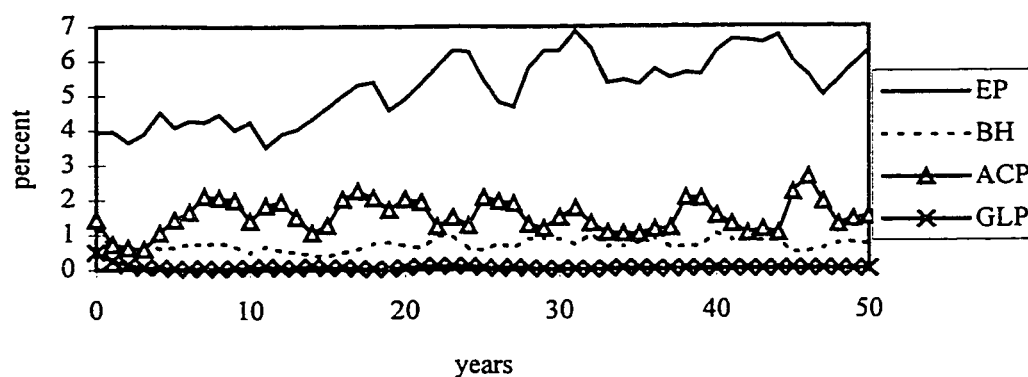


Figure 6.4. WCP rotation, percentage of ecodistrict through 50 year simulation of the four ecodistricts.

Table 6.5. Change in soil indicators over initial value after 50 years for an average annually cropped hectare in each ecodistrict.

Soil Indicators	Ecodistrict			
	Eston	Beechy	Antelope Creek	Gull Lake
SOMC (kg/ha)	4947.18	2799.67	3968.79	3383.16
Soil N (kg/ha)	44.87	25.82	29.16	26.16
Net N (kg/ha)	-504.17	167.64	-10.03	175.41
N Fert (kg/ha)	1503.72	1559.46	1567.59	1569.83
CO ₂ (kg/ha)	75697.59	52150.06	58214.48	51326.74

additions to SOMC stocks. In addition, nitrogen fertilizer requirements were smaller in the Eston Plain ecodistrict due to the greater soil nitrogen levels, further enhancing net revenues and land rent. At a rotation scale, the WCP rotation in the Eston Plain ecodistrict contributed significantly to the SOMC stock of the soil (Table 6.6). The large additions of SOMC provided by the WCP rotation resulted in more favourable conditions for this rotation over the course of the simulation.

The SOMC - yield feedback process had the opposite effect in the Beechy Hills and Gull Lake ecodistricts, to that reported for the Eston Plain ecodistrict. Smaller crop yields in these ecodistricts, due to the lower soil water availability in the coarser textured soils, resulted in smaller additions to the SOMC stock, greater nitrogen fertilizer requirements (Table 6.5) and decreasing future yields. In addition, the combination of low revenues and limited soil water in these ecodistricts provided little incentive to add nitrogen fertilizer (Table 6.5; Table 6.6), thereby further limiting future yields. The land use effect in these ecodistricts was to allocate more land to those rotations that provided more consistent yields and imposed less of a water limitation with more frequent summer fallow management (WF and WWF). The land use decisions that were simulated in these two ecodistricts limited the increases in the SOMC stocks due to two processes: (1) greater fallow area - fallow increases decomposition rates; and (2) greater erosion rates on fallow and lower clay content soils - soil erosion decreases SOMC stocks.

A consequence of the high yields reported for the Eston Plain ecodistrict is that the larger SOMC stocks contributed greater soil and residue CO₂ emissions (Table 6.7). In addition, the greater area allocated to the more input intensive WCP and WFP rotations contributed greater levels of embodied carbon in the Eston Plain ecodistrict.

Table 6.6. Soil quality indicators as a cumulative stock at 50 years for four rotations in the four target ecodistricts (kg/ha).

Ecodistrict	Rotation	Soil Indicator (kg/ha)				
		SOMC	Soil N	Net N	N Fert	CO ₂
Eston	WF	288.12	40.21	-469.48	1390.6	69482.40
	WWF	8649.89	49.03	-562.94	1402.55	87887.94
	WCP	19239.66	48.56	-533.08	1426.94	79524.06
	WFP	3940.67	47.14	-458.65	1412.30	63348.38
Beechy	WF	405.90	23.04	39.05	1871.18	49350.58
	WWF	5425.17	28.91	304.78	1861.12	55634.73
	WCP	6686.28	24.65	571.33	1873.59	36424.18
	WFP	620.15	27.62	5.07	1887.96	36519.80
Antelope Creek	WF	1176.16	26.07	-107.60	2192.29	55314.63
	WWF	7307.71	32.95	82.41	1656.99	62882.01
	WCP	8050.22	28.08	574.53	1792.86	41175.40
	WFP	2332.41	31.54	5.41	1229.41	42874.03
Gull Lake	WF	876.32	23.35	50.69	1012.82	48630.13
	WWF	6393.08	29.47	325.55	804.33	54567.45
	WCP	4270.55	24.51	425.39	985.77	31315.56
	WFP	1117.08	28.14	20.21	826.81	36133.88

Table 6.7. Carbon balance as a total value over the 50 year simulations for each ecodistrict.

Carbon Balance	Ecodistrict			
	Eston	Beechy	Antelope	Gull Lake
N Fert. embodied C (Mt/50yr)	1.13	0.29	0.33	0.16
P Fert. embodied C (Mt/50yr)	0.11	0.03	0.03	0.01
Chem. embodied C (Mt/50yr)	5.52	1.19	1.38	0.63
Soil CO ₂ emissions (Mt/50yr)	46.49	7.89	10.07	4.17
Total CO ₂ emissions (Mt/50yr)	53.25	9.40	11.81	4.97
SOMC (Mt/50yr)	3.04	0.41	0.68	0.27
Hay stored C (Mt/50yr)	1.20	0.39	0.61	0.39
Total C stored (Mt/50yr)	4.24	0.80	1.29	.66
Net CO ₂ emissions (Mt/50yr)	49.01	8.60	10.52	4.31
Net CO ₂ /ha of crop and hay(t/50yr)	74.35	55.57	55.29	49.83

The decreased area of native land in all ecodistricts (Table 6.4), and the decreased area of hay land in all but the Eston Plain ecodistrict negatively affected those indicator species that prefer dense cover (Figure 6.5). Those species that prefer sparse cover (i.e. Horned Lark, Killdeer) experienced an increase in preferred habitat in the Beechy Hills and Gull Lake Plain ecodistricts due to the increases in land allocated to fallow and cropland, with increases in WF and WWF rotations (Figure 6.5). With respect to the context for habitat areas, fallow and cropland offer sparse cover and are frequently disturbed by tillage operations. As a result, fallow and cropland are a more hostile habitat context to species that prefer more dense cover. The increases in fallow and cropland area in the Beechy Hills, Antelope Creek Plain and Gull Lake Plain ecodistricts (Figure 6.6) implies that habitat quality may be decreasing with respect to the context for habitat areas. In contrast, in the Eston Plain ecodistrict both fallow and cropland decreased and tame hay and native pasture increased, thereby improving the context for the native habitat areas. In all ecodistricts the wetland index is highly variable, reflecting precipitation variability with a modest upward trend over the course of the simulations, implying an improvement in wetland conditions (Figure 6.7).

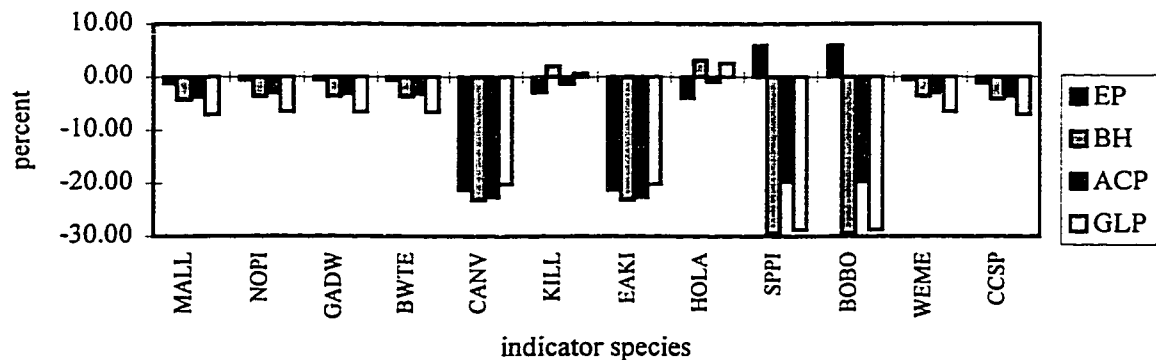


Figure 6.5. Percentage change in preferred habitat area over 50 year simulations in the four ecodistricts.

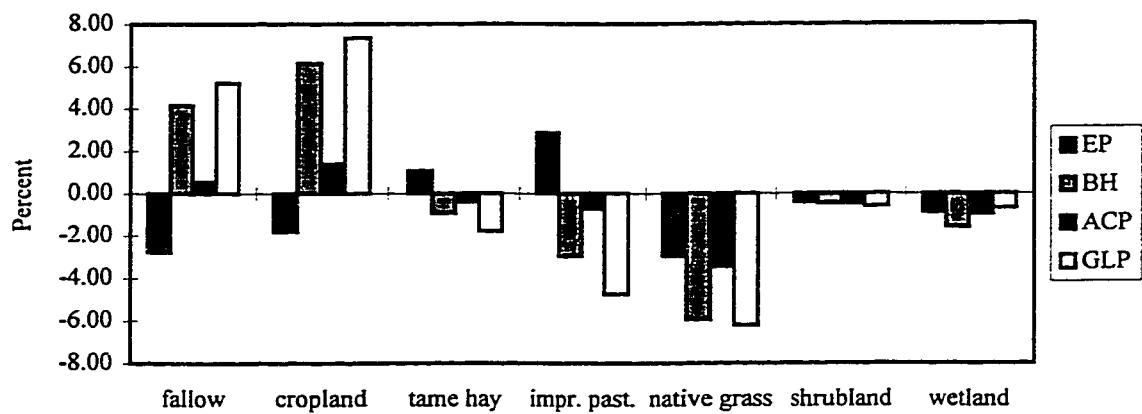


Figure 6.6. Percentage change in relative habitat abundance over 50 year simulation for the four ecodistricts.

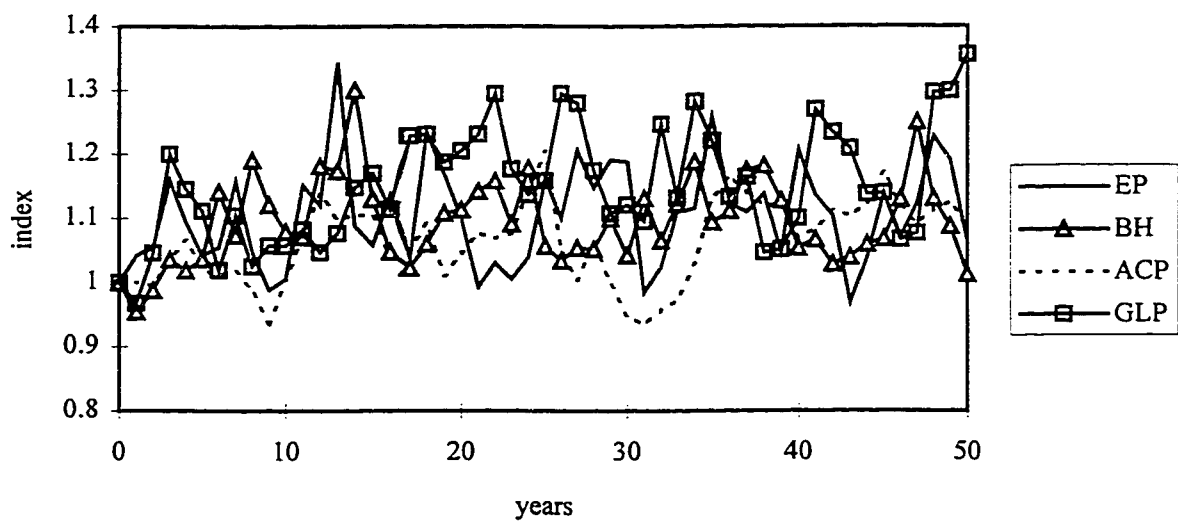


Figure 6.7. Wetland index over 50 year simulation for the four ecodistricts.

6.4 SUMMARY AND SYNTHESIS

An important result coming out of the baseline simulations of the four target ecodistricts is the importance of the biophysical and climatic context of the agroecosystem to the productivity of the systems. In general, the crop rotations modeled were relatively more sustainable in agroecosystems with clay textured soils than in the loam and sandy soil regions.

In the Eston Plain ecodistrict the economic indicators suggested that components of the man-made capital stock would be maintained. In addition, the soil indicators showed that the soil natural capital improved over the course of the simulations. The economic viability of all available rotations in this ecodistrict ensured a more diverse landscape including increases in tame forage land. This increase in tame forage land, which was accompanied by decreases in crop and fallow land, resulted in smaller losses of habitat for many of the indicator species.

In contrast there was evidence that aspects of this agroecosystem were not on a sustainable path. Although the total nitrogen deficit was quite small in absolute terms, it did indicate that the soil nitrogen capital was being degraded at a greater rate than in the other ecodistricts. In addition, the cropping practices in place in the Eston Plain ecodistrict were releasing relatively large levels of CO₂ emissions, and the percentage of the landscape that was preferred habitat for most of the indicator species (native cover) was smaller than all other ecodistricts.

The biophysical characteristic of the Beechy Hills and Gull Lake Plain ecodistricts restricted the annual crop production options to WF and WWF. As a result, SOMC stocks and soil nitrogen levels were smaller in these ecodistricts. The net nitrogen balance indicated a small surplus for all rotations revealing that soil nitrogen capital was not being run down. Land rent values in Beechy Hills and Gull Lake Plain ecodistricts were approximately 25 percent lower than those reported for the other two ecodistricts, indicating a lower probability of maintaining the man-made capital stock, and therefore a lower probability of economic sustainability. Larger initial areas of perennial cover (tame forage and native) indicated better habitat conditions in these two

ecodistricts. However, the rate of conversion of these areas to annual crop was the greatest among the ecodistricts.

The biophysical constraints in the Antelope Creek Plain ecodistrict were less severe than the Beechy Hills and Gull Lake Plain ecodistricts but more severe than the Eston Plain ecodistrict. In the Antelope Creek Plain WF and WWF were more productive than in the Beechy Hills and Gull Lake Plain ecodistricts, and WCP was allocated to a small quantity of land. As a result, the economic indicators suggested a more economically sustainable agroecosystem than the Beechy Hills and Gull Lake Plain ecodistricts. SOMC stocks, soil nitrogen and virtually balanced soil nitrogen dynamics indicated maintenance of the soil capital stock. Total CO₂ emissions were greater in this ecodistrict than the Beechy Hills and Gull Lake Plain ecodistricts, but the greater rate of carbon sequestration in annual cropland, and the larger areas of hay land, resulted in relatively low emission levels at a per hectare scale. In terms of habitat, fallow and cropland increased only slightly and native lands decreased moderately resulting in changes in habitat that were midway between the other ecodistricts.

The cropping systems available to the farms in these simulations seemed marginally conducive to agroecosystem sustainability. The WF and WWF rotations were the only economically viable rotations in all ecodistricts. Between 61 and 74 percent of the total land base in these ecodistricts was allocated to the WF and WWF rotations. Domination of the landscape by wheat and fallow makes for homogeneous agroecosystems. These low diversity systems have lower levels of economic and ecological resilience and therefore lower probability of sustainability. While the WF and WWF rotations require less inputs, and as a result were responsible for less embodied CO₂ emissions, the higher rates of decomposition and erosion associated with these rotations contributed to the net CO₂ emissions. In general, the menu of management choices available in these simulations seemed more appropriate to maintaining the capital stock of those agroecosystems with more moderate biophysical constraints, Eston Plain, than in the more constrained environment found in the Beechy Hills and Gull Lake Plain Ecodistricts.

6.5 CONCLUSION

The results presented in this chapter provide insight into the relative sustainability of selected ecodistricts. The changes in the natural and man-made capital stock in the simulations signals the direction of change of the ecodistrict with respect to a long-term goal of sustainability. The results presented highlighted the importance of biophysical constraints in determining the sustainability of an agroecosystem. In addition, the results and discussion of this chapter have highlighted the importance of feedback relationships, particularly with respect to SOMC in annual crop land. However, the discussion also indicates that determining whether an agroecosystem is sustainable or not is not straight forward. Sustainable processes in terms of one indicator may result in changes in another indicator that are not sustainable. While this chapter focused on results relevant to a cross-section of ecodistricts, the following chapter presents simulation results from a single ecodistrict with the imposition of a range of policy and climate change forces.

CHAPTER 7

CARBON EMISSIONS POLICY AND CLIMATE CHANGE

7.1 INTRODUCTION

The discussion in Chapter Six focused on baseline simulation output for the four target ecodistricts. This chapter assesses the relative sustainability of simulated landscapes by concentrating on system changes in a single ecodistrict, under the influence of economic (carbon credit and carbon tax policies) and environmental perturbations (climate change). The discussion provides insight into agroecosystem feedbacks, and how system level changes alter the sustainability of the complex system. The next section concentrates on output from non-climate change simulations (Section 7.2). Section 7.3 examines the same policy scenarios under simulated climate change. Finally, Section 7.4 provides a brief synthesis of the results presented in this chapter concentrating on agroecosystem sustainability.

7.2 NON-CLIMATE CHANGE

All simulations in this chapter focus on the Antelope Creek ecodistrict. The non-climate change simulations use climatic parameters that are consistent with ecodistrict weather data (Table 4.3). The discussion in this section focuses on simulation output with the imposition of carbon credit and carbon tax policies, in a non-climate change environment. The discussion in this chapter, as in Chapter Six, highlights the effect of the system linkages mapped out in Figure 6.1.

7.2.1 Carbon Credit

A carbon credit serves as a subsidy to those land use practices that sequester soil carbon. It is assumed that a carbon credit applies only to the incremental carbon sequestered in annual crop land, and tame forage soil. In SAM native vegetated soils are assumed to be in equilibrium with respect to organic carbon. Simulations were performed with carbon credit levels of \$25.00, \$75.00 and \$125.00 per tonne of sequestered carbon. The principal consequences of the carbon credit are as follows:

- 1. Annual crop and hay revenues increased in proportion to the rate that carbon was sequestered in the soil.*

Within SAM a carbon credit resulted in an increase in revenues for tame forage production proportional to the rate of carbon sequestration imposed on tame forage soils (0.7 t/ha/yr). For example, a \$25.00 carbon credit translated into an approximately \$17.50/ha increase in hay revenues (Table 7.1). For annual crops revenues increased proportional to the simulated rate of soil carbon sequestration. However as carbon credit levels increased, the SOMC stocks in each rotation did not all increase. Figure 7.1 shows that WFP and WCP revenues decreased with greater carbon credit levels, while WF and WWF revenues increased. The baseline simulations of Chapter Six showed that in the Antelope Creek Plain ecodistrict the WFP rotation was not profitable and contributed very little to the SOMC stock. Therefore, a carbon credit resulted in nitrogen fertilizer rates decreasing in WFP (Figure 7.2), and land allocated to rotations that sequestered more carbon (Figure 7.3; Table 7.2) and were therefore more profitable. With a carbon credit, the farms' management decisions accounted for increasing the SOMC stocks of their soils. At higher carbon credit levels the marginal productivity of nitrogen fertilizer, in terms of SOMC increments, was greater with WF and WWF, which are less limited by water than the continuous crop rotation, WCP (Figure 7.3). WF and WWF revenues increased (Figure 7.1) and nitrogen fertilizer inputs on these rotations were set to optimize SOMC, which translates into optimized revenues under a carbon credit policy (Figure 7.2; Figure 7.3).

Table 7.1. Change in land rent and hay net revenue (\$/ha) due to carbon credit (cc), and carbon tax (ct) policies within a non-climate change environment.

Scenario	Land Rent (change in \$/ha)	Net Revenue Hay (change in \$/ha)
Baseline (\$/ha)	30.60	62.61
\$25 C credit	+5.75	+17.42
\$75 C credit	+21.50	+52.62
\$125 C credit	+28.39	+87.33
\$25 C tax	-4.81	0.00
\$75 C tax	-17.79	0.00
\$125 C tax	-26.51	0.00

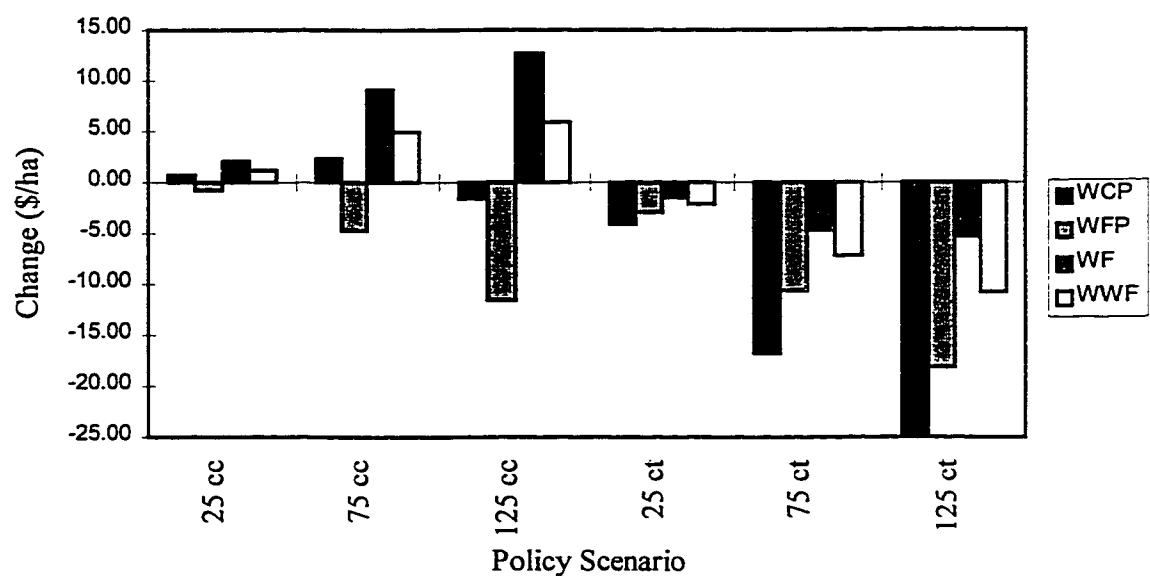


Figure 7.1. Change in hectare revenue (\$/ha) due to carbon credit (cc), and carbon tax (ct) policies within a non-climate change environment.

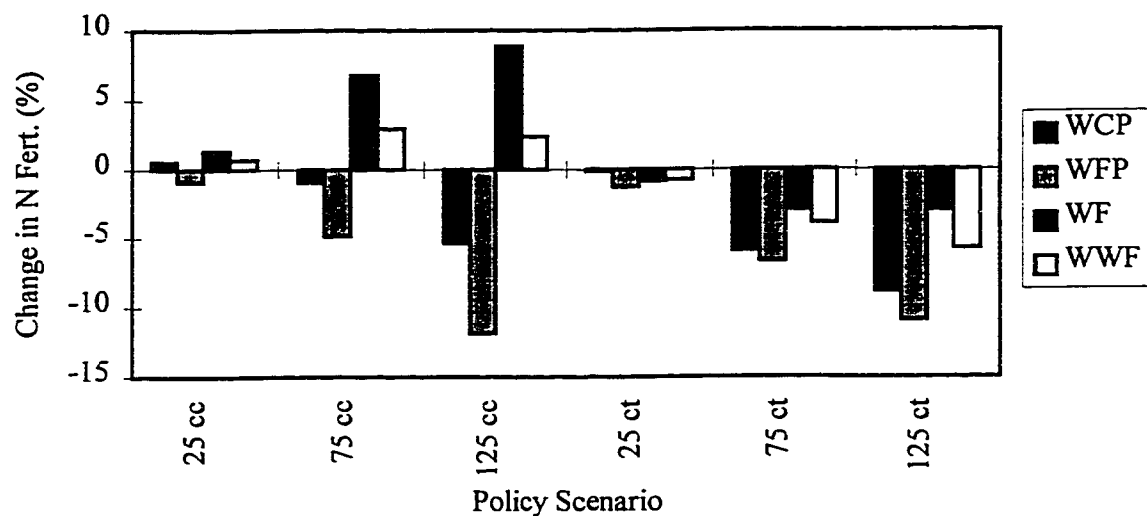


Figure 7.2. Percentage change in nitrogen fertilizer input for the four rotations, due to carbon credit (cc), and carbon tax (ct) policies within a non-climate change environment.

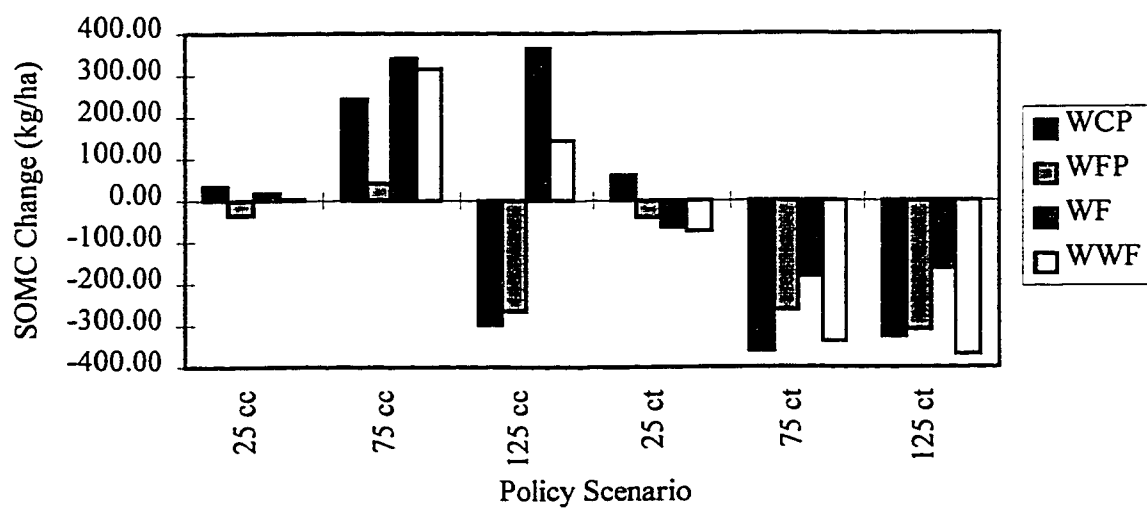


Figure 7.3. Change in SOMC (kg/ha) in the four rotations due to carbon credit (cc), and carbon tax (ct) policies within a non-climate change environment.

Table 7.2. Percentage change in land use with carbon credit, and carbon tax policies under non-climate change environment.

Scenarios	Land Use Categories (% change)					
	WCP	WFP	WF	WWF	Hay	Native
Baseline (ha)	3479	1083	93161	78429	15875	39139
\$25 C credit	+53.39	-62.10	-8.52	-7.60	+87.91	-2.33
\$75 C credit	-32.82	-99.77	-30.35	-22.00	+340.50	-5.07
\$125 C credit	-47.02	-100.00	-49.24	-33.48	+525.38	-8.73
\$25 C tax	+12.98	-29.15	-1.59	-1.50	+26.64	-1.26
\$75 C tax	-38.00	-82.93	-4.94	-5.43	+80.51	+0.18
\$125 C tax	-56.89	-95.35	-8.43	-10.45	+150.58	+0.10

2. The demand for land increases which results in an increase in land rent.

The structure of the economic component of SAM is such that increased revenues, with non-land input costs kept relatively constant, resulted in an upward shift in the derived demand for land for each rotation, and an increase in the overall demand for annual cropland. Under a carbon credit, the increase in demand for land in each rotation is proportional to the quantity of soil sequestered carbon provided by that rotation. As discussed earlier, the increases in SOMC stock were greater in the WF and WWF rotations, with increasing carbon credit levels, due to the marginal productivity of nitrogen fertilizer in terms of yield and SOMC. As a result, the derived demand for land for WF and WWF increased while the derived demand for WFP and WCP decreased. Overall however the demand for annual crop land increased with increased carbon credit levels. Given a fixed supply of land this increase in demand results in an increase in land rent (Table 7.1). In the SAM simulations the supply of annual cropland was not constant, but decreased, further increasing land rent. Total annual cropland area decreased during the simulations due to increased allocation of land to hay production (Table 7.2). This is discussed in more detail in the following section.

The increase in land rent, discussed above, resulted in a decrease in native land stocks (Table 7.2). The greater returns to land associated with increases in carbon credit

rates created an economic incentive to convert native lands to annual crop and tame forage production.

3. Land was allocated to those land uses that have greater rates of soil carbon sequestration .

In the SAM simulations a carbon credit resulted in a reallocation of land from annual crop production to tame hay production (Table 7.2). This shift in land use was in response to the extra revenues that could be gained from sequestering carbon. Within the model hay land sequesters greater quantities of carbon than any other land use. Total annual cropland area decreased and within annual cropland there was a relative shift of land use towards the WWF rotation (Table 7.2). The WF rotation reported larger increases in yields and SOMC (Figure 7.3) in response to increases in nitrogen fertilizer (Figure 7.2) due to less water limitation with greater frequency of fallow. However, the WWF rotation contributed greater overall quantities of SOMC than WF and as a result takes advantage of the SOMC - yield feedback process discussed in Chapter Six. The larger SOMC stocks in WWF resulted in greater levels of soil nitrogen, less demand for nitrogen fertilizer input (Figure 7.2) and more stable yields leading to larger decreases in economic risk (Figure 7.4). The economic conditions created by a carbon credit result in WWF being the most economically attractive annual crop rotation.

4. Quantities of soil sequestered carbon increased and total CO₂ emissions decreased

The relative shift of annual cropland toward the WWF rotation (Table 7.2), and the increase in nitrogen fertilizer inputs in the WF and WWF rotation (Figure 7.2; Table 7.3), resulted in an increase in SOMC stocks at the ecodistrict scale (Table 7.3). The increase in SOMC also resulted in increases in soil nitrogen. The greater nitrogen surplus reported in Table 7.3 was a consequence of the larger levels of nitrogen fertilizer added to the system as carbon credit levels rise. Ecodistrict scale soil sequestered carbon increased primarily as a result of the increased area of tame forage (Table 7.4).

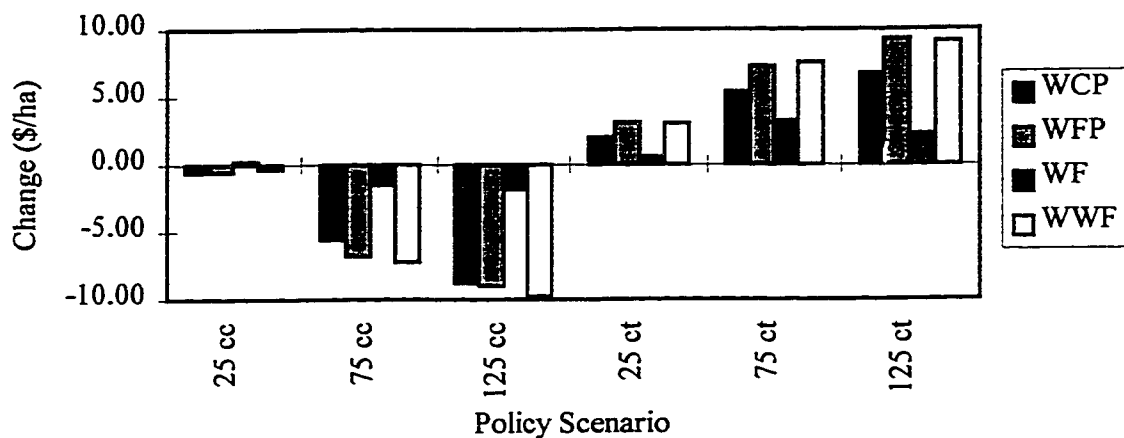


Figure 7.4. Change in economic risk (\$/ha) for the four rotations due to carbon credit (cc), and carbon tax (ct) policies within a non-climate change.

Table 7.3. Changes in soil sustainability indicators due to carbon credit and carbon tax policies under a non-climate change environment.

Policy Scenarios	Soil Indicators				
	SOMC	Soil Nitrogen	Net Nitrogen (kg/ha)	Nitrogen Fertilizer	Soil CO ₂ Emissions
Baseline (kg/ha)	3968.79	29.16	-10.03	1567.59	58214.48
\$25 C credit	+73.14	+0.01	+7.36	+20.75	+78.75
\$75 C credit	+560.37	+0.46	+39.27	+91.30	+1281.56
\$125 C credit	+588.89	+0.29	+62.85	+107.16	+968.65
\$25 C tax	-64.98	-0.02	-19.91	-12.11	-135.27
\$75 C tax	-288.16	-0.29	-42.77	-57.22	-731.03
\$125 C tax	-296.94	-0.23	-62.13	-72.18	-486.25

Table 7.4. Carbon dioxide (CO₂) balance with carbon credit and carbon tax policies under non-climate change environment.

Policy Scenarios	CO ₂ Balance (Mt/50yr)				
	Total Emission	Total Soil Sequestered C	Net Emission	Change in Emissions	Cost of decreased CO ₂ (\$/tonne)
Baseline	11.81	1.29	10.51	-	-
\$25 C credit	11.13	1.65	9.48	-1.04	\$39.77
\$75 C credit	9.98	2.43	7.55	-2.96	\$61.43
\$125 C credit	8.91	2.95	5.96	-4.55	\$80.85
\$25 C tax	11.62	1.37	10.24	-0.27	\$156.30
\$75 C tax	11.18	1.52	9.66	-0.85	\$144.91
\$125 C tax	10.84	1.73	9.11	-1.40	\$141.58

The carbon balance for the Antelope Creek Plain ecodistrict is reported in Table 7.4. The dominant source of CO₂ emissions in these simulations was annually cropped soil. The decrease in total emissions as the carbon credit increased was primarily due to a shift in land allocation toward tame forage production. The total soil sequestered carbon reported in Table 7.4 includes annual cropland and tame forage land (tame hay and improved pasture). The cost of decreased CO₂ emissions was calculated as the total payments made for sequestered carbon divided by the change in net emissions over the 50 year simulation. The simulations reported that the \$25.00 per tonne carbon credit rate was the most cost effective. At this rate nitrogen fertilizer input did not increase dramatically, WCP area increased (low decomposition rate), while WF area decreased (high decomposition rate). Moderate increases in soil CO₂ emissions (Table 7.3) were off-set by moderate decreases in total input embodied carbon emissions (due to decreased annual crop land) and a moderate increase in sequestration (Table 7.4). All of these factors contributed to the relatively low cost per tonne of decreased carbon emissions. At carbon credit rates above \$25/t the shift in annual crop land to WF and WWF resulted in larger increases in soil CO₂ emissions. At these higher carbon credit rates the decrease in net emissions was gained primarily through increased sequestration, which must be paid for under a carbon credit policy, making these policies more expensive.

It should be noted that total emissions calculated in SAM did not include the mineralization of soil carbon caused by the conversion of native land to annual cultivation. It has been estimated that virgin prairie soils contains approximately 80 t/ha of SOMC (Anderson, 1992). The assumed baseline SOMC stock on annually cropped land in SAM was 50 t/ha. SOMC stocks in the Antelope Creek Plain ecodistrict at the end of the 50 year simulation period ranged from 51 to 58 t/ha. Therefore, the conversion of native lands to annual cultivation may be an important source of CO₂ emissions. This is particularly important under a carbon credit policy which provides an economic incentive to convert native land.

5. Land use changes in general benefit indicator species that prefer dense habitat but decrease the area of native habitat in the agroecosystem.

As discussed earlier, the increase in land rent associated with an increase in carbon credit resulted in an economic incentive to convert native land to crop or forage production (Table 7.2). The changes in preferred habitat for the avian indicator species reflects this land use trend. Total habitat area increased for habitat generalists that prefer dense cover (Mallard, Pintail, Gadwall, Blue-Winged Teal, Sprague's Pipit, Bobolink, Western Meadowlark, Clay-Colored Sparrow) (Table 7.5). However, although total preferred habitat increased, the most productive habitat for these species often includes some form of native cover. Therefore, the overall habitat quality for these species may decline (in particular Mallard, Sprague's Pipit, Western Meadowlark and Clay-Colored Sparrow). Those species that are native habitat specialists (Canvasback, Eastern Kingbird) experienced a decline in total habitat area. The species that prefer sparse cover experienced moderate declines (Horned Lark), or slight increases (Killdeer) in habitat area.

The increased area of tame forage land, and decreased area of fallow and cropland indicate that the context for habitat areas have a lower probability of being hostile thereby providing higher quality habitat. The wetland index values showed little trend in response to changing carbon credit levels (Table 7.6). Land uses with

Table 7.5. Percentage change in preferred habitat of avian indicator species due to carbon credit, and carbon tax under non-climate change and climate change environment.

	Avian Indicator Species											
	Mallard	Pintail	Gadwall	BW Teal	Cnvbck	Killdeer	Eastern Kingbird	Horned Lark	Sprague's Pipit	Bobolink	Western Meadowlark	CC Sparrow
Baseline (ha)	150934	146313	147086	147086	2448	219372	31267	213795	43296	43296	146313	150160
\$25 C credit	+4.43	+4.64	+4.61	+4.61	-2.33	+0.28	-2.33	-2.00	+30.76	+30.76	+4.64	+4.46
\$75 C credit	+16.53	+17.21	+17.09	+17.09	-5.07	+2.25	-5.07	-6.57	+121.64	+121.64	+17.21	+16.64
\$125 C credit	+25.06	+26.13	+25.95	+25.95	-8.73	+2.28	-8.73	-10.81	+187.11	+187.11	+26.13	+25.24
\$25 C tax	+1.68	+1.78	+1.76	+1.76	-1.26	+0.62	-1.26	-0.06	+8.97	+8.97	+1.78	+1.70
\$75 C tax	+3.83	+3.95	+3.93	+3.93	+0.18	+0.80	+0.18	-1.28	+29.64	+29.64	+3.95	+3.85
\$125 C tax	+7.87	+8.12	+8.07	+8.07	+0.10	+2.22	+0.10	-1.65	+55.28	+55.28	+8.12	+7.91
Climate Chng (ha)	159325	154287	155131	155131	2669	218470	34089	205516	66775	66775	154287	158482
\$25 C credit	+4.33	+4.50	+4.47	+4.47	-0.80	+0.56	-0.79	-2.12	+23.40	+23.40	+4.50	+4.36
\$75 C credit	+14.26	+14.80	+14.71	+14.71	-2.36	+2.71	-2.36	-5.59	+73.21	+73.21	+14.80	+14.34
\$125 C credit	+21.06	+21.81	+21.68	+21.68	-1.87	+3.14	-1.87	-9.13	+108.41	+108.41	+21.81	+21.18
\$25 C tax	+3.66	+3.76	+3.75	+3.75	+0.44	+1.85	+0.44	+0.33	+14.56	+14.56	+3.76	+3.68
\$75 C tax	+2.90	+3.03	+3.01	+3.01	-1.10	+0.48	-1.10	-1.41	+16.30	+16.30	+3.03	+2.92
\$125 C tax	+4.99	+5.19	+5.15	+5.15	-0.97	+0.76	-0.97	-2.33	+27.07	+27.07	+5.19	+5.02

Table 7.6. Percentage change in wetland index and habitat diversity index due to carbon credit and carbon tax policies under a non-climate change environment.

	Wetland Index (% change)	Habitat Diversity (% change)
Baseline (index)	1.06	0.37
\$25 C credit	-0.38	-5.91
\$75 C credit	+2.66	-13.28
\$125 C credit	+1.69	-16.07
\$25 C tax	+0.47	-1.83
\$75 C tax	-0.75	-5.22
\$125 C tax	+1.36	-8.27

larger SOMC stocks have higher water infiltration rates, and concomitant lower levels of runoff. However, the wetland index does not capture the effect of larger quantities of snow trapped on hay land. Therefore, greater levels of runoff may be associated with increased area of tame forage despite the higher infiltration rate. The habitat diversity index indicates that with increased carbon credit levels habitat diversity decreased (Table 7.6). This is caused by the decreasing area of WCP, WFP and native land, and the increasing dominance of the landscape by hay, WF and WWF.

7.2.2 Carbon Tax

A carbon tax introduces a different set of economic signals to farms than a carbon credit. Carbon tax provides an economic disincentive to management practices that require inputs with high levels of embodied carbon. The inputs targeted by the carbon tax are nitrogen fertilizer, phosphorous fertilizer and chemical pesticides. Within SAM the imposition of a carbon tax results in a parallel and upward shift of the fertilizer and chemical input supply relationships at the industry scale, or an increase in chemical and fertilizer input costs at the farm scale. The magnitude of input price increases are proportional to the quantity of embodied carbon in each of the inputs. The consequences of these changes are:

1. *Land is allocated to production systems that require less inputs.*

The SAM simulations reported that, in general, annual crop production area decreased with concomitant increases in hay production under a carbon tax (Table 7.2). It was assumed that hay input costs are unaffected by a carbon tax. However, the increase in hay land was of a much smaller magnitude than under an equivalent carbon credit. This can be partially explained by the difference between the opportunity cost of cultivated land and hay revenues under the two policy scenarios. Hay revenues did not increase and land rent decreased under a carbon tax. (Table 7.1). At a carbon tax level of \$75/t, the relative increase in hay revenue was \$17.79 (0 (change in hay revenue) - (-\$17.19(change in land rent))). In contrast, at a carbon credit level of \$75.00/tonne the relative increase in hay revenue, with respect to land rent, was \$31.12/ha (\$52.62 - \$21.50) (Table 7.1). Therefore, the economic incentive to convert cultivated land to tame forage production was smaller with a tax than a credit.

Within the annual crop land stock there was a relative shift of land from the more input intensive rotations (WCP) to the less input intensive rotations (WF) (Table 7.2). Increased input costs with a carbon tax resulted in less fertilizer applied to all rotations (Figure 7.2; Table 7.3) causing lower and more variable yields and lower and more variable revenues for each rotation (Figure 7.1; Figure 7.4). It should be noted that these changes in WF revenues were smaller than any of the other rotations due to its limited input requirements. The economic conditions created by the carbon tax resulted in WF being the least economically unattractive crop rotation.

The relative shift of annual cropland to WF, in combination with a decrease in nitrogen fertilizer input, resulted in decreases in SOMC stocks and soil nitrogen at the ecodistrict scale (Table 7.3). The SOMC - yield positive feedback process resulted in decreases in annual crop yields and further decreases in future soil quality.

2. Demand for annual cropland decreased resulting in decreases in land rent.

An increase in input costs and non-increasing gross revenues resulted in a decrease in the derived demand for land for each rotation, proportional to the level of input embodied carbon, and a decrease in overall demand for annual crop. The decrease in land demand resulted in a decrease in land rent (Table 7.1). However, negative feedback prevented a change in land rent from being as great as that under an equivalent carbon credit level. The carbon tax decreased annual crop revenues, (Figure 7.1) which reduced nitrogen fertilizer input (Figure 7.2), yield and future revenues, which decreased land rent. This effect was off-set by decreased annual crop revenues causing an allocation of land to hay production (Table 7.2) which decreased the supply of annual crop land putting upward pressure on land rent.

3. Decrease in CO₂ emissions at an ecodistrict scale.

Total CO₂ emission decreased very little with increasing carbon tax levels (Table 7.4). Soil was the dominant source of CO₂ in SAM simulations making this result primarily a land allocation issue. The greater area maintained in annual crop production, under a carbon tax, and the relative shift to WF which had a higher SOMC decomposition rate, resulted in greater CO₂ contributions per hectare of cropland area. In addition, smaller tame forage area resulted in smaller total quantities of soil sequestered carbon. Therefore, the total decrease in net CO₂ was relatively small compared to carbon credit levels, and the cost of decreased CO₂ was relatively high (Table 7.4). The \$125.00/t tax rate was the least expensive (Table 7.4) due to the area of tame forage land at this tax level. Total emissions decreased by approximately seven percent between \$25.00/t and \$125.00/t. However soil sequestered carbon increased by 26 percent in this same interval.

4. Habitat changes were beneficial to most of the indicator species.

The relatively low land rent under a carbon tax resulted in little economic incentive to convert native land to crop or forage production. Increases in tame forage area, decreases in annual crop land and stable native land areas (Table 7.2) resulted in stable or moderate increases in preferred habitat for most of the indicator species (Table 7.5). The only species that experienced a decline in habitat area was Horned Lark. However, at a landscape scale the total area of preferred habitat for this species was relatively large making a one to two percent decrease in habitat insignificant. Although the total area of annual cultivation decreased, the relative shift in land use toward WF resulted in a smaller decrease in fallow area than at an equivalent carbon credit rate. As a result, habitat context had a higher probability of being a hostile area.

Wetland index values showed little trend in response to carbon tax levels (Table 7.6). Greater variation in wetland index values can be attributed to small differences in precipitation patterns between simulations rather than changes in runoff as a result of land management. The habitat diversity index decreased at a smaller rate than reported under a carbon credit reflecting: (1) maintenance of native area; (2) smaller increases in tame forage land; and (3) smaller decrease in annual crop land (Table 7.6).

7.3 CLIMATE CHANGE

The discussion in this section focuses on simulation output with the implementation of carbon credit and carbon tax policies, under a climate change environment. The discussion will highlight results that are inconsistent with those reported under a non-climate change scenario.

The effect of climate change alone on the simulated Antelope Creek Plain ecodistrict is significant. Lower and more variable precipitation caused the yields of all crops to decrease and become more variable. Lower yields resulted in a decrease in farm revenues. The magnitude of this decrease was proportional to the effect that climate change had on the yields of the crops in the relevant rotation. The decreased revenues

resulted in a downward shift in the derived demand for land in each rotation, and a decrease in the overall demand for land. The decrease in demand for land resulted in a decrease in land rent compared to the non-climate change baseline value for the Antelope Creek Plain ecodistrict (Table 7.7). Lower land rent for annual crop land resulted in less land allocated to annual crop production and more land allocated to tame forage and native cover (Table 7.7). The higher frequency of fallow in WF resulted in relatively smaller decreases in yield, revenues, derived demand for land and the land allocated to this rotation, compared to WWF. Lower annual crop yields, in combination with a relative shift of annual crop land allocation to WF, resulted in lower SOMC stocks, lower soil nitrogen and less nitrogen fertilizer input under climate change. The above conditions also contributed to higher SOMC decomposition rates. For example, while the SOMC stock under climate change was 39 percent of the non-climate change stock, soil CO₂ emissions were 78 percent (Table 7.7). At the ecodistrict scale, the decrease in annual crop land and increase in tame forage land resulted in lower CO₂ emissions, and greater carbon sequestration.

7.3.1 Carbon Credit and Climate Change

A carbon credit under a climate change environment had similar consequences for the system as reported under non-climate change. With increasing carbon credit rates land rent increased (Table 7.8) and net hay revenue increased (Table 7.8). The decrease in annual cropland area exhibited a different pattern than simulated under a carbon credit. Decreased precipitation and soil water resulted in a relative shift of crop land to WF rather than WWF (Table 7.9). The climate change environment provided a competitive advantage to WF, with its higher frequency of water conserving fallow management. The higher net revenues and lower economic risk of the WF rotation caused larger nitrogen fertilizer applications and higher yields, relative to other rotations, as credit rates increased.

The decreased yields and relative shift towards WF, associated with climate change, resulted in smaller increases in SOMC with increasing carbon credit levels

Table 7.7. Comparison of baseline parameters for non-climate change and climate change scenarios.

	Non-climate change	Climate change
Land Rent (\$/ha)	30.60	-15.99
Net Revenue Hay (\$/ha)	62.61	58.45
WCP (ha)	3479	2785
WFP (ha)	1083	685
WF (ha)	93161	87276
WWF (ha)	78429	61009
Hay (ha)	15875	36880
Native (ha)	39139	42670
SOMC (kg/ha)	3968	1533
Soil N (kg/ha)	29.16	24.42
Net N (kg/ha)	-10.03	+5.40
N fertilizer (kg/ha)	1567.59	1203.81
Soil CO ₂ (kg/ha)	58214	45336
Total CO ₂ emission (Mt/50yr)	11.81	8.55
Total Soil C Sequestered (Mt/50yr)	1.29	1.38

(Table 7.10). The greater positive nitrogen balance indicates that a moisture limitation reduced nitrogen use efficiency and yields, which minimized nitrogen exports from the system (Table 7.10). As mentioned earlier, the high rate of SOMC decomposition that is associated with fallow management resulted in greater soil CO₂ emissions than would be predicted from SOMC stocks.

Climate change had little effect on stocks of soil sequestered carbon within SAM. The smaller SOMC stocks in cultivated land were offset by larger areas of land allocated to tame forage. The cost of decreasing CO₂ emissions under climate change was greater than reported under non-climate change (Table 7.11) due to decreases in CO₂ emissions primarily attained by sequestration, rather than reduced inputs. The higher input rotations, WCP and WFP, were not economically viable under climate change and were produced on very small areas in the baseline climate change simulation. As a result these rotations could not be further reduced, thereby reducing input associated CO₂ emissions.

Table 7.8. Change in land rent and hay net revenue (\$/ha) due to carbon credit and carbon tax policies within a climate change environment.

Climate Change Scenario	Land Rent (change in \$/ha)	Net Revenue Hay (change in \$/ha)
Baseline (\$/ha)	-15.99	58.45
\$25 C credit	+5.05	+17.39
\$75 C credit	+13.32	+52.43
\$125 C credit	+21.44	+87.41
\$25 C tax	-6.26	-0.11
\$75 C tax	-16.67	0.06
\$125 C tax	-23.66	0.00

Table 7.9 Percentage change in land use with carbon credit and carbon tax policies under climate change environment.

Climate Change Scenarios	Land Use Categories (% change)					
	WCP	WFP	WF	WWF	Hay	Native
Baseline (ha)	2785	685	87276	61009	36880	42670
\$25 C credit	-12.61	-56.59	-7.17	-12.17	+43.02	-0.79
\$75 C credit	-73.97	-98.77	-22.54	-33.72	+134.46	-2.36
\$125 C credit	-71.26	-97.11	-38.55	-47.98	+197.81	-1.87
\$25 C tax	-26.56	-34.03	-0.80	-6.58	+26.00	0.44
\$75 C tax	-37.32	-84.43	-4.80	-6.61	+30.40	-1.10
\$125 C tax	-48.70	-97.13	-9.13	-10.56	+49.79	-0.97

Table 7.10 Changes in soil sustainability indicators with carbon credit and carbon tax policies under a climate change environment.

Policy Scenarios under Climate Change	Soil Indicators				
	SOMC	Soil Nitrogen	Net Nitrogen (kg/ha)	Nitrogen Fertilizer	Soil CO ₂ Emissions
Baseline (kg/ha)	1533.31	24.42	+5.40	1203.81	45336.52
\$25 C credit	+35.27	+0.10	+19.35	+21.88	+395.08
\$75 C credit	+78.19	-0.07	+54.42	+56.93	+392.11
\$125 C credit	+98.04	-0.21	+84.65	+97.75	+682.65
\$25 C tax	-39.59	-0.01	-11.39	-23.07	-65.09
\$75 C tax	-94.71	-0.02	-22.29	-43.56	-652.70
\$125 C tax	-90.79	+0.23	-46.57	-54.96	-145.09

The relatively stable native land stock, in combination with large areas of tame forage resulted in larger areas of baseline preferred habitat than under non-climate change (Table 7.5). There was a positive relationship between carbon credit rates and habitat for most species, except Horned Lark. In addition, the land use mosaic was beneficial for most species with respect to high priority habitat and habitat context as well.

An increasing wetland index, with increasing carbon credit, reflects the decreased SOMC stock, relative increase in fallow land and the associated increase in erosion rates, causing lower water infiltration rates and greater water runoff (Table 7.12). However, the absolute value of the wetland index was much lower than in the non-climate change scenarios reflecting decreased precipitation. The increased habitat diversity index reflects the change to more balance land use with increasing carbon credit.

7.3.2 Carbon Tax and Climate Change

Within a climate change environment the imposition of a carbon tax resulted in very similar trends as reported for the non-climate change scenario. With increased carbon tax rates, land rent decreased and hay revenues remained constant (Table 7.8), resulting in a re-allocation of land from annual crop to tame forage production (Table 7.9). WF was the most economically attractive annual crop rotation, and as a result is allocated to relatively more land than the other rotations as carbon tax rates increase. In addition, lower crop revenues reduced nitrogen fertilizer use which decreased SOMC stocks and soil nitrogen, and the soil nitrogen deficit increased (Table 7.10). The SOMC stock remained constant as the tax rate increased from \$75.00/t to \$125.00/t with crop yields already so low that further reduction in nitrogen had little effect on residue additions to the soil.

Table 7.11. Carbon dioxide (CO₂) balance with a carbon credit and carbon tax under climate change environment.

CO ₂ Balance (Mt/50yr)					
Policy Scenarios	Total Emission	Total Soil Sequestered C	Net Emission	Change in Emissions	Cost of decreased CO ₂ (\$/tonne)
Climate Change	8.55	1.380	7.17	-	-
\$25 C credit	8.25	1.630	6.62	-0.545	\$74.77
\$75 C credit	7.29	2.298	4.99	-2.177	\$79.18
\$125 C credit	6.68	2.748	3.93	-3.234	\$106.25
\$25 C tax	8.50	1.428	7.07	-0.097	\$381.02
\$75 C tax	8.19	1.565	6.62	-0.547	\$195.87
\$125 C tax	8.02	1.723	6.30	-0.870	\$198.84

Table 7.12. Percentage change in wetland index and habitat diversity index due to carbon credit and carbon tax policies under a climate change environment.

Climate Change	Wetland Index (% change)	Habitat Diversity (% change)
Baseline (index)	0.88	0.31
\$25 C credit	-1.78	2.33
\$75 C credit	0.35	6.44
\$125 C credit	1.35	7.49
\$25 C tax	-0.82	0.60
\$75 C tax	0.96	2.13
\$125 C tax	-0.64	3.53

Habitat diversity increased slightly with an increase in carbon tax, which is inconsistent with the result obtained under non-climate change. This reflects the more balanced land use mosaic that occurred from climate change forces alone (less fallow and cropland, more hay and native land) and the smaller relative changes required to balance this statistic with increasing carbon tax.

7.4 SUSTAINABILITY SYNTHESIS

To determine whether an agroecosystem is sustainable, or on a sustainable pathway, a cross-disciplinary assessment is required. This thesis assessed whether the components of the capital stock, natural and man-made, are being degraded or conserved within the defined agroecosystems. A summary of the changes in selected indicators, relative to the baseline simulations, in the non-climate change and climate change scenarios is presented in Figures 7.5 and 7.6 respectively.

The bars on the positive and negative zones of Figures 7.5 and 7.6 represent conservation or degradation of the capital stock component. The largest measured change relative to the baseline is set equal to one. For the remainder of the simulations the value of the indicator is expressed in relative proportion to one. For example, in Figure 7.5 the largest increase in SOMC stocks was +588 t/ha for the \$125.00/t carbon credit level. In proportion, SOMC increased only 0.12 at \$25.00/t carbon credit ($73.14 / 588.89 = 0.12$).

The land use stocks (fallow, cropland, hay, native) represent changes in habitat. The majority of the indicator species prefer hay and native cover, therefore increases in these land uses were considered positive changes whereas increases in fallow and cropland were considered negative changes. Changes in habitat were not weighted.

The model output indicates the relative sustainability of the scenario simulations. For example, in the non-climate change environment, the \$125.00/t carbon credit policy resulted in the maximum movement toward sustainability in terms of SOMC, net CO₂ emissions, land rent and habitat improvement, with respect to fallow, crop land and tame forage area. However, the \$125.00/t carbon credit decreased native land and increased soil emissions. This result illustrates the difficulty in developing an absolute assessment of agroecosystem sustainability. Native lands are very important in terms of such natural capital components as biodiversity, pollination services and aesthetics. Soil CO₂ is an important contributor to global warming and may signal a decrease in soil function. Based on weighted relative changes in the indicators this system may or may not be considered on the path to sustainability. Similar trade-offs are evident in evaluating all of the non-climate change and climate change scenarios. It is beyond the scope of this

thesis to assign weights to changes in sustainability indicators and to develop an absolute ranking of policies in terms of sustainability. The weighting of indicators would involve evaluating system thresholds for integrity (how low can an indicator get before irreversibly damaging the system) and societal preferences. However, the summary of indicators provided in Figures 7.5 and 7.6 does contribute insight into the benefits and disadvantages of the different policies with respect to agroecosystem sustainability. The nature of the Sustainable Agroecosystem Model ensures that the changes in the indicators do not simply represent a one time cause and effect, but a more dynamic change based on a series of feedbacks from within the system. In this way the output from the model can be evaluated, in context, to assess the changes of the indicators.

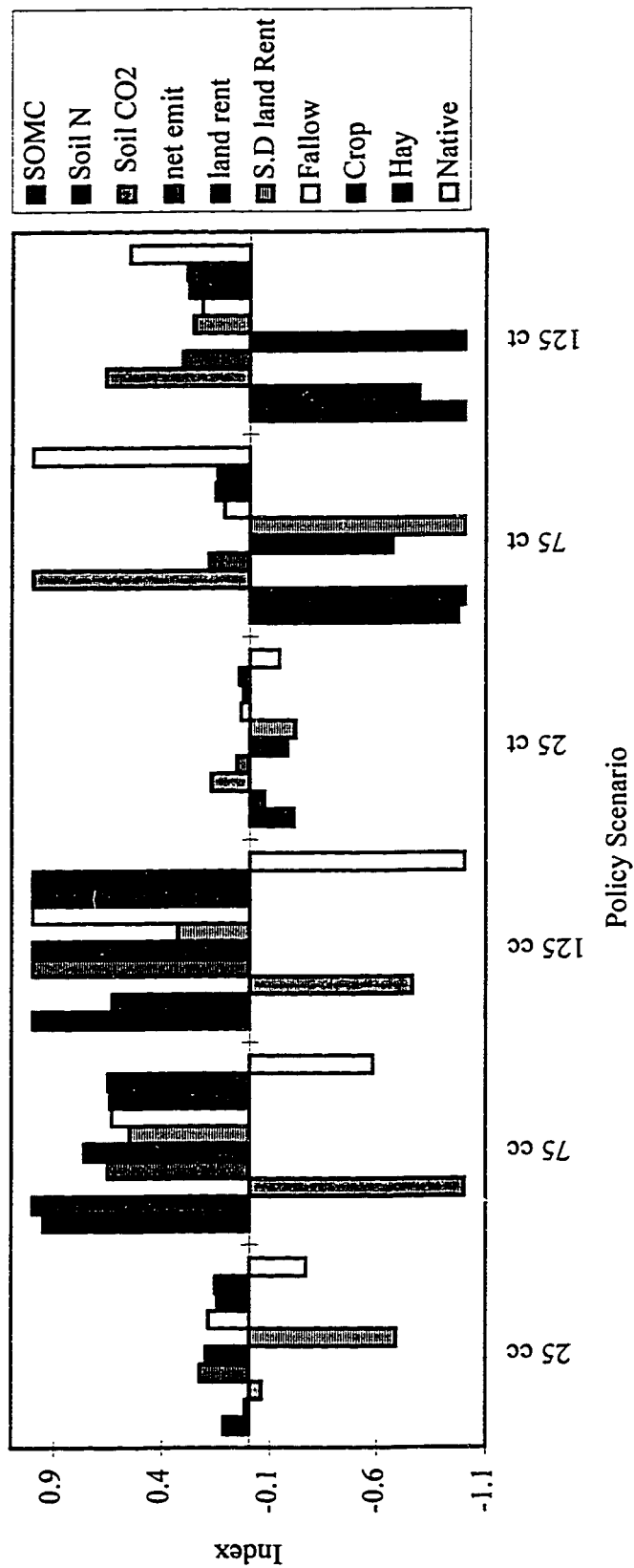


Figure 7.5. Summary of proportional change of sustainability indicators with a carbon credit (cc), and carbon tax (ct) in a non-climate change environment.

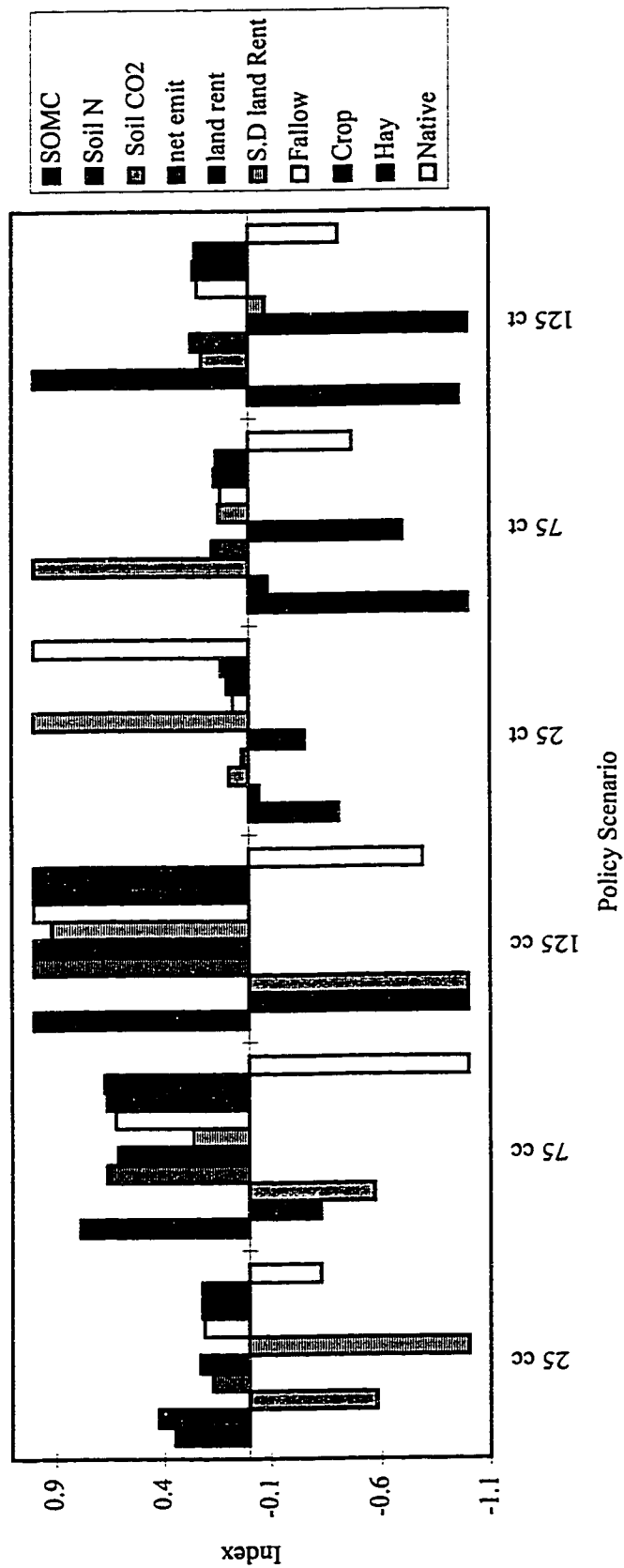


Figure 7.6. Summary of proportional change of sustainability indicators with a carbon credit (cc), and carbon tax (ct) in a climate change environment.

CHAPTER 8

SUMMARY AND CONCLUSIONS

8.1 SUMMARY

Agricultural production systems and their environmental context are viewed within this study as a single but interdependent complex system. The agroecosystems of Saskatchewan are coming under increasing pressure to produce food to meet local and global demands. A number of signals including soil degradation, loss of biodiversity, water pollution and the loss of rural communities are indicating that the agroecosystems are not being managed in a sustainable way. This study examines agroecosystem sustainability in the Brown soil zone of southwestern Saskatchewan.

The primary drivers of environmental degradation within an agroecosystem are the land use and management decisions of farms. Management decisions are made based on a number of social, ethical, agronomic, and economic signals. Input and output prices comprise an important component of the economic signals that farm decision makers react to. However, many components of the agroecosystem can become highly degraded without significant or recognizable changes in price signals. Policy instruments, in a range of forms, have been identified as useful tools to correct for these market failures.

The analysis of the economic, social, or environmental effect of policy has traditionally employed some form of modeling. The limitation of many of these models with respect to agroecosystem change is that they often incorporate a single disciplinary perspective, or include other disciplines as a static constraint. In addition, the models

often assume a single objective function that represents an equilibrium position for the system.

Agroecosystems are complex, temporally dynamic systems, encompassing multiple feedback and linkages between components of the system. Sustainability is characterized in this study as a long term goal, rather than a precisely defined state. The process used to determine whether an agroecosystem is on a sustainable path is informed by the concept of strong sustainability. Strong sustainability is met when each of the components of the natural and man-made capital stock is maintained intact separately.

The framework adopted in this study is a weakly integrated, multi-disciplinary simulation model. The model includes soils, economic, and ecological component models that are linked to represent the critical relationships and feedback processes of a portion of an agroecosystem. The soils model simulates changes in soil quality and yield as interdependent processes constrained by landscape specific biophysical characteristics, and influenced by agricultural management. The economic model simulates land allocation and cropping decisions, based on a profit maximization criteria, on annual cropland, tame forage and native land. The ecological model primarily uses land use output from the economic model to calculate landscape scale changes in habitat. Output from each of these components is used to evaluate changes, over time, of components of the capital stock. The changes in the capital stock give insight into whether the target agroecosystem is on a sustainable path.

To evaluate the sustainability of agroecosystems as influenced by biophysical characteristics, baseline simulations were performed representing four distinct ecodistricts. The sustainability of a single ecodistrict, subjected to economic perturbations, in the form of carbon tax and carbon credit policies, and climatic perturbations, in the form of climate change, was also assessed.

8.2 PRIMARY INSIGHTS

The most important insights into the modeling and sustainability of agroecosystems provided by this study are:

1. The weakly integrated modeling framework is a simple, effective and very flexible structure for setting up dynamic, systems-based models. This structure enables modeling different aspects of a system through the construction and linking of appropriate disciplinary models within the existing framework. In addition, the framework is very amenable to imposing policy or environmental shocks aimed at different aspects of the system.
2. The concept of strong sustainability facilitates an evaluation of sustainability based on absolute changes in components of the capital stock. Strong sustainability does not require that capital stock components be valued using a common currency as is required under the concept of weak sustainability.
3. Feedback processes are very important to changes in agroecosystem structure and function over time. For example, SOMC stocks influence crop yields through soil nutrient and water provision, yields affect SOMC by: (1) altering the quantity and quality of plant residues returned to the soil; and (2) changing revenues which further influence cropping, management, and input decisions. These land use changes feedback to affect yields and SOMC stocks in future time periods.
4. Significant feedback from the ecological systems to the economic and soil systems were difficult to quantify and model. For example, losses of biodiversity may decrease system function including pollination services, pest control, and waste assimilation, all of which have consequences for agricultural production. The difficulty in quantifying the feedback process, in many cases, and quantifying the value of the feedback process to the system, in most cases, is partly due to a lack of data relating system structure and function within an agroecosystem.
5. Biophysical constraints have a large effect on agroecosystem sustainability. Soil texture and climatic characteristics limit the land use and management options available to the farm. Within the model, those ecodistricts with greater biophysical constraints are limited to only two economically viable annual crop rotations. Farms in these landscapes have fewer management options and as a result are less able to respond to future shocks in a sustainable way. The biophysical constraints are

intensified under a climate change environment. This highlights the importance of landscape-appropriate policy initiatives in response to sustainability issues.

6. The relative sustainability of the modeled agroecosystem is enhanced to a greater degree by a carbon credit than a carbon tax policy. This assessment is based on the magnitude of the relative changes in the indicators. However, the CO₂ emissions from soil, native land, and soil nitrogen (under climate change) indicators suggest that these components of the capital stock are being degraded under a carbon credit policy.
7. No policy, under climate change or non-climate change, resulted in system wide improvement or degradation of all components of the capital stock. This result reveals the difficulty of identifying a policy, or set of policies, that will engender a sustainable system using value free criteria. Identifying those capital stock components in which degradation will not be tolerated, or accepted, is a process requiring knowledge of societal preferences, ethical responsibilities, degradation thresholds, and system co-evolution.

8.3 CONCLUSIONS

Evaluating the sustainability of an agroecosystem, or whether the system is on the path towards sustainability, requires a process that captures the relevant linkages and feedback processes between the components of the system. The Sustainable Agroecosystem Model explicitly includes a number of these processes and can thereby capture the co-evolutionary changes in the system.

The relative sustainability of the modeled agroecosystems is limited by the available cropping systems that are economically viable, as dictated by the biophysical constraints of the landscape. Those landscapes where rotations with a lower frequency of fallow are economically viable experience greater improvements in the soil capital stock. However, the tradeoff in these landscapes is greater levels of CO₂ emissions and less wildlife habitat. The biophysical constraints of a landscape are intensified with the imposition of climate change. The modeled annual cropping systems are less

economically viable under climate change resulting in greater quantities of land allocated to tame forage and native cover, which is beneficial to certain components of the capital stock. An overriding characteristic of the model output is that improvement in certain capital stock components, are accompanied by the degradation of others.

The model developed in this thesis is appropriate for the assessment of the relative sustainability of target landscapes. The output provides insight into the improvement or degradation of the capital stock relative to the initial, simulated conditions, and relative to simulations of the same landscape under different policy and climatic scenarios, and to a lesser extent, relative to other landscapes. Output from this model is not appropriate for determining the absolute conditions of a target agroecosystem.

8.4 RECOMMENDATIONS FOR FURTHER RESEARCH

An important result of a modeling process, such as that undertaken in this thesis, is the development of new perspectives on how to improve the process.

1. Annual cropping options should be expanded to include different combinations of the same crops, and alternative crop varieties and management prescriptions. By including only three crop varieties in four different rotations the present model provided very limited management options to the farms, particularly in the drier ecodistricts. The model should be expanded to include such options as longer rotations, organic rotations and rotations that include alternative crops.
2. An important factor affecting the adoption of new conservation management by agricultural firms is the lack of, or unavailability of technical information about these strategies (Stonehouse, 1996). Within the present model, changes in extension policy can not be simulated. The model could be expanded by including “management skill” as a production input. Changes in extension policy would change the level of management skill in agricultural firms and thereby decrease the cost of adopting new production technologies.

3. The limitations inherent in the fixed proportion assumption of the economic model were decreased by developing an endogenous procedure to determine certain input requirements such as nitrogen fertilizer input levels. However, the model would better represent agricultural firm's decision making by increasing the capacity for substitution between selected input pairs in the economic framework.
4. The characterization of farm decision makers as profit maximizers is fairly restrictive and does not capture a number of factors included in the decision making process. While it would likely be fruitless to attempt to include the many ethical, moral, and cultural components of decision making, the existing framework may be relaxed to include some of these components.
5. The model assumes that farm decision makers are myopic with respect to the effect that soil quality has on future rents and land prices. Although some research has shown that this is a reasonable assumption, SAM could be useful in determining the value of increased soil quality, or the user cost of soil degradation.
6. The lack of spatially explicit relationships in the model limits the applicability of model output to highly heterogeneous landscapes. One way to make the model more spatially explicit is to link the procedures to a GIS database following the procedures detailed by Bockstael et al., (1995). This linkage would make land use decisions appropriate to the local land capability. This improvement to the model would also facilitate the development of a much more meaningful ecological component.

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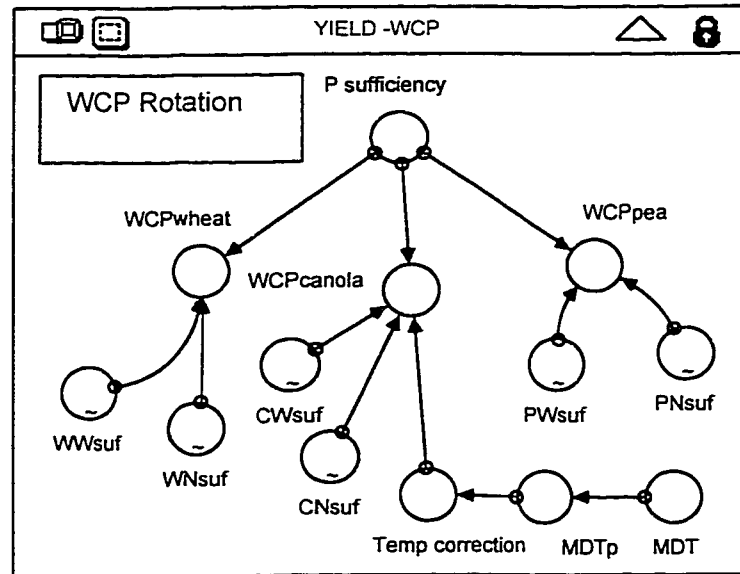
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APPENDIX A

SAM Modeling Sectors and Equations

From STELLA



Yield -WCP

MDT = 2

MDTp = IF TIME=0 THEN MDT ELSE NORMAL(MDT,1)

P_sufficiency = 1-EXP(-.19*(if available_P-2<=0 then 0.00001 Else available_P-2))

Temp_correction = MDTp*-.28

WCPcanola = IF 3.1*P_sufficiency*CNsuf*CWsuf+Temp_correction>0 THEN

3.1*P_sufficiency*CNsuf*CWsuf+Temp_correction ELSE 0

WCPpea = 2.7*P_sufficiency*PNsuf*PWsuf

WCPwheat = 4*P_sufficiency*WNsuf*WWsuf

CNsuf = GRAPH(CavailN)

(0.00, 0.00), (20.0, 0.1), (40.0, 0.4), (60.0, 0.6), (80.0, 0.7), (100, 0.805), (120, 0.855),
(140, 0.91), (160, 0.95), (180, 1.00), (200, 1.00)

CWsuf = GRAPH(WCPavailH2O)

(0.00, 0.00), (5.00, 0.05), (10.0, 0.15), (15.0, 0.35), (20.0, 0.73), (25.0, 0.88), (30.0,
0.95), (35.0, 0.99), (40.0, 0.93), (45.0, 0.78), (50.0, 0.6)

PNsuf = GRAPH(PavailN)

(0.00, 0.23), (2.50, 0.33), (5.00, 0.43), (7.50, 0.53), (10.0, 0.63), (12.5, 0.73), (15.0,
0.83), (17.5, 0.93), (20.0, 1.00), (22.5, 1.00), (25.0, 1.00)

PWsuf = GRAPH(WCPavailH2O)

(0.00, 0.00), (5.00, 0.00), (10.0, 0.1), (15.0, 0.25), (20.0, 0.5), (25.0, 0.8), (30.0, 0.95),
(35.0, 1.00), (40.0, 0.99), (45.0, 0.95), (50.0, 0.7)

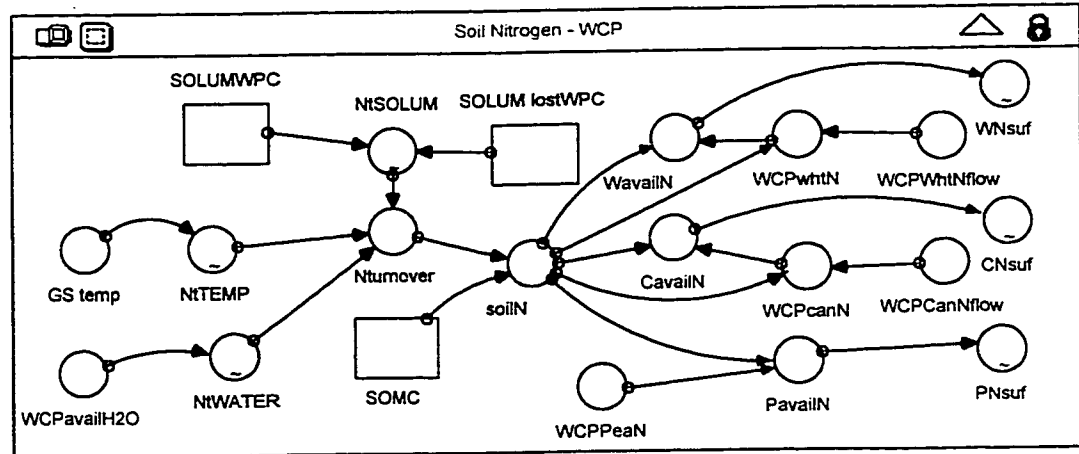
WNsuf = GRAPH(WavailN)

(0.00, 0.00), (10.0, 0.19), (20.0, 0.34), (30.0, 0.47), (40.0, 0.57), (50.0, 0.65), (60.0,
0.72), (70.0, 0.77), (80.0, 0.82), (90.0, 0.84), (100, 0.88), (110, 0.9), (120, 0.92), (130,

0.93), (140, 0.95), (150, 0.96), (160, 0.97), (170, 0.975), (180, 0.98), (190, 0.99), (200, 0.995)

WWsuf = GRAPH(WCPavailH2O)

(0.00, 0.00), (2.50, 0.04), (5.00, 0.08), (7.50, 0.12), (10.0, 0.16), (12.5, 0.23), (15.0, 0.385), (17.5, 0.535), (20.0, 0.665), (22.5, 0.775), (25.0, 0.86), (27.5, 0.92), (30.0, 0.96), (32.5, 0.98), (35.0, 0.995), (37.5, 0.99), (40.0, 0.955), (42.5, 0.895), (45.0, 0.805), (47.5, 0.705), (50.0, 0.6)



Soil Nitrogen - WCP

$SOLUWPC(t) = SOLUWPC(t - dt)$

INIT SOLUWPC = 60

$SOLU_lostWPC(t) = SOLU_lostWPC(t - dt)$

INIT SOLU_lostWPC = 0

$SOMC(t) = SOMC(t - dt)$

INIT SOMC = 50000 {for brown soil zone, Voroney et al., 1981}

$CavailN = soilN + WCPcanN + 20$

GS_temp = 20

$Nturnover = NtSOLU * NtTEMP * NtWATER$

$PavailN = soilN + WCPPeaN$

$soilN = (Nturnover * (SOMC/10))$

$WavailN = soilN + WCPwhtN$

$WCPavailH2O = total_water_ * Storage_factor$

$WCPcanN = \text{if } soilN + WCPcanNflow + 20 < 200 \text{ then } WCPcanNflow \text{ else } 200 - soilN$

$WCPcanNflow = CanN - DELAY(CanN, 1, 50)$

$WCPPeaN = 11.2$ {kg/ha, static application rate of Nitrogen on stubble peas}

$WCPwhtN = \text{if } soilN + WCPwhtNflow < 200 \text{ then } WCPwhtNflow \text{ else } 200 - soilN$

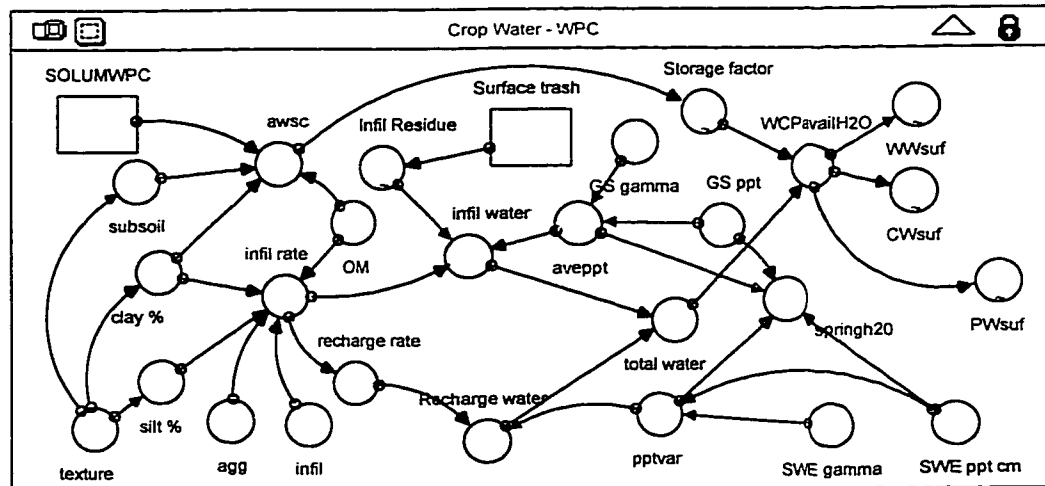
$WCPwhtNflow = WhtN - DELAY(WhtN, 1, 50)$

$CNsuf = GRAPH(CavailN)$

(0.00, 0.00), (20.0, 0.1), (40.0, 0.4), (60.0, 0.6), (80.0, 0.7), (100, 0.805), (120, 0.855),

(140, 0.91), (160, 0.95), (180, 1.00), (200, 1.00)

$NtSOLU = GRAPH(1 - (SOLU_lostWPC / (SOLU_lostWPC + SOLUWPC)))$
 (0.00, 0.0015), (0.0833, 0.00269), (0.167, 0.00393), (0.25, 0.00515), (0.333, 0.00609),
 (0.417, 0.00704), (0.5, 0.00798), (0.583, 0.00906), (0.667, 0.0102), (0.75, 0.0119),
 (0.833, 0.0134), (0.917, 0.0145), (1.00, 0.0149)
 $NtTEMP = GRAPH(GS_temp)$
 (5.00, 0.39), (7.08, 0.505), (9.17, 0.61), (11.3, 0.72), (13.3, 0.82), (15.4, 0.89), (17.5,
 0.945), (19.6, 0.975), (21.7, 0.975), (23.8, 0.98), (25.8, 0.985), (27.9, 0.99), (30.0, 0.995)
 $NtWATER = GRAPH(WCPavailH2O)$
 (5.00, 0.055), (8.33, 0.125), (11.7, 0.19), (15.0, 0.275), (18.3, 0.345), (21.7, 0.435),
 (25.0, 0.51), (28.3, 0.59), (31.7, 0.685), (35.0, 0.765), (38.3, 0.855), (41.7, 0.94), (45.0,
 1.00)
 $PNsuf = GRAPH(PavailN)$
 (0.00, 0.23), (2.50, 0.33), (5.00, 0.43), (7.50, 0.53), (10.0, 0.63), (12.5, 0.73), (15.0,
 0.83), (17.5, 0.93), (20.0, 1.00), (22.5, 1.00), (25.0, 1.00)
 $WNsuf = GRAPH(WavailN)$
 (0.00, 0.00), (10.0, 0.19), (20.0, 0.34), (30.0, 0.47), (40.0, 0.57), (50.0, 0.65), (60.0,
 0.72), (70.0, 0.77), (80.0, 0.82), (90.0, 0.84), (100, 0.88), (110, 0.9), (120, 0.92), (130,
 0.93), (140, 0.95), (150, 0.96), (160, 0.97), (170, 0.975), (180, 0.98), (190, 0.99), (200,
 0.995)



Crop Water - WCP

$SOLUWPC(t) = SOLUWPC(t - dt)$

INIT SOLUWPC = 60

$Surface_trash(t) = Surface_trash(t - dt)$

INIT Surface_trash = 4200 {assumed initial surface trash value in kg/ha}

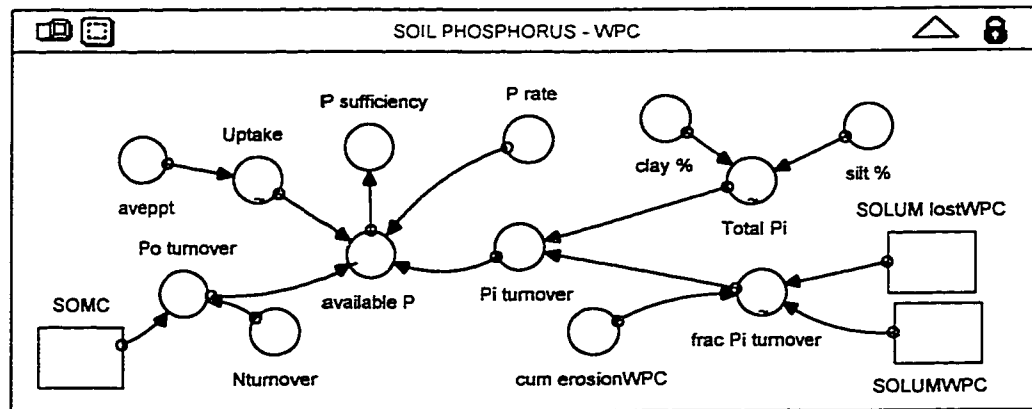
agg = 1

aveppt = IF TIME=0 THEN GS_ppt ELSE GS_gamma

```

awsc = ((1.3*SOLUMWPC/100)*(0.99+(.5595*clay_%)+(2.386*(.588*OM))-
(.753+(.334*clay_%)+(1.855*(.588*OM))))+(1.4*(100-SOLUMWPC)/100)*subsoil
clay_% = if texture=1 then 60 else if texture=2 then 35 else if texture=3 then 25 else if
texture=4 then 10 else 5
GS_gamma = M19Sd5
GS_ppt = 18
infil = 1
infil_rate_ = 1-((2.1*((silt_*(100-clay_%))^1.14)*10^(-4)*(12-OM)+3.25*(agg-
2)+2.5*(infil-3))/100)
infil_water = aveppt*infil_rate_+(aveppt*(1-infil_rate_)*Infil_Residue)
OM = (SOMC*1.742)/(SOLUMWPC*1300)
pptvar = IF TIME=0 THEN SWE_ppt_cm ELSE SWE_gamma
recharge_rate = infil_rate_*.5
Recharge_water = pptvar*recharge_rate
silt_% = if texture=1 then 30 else if texture=2 then 35 else if texture=3 then 45 else if
texture=5 then 30 else 10
springh20 = (aveppt+pptvar)/(GS_ppt+SWE_ppt_cm)
subsoil = if texture=1 then 17.7 else if texture=2 then 14 else if texture=3 then 12.1 else
if texture=4 then 8.8 else if texture=5 then 5.1 else 0
SWE_gamma = M17Sd5
SWE_ppt_cm = 14
texture = 2
total_water_ = infil_water+Recharge_water
WCPavailH2O = total_water_*Storage_factor
CWsuf = GRAPH(WCPavailH2O)
(0.00, 0.00), (5.00, 0.05), (10.0, 0.15), (15.0, 0.35), (20.0, 0.73), (25.0, 0.88), (30.0,
0.95), (35.0, 0.99), (40.0, 0.93), (45.0, 0.78), (50.0, 0.6)
Infil_Residue = GRAPH((Surface_trash/.45))
(0.00, 0.00), (375, 0.161), (750, 0.435), (1125, 0.595), (1500, 0.72), (1875, 0.8), (2250,
0.835), (2625, 0.89), (3000, 0.925), (3375, 0.95), (3750, 0.97), (4125, 0.985), (4500,
1.00)
PWsuf = GRAPH(WCPavailH2O)
(0.00, 0.00), (5.00, 0.00), (10.0, 0.1), (15.0, 0.25), (20.0, 0.5), (25.0, 0.8), (30.0, 0.95),
(35.0, 1.00), (40.0, 0.99), (45.0, 0.95), (50.0, 0.7)
Storage_factor = GRAPH(awsc)
(0.00, 0.285), (1.92, 0.495), (3.83, 0.54), (5.75, 0.575), (7.67, 0.605), (9.58, 0.64), (11.5,
0.695), (13.4, 0.755), (15.3, 0.825), (17.3, 0.89), (19.2, 1.00), (21.1, 1.00), (23.0, 1.00)
WWsuf = GRAPH(WCPavailH2O)
(0.00, 0.00), (2.50, 0.04), (5.00, 0.08), (7.50, 0.12), (10.0, 0.16), (12.5, 0.23), (15.0,
0.385), (17.5, 0.535), (20.0, 0.665), (22.5, 0.775), (25.0, 0.86), (27.5, 0.92), (30.0, 0.96),
(32.5, 0.98), (35.0, 0.995), (37.5, 0.99), (40.0, 0.955), (42.5, 0.895), (45.0, 0.805), (47.5,
0.705), (50.0, 0.6)

```

Soil Phosphorus - WPC

$SOLUMWPC(t) = SOLUMWPC(t - dt)$

INIT SOLUMWPC = 60

$SOLUM_lostWPC(t) = SOLUM_lostWPC(t - dt)$

INIT SOLUM_lostWPC = 0

$SOMC(t) = SOMC(t - dt)$

INIT SOMC = 50000 {for brown soil zone, Voroney et al., 1981}

$available_P = (Po_turnover + Pi_turnover + ((P_rate * 1000) * .25)) * Uptake$

$aveppt = \text{IF TIME}=0 \text{ THEN GS_ppt ELSE GS_gamma}$

$clay_ \% = \text{if texture}=1 \text{ then } 60 \text{ else if texture}=2 \text{ then } 35 \text{ else if texture}=3 \text{ then } 25 \text{ else if texture}=4 \text{ then } 10 \text{ else } 5$

$cum_erosionWPC = (time * ErosionWPC)$

$Nturnover = NtSOLUM * NtTEMP * NtWATER$

$Pi_turnover = (Total_Pi * frac_Pi_turnover)$

$Po_turnover = Nturnover * (SOMC / 10) * .1$

$P_rate = .0336$ {t/ha, application rate of actual P for all crops}

$P_sufficiency = 1 - \text{EXP}(-.19 * (\text{if } available_P - 2 \leq 0 \text{ then } 0.00001 \text{ Else } available_P - 2))$

$silt_ \% = \text{if texture}=1 \text{ then } 30 \text{ else if texture}=2 \text{ then } 35 \text{ else if texture}=3 \text{ then } 45 \text{ else if texture}=5 \text{ then } 30 \text{ else } 10$

$frac_Pi_turnover =$

$GRAPH((cum_erosionWPC / 130) / (SOLUMWPC + SOLUM_lostWPC))$

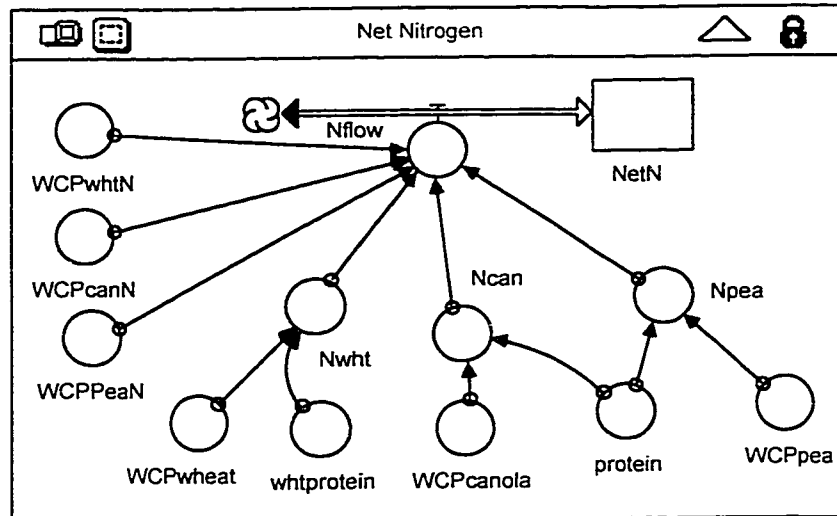
(0.00, 0.04), (0.0833, 0.039), (0.167, 0.0382), (0.25, 0.036), (0.333, 0.0338), (0.417, 0.0292), (0.5, 0.019), (0.583, 0.012), (0.667, 0.007), (0.75, 0.0036), (0.833, 0.0008), (0.917, 0.0002), (1.00, 0.0002)

$Total_Pi = GRAPH(clay_ \% + silt_ \%)$

(0.00, 468), (8.33, 650), (16.7, 800), (25.0, 950), (33.3, 1070), (41.7, 1210), (50.0, 1310), (58.3, 1410), (66.7, 1510), (75.0, 1620), (83.3, 1730), (91.7, 1850), (100, 2000)

$Uptake = GRAPH(aveppt)$

(0.00, 0.13), (3.75, 0.21), (7.50, 0.28), (11.3, 0.355), (15.0, 0.445), (18.8, 0.515), (22.5, 0.605), (26.3, 0.71), (30.0, 0.795), (33.8, 0.875), (37.5, 0.945), (41.3, 1.00), (45.0, 1.00)



Net Nitrogen

$NetN(t) = NetN(t - dt) + (Nflow) * dt$

INIT NetN = 0

INFLOWS:

$Nflow = (WCPwhtN - Nwht) + (WCPcanN - Ncan) + (WCPPeaN - Npea)$

$Ncan = protein / 5.7 / 100 * WCPcanola * 1000 \text{ \{kg/ha\}}$

$Npea = protein / 5.7 / 100 * WCPpea * 1000 * 0.2 \text{ \{kg/ha\}}$

$Nwht = whtprotein / 5.7 / 100 * WCPwheat * 1000 \text{ \{kg/ha\}}$

protein = 22

$WCPcanN = \text{if } soilN + WCPCanNflow + 20 < 200 \text{ then } WCPCanNflow \text{ else } 200 - soilN$

$WCPcanola = \text{IF } 3.1 * P_sufficiency * CNsuf * CWsuf + Temp_correction > 0 \text{ THEN } 3.1 * P_sufficiency * CNsuf * CWsuf + Temp_correction \text{ ELSE } 0$

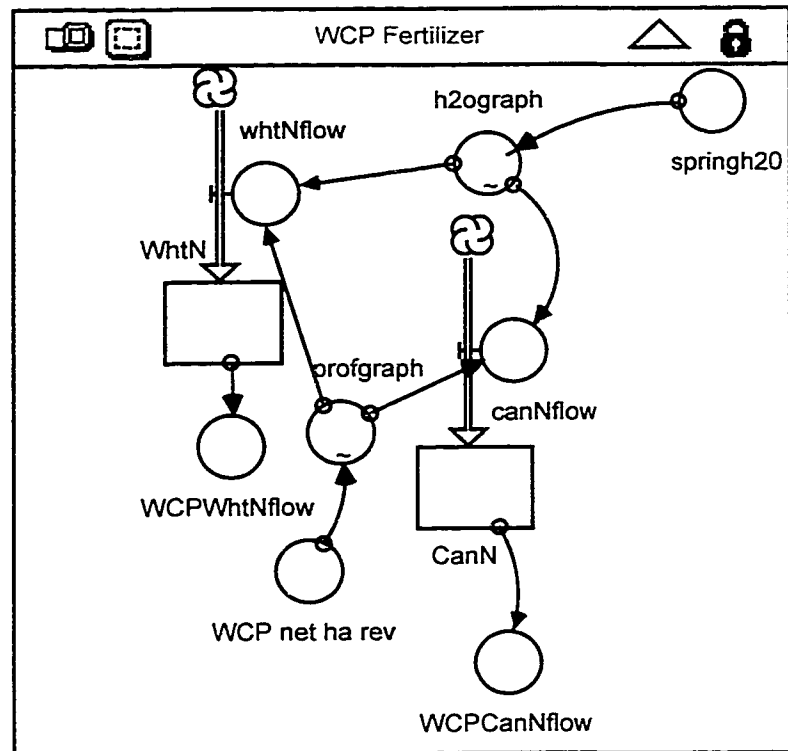
$WCPpea = 2.7 * P_sufficiency * PNsuf * PWsuf$

$WCPPeaN = 11.2 \text{ \{kg/ha, static application rate of Nitrogen on stubble peas\}}$

$WCPwheat = 4 * P_sufficiency * WNsuf * WWsuf$

$WCPwhtN = \text{if } soilN + WCPWhtNflow < 200 \text{ then } WCPWhtNflow \text{ else } 200 - soilN$

whtprotein = 13



WCP Fertilizer

$\text{CanN}(t) = \text{CanN}(t - dt) + (\text{canNflow}) * dt$

INIT CanN = 100

INFLOWS:

$\text{canNflow} = 180 * \text{h2ograph} * \text{profgraph}$

$\text{WhtN}(t) = \text{WhtN}(t - dt) + (\text{whtNflow}) * dt$

INIT WhtN = 100

INFLOWS:

$\text{whtNflow} = (200 * \text{h2ograph} * \text{profgraph})$

$\text{springh20} = (\text{aveppt} + \text{pptvar}) / (\text{GS_ppt} + \text{SWE_ppt_cm})$

$\text{WCPCanNflow} = \text{CanN} - \text{DELAY}(\text{CanN}, 1, 50)$

$\text{WCPWhNflow} = \text{WhtN} - \text{DELAY}(\text{WhtN}, 1, 50)$

$\text{WCP_net_ha_rev} = \text{WCP_ha_rev} -$

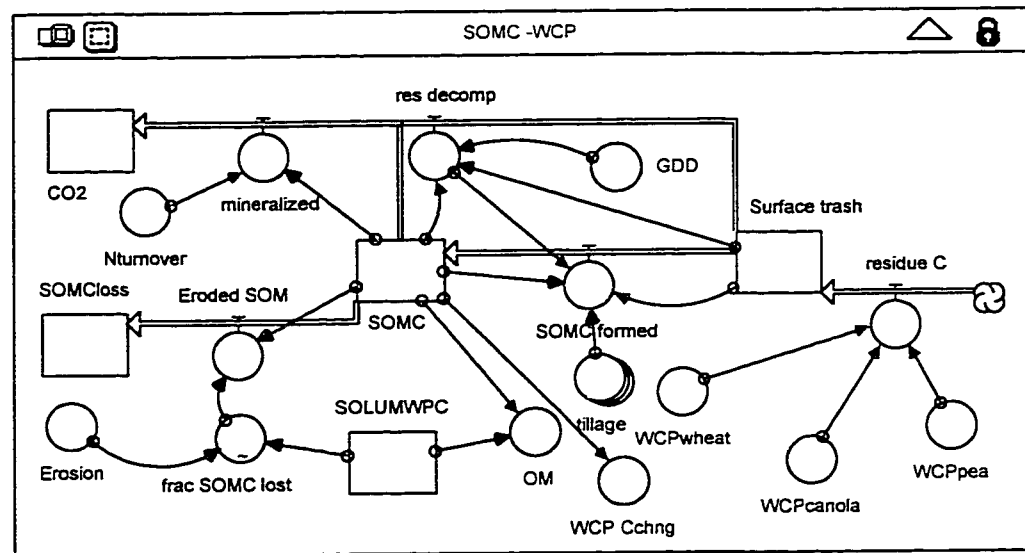
$(\text{other_WCP_input_p} + \text{WCP_ha_chem_P} + (\text{MAX}(\text{Equil_Land_P}, 0)) + \text{WCP_N_cost} + \text{WCP_P_cost})$

$\text{h2ograph} = \text{GRAPH}(\text{springh20})$

(0.00, 0.25), (0.25, 0.25), (0.5, 0.25), (0.75, 0.38), (1.00, 0.5), (1.25, 0.75), (1.50, 1.00), (1.75, 1.00), (2.00, 1.00)

$\text{profgraph} = \text{GRAPH}(\text{WCP_net_ha_rev})$

(-300, 0.255), (-285, 0.25), (-269, 0.25), (-254, 0.25), (-238, 0.25), (-223, 0.255), (-208, 0.255), (-192, 0.25), (-177, 0.255), (-162, 0.255), (-146, 0.255), (-131, 0.25), (-115, 0.25), (-100, 0.25), (-84.6, 0.25), (-69.2, 0.25), (-53.8, 0.25), (-38.5, 0.25), (-23.1, 0.3), (-7.69, 0.45), (7.69, 0.585), (23.1, 0.725), (38.5, 0.83), (53.8, 0.84), (69.2, 0.83), (84.6, 0.83), (100.0, 0.83), (115, 0.83), (131, 0.83), (146, 0.84), (162, 0.84), (177, 0.84), (192, 0.84), (208, 0.84), (223, 0.84), (238, 0.84), (254, 0.845), (269, 0.845), (285, 0.845), (300, 0.85)



SOMC -WCP

$CO_2(t) = CO_2(t - dt) + (mineralized + res_decomp) * dt$

INIT CO2 = 0

INFLOWS:

$mineralized = SOMC * (Nturnover)$

$res_decomp = \text{if } SOMC > INIT(SOMC) \text{ then } Surface_trash - Surface_trash * (EXP(1.95 * .2 * -0.0004 * GDD)) \text{ else } 0$

$SOLUMWPC(t) = SOLUMWPC(t - dt)$

INIT SOLUMWPC = 60

$SOMC(t) = SOMC(t - dt) + (SOMC_formed - mineralized - Eroded_SOM) * dt$

INIT SOMC = 50000 {for brown soil zone, Voroney et al., 1981}

INFLOWS:

$SOMC_formed = \text{if } SOMC < INIT(SOMC) * 1.25 \text{ then } (Surface_trash - res_decomp) * \text{tillage}[wheat_canola_peas] \text{ else } (Surface_trash - res_decomp)$

$* \text{tillage}[wheat_canola_peas] - 0.05$

$* \text{tillage}[wheat_canola_peas] - 0.05$

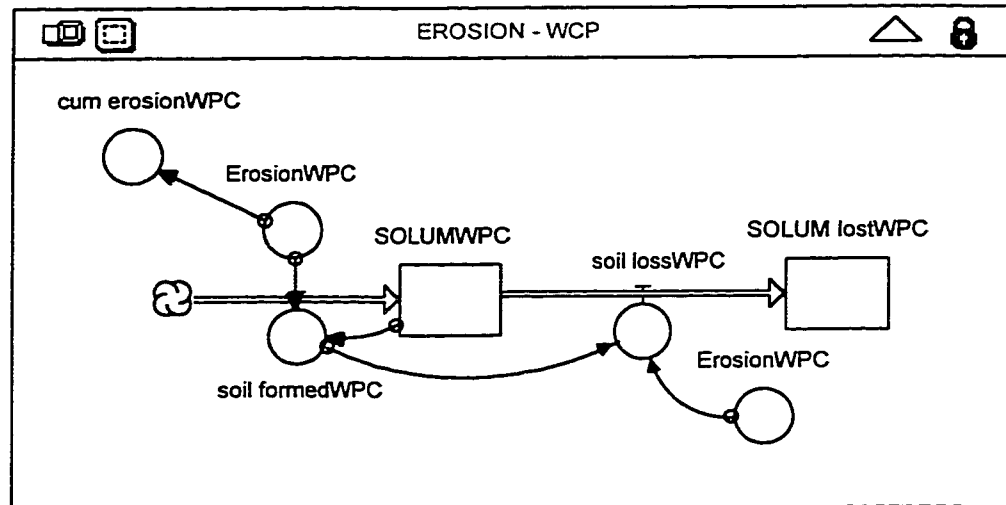
OUTFLOWS:

$mineralized = SOMC * (Nturnover)$

```

Eroded_SOM = frac_SOMC_lost*SOMC
SOMCloss(t) = SOMCloss(t - dt) + (Eroded_SOM) * dt
INIT SOMCloss = 0
INFLOWS:
Eroded_SOM = frac_SOMC_lost*SOMC
Surface_trash(t) = Surface_trash(t - dt) + (residue_C - res_decomp - SOMC_formed) *
dt
INIT Surface_trash = 4200 {assumed initial surface trash value in kg/ha}
INFLOWS:
residue_C = (WCPwheat*1000
*.45*2.2+WCPcanola*1000*.45*2.2+WCPpea*1000*.45*1.2)/3
OUTFLOWS:
res_decomp = if SOMC > INIT(SOMC) then Surface_trash-Surface_trash
*(EXP(1.95*.2*-0.0004*GDD)) else 0
SOMC_formed = if SOMC < INIT(SOMC) * 1.25 then (Surface_trash - res_decomp) *
tillage[wheat_canola_peas] else (Surface_trash - res_decomp)
*tillage[wheat_canola_peas] -0.05
Erosion = 0
GDD = 1507
Nturnover = NtSOLUM*NtTEMP*NtWATER
OM = (SOMC*1.742)/(SOLUMWPC*1300)
tillage[wheat_fallow] = 0.13
tillage[wheat_fallow_peas] = 0.2
tillage[wheat_canola_peas] = 0.2
tillage[wheat_wheat_fallow] = 0.15
WCPcanola = IF 3.1*P_sufficiency*CNsuf*CWsuf+Temp_correction>0 THEN
3.1*P_sufficiency*CNsuf*CWsuf+Temp_correction ELSE 0
WCPpea = 2.7*P_sufficiency*PNsuf*PWsuf
WCPwheat = 4*P_sufficiency*WNsuf*WWsuf
WCP_Cchng = SOMC-DELAY(SOMC,1,INIT(SOMC))
frac_SOMC_lost = GRAPH((Erosion/130)/SOLUMWPC)
(0.00, 0.00), (0.0833, 0.195), (0.167, 0.315), (0.25, 0.45), (0.333, 0.545), (0.417, 0.63),
(0.5, 0.71), (0.583, 0.775), (0.667, 0.84), (0.75, 0.89), (0.833, 0.94), (0.917, 0.98), (1.00,
1.00)

```



Erosion - WCP

$SOLUMWPC(t) = SOLUMWPC(t - dt) + (soil_formedWPC - soil_lossWPC) * dt$

INIT SOLUMWPC = 60

INFLOWS:

$soil_formedWPC = \text{if } SOLUMWPC \leq 10 \text{ then } ErosionWPC/130 \text{ else } 0$

OUTFLOWS:

$soil_lossWPC = (ErosionWPC/130) - soil_formedWPC$

$SOLUM_lostWPC(t) = SOLUM_lostWPC(t - dt) + (soil_lossWPC) * dt$

INIT SOLUM_lostWPC = 0

INFLOWS:

$soil_lossWPC = (ErosionWPC/130) - soil_formedWPC$

$cum_erosionWPC = (time * ErosionWPC)$

ErosionWPC = 0


```

WCPPeaN = 11.2 {kg/ha, static application rate of Nitrogen on stubble peas}
WCPwheat = 4*P_sufficiency*WNsuf*WWsuf
WCPwhtN = if soilN+WCPWhtNflow<200 then WCPWhtNflow else 200-soilN
WCP_Cchng = SOMC-DELAY(SOMC,1,INIT(SOMC))
WCP_ch_C = C_tax*.2856 {$/ha due to carbon tax, .2856 represents the embodied C in
the pesticide inputs for the WCP rotation}
WCP_Cpay = IF(WCP_Cchng/1000)*C_credit>0 THEN (WCP_Cchng/1000)*C_credit
ELSE 0
WCP_ha_chem_P =
((Can_chem_P+SPea_chem_P+Wheat_CC_chem_P)/3)+WCP_ch_C
WCP_ha_rev =
(((Wheat_Price*WCPwheat)+(Canola_Price*WCPcanola)+(Pea_Price*WCPpea))/3)+W
CP_Cpay
WCP_net_ha_rev = WCP_ha_rev-
(other_WCP_input_p+WCP_ha_chem_P+(MAX(Equil_Land_P,0))+WCP_N_cost+WC
P_P_cost)
WCP_N_cost =
(((WCPcanN+WCPPeaN+WCPwhtN)*N_fert_price)/1000/3)+(((WCPcanN+WCPPeaN
+WCPwhtN)*C_rate_N)/3)
WCP_P_cost = (P_cost+P_cost+P_cost)/3 {$/ha, average cost of P input for WCP
rotation}
WCP_rot_profit = WCP_net_ha_rev*WCP_hectares
Wheat_CC_chem_P = base_wheat_CC_chem_P+Bo_wheat_CC_chem_P
Wheat_Price = base_wheat_price+Bo_wheat_price

```



```

base_fert_cost[wheat_fallow] =
base_N_cost[wheat_fallow]+base_P_cost[wheat_fallow]
base_fert_cost[wheat_fallow_peas] =
base_N_cost[wheat_fallow_peas]+base_P_cost[wheat_fallow_peas]
base_fert_cost[wheat_canola_peas] =
base_N_cost[wheat_canola_peas]+base_P_cost[wheat_canola_peas]
base_fert_cost[wheat_wheat_fallow] =
base_N_cost[wheat_wheat_fallow]+base_P_cost[wheat_wheat_fallow]
base_ha_revenue[wheat_fallow] =
(((base_WFw*base_wheat_price))/2)+(0*(base_WCPc+base_canola_price+base_pea_pr
ice+base_WCPp+base_WCPw+base_WFPp+base_WFPw+base_WWFsw+base_WWFw)
)
base_ha_revenue[wheat_fallow_peas] =
(((base_WFPw*base_wheat_price)+(base_WFPp*base_pea_price))/3)+(0*(base_canola
_price+base_WCPp+base_WFw+base_WCPc+base_WCPw+base_WWFsw+base_WWF
w))
base_ha_revenue[wheat_canola_peas] =
(((base_wheat_price*base_WCPw)+(base_canola_price*base_WCPc)+(base_pea_price*b
ase_WCPp))/3)+(0*(base_WFw+base_WFPp+base_WFPw+base_WWFsw+base_WWF
w))
base_ha_revenue[wheat_wheat_fallow] =
(((base_wheat_price*base_WWFw)+(base_wheat_price*base_WWFsw))/3)+(0*(base_W
CPw+base_canola_price+base_pea_price+base_WCPc+base_WCPp+base_WFPp+base
_WFPw+base_WFw))
base_land_price = 49.42 {per hectare, investment price per ha of land sask ag and food
crop planning guide 1997}
base_pea_price = 181 {$ per tonne}
base_WCPc = INIT(WCPcanola)
base_WCPp = INIT(WCPpea)
base_WFPp = INIT(WFPpeas)
base_WFPw = INIT(WFPwheat)
base_WFw = INIT(WFwheat)
base_wheat_price = 167 {$ per tonne}
base_WWFsw = INIT(WWFstubwheat)
base_WWFw = INIT(WWFwheat)
Cultivated_ha = Total_Cult_Ha
ha_cropped[wheat_fallow] = WF_rot+(0*(WCP_rot+WFP_rot+WWF_rot))
ha_cropped[wheat_fallow_peas] = WFP_rot+(0*(WCP_rot+WF_rot+WWF_rot))
ha_cropped[wheat_canola_peas] = WCP_rot+(0*(WF_rot+WFP_rot+WWF_rot))
ha_cropped[wheat_wheat_fallow] = WWF_rot+(0*(WCP_rot+WFP_rot+WF_rot))
ha_revenue[wheat_fallow] =
WF_ha_rev+(WFP_ha_rev+WCP_ha_rev+WWF_ha_rev)*0
ha_revenue[wheat_fallow_peas] =
WFP_ha_rev+(WF_ha_rev+WCP_ha_rev+WWF_ha_rev)*0

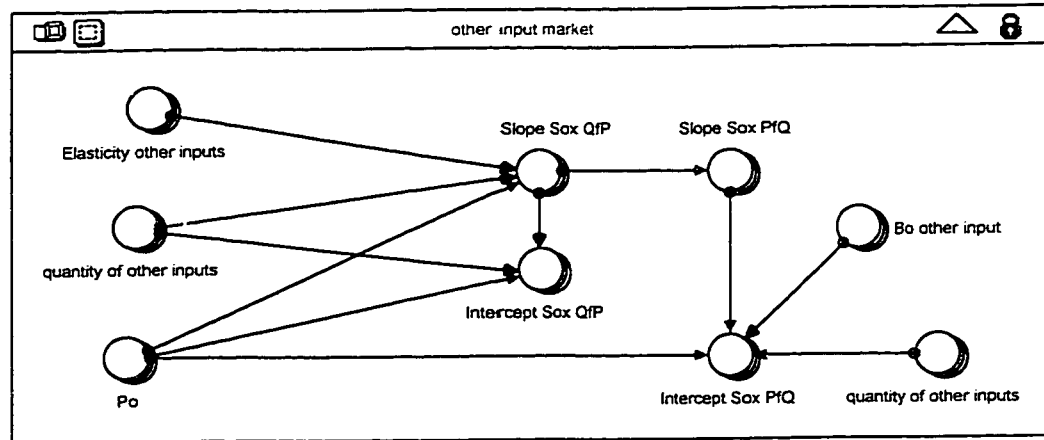
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ha_revenue[wheat_canola_peas] =
WCP_ha_rev+(WF_ha_rev+WFP_ha_rev+WWF_ha_rev)*0
ha_revenue[wheat_wheat_fallow] =
WWF_ha_rev+(0*(WCP_ha_rev+WFP_ha_rev+WF_ha_rev))
Land_Prop[wheat_fallow] = .55 {proportion of cultivated land at t=0 that is dedicated to
the WF rotation}
Land_Prop[wheat_fallow_peas] = .03
Land_Prop[wheat_canola_peas] = .02
Land_Prop[wheat_wheat_fallow] = .40
Po[wheat_fallow] = base_ha_revenue[wheat_fallow]-
(base_Fert_cost[wheat_fallow]+base_land_price+base_Chem_cost[wheat_fallow])
{assumption of the FPM that total revenue will be equal to total costs}
Po[wheat_fallow_peas] = MAX(120,base_ha_revenue[wheat_fallow_peas]-
(base_Fert_cost[wheat_fallow_peas]+base_land_price+base_Chem_cost[wheat_fallow_
peas])) {assumption of the FPM that total revenue will be equal to total costs}
Po[wheat_canola_peas] = MAX(135,base_ha_revenue[wheat_canola_peas]-
(base_Fert_cost[wheat_canola_peas]+base_land_price+base_Chem_cost[wheat_canola_
peas])) {assumption of the FPM that total revenue will be equal to total costs}
Po[wheat_wheat_fallow] = base_ha_revenue[wheat_wheat_fallow]-
(base_Fert_cost[wheat_wheat_fallow]+base_land_price+base_Chem_cost[wheat_wheat
_fallow]) {assumption of the FPM that total revenue will be equal to total costs}
quantity_of_chem[Rotation] = ha_cropped[Rotation]
quantity_of_Fert[Rotation] = 1*ha_cropped[Rotation]
quantity_of_other_inputs[Rotation] = 1*ha_cropped[Rotation]
Total_Cult_Ha = 0
WCPcanola = IF 3.1*P_sufficiency*CNsuf*CWsuf+Temp_correction>0 THEN
3.1*P_sufficiency*CNsuf*CWsuf+Temp_correction ELSE 0
WCPpea = 2.7*P_sufficiency*PNsuf*PWsuf
WCPwheat = 4*P_sufficiency*WNsuf*WWsuf
WCP_ha_rev =
(((Wheat_Price*WCPwheat)+(Canola_Price*WCPcanola)+(Pea_Price*WCPpea))/3)+W
CP_Cpay
WCP_rot = Total_Cult_Ha*Land_Prop[wheat_canola_peas] {Assume that 5% of
cultivated area is in WCP rotation}
WFPpeas = 2.7*PsufWFP*PNsufWFP*PWsufWFP
WFPwheat = 4*PsufWFP*WNsufWFP*WWsufWFP
WFP_ha_rev = (((WFPwheat*Wheat_Price)+(WFPpeas*Pea_Price))/3)+WFP_Cpay
WFP_rot = Total_Cult_Ha*Land_Prop[wheat_fallow_peas] {assume that 15% of
cultivated area in WFP rotation}
WFwheat = 4*PsufWF*WNsufWF*WWsufWF
WF_ha_rev = ((Wheat_Price*WFwheat)/2)+WF_Cpay
WF_rot = Total_Cult_Ha*Land_Prop[wheat_fallow]
WWFstubwheat = 4*PsufWWF*sWNsufWWF*WstubWsufWWF
WWFwheat = 4*PsufWWF*WNsufWWF*WWsufWWF
WWF_ha_rev = ((Wheat_Price*(WWFwheat+WWFstubwheat))/3)+WWF_Cpay

```

$WWF_rot = Total_Cult_Ha * Land_Prop[wheat_wheat_fallow]$ {assume that 30% of the baseline landscape is in the WWF rotation}



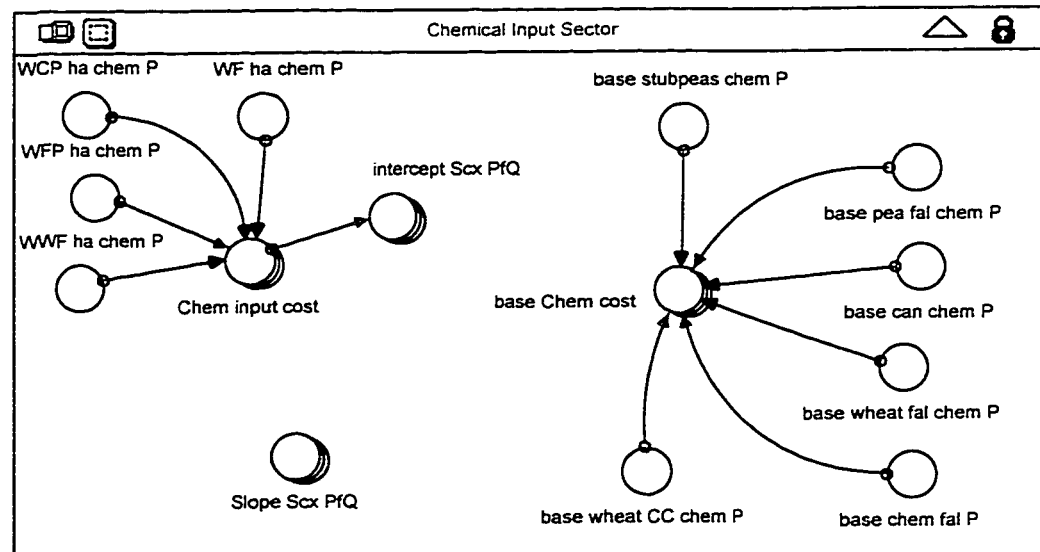
Other Input Market

```

Bo_other_input[wheat_fallow] = 0
Bo_other_input[wheat_fallow_peas] = 0
Bo_other_input[wheat_canola_peas] = 0
Bo_other_input[wheat_wheat_fallow] = 0
Elasticity_other_inputs[wheat_fallow] = 1.5
Elasticity_other_inputs[wheat_fallow_peas] = 5
Elasticity_other_inputs[wheat_canola_peas] = 17
Elasticity_other_inputs[wheat_wheat_fallow] = 3
Intercept_Sox_PfQ[Rotation] = (Po[Rotation]+Bo_other_input[Rotation])-
(quantity_of_other_inputs[Rotation]*Slope_Sox_PfQ[Rotation])
Intercept_Sox_QfP[Rotation] = quantity_of_other_inputs[Rotation]-
(Slope_Sox_QfP[Rotation]*Po[Rotation])
Po[wheat_fallow] = base_ha_revenue[wheat_fallow]-
(base_Fert_cost[wheat_fallow]+base_land_price+base_Chem_cost[wheat_fallow])
{assumption of the FPM that total revenue will be equal to total costs}
Po[wheat_fallow_peas] = MAX(120,base_ha_revenue[wheat_fallow_peas]-
(base_Fert_cost[wheat_fallow_peas]+base_land_price+base_Chem_cost[wheat_fallow_
peas])) {assumption of the FPM that total revenue will be equal to total costs}
Po[wheat_canola_peas] = MAX(135,base_ha_revenue[wheat_canola_peas]-
(base_Fert_cost[wheat_canola_peas]+base_land_price+base_Chem_cost[wheat_canola_
peas])) {assumption of the FPM that total revenue will be equal to total costs}
Po[wheat_wheat_fallow] = base_ha_revenue[wheat_wheat_fallow]-
(base_Fert_cost[wheat_wheat_fallow]+base_land_price+base_Chem_cost[wheat_wheat_
_fallow]) {assumption of the FPM that total revenue will be equal to total costs}
quantity_of_other_inputs[Rotation] = 1*ha_cropped[Rotation]

```

$\text{Slope_Sox_PfQ[Rotation]} = 1/\text{Slope_Sox_QfP[Rotation]}$
 $\text{Slope_Sox_QfP[wheat_fallow]} =$
 $(\text{Elasticity_other_inputs[wheat_fallow]} * \text{quantity_of_other_inputs[wheat_fallow]}) / \text{Po[wheat_fallow]}$
 $\text{Slope_Sox_QfP[wheat_fallow_peas]} =$
 $(\text{Elasticity_other_inputs[wheat_fallow_peas]} * (\text{quantity_of_other_inputs[wheat_fallow_peas]}) / \text{Po[wheat_fallow_peas]})$
 $\text{Slope_Sox_QfP[wheat_canola_peas]} =$
 $(\text{Elasticity_other_inputs[wheat_canola_peas]} * \text{quantity_of_other_inputs[wheat_canola_peas]}) / \text{Po[wheat_canola_peas]}$
 $\text{Slope_Sox_QfP[wheat_wheat_fallow]} =$
 $\text{Elasticity_other_inputs[wheat_wheat_fallow]} * \text{quantity_of_other_inputs[wheat_wheat_fallow]} / \text{Po[wheat_wheat_fallow]}$



Chemical Input Sector

base_can_chem_P = 52.46 {\$/ha, pesticide costs for direct seeded canola, from saskatchewan crop production data}

base_Chem_cost[wheat_fallow] =

$(\text{base_wheat_fal_chem_P} + \text{base_chem_fal_P}) / 2 + (0 * (\text{base_can_chem_P} + \text{base_pea_fal_chem_P} + \text{base_stubpeas_chem_P} + \text{base_wheat_CC_chem_P}))$

base_Chem_cost[wheat_fallow_peas] =

$(\text{base_wheat_CC_chem_P} + \text{base_chem_fal_P} + \text{base_pea_fal_chem_P}) / 3 + (0 * (\text{base_can_chem_P} + \text{base_stubpeas_chem_P} + \text{base_wheat_fal_chem_P}))$

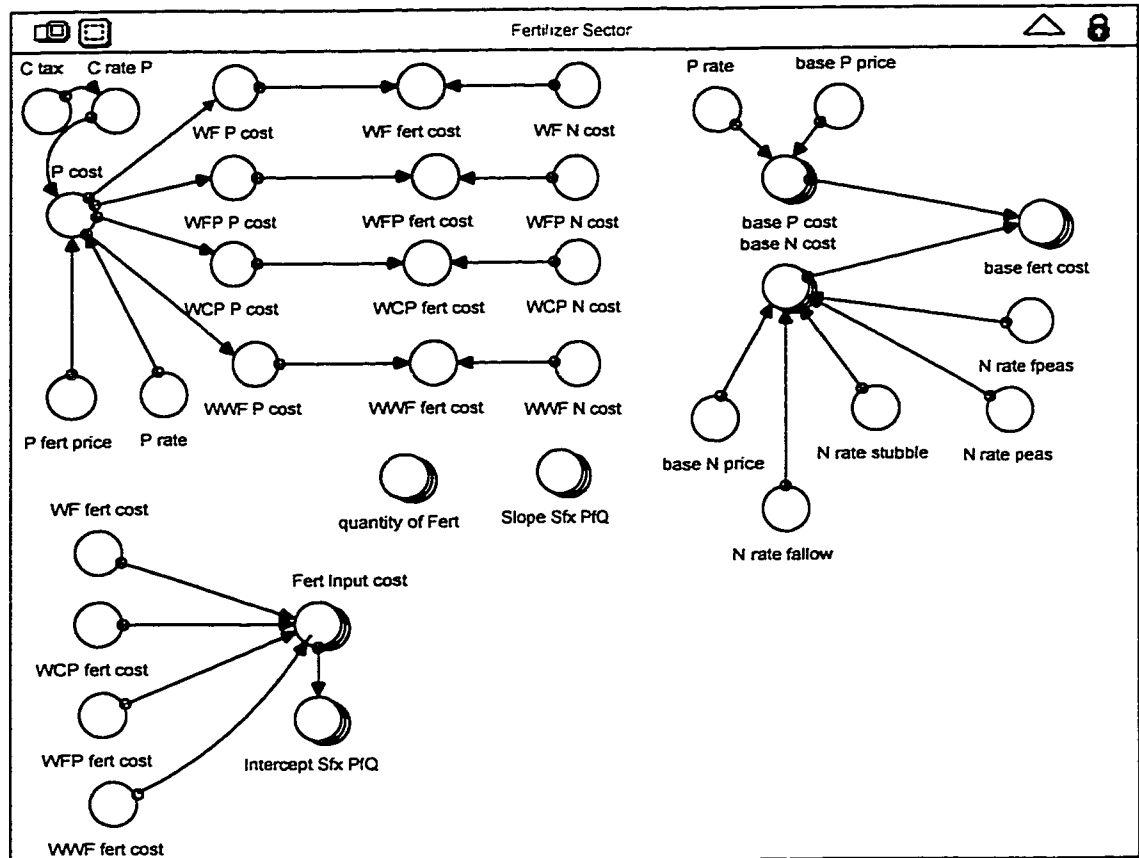
base_Chem_cost[wheat_canola_peas] =

$(\text{base_wheat_CC_chem_P} + \text{base_can_chem_P} + \text{base_stubpeas_chem_P}) / 3 + (0 * (\text{base_chem_fal_P} + \text{base_pea_fal_chem_P} + \text{base_wheat_fal_chem_P}))$

```

base_Chem_cost[wheat_wheat_fallow] =
((base_wheat_fal_chem_P+base_wheat_CC_chem_P+base_chem_fal_P)/3+(0*(base_can
_chem_P+base_pea_fal_chem_P+base_stubpeas_chem_P))
base_chem_fal_P = 36.55 {$/ha, pesticide cost for fallow, from Saskatchewan crop
production data}
base_pea_fal_chem_P = 101.06 {$/ha, pesticide cost for fallow seeded canola, from
Saskatchewan crop production data}
base_stubpeas_chem_P = 101.06 {$/ha cost of pesticides for stubble seeded peas from
crop production stats,}
base_wheat_CC_chem_P = 43.51 {$/per ha, direct seeded wheat onto stubble, from
saskatchewan crop production data}
base_wheat_fal_chem_P = 32.74 {$/ha, pesticide costs for fallow seeded spring wheat,
from Saskatchewan crop production data}
Chem_input_cost[wheat_fallow] =
WF_ha_chem_P+(0*(WCP_ha_chem_P+WFP_ha_chem_P+WWF_ha_chem_P))
Chem_input_cost[wheat_fallow_peas] =
WFP_ha_chem_P+(0*(WCP_ha_chem_P+WF_ha_chem_P+WWF_ha_chem_P))
Chem_input_cost[wheat_canola_peas] =
WCP_ha_chem_P+(0*(WFP_ha_chem_P+WF_ha_chem_P+WWF_ha_chem_P))
Chem_input_cost[wheat_wheat_fallow] =
WWF_ha_chem_P+(0*(WCP_ha_chem_P+WFP_ha_chem_P+WF_ha_chem_P))
intercept_Scx_PfQ[Rotation] = Chem_input_cost[Rotation]
Slope_Scx_PfQ[Rotation] = 0
WCP_ha_chem_P =
((Can_chem_P+SPea_chem_P+Wheat_CC_chem_P)/3)+WCP_ch_C
WFP_ha_chem_P =
((pea_fal_chem_P+Chem_Fal_P+Wheat_CC_chem_P)/3)+WFP_ch_C
WF_ha_chem_P = ((Chem_Fal_P+Wheat_fal_chem_p)/2)+WF_ch_C
WWF_ha_chem_P =
((Chem_Fal_P+Wheat_fal_chem_p+Wheat_CC_chem_P)/3)+WWF_ch_C

```



Fertilizer Sector

base_fert_cost[wheat_fallow] =

base_N_cost[wheat_fallow]+base_P_cost[wheat_fallow]

base_fert_cost[wheat_fallow_peas] =

base_N_cost[wheat_fallow_peas]+base_P_cost[wheat_fallow_peas]

base_fert_cost[wheat_canola_peas] =

base_N_cost[wheat_canola_peas]+base_P_cost[wheat_canola_peas]

base_fert_cost[wheat_wheat_fallow] =

base_N_cost[wheat_wheat_fallow]+base_P_cost[wheat_wheat_fallow]

base_N_cost[wheat_fallow] = (base_N_price*N_rate_fallow)/2

+(0*(N_rate_peas+N_rate_stubble+N_rate_fpeas)) {\$/ha for average hectare in WF rotation}

base_N_cost[wheat_fallow_peas] =

((base_N_price*N_rate_stubble)+(base_N_price*N_rate_fpeas))/3+(0*N_rate_stubble+N_rate_peas+N_rate_fallow) {\$/ha cost of N on average WFC ha}

base_N_cost[wheat_canola_peas] =

((base_N_price*N_rate_stubble)+(base_N_price*N_rate_stubble)+(base_N_price*N_rate_peas))/3+(0*N_rate_fallow+N_rate_fpeas)

base_N_cost[wheat_wheat_fallow] =

((base_N_price*N_rate_fallow)+(base_N_price*N_rate_stubble))/3+(0*(N_rate_peas+N_rate_fpeas))

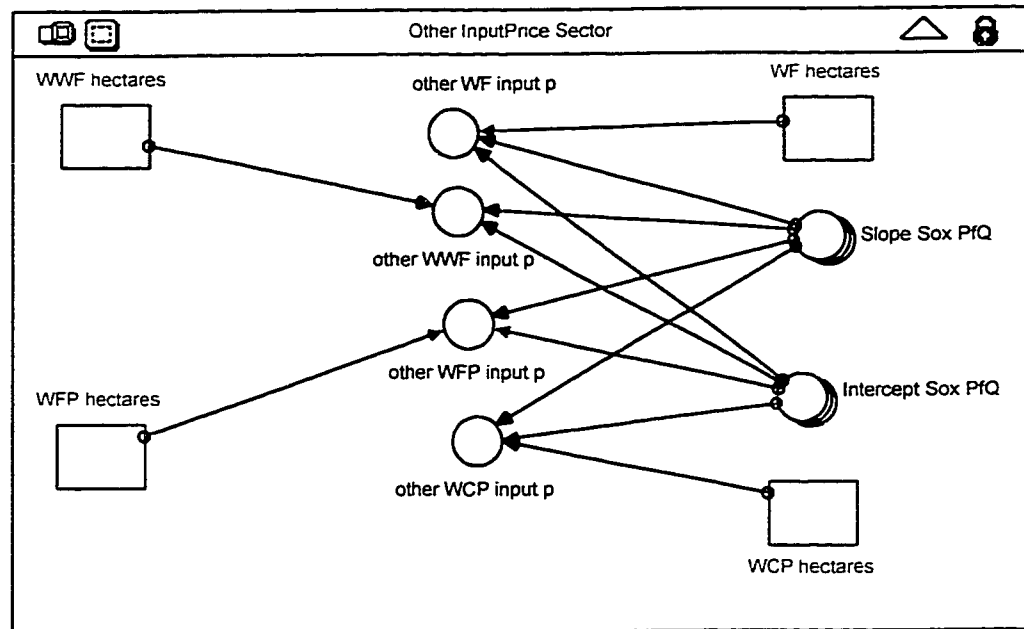
$\text{base_N_price} = 537$ {\$/tonne of actual N based on \$315/t for 46-0-0, \$0.68N/kg}
 $\text{base_P_cost}[\text{wheat_fallow}] = (\text{base_P_price} * \text{P_rate}) / 2$ {cost of P for WF rotation based on baseline statistics}
 $\text{base_P_cost}[\text{wheat_fallow_peas}] = ((\text{base_P_price} * \text{P_rate}) * 2) / 3$
 $\text{base_P_cost}[\text{wheat_canola_peas}] = ((\text{base_P_price} * \text{P_rate}) * 3) / 3$
 $\text{base_P_cost}[\text{wheat_wheat_fallow}] = ((\text{base_P_price} * \text{P_rate}) * 2) / 3$
 $\text{base_P_price} = 735$ {\$/tonne of actual P based on \$400/tonne for 12-51-0, or \$.617/kg}
 $\text{C_rate_P} = .255 * \text{C_tax}$ {calculates the cost per tonne of p fertilizer based on a carbon tax}
 $\text{C_tax} = 0$ {\$/tonne}
 $\text{Fert_Input_cost}[\text{wheat_fallow}] =$
 $\text{WF_fert_cost} + (0 * (\text{WFP_fert_cost} + \text{WCP_fert_cost} + \text{WWF_fert_cost}))$
 $\text{Fert_Input_cost}[\text{wheat_fallow_peas}] =$
 $\text{WFP_fert_cost} + (0 * (\text{WCP_fert_cost} + \text{WF_fert_cost} + \text{WWF_fert_cost}))$
 $\text{Fert_Input_cost}[\text{wheat_canola_peas}] =$
 $\text{WCP_fert_cost} + (0 * (\text{WF_fert_cost} + \text{WFP_fert_cost} + \text{WWF_fert_cost}))$
 $\text{Fert_Input_cost}[\text{wheat_wheat_fallow}] =$
 $\text{WWF_fert_cost} + (0 * (\text{WCP_fert_cost} + \text{WFP_fert_cost} + \text{WF_fert_cost}))$
 $\text{Intercept_Sfx_PfQ}[\text{Rotation}] = \text{Fert_Input_cost}[\text{Rotation}]$
 $\text{N_rate_fallow} = .0338$ {t/ha, actual N rate on fallow seeded crops, based on Sask Ag. and Food Farm Facts information}
 $\text{N_rate_fpeas} = .01$ {t/ha}
 $\text{N_rate_peas} = .0225$ {t/ha actual N, based on Sask ag. and food values}
 $\text{N_rate_stubble} = .0676$ {t/ha actual N, Sask. AG. and Food}
 $\text{P_cost} = (\text{P_fert_price} * \text{P_rate}) + (\text{P_rate} * \text{C_rate_P})$ {\$/ha, all crops within a soil zone have same rates of P application, Sask. Ag and Food}
 $\text{P_fert_price} = \text{base_P_price} + \text{Bo_P_price}$
 $\text{P_rate} = .0336$ {t/ha, application rate of actual P for all crops}
 $\text{quantity_of_Fert}[\text{Rotation}] = 1 * \text{ha_cropped}[\text{Rotation}]$
 $\text{Slope_Sfx_PfQ}[\text{Rotation}] = 0$
 $\text{WCP_fert_cost} = \text{WCP_N_cost} + \text{WCP_P_cost}$
 $\text{WCP_N_cost} =$
 $(((\text{WCPcanN} + \text{WCPPeaN} + \text{WCPwhtN}) * \text{N_fert_price}) / 1000 / 3) + (((\text{WCPcanN} + \text{WCPPeaN} + \text{WCPwhtN}) * \text{C_rate_N}) / 3)$
 $\text{WCP_P_cost} = (\text{P_cost} + \text{P_cost} + \text{P_cost}) / 3$ {\$/ha, average cost of P input for WCP rotation}
 $\text{WFP_fert_cost} = \text{WFP_N_cost} + \text{WFP_P_cost}$
 $\text{WFP_N_cost} =$
 $(((\text{WFPWhtN} * \text{N_fert_price}) + (\text{WFPPeaN} * \text{N_fert_price})) / 1000 / 3) + ((\text{WFPPeaN} + \text{WFPWhtN}) * \text{C_rate_N}) / 3$
 $\text{WFP_P_cost} = (\text{P_cost} + \text{P_cost}) / 3$ {\$/ha, average cost of P input}
 $\text{WF_fert_cost} = \text{WF_N_cost} + \text{WF_P_cost}$
 $\text{WF_N_cost} = ((\text{WfWhtN} * \text{N_fert_price}) / 2 / 1000) + ((\text{WfWhtN} * \text{C_rate_N}) / 2)$
 $\text{WF_P_cost} = \text{P_cost} / 2$ {\$/ha, average cost per ha of P input}
 $\text{WWF_fert_cost} = \text{WWF_N_cost} + \text{WWF_P_cost}$


```

WCP_hectares(t) = WCP_hectares(t - dt)
INIT WCP_hectares = WCP_rot
WFP_hectares(t) = WFP_hectares(t - dt)
INIT WFP_hectares = WFP_rot
WF_hectares(t) = WF_hectares(t - dt)
INIT WF_hectares = WF_rot
WWF_hectares(t) = WWF_hectares(t - dt)
INIT WWF_hectares = WWF_rot
Active_Ha =
TransitionLand+WCP_hectares+WFP_hectares+WF_hectares+WWF_hectares {area of
ecodistrict that is dedicated to annual crop production}
Equil_Land_P = (Intercept_Dl_QfP-Intercept_Sl_QfP)/(Slope_Sl_QfP-
Slope_Dl_QfP){calculates the market clearing land price at the current state}
Equil_Land_Q[Rotation] = IF
(Intercept_Slx_QfP[Rotation]+(Slope_Slx_QfP[Rotation]*Equil_Land_P))>0 THEN
Intercept_Slx_QfP[Rotation]+(Slope_Slx_QfP[Rotation]*Equil_Land_P) ELSE 0
ha_revenue[wheat_fallow] =
WF_ha_rev+(WFP_ha_rev+WCP_ha_rev+WWF_ha_rev)*0
ha_revenue[wheat_fallow_peas] =
WFP_ha_rev+(WF_ha_rev+WCP_ha_rev+WWF_ha_rev)*0
ha_revenue[wheat_canola_peas] =
WCP_ha_rev+(WF_ha_rev+WFP_ha_rev+WWF_ha_rev)*0
ha_revenue[wheat_wheat_fallow] =
WWF_ha_rev+(0*(WCP_ha_rev+WFP_ha_rev+WF_ha_rev))
Intercept_Dlx_PfQ[Rotation] = ha_revenue[Rotation]-
(Intercept_Sfx_PfQ[Rotation]+Intercept_Sox_PfQ[Rotation]+intercept_Scx_PfQ[Rotati
on]) {FPM constraint of zero profits}
Intercept_Dlx_QfP[Rotation] = (0-
Intercept_Dlx_PfQ[Rotation])/Slope_Dlx_PfQ[Rotation] {this calculates the quantity
intecept for the inverse demand curve for each individual land demand curve, Qf(P)}
Intercept_Dl_PfQ = (0-Intercept_Dl_QfP)/Slope_Dl_QfP
Intercept_Dl_QfP = ARRAYSUM(Intercept_Dlx_QfP[*])
intercept_Scx_PfQ[Rotation] = Chem_input_cost[Rotation]
Intercept_Sfx_PfQ[Rotation] = Fert_Input_cost[Rotation]
Intercept_Slx_QfP[wheat_fallow] = Intercept_Sl_QfP-
(Intercept_Dlx_QfP[wheat_fallow_peas]+Intercept_Dlx_QfP[wheat_canola_peas]+Inter
cept_Dlx_QfP[wheat_wheat_fallow])
Intercept_Slx_QfP[wheat_fallow_peas] = Intercept_Sl_QfP-
(Intercept_Dlx_QfP[wheat_fallow]+Intercept_Dlx_QfP[wheat_canola_peas]+Intercept_
Dlx_QfP[wheat_wheat_fallow])
Intercept_Slx_QfP[wheat_canola_peas] = Intercept_Sl_QfP-
(Intercept_Dlx_QfP[wheat_fallow]+Intercept_Dlx_QfP[wheat_fallow_peas]+Intercept_
Dlx_QfP[wheat_wheat_fallow])

```

$\text{Intercept_Slx_QfP[wheat_wheat_fallow]} = \text{Intercept_Sl_QfP} -$
 $(\text{Intercept_Dlx_QfP[wheat_fallow]} + \text{Intercept_Dlx_QfP[wheat_fallow_peas]} + \text{Intercept_Dlx_QfP[wheat_canola_peas]})$
 $\text{Intercept_Sl_QfP} = \text{Active_Ha}$
 $\text{Intercept_Sox_PfQ[Rotation]} = (\text{Po[Rotation]} + \text{Bo_other_input[Rotation]}) -$
 $(\text{quantity_of_other_inputs[Rotation]} * \text{Slope_Sox_PfQ[Rotation]})$
 $\text{Slope_Dlx_PfQ[Rotation]} = 0 -$
 $(\text{Slope_Sfx_PfQ[Rotation]} + \text{Slope_Scx_PfQ[Rotation]} + \text{Slope_Sox_PfQ[Rotation]})$
 $\text{Slope_Dlx_QfP[Rotation]} = 1 / \text{Slope_Dlx_PfQ[Rotation]} \text{ \{Slope of individual demand curve for land where Qf(P)\}}$
 $\text{Slope_DI_PfQ} = 1 / \text{Slope_DI_QfP}$
 $\text{Slope_DI_QfP} = \text{ARRAYSUM}(\text{Slope_Dlx_QfP[*]})$
 $\text{Slope_Scx_PfQ[Rotation]} = 0$
 $\text{Slope_Sfx_PfQ[Rotation]} = 0$
 $\text{Slope_Slx_QfP[wheat_fallow]} = \text{Slope_Sl_QfP} -$
 $(\text{Slope_Dlx_QfP[wheat_fallow_peas]} + \text{Slope_Dlx_QfP[wheat_canola_peas]} + \text{Slope_Dlx_QfP[wheat_wheat_fallow]})$
 $\text{Slope_Slx_QfP[wheat_fallow_peas]} = \text{Slope_Sl_QfP} -$
 $(\text{Slope_Dlx_QfP[wheat_fallow]} + \text{Slope_Dlx_QfP[wheat_canola_peas]} + \text{Slope_Dlx_QfP[wheat_wheat_fallow]})$
 $\text{Slope_Slx_QfP[wheat_canola_peas]} = \text{Slope_Sl_QfP} -$
 $(\text{Slope_Dlx_QfP[wheat_fallow]} + \text{Slope_Dlx_QfP[wheat_fallow_peas]} + \text{Slope_Dlx_QfP[wheat_wheat_fallow]})$
 $\text{Slope_Slx_QfP[wheat_wheat_fallow]} = \text{Slope_Sl_QfP} -$
 $(\text{Slope_Dlx_QfP[wheat_fallow]} + \text{Slope_Dlx_QfP[wheat_fallow_peas]} + \text{Slope_Dlx_QfP[wheat_canola_peas]})$
 $\text{Slope_Sl_QfP} = 0$
 $\text{Slope_Sox_PfQ[Rotation]} = 1 / \text{Slope_Sox_QfP[Rotation]}$



Other Input Price Sector

WCP_hectares(t) = WCP_hectares(t - dt)

INIT WCP_hectares = WCP_rot

WFP_hectares(t) = WFP_hectares(t - dt)

INIT WFP_hectares = WFP_rot

WF_hectares(t) = WF_hectares(t - dt)

INIT WF_hectares = WF_rot

WWF_hectares(t) = WWF_hectares(t - dt)

INIT WWF_hectares = WWF_rot

Intercept_Sox_PfQ[Rotation] = (Po[Rotation]+Bo_other_input[Rotation])-(quantity_of_other_inputs[Rotation]*Slope_Sox_PfQ[Rotation])

other_WCP_input_p =

Intercept_Sox_PfQ[wheat_canola_peas]+(Slope_Sox_PfQ[wheat_canola_peas]*WCP_hectares)

other_WFP_input_p =

Intercept_Sox_PfQ[wheat_fallow_peas]+(Slope_Sox_PfQ[wheat_fallow_peas]*WFP_hectares)

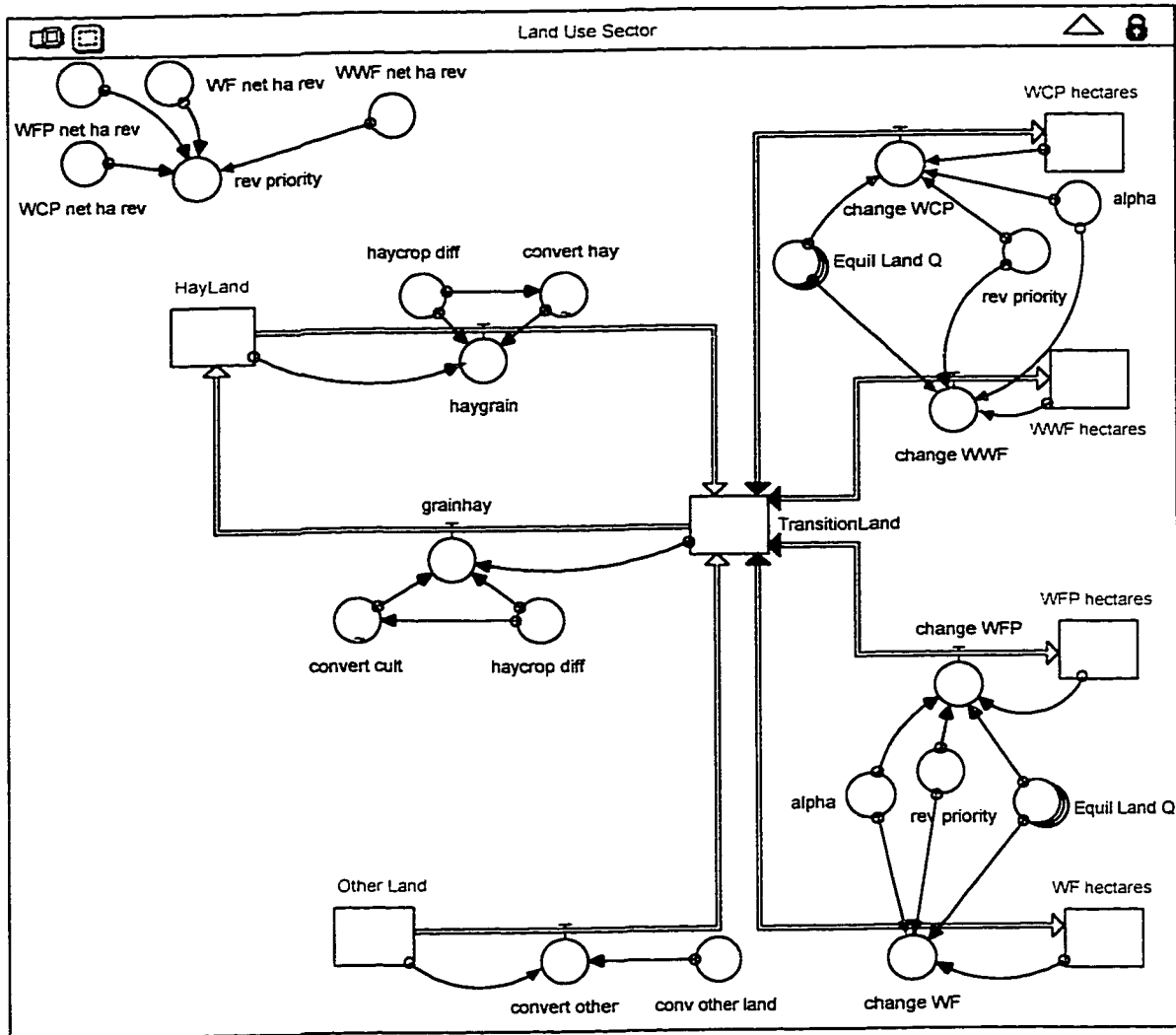
other_WF_input_p =

Intercept_Sox_PfQ[wheat_fallow]+(Slope_Sox_PfQ[wheat_fallow]*WF_hectares)

other_WWF_input_p =

Intercept_Sox_PfQ[wheat_wheat_fallow]+(Slope_Sox_PfQ[wheat_wheat_fallow]*WWF_hectares)

Slope_Sox_PfQ[Rotation] = 1/Slope_Sox_QfP[Rotation]



Land Use Sector

HayLand(t) = HayLand(t - dt) + (grainhay - haygrain) * dt

INIT HayLand = 1000

INFLOWS:

grainhay = IF haycrop_diff < -30 AND haycrop_diff > -40 THEN
(convert_cult * TransitionLand) ELSE IF haycrop_diff < -30 THEN (0.5 * TransitionLand)
ELSE 0

OUTFLOWS:

haygrain = IF (haycrop_diff > 20) AND (haycrop_diff < 30) THEN
(HayLand * convert_hay) ELSE IF (haycrop_diff > 30) THEN (0.5 * HayLand) ELSE 0

Other_Land(t) = Other_Land(t - dt) + (- convert_other) * dt

INIT Other_Land = 1

OUTFLOWS:

convert_other = MIN(conv_other_land, (.05 * (INIT(Other_Land))))

TransitionLand(t) = TransitionLand(t - dt) + (haygrain + convert_other - change_WCP -
change_WF - grainhay - change_WFP - change_WWF) * dt

INIT TransitionLand = 1

INFLOWS:

haygrain = IF (haycrop_diff>20) AND (haycrop_diff<30) THEN
(HayLand*convert_hay) ELSE IF (haycrop_diff>30) THEN (0.5*HayLand) ELSE 0
convert_other = MIN(conv_other_land,(.05*(INIT(Other_Land))))

OUTFLOWS:

change_WCP = IF (Equil_Land_Q[wheat_canola_peas]-WCP_hectares)<0 THEN
(alpha*(Equil_Land_Q[wheat_canola_peas]-WCP_hectares)) ELSE IF
(Equil_Land_Q[wheat_canola_peas]-WCP_hectares)>0 AND (rev_priority=1) THEN
(alpha*(Equil_Land_Q[wheat_canola_peas]-WCP_hectares)) ELSE 0
change_WF = IF (Equil_Land_Q[wheat_fallow]-WF_hectares)<0 THEN
(alpha*(Equil_Land_Q[wheat_fallow]-WF_hectares)) ELSE IF
(Equil_Land_Q[wheat_fallow]-WF_hectares)>0 AND (rev_priority=3) THEN
(alpha*(Equil_Land_Q[wheat_fallow]-WF_hectares)) ELSE 0
grainhay = IF haycrop_diff<-30 AND haycrop_diff>-40 THEN
(convert_cult*TransitionLand) ELSE IF haycrop_diff<-30 THEN (0.5*TransitionLand)
ELSE 0

change_WFP = IF (Equil_Land_Q[wheat_fallow_peas]-WFP_hectares)<0 THEN
alpha*(Equil_Land_Q[wheat_fallow_peas]-WFP_hectares) ELSE IF
(Equil_Land_Q[wheat_fallow_peas]-WFP_hectares)>0 AND rev_priority=2 THEN
alpha*(Equil_Land_Q[wheat_fallow_peas]-WFP_hectares) ELSE 0
change_WWF = IF (Equil_Land_Q[wheat_wheat_fallow]-WWF_hectares)<0 THEN
alpha*(Equil_Land_Q[wheat_wheat_fallow]-WWF_hectares) ELSE IF
(Equil_Land_Q[wheat_wheat_fallow]-WWF_hectares)>0 AND (rev_priority=4) THEN
alpha*(Equil_Land_Q[wheat_wheat_fallow]-WWF_hectares) ELSE 0
WCP_hectares(t) = WCP_hectares(t - dt) + (change_WCP) * dt
INIT WCP_hectares = WCP_rot

INFLOWS:

change_WCP = IF (Equil_Land_Q[wheat_canola_peas]-WCP_hectares)<0 THEN
(alpha*(Equil_Land_Q[wheat_canola_peas]-WCP_hectares)) ELSE IF
(Equil_Land_Q[wheat_canola_peas]-WCP_hectares)>0 AND (rev_priority=1) THEN
(alpha*(Equil_Land_Q[wheat_canola_peas]-WCP_hectares)) ELSE 0
WFP_hectares(t) = WFP_hectares(t - dt) + (change_WFP) * dt
INIT WFP_hectares = WFP_rot

INFLOWS:

change_WFP = IF (Equil_Land_Q[wheat_fallow_peas]-WFP_hectares)<0 THEN
alpha*(Equil_Land_Q[wheat_fallow_peas]-WFP_hectares) ELSE IF
(Equil_Land_Q[wheat_fallow_peas]-WFP_hectares)>0 AND rev_priority=2 THEN
alpha*(Equil_Land_Q[wheat_fallow_peas]-WFP_hectares) ELSE 0
WF_hectares(t) = WF_hectares(t - dt) + (change_WF) * dt
INIT WF_hectares = WF_rot

INFLOWS:

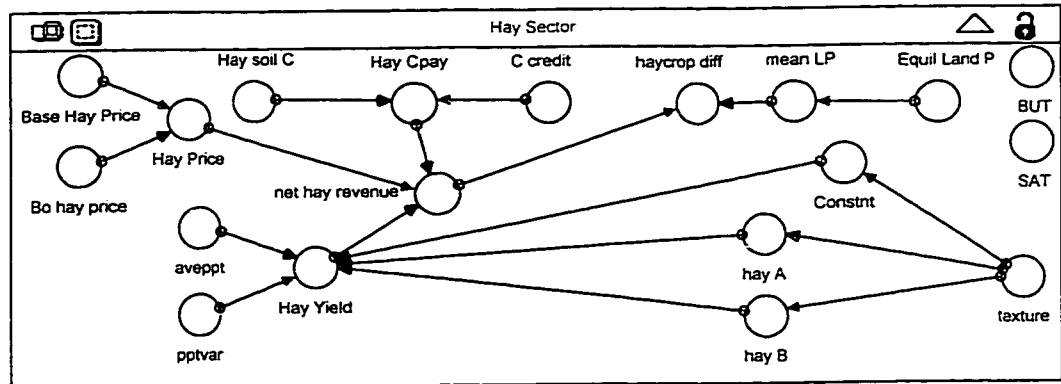
change_WF = IF (Equil_Land_Q[wheat_fallow]-WF_hectares)<0 THEN
(alpha*(Equil_Land_Q[wheat_fallow]-WF_hectares)) ELSE IF

```

(Equil_Land_Q[wheat_fallow]-WF_hectares)>0 AND (rev_priority=3) THEN
(alpha*(Equil_Land_Q[wheat_fallow]-WF_hectares)) ELSE 0
WWF_hectares(t) = WWF_hectares(t - dt) + (change_WWF) * dt
INIT WWF_hectares = WWF_rot
INFLOWS:
change_WWF = IF (Equil_Land_Q[wheat_wheat_fallow]-WWF_hectares)<0 THEN
alpha*(Equil_Land_Q[wheat_wheat_fallow]-WWF_hectares) ELSE IF
(Equil_Land_Q[wheat_wheat_fallow]-WWF_hectares)>0 AND(rev_priority=4) THEN
alpha*(Equil_Land_Q[wheat_wheat_fallow]-WWF_hectares) ELSE 0
alpha = 0.5
conv_other_land = IF (Change_P>0) THEN ((gamma/(Change_P^delta))*Other_Land)
ELSE 0
Equil_Land_Q[Rotation] = IF
(Intercept_Slx_QfP[Rotation]+(Slope_Slx_QfP[Rotation]*Equil_Land_P))>0 THEN
Intercept_Slx_QfP[Rotation]+(Slope_Slx_QfP[Rotation]*Equil_Land_P) ELSE 0
haycrop_diff = mean_LP-net_hay_revenue
rev_priority = IF(WCP_net_ha_rev>WFP_net_ha_rev)
AND(WCP_net_ha_rev>WF_net_ha_rev)AND(WCP_net_ha_rev>WWF_net_ha_rev)
THEN 1 ELSE IF(WFP_net_ha_rev>WCP_net_ha_rev)
AND(WFP_net_ha_rev>WF_net_ha_rev)AND(WFP_net_ha_rev>WWF_net_ha_rev)
THEN 2 ELSE IF(WF_net_ha_rev>WCP_net_ha_rev)
AND(WF_net_ha_rev>WFP_net_ha_rev)AND(WF_net_ha_rev>WWF_net_ha_rev)
THEN 3 ELSE IF
(WWF_net_ha_rev>WCP_net_ha_rev)AND(WWF_net_ha_rev>WFP_net_ha_rev)AND
(WWF_net_ha_rev>WF_net_ha_rev) THEN 4 ELSE(0)
WCP_net_ha_rev = WCP_ha_rev-
(other_WCP_input_p+WCP_ha_chem_P+(MAX(Equil_Land_P,0))+WCP_N_cost+WC
P_P_cost)
WFP_net_ha_rev = WFP_ha_rev-
(other_WFP_input_p+WFP_ha_chem_P+(MAX(Equil_Land_P,0))+WFP_N_cost+WFP
_P_cost)
WF_net_ha_rev = WF_ha_rev-
(other_WF_input_p+WF_ha_chem_P+(MAX(Equil_Land_P,0))+WF_N_cost+WF_P_c
ost)
WWF_net_ha_rev = WWF_ha_rev-
(other_WWF_input_p+WWF_ha_chem_P+(MAX(Equil_Land_P,0))+WWF_N_cost+W
WF_P_cost)
convert_cult = GRAPH(haycrop_diff)
(-40.0, 0.498), (-39.5, 0.468), (-38.9, 0.428), (-38.4, 0.403), (-37.9, 0.38), (-37.4, 0.363),
(-36.8, 0.35), (-36.3, 0.325), (-35.8, 0.29), (-35.3, 0.25), (-34.7, 0.235), (-34.2, 0.208), (-
33.7, 0.178), (-33.2, 0.153), (-32.6, 0.125), (-32.1, 0.1), (-31.6, 0.0775), (-31.1, 0.0625),
(-30.5, 0.0425), (-30.0, 0.00)
convert_hay = GRAPH(haycrop_diff)
(20.0, 0.0075), (20.5, 0.148), (21.1, 0.208), (21.6, 0.243), (22.1, 0.273), (22.6, 0.31),
(23.2, 0.335), (23.7, 0.355), (24.2, 0.373), (24.7, 0.39), (25.3, 0.408), (25.8, 0.425),

```

(26.3, 0.443), (26.8, 0.455), (27.4, 0.47), (27.9, 0.478), (28.4, 0.488), (28.9, 0.495), (29.5, 0.498), (30.0, 0.5)



Hay Sector

```
aveppt = IF TIME=0 THEN GS_ppt ELSE GS_gamma
```

Base Hay Price = 63 {\$/tonne}

Bo hay price = 0

$\text{BUT} = 30$ {Break-up threshold}

Constnt = If texture=1 THEN 1.157 ELSE IF texture=3 THEN 1.307 ELSE IF texture=4 THEN 0.484 ELSE .5 {values derived from regressions, .5 value proxy for soils not regressed}

C credit = 0 {\$/tonne of C sequestered paid as a credit to the landowner}

$$\text{Equil_Land_P} = (\text{Intercept_DI_QfP} - \text{Intercept_SI_QfP}) / (\text{Slope_SI_QfP} - \text{Slope_DI_QfP}) \{ \text{calculates the market clearing land price at the current state} \}$$

```
haycrop_diff = mean_LP-net_hay_revenue
```

```
hay_A = If texture=1 THEN 0.041 ELSE IF texture=3 THEN 0.032 ELSE IF texture=4
THEN 0.078 ELSE .05 {values derived from regressions, .05 value proxy for soils not
regressed}
```

```
hay_B = If texture=1 THEN 0.047 ELSE IF texture=3 THEN 0.062 ELSE IF texture=4
THEN 0.036 ELSE .05 {values derived from regressions, .05 value proxy for soils not
regressed}
```

$$\text{Hay_Cpay} = \text{C_credit} * \text{Hay_soil_C} \text{ \{ \$/ha/yr, C credit payed to landowner for sequestered carbon \}}$$
$$\text{Hay Price} = \text{Base Hay Price} + \text{Bo hay price}$$

Hay soil C = 0.7 {t/ha/yr, C sequestered by soil under grass cover}

$$\text{Hay Yield} = \text{Constnt} + (\text{hay_A} * \text{aveppt}) + (\text{hay_B} * \text{pptvar}) \{t/ha\}$$

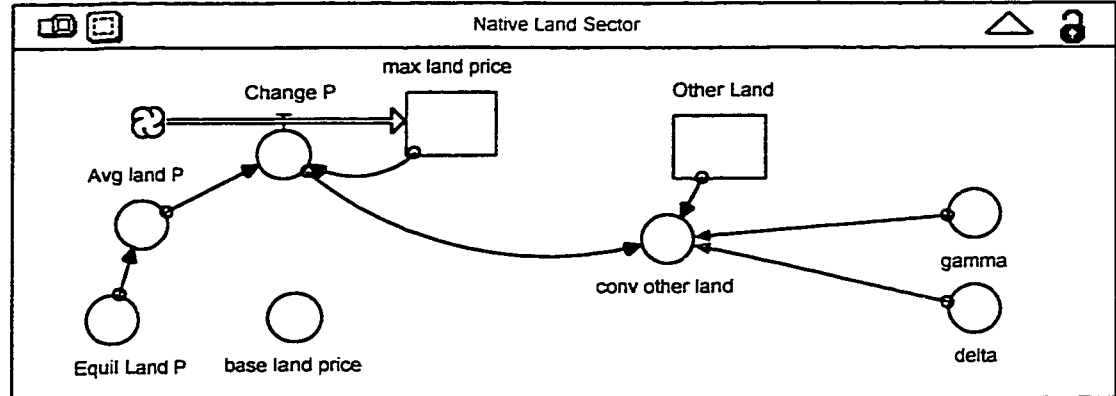
```
mean_LP = MEAN(Equil_Land_P, DELAY(Equil_Land_P, 1, INIT(Equil_Land_P)))
```

$$\text{net hay revenue} = ((\text{Hay Price} * \text{Hay Yield}) / 3) + \text{Hay_Cpay}$$

```
pptvar = IF TIME=0 THEN SWE_ppt_cm ELSE SWE_gamma
```

SAT = -40 {Set-aside threshold}

```
texture = 0
```

Native Land Sector

$\text{max_land_price}(t) = \text{max_land_price}(t - dt) + (\text{Change_P}) * dt$

INIT $\text{max_land_price} = \text{Equil_Land_P}$ {otherwise set at 49.42}

INFLOWS:

$\text{Change_P} = \text{IF } (\text{Avg_land_P} - \text{max_land_price}) > 0 \text{ THEN } (\text{Avg_land_P} - \text{max_land_price})$
ELSE 0

$\text{Other_Land}(t) = \text{Other_Land}(t - dt)$

INIT $\text{Other_Land} = 1$

$\text{Avg_land_P} =$

$\text{MEAN}(\text{Equil_Land_P}, \text{DELAY}(\text{Equil_Land_P}, 1, \text{INIT}(\text{Equil_Land_P})), \text{DELAY}(\text{Equil_Land_P}, 2, \text{INIT}(\text{Equil_Land_P})))$

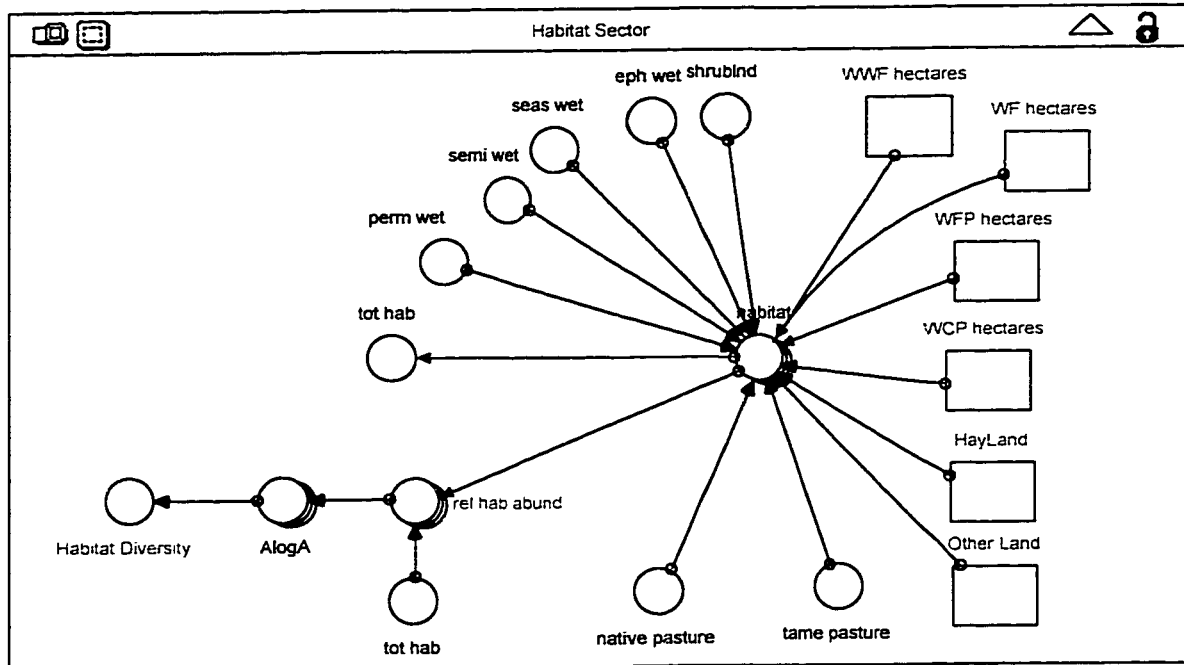
$\text{base_land_price} = 49.42$ {per hectare, investment price per ha of land sask ag and food crop planning guide 1997}

$\text{conv_other_land} = \text{IF } (\text{Change_P} > 0) \text{ THEN } ((\text{gamma} / (\text{Change_P}^{\text{delta}})) * \text{Other_Land})$
ELSE 0

$\text{delta} = -1.010871$ {exponent for denominator in other land quality calculation, determined empirically}

$\text{Equil_Land_P} = (\text{Intercept_DI_QfP} - \text{Intercept_SI_QfP}) / (\text{Slope_SI_QfP} - \text{Slope_DI_QfP})$ {calculates the market clearing land price at the current state}

$\text{gamma} = 0.035634$ {denominator for other land quality calculation, determined empirically}



Habitat Sector

HayLand(t) = HayLand(t - dt)

INIT HayLand = 1000

Other_Land(t) = Other_Land(t - dt)

INIT Other_Land = 1

WCP_hectares(t) = WCP_hectares(t - dt)

INIT WCP_hectares = WCP_rot

WFP_hectares(t) = WFP_hectares(t - dt)

INIT WFP_hectares = WFP_rot

WF_hectares(t) = WF_hectares(t - dt)

INIT WF_hectares = WF_rot

WWF_hectares(t) = WWF_hectares(t - dt)

INIT WWF_hectares = WWF_rot

AlogA[Habitat_type] =

rel_hab_abund[Habitat_type]*(LOG10(rel_hab_abund[Habitat_type]))

eph_wet = 1000

habitat[fallow] =

(WF_hectares*.5)+(WFP_hectares*.333)+(WWF_hectares*.333)+(0*(HayLand+Other_Land+WCP_hectares+native_pasture+tame_pasture+eph_wet+perm_wet+seas_wet+semi_wet+shrubland)) {area of land in fallow in the landscape at each time step}

habitat[cropland] =

(WFP_hectares*.66666)+(WF_hectares*.5)+(WWF_hectares*.666)+(WCP_hectares)+(0*(HayLand+Other_Land+native_pasture+tame_pasture+eph_wet+perm_wet+seas_wet+semi_wet+shrubland)) {area of landscape in non-continuous crop at each time step}

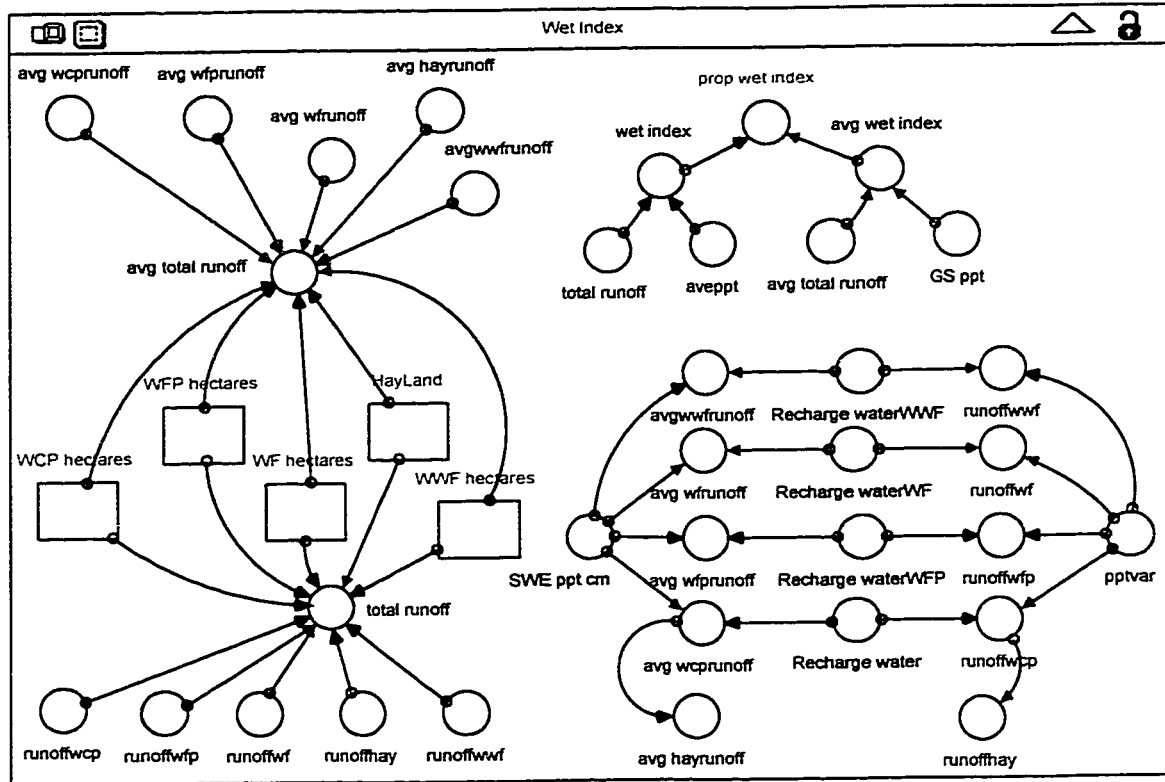
habitat[tame_hay] = ((1-

(tame_pasture/(INIT(HayLand))))*HayLand)+(0*(Other_Land+WCP_hectares+WFP_h

```

ectares+WF_hectares+WWF_hectares+native_pasture+eph_wet+perm_wet+seas_wet+s
emi_wet+shrublnd)) {area of the landscape dedicated to tame hay at each time step}
habitat[imp_pasture] =
((tame_pasture/(INIT(HayLand)))*HayLand)+(0*(WCP_hectares+WFP_hectares+WF_
hectares+WWF_hectares+native_pasture+Other_Land+eph_wet+perm_wet+seas_wet+s
emi_wet+shrublnd)) {just an estimate}
habitat[native_grass] =
((native_pasture/(INIT(Other_Land)))*Other_Land)+(0*(HayLand+WCP_hectares+WF
P_hectares+WF_hectares+WWF_hectares+tame_pasture+eph_wet+perm_wet+seas_wet
+semi_wet+shrublnd)) {document}
habitat[shrub] =
((shrublnd/(INIT(Other_Land)))*Other_Land)+(0*(HayLand+WCP_hectares+WFP_hec
tares+WF_hectares+WWF_hectares+native_pasture+tame_pasture+eph_wet+perm_wet
+seas_wet+semi_wet))
habitat[ephemeral_wetland] =
((eph_wet/(INIT(Other_Land)))*Other_Land)+(0*(HayLand+WCP_hectares+WFP_hect
ares+WF_hectares+WWF_hectares+native_pasture+tame_pasture+perm_wet+seas_wet
+semi_wet+shrublnd))
habitat[seasonal_wetland] =
((seas_wet/(INIT(Other_Land)))*Other_Land)+(0*(HayLand+WCP_hectares+WFP_hec
tares+WF_hectares+WWF_hectares+native_pasture+tame_pasture+eph_wet+perm_wet
+semi_wet+shrublnd))
habitat[semipermwet] =
((semi_wet/(INIT(Other_Land)))*Other_Land)+(0*(HayLand+WCP_hectares+WFP_he
ctares+WF_hectares+WWF_hectares+eph_wet+native_pasture+perm_wet+seas_wet+sh
rublnd+tame_pasture))
habitat[perm_wetland] =
((perm_wet/(INIT(Other_Land)))*Other_Land)+(0*(HayLand+WCP_hectares+WFP_he
ctares+WF_hectares+WWF_hectares+native_pasture+tame_pasture+eph_wet+seas_wet
+semi_wet+shrublnd))
Habitat_Diversity = -1*(ARRAYSUM(AlogA[*])) {calculates the degree to which an
ecodistrict is dominated by a few or many land uses}
native_pasture = 1
perm_wet = 1000
rel_hab_abund[Habitat_type] = habitat[Habitat_type]/tot_hab {calculates the proportion
of the habitat landscape is dedicated to the particular habitat type}
seas_wet = 1000
semi_wet = 1000
shrublnd = 1000
tame_pasture = 1
tot_hab = ARRAYSUM(habitat[*])

```



Wet Index

HayLand(t) = HayLand(t - dt)

INIT HayLand = 1000

WCP_hectares(t) = WCP_hectares(t - dt)

INIT WCP_hectares = WCP_rot

WFP_hectares(t) = WFP_hectares(t - dt)

INIT WFP_hectares = WFP_rot

WF_hectares(t) = WF_hectares(t - dt)

INIT WF_hectares = WF_rot

WWF_hectares(t) = WWF_hectares(t - dt)

INIT WWF_hectares = WWF_rot

aveppt = IF TIME=0 THEN GS_ppt ELSE GS_gamma

avgwwfrunoff = SWE_ppt_cm-Recharge_waterWWF

avg_hayrunoff = avg_wcprunoff

avg_total_runoff =

(INIT(WF_hectares)*avg_wfrunoff)+(INIT(WFP_hectares)*avg_wfprunoff)+(INIT(WCP_hectares)*avg_wcprunoff)+(INIT(HayLand)*avg_hayrunoff)+(INIT(WWF_hectares)*avgwwfrunoff) {winter runoff given initial land use and average precipitation}

avg_wcprunoff = SWE_ppt_cm-Recharge_water

avg_wet_index =

(.3*((DELAY(avg_total_runoff,2))+DELAY(GS_ppt,2)))+(0.6*((DELAY(avg_total_runoff,1))+DELAY(GS_ppt,1)))+(avg_total_runoff) {average wet index}

avg_wfprunoff = SWE_ppt_cm-Recharge_waterWFP

```

avg_wfrunoff = SWE_ppt_cm-Recharge_waterWF {runoff cm/ha given average non
growing season precipitation}
GS_ppt = 35
pptvar = IF TIME=0 THEN SWE_ppt_cm ELSE SWE_gamma
prop_wet_index = wet_index/avg_wet_index {gives the proportion that the wet index
makes up of the average index}
Recharge_water = pptvar*recharge_rate
Recharge_waterWF = pptvar*recharge_ratewf
Recharge_waterWFP = pptvar*recharge_ratewfp
Recharge_waterWWF = pptvar*recharge_ratewwf
runoffhay = runoffwcp
runoffwcp = pptvar-Recharge_water
runoffwf = pptvar-Recharge_waterWF {average runoff cm/ha for WF based on baseline
land use}
runoffwfp = pptvar-Recharge_waterWFP
runoffwwf = pptvar-Recharge_waterWWF
SWE_ppt_cm = 16
total_runoff =
(WF_hectares*runoffwf)+(WFP_hectares*runoffwfp)+(WCP_hectares*runoffwcp)+(Ha
yLand*runoffhay)+(WWF_hectares*runoffwwf) {total annual runoff at landscape level}
wet_index =
(.3*((DELAY(total_runoff,2))+DELAY(aveppt,2)))+(0.6*((DELAY(total_runoff,1))+
DELAY(aveppt,1)))+(total_runoff) {gives weighted value for water conditions in
wetlands}

```